

Forestry contributed to warming of forest ecosystems in northern Germany during the extreme summers of 2018 and 2019

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Abstract

- Forest management influences a variety of ecosystem structures and processes relevant to meso- and microclimatic regulation, but little research has been done on how forest management can mitigate the negative effects of climate change on forest ecosystems.
- We studied the temperature regulation capacity during the two Central European extreme summers in 2018 and 2019 in Scots pine plantations and European beech forests with different management-related structural characteristics.
- We found that the maximum temperature was higher when more trees were cut and canopy was more open. Logging 100 trees per hectare increased maximum temperature by 0.21–0.34 K at ground level and by 0.09–0.17 K in 1.3 m above ground. Opening the forest canopy by 10% significantly increased T_{\max} , measured 1.3 m above ground by 0.46 K (including pine and beech stands) and 0.35 K (only pine stands). At ground level, T_{\max} increased by 0.53 K for the model including pine and beech stands and by 0.41 K in pure pine stands. Relative temperature cooling capacity decreased with increasing wood harvest activities, with below average values in 2018 (and 2019) when more than 656 (and 867) trees per hectare were felled. In the pine forests studied, the relative temperature buffering capacity 1.3 m above ground was lower than average values for all sample plots when canopy cover was below 82%. In both study years, mean maximum temperature measured at ground level and in 1.3 m was highest in a pine-dominated sample plots with relatively low stand volume ($177 \text{ m}^3 \text{ ha}^{-1}$) and 9 K lower in a sample plot with relatively high stock volumes of *Fagus sylvatica* ($>565 \text{ m}^3 \text{ ha}^{-1}$). During the hottest day in 2019, the difference in temperature peaks was more than 13 K for pine-dominated sample plots with relatively dense (72%) and low (46%) canopy cover.
- Structural forest characteristics influenced by forest management significantly affect microclimatic conditions and therefore ecosystem vulnerability to climate change. We advocate keeping the canopy as dense as possible (at least 80%) by

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maintaining sufficient overgrowth and by supporting deciduous trees that provide effective shade.

KEY WORDS

climate change, cooling, drought events, forest canopy, forest functionality, regulating ecosystem services, temperature regulation

1 | INTRODUCTION

In Central Europe, the weather extremes of the past years, in combination with insect outbreaks and other calamities, have already contributed to large-scale tree dieback (Schuldt et al., 2020; Seidl et al., 2017). Heat and drought stress, forest fires, storms and late frost events, as well as associated pests and diseases, often occur in complex interaction with each other and cause impacts with unpredictable outcomes (Allen et al., 2010; Carnicer et al., 2011). Extreme temperatures, for example, can cause physiological damage and have an impact on the vitality, growth and mortality of trees (Buras et al., 2018). This can trigger a decline in productivity, carbon sequestration and woody biomass (Smith et al., 2020; Vieira et al., 2020). Warmer temperatures can directly impair organisms, but also drive a nonlinear rise in water vapour deficit and contribute to the desiccation of plants (Hatfield & Prueger, 2015). Higher temperatures can increase the production of seeds by trees but compromise their establishment and survival due to increased water stress (Ibáñez et al., 2017). Although species have different mechanisms to cope with drought (Volaire, 2018), tree vitality and productivity already declined in response to recent drought events (Rohner et al., 2021; Senf et al., 2020).

The extreme drought in 2018 severely affected forest stands in Germany (Ionita et al., 2021), and, as a consequence, about twice as many trees died in 2019 compared to 2018 and about 80% of all living trees showed poor vitality (BMEL, 2020). Climate conditions, which are currently perceived as extreme (Büntgen et al., 2021), could represent the new ‘normal’ in the near future (Hari et al., 2020; Scharnweber et al., 2020). It is, therefore, of high interest that which practices of forest management such as thinning or keeping close canopies have the potential to attenuate the adverse effects of heat waves within forest stands.

Recent studies conclude that thinning can reduce drought impacts (Ameztegui et al., 2017; D’Amato et al., 2013; Del Río et al., 2017; Gebhardt et al., 2014; Giuggiola et al., 2013, 2016; Ma et al., 2010; Primicia et al., 2013; Simonin et al., 2007; Sohn, Hartig, et al., 2016). However, a critical challenge for forest management is to support the ecosystem capacity for microclimate regulation, especially in times of frequently recurring dry and hot years, when precipitation is absent for longer periods of drought. Here, microclimate regulation implies the attenuation of summer peak temperatures, moderation of mean temperatures and the buffering of temperature fluctuations in the forest interior. An important outcome of microclimatic regulation is the stabilization of habitat conditions for species affected by shifting

microclimatic conditions (De Frenne et al., 2013; Milling et al., 2018; Suggitt et al., 2011; Tuff et al., 2016; Varner & Dearing, 2014; Zellweger et al., 2020). With growth and development of a forest, the canopy structure changes, water uptake potential improves and transpiration reduces extreme temperatures (Holdaway et al., 2010). The canopy structure of a forest stand resulting from forest management directly influences temperature and vapour pressure deficit in the forest interior (Jucker et al., 2018), where the release of water vapour by plants is often the only source of humidity, particularly in seasons with deficient precipitation (Moreira et al., 1997). As a consequence, the forest canopy cools soil and air during warm days and buffers temperature fluctuations (Jin et al., 2019). In this way, forest interior temperatures are moderated across seasons (Zellweger et al., 2019), and temperature differences inside and outside forests are higher when macroclimatic conditions become more extreme (De Frenne et al., 2019). Structural characteristics that determine a within stand’s microclimate include the predominant tree species (De Abreu-Harbich et al., 2015), tree vitality (Sanusi & Livesley, 2020), biomass volume (Norris et al., 2012) as well as elevation and canopy cover (Ma et al., 2010).

Microclimatic regulation is an important ecosystem service facilitated especially by trees, woodlands or forests, which is increasingly appreciated and taken into account in urban development and planning (Fung & Jim, 2019). In the context of urban adaptations to climate change, microclimate regulation for the mitigation of heat island effects by green infrastructure has been investigated much more intensively (Kong et al., 2014; Lindén et al., 2016; Shashua-Bar et al., 2009; Wang et al., 2018) than in open landscapes and within forest ecosystems. Despite the fact that forest management influences a variety of forest ecosystem structures and processes relevant to meso- and microclimatic regulation, so far, there is a knowledge gap of how forest management could contribute to a reduction of temperature extremes, mean temperatures and temperature fluctuations within forest stands and could thus improve climate change adaption at the stand or even at the landscape level.

There is plenty of evidence and several conceptual frameworks have been suggested that may encourage forest managers to reflect about temperature management in forests for stabilising the within-stand microclimate during extreme summer heat as contribution to preserving tree vitality and productivity under climate change. An open question is to what extent forest characteristics that are directly shaped by forest management operations (such as thinning, harvesting intensity and nature conservation) influence within-stand temperatures under

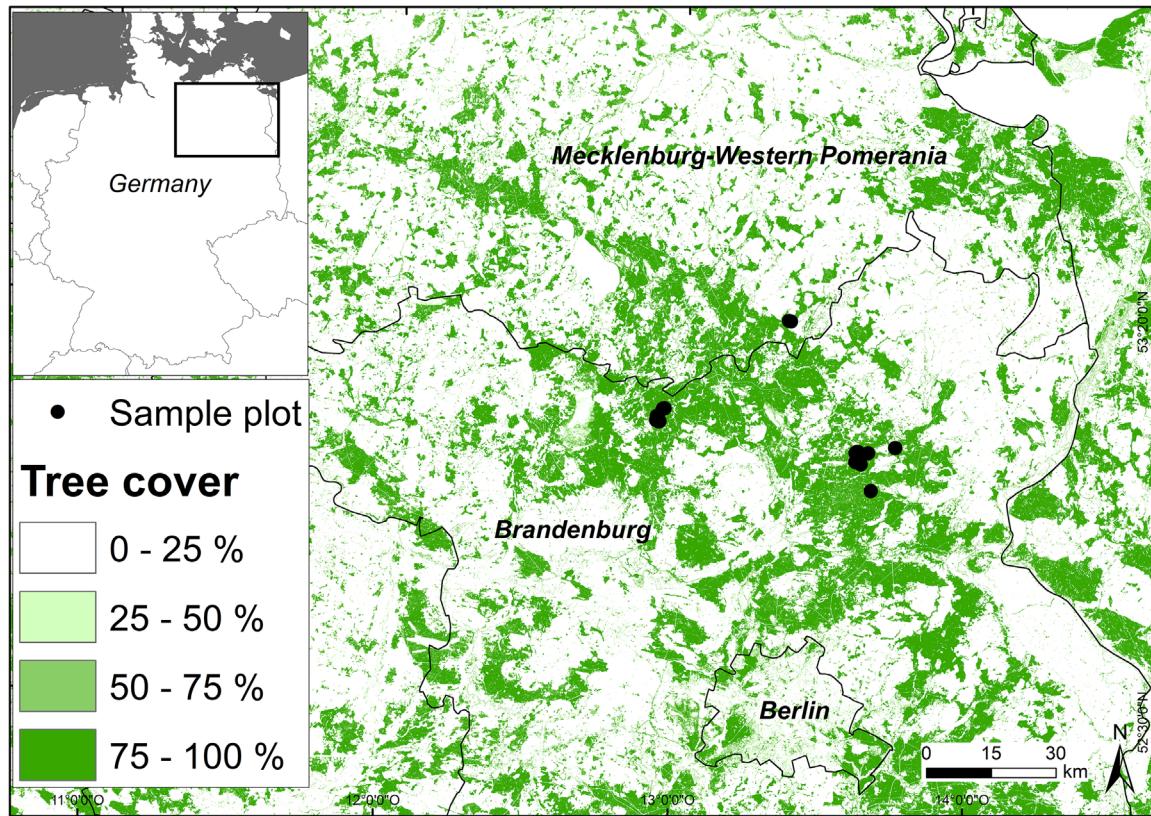


FIGURE 1 Sample plots are located in the north-eastern lowlands of Germany belonging to the largest remaining forest landscape in northern Germany. Tree cover as of 2000 in green (Hansen et al., 2013)

extreme climatic conditions in exceptionally hot periods in a temperate region.

This study aims at analysing the effects of forest management on the microclimatic regulation of forest stands during the two extremely hot and dry summers in 2018 and 2019 (see, e.g. Buras et al., 2020; Kornhuber et al., 2019; Vogel et al., 2019) by comparing forests in northern Germany across a gradient of different structural characteristics resulting from silvicultural treatments such as thinning and wood harvest. We conducted on-site microclimatic measurements in stands dominated by *Pinus sylvestris* and *Fagus sylvatica* with a high temporal and spatial resolution in order to quantify the temperature regulation of forest stands in relation to stand structural characteristics. We hypothesized that forest management activities, such as tree harvesting, would affect microclimate regulation negatively by causing higher within-stand temperatures.

2 | MATERIALS AND METHODS

2.1 | Study area

The study area is located in a post-glacial landscape in the north-eastern lowlands of Germany comprising forest stands on moraines and outwash plains (Figure 1). The investigated forest sites mainly comprise monocultures of *Pinus sylvestris* (henceforth referred to as *P.*

sylvestris or pine), but also pine stands with broadleaved understorey as well as stands of *Fagus sylvatica* (*F. sylvatica* or European beech). The forest stands are of different age (overstory trees with an age between ca. 20 and 300 years), which experienced different silvicultural management in the past including regular thinning (e.g. about every 5 years), harvesting, enlarging the volume of fresh downed deadwood by felling and plantings but also include stands without interventions since 20–70 years. The diverse character of the different sites represents a gradient of management intensity rather than distinct treatments. The resulting differences in stand structure and compositional attributes were analysed in relation to microclimatic indicators.

Forest stands were codified as a specific site. Two to six sample plots were surveyed per site and located in the centre of forest stands with a minimum distance of 50 m to each other. A total of 68 sample plots were investigated in 2018 (thereof six beech sample plots within two sites) and 101 sample plots in 2019 (thereof nine beech sample plots within three sites) (Table 1). All data were collected within concentric circular sample plots of 0.1 and 0.03 ha (Figure 2, left). At the centre, a wooden pole was installed and two microclimatic data-loggers were mounted on the north-facing side (Figure 2, right).

Weather data from the closest weather station in Angermünde, located in about 30–80 km distance to the investigated sites, showed that the annual mean temperature was extremely high in the study period, that is 2°C higher in the year 2019 compared to the reference period (8.9°C averaged over 1981–2010) (DWD, 2020). In the

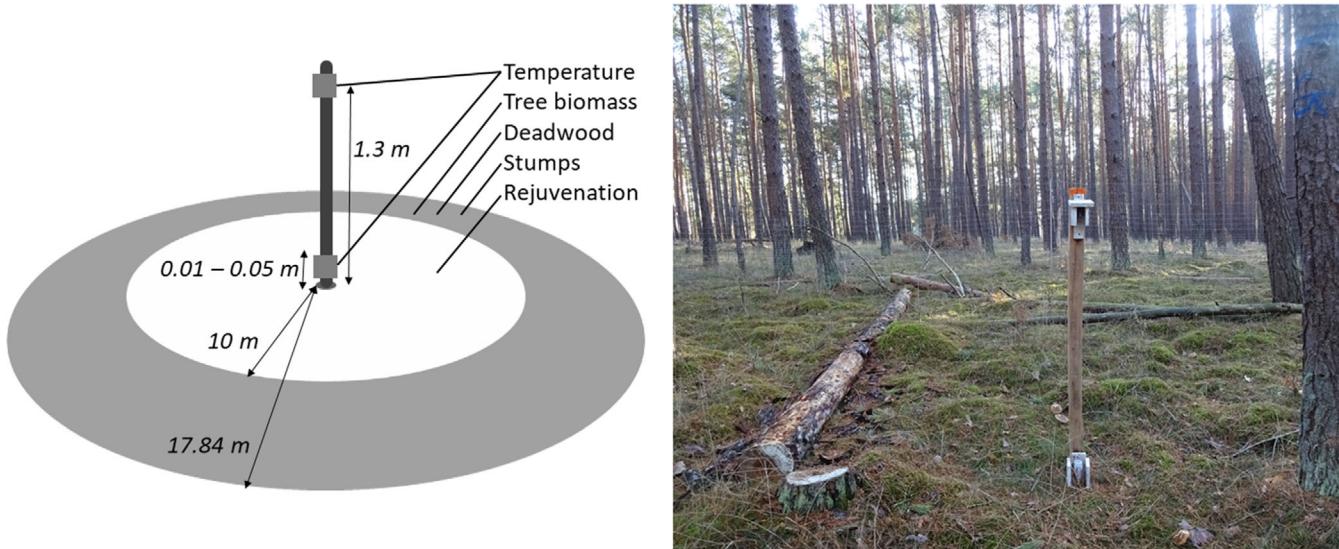


FIGURE 2 Schematic visualisation of measurements taken in a sample plot (left) and sample plot with two dataloggers in a half-open wooden box (right)

TABLE 1 Number of sample plots in the study period for the three study areas

Year	2018	2019
Wittwesee (<i>Pinus sylvestris</i>)	42	60
Reiersdorf		
<i>Pinus sylvestris</i>	20	32
<i>Fagus sylvatica</i>	3	6
Heilige Hallen (<i>Fagus sylvatica</i>)	3	3
Total	68	101

year 2018, the daily maximum temperature was above 25°C for 82 days, which is 45 days more than in the reference period (1981–2010). Mean summer temperature in the reference period was 17.3°C (1981–2010), 19.9°C in 2018, and 20.3°C in 2019.

2.2 | Microclimate

Temperature was measured at 1.3 m above ground using HOBO UA-001-64 Pendant data-loggers, and at ground level with HOBO U23-001 Pro V2 data-loggers (Onset Computer Corporation, Bourne, USA). Data-loggers were protected from direct sunlight by a white-painted, half-open, wooden box. The data-loggers were initially left for at least 12 h to acclimatize to ambient conditions before temporally synchronized records were taken at 30 min intervals from May to October in 2018 and 2019.

As an ANOVA showed that the variables considered (T_{\max} , RTC and RTB, see definitions below) differed significantly between the two height levels ($p < 0.001$), they were analysed separately for each height level. For each year, all records of days with an average temperature above 20°C (resulting in 45 days in 2018 and 39 days in 2019), cal-

culated as mean over all sample plots and data-loggers of both height levels, were extracted, and the following microclimatic indicators were computed following Blumroeder et al. (2019):

- Maximum temperature (T_{\max}) calculated as mean of the five highest temperature records per day.
- Relative temperature cooling capacity (RTC) represents the capacity of lowering daily mean temperature records. It is calculated as reciprocal of the daily mean temperature of each plot divided by the daily mean temperature over all sample plots. Indicator values > 1 indicate that the daily mean temperature was lower than the average over all sample plots, meaning that a relative cooling effect was generated.
- Relative temperature buffering capacity (RTB) represents the deviation of temperature records from the daily mean temperature. It is calculated as reciprocal of the standard deviation of daily mean temperature of each plot divided by the standard deviation of daily mean temperature of all plots. Indicator values > 1 indicate that the temperature was less variable over a day compared to the average over all sample plots, meaning that a buffering effect was generated by stabilising the variation of daily temperatures.

2.3 | Stand characteristics and forest management

Silvicultural treatments such as tree felling and tree biomass extraction directly or indirectly impact forest stand attributes. We surveyed indicators that are primarily influenced by forest management activities and tree harvesting in particular (i.e. canopy cover, stand volume, Stand Density Index (SDI), number of cut stumps, deadwood volume and regeneration density).

Canopy cover was assessed during the vegetation period in the year 2019, using a spherical crown densiometer with a convex mirror (Model A, Forestry Suppliers, Inc., Mississippi, USA) showing 24

squares of 0.25 inches engraved on the surface (Lemmon, 1956). The number of squares in the mirror occupied by tree crowns and stems was counted at the centre of each sample plot into the four cardinal directions, multiplied by 1.04 to obtain the estimated overstory density percentage and averaged per sample plot to determine the density of the overstory by tree crowns (Lemmon, 1956). This indicator represents the proportion of sky that is covered by the tree crowns.

In wintertime, all tree individuals with a diameter at breast height of at least 6 cm were recorded within a 0.1-ha circular sample plot around the data-loggers. Tree species, apex height and circumference at 1.3 m height were assessed. Tree volume was calculated for each single living tree based on diameter and height (Lockow, 2007), and was aggregated per sample plot and extrapolated to 1 ha to quantify the stand volume.

SDI (Reineke, 1933) was calculated based on the number of trees per unit area and quadratic mean diameter (Curtis & Marshall, 2000) and indicates the stocking density of trees in a stand. The higher the index value, the more crowded is a stand.

Tree stumps are defined as snags of less than 1.3-m height. Stumps result either by harvesting operations (cut stems) or natural disturbance and dieback (broken stems). All cut tree stumps within the 0.1-ha sample plots were counted, extrapolated to 1 ha and used as proxy indicator for management history and harvesting intensity in the further analysis.

Standing and downed deadwood were also recorded within a sample plot encompassing a 0.1 ha circle around the data-loggers. For each snag taller than 1.3 m within a sample plot, volume was calculated similarly to stand volume for unbroken snags and based on minimum and maximum diameter as well as length for broken snags. The volume of downed deadwood with a minimum diameter of 5 cm and minimum length of 1.3 m was determined on the basis of the diameters at both ends and length. For the further analysis, all types of standing and downed deadwood were combined as the total volume of deadwood per hectare, similar to stand volume.

Trees with a diameter smaller than 6 cm at breast height were classified as tree regeneration. Regeneration density (N/ha) was recorded as the number of trees within a sample plot encompassing a circle with 10-m radius and extrapolated to 1 ha.

2.4 | Statistical analyses

Stand characteristics (canopy cover, stand volume, SDI, number of stumps, deadwood volume and regeneration density) were tested for correlations to avoid multicollinearity and used in the further analysis because correlation coefficients between all predictor variables were <0.7 (Dormann et al., 2013).

Linear mixed effects models fitted by restricted maximum likelihood (REML) and t-tests using Satterthwaite's method of the R-package lmerTest (Kuznetsova et al., 2017) were used to analyse the relationships between structural stand characteristics resulting from forest management and the microclimate indicators from the two height levels (Nakagawa & Schielzeth, 2013). Canopy cover (only available for the year 2019), stand volume, SDI, number of stumps, deadwood vol-

ume and regeneration density were modelled as fixed effects; sites were modelled as random effect to account for spatial autocorrelation, and the response variable was a microclimatic indicator per height level. Due to the limited sample size of 68 (2018) and 101 (2019) plots and the relatively high number of six (2018) to seven (2019) fixed-effect predictor variables, we did not model and test interaction terms.

The models were fitted separately for the years 2018 and 2019 because data for canopy cover were available only for 2019. All models were fitted once using the entire dataset including all sample plots and once again for the pine monocultures only, because the dataset for beech forests was too small to be considered separately (only six sample plots within two sites in 2018 and nine sample plots within two sites in 2019).

We assessed the assumptions of linear (mixed) models, including the normal distribution and homoscedasticity of the residuals, using QQ-plots and residuals versus fitted values plots. For each year and microclimatic indicator, we generated figures for the significant fixed effects of the model with the highest marginal R^2 . All statistical analyses were conducted in R Studio (R Development Core Team, 2008).

3 | RESULTS

3.1 | Maximum temperature

In both years, T_{\max} measured at ground level and in 1.3 m was highest in a pine-dominated sample plot with relatively low living tree biomass ($177 \text{ m}^3 \text{ ha}^{-1}$). In a sample plot with relatively high stock volumes of *F. sylvatica* ($>565 \text{ m}^3 \text{ ha}^{-1}$), T_{\max} was 9 K lower than in the warmest plot.

Linear mixed effects models showed that the proportion of variance explained by the fixed factors was highest for the datasets including only pine sample plots. The number of stumps, regeneration density and SDI were significant in the year 2018 for T_{\max} in 1.3 m and in 2019, when also data for canopy cover were available, canopy cover was significant as well (Table 2; Figure 3).

The number of stumps resulting from harvesting activities was significantly associated with T_{\max} (Table 2). Across all years and sample plots, when the number of stumps increased by 100, maximum temperature increased by 0.21–0.34 K at ground level and by 0.09–0.17 K in 1.3 m (Table 2; Figure 3a). Since the number of stumps is a strong proxy for the intensity of harvesting, this indicates a strong effect of tree harvesting on maximum temperature.

In 2019, when data for canopy cover were available and included in the linear mixed effects models, canopy cover influenced the average maximum temperature (T_{\max}) in all tested datasets (Table 2). Opening canopy cover by 10% increased T_{\max} 1.3 m above ground by 0.46 K (including pine and beech stands) and 0.35 K (only pine stands) (Figure 3g). At ground level, T_{\max} increased by 0.53 K for the model including pine and beech stands and 0.41 K in pure pine stands.

Tree regeneration also showed significant effects on T_{\max} across all datasets, except for T_{\max} in 1.3 m with the dataset including all sample plots in 2018 (Table 2). An increase by 1000 regenerating trees per

TABLE 2 Estimates of all fixed effect parameters of the linear mixed effect models

Microclimate indicator	Year	Sample	Level	Canopy cover (%)	Stand volume ($m^3 ha^{-1}$)	Stand Density Index (Reineke, 1933)	Stumps (N)	Deadwood volume ($m^3 ha^{-1}$)	Regeneration (N)	Marginal R^2	Conditional R^2	
T_{max}	2018	All plots	Top	NA	-2.69×10^{-3}	ns	1.48×10^{-3}	ns	ns	0.44	0.85	
		Ground	NA	ns	-3.05×10^{-3}	ns	3.41×10^{-3}	ns	1.20×10^{-4}	0.62	0.79	
		Only pine	Top	NA	ns	-3.09×10^{-3}	1.74×10^{-3}	ns	5.51×10^{-5}	0.71	0.74	
	2019	Ground	NA	ns	-2.42×10^{-3}	3.48×10^{-3}	ns	1.03×10^{-4}	0.57	0.73		
		Top	-4.60×10^{-2}	-1.53×10^{-3}	ns	9.56×10^{-4}	ns	5.37×10^{-5}	0.58	0.84		
		Only pine	Top	-5.31×10^{-2}	-3.31×10^{-3}	ns	2.11×10^{-3}	ns	7.81×10^{-5}	0.5	0.81	
RTC	2018	Ground	NA	ns	-1.66×10^{-3}	9.22×10^{-4}	ns	6.92×10^{-5}	0.63	0.75		
		Top	ns	-4.07×10^{-2}	ns	ns	2.11×10^{-3}	ns	1.01×10^{-4}	0.39	0.73	
		Only pine	Top	NA	ns	ns	ns	-3.75×10^{-5}	ns	0.02	0.96	
	2019	Ground	NA	ns	ns	-5.34×10^{-5}	ns	ns	ns	0.26	0.75	
		Top	NA	ns	ns	ns	-4.47×10^{-5}	1.39×10^{-4}	ns	0.02	0.92	
		Only pine	Top	3.08×10^{-4}	ns	ns	-1.82×10^{-5}	ns	-6.05×10^{-7}	0.16	0.74	
RTB	2018	Ground	ns	ns	ns	-3.92×10^{-5}	ns	ns	ns	0.23	0.72	
		Top	ns	ns	ns	-1.77×10^{-5}	ns	-1.02×10^{-6}	0.12	0.82		
		Only pine	Top	NA	ns	-4.00×10^{-5}	ns	-1.56×10^{-6}	0.1	0.66		
	2019	Ground	ns	2.18×10^{-4}	ns	ns	ns	ns	0.07	0.97		
		Top	NA	ns	ns	ns	ns	ns	ns	0.09	0.91	
		Only pine	Top	NA	ns	3.14×10^{-4}	-1.25×10^{-4}	ns	-4.42×10^{-6}	0.72	0.87	

Note: Each line in the table corresponds to a different model. For each model, the parameter estimates as well as the marginal and conditional R^2 are reported. The significance of the estimates is indicated by the font: bold for $p < 0.05$ and italics for $p < 0.1$ (ns = non-significant).

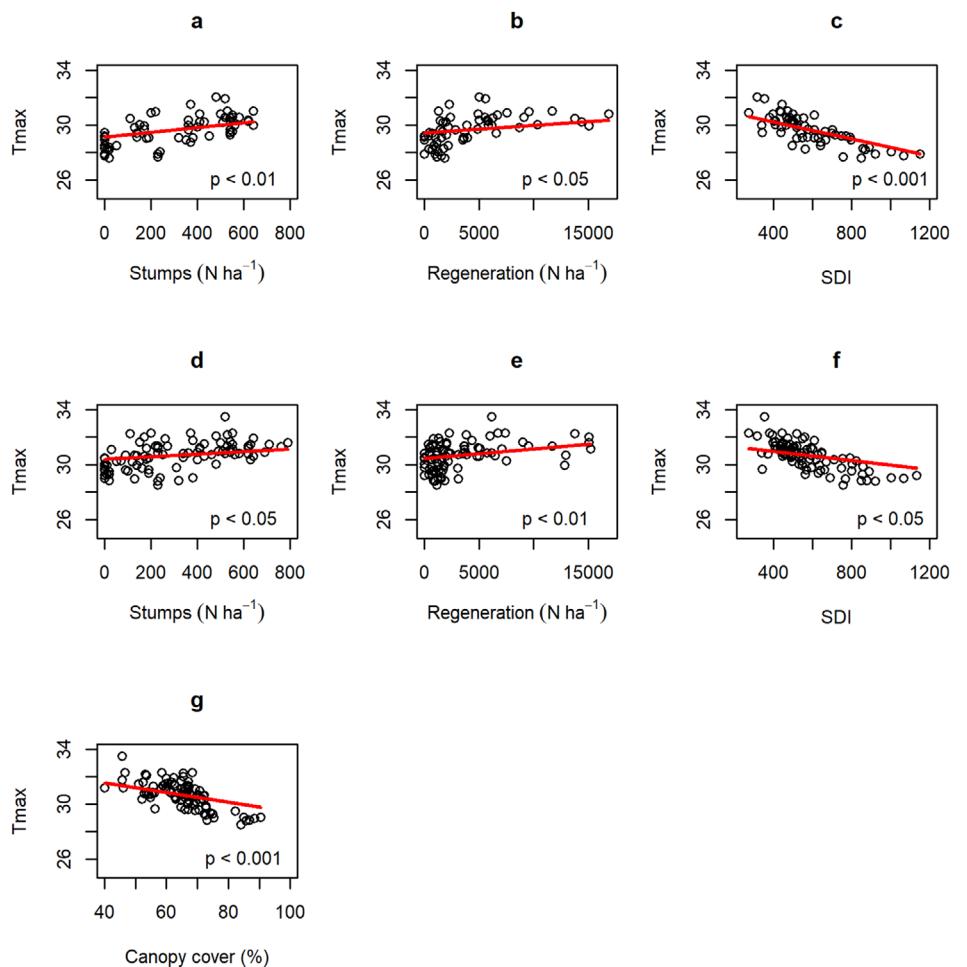


FIGURE 3 Maximum temperature in 1.3 m above ground in sites dominated by *Pinus sylvestris* in 2018 (a–c; $R^2_{\text{marginal}} = 0.71$; $R^2_{\text{conditional}} = 0.74$) in 2019 (d–g; $R^2_{\text{marginal}} = 0.63$; $R^2_{\text{conditional}} = 0.65$) in relation to the number of cut trees (stumps; a and d), the number of trees < 6 cm DBH (regeneration; b and e), Stand Density Index (SDI; c and f) and canopy cover (g). The points show the data and the red lines model predictions. The predictions were generated by only varying the variable shown at the x-axis, while all other variables were fixed at their observed mean value. The p -values refer to the test that the slope estimate of the variable shown on the x-axis equals 0. The degrees of freedom were approximated using Satterthwaite's correction method

hectare was related to an increase in maximum temperature by around 0.1 K (Figure 3b).

Decreasing stand volume by 100 m^3 per hectare significantly increased T_{max} by 0.31–0.33 K at ground level and by 0.15–0.27 K in 1.3 m above ground in the models including all sample plots but not for the dataset comprising only pine stands (Table 2). When only stands dominated by *P. sylvestris* were considered, SDI had a negative effect on T_{max} , meaning that the denser a forest stand is stocked, the lower the maximum temperature (Table 2; Figures 3c and 3f).

3.2 | Relative temperature cooling capacity

The intensity of harvesting – indicated by the number of cut stumps – results in a significantly lower RTC at ground level for all models with all datasets (Table 2). At 1.3 m above ground, only in 2019 the number of stumps showed an effect on RTC (Table 2).

The two models with the highest marginal R^2 (RTC at ground level including all sample plots) showed that only the number of stumps had a significant effect (RTC decreased with more trees being cut). RTC fell below the average in 2018 (and 2019) when more than 656 (and 867) trees per hectare have been logged (Figure 4).

3.3 | Relative buffering capacity

In 2018, the model that best explained the variance of RTB showed that the number of stumps, regeneration density and SDI were significant predictors (Table 2). RTB was lower when more trees were felled, more trees regenerate and stands are less dense (Figure 5). In 2019, when canopy cover was also tested, this was significant for all datasets (Table 2). In the studied pine forest, RTB in 1.3 m was lower than the average of all sample plots when canopy cover was below 82% (Figure 5g). RTB in pine forests at ground level was lower than the

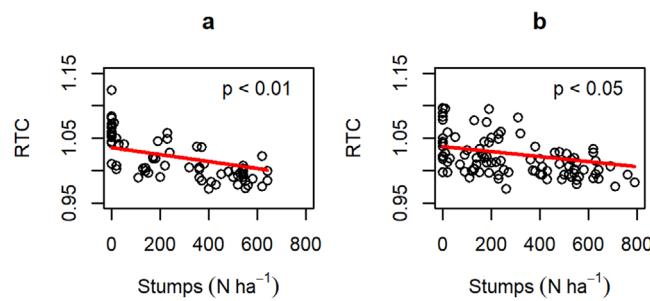


FIGURE 4 Relative temperature cooling capacity at ground level across all study plots including sites dominated by *Pinus sylvestris* and *Fagus sylvatica* in 2018 (a; $R^2_{\text{marginal}} = 0.26$; $R^2_{\text{conditional}} = 0.75$) in 2019 (b; $R^2_{\text{marginal}} = 0.23$; $R^2_{\text{conditional}} = 0.72$). The points show the data and the red lines model predictions. The predictions were generated by only varying the variable shown at the x-axis, while all other variables were fixed at their observed mean value. The p-values refer to the test that the slope estimate of the variable shown on the x-axis equals 0. The degrees of freedom were approximated using Satterthwaite's correction method

average if canopy cover was less than 49%. Including beech plots into the dataset, Imer showed that temperature fluctuation in 1.3 m above ground was above average when canopy cover was below 65%.

3.4 | Diurnal temperature variation

The highest daily mean temperature in the year 2019 was measured on June 26, with 26.6°C calculated over all measurements. On that day, peak temperature values at ground level differed by more than 13 K between pine-dominated sample plots with relatively dense and open canopy (72% vs. 46%, respectively) (Figure 6). Considering only pine stands, canopy cover significantly influenced temperature over the course of a day ($p < 0.05$). The highest peak temperature in 2019 was measured on June 30 in a pine stand ($177 \text{ m}^3 \text{ ha}^{-1}$) and exceeded 45°C, while maximum temperatures on the same day in beech stands remained below 35°C. On the same day, the range between minimum and maximum temperatures in beech stands was below 20°C, while the variation in pine plots was up to 35°C.

4 | DISCUSSION

Our findings confirm our hypothesis that forest management can significantly mediate forests' ability to dampen extreme temperatures, moderate mean temperature and temperature variability. Forests with high volumes of living trees, for example due to high stand age and low rates of timber extraction in the past, are effectively cooling landscape elements (compare Chen et al., 1999; Frey et al., 2016; Norris et al., 2012). For the reduction of maximum temperatures in the forest interior, the most decisive driver is the degree of canopy openness, but also the quantity of logged trees is relevant, and both variables are directly controlled by forest management (in terms of reduc-

ing harvesting activities and by developing denser, multi-layered forest stands). Our finding that a 10% reduction in canopy closure causes an increase in forest interior temperature is consistent with results from Thom et al. (2020), who found that an increase in surface light by 10%, resulting from the opening of the canopy in European beech forests, caused an increase in maximum temperature by 0.42°C. A study in Chinese forests revealed a 0.83 K increase in surface temperature (Kong et al., 2014). Minimizing the temperature of the forest interior contributes to climate regulation in the wider landscape and positively influence water and carbon cycles (Ellison et al., 2017). Microclimate regulation can therefore buffer adverse effects of climate change (Thom et al., 2020).

In the two record heat and drought years 2018 and 2019, denser and less thinned forests showed substantial microclimate regulation. Effective forest management aiming at a continuous forest cover and more complex structures instead of homogenous even-aged monocultures thus allows for stabilization of microclimatic conditions in the forest interior and counteracts extreme macroclimatic conditions to be expected under climate change. This is not only of direct relevance for the growth and survival of all woody species, but also for other forest organisms such as the ground vegetation or the forest edaphon on the edge of their thermal tolerance limits (De Frenne et al., 2013; Duffy et al., 2015; Martius et al., 2004). However, other ecosystem services, such as cultural services, and species richness (e.g. beetles) can be lower in forests with denser canopies and high carbon stocks (Felipe-Lucia et al., 2018; Sabatini et al., 2019; Seibold et al., 2016).

Differences in morphological characteristics between the tree species *P. sylvestris* and *F. sylvatica*, such as the shape of leaves, specific crown architecture and density, may explain species-specific shading properties and corresponding signatures of microclimate regulation (compare De Abreu-Harbich et al., 2015). The availability of water is also crucial for temperature regulation mechanisms (Davis et al., 2019). The overall decrease of heat and the avoidance of high maximum temperatures is of physiological relevance for forest trees and other forest organisms (Hatfield & Prueger, 2015; Suggitt et al., 2011), but also effectively contributes to the reduction of water deficits as evaporation increases nonlinearly with increasing temperatures (Nkemdirim, 1991).

Studies showed that certain drought impacts can be mitigated by thinning as thinned stands recover faster from growth reduction, but thinning also reduces wood quality, volume increment, litter mass and stand structural diversity (Del Río et al., 2017). Short-term improvements in the resistance of radial growth of trees to drought stress are more likely to be observed at wetter than at drier sites and therefore also depend on water supply (Laurent et al., 2003) and tree age as younger stands show that stand density reductions improved drought resistance and resilience but at higher age higher densities showed greatest drought resistance and resilience (D'Amato et al., 2013). Although thinning supports the recovery of radial growth after drought events, it hardly affects the resistance to drought and can lead to higher evaporation in the short term due to higher wind exposure (Sohn, Hartig, et al., 2016). In addition, the magnitude of short- and medium-term resistance, recovery and resilience of radial growth was affected by

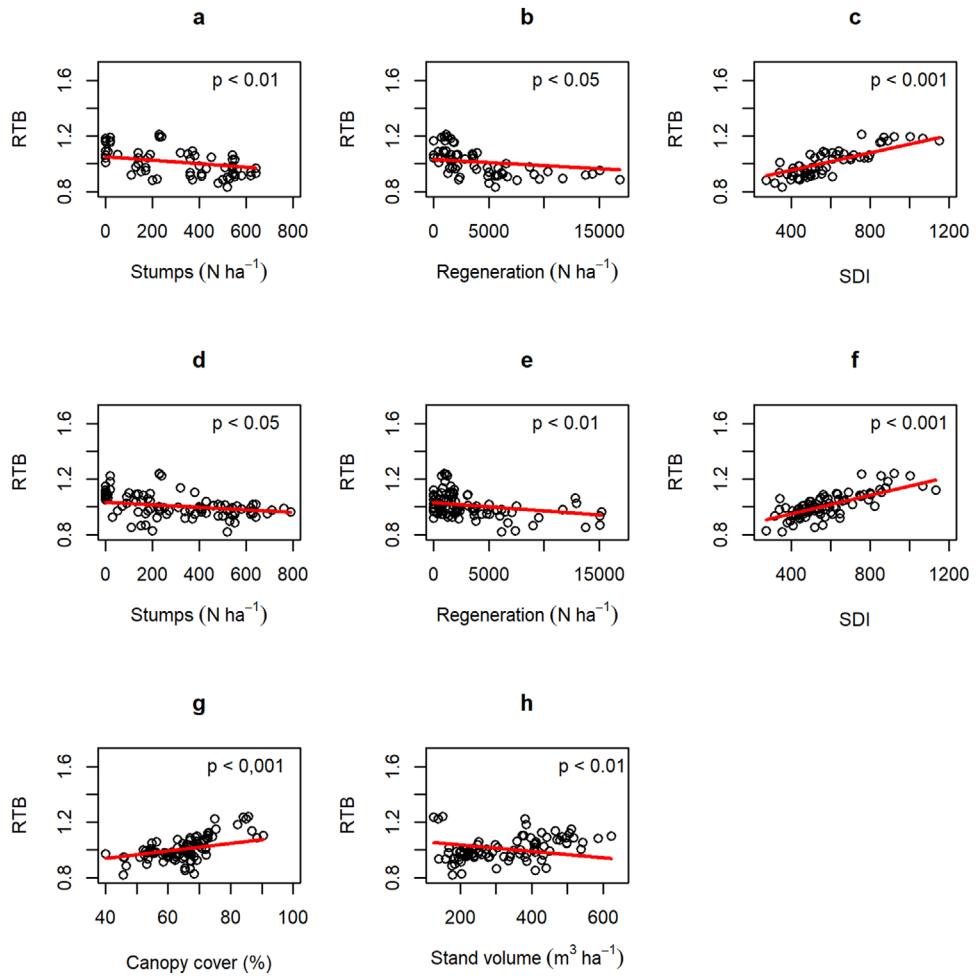


FIGURE 5 Relative temperature buffering capacity measured in 1.3 m above ground in sites dominated by *Pinus sylvestris* in 2018 (a–c; $R^2_{\text{marginal}} = 0.72$; $R^2_{\text{conditional}} = 0.87$) in 2019 (d–h; $R^2_{\text{marginal}} = 0.67$; $R^2_{\text{conditional}} = 0.79$) in relation to the number of cut trees (stumps; a and d), the number of trees <6 cm DBH (regeneration; b and e), Stand Density Index (SDI; c and f), canopy cover (g) and stand volume (h). The points show the data and the red lines model predictions. The points show the data and the red lines model predictions. The predictions were generated by only varying the variable shown at the x-axis, while all other variables were fixed at their observed mean value. The p-values refer to the test that the slope estimate of the variable shown on the x-axis equals 0. The degrees of freedom were approximated using Satterthwaite's correction method

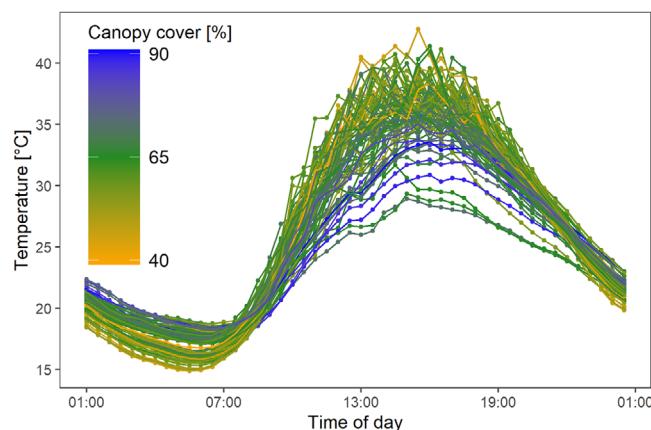


FIGURE 6 Diurnal temperature variation on the hottest day in the year 2019 measured in pine-dominated sample plots ($n = 88$) at ground-level

thinning (Sohn, Saha, et al., 2016), whereas thinning did not improve leaf-level efficiency and intrinsic water use efficiency (Fernández et al., 2015; Sohn, Saha, et al., 2016). Under certain conditions, the competition for water and water stress levels in forest stands can be reduced by thinning (Giuggiola et al., 2013; Sohn, Saha, et al., 2016). Although air and soil temperature as well as wind speed increase by thinning, soil moisture was found to be higher in thinned forest stands (Ma et al., 2010). However, higher transpiration rates of thinned canopies can lead to lower soil humidity despite increased throughfall (Primicia et al., 2013). For example Lagergren et al. (2008) found that thinning can elevate transpiration by 20% compared to unthinned stands and induce a sevenfold increase of transpiration at the peak of a drought. Higher evaporation in turn can compensate for the positive effect of thinning related to increased throughfall of precipitation and lower overstory transpiration (Simonin et al., 2007). Despite short-term decreases in stand-level transpiration (25% by moderate, 50% by heavy thinning) and higher soil water availability due to lower interception and

transpiration, single tree transpiration and additional understorey evapotranspiration can be observed in the mid-term after thinning (Gebhardt et al., 2014). Also, water vapor pressure deficit increases by thinning across heights in a stand (Rambo & North, 2009).

The benefits of thinning depend on the local climate conditions and cannot be generalized (Ameztegui et al., 2017). Clearly, it must be reflected more critically in times of frequently recurring dry and hot years, when precipitation is absent for longer periods of drought. Then, potential advantages of thinning can turn into a disadvantage, because higher water losses through evaporation become the decisive stressor in forests that experience more intense heat. It is also known that forest openings and clearings increase ambient and soil temperatures, which in turn negatively impact water availability, especially during periods of low precipitation (Redding et al., 2003). The larger the openings of forest canopies, the higher the air and soil temperature (Latif & Blackburn, 2010). At forest edges, soil moisture can be similar to open areas (Erdős et al., 2019). As a consequence of our findings, we recommend minimising heating and evaporation effects within the forest interior by avoiding the creation of artificial canopy gaps due to silvicultural operations, including intensive thinning and clearcutting as well as introducing road and skidding trail infrastructure. In this context, the fragmentation of the forests by roads and infrastructure as well as the opening of the canopy by the construction or maintenance of skidding and extraction trails must be discussed. Typically, wood harvest in production forests in Germany takes place every 5 years, and usually skidding trails with 20–40 m distance to each other are cut through the forest. The concomitant opening of the canopy creates internal forest edges and potential edge effects within a forested area that can reduce the microclimate regulation capacity and increases the risk of heat and drought stresses from the edges towards the forest interior (Duncan et al., 2019; Reed et al., 1996). Road infrastructure causes higher air and canopy temperatures as well as vapour pressure deficit (Delgado et al., 2007; Pohlman et al., 2007). Although increased evaporation on edges of fragmented forests can contribute to landscape cooling, in dry seasons the effect intensifies desiccation of the forest interior (Mendes & Prevedello, 2020). Increased tree mortality at forest edges indicates higher stress level in times of water shortage and heat influence (Brun et al., 2020).

Adapting forest management to climate change means, first and foremost, reducing the sensitivity of trees to drought events as much as possible. Extremely low precipitation and high temperature, depleted soil moisture and increased evapotranspiration were responsible for recent mid-spring droughts in central Europe and are supposed to continue in the long term due to climate change-induced phenomena of atmospheric circulation (Ionita et al., 2020). According to our results, high stock and dense canopy provide an insurance against heat and drought events. This is in contrast to a promotion of thinning as management strategy to adapt forests to climate change and reduce related drought impacts. We argue that microclimate management for cooler and less volatile forest interior temperature is a crucial element of ecosystem-based adaptation to climate change.

Forest microclimate regulation can be used as a proxy indicator for forest functionality (Chen et al., 1999) and ecological effects of

forest management and forest certification (Blumroeder et al., 2019). The evident patterns of cooling intensity can be transferred to other forests, although different ecosystems in other geographic regions exhibit divergent microclimatic features due to differences in albedo and evaporation (Li et al., 2015).

In some regions in Germany, especially in the federal state of Brandenburg, pine plantations cover about 70% of the forested area and urgently need to develop into diverse mixed and more self-regulating, cooler and resilient forest ecosystems in order to reduce their vulnerability to climate change. This is also in line with the insurance hypothesis that highlights the importance of biodiversity for maintaining ecosystem functions and processes (Loreau et al., 2001; Naeem & Li, 1997; Yachi & Loreau, 1999). On the one hand, tree rejuvenation is an ecosystem process that depends on suitable microclimatic conditions (Aussenac, 2000; Dingman et al., 2013). On the other hand, supporting tree regeneration by thinning can induce higher temperature extremes in the forest interior. Our findings are relevant for the implementation of forest management to transform such structurally homogenous stands. We advocate keeping the canopy as dense as possible, specifically with a cover of at least 80%, by maintaining sufficient overgrowth and by additional layers of any native deciduous tree species to establish multi-layered uneven-aged stands that provide effective shade. The trade-off between adequate light availability for understory plant growth, which is necessary for the forest to evolve into more resilient ecosystems, and the need to maintain protective shade is becoming increasingly apparent under climate change conditions, especially in extremely hot and dry years. Of central importance is the risk that additional heat, critically high temperatures, soil dehydration or even sunburn of exposed trees (e.g. *F. sylvatica*) can jeopardize the success of forest development.

Forest microclimate regulation is a key ecosystem service that influences other services (Tuff et al., 2016); its socioeconomic importance goes far beyond timber production and also has relevance for human health and recreation. Thus, foresters should assume increased responsibility for mitigating the landscape microclimate crisis that is exacerbating the negative impacts of the global climate crisis.

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

AUTHORS' CONTRIBUTIONS

JB and PLI conceived the study and the conceptual framework. JB developed the research design, conducted the fieldwork, prepared the data, computed the data analysis, and wrote the manuscript. PLI supervised all working steps and contributed to writing of the manuscript. FM provided support for data pre-processing and statistical analyses. WH provided specific advice. All authors contributed to the revision and finalization of the manuscript.

DATA AVAILABILITY STATEMENT

Data is deposited at Zenodo (Blumroeder, 2021).

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