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Crown die-back of peri-urban forests after combined heatwave and drought was species-specific, size-dependent, and also related to tree neighbourhood characteristics

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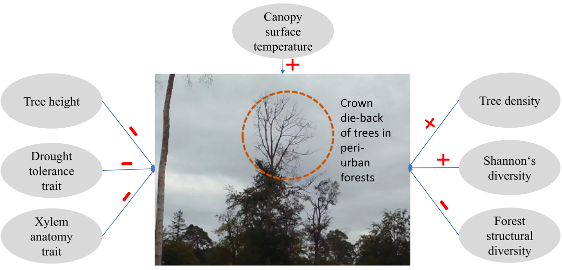
HIGHLIGHTS

- Peri-urban forests experience severe dieback after heatwave-compounded drought events.
- Crown die-back are species-specific and size dependent.
- Among 35 common species, 9 decline, 3 increase dieback significantly along tree height.
- Trees with diffuse porous xylem or high drought tolerance had lower dieback.
- High structural diversity, low Shannon diversity and tree density had lower dieback.

GRAPHICAL ABSTRACT

The hot, dry summer in southwest Germany induced crown die-back of peri-urban forests



Crown die-back pattern are species-specific and size-dependent, and also associated with species drought and xylem traits, high heat load, high tree density, high Shannon's diversity, and low structural diversity (Gini index)

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ABSTRACT

The Rhine River valley of Germany has been facing recurrent and intense spells of drought and heatwaves threatening the health of trees in peri-urban forests. Crown damage intensified by climate change accelerates tree mortality, threatening its ecological, economic, and social benefits; however, the pattern of crown die-back in peri-urban forests remained unclear. We performed a field inventory to estimate the crown die-back of 2578 trees of 51 species from 68 randomly selected peri-urban forest plots in Karlsruhe region on the right bank of the Rhine, after the catastrophic summer heatwave and drought of 2018. We related crown die-back to species-specific drought tolerance, wood anatomical traits, tree size, canopy surface temperature, tree density, Shannon's diversity and Gini coefficient for tree height. Regression results indicate that small-size trees were found to be more susceptible to canopy damage than large trees, with a 1-meter increase in tree height associated with a 0.8 % reduction in crown die-back. This size-dependent process is also species-specific. Among the 12 species with significant ($p < 0.05$) linear relationship between height and die-back, 9 species demonstrated negative correlations and 3 species showed positive relationships. Species tolerant to drought or cavitation (e.g., trees with diffuse porous xylem, 21 species) had significantly lower crown dieback. For example, with a 1-point-scale

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increase in drought tolerance crown die-back declined 14.35 %. Trees that experienced high canopy surface temperature and grew with high tree density and species diversity (Shannon's diversity) had more crown die-back. However, high structural diversity (Gini coefficient) was related to lower crown die-back. Our results suggested that future research should focus more on tree species-specific hydraulic and thermal traits and tree density and structure management to improve tree health and species selection in peri-urban forests under future climate change.

1. Introduction

Peri-urban forests are characterized as woodlands with a minimum of 0.5 ha in size with at least 10 % canopy cover located at the city's periphery or within the administrative boundary of cities, and as an essential green infrastructure, they provide multiple benefits and ecosystem services for citizens (Cueva et al., 2022). The importance of improving the quality of life and mitigating climate change has been brought to public attention, especially during and after the pandemic (Beckmann-Wübbelt et al., 2021). However, peri-urban forests' persistence and ecosystem services are threatened by urbanization and climate change (Esperon-Rodriguez et al., 2022a). Trees are vulnerable to extreme heat and prolonged drought events, which are predicted to be more frequent and severe under fast urbanization and climate change (Choat et al., 2018; Marchin et al., 2022a). Drought and heat stress are usually combined as a heatwave-compounded drought event. Globally, combined heatwave and drought events drive widespread crown die-back, and many trees eventually die when severe crown die-back continues (Hammond et al., 2022; Matusick et al., 2018; Zhang and Brack, 2021). However, those observations mostly focus on forests in rural areas, and the systematic and large-scale assessments crown die-back in peri-urban forests bordering large cities with urban heat islands are rare (Hartmann et al., 2022; Marchin et al., 2022b).

Drought is generally considered the driving or predisposing factor for tree mortality during large-scale forest die-back events (Choat et al., 2018; Matusick et al., 2018). The mechanism that induces tree mortality during drought is usually attributed to two reasons, and one important reason is hydraulic failure that is caused by xylem dysfunction and cavitation; the other one is carbon starvation (Adams et al., 2017; McDowell et al., 2022). Drought-induced hydraulic failure and heatwave increased stomatal conductance, exacerbating hydraulic stress (Marchin et al., 2022a; Matusick et al., 2018). Trees experiencing different heat loads are supposed to have varied susceptibility to crown damage, concurrently tree species with different drought tolerance and xylem anatomy types may respond in different ways and have different die-back percentages under the same drought stress (Bennett et al., 2015; Camarero, 2021; Marchin et al., 2022b). Tree health and mortality under heat and drought stress have become a topic of concern among researchers and other stakeholders as climate change accelerates (Choat et al., 2018; Marchin et al., 2022b; Matusick et al., 2018). However, predicting tree mortality remains an unattainable task, and the application of artificial intelligence in the field of tree failure prediction may provide another possibility (Jahani and Saffariha, 2021). Possible predictors of tree failure include tree physiology, site factors, and ecological interactions with other organisms (Trugman et al., 2021). Urban and peri-urban forests are compounded social-ecological systems, the prediction of tree failure is important for forest hazard management (Jahani and Saffariha, 2021) and the purposeful removals by humans can also be a key element for urban tree mortality (Hilbert et al., 2019). Using species distribution and climate data to predict future forest transformation has been discussed a lot (Esperon-Rodriguez et al., 2022a, 2022b), but species geographic distribution not always reflect their climate change adaptation capacity, especially when it comes to vulnerability of trees to crown die-back in peri-urban environment that is intrinsically hotter than forests in rural areas (Bocsi et al., 2016; Hanley et al., 2021). Also, these data are incomplete and may not necessarily reflect the species' fundamental niches. Climate similarity and geographical proximity cannot predict species composition in peri-urban forests, thus, on-site species-specific investigations are required to select suitable tree species for future peri-urban forests, although studies are rare (Hanley et al., 2021).

The size and vitality of trees in forests are related to sustainable supply of ecosystem services. The failure of trees in peri-urban forests can dramatically change the urban landscape and reduce human well-being (Beckmann-Wübbelt et al., 2023). Crown damage and tree mortality are driven on the one hand by climate change, while this process is partly species and size dependent and also interacts with management strategy and site conditions (Camarero, 2021; Nolan et al., 2021). Big and tall trees have exposed canopy and higher evaporative demand and metabolic requirements, and they were observed to be more vulnerable to crown damage than small trees (Bennett et al., 2015; McDowell and Allen, 2015; Stovall et al., 2019). On the other hand, shorter, understorey trees have a much shallow root system and can be also susceptible to crown damage and mortality rate due to drought as well as competition for other resources due to high stem density (Brun et al., 2020; Nolan et al., 2021). Intra- and inter-species interactions in forest community can also be a factor for crown health (Chakraborty et al., 2017). Plant interactions are synergistic or competitive for resource uptake, and which type of interaction plays a key role is somehow depend on tree species and site conditions, e.g. species diversity, functional traits, and structural diversity (Bertness and Callaway, 1994). These interactions can be facilitative, competitive, or have no effect on crown dieback. Thus, the interrelationship between tree size, stem density, tree interactions, and crown die-back needs to be better understood in peri-urban forests.

The differential responses of trees to extreme climate events and the predisposing factors that make some trees more vulnerable than others remain poorly understood in peri-urban forests near large cities. Previous studies of mortality events in natural forests in rural regions either relied on remote sensing observations or had limited sample sizes or species diversity (Marchin et al., 2022b; Neycken et al., 2022; Stovall et al., 2019). Tree species respond differently during extreme drought; many trees showed severe crown die-back or eventually died, while other co-occurring trees revealed only minor damage (Camarero, 2021; Neycken et al., 2022). Large-scale plot-based crown die-back observations for multiple tree species in peri-urban forests are even rarer. The ground-based on-site survey of tree crown damage is necessary for understanding the true respond of tree health in urban environments, and facilitate the adaptation of the management to increase forest resilience and species selection in peri-urban environments.

Drought hazard has long been a critical environmental issue in southwest Germany (Erfurt et al., 2019). The most recent significant drought occurred in 2018, and the summer of 2018 is ranked as a top five drought event in the research area in Karlsruhe and Rheinstetten region (Erfurt et al., 2019; Tijdeman and Menzel, 2021). The study region has experienced urban sprawling over the last decades and was considered to have high sensitivity to climate change as well as urbanization (Rannow et al., 2010; Siedentop and Fina, 2010). We conducted inventory in peri-urban forests in the 2019 and 2020 post-drought period in Karlsruhe and Rheinstetten region of southwest Germany.

Crown die-back is important for the measurements of crown health conditions and tree vitality as the crown is the most important functional part of the tree (Chakraborty et al., 2017). The start of crown die-back from the upper part of the stem is related to the stressed that trees are facing. Our hypotheses were: 1) crown die-back would vary greatly among tree species and changed along tree size, and 2) plant functional traits (i.e. xylem anatomy, drought tolerance), tree neighbourhood characteristics (i.e. tree density, species diversity, forest structural diversity) and canopy heat exposure (averaged for three years) would also affect crown die-back.

2. Method

2.1. Study area

This study was conducted in the municipal area of Karlsruhe and Rheinstetten in the State of Baden-Württemberg, southwest Germany (Fig. 1). Both municipalities locate on the right bank of the Upper Rhine Trough. The prevailing atmospheric conditions are semi-continental climate with warm, humid air, high irradiation combined low wind speeds, average temperature of 10,5 °C and total precipitation of 717 mm per year (reference period: 1981–2010) (German Weather Service, 2023). In 2003, the Karlsruhe weather station in Germany set a record for the highest air temperature of 40.2 °C, which lasted until 2015. In 2018, both municipalities experienced extreme heatwave and severe drought, and this year was characterized by the longest total time in soil moisture stress in Southwest German, with a median total duration of 91 days (Erfurt et al., 2019; Tijdeman and Menzel, 2021). Soil fertility and stand level mean annual basal increment are comparable in this study area because all plots locate in a similar geology. The soil type in this area is cambisol with alluvium as well as colluvium from Rhine river and its tributaries (Gauer and Aldinger, 2005).

2.2. Surveyed plots and field data collection

Community and municipality-owned forests within the city's administrative boundaries are part of peri-urban forests (PF) in southwest Germany. The PF inventory data were collected in the summers of 2019 and 2020 following the ground-based methodology proposed by the United States Forest Service's Urban Forestry Section at Syracuse, New York (Nowak et al., 2008). Firstly, we divided the study area into a 1360 × 1360 m grid nets. Then, two plots were randomly selected within each grid cell. For each plot, the sample size is 400 m^2. Finally, 68 plots from peri-urban forests areas were sampled (Fig. 1). All trees and shrubs >2.5 cm DBH (diameter at breast height, cm) were identified to species level. A total of 2578 trees that belongs to 51 tree or shrub species were identified. DBH, crown width (N-S, E-W, m), total tree height (m), crown height (m), crown missing (%) and crown die-back (%) of each individual trees were measured. Besides these variables, we also investigated tree health status according to i-Tree pest detection methodology (IPED, 2010), which created for city trees. These tree health variables include: Epicormics shoots (count); Wilted foliage (%); Defoliation (%); Discoloured foliage (%); Abnormal foliage (%); Insect signs (%); BB insect signs (%); Insect presence (%); Disease signs (%); Loose bark (%).

DBH was measured using a measuring tape. Tree height, crown width and crown height, were measured using the TruPulse 360B, a Laser rangefinder. Crown missing is visually estimated as the percentage

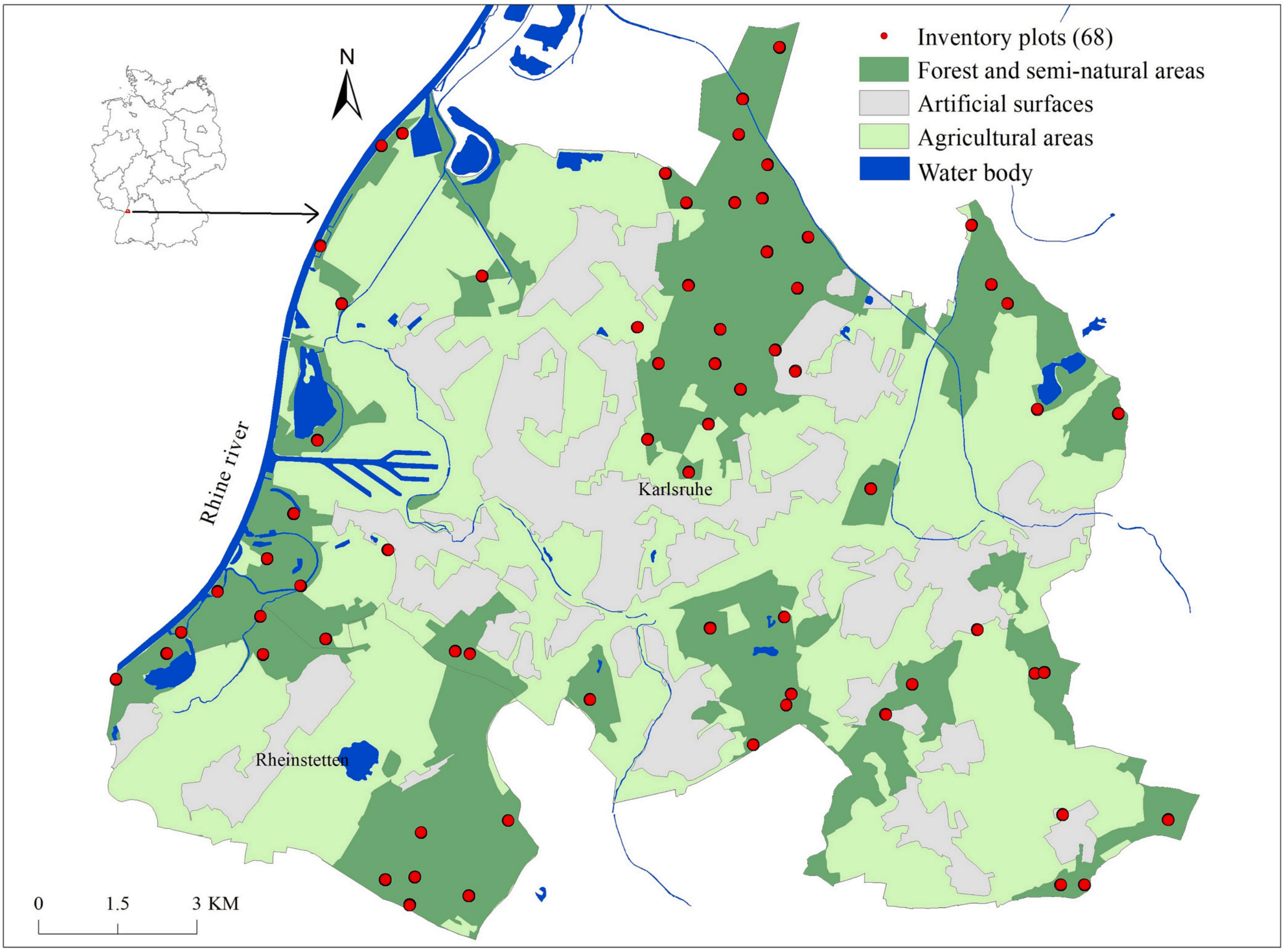

Fig. 1. Location of the municipal area of Karlsruhe and Rheinstetten with the distribution of 68 randomly sampled peri-urban forest plots.

of the crown volume that is not occupied by branches and leaves due to pruning, dieback, defoliation, uneven crown, or dwarf or sparse leaves, and consider the natural crown shape for the particular species. Crown die-back is the volume proportion of dead branches relative to the volume of the total potential crown of the healthy tree. It is the dying back of branches and branch tips generally in the upper and outer portions of the tree crown, and was visually estimated in 5 % classes, ranging from 0 % (no crown die-back) to 100 % (fully dead crown). Percent dieback does not include normal, natural branch dieback, i.e., self-pruning due to crown competition or shading in the lower portion of the crown. However, branch dieback on side(s) and top of crown area due to shading from a building or another tree would be included (i-Tree eco, 2021). Two trained observers provided independent visual estimates of crown missing and die-back, and the final result was averaged from these two measurements. Based on crown width, crown height and crown missing, crown projection area (CPA, m^2) and crown volume (m^3) were calculated. CPA was calculated by multiplying the radius of the crown width in the N-S direction by the radius of the crown width in the E-W direction, and then multiplying by π. Crown volume was calculated based on shape-specific volume calculations of (Franceschi et al., 2022) and take into account the crown missing. The characteristics of each plot can be found in Appendix Table 1.

2.3. Variables selection and tobit model

We checked the distribution of crown die-back for all 2578 trees in R (R version 4.2.0, Copyright (C) 2022) (Appendix Fig. 2). The 47 % of trees were assessed as 0 % (963 trees) and 100 % crown die-back (239 trees). The general regression method cannot explain the difference in crown die-back between the limit value and the non-limiting observation value to ensure the consistency of the estimates. Therefore, a tobit regression, which can handle censored data, was used to analyse the influencing factors on crown die-back for all 2578 trees from 51 tree species (Shuai and Fan, 2020). Tobit regression is also known as a censored regression model, and is designed to estimate linear relationships between variables when there is either left- or right-censoring in the dependent variable. The development of the model is as follows:

$$y_i^* = \beta_0 + \beta_i x_i + \varepsilon_i \ \varepsilon_i \sim N\left(0, \sigma^2\right)$$

$$y_i = 0, \text{ if } y_i^* \leq 0$$

$$y_i = y_i^*, \text{ if } 100 > y_i^* > 0$$

$$y_i = 100, \text{ if } y_i^* \geq 100$$

Among them, the interpreted variable y_i is the crown die-back of the i-th tree. x_i is the explanatory variable, β_i is the unknown parameters of the i-th explanatory variable, and ε_i refers to the standard error of this model. y_i is observed variable, while y_i^* is latend variable. These coefficients are calculated using the maximum log-likelihood function, which indicates the consistency of the model. The explanatory variable x_i takes the actual observation value, and the interpreted variable y_i^* is in a restricted manner. When the value $0 > y_i^* > 100$, it takes the actual observation; when $y_i^* \leq 0$, the observation is censored to 0; when $y_i^* \geq 100$, the observation is censored to 100.

We firstly checked the collinearity (Dormann et al., 2013) of possible explanatory variables using spearman correlation in SPSS software (IBM Corp., Released 2021) and vif (variance inflation factor) function from 'car' package (Fox and Weisberg, 2019) in R (R Core Team, 2022). The multicollinearity result can be found in Appendix Table 2 and Appendix Fig. 2. After checking the collinearity, DBH, tree height, CPA and crown volume are highly correlated with spearman correlation coefficient larger than 0.7 and VIF > 2, and we choose tree height as the only tree size variable. Other variables included in tobit model are described below.

The explanatory variables that included in the tobit model are as follows: tree height and tree density (number of trees per plot per ha) which represent tree size and competition; xylem anatomy type (Schoch et al., 2004; *The wood database,* 2022) and plant drought tolerance (Niinemets and Valladares, 2006) which underlies plant hydraulic function and water use strategy (For each tree species, plant drought tolerance ranking and xylem anatomy type information can be found in Appendix Table 3); canopy surface temperature, retrieved from Landsat 8 TRIS images on three typical hot summer days from 2018 to 2020, which represent the high heat differences among tree canopy surface (Appendix Fig. 3); H index (Shannon-Weiner Index), which described species diversity and evenness; and Gini coefficient for tree height,

Table 1
Variables description included in Tobit model.

Variables	Levels or ranges	Definition and/or citations
Tree height	1.2 to 45 m	Height from the ground to the top of the tree (alive or dead).
Tree density	123 to 3877 trees per plot per ha	The number of trees per plot per hectare.
Xylem anatomy types	Diffuse porous Diffuse to semi-ring porous Ring porous Non-porous	Xylem is one of the transport tissues in vascular plants, and its basic function is to transport water from roots to stems and leaves. There are four types of xylem anatomy types according to the size and distribution of vessels and tracheid. Plant xylem anatomy types were retrieved from (Schoch et al., 2004; *The wood database,* 2022).
Drought tolerance	1, very intolerant; 2, intolerant; 3, moderately tolerant; 4, tolerant; 5, very tolerant	Drought tolerance ranking, based on site characteristics of species dispersal (total annual precipitation, ratio of precipitation to potential vapor transpiration (P:PET ratio) and duration of the dry period) and physiological potentials of species (minimum soil water potential that can be tolerated over the long term with <50 % of foliage damage or dieback), referenced from (Niinemets and Valladares, 2006)
Shannon-Weiner Index	0 to H_{max} (which is 2.04 in this research)	Shannon's diversity Index which considers both species richness and evenness, also known as H index, $H = -\sum_{i=1}^{S} p_i \times lnp_i$, where pi = proportion of individuals of species i, and ln is the natural logarithm, and S = species richness.
Gini coefficient	0 (perfect equality, e.g. same height in a plot) to 1 (maximum structural diversity)	Gini coefficient for tree height is a measure of statistical dispersion, the forest structural diversity, e.g. tree height diversity, within a plot (Weiner, 1985). We calculate Gini coefficient using Gini package (Du et al., 2021) in R.
Canopy surface temperature	28 to 40 °C	Averaged surface temperature at tree canopy (Land surface temperature at tree canopy) retrieved from Landsat 8 TIRS images (path/row is 195/26) on August 6th 2018, July 7th 2019, and August 8th 2020, three hot summer days at 12:16 pm local time, provided by German Federal Meteorological Office "Deutscher Wetterdienst". The air temperature of these three images when the satellite passed over were 30 °C, 32.8 °C, and 32.2 °C, respectively.

which measured forest structural diversity. More details can be found in Table 1. The tobit model was conducted using 'AER' (Kleiber and Zeileis, 2008) and 'censReg' (Henningsen, 2022) packages in R (R Core Team, 2022).

2.4. Species and size effect on crown die-back

To be more specific about which tree species, at what height extent are more resilient to crown damage, we ranked and plotted the crown die-back of 35 most common tree species (species with a sample size >8 trees were considered common species in this research) using 'ggplot2' package in R, and also checked and plotted the linear correlations between tree height and crown die-back for each 35 common tree species using 'lm' function in R (R Core Team, 2022).

3. Results

3.1. Characteristics of tree species and their health status

A total of 2578 trees of 51 species were investigated in this research (Table 2). The most common tree species in the research area was *Fagus sylvatica*, accounting for 17.8 % of all sampled trees, followed by *Prunus serotina* (accounting for 13.9 %), *Acer pseudoplatanus* (12.8 %) and *Carpinus betulus* (8.3 %). The median crown die-back for these species were 5 %, 5 %, 0 % and 2.5 %, respectively, ranging from 0 % to 100 %. The median DBH for all trees was 7 cm, ranging from 2.5 to 123 cm, and the median tree height was 8 m, ranging from 1.2 to 45 m. The median for all health variables were 0 count or percent, except for crown die-back. There were no outbreaks of insects or tree disease in the study area during the investigation period.

3.2. Factors affect crown die-back

Tree height posed negative effects on crown die-back for all 2578 trees from 51 tree species (Table 3). For every 1 m taller a tree grows, the crown die-back decreased by 0.81 % (95 % CI −1.12, −0.51) ($p < 0.001$). Plant drought tolerance and Gini coefficient also had negative effects on crown die-back, and for every 1-point increase in the tolerance ranking or Gini coefficient, the crown die-back decreased by 14.35 % (95 % CI −18.44, −10.25) and 41.78 % (95 % CI −78.03, −5.52), respectively ($p < 0.001$ and $p < 0.05$). As refers to xylem anatomy type, we choose ring porous trees as reference group. Compare to ring porous trees, crown die-back of trees with diffuse porous and diffuse to semi ring porous decreased 29.13 % (95 % CI −36.73, −21.53) and 23.35 % (95 % CI −30.83, −15.8) respectively ($p < 0.001$), whereas crown die-back of non-porous trees increased 20.97 % (95 % CI 12.09, 29.86) ($p < 0.001$). Tree density, Shannon-Weiner Index and canopy surface temperature both increased crown die-back. With tree density increased one trees per plot per ha, the crown die-back increased 0.01 % (95 % CI 0.00, 0.01) ($p < 0.001$). With Shannon-Weiner Index increased 1 point, the crown die-back increased 14.21 % (95 % CI 8.93, 19.49) ($p < 0.001$). We also found a 1 °C increase in canopy surface temperature increased crown die-back by 1.75 % (95 % CI 0.57, 2.92) ($p < 0.01$).

3.3. Crown die-back variations across common tree species and tree height

We ranked the average crown die-back of the 35 most common tree species (with sample size over 8 trees) in Fig. 2. The mean and median crown die-back of all trees are 20 % and 5 %, respectively. Among the 35 common tree species, 17 species had median crown die-back lower than 5 %. Crown die-back varied greatly from 100 % in *Magnolia* × *soulangeana* to 0 % in *Acer negundo*. The top 10 crown die-back species were *Magnolia* × *soulangeana* 100 % (100–100 %), *Euonymus europaeus* 90 % (67.5–92.5 %), *Picea abies* 65 % (27.5–100 %), *Abies alba* 45 % (25–100 %), *Fraxinus excelsior* 45 % (20–100 %), *Populus alba* 26.3 % (1.3–100 %), 33.8 % (13.8–76.9 %) of *Sambucus nigra*, 5 % (0–100 %) of *Betula pendula*, 10 % (0–42.5 %) of *Crataegus monogyna*, and 5 % (0–58.75 %) of *Pseudotsuga menziesii* (Fig. 2).

Tree height as an explanatory variable was either positive, negative or had no influence on crown die-back, and the relationships between tree height and crown damage varied across species (Fig. 3 and Appendix Fig. 4). For all 35 common tree species with a sample size of >8 trees, only 12 tree species showed significant relationship between height and die-back (Fig. 3, $p < 0.05$). Among the 12 tree species, 9 species (*Acer pseudoplatanus*, *Fraxinus excelsior*, *Picea abies*, *Pinus sylvestris*, *Populus alba*, *Prunus avium*, *Prunus padus*, *Prunus serotine*, *Pseudotsuga menziesii*) showed a decrease in crown die-back as tree height increased. Only 3 species (*Populus* × *canadensis*, *Quercus rubra*, *Sorbus aucuparia*) showed increasing crown dieback with increasing tree height. The relationship between tree height and crown die-back of all 35 common tree species was displayed in Appendix Fig. 4.

4. Discussion

4.1. Crown die-back are species-specific and size dependent

Globally, forest die-off events under climate change are accelerating, and many studies demonstrated that climate change poses great mortality risk on forests, especially for large trees (Bennett et al., 2015; McDowell and Allen, 2015; Stovall et al., 2019). Large trees with big canopy exposed to higher solar radiation and leaf-to-air vapor pressure deficits could be more vulnerable to drought due to higher water requirements for evaporation and transpiration (Bennett et al., 2015; Camarero, 2021). However, there are also many reports of higher mortality in shorter understory trees (Brun et al., 2020; Nolan et al., 2021). Over the post-heat and drought period, short and understory trees in peri-urban forests were more susceptible to crown damage than tall trees with large crowns in our study (Table 3 and Fig. 3). A possible explanation could be the different drought exposure of large and small trees. Large trees can store more water and have the highest hydraulic capacitance, and deep root systems allow them to uptake groundwater more easily and efficiently, and relieve water stress during drought years (Camarero, 2021; Choat et al., 2018). Another possible explanation could be that this size-dependent crown damage disagreement highlights the system-dependent response of trees, not only the tree physiology and site factors but also the ecological interactions with biotic agents that obscure understanding of drought exposure and vulnerability (Camarero, 2021; Trugman et al., 2021). In general, biological agents such as bark beetles tend to preferentially attack larger trees. However, in our research, the biotic stresses (e.g. insects' infection and disease) was very low and wasn't a threat to large trees (Table 2). The crown die-back patterns of urban trees during record-breaking droughts and heatwaves in Australia also showed a negative correlation between diameter at breast height and crown die-back, which is consistent with our research results, indicating a higher crown damage in smaller trees (Marchin et al., 2022b). The establishment period after tree planting was generally viewed as the life stage with the highest mortality for urban trees (Hilbert et al., 2019). Newly planted city trees in Germany are usually irrigated by the city horticulture departments in the first few years of establishment, but this management strategy does not apply to trees in peri-urban forests. Considering the high crown damage of small trees in peri-urban forests, more nursing management need to be taken to help trees better adapt to the accelerating heatwave and drought events.

Tree size-dependent crown die-back pattern was shared among most species, but with some exceptions (Fig. 3, *Populus* × *canadensis*, *Quercus rubra*, *Sorbus aucuparia*), indicating that this process was also species-specific. During the post combined heatwave and drought period, crown die-back varied greatly among tree species (Fig. 2), and trees with ring porous or non-porous usually become more vulnerable to crown damage than trees with diffuse and diffuse to semi-ring porous species

Table 2

All 2578 trees of 51 tree species in the suburban area of Karlsruhe, Germany are listed according to their sample size. The DBH, tree height, crown width, crown projection area (CPA), crown volume and ten other tree health variables with units, which followed by i-PED (IPED, 2010), and the number of trees that surveyed (sample size) are provided. All variables are described as median (min, max), except for sample size.

Species	DBH cm	Tree height m	CPA m^2	Crown volume m^3	Crown die-back %	Epicormics count	Wilted foliage %	Defoliation %	Discoloured foliage %	Abnormal foliage %	Insect signs %	BB insect signs %	Insect presence %	Disease signs %	Loose bark %	Sample size
Fagus sylvatica	6 (2.5, 78)	7.5 (2, 32)	15 (0, 300)	34 (0, 3330)	5 (0, 100)	0 (0, 500)	0 (0, 100)	0 (0, 90)	0 (0, 90)	0 (0, 100)	0 (0, 80)	0 (0, 10)	0 (0, 10)	0 (0, 10)	0 (0, 20)	460
Prunus serotine	6.2 (2.5, 53)	7 (1.4, 35)	13 (0, 113)	19 (0, 538)	5 (0, 100)	0 (0, 100)	0 (0, 50)	0 (0, 98)	0 (0, 60)	0 (0, 20)	0 (0, 100)	0 (0, 20)	0 (0, 10)	0 (0, 10)	0 (0, 30)	357
Acer pseudoplatanus	5 (2.5, 57)	7.5 (1.8, 28.5)	10 (0, 145)	18 (0, 1308)	0 (0, 100)	0 (0, 40)	0 (0, 100)	0 (0, 100)	0 (0, 90)	0 (0, 90)	0 (0, 100)	0 (0, 10)	0 (0, 10)	0 (0, 20)	0 (0, 10)	331
Carpinus betulus	7 (2.5, 46.8)	8 (2, 26)	20 (0, 228)	52 (0, 1927)	2.5 (0, 100)	0 (0, 120)	0 (0, 30)	0 (0, 60)	0 (0, 50)	0 (0, 30)	0 (0, 60)	0 (0, 10)	0 (0, 0)	0 (0, 0)	0 (0, 0)	213
Pinus sylvestris	33.8 (7.5, 68)	23 (5, 35)	28 (0, 201)	102 (0, 926)	20 (0, 100)	0 (0, 20)	0 (0, 60)	0 (0, 100)	0 (0, 100)	0 (0, 30)	0 (0, 20)	0 (0, 60)	0 (0, 0)	0 (0, 0)	0 (0, 70)	106
Fraxinus excelsior	11 (3, 75)	12.5 (2.5, 31.6)	10 (0, 201)	22 (0, 1448)	45 (0, 100)	0 (0, 20)	0 (0, 15)	0 (0, 80)	0 (0, 40)	0 (0, 0)	0 (0, 20)	0 (0, 100)	0 (0, 50)	0 (0, 50)	0 (0, 100)	79
Prunus avium	9.5 (2.5, 64)	8.5 (2.5, 25)	13 (0, 79)	20 (0, 562)	17.5 (0, 100)	0 (0, 60)	0 (0, 20)	10 (0, 100)	0 (0, 25)	0 (0, 10)	0 (0, 40)	0 (0, 20)	0 (0, 20)	0 (0, 80)	0 (0, 70)	79
Picea abies	11 (2.5, 56)	9 (2, 32)	4 (0, 123)	2 (0, 1516)	65 (0, 100)	0 (0, 0)	0 (0, 70)	0 (0, 85)	0 (0, 80)	0 (0, 10)	0 (0, 0)	0 (0, 50)	0 (0, 10)	0 (0, 0)	0 (0, 20)	73
Prunus padus	5 (2.5, 23.5)	8 (1.8, 14)	13 (2, 223)	36 (2, 323)	10 (0, 70)	0 (0, 15)	0 (0, 15)	20 (10, 60)	10 (0, 40)	0 (0, 0)	0 (0, 35)	0 (0, 0)	0 (0, 0)	0 (0, 30)	0 (0, 0)	69
Pseudotsuga menziesii	8.5 (3, 79)	6 (2.5, 38)	7 (0, 165)	24 (0, 1835)	5 (0, 100)	0 (0, 30)	0 (0, 10)	10 (0, 95)	0 (0, 20)	0 (0, 0)	0 (0, 0)	0 (0, 10)	0 (0, 0)	0 (0, 10)	0 (0, 15)	67
Tilia cordata	17.3 (3, 51)	14 (3.5, 29.5)	31 (3, 245)	130 (6, 2767)	6.3 (0, 70)	0 (0, 26)	0 (0, 15)	10 (0, 80)	0 (0, 85)	0 (0, 50)	0 (0, 100)	0 (0, 60)	0 (0, 0)	0 (0, 10)	0 (0, 40)	62
Cornus sanguinea	4 (2.5, 42)	4.5 (2, 17)	10 (0, 50)	11 (0, 179)	0 (0, 100)	0 (0, 30)	0 (0, 10)	0 (0, 35)	0 (0, 30)	0 (0, 10)	0 (0, 60)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	58
Acer campestre	4 (2.5, 17)	6 (3, 12)	9 (0, 28)	18 (0, 120)	2.5 (0, 100)	1.5 (0, 60)	0 (0, 10)	0 (0, 30)	0 (0, 60)	0 (0, 0)	0 (0, 50)	0 (0, 60)	0 (0, 0)	0 (0, 0)	0 (0, 10)	50
Quercus robur	24 (2.5, 92)	21 (3, 34)	38 (0, 201)	200 (0, 1458)	12.5 (0, 100)	1.5 (0, 50)	0 (0, 80)	0 (0, 70)	0 (0, 70)	0 (0, 50)	0 (0, 20)	0 (0, 0)	0 (0, 20)	0 (0, 0)	0 (0, 10)	50
Corylus avellane	6.2 (3, 68.2)	6 (1.5, 13)	17 (1, 113)	36 (1, 424)	5 (0, 72.5)	0 (0, 28)	0 (0, 10)	10 (0, 70)	0 (0, 90)	0 (0, 40)	0 (0, 40)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 20)	44
Quercus rubra	13.8 (2.5, 64.6)	15 (2.5, 24.5)	13 (3, 178)	69 (5, 1587)	3.8 (0, 35)	0 (0, 11)	0 (0, 10)	0 (0, 30)	0 (0, 10)	0 (0, 10)	10 (0, 70)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 10)	42
Sambucus nigra	5 (2.5, 10)	4.5 (2.5, 7)	7 (0, 29)	9 (0, 56)	33.8 (0, 100)	0 (0, 0)	0 (0, 90)	0 (0, 80)	0 (0, 100)	0 (0, 40)	0 (0, 100)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 30)	42
Populus alba	26 (7, 97)	22 (4.5, 30)	33 (3, 178)	146 (5, 1657)	0 (0, 12.5)	0 (0, 0)	0 (0, 0)	0 (0, 50)	0 (0, 50)	0 (0, 0)	0 (0, 50)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	38
Alnus glutinosa	22.5 (4.5, 63)	11.5 (3, 45)	19 (0, 96)	52 (0, 866)	10 (0, 100)	0 (0, 12)	0 (0, 0)	15 (0, 80)	0 (0, 10)	0 (0, 0)	0 (0, 20)	0 (0, 80)	0 (0, 30)	0 (0, 30)	0 (0, 40)	36
Ulmus minor	5.7 (2.6, 14.2)	5.5 (1.4, 14)	11 (1, 39)	18 (0, 120)	5 (0, 52.5)	0 (0, 90)	0 (0, 10)	0 (0, 15)	0 (0, 15)	0 (0, 15)	0 (0, 50)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 10)	34
Crataegus monogyna	6.3 (2.5, 21.3)	6 (1.2, 10)	8 (0, 39)	12 (0, 126)	10 (0, 100)	0 (0, 15)	0 (0, 10)	0 (0, 30)	0 (0, 0)	0 (0, 70)	0 (0, 10)	0 (0, 0)	0 (0, 40)	0 (0, 0)	0 (0, 30)	30
Salix alba	33 (3, 123)	18 (2.5, 30)	28 (0, 114)	124 (0, 742)	10 (0, 100)	0 (0, 40)	0 (0, 10)	10 (0, 50)	10 (0, 50)	0 (0, 50)	0 (0, 10)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 40)	29
Euonymus europaeus	4 (2.5, 10)	5 (2.5, 7.5)	6 (0, 26)	7 (0, 37)	90 (0, 100)	0 (0, 0)	0 (0, 0)	70 (0, 90)	0 (0, 10)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	27
Magnolia × soulangeana	6.5 (2.5, 61)	5.8 (1.5, 18)	0 (0, 13)	0 (0, 26)	100 (5, 100)	0 (0, 42)	0 (0, 40)	0 (0, 90)	0 (0, 100)	0 (0, 0)	0 (0, 10)	0 (0, 70)	0 (0, 20)	0 (0, 10)	0 (0, 0)	26
Abies alba	4 (2.5, 17)	5 (2.2, 15)	3 (0, 20)	3 (0, 145)	45 (0, 100)	0 (0, 5)	0 (0, 10)	0 (0, 80)	0 (0, 15)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	25
Acer platanoides	7.5 (3, 44)	9.5 (4, 25)	14 (4, 143)	39 (5, 1171)	0 (0, 20)	0 (0, 4)	0 (0, 0)	0 (0, 15)	0 (0, 10)	0 (0, 0)	0 (0, 10)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	23
Populus nigra	63 (4, 75)	19 (5.5, 28)	6 (0, 116)	11 (0, 511)	10 (7.5, 50)	0 (0, 0)	0 (0, 20)	0 (0, 15)	0 (0, 0)	0 (0, 0)	0 (0, 80)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	15
Betula pendula	5 (2.5, 34)	8.5 (4, 15)	6 (0, 24)	14 (0, 119)	5 (0, 100)	0 (0, 50)	0 (0, 30)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	14

(*continued on next page*)

Table 2 (*continued*)

Species	DBH cm	Tree height m	CPA m^2	Crown volume m^3	Crown die-back %	Epicormics count	Wilted foliage %	Defoliation %	Discoloured foliage %	Abnormal foliage %	Insect signs %	BB insect signs %	Insect presence %	Disease signs %	Loose bark %	Sample size
Sorbus aucuparia	39.5 (6, 49)	29 (4, 35)	64 (7, 118)	315 (7, 1236)	20 (0, 50)	0 (0, 1)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 10)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	11
Tilia platyphyllos	11 (3, 17.5)	10 (4, 18)	16 (6, 57)	64 (5, 298)	0 (0, 25)	0 (0, 22)	0 (0, 10)	10 (0, 70)	0 (0, 80)	0 (0, 10)	0 (0, 90)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	11
Robinia pseudoacacia	10 (3.5, 41)	11 (5, 30)	31 (15, 51)	82 (14, 317)	10 (0, 25)	0 (0, 0)	0 (0, 10)	0 (0, 40)	0 (0, 50)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	9
Acer negundo	12.3 (3, 18.5)	7.5 (4.5, 10)	13 (2, 24)	27 (2, 118)	0 (0, 0)	8.5 (0, 30)	0 (0, 10)	0 (0, 20)	10 (0, 30)	10 (0, 40)	0 (0, 20)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	8
Populus × *canadensis*	7 (3, 33.4)	8 (2.5, 19)	7 (0, 64)	6 (0, 420)	32.5 (0, 100)	0 (0, 0)	0 (0, 20)	0 (0, 10)	0 (0, 10)	0 (0, 0)	0 (0, 20)	0 (0, 0)	0 (0, 10)	0 (0, 10)	0 (0, 0)	8
Prunus spinosa	3.3 (2.5, 12)	5.5 (4.5, 11)	10 (3, 64)	16 (5, 361)	2.5 (0, 90)	0 (0, 8)	0 (0, 0)	0 (0, 30)	0 (0, 0)	0 (0, 0)	0 (0, 80)	0 (0, 0)	0 (0, 10)	0 (0, 0)	0 (0, 0)	8
Taxus baccata	10.4 (3, 31.8)	5.8 (2.5, 10)	15 (2, 79)	35 (2, 327)	15 (0, 92.5)	0 (0, 0)	0 (0, 50)	0 (0, 90)	0 (0, 30)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	8
Ulmus glabra	6.8 (3, 18.2)	6 (3.5, 7)	13 (5, 14)	12 (5, 38)	5 (0, 45)	0 (0, 17)	0 (0, 0)	10 (0, 20)	0 (0, 20)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	4
Larix decidua	40 (30.5, 53)	28 (24, 30)	39 (39, 44)	92 (79, 138)	10 (5, 10)	0 (0, 200)	0 (0, 10)	0 (0, 10)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	3
Populus tremula	30.5 (13.5, 100)	18 (17, 39.5)	28 (13, 436)	177 (75, 5513)	0 (0, 15)	0 (0, 10)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	3
Salix caprea	11.5 (9, 63.5)	6 (4.5, 14)	24 (18, 155)	44 (26, 1031)	57.5 (42.5, 85)	0 (0, 8)	0 (0, 0)	60 (45, 90)	0 (0, 0)	0 (0, 0)	90 (70, 100)	0 (0, 65)	0 (0, 0)	0 (0, 0)	0 (0, 0)	3
Ulmus laevis	19.1 (8.5, 27)	8 (7, 15)	20 (16, 44)	46 (19, 280)	0 (0, 25)	10 (0, 65)	0 (0, 0)	0 (0, 30)	20 (0, 20)	0 (0, 0)	0 (0, 30)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	3
Viburnum opulus	5 (5, 7.8)	3.5 (3, 5.5)	11 (0, 20)	3 (0, 41)	82.5 (30, 100)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	3
Betula pubescens	38.5 (36, 41)	29 (26, 32)	73 (51, 96)	470 (307, 632)	10 (0, 20)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	2
Castanea sativa	3.8 (3.5, 4)	5 (5, 5)	7 (6, 7)	9 (9, 9)	2.5 (0, 5)	0 (0, 0)	0 (0, 0)	7.5 (0, 15)	15 (0, 30)	0 (0, 0)	7.5 (0, 15)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	2
Cornus mas	6.3 (3, 9.6)	5.8 (5, 6.5)	17 (15, 20)	27 (23, 31)	17.5 (10, 25)	0 (0, 0)	0 (0, 0)	10 (0, 20)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	2
Crataegus laevigata	3.7 (3, 4.4)	3.8 (3, 4.5)	14 (13, 16)	20 (13, 27)	0 (0, 0)	4 (2, 6)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	2
Ilex aquifolium	3.5 (3, 4)	4.5 (4, 5)	13 (13, 13)	31 (27, 35)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	5 (0, 10)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	2
Ligustrum vulgare	3.8 (3.5, 4)	3.5 (3, 4)	16 (13, 20)	23 (20, 25)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	2
Quercus petraea	17 (7.5, 26.5)	11.5 (11, 12)	24 (20, 28)	67 (64, 70)	1.3 (0, 2.5)	0 (0, 0)	0 (0, 0)	0 (0, 0)	15 (10, 20)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	2
Crataegus nitida	6 (6, 6)	7.5 (7.5, 7.5)	16 (16, 16)	48 (48, 48)	0 (0, 0)	30 (30, 30)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	1
Prunus cerasifera	12.7 (12.7, 12.7)	3 (3, 3)	7 (7, 7)	12 (12, 12)	90 (90, 90)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	1
Rhamnus cathartica	4 (4, 4)	7.5 (7.5, 7.5)	13 (13, 13)	23 (23, 23)	5 (5, 5)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	0 (0, 0)	1
In total	7 (2.5, 123)	8 (1.2, 45)	12.6 (0, 435.5)	26.5 (0, 5512.75)	5 (0, 100)	0 (0, 500)	0 (0, 100)	0 (0, 100)	0 (0, 100)	0 (0, 100)	0 (0, 100)	0 (0, 100)	0 (0, 50)	0 (0, 80)	0 (0, 100)	2578

Table 3
Summary of Tobit regression result explaining the variations of tree crown die-back in peri-urban forest during post-drought period. This is the regression result of the overall 2578 trees from all 51-tree species. β (95 % CI) is the beta coefficient with 95 % confidence intervals that explain the influence degree of explanatory variables on target variable of die-back; Marg. Eff. is the marginal effects which is the probability that the dependent variable is uncensored.

Variables	β (95 % CI)	Marg. Eff.
(Intercept)	3.77 (−40.38, 47.93)	
Tree height	−0.81*** (−1.12, −0.51)	−0.44***
Drought tolerance	−14.35*** (−18.44, −10.25)	−7.72***
Tree density	0.01*** (0, 0.01)	0***
Diffuse porous xylem	−29.13*** (−36.73, −21.53)	−15.68***
Diffuse to semi ring porous	−23.35*** (−30.83, −15.87)	−12.57***
Non-porous xylem	20.97*** (12.09, 29.86)	11.29***
Canopy surface temperature	1.75** (0.57, 2.92)	0.94**
Tree species diversity (Shannon-Weiner Index)	14.21*** (8.93, 19.49)	7.65***
Structural diversity (Gini coefficient)	−41.78* (−78.03, −5.52)	−22.49*

*** $p < 0.001$.
** $p < 0.01$.
* $p < 0.05$.

(Table 3). There are rarely comparative studies of tree health or mortality in response to drought among diffuse porous, ring porous, and conifers (Kannenberg et al., 2019). However, wood formation is highly sensitive to climate, and wood porosity can be an important predictor of species sensitivity to projected intensification drought (D'Orangeville et al., 2022; Fuchs et al., 2021). Ring porous and non-porous species were found to activate cambium and wood cell lignification earlier than diffuse-porous trees (Michelot et al., 2012). While, the later growth window in diffuse porous species, coinciding with peak evapotranspiration and lower water availability, exposes diffuse species to a higher water deficit in a case study in North America (D'Orangeville et al., 2022). We are not sure whether this coinciding of growth peak and drought exposure in diffuse-porous wood implies high vulnerability or drought adaptation, and whether this coinciding contributes to crown damage remains unknown and warrants further field ecophysiological research in peri-urban forests. The growth of diffuse porous species Maple and Beech also unimodally peaked in June in Germany, while ring porous Oak growth was more sensitive to air humidity and had multiple irregular growth peaks during the year (Tumajer et al., 2022). Crown die-back is an inevitable consequence of hydraulic failure, and trees with most crown die-back exhibit high rates of natural embolism (Nolan et al., 2021). We attribute this variation in wood xylem-dependent crown damage to the differentiation of growth patterns, drought sensitivity, and, most importantly, embolism resistance. The cavitation will decline hydraulic conductance in the soil–root pathway, exacerbating plant hydraulic deficits and eventually forming crown die-back if severe drought occurs (Adams et al., 2017; Walthert et al., 2021). Ring porous woods are evolved based on diffuse porous wood, and have a higher maximum water transport capacity than diffuse-porous species but are less resistant to embolism (Taneda and Sperry, 2008). Different from low embolism resistance of ring porous and low water transport capacity of non-porous species, diffuse porous species may able to maintain high water transportation and low embolism in a good balance during and after severe drought and heatwaves. More evidence in future research is needed to fill this knowledge gap.

Crown damage and tree mortality under drought has always been a heated topic, and trees with low drought tolerance were proved to be more susceptible to crown damage and mortality risk by many studies (Choat et al., 2018; Marchin et al., 2022b; Walthert et al., 2021), as well as this research. We found that for every 1-point scale increase in drought tolerance ranking of a tree, crown die-back decreased by 14.35 % (Table 3). However, this tolerance ranking was estimated for species level, not measured for single tree in situ, and we didn't measure site water availability, either. Thus, this tolerance ranking of species was not always consistent with crown die-back. For example, common spindle (*Euonymus europaeus*) had a drought tolerance ranking of 3.04, indicating moderately tolerant, but had a median crown die-back of 90 %, meanwhile Norway maple (*Acer platanoides*) had a drought tolerance ranking of 2.73, and had a median crown die-back of only 10 % (Table 2 and Appendix Table 1). Plant drought traits could influence branch die-back patterns. *Populus bolleana* using a drought avoidance strategy (intolerant to drought) showed a higher possibility of damage for branches in the upper canopy, causing die-back in the upper branches first, while drought-tolerant *Haloxylon ammodendron* showed relative consistent symptoms of drought stress with increasing plant stature (Xu et al., 2023). Apart from species-specific plant functional traits, tree neighbourhood or site characteristics, including inter-species interactions were also important factors that affect crown health or tree mortality (Camarero, 2021; Trugman et al., 2021). Given that we sampled 68 plots in the municipal area of Karlsruhe and Rheinstetten, the characteristics of tree neighbourhood were rather important for crown die-back variations within and between species.

4.2. Tree neighbourhood characteristics and canopy surface temperature also affect die-back

Peri-urban forests bordering the municipal area of Karlsruhe and Rheinstetten had a high tree density, 936 trees per ha on average (Appendix Table 2), and this could increase competition for light, water, soil fertility, and other resources within neighbourhoods. Competition is an important phenomenon in the growth process of trees and the dominant factor in the establishment of forest structure, and was proved to be an important predisposing factor of crown die-back in natural forest (Bakys et al., 2013; Chakraborty et al., 2017; Ohno et al., 2010), but less research was done for urban and peri-urban forests. In this research, we sampled 2578 trees of 51 species from 68 plots in peri-urban forests, and found a negative effect of Gini coefficient of tree height ($p < 0.05$) and a positive effect of tree density ($p < 0.001$) on crown die-back (Table 3). The Gini coefficient was originally proposed to measure income inequality and was later introduced to measure forest structural diversity (Weiner, 1985; Zeller and Pretzsch, 2019). Structural diversity fostered various ecosystem functions because of niche complementarity effect. A higher Gini coefficient of tree height within a plot indicated a more diverse forest structure and competition for water, nutrient and light could be common due to high tree density, and the most obvious benefits of a higher Gini coefficient could be better above-ground space use and higher light availability and efficiency for trees. With increasing light availability, crown die-back was decreasing (Chakraborty et al., 2017). Additionally, tree density posed a significantly positive effect on crown die-back, even though its coefficient was rather small (Table 3, $p < 0.001$). High density of trees can be a long-term stressor or predisposing factor for crown die-back, especially when other stressors are present, e.g. drought stress (Chakraborty et al., 2017), and proper thinning can reduce crown die-back probability (Ohno et al., 2010). In the process of peri-urban forest management, the management of forest structure and tree density is as important as the selection of suitable tree species. Medium density, uneven tree ages or varying tree size may be a suitable strategy for healthy trees in peri-urban forests.

According to Bertness and Callaway's stress gradient hypothesis, interactions between tree species is more synergistic than competitive under increasing stress (e.g. drought stress) for resource uptake (Bertness and Callaway, 1994). However, we found a positive association between Shannon's diversity index and crown die-back (Table 3, $p < 0.001$), implying that inter-specific competition, rather than synergy effect might be the key neighbourhood interactions between species in peri-urban forests. One possible explanation could be the less niche complementarity between tree species, for example, high species diversity does not necessary led to a diverse in trees with different types of

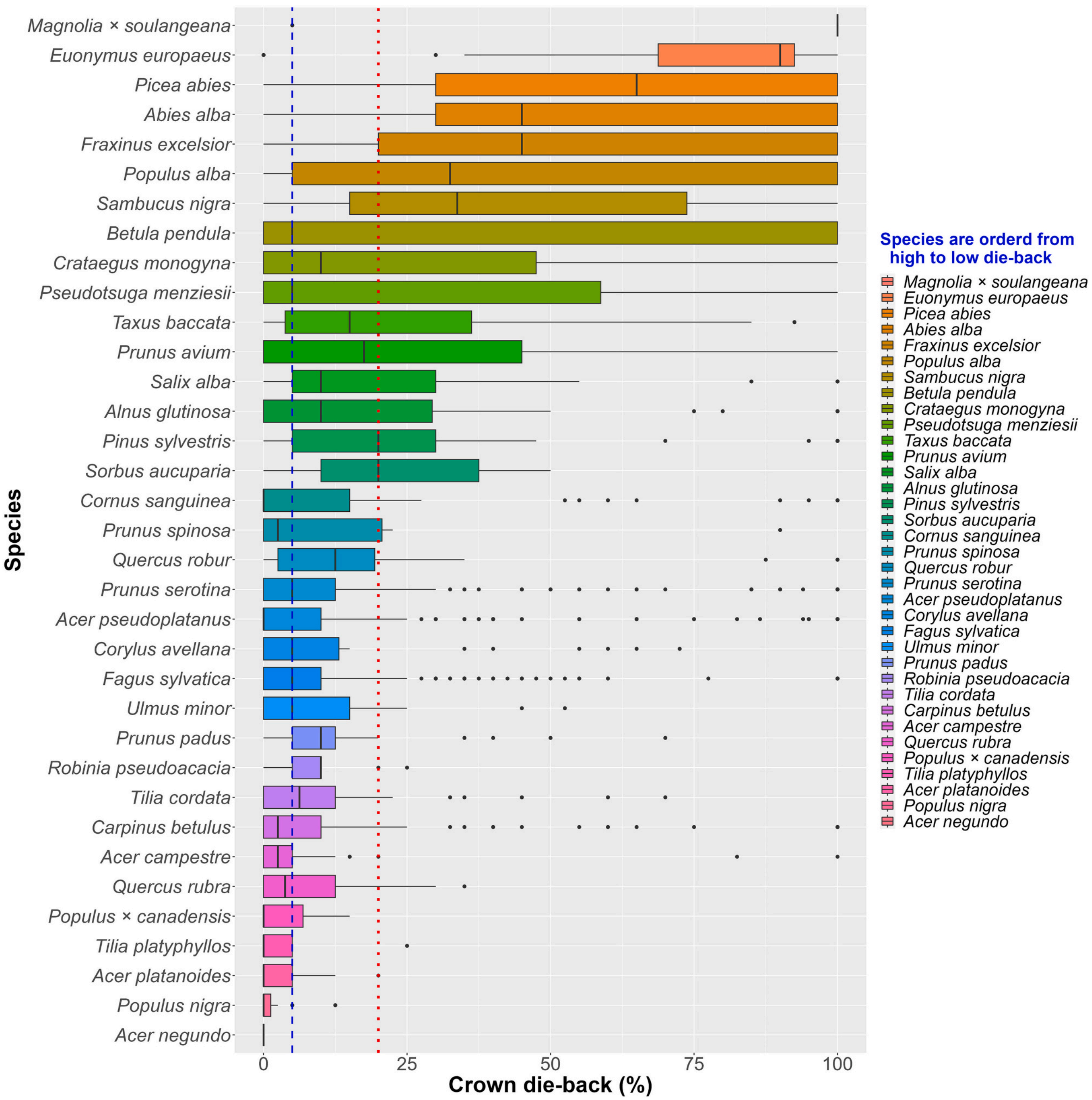

Fig. 2. The boxplots of crown die-back (%) for 35 common tree or shrub species (with >8 trees for each species and 2542 trees in total) sorted by averaged crown die-back (%) from largest to smallest. The dotted red line is the average crown die-back (20 %) and the dotted blue line is the median crown die-back (5 %) of all trees. Crown die-back indicates the morphology estimates of crown health, visually estimated as the dying back of branches and branch tips, generally located in the upper and outer portions of the tree crown.

roots (e.g., trees with diverse rooting depth), and this can cause a poor hydraulic redistribution (roots take up water from moist soil and release it into drier soil), a low water use efficiency, and a high competition for water uptake on sites (Pretzsch et al., 2017). Another explanation can be attribute to that soil nutrient conditions in our study area are relatively good and comparable between plots, this fertile soil do not necessarily exist nutrient stress gradients, thus positive diversity effect on plant stress may not be visible (Gauer and Aldinger, 2005). Considering the effect of high density and potential competition on crown die-back, inter-specific competition may exacerbate crown damage. There were limited direct evidence concerning tree species diversity effect on crown die-back (Marchin et al., 2022b; Walthert et al., 2021), inter-specific interactions effect on crown damage can be complex and indirect, and its intrinsic response mechanisms require more and in-depth research.

Our observations highlighted the fact that plant neighbourhood interactions (tree density, Gini coefficient, Shannon's diversity index) played a key role to the probability of crown damage. Furthermore, the same tree species in different context may respond differently to the same stress. For example, European beech (*Fagus sylvatica* L.) in natural forest has strongly suffered from the exceptional 2018 drought and subsequent dry years that hit Central Europe (Walthert et al., 2021), but was less impact by the same drought in peri-urban forest of our study region (Fig. 2). In our observation, combined heatwave and droughts put some trees at risk; for trees experiencing high heat at canopy, high crown die-back was exhibited. We found a 1 °C increase in canopy surface temperature increased crown die-back by 1.75 % (Table 3, $p < 0.01$). On the one hand, higher temperatures generally lead to more evaporation and transpiration, thus drying soil and trees more quickly than at lower

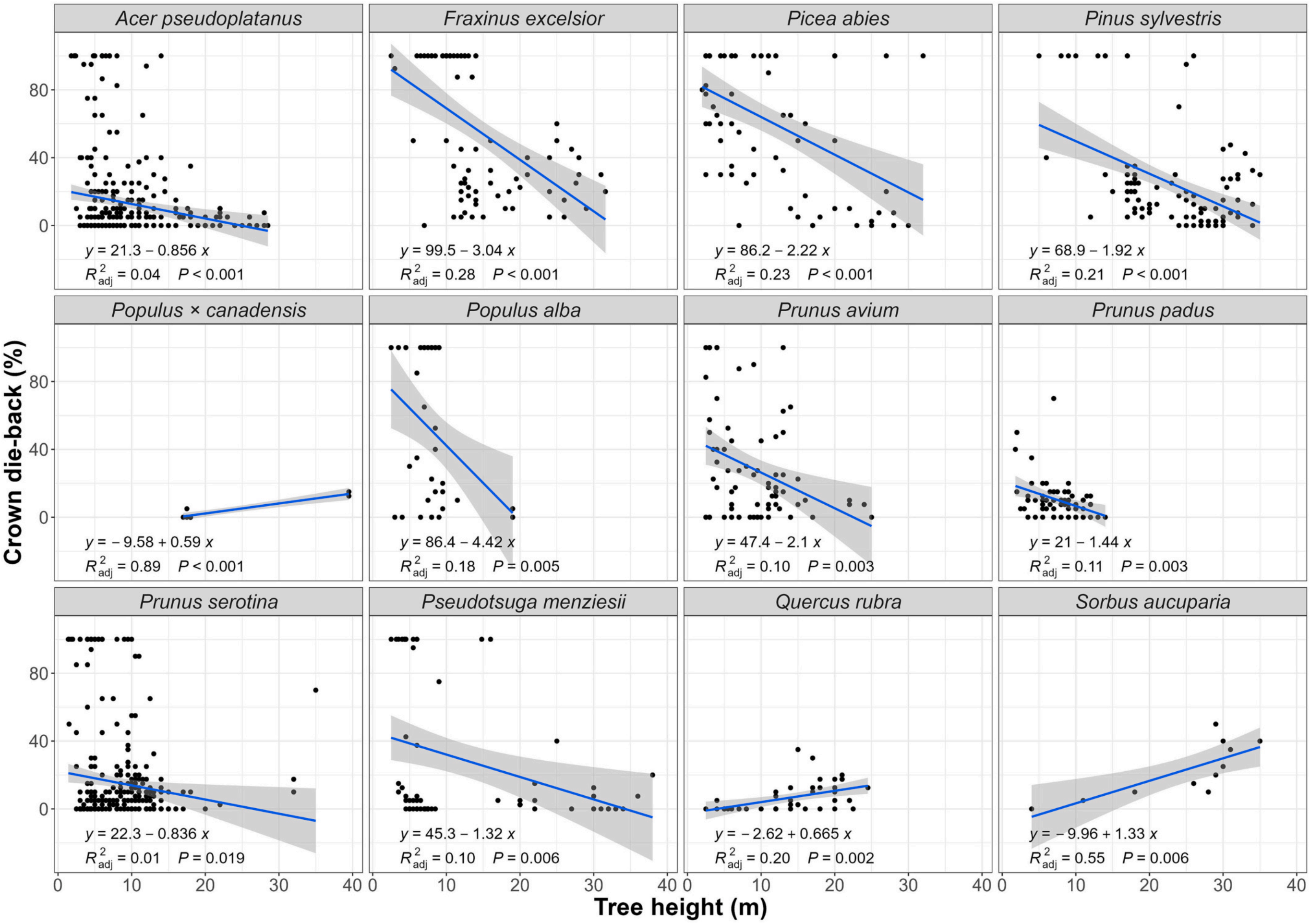

Fig. 3. Tree height as an independent variable that explained the variation of crown die-back, and the relationships between tree height and crown die-back was varied across tree species (species with sample size >8 trees and showed significant relationship ($p < 0.05$) between height and die-back were selected, and the relationships between tree height and crown die-back of all 35 common tree species was displayed in Appendix Fig. 4). The black dots are each individual sampling trees, and the grey band indicates the 95 % confidence interval of the standard error of the model estimates.

temperatures (Choat et al., 2018). On the other hand, drought-stressed trees tend to have lower thermal safety margins (TSM) and dramatically open stomata during heat waves to prevent overheating, but this will exceed leaf turgor loss point and increase mortality risk (Marchin et al., 2022a). Given that human thermal comfort was threatened by the urban heat islands effect under climate change, more trees are needed to improve the living environment (Gangwisch et al., 2023). Notably, we should pay more attention to crown die-back of trees grow on hotter sites such as forest near cities to maximize their cooling effects.

4.3. Limitations and outlook

Due to the limited time, COVID pandemic, we only did one-time sampling during the research period, and crown die-back was surveyed in 2019 and 2020, within two years after the severe combined heatwave and drought events in 2018. Additionally, we didn't measure drought stress level or the actual drought exposure in situ, and for heat exposure, we used the averaged three typical hot summer day from 2018 to 2020. Crown dieback was best explained by plant xylem anatomy traits, drought tolerance traits, tree neighbourhood characteristics (such as Gini coefficient and Shannon's diversity index and tree density), tree size and heatwave exposure also intensify crown dieback. However, there are still other variables not considered by us, such as management strategies (this is important for peri-urban forests, e.g. thinning, soil fertility), actual light availability, soil conditions, etc. (e.g. soil water holding capacity) (Giraldo-Charria et al., 2019; Hietala et al., 2018; Speak and Salbitano, 2023). In addition, species traits used in this study was retrieved from a database (Niinemets and Valladares, 2006), not measured in situ, especially for drought tolerance traits, which can vary greatly among individuals and sites and therefore it is very difficult to assume data from previous studies reflect what is happening in each plot at a specific time. More variables should be considered when crown damage are surveyed in the future, such as leaf thickness, stomatal conductance, leaf water potential, wood density, vessel or tracheid diameter, etc.

The regression result of this research showed crown die-back pattern in peri-urban forests under combined heatwave and drought events, however long-term monitoring of tree health, drought and microclimate inside forest are needed. The application of artificial intelligence and machine learning in the field of tree failure prediction should be used to solve the complicated system-dependent response of trees to drought and other stressors (Jahani and Saffariha, 2021).

5. Conclusions

Natural forest mortality under climate change is accelerating, but few researches focuses specifically on peri-urban forests bordering to a large city. We, for the first time, observed crown damage of 2578 trees of 51 species in peri-urban forests after the combined heatwave and drought events that hit central Europe and our study area in 2018. Our study identified that plant xylem and drought tolerance traits played a key role for the vulnerability of crown damage. Trees with diffuse or diffuse to semi-ring porous had low crown dieback, possibly due to the ability to maintain a balance between high water transportation and low embolism under drought and heat stress. Drought-tolerance trees showed low crown dieback, which hints for the potential species selection of peri-urban forest for future climate change. The crown dieback pattern of peri-urban trees was not only species-specific, but also size-dependent. Small trees are generally more susceptible to crown damage than large trees. This size-dependent crown dieback pattern is common to most species, with some exceptions. Small trees in peri-urban forests need more care by the practitioners in dry and hot climates. The greater mortality risk of small trees may modify future forest succession, whereas crown damage of large trees causes disproportionate carbon and ecosystem function losses. Tree neighbourhood characteristics also played a key role on crown health, trees growing in highly diverse structures, low tree density, low Shannon's diversity plot tend to have low crown die-back, probably owing to the low light, resource and interspecific competition within neighbourhood. High canopy surface temperature posed great risk for crown health, trees growing at plots that likely to suffer from heat risk were more vulnerable to crown damage. With the increasing frequency and severity of drought and heatwaves, Trees in peri-urban forests will suffer more. We ranked die-back of all common tree species in the research area, and many of them had median crown die-back lower than 5 %, which may have higher potential to survive from future climate change. This approach could help urban practitioners in species selection.

CRediT authorship contribution statement

Hailiang Lv: Data curation, Formal analysis, Funding acquisition, Methodology, Validation, Visualization, Writing – original draft, Writing – review & editing. **Marcel Gangwisch:** Data curation, Formal analysis. **Somidh Saha:** Conceptualization, Funding acquisition, Investigation, Methodology, Resources, Supervision, Writing – review & editing.

Declaration of competing interest

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

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

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