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## RESEARCH ARTICLE

# Soil moisture and temperature dynamics in juvenile and mature forest as a result of tree growth, hydrometeorological forcings, and drought

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**Abstract**

Afforestation, as one of the major drivers of land cover change, has the potential to provide a wide range of ecosystem services. Aside from carbon sequestration, afforestation can improve hydrological regulation by increasing soil water storage capacity and reducing surface water runoff. However, afforested areas are rarely studied over time scales appropriate to determine when changes in soil hydrological processes occur as the planted (mixed) forests establish and grow. This study investigates the seasonal soil moisture and temperature dynamics, as well as the event-based responses to precipitation and dry periods, for a mature and a juvenile forest ecosystem over a 5-year time period. Generally, soil moisture was higher in the juvenile forest than in the mature forest, suggesting a lower physiological water demand. Following the 2018 drought, soil moisture dynamics in the growing juvenile plantation began to match those of the mature forest, owing to canopy development and possibly also to internal resilience mechanisms of the young forest to these external hot weather perturbations. Soil temperature dynamics in the juvenile plantation followed air temperature patterns closely, indicating lower thermal regulation capacity compared to the mature forest. While our findings show that an aggrading juvenile plantation achieves mature forest shallow soil moisture storage dynamics at an early stage, well before physiological maturity, this was not the case for soil temperature. Our results shed light on long-term trends of seasonal and event-based responses of soil moisture and temperatures in different-aged forest systems, which can be used to inform future assessments of hydrological and ecosystem responses to disturbances and forest management.

**KEYWORDS**

afforestation, BIFoR, drought, forest hydrology, micro-climate, soil moisture

## 1 | INTRODUCTION

Forests represent a key global carbon store and afforestation is an important tool for sequestering atmospheric carbon (Lewis et al., 2019). The increased awareness of the mitigating role that forests could play is supported by global initiatives including The Bonn Challenge (The Bonn Challenge, accessed Sept. 2020. <https://www.bonnchallenge.org/>), the UN's 'Decade on Ecosystem Restoration' (2021–2030), and the UN's REDD+ initiative. In the UK, the restoration of forests in agricultural settings helps to achieve the targets set by Net Zero by 2050, as stipulated in Environmental Land Management Policy (Land use: Policies for a Net Zero UK, 2020). Forests have a significant impact on hydrological processes, and well-planned afforestation has the potential to mitigate water management challenges in the context of global change (Cao et al., 2011; Ellison et al., 2012; Levia et al., 2020). Forests act as important modulators of water, nutrients and energy fluxes across the soil-vegetation-atmosphere interface (Ellison et al., 2017). Through their different water storage mechanisms from canopy down to deep root system, forests provide hydrological buffering capacity that is critical to moderate overland flow paths and thus, flood risk, as well as mitigating drought impacts through increased catchment storage (Choat et al., 2012). Consequently, afforestation influences runoff generation processes (Levia et al., 2020) and regulates soil moisture that provides the catchment template for flooding and hydrological drought (Brodribb et al., 2020) that are expected to be exacerbated under climate change. In addition, afforestation impacts (micro)meteorological conditions and boundary layer energy balances (Hannah et al., 2008), biogeochemical cycling in soils (Levia et al., 2020) and the landscape resilience to fire (Spence et al., 2020) as well as affecting soil habitat conditions, including the spread of soil and plant pathogens (Krause et al., 2013).

Forest soil moisture content directly impacts the system's capacity to absorb precipitation, preventing or reducing surface runoff generation during storm events, with infiltration rates varying significantly by soil type, tree species and forest management practice (Zimmermann et al., 2006). Additionally, spatial patterns of soil moisture deficit directly control the intensity of root water uptake to satisfy the photosynthetic water demand and growth of trees and sub-canopy vegetation (Seneviratne et al., 2010). Concurrently, tree physiology features such as canopy architecture, leaf area index, and stand age can have a direct impact on water inputs by controlling interception, throughfall, and stem-flow (Levia et al., 2020), thereby influencing infiltration processes. Physiology features also affect water outputs through root water uptake, decreased runoff and evapotranspiration (ET). The balance and interactions between these governing processes determines the spatial patterns and temporal dynamics of forest soil moisture (Naithani et al., 2013).

Soil moisture also influences plant nutrient availability, carbon sequestration by plants and soils, gaseous exchanges, including greenhouse gases, the leaching of nutrients into groundwater (Schlesinger et al., 2016), as well as plant-microbe interactions processes and nutrient cycling that ultimately affect the health and resilience of forests under climate change (Braganza et al., 2013). Changes in soil moisture dynamics, ranging from saturation to water deficit, affect the balance between aerobic and anaerobic microbial metabolism in soils. Water-

saturated forest soils, for example, have lower nitrogen mineralization rates (Ullah & Moore, 2009) and can also act as a source for atmospheric methane (CH<sub>4</sub>), whereas lower moisture conditions result in higher CH<sub>4</sub> absorption (Ullah & Moore, 2011).

The establishment of aggrading juvenile forests is expected to exert increasing influence on soil hydrological processes at various spatial and temporal scales. This complex relationship is expected to stabilize once fully-grown trees can effectively regulate the forest stand microclimate, as well as the nutrient and site hydrology (Douglas, 2018). However, inadequate consideration of the inherent relationship between abiotic factors and trees can limit afforestation success. For instance, the selection of unsuitable tree species combined with inappropriate high initial planting density accelerates soil drying, thus threatening the overall tree survival rate (Nan et al., 2020). Yang et al. (2012) also reported soil moisture depletion following a change in land cover to forest plantation in the semi-arid Loess Plateau in China, where warming and drying trends increased the overall moisture stress (Pu et al., 2006; Yao et al., 2005). Similarly, forest plantation failures were reported in Mediterranean drylands (Del Campo et al., 2020; Rey Benayas et al., 2015). Successes in afforestation are found when reforested areas following ideal planting criteria, and lead to higher infiltration rate and improved soil moisture retention (Ilstedt et al., 2016; Mapa, 1995). Given these challenges in afforestation, characterizing soil moisture dynamics on afforested sites is crucial in identifying techniques that may optimize tree growth processes and, thus, carbon sequestration from the atmosphere (Lewis et al., 2019; Pérez-Silos et al., 2021), as well as forest water-related ecosystem services.

As reviewed by Jones et al. (2022), our understanding of the interplay between afforestation, soil moisture, and temperature dynamics, particularly in the long term, remains limited. This knowledge gap is primarily attributed to the different impacts of different stages of forest growth on soil moisture dynamics which result from variation in tree water use, evapotranspiration (Liang et al., 2018; Porporato et al., 2002) and the co-evolution of roots and soil structure (Carminati, 2013). Multi-annual observations are critical for properly addressing this knowledge gap. By making multi-annual observations, it also become possible to capture the effect of external disturbances, such as drought events, on the internal resilience of young forests (Au et al., 2022). Integrating this valuable information will aid in bridging the existing knowledge gap, which is crucial for supporting efforts to better understand afforestation and restoration approaches enhancing their efficacy.

This paper presents the findings of a study that compared soil moisture and temperature dynamics between a juvenile forest and a neighbouring mature forest ecosystem. While the initial focus was to understand the different hydrological processes in the two forest ecosystems, a drought event impacted the internal dynamics of the forests. As a result, this study presents not only different hydrological processes in two forest systems, but also investigates the hydrological evolution of a juvenile forest ecosystem following external perturbations. The multi-annual observations facilitated this opportunity, enabling a comprehensive understanding of how differently juvenile forest stand responded to external disturbances like drought and how it could provide water ecological functions similar to those of a mature woodland, such as flood regulation.

## 2 | MATERIALS AND METHODS

### 2.1 | Field site description

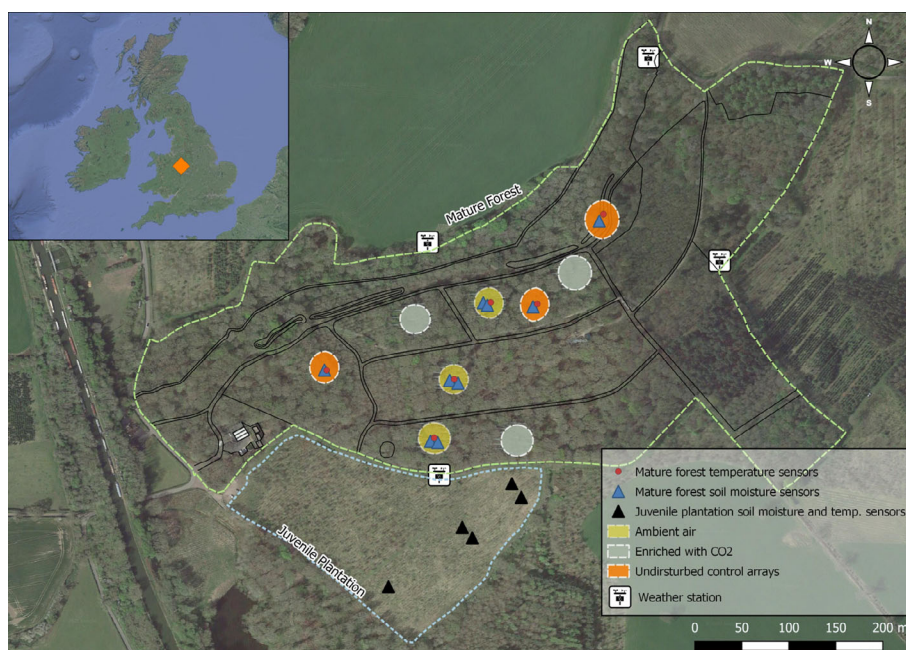
The study site is located in rural Staffordshire, Central England, UK and consists of a mature temperate deciduous forest patch hosting the Birmingham Institute of Forest Research Free-Air Carbon Dioxide Enrichment (BIFoR FACE) facility (Hart et al., 2019; MacKenzie et al., 2021) and an adjacent juvenile forest plantation established in 2014 over previous arable farmland (52°48'3.6" N, 2°18'0" W, 106 m above mean sea level (amsl)). The annual average above-canopy temperature was 10.6 ( $\pm 0.8$ )°C; using 1 min average from 2016 to 2019, with a maximum of 24.3°C and a minimum of -4.6°C. The annual rainfall averaged 676 ( $\pm 66$ ) mm.

The BIFoR FACE facility consists of six approximately circular experimental arrays of 15 m radius with an open infrastructure 2–3 m taller than the tree canopy (Hart et al., 2019). Three arrays are treated with CO<sub>2</sub>-enriched air to maintain +150 ppmv above ambient CO<sub>2</sub> (eCO<sub>2</sub>) during daytime and growing season, and three are treated with ambient air only (control, shown in yellow in Figure 1). In addition, the facility includes three additional arrays with no physical infrastructure (undisturbed, shown in orange in Figure 1). The FACE infrastructure was built in 2015 causing minimal disturbance to the forest (Hart et al., 2019). The eCO<sub>2</sub> treatment started in April 2017 and has been active during leaf out daylight hours April to October, inclusive, each year since. The BIFoR FACE forest (from now on called the 'mature forest', Figure 1) is a mature deciduous woodland covering approximately

19 ha (Hart et al., 2019), dominated by *Quercus robur* (English oak) planted around 1850, coppiced for *Corylus avellana* (common hazel) and remaining largely unmanaged for the last three decades until 2015. Other native tree species including *Crataegus monogyna* (common hawthorn), *Acer pseudoplatanus* (sycamore maple) and *Ilex aquifolium* (holly) have self-set sporadically (Hart et al., 2019). The average height of the oak trees is 25 m, while the other sub-dominant species usually reach approximately 10 m, creating a dense and complex multi-layered canopy structure. The soil depth of the mature forest presents a generally sandy loam layer at a depth of 40 cm, and mostly loamy sand and sand conditions until 1 m (Hollis et al., 2021). The underlying geology is composed of superficial till deposits with a limited area of glaciofluvial deposits overlying sandstone of the Helsby Sandstone formation (BGS, 2020). The adjacent plantation (Figure 1), is a 4.66 ha aggrading mixed-species deciduous woodland (named 'juvenile plantation', Figure 1) planted in spring 2014 over a previously ploughed field (plough depth 0.4 m), characterized by a soil depth of approximately 1 m with mostly sandy loam characteristics. The estimated available water holding capacity (AWC) at both sites is 1.2 mm per cm depth of soil, based on the predominant sandy loam conditions (Table 1).

The juvenile plantation was afforested with 75% *Quercus robur* (English oak), 8% *Betula pendula* (silver birch), 7% *Prunus avium* (wild cherry), 5% *Corylus avellana* (common hazel), and smaller amounts of other deciduous tree species (detail in Supplementary information S1). Saplings less than 1 m tall were planted at a density of 2500 trees ha<sup>-1</sup> in rows approximately 2–2.5 m apart, with 1.8 m distance between trees within the same row.

**FIGURE 1** Field site map. Paired plots of juvenile plantation (blue dotted outline) and mature forest (green dotted outline) with the latter placed within the BIFoR FACE facility, as well as sensor locations in the FACE facility's control arrays (treated with ambient air, in yellow).



**TABLE 1** Available water holding capacity (mm/cm depth of soil) for both sites.

Texture	Field capacity (FC)	Wilting point (PWP)	Available water (AWC)
Sandy loam	2.0 mm/cm depth of soil	0.8 mm/cm depth of soil	1.2 mm/cm depth of soil

Note: Figures are reference averages and vary with soil structure and organic matter content (Blencowe et al., 1960).

## 2.2 | Experimental setup

The data analysed in this study cover the period from 4th April 2016 until 1st March 2020. Air temperature, precipitation, soil moisture and soil temperature were taken at a sub-hourly resolution and were aggregated into hourly and daily data before analysis. Prior to March 2019, weather data was collected from the 'Weather Underground' sensor network (<https://www.wunderground.com/>) using a range of weather stations in the village of Gnosall, approximately 3.5 km from the site. In early 2019, four heavily instrumented meteorological towers were installed outside the boundary of the mature forest and equipped with rain gauges (TR-525M, Texas Electronics) and air temperature sensors (HMP155, Vaisala) at 25 m. Furthermore, 30-year average precipitation and temperature data for climatological normal (CN) calculation were acquired from the MET office weather station at Shawbury (MET office, 2020), located nearly 20 km from the research site. Soil moisture was measured by two different sets of probes. In the mature forest, CS655 probes, Campbell Scientific (0.12 m) were inserted diagonally from the surface inside the control and undisturbed BIFoR FACE arrays (Figure 1; the CO<sub>2</sub>-enriched arrays were not used in this study to avoid potential interferences of the elevated CO<sub>2</sub> regime on the soil moisture dynamics). Each location currently hosts three sensors in triangular formation spaced approximately 1 m apart about 2 m from the nearest mature *Q. robur*. A group of 2–3 sensors in close proximity were averaged and considered as a single point measurement.

The number of sensors was gradually expanded over time, with six locations recorded since October 2016, seven since March 2017 and nine since December 2018. Soil temperature was measured by 107 thermistors, Campbell Scientific, buried horizontally at 0.1 m since early 2018 (one location per array). Soil moisture sensors were installed in the juvenile plantation in the summer of 2015. Soil moisture and soil temperature were measured with 5TM probes, Decagon Devices (now METER group), buried at five locations at 0.1 m (one sensor per location; Figure 1). One of these locations was permanently damaged in winter 2017 leaving four locations active for the remaining period. The soil moisture data analysed in this study cover the period from 4th April 2016 until 1st March 2020. For this specific period, data from 64 observations (from 28th March 2019 to 23th May 2019) were missing in the juvenile plantation dataset and were filled using linear interpolation (detail in Supplementary information S2) to allow the time series decomposition.

## 2.3 | Juvenile plantation monitoring

Understanding the interactions between climatic and hydrological conditions, site conditions, general forest dynamics, and relative land use changes relies strongly on knowledge of tree growth. Diameter at breast height (DBH) and tree height represent the key measurements for describing forest stand growth and density, as well as suggesting appropriate management strategies. We collected data on the DBH at 1.35 m and tree height using a combination of an extensible height

stick and clinometer for the tallest trees during two surveys in 2019 and 2021. Using these measurements, we calculated the basal area as a sum of the transversal areas at 1.35 m of all the trees surveyed within the plantation. Finally, by comparing data from the two surveys, we estimated the mortality rates across species.

## 2.4 | Time series analysis

Analysing the dynamics of soil moisture over time is essential for determining the relationship between forest development and hydrological processes. Soil moisture patterns vary annually and seasonally due to changes of precipitation regime, forest phenology (i.e., leaf-on vs. leaf-off), forest growth, and soil properties. In this context, time series decomposition became an important tool for identifying drivers of hydrological processes change by isolating the salient features of the data in sub-series. We transformed the soil moisture dynamics of the mature forest and juvenile plantation using the seasonal and trend decomposition using the LOESS (STL) procedure (Cleveland et al., 1990) over a 5-year time period (2016–2020). This method performs an additive decomposition that allows the trend ( $T_T$ ), seasonal ( $S_T$ ) and remainder components ( $R_T$ ) (Harvey & Peters, 1990) to be isolated and analysed (Equation 1):

$$Y_T = T_T + S_T + R_T \quad (1)$$

The core of the process is the Loess smoother application which defines the explanatory variables, the value closest to the point whose response is being estimated by fitting a locally weighted polynomial regression over a number of observations. The STL procedure is carried out via an iterative cycle consisting of an inner and outer loop. Each passage of the inner loop applies a moving average smoother to the seasonal and trend components while the subsequent outer loop calculates the remainder component. Finally, the Loess process is used to compute the weight of the remainder component's extreme values. Further iterations of the inner cycle uses the calculated weights to reduce the effects of extreme values identified by the previous outer loop. For more detailed information about the STL method, see Cleveland et al. (1990). The STL function in the R software (R Core Team, 2022) used for time-series analysis required the definition of the seasonal component smoothing parameter  $n_s$  which should be an odd integer number corresponding to the years of observation. In our case, given the study period from 2016 to 2020,  $n_s$  was set to 5.

## 2.5 | Events analysis

To characterize the mature forest and juvenile plantation response to single meteorological events we selected four 2-day periods with precipitation events of contrasting magnitude (5 and 20 mm) for 2016 and 2019. In addition, two 2-month summer periods from mid-June to mid-August in 2016 and 2019 were selected and analysed using the statistical methods described below.



## 2.6 | Temperature regulation

We investigated the canopy cover thermoregulation capacity in both the mature forest and juvenile plantation by monitoring the mean daily air and soil temperatures of topsoil (10–12 cm) from 2018 to 2020. The temperature difference ( $\Delta T$ ) was calculated between air temperature and topsoil temperature.

## 2.7 | Statistical methods

We used two statistical methods to determine the effects of the dry periods on soil moisture dynamics. Firstly, to evaluate the difference between soil moisture observations in both mature forest and juvenile plantation we applied the two-sample Kolmogorov–Smirnov test (KS; Hazewinkel, 2001) using the *stats* package in R. This approach compares two samples determining whether they are from the same distribution (null hypothesis,  $H_0$ ), or distinct ones (alternative hypothesis,  $H_1$ ). The test is carried out at a predefined statistical significance level  $\alpha$  of 5%. Given a first sample of size  $m$  with an observed cumulative distribution function  $F(x)$  and a second sample of size  $n$  with an observed cumulative distribution function of  $G(x)$ , the Kolmogorov–Smirnov statistic  $D$  is given by:

$$D_{n,m} = \max |F(x) - G(x)| \quad (2)$$

If  $D_{n,m}$  is greater than  $D_{n,m,\alpha}$ , the null hypothesis at significance level  $\alpha$  is rejected where  $D_{n,m,\alpha}$  is the critical value. For sufficiently large  $m$  and  $n$ :

$$D_{n,m,\alpha} = c(\alpha) \sqrt{\frac{m+n}{mn}} \quad (3)$$

Secondly, we applied the piecewise linear regression (PLR) method, also known as segmented linear regression, to investigate the difference in the system's temporal response to the dry periods. In ecological studies, this analysis has been widely used (Ficetola & Denoël, 2009; Shea & Vecchione, 2002; Toms & Lesperance, 2003;

Toms & Villard, 2015) to identify thresholds that reflect step changes in the studied variable dynamics or the processes governing them. The statistical model for only one breakpoint at  $t = \varphi$  is given by:

$$y_i = \begin{cases} \beta_0 + \beta_1 t_i + e_i & \text{for } t_i \leq \varphi \\ \beta_0 + \beta_1 t_i + \beta_2 (t_i - \varphi) + e_i & \text{for } t_i > \varphi \end{cases} \quad (4)$$

where  $y_i$  is the value for the  $i$ th observation of soil moisture,  $t_i$  is the corresponding value for the independent variable,  $\varphi$  is the breakpoint (the threshold), and  $e_i$  are assumed to be the independent, additive errors with mean zero and constant variance. The slopes of the lines are respectively  $\beta_1$  and  $\beta_1 + \beta_2$ , so  $\beta_2$  can be considered as the difference in slopes. The *strucchange* package in R (Zeileis et al., 2002) has been used to determine the optimal number of breakpoints and their temporal location.

## 3 | RESULTS

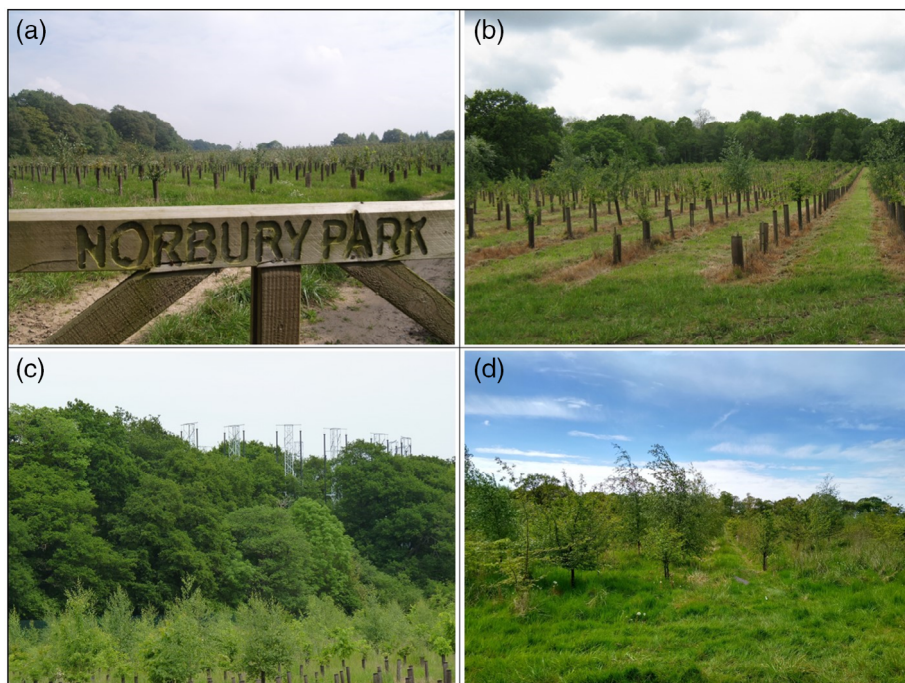
### 3.1 | Juvenile plantation survey

The first tree survey in the juvenile plantation was carried out in January 2019, with a tree density of 1340 tree  $\text{ha}^{-1}$  estimated. Because of the limited dimension of tree diameter, only the heights were measured during this survey. The average height of *Quercus robur* was 1.6 m, ranging from a maximum of 2.5 m to a minimum of 1 m. *Betula pendula* and *Prunus avium* had higher average height values of 3.4 and 3.2 m with maximum values of 5 and 4.5 m, respectively. During the second tree survey, which took place in 2020, the estimated density was 975 trees  $\text{ha}^{-1}$  with a basal area of 1.2  $\text{m}^2 \text{ha}^{-1}$ . *Quercus robur* and *Prunus avium* had higher death rates, losing 33% and 16% of their respective population (Table 2).

In comparison, the average oak height increased to 2 m, with some individuals reaching 3.3 m. Similarly, *Betula pendula* and *Prunus avium* reached an average height of 5 and 4 m, respectively, with maximum values of 6.50 and 6 m. Figure 2 shows the growth of juvenile plantation from 2014 to 2019.

**TABLE 2** Average, maximum, minimum heights, as well as mortality rates across all species within the juvenile plantation following a tree survey in 2020.

Common name	Latin name	Average height (m)	Maximum height (m)	Minimum height (m)	Mortality rates from 2019 survey
English Oak	<i>Quercus robur</i>	2	4.50	1	33%
Silver Birch	<i>Betula pendula</i>	5	6.50	3	3%
Wild Cherry	<i>Prunus avium</i>	4	6	1.50	16%
Hazel	<i>Corylus avellana</i>	2.50	4.50	1	5%
Hornbeam	<i>Carpinus betulus</i>	3	4.50	1.50	0%
Rowan	<i>Sorbus aucuparia</i>	3.5	4.50	1	8%
Field Maple	<i>Acer campestre</i>	3	4	1	0%
Chestnut	<i>Castanea sativa</i>	2	4	1	22%
Whitebeam	<i>Sorbus aria</i>	2	2.50	0.70	0%
Wild Pear	<i>Pyrus communis</i>	3	3.50	1.50	0%
Crab Apple	<i>Malus sylvestris</i>	2.7	3	2.50	33%



**FIGURE 2** Juvenile plantation stages from planting in 2014 (a) to further development in 2016 (b), 2017 from a different camera angle showing both the mature forest in BIFoR (top) and juvenile forest (bottom) (c), and 2019 (d) respectively. In panels B and C, early tree mortality can be observed in the foreground.

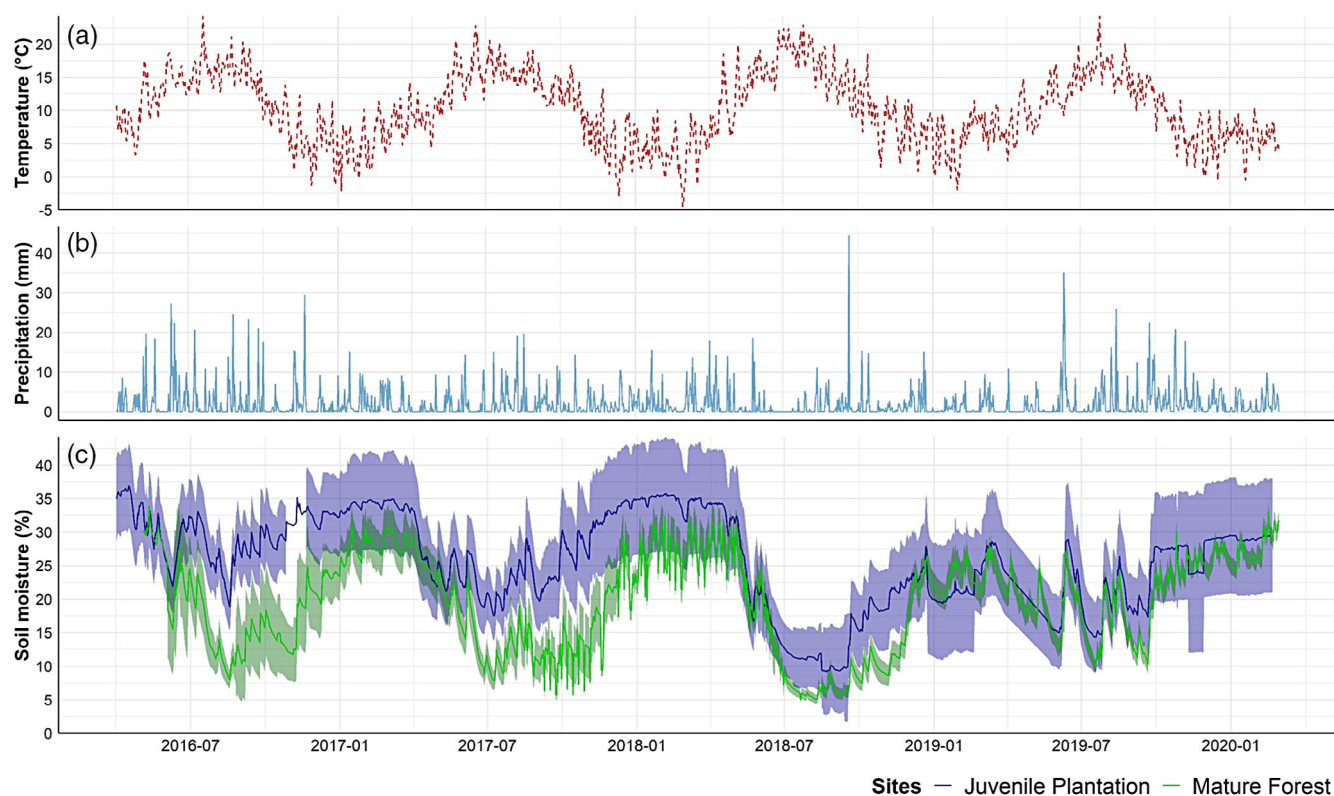
### 3.2 | Soil moisture dynamic and hydrometeorological conditions

Average daily soil moisture at the juvenile plantation and mature forest from 2016 to 2020 is shown in Figure 3, in which two distinct phases are identified. The first period, from April 2016 to June 2018, was characterized by a similar seasonal pattern for both sites (Figure 3c). From September to April, soil moisture levels generally increased, with minimum and maximum values of 8%–32% and 20%–36% in mature and juvenile forests, respectively. Peak soil moisture values synchronized for all years with some minor delays in the mature forest compared to the juvenile plantation (Figure 3c). When considering seasonal variability, the largest absolute difference in soil moisture is observed from early summer to mid-autumn (e.g.,  $\Delta_{sm} = 15\%$ ) compared to more similar soil moisture observations in winter and spring. This seasonal pattern is also reflected in the standard error (SE) that shows a larger range in summer compared to winter. Substantial precipitation events in 2016 and 2017 resulted in a soil moisture increase, but the absolute difference between the forests remained consistently larger over this first period. Note that the SE for the mature forest is smaller due to the larger aggregated samples in the studied arrays. The second identified period starts in June 2018 and co-occurs with a dry period. In the following months, soil moisture in the mature and juvenile plantation declined to minima of 5% and 10%, respectively. This extended drought ended with heavy precipitation events (1 August and 30 September), which only partially restored seasonal soil moisture levels. The absolute difference in soil moisture was altered after this dry stage, resulting in similar soil moisture conditions in both forest types, both on seasonal and annual time scales. For example, soil moisture in 2019 varied only 3%–4% on average from January to July, based on a total rainfall of 338 mm, which is nearly the same amount that fell during the same period in 2018

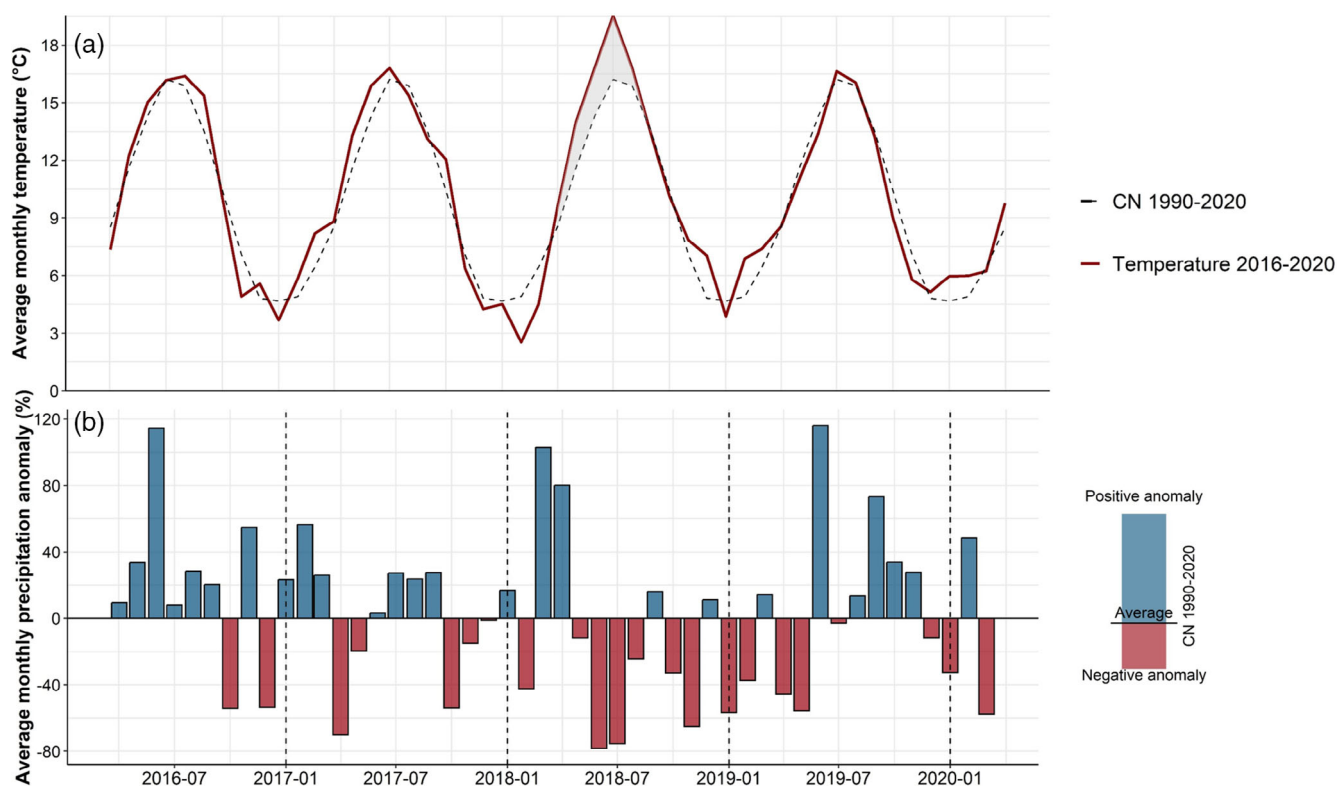
(341 mm). The SE for the juvenile plantation increased during this period peaking at nearly 22% in late 2019. In the mature forest, however, the SE decreased over time due to installation of additional soil moisture probes overtime (MacKenzie et al., 2021).

Figure 4 depicts the monthly average temperature and percentage precipitation anomalies from 2016 to 2020 in relation to the 1990–2020 climatological normal (CN). In particular, 2018 differed more from the other years of the series (Figure 4a), with below-average temperature ( $-1.5^{\circ}\text{C}$ ) winter months followed by a 10-month period characterized by several positive thermal anomalies (peaking in July with  $+3.3^{\circ}\text{C}$ ) that ended in early 2019. Related to this, Figure 4b highlights how summer months of 2018 were affected by a severe drought, measuring  $-1.7$  SPI across the UK (UKCEH, 2022), with annual rainfall being reduced to 79% in July. Despite the period 2018–2019 being drier and warmer than normal, the start of 2019 represented a shift to generally wetter and colder conditions. In 2019, the mean annual air temperature dropped to  $+9.75 (\pm 4.28)^{\circ}\text{C}$ , in line with the CN, while a mean annual precipitation of  $775 (\pm 35.4)$  mm represented an estimated 19% increase compared to 2018. However, when focusing on the summer months of 2018 and 2019, the difference in precipitation is estimated to be 60% with 148.6 and 368 mm, respectively.

Decomposition of soil moisture time series revealed significant differences between the juvenile plantation and the mature forest (Figure 5a). Although soil moisture decreased for both forest systems from 2016 to 2018 (Figure 5b), the decline was greater in the juvenile plantation than in the mature forest (KS test:  $D = 0.590$ ,  $p < 0.05$ ). In 2019, soil moisture in the mature forest returned to pre-drought levels in less than 1.5 years, whereas soil moisture in the juvenile plantation reached a 10% lower value than before (Figure 5b). The application of piecewise linear regression (details can be found in Supplementary information S3) to soil moisture trends identified three

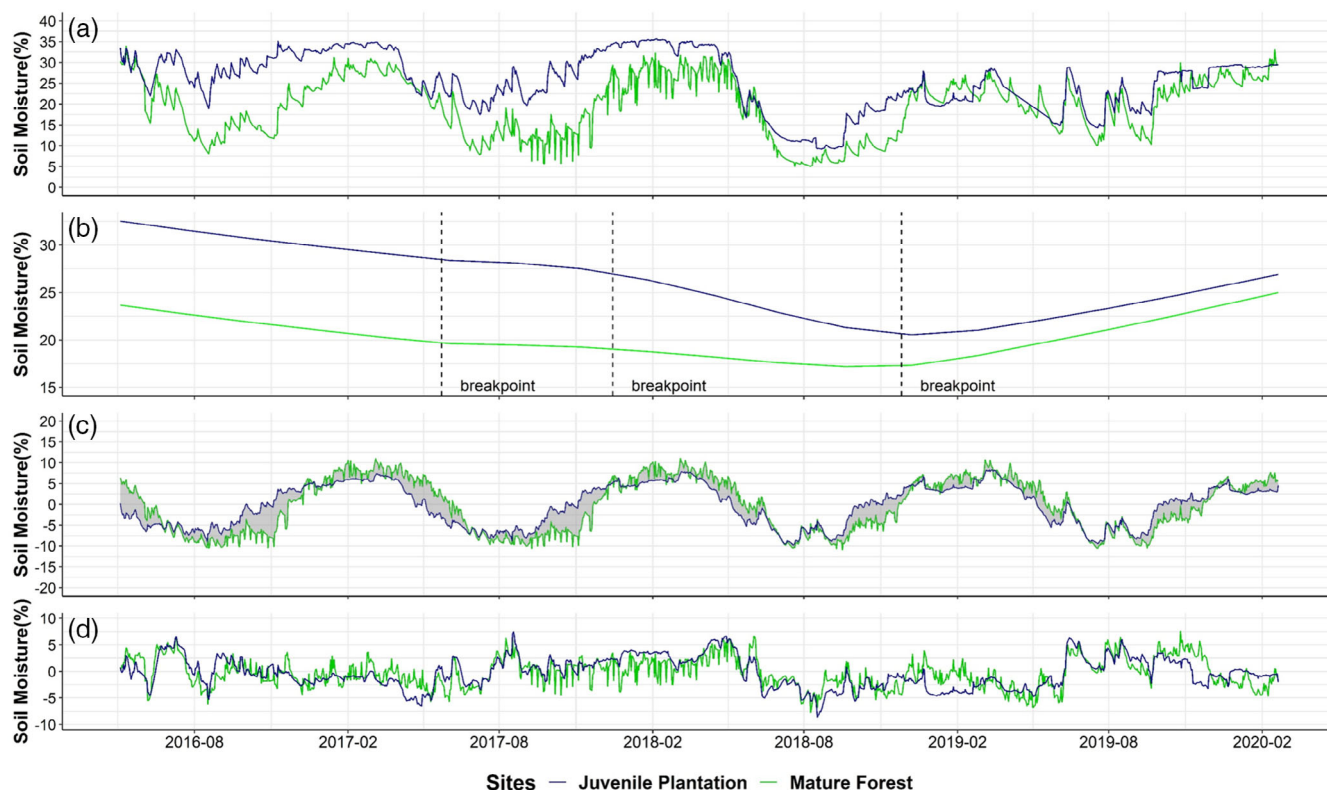


**FIGURE 3** Daily air temperature (a), precipitation (b), and soil moisture (c) observations for the juvenile plantation and mature forest. Daily mean soil moisture plots (C) for both plots include the relative standard error of the mean (SE). Note that soil moisture observations at BIFoR FACE are only used for control and undisturbed arrays.



**FIGURE 4** Monthly averages of air temperature (a) and precipitation (b) in the research area from 2016 to 2020 in relation to the monthly climatological normal (CN) calculated for the period 1991–2020. Thirty years weather data acquired from a weather station located at Shawbury, 15 km away (UK Met office, 2020).





**FIGURE 5** Soil moisture time series decomposition over the period 2016–2020. Daily mean soil moisture at mature and juvenile forest (a); long-term trend in soil moisture with dashed lines representing the breakpoints (b); soil moisture seasonal component with grey filling area highlighting the difference between both forest ecosystems (c); soil moisture residual component (d).

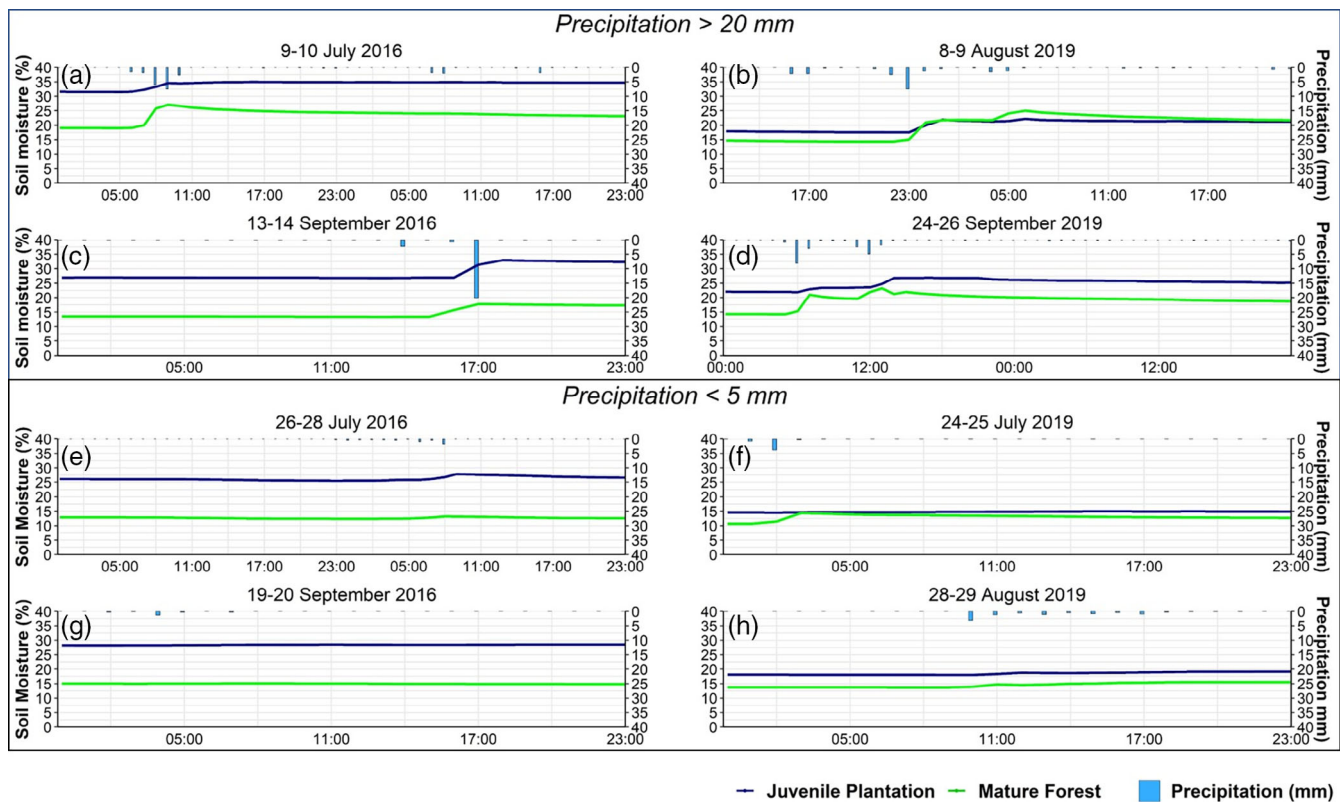
major breakpoints that occurred in May 2017, December 2017, and November 2018 (dashed line in Figure 5b). The slopes of linear regression calculated between December 2017 and November 2018 are  $-0.030$  and  $0.020$ , respectively, indicating multiple distinct soil moisture dynamics that coincide with the identified two periods in Figure 3. Figure 5c also depicts the progressive synchronization of seasonal soil moisture dynamics, as evidenced by the reduction of the summer soil moisture difference between juvenile and mature forest from 2016 (KS test:  $D = 0.46$ ,  $p$ -value  $< 0.05$ ) to 2019 (KS test:  $D = 0.15$ ,  $p$ -value  $= 0.2435$ ). Residual components (Figure 5d) of soil moisture in both forest sites confirm these findings, with their dynamics aligning more during the summer of 2019.

### 3.3 | Event based analysis of temporal differences in soil moisture responses

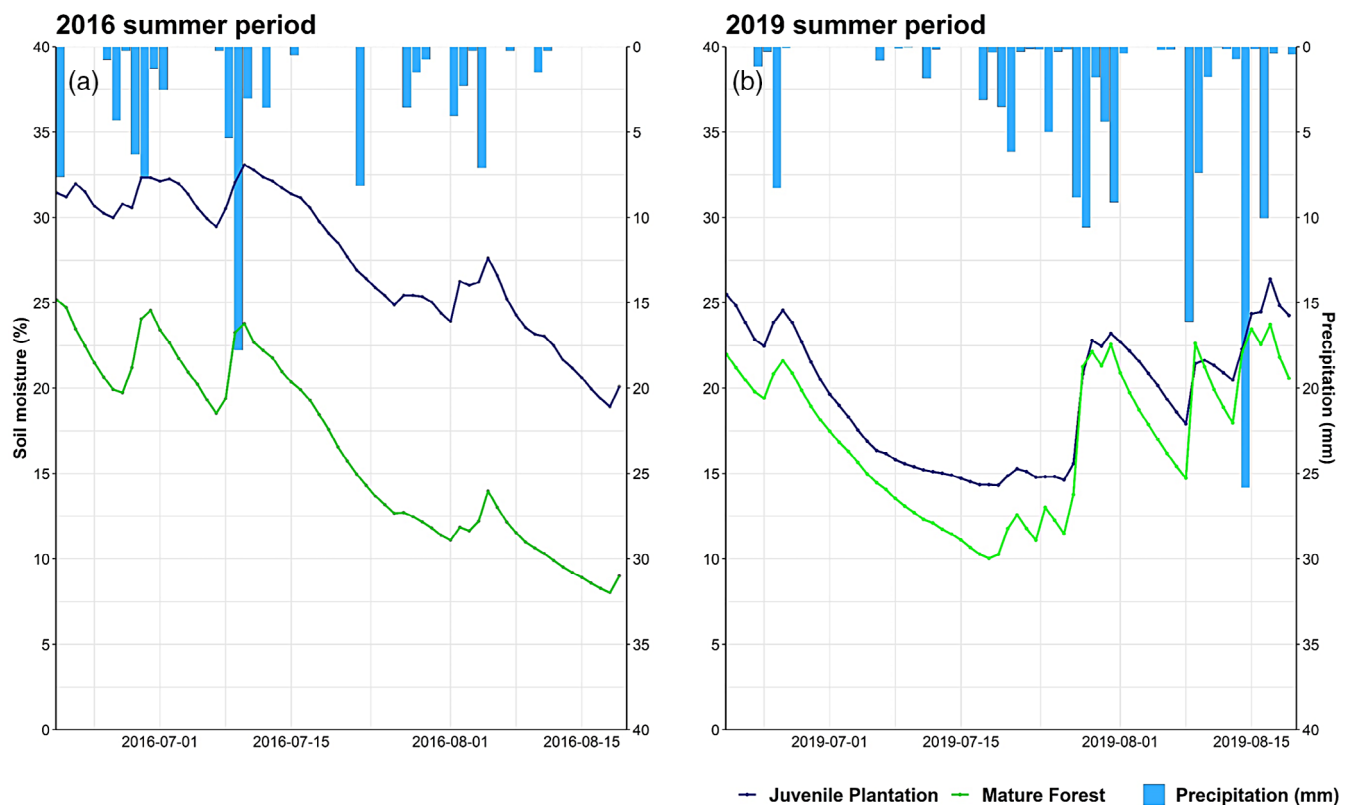
Figure 6 shows hourly mean soil moisture in juvenile and mature forest measurements in response to selected precipitation events ( $P > 20$  mm and 5 mm) during the summers of 2016 and 2019. Regardless of precipitation magnitude, juvenile plantation had higher soil moisture values in 2016 compared to the mature forest (KS test:  $D = 0.54$ ,  $p$ -value  $< 0.05$ ), with an average difference ranging from 12% to 15%. In 2019, however, this dissimilarity between the two sites is no longer significant (KS test:  $D = 0.12$ ,  $p$ -value  $> 0.05$ ) with the soil moisture differences reduced to 5% or less. Additionally,

mature forest soil moisture is generally lower than that in the juvenile plantation, but this pattern was reversed on 8–9 August (Figure 6). Soil moisture exceeded that of the plantation by approximately 3%, following a 23.5 mm rainfall event. Overall, soil moisture in the mature forest increases more compared to the plantation following heavy rain events. On the other hand, it appears that smaller events ( $P < 5$  mm) cause negligible soil moisture response at the monitored soil depths in all cases (Figure 6).

To further characterize the two identified periods that show pre- and post-drought responses of soil moisture dynamics in both forest ecosystems, we selected two summer periods in 2016 and 2019. The first summer period, which lasted from 20 June to 20 August 2016 had total rainfall of 90.7 mm and an average temperature of  $+15.3$  ( $\pm 1$ )°C. During this time, both the mature forest and juvenile plantation showed a progressive soil moisture decline of 16% and 14%, respectively, reaching lows of 10% and 20% (Figure 7a). The second selected summer period in 2019 (from 20 June to 20 August) had higher precipitation (total of 130.2 mm) and slightly warmer conditions than 2016 (mean temperature of  $16.03$ °C). The soil moisture showed an interesting dynamic here, as the first month was characterized by drier conditions (19.6 mm) resulting in a 12% and 10% decrease for both mature forest and juvenile plantation, respectively (Figure 7b). Following this initial period, increased precipitation (104.4 mm) resulted in a similar sharper soil moisture response in both forest ecosystems, nearly restoring the initial conditions. Despite forest soil moisture dynamics continuing to differ significantly, there is a



**FIGURE 6** Effect of selected precipitation events on soil moisture. Panels show hourly soil moisture at the mature forest and juvenile plantation after precipitation events of different intensity ( $P > 20$  mm and 5 mm) in summers 2016 and 2019.



**FIGURE 7** Soil moisture in mature forest and juvenile plantation relative to the summer 2016 (panel a) and summer 2019 (panel b).

shift toward a new distinct period, with the stark difference between systems becoming minimal (KS test:  $D(2016) = 0.71$ ,  $D(2019) = 0.34$ ,  $p\text{-value} < 0.05$ ).

### 3.4 | Buffering impact on air temperature

From 2018 to 2020, the average daily dynamics of air and topsoil temperatures show that the mature forest buffers large temperature differences. Its microclimate is effectively modulated with topsoil temperature showing decreasing by only 0.2% over this period. This is remarkable given that 2019 had nearly one degree below the mean annual air temperature of  $+9.77^{\circ}\text{C}$  (Figure 8a). Seasonally, the mature forest topsoil temperature is characterized by higher temperature during the colder periods and lower temperatures in warmer ones compared to air temperature (KS test:  $D = 0.18$ ,  $p\text{-value} < 0.05$ ). The extent of the mature forest buffering effect is clearest between 1 May and 1 September 2018, when there is  $4^{\circ}\text{C}$  average difference between the topsoil and air temperature (Figure 8b). For the same period, the buffering effect of the juvenile plantation is far inferior, with average differences between topsoil and air temperature less than  $1^{\circ}\text{C}$  (KS test:  $D = 0.06$ ,  $p\text{-value} > 0.05$ ; Figure 8c).

## 4 | DISCUSSION

### 4.1 | Increasing similarity in soil moisture dynamics between juvenile plantation and mature forest

Based on the presented soil moisture observations, the juvenile plantation showed a more conservative water balance, less fluctuation

between soil moisture contents, than the mature forest (Figure 3). Soil moisture conditions depend on different water fluxes that relate to vegetation type, hydrometeorological conditions, and soil properties (Seneviratne et al., 2012). Despite having comparable soil textures (Table 1), the mature forest and juvenile plantation exhibit variations in soil properties and soil moisture dynamics due to their different histories. In particular, the juvenile plantation, which was only 6 years old at the time of the drought reported in this study, is characterized by an open canopy (Figure 2) and possibly low levels of organic matter in the soil. In fact, afforestation efforts often take several decades to improve various soil properties (Korkanç, 2014; Mongil-Manso et al., 2022; Yao et al., 2023). As a result, the variation in soil moisture observed within the juvenile plantation is most likely caused by the increased physiological water demand of the trees after the drought, as well as differences in forest structure compared to the mature forest.

The developed canopy structure of an undisturbed forest ecosystem, for example, plays an important role in regulating soil moisture oscillation due to increased rainfall interception and reduced throughfall, as well as shade that influences the understory microclimate and the evaporation from the soil (He et al., 2013). Similarly, water consumption to meet photosynthetic water demand in the mature forest is likely to result in higher ET losses and soil water deficits due to a more developed root network (Landsberg et al., 2017). Interestingly, the developed ground cover vegetation in the juvenile plantation does not appear to cause additional soil water uptake, as our results show over the period preceding the drought, which would reduce soil moisture to a similar level as in the mature forest (3C). This relates to the principal difference in functional rooting depth between trees and grasses, implying that trees respond primarily to deeper soil moisture availability (10–20 cm) compared to grasses that have a larger



**FIGURE 8** Difference between topsoil temperature and air temperature observed at the mature forest and juvenile plantation (2016–2020). Panel a shows daily air temperature and topsoil temperature collected at the control and undisturbed arrays within BIFoR FACE, panels b and c show the daily difference between topsoil and air temperature for both sites.

presence in the shallow soil layers (Weltzin & McPherson, 1997). This interpretation is coherent with the moisture dynamics at the different depths within the juvenile plantation during the dry stress period, where similar dynamics in soil moisture can be detected (details in the supplemental material S4). In this context, the drought acted effectively as a 'renewal event', eliminating previous moisture dynamics (Viola et al., 2008). Although vertical moisture profile dynamics is critical for understanding the overall response and adaptation to drought, the 10 cm depth is a reliable indicator of the hydrological processes in action. Furthermore, the 10 cm depth provides valuable information on the development of flood capacity regulation by the juvenile plantation (Figures 6 and 7), which is a valuable water ecological service provided by forest ecosystems (Jenkins & Schaap, 2018). This effect of functional root depth is also reflected in the analysis of high rainfall events, where the mature forest shows a greater adaptability to uptake soil moisture. This could be related to retaining soil moisture in deeper soil layers and/or a more extensive root network (Holdo & Nippert, 2015), although this falls outside the direct scope of this research.

Notably, Figure 5 shows how soil moisture changes highlight differences in resilience (Lloret et al., 2011) between the two forest ecosystems, with the mature forest showing higher resistance and recovery. Nevertheless, differences in soil moisture between the mature forest and juvenile plantation became smaller beginning in September 2018. This pattern becomes more evident analysing the soil moisture trends (Figure 5b) and the progressive alignment of seasonal soil moisture (Figure 5c) which suggests that the hydrological functionality of both forest ecosystems becomes similar after the 2018 drought. One explanation might be the rapid growth of trees observed within the juvenile plantation, which was 6 years old in 2018 (Figure 2). This increases overall the associated water photosynthetic needs with greater contribution from the fast-growing species (Cao et al., 2011) such as *Betula pendula*, *Prunus avium*, and *Corylus avellana* accounting for 20% of the total population. A mixed species plantation is likely to exhibit higher growth rates and carbon storage even during dry periods compared to monoculture due to their different water and nutrient acquisition strategy (Liu et al., 2018). Furthermore, the change in soil moisture dynamics could relate to dry and warm conditions, which forced the trees to adapt their belowground surface to maintain plant water and nutrient uptake. Trees can use one of two strategies to overcome periods of water stress: either increase fine root biomass (FRB) formation to maintain absorbing surface (extensive approach) or modify root morphology and physiology to maximize uptake efficiency per root mass (intensive approach) (Löhmus et al., 2006). It has been shown before that drought-induced decreases FRB can be compensated by higher growth rates (Joslin et al., 2000) during favourable periods, resulting in soil moisture dynamic changes as shown, for instance, by the juvenile plantation seasonal dynamics. Related to these studies, it could be that the meteorological drought in 2018 (SPI-1.7. UKCEH, 2022) and persistent higher topsoil temperature in the juvenile plantation (Figure 8a) could be driving root growth (Joslin et al., 2000; Kwatcho Kengdo et al., 2022; Salazar et al., 2020). Similarly, mycorrhizal network

growth pulses may play an important role in forest ecosystem resilience (Simard et al., 2012). Tree C allocation to mycorrhizal fungi represent an essential strategy in sustaining nutrient and water uptake over stress periods (Hawkes et al., 2011; Simard et al., 2012; Wang et al., 2021). In fact, during a drought an extensive hyphal network that allows for water movement and redistribution (Bingham & Simard, 2011) can be critical to plant survival (Neumann & Carbon, 2012; Querejeta et al., 2007) and resilience. Finally, more research is required to understand how severe perturbations, such as droughts, can erode natural soil structure resilience, potentially leading to an alternative stable state in an ecosystem for either mature forests or juvenile plantations. Hydrological extremes, such as droughts, can alter soil hydraulic parameters, resulting in different soil moisture trajectories with a domino effect on all terrestrial ecosystems processes like soil respiration, nutrient cycling, and net primary productivity (Robinson et al., 2016). Overall, a closer correlation of the water dynamics between the juvenile plantation and the mature forest over time suggests that key aspects of a mature forest hydrological functionality can be achieved on shorter times scales compared to other ecological goals, and likely in response to the drought period. This is an important finding for areas where reforestation is introduced to restore local water resources.

It is also worth noting that trees in both forests are suffering from water stress. Unfavourable climatic conditions, most notably a lack of precipitation, rising air temperatures, and increasingly frequent and extended dry periods, have serious, but varying, long-term implications for mixed forest ecosystems (Češljarić et al., 2022; Niinemets & Valladares, 2006). In particular, juvenile plantations tend to be more sensitive to water stress due to high tree density and the adoption of genetic provenance with low drought tolerance, resulting in lower planting vigour and higher mortality ratio when compared to a naturally regenerated forest (Navarro-Cerrillo et al., 2018). For example, tree oak mortality has been known to occur approximately 2 years after dry stress, which may explain their high mortality rates observed in the juvenile plantation during the 2020 tree survey (Table 2). With this work, we show that long-term observation of soil moisture dynamics in different forest types and management conditions shows differences in drought resilience, but more research is needed to provide guidance to forest management under climate change conditions where dry spells and droughts are expected to occur more often and with increased severity (Dai, 2013; Hari et al., 2020; Stagge et al., 2017).

## 4.2 | Differences in forest soil moisture responses to wetting and drying events

The similarity in soil moisture response is also echoed when focusing on both storm events and dry spells in 2016 and 2019. The 2016 events show that the juvenile plantation had 12% to 15% wetter conditions compared to the mature forest and soils were close to fully saturated at times (Figure 6). In 2019, soil moisture responses in the juvenile plantation were more similar to those of the mature forest



after storm events, indicating an initial buffer effect of the young trees (Figure 6). Summer periods in 2016 and 2019 showed overall higher values for the juvenile plantation indicating inferior water uptake of growing trees when compared to the mature forest (Figure 7). In 2016, the decline in soil moisture is similar in both forest ecosystems considering an absolute difference between systems. This compared to the soil moisture in 2019 that shows several overlapping dynamics in response to precipitation events (Figure 7b). The observed response to wetting and drying events provides further evidence in support of the prior interpretation that the growth (Figure 2) of trees has led to increased water balance buffering capacity. Moreover, it indicates that the juvenile plantation exhibits recovery mechanisms that allow it to recover from water stress. In fact, when compared to a mature forest, the altered response of juvenile plantation soil moisture to storm events in 2019 may indicate increased infiltration due to the growth of the root network (Ilstedt et al., 2016; Jones et al., 2022; Lange et al., 2013; Zhang et al., 2014). This may lead to a change in water storage in shallow soil and relative flood capacity regulation, which is one of the most important water-ecosystem services provided by a forest stand in a watershed (Crossman et al., 2019).

### 4.3 | Buffering impact on air temperature

Given the low fluctuations in topsoil temperature, the mature forest's thermal buffering capacity is evident (Figure 8a). However, the larger fluctuations of the juvenile plantation reveal a significantly lower thermal regulation capacity during the monitored period. This is due to the less developed forest and canopy structure of the juvenile plantation (Figure 2), which also explains the higher (close to the air temperature) soil temperature (Figure 8a), as opposed to the observed reduced summer topsoil temperature and temperature anomalies in the mature forest. In fact, as illustrated in Figure 8b, the mature forest cools summer air and warms the cold winter air, thereby buffering temperature changes (Jin et al., 2019). As a result of extreme climate conditions, temperature differences between inside and outside forests increase (De Frenne et al., 2013). Despite the growing canopy cover, the juvenile plantation has not (yet) attained similar thermal buffering capacity (Figure 8c).

### 4.4 | Limitations and uncertainty in data

While the mature forest investigated here has been thoroughly studied, the neighbouring juvenile plantation lacks detailed tree physiology observations, which limits a detailed interpretation of the role of forest conditions on soil moisture and temperature differences. To move beyond the empirical findings presented here, toward causal mechanistic understanding of the effect of tree growth and forest management on hydrological processes, additional research within the juvenile plantations such as rooting depth, sap flow, and interception losses will be required.

## 5 | CONCLUSIONS

In this work, we compared soil moisture and temperature observations of a juvenile and mature forest that revealed significant differences in seasonal patterns as well as event. The ongoing monitoring shows a transition in hydrological trends, as well as changes in precipitation response, particularly within the juvenile forest. These alterations suggest that there has been a change in water storage in shallow soil as a result of increased infiltration, and subsequently improvement in the ability of the juvenile plantation to control flooding. We found that shallow soil moisture dynamics of the juvenile plantation match those of the mature forest in less than 10 years (8 years after planting), which is most likely due to adaptation strategies of the plantation following the 2018 drought. This event may have promoted modification of the tree root structure (i.e., biomass and morphology) to maintain adequate water and nutrient uptake. Additionally, the impact of drought on the hydrological function of the juvenile plantation highlights a new challenge of climate change in temperate regions which needs to be considered when predicting outcome of reforestation schemes. The observed effects on the soil moisture and hydrological function of the juvenile plantation may be due to the complementarity nutrient and water acquisition strategies of mixed species, which might not be case in traditional monoculture forestry system. Finally, thermal regulation capacity remained dissimilar between juvenile plantation and mature forest suggesting how other forest ecosystem functions take longer to establish. This study highlights the value and the challenges of continuous long-term observation at high frequency to complement the analysis of seasonal and interannual behaviour with investigating the trends in event-based responses. Further research on this long-term observatory will continue to facilitate analyses in ecosystem and hydrological responses to forest management practices, as well as provide invaluable insights into the complex relationship between land use changes, such as reforestation, and drought.

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## DATA AVAILABILITY STATEMENT

The data are available upon request; an open data repository for a subset of core data is under construction.

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## SUPPORTING INFORMATION

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