Received: 26 February 2019 | Revised: 16 May 2019 | Accepted: 14 June 2019
DOI: 10.1002/ppp3.10064

Plants People Planet PPP Open Access

RESEARCH ARTICLE

# Assessing the vulnerability of Australia's urban forests to climate extremes

**Manuel Esperon-Rodriguez**[1] | **Sally A. Power**[1] | **Mark G. Tjoelker**[1] | **Linda J. Beaumont**[2] | **Hugh Burley**[2] | **Dayenari Caballero-Rodriguez**[3] | **Paul D. Rymer**[1]

[1]Hawkesbury Institute for the Environment, Western Sydney University, Penrith, NSW, Australia
[2]Department of Biological Sciences, Macquarie University, North Ryde, NSW, Australia
[3]Smithsonian Tropical Research Institute, Ancón, Panamá

**Correspondence**
Manuel Esperon-Rodriguez, Hawkesbury Institute for the Environment, Locked Bag 1797, Penrith, NSW 2751, Australia.
Email: m.esperon-rodriguez@westernsydney.edu.au

**Funding information**
Hort Innovation, with co-investment from Macquarie University, Western Sydney University and the NSW Office of Environment and Heritage and contributions from the Australian Government

## Societal Impact Statement

Urban forests are recognized for the multiple benefits they provide to city-dwellers. However, climate change will affect tree species survival and persistence in urban ecosystems. Tree failures will cause economic losses and jeopardize the delivery of societal benefits. The impacts of climate change will depend on the species' resilience and adaptive capacity, as well as management actions which may ameliorate some of the negative impacts. Here, we assessed the potential vulnerability of Australia's urban forests to climate extremes. Our results can be used for future urban planning aiming to incorporate species that are well-adapted to the hotter, drier climates expected with climate change.

## Summary

- Urban forests (UFs) are recognized for the multiple benefits they provide to city-dwellers. However, global climate change—particularly predicted increases in the frequency and intensity of heatwaves and drought—will affect tree species' performance and survival in urban ecosystems.
- Here, we assessed species composition and potential vulnerability of UFs in 22 Australian significant urban areas (SUAs) to heat and/or moisture stress. We quantified species' realized climatic niches across their known distribution, and assessed the extent to which baseline climate in the SUAs where a particular species is planted fell within its niche. We used three environmental variables to group species based on their potential climate vulnerability.
- UFs varied in species composition and climate vulnerability across the continent. In general, neither climate similarity nor geographical proximity were good predictors of species composition among UFs. Of 1,342 tree species assessed (68.4% natives), 53% were considered potentially vulnerable to heat and/or moisture stress in at least one city where they are currently planted.
- Our results highlight the climate vulnerability of current plantings across Australian SUAs and can be used to direct future species selection that considers the species' climate of origin and climatic niche. UF planning can incorporate species from SUAs with similar climates and with low vulnerability to contemporary, as well as

---

future climate conditions. Species with high climate vulnerability, in contrast, may require more intensive management to avoid failure under future hotter, drier climate conditions.

**KEYWORDS**
climate change, climate niche, landscape planting, species composition, species distribution, species selection, tree inventory, urban planning

## 1 | INTRODUCTION

Plants sustain life on our planet by regulating major biogeochemical cycles (Bolin & Cook, 1983; Melillo, Field, & Moldan, 2003). Plants also provide multiple benefits to people, in the form of goods (e.g. food, raw material) and services (e.g. aesthetic, physiological) (Balick & Cox, 1996; Lohr, 2007). In urban ecosystems, plants contribute significantly to human welfare via their contributions to carbon sequestration, pollutant assimilation, heat mitigation, increased biodiversity, improvement of human health, cultural value, and social integration (e.g. Alvey, 2006; Ballinas & Barradas, 2016; Germann-Chiari & Seeland, 2004; Nowak, Hoehn, & Crane, 2007; Oliveira, Andrade, & Vaz, 2011; Tzoulas et al., 2007).

Urban forests—a collection of trees that grow within and around human settlements—are unique ecosystems, typically existing in highly fragmented environments, with a distinctive species composition and structure (Williams et al., 2009). Human needs and preferences play a decisive role in determining the composition of urban forests (Gerstenberg & Hofmann, 2016; Sæbø, Benedikz, & Randrup, 2003), which can vary depending on the type of urban setting, its location, and the desired benefits associated with the planting (Song, Tan, Edwards, & Richards, 2017). For instance, cities typically have areas with significantly higher temperatures compared to the surrounding non-urban areas—the urban heat island effect. In such locations, species with broad canopies and high transpiration rates might be selected to aid with heat mitigation (Ballinas & Barradas, 2016). Although, climate might not necessarily dictate which species are planted in an urban forest, the species' climate of origin has been identified as a key factor determining species' survival and performance (Kendal et al., 2018). In addition, many species are able to grow successfully in climatic conditions beyond those of their natural distribution (Booth, 2017; Booth, Nix, Busby, & Hutchinson, 2014).

The unique characteristics of the urban environment may increase the vulnerability of some species to climate change. Given that there are thermal and aridity limits to species distributions (Stuart-Haëntjens et al., 2018; Woodward & Williams, 1987), predicted changes in climate (means and extremes) may threaten both existing and future urban forests. The impervious surfaces throughout urban areas generally have high heat retention properties and may alter the retention, infiltration, and reuse of water compared to natural substrates (Kumar et al., 2016; Norton et al., 2015). Extremes of low soil moisture availability will be exacerbated by changes in rainfall seasonality and associated increases in the duration and intensity of drought (De Sherbinin, Schiller, & Pulsipher, 2007). Additionally, the urban heat island effect will be aggravated by projected increases in temperature (Corburn, 2009), with expected increases in the magnitude and duration of heatwaves impacting plant and animal species as well as human populations (King & Karoly, 2017; Perkins-Kirkpatrick & Gibson, 2017). Thus, assessing the relationships between species occurrence and climatic conditions, such as temperature and precipitation, can provide valuable information on tree species' potential vulnerability to climate change (Busby, 1988).

Extreme climate conditions are likely to affect species performance and, ultimately, alter the composition of urban forests. This issue is important because urban forests typically include species that are geographically and climatically distant from their natural distributions (Kendal et al., 2018). For species that originate from cooler climates, increases in temperature may limit their persistence in warmer cities. However, rising temperatures will make cities in cooler climates more suitable for species that originate from warmer climates (Jenerette et al., 2016; Kendal et al., 2018). Similarly, in areas where trees receive irrigation, plants are unlikely to be constrained by ambient rainfall and will be less vulnerable to reductions in precipitation (Vogt et al., 2017). Despite the role that climatic suitability may play in shaping species choice and performance in urban areas, few comprehensive studies have explored tree species composition of urban forests across wide-ranging climate zones (but see Jenerette et al., 2016; Ramage, Roman, & Dukes, 2013).

Urban forests require species that can tolerate the climatic conditions likely to occur during the life-span of the individuals (McPherson, Berry, & Doorn, 2018). However, species selection requires consideration of many underlying complexities, including land use, land ownership, competing priorities, and governmental policy (e.g. biosecurity protocols). Furthermore, selection must consider climate adaptation, disease resistance, and phenotypic plasticity, along with aesthetic and social factors (Roman et al., 2015; Sæbø et al., 2003). Thus, evaluating relationships between existing urban forest species and climate provides a basis for identifying vulnerabilities. Vulnerability is defined here as the potential for an adverse impact of external environmental stressors (e.g. heat and moisture) on trees in urban environments. Vulnerability is influenced by the exposure and sensitivity of a system to climate change, as well as its capacity to adapt to this change (IPCC, 2014).

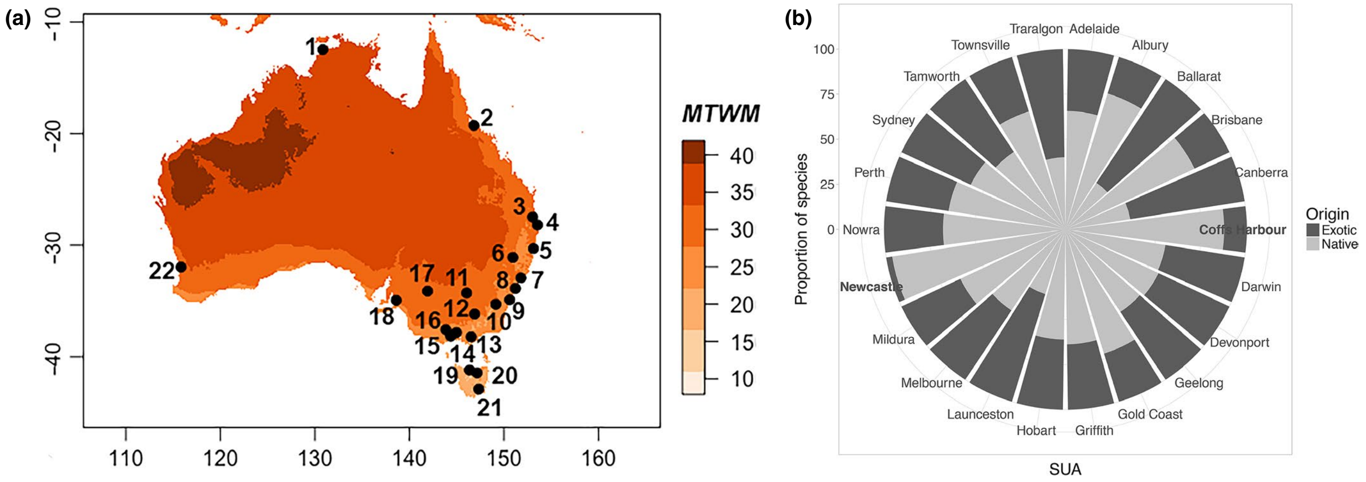

**FIGURE 1** (a) Maximum temperature of warmest month (*MTWM*, °C; interpolations of observed data representative of 1960–1990, data from WorldClim <worldclim.org/>) and location of the 22 significant urban areas (SUAs) assessed in this study: 1–Darwin, NT; 2–Townsville, QLD; 3–Brisbane, QLD; 4–Gold Coast, QLD; 5–Coffs Harbour, NSW; 6–Tamworth, NSW; 7–Newcastle, NSW; 8–Sydney, NSW; 9–Nowra, NSW; 10–Canberra, ACT; 11–Griffith, NSW; 12–Albury, NSW; 13–Traralgon, VIC; 14–Melbourne, VIC; 15–Geelong, VIC; 16–Ballarat, VIC; 17–Mildura, VIC; 18–Adelaide, SA; 19–Devonport, TAS; 20–Launceston, TAS; 21–Hobart, TAS; and 22–Perth, WA. (b) Proportion of native and exotics species in each SUA

The aim of this study was to (a) assess the tree species composition of urban forests within 22 of Australia's Significant Urban Areas (SUAs); (b) assess the potential sensitivity of tree species in urban environments to extreme temperature and precipitation conditions; and (c) undertake a categorization of potential vulnerability. Combined, the 22 SUAs capture a wide range of urban climates, from cool and wet to hot and dry. We collated lists of tree species planted in urban areas across Australia, evaluated the breadth of species' realized climate niches based on native and non-native occurrence data, and compared these to the extremes of climate currently experienced in each of the 22 SUAs where the species are planted. Species' vulnerability was assessed by determining whether it is likely to be affected, adversely or beneficially, when exposed to climatic extremes that are experienced in the SUAs where they are planted. The resulting categorization of potential climate vulnerability can be used for future urban planning and species selection.

## 2 | METHODS

### 2.1 | General approach

To summarize our approach, we collated lists of tree species grown in Significant Urban Areas (SUAs) throughout Australia, this information was used independently and solely to determine vulnerability. Additionally, for each species, we obtained georeferenced occurrence records from natural history database collections. We then extracted environmental data for all occurrence records, including both native and non-native occurrence records, and SUAs included in this study. Species' niches were quantified and compared to the climate of each SUA in which the species is known to be grown based on the tree species lists.

### 2.2 | Tree species lists and species occurrence records

SUAs are geographical units that describe urban agglomerations comprising more than 10,000 people within a single labor market. SUAs are not necessarily a single urban center, as they can also represent a cluster of related centers with a core urban population (ASGS, 2011).

We compiled lists of tree species growing within each SUA based on data from multiple sources, including local council tree inventories and reports, roads/utility authorities reports, the National Register of Big Trees (www.nationalregisterofbigtrees.com.au), and published literature (Kendal et al., 2017). In total, we gathered tree species occurrence data for 50 of Australia's 101 SUAs. However, species lists were limited for 28 of these. Hence, the climate vulnerability section of our study was restricted to 22 SUAs; seven SUAs are from the state of New South Wales (NSW), five from Victoria (VIC), three from Queensland (QLD) and Tasmania (TAS), and one each from the Australian Capital Territory (ACT), the Northern Territory (NT), South Australia (SA) and Western Australia (WA) (Figure 1a).

For each tree species reported, we obtained global occurrence records from the Global Biodiversity Information Facility (GBIF; www.gbif.org) and the Atlas of Living Australia (ALA; www.ala.org.au/) using the *rgbif* and *ALA4R* packages in R (R Core Team, 2018; https://github.com/AtlasOfLivingAustralia/ALA4R), thereby capturing the entirety of the species' realized climate niche by including records from its native and exotic ranges. GBIF and ALA databases are curated repositories of species occurrences, comprising herbarium and museum specimens, survey and atlas data and citizen-science based observations. Occurrence records were filtered and cleaned by removing spatially invalid or suspect records that could lead to miscalculation of species' climate tolerances, records collected prior to 1950, and duplicate records.

Taxonomy and endemicity were standardized and verified against the Global Biodiversity Information Facility (GBIF, www.gbif.org) and then against The Plant List (TPL; www.theplantlist.org) and the Australian Plant Name Index (https://www.anbg.gov.au) using *Taxonstand*, *taxize*, and *taxizehelper* packages (Cayuela, Stein, & Oksanen, 2017; Chamberlain & Szocs, 2013; Quintans, 2018) in R (R Core Team, 2018). We used the accepted binomials from The Plant List taxonomy for our analysis.

## 2.3 | Urban forest composition

Australia's urbanization is largely concentrated in the temperate south-east region and tree inventories are developed only by some local councils. In order to enable valid species composition analysis (Legendre & Legendre, 2012) for assessing whether SUAs with similar Köppen climatic conditions (Stern, Hoedt, & Ernst, 2000) also have similar urban forest compositions, we included a subset of 11 of the 22 SUAs that have relatively larger spatial coverage among the tree inventory data. For these SUAs, we developed a species presence–absence matrix using non-metric multidimensional scaling (NMDS). NMDS projects multivariate data along latent axes based on distances between assemblages but preserves the underlying dissimilarity structure of the original dataset. The distance between SUAs in the ordination space reflects the dissimilarity in species composition, such that SUAs with similar scores are expected to have similar composition (Legendre & Legendre, 2012). Statistical analyses were performed using R version 3.4.4 (R Core Team, 2018), specifically we used the *vegan* package (Oksanen, 2019) and the function `metaMDS`, without auto-transformation and Jaccard coefficient as the metric for the presence–absence community-by-site matrix.

## 2.4 | Environmental data

We assessed the realized climate niche of each tree species occurring in one or more of the 22 SUAs, based on three variables. Climate data (interpolations of observed data, representative of 1960–1990) for each SUA and across each species set of global occurrence records were obtained from WorldClim version 1.4 (http://worldclim.org) at 30 arc-seconds (~1 km) resolution. We selected two variables describing extreme climate: maximum temperature of the warmest month (*MTWM*) and precipitation of the warmest quarter (*PWQ*). We point out that these extreme variables are averages from the WorldClim baseline. Data on actual extreme climates are not readily available. These variables are known to influence ecophysiological functions, and thus species' distributions (O'Donnell & Ignizio, 2012). *MTWM* provides information on how species' distributions relate to warm temperature extremes, whereas *PWQ* is indicative of the importance of water availability to a species' distribution (O'Donnell & Ignizio, 2012). Furthermore, these variables are important for the establishment and performance of species in urban environments (Jenerette et al., 2016).

Additionally, we downloaded data on global potential evapotranspiration (Global-*PET*) from the Consortium for Spatial information (CGIAR-CSI; http://www.cgiar-csi.org) at 30 arc-seconds (~1 km) resolution. These data are representative for the 1970–2000 period, are related to drivers of evapotranspiration and rainfall deficit, and are important indicators of potential vegetative growth (Zomer et al., 2007). Furthermore, *PET* is related to the hydrological cycle, drought, and aridity (Sahin, 2012). All environmental data were transformed to the Australian Albers Equal-Area Conic projection (EPSG:3577).

For all global occurrence records of each species, we extracted values of the three environmental variables. Based on these global records, we then estimated each species' realized climate niche by calculating the mean, median, and 5th/95th percentiles of each variable (Figure S1). For each SUA, we placed a grid (1 × 1 km) over its area and extracted values of the three environmental variables at each cell. Then, we calculated the mean, median, and 5th/95th percentiles of each variable.

For species and SUAs, we used the climate thresholds estimated from the 95th percentile of *MTWM* and the 5th percentile of *PWQ*, and the mean *PET*. Hereafter, when we refer to *MTWM*, *PWQ*, and *PET*, we imply the use of the 95th and 5th percentiles and the mean respectively. We used these thresholds to assess the extremes of these variables as indicative of niche tolerance (i.e. species' thermal and moisture stress tolerance for survival and growth) (Martinez, Arenas, Trilla, Viejo, & Carreño, 2015). For each city, we used the same *MTWM*, *PWQ*, and *PET* thresholds derived from the SUA's climate data. These SUA climate data were plotted and compared with climate thresholds of all the individual species constituting its urban forest. We then identified which species, within a given city, were at the upper or lower limits of their realized climate niche for temperature (*MTWM*), precipitation (*PWQ*), and *PET* respectively (Figure S2).

Finally, for each SUA, we placed each species into one of four categories. *Category 1* includes species considered as potentially vulnerable to both high temperatures and moisture deficit (heat/moisture vulnerable) because their extreme *MTWM* values were below and *PWQ* values above the SUA values. For these species, the SUA was both warmer and drier than the species' high temperature and low precipitation thresholds. Species in *category 2* have values of *PWQ* and *MTWM* above the SUA values; these species were considered only as moisture vulnerable in particular SUAs that were cooler than the high temperature threshold, but drier than the low precipitation threshold. *Category 3* includes species that are heat vulnerable. *MTWM* and *PWQ* species values for *category 3* were below the SUA values, meaning the SUA was warmer than the species' high temperature threshold, but wetter than the low precipitation threshold. Finally, species that are unlikely to be vulnerable to either heat or moisture deficit were placed into *category 4*. These species have values of *MTWM* above and *PWQ* below the SUA values, meaning the SUA is both cooler and wetter than the species' high temperature and low precipitation thresholds. Subsequently, we evaluated cases where the species' *PET* values were higher or lower than the SUA's *PET* values. In those cases where species' *PET* > SUA's *PET*, we highlighted their increased vulnerability (i.e. *PET* vulnerable) (Figure 2).

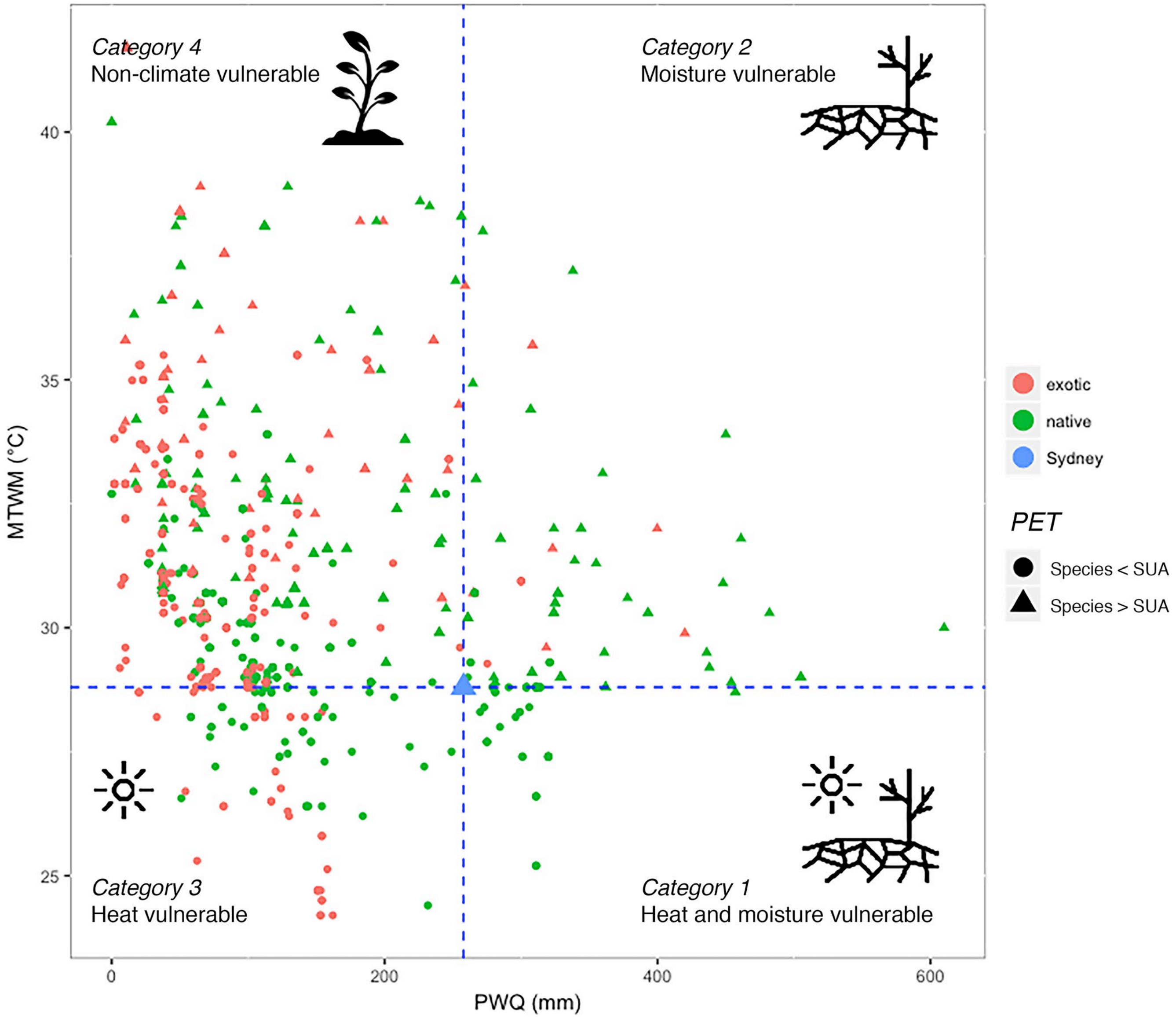

**FIGURE 2** The maximum temperature of the warmest month (*MTWM*) and precipitation of the warmest quarter (*PWQ*) were used to divide tree species (green and blue dots—exotic and native, respectively) into four categories of potential vulnerability to extreme climate. Category 1 includes species whose *MTWM* (95th percentile) values are below and *PWQ* (5th percentile) values are above the significant urban area (SUA) values, respectively (blue triangle—Sydney, NSW, for this example). The category 1 species were considered as potentially vulnerable to heat and moisture stress. Species in category 2 have values of *PWQ* and *MTWM* above the SUA values; these species are considered moisture vulnerable. In contrast, category 3 includes species that are heat vulnerable. *MTWM* and *PWQ* values for these species are below the SUA values. Finally, species with no apparent climate vulnerability fell under category 4. These species have values of *MTWM* above and *PWQ* below the SUA values. Additionally, we included the vulnerability to potential evapotranspiration (*PET*). Vulnerable species are those whose *PET* values are higher that of the SUA's *PET* values (species > SUA)

## 3 | RESULTS

### 3.1 | Urban forest composition

Based on our species lists, we found 1,342 tree species from 138 families reported to be planted and growing in the 22 SUAs. Of these species, 918 (68.4%) were Australian natives and 424 (31.6%) were exotics. Across all SUAs, we found a mean of 161 species (±134 standard deviation), ranging from 36 species in Hobart (TAS) to 423 species in Melbourne (VIC), with a mean of 101 natives (±89) and 60 exotics (±58). Newcastle (NSW) and Melbourne (VIC) had the highest number of native and exotic species reported (295 and 195 species, respectively). In contrast, Ballarat (VIC; 13 species) had the lowest number of natives and Coffs Harbour (NSW; six species) the lowest number of exotic species reported (Figure 1b).

Urban forest composition varied considerably across SUAs. Although our sample size was small (*n* SUAs = 11), we found some indication that species composition is more similar amongst SUAs with similar climates compared to SUAs with contrasting climates. Indeed, the humid subtropical SUAs were separated from the other SUAs,

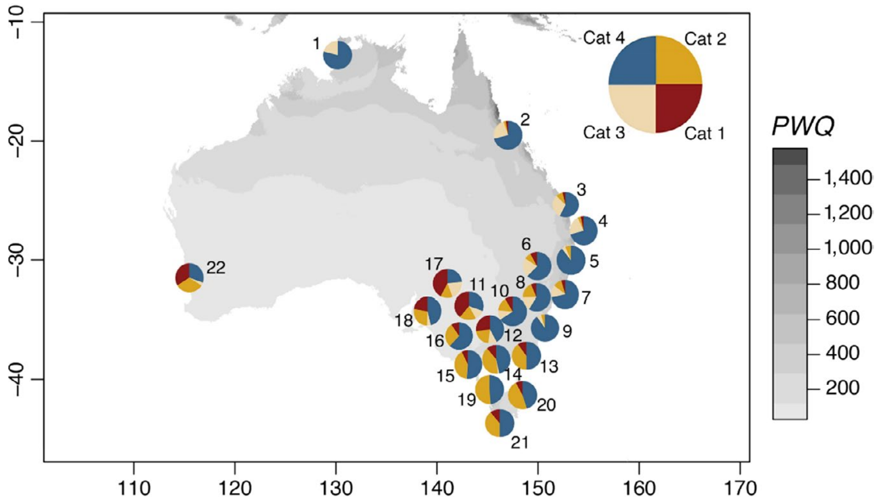

**FIGURE 3** Precipitation of the warmest quarter (*PWQ*, mm; interpolations of observed data (1960–1990) from WorldClim; <worldclim.org/>) and proportion of tree species in each of four potential climate vulnerability categories: heat and moisture vulnerable (Cat 1); moisture vulnerable (Cat 2); heat vulnerable (Cat 3); and non-vulnerable (Cat 4), in 22 significant urban areas (SUAs) within Australia. For more details on climate categories see Figure 2; for names of SUAs see Figure 1a

while the colder semi-arid SUAs of Mildura (VIC) and Griffith (NSW) were clustered together (Figure S3). However, for some SUAs, it appears that neither geographical distance nor climate similarity play a key role in determining urban forest composition. For example, SUAs with markedly different climates, such as Mediterranean Adelaide (SA), cold semi-arid Mildura and coastal Melbourne (VIC) were clustered together. Similarly, although geographically and climatically close, Newcastle and Sydney (NSW) were separated on both NDMS axes. Meanwhile, Melbourne and Perth (WA), which are geographically separated by ~3,500 km and in different Köppen climate zones, were located close to each other on the NMDS plot (Figure S3).

## 3.2 | Categorization of potential climate vulnerability

Higher proportions of heat vulnerable species occur in SUAs further north in progressively warmer climates. Moving south, the proportion of species within Australia that are neither vulnerable to heat nor moisture deficit declines, and the proportion of moisture vulnerable species increases. The highest proportions of species vulnerable to both heat and moisture are found in cold, semi-arid environments (e.g. Griffith and Mildura) and in Mediterranean climates (e.g. Perth and Adelaide) (Figure 3). In general, we found a greater proportion of heat/moisture and moisture vulnerable species in SUAs with lower *PWQ*. Unsurprisingly, higher proportions of heat vulnerable species are found in SUAs with higher *MTWM*. Proportions of non-vulnerable species are higher in SUAs with higher *PWQ* (Figure S4).

Categories of vulnerability are species-specific at each SUA; hence, the same species can fall into multiple categories depending upon the climate of the particular SUA where they are planted. For instance, 38 species (29 natives and nine exotics) were classified in all four categories across various SUAs, where the European species *Quercus robur* and *Betula pendula* and the native *Angophora costata* were categorized as heat/moisture vulnerable (*category 1*) in multiple SUAs (Dataset S1). As such, 706 of the 1,342 species (53%) fall into at least one climate vulnerability category across their various SUAs. In contrast, 992 species have at least one SUA in which the species was placed in *category 4* (i.e. vulnerable to neither heat nor moisture stress). The species' average in each category varied across the 22 SUAS, although the number of species that are neither heat nor moisture vulnerable (*category* 4) was higher across SUAs for both native and exotic species (Table S1; Figure 4a).

A small proportion of species were classified as vulnerable to *PET* (i.e. species' *PET* > SUA's *PET*). Across all SUAs, 1% (±1.4%) of the species defined as heat/moisture vulnerable (*category 1*) were also vulnerable to *PET*, with the highest proportion in Adelaide, SA (3.8%). Of the species vulnerable to moisture stress (*category 2*), 14.9% ± 15.1% were also *PET* vulnerable (highest proportion at Devonport, TAS 45.9%). Only 0.1% ± 0.2% of the species vulnerable to heat stress (*category 3*) was also *PET* vulnerable (highest proportion at Newcastle, NSW 0.98%). In contrast, amongst those species considered to be not vulnerable under current climate (*category 4*), almost a third 32.6% ± 20.1% were vulnerable to *PET* (highest proportion at Nowra, NSW 66.7%) (Figure 4b; Figure S5).

# 4 | DISCUSSION

## 4.1 | Urban forest composition and potential vulnerability to extreme climate

The composition of urban forests shows patterns of climate vulnerability evident across the 11 SUAs traversing all states and biomes in Australia. For instance, similarities in urban forest composition decreased (i.e. fewer common species) between SUAs with contrasting climates (e.g. compare humid subtropical and cold semi-arid, Figure S3) and increasing geographic separation (e.g. compare Newcastle, Perth, Figure S3). Conversely, SUAs within different Köppen climate zones but with similar compositions (e.g. Melbourne, Mildura, and Perth) likely result from incorporation of species from other biomes and biogeographical provinces into their urban forests (Jenerette et al., 2016). Finding that climate and geographic distance does not necessarily have a strong effect on the urban composition of some SUAs indicates that other factors (e.g. aesthetic or historic

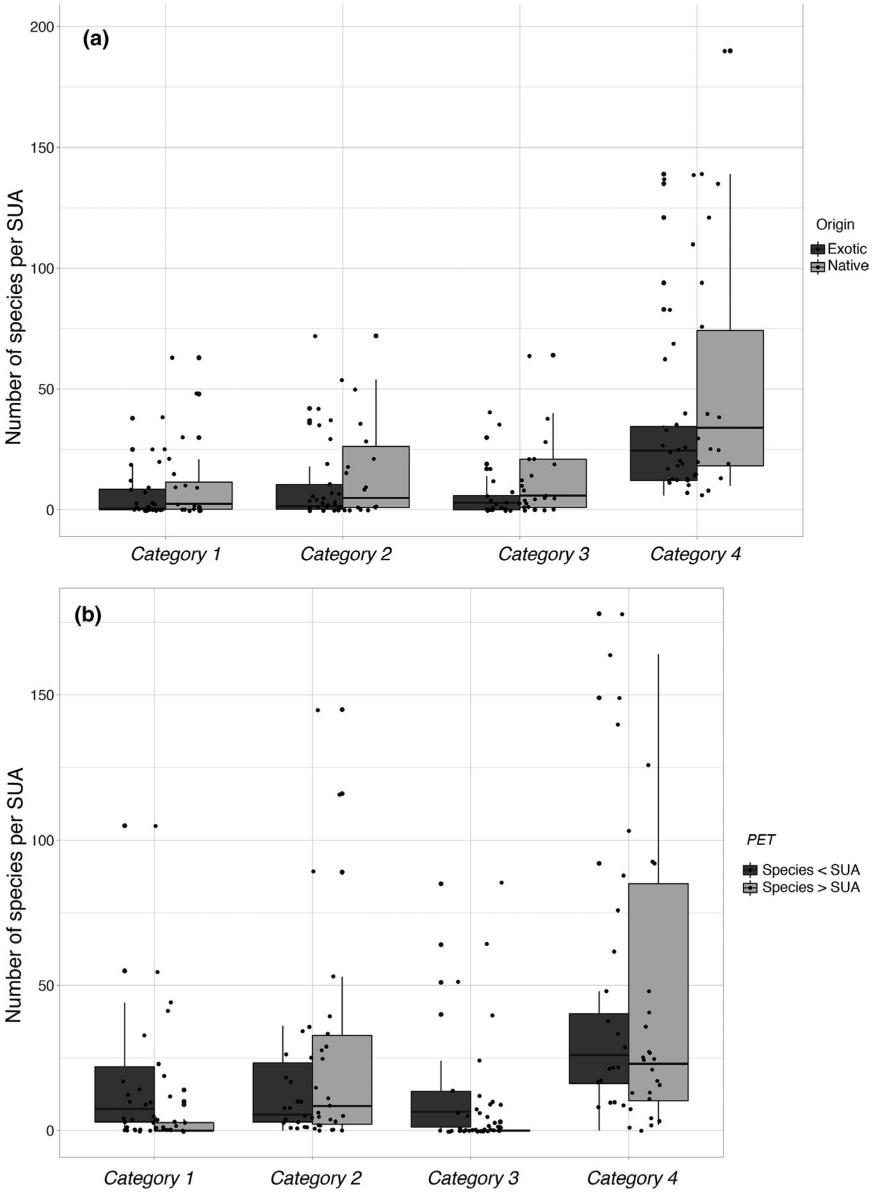

**FIGURE 4** Box-and-whisker plots of categories of potential vulnerability to extreme climate of tree species inventoried across 22 significant urban areas of Australia grouped by (a) their geographic origin (native and exotic) and (b) vulnerability to evapotranspiration (*PET*), where species' *PET* > SUA's *PET* represents potential vulnerability. Categories represent species that are: *category 1*–heat and moisture vulnerable; *category 2*–moisture vulnerable; *category 3*–heat vulnerable; and *category 4*–neither heat nor moisture vulnerable, based on climate thresholds of individual species and the various urban areas in which they are found

preferences) are driving species selection in these Australian SUAs with less consideration given to potential climate vulnerability of planted species.

Future changes in temperature and precipitation patterns will undoubtedly affect the performance and survival of urban forests globally. Rainfall is well-known to play an important role in determining species distributions (Moles et al., 2014; O'Donnell & Ignizio, 2012), and both natural and urban forests are increasingly threatened by drought-induced tree mortality (Anderegg et al., 2015; Klockow, Vogel, Edgar, & Moore, 2018; Vogt et al., 2017). However, the adverse effects of limited rainfall can be reduced in areas with low potential evapotranspiration and also mitigated by irrigation (Jenerette et al., 2016; Vogt et al., 2017). A somewhat surprising result was that relatively few species were found to be vulnerable to *PET*, presumably because vulnerability to heat and moisture already captures vulnerability to *PET*.

Another factor to consider is changes in rainfall seasonality, which will affect soil water balance and plant growth (Feng, Vico, & Porporato, 2012). Rainfall seasonality across Australia varies geographically. Southern Australia experiences predominantly winter rainfall, while northern Australia experiences summer rainfall. Our finding that the two SUAs experiencing a Mediterranean climate also have the highest proportions of species vulnerable to moisture stress (Figure 3) may be related to the seasonality of rainfall in these areas, although we caution that our ability to attribute contrasting vulnerability to rainfall seasonality is limited given only two SUAs contrast.

Extreme temperature is a major factor limiting plant growth and performance (Larcher, Kainmüller, & Wagner, 2010). As heat waves will likely increase in intensity, frequency, and magnitude (McAlpine et al., 2009; Perkins-Kirkpatrick & Gibson, 2017), existing plantings will become more vulnerable to these extremes. We found a higher proportion of heat vulnerable species in warmer SUAs, such as Darwin (NT) and Townsville (QLD), indicating that these trees are potentially more at risk from future increases in temperature. In contrast, cooler and wetter SUAs (such as Devonport [TAS] and Traralgon [VIC]), have either no, or a low proportion of, heat vulnerable species. Trees in these SUAs have the capacity to continue to perform well in the face of modest, future warming, though an increase in heat wave intensity, and duration would negatively affect tree performance.

## 4.2 | Caveats and limitations

The lack of widespread tree inventories and occurrence records constitutes a key limitation to this type of work. Inevitably, our findings are biased toward those SUAs with the resources to develop comprehensive tree inventories. Hence, our results represent an approximation of urban forest composition and perhaps a more limited indication of species' potential climate vulnerability than if data from more SUAs were available. More detailed tree inventories would certainly improve the accuracy of comparisons of urban forest composition across spatial scales in future studies. We used global occurrence records to estimate species' realized niche. However, we note that sampling effort is unequal across regions and might have spatial and temporal bias (Haque, Nipperess, Gallagher, & Beaumont, 2017). Thus, occurrence records used to estimate species' realized niche might not represent the entire species' climatic niche. As such, the breath of species' climatic tolerances can only be approximated and, for most species, their plasticity and adaptive capacity in novel climates is unknown.

Secondly, climate within each SUA can be highly variable. Here, we used 1 km grid data, which does not account for the fine grain of heterogeneity in urban centers. For instance, our analysis cannot explicitly consider locations where temperature is exacerbated by the urban heat island effect or locations where microclimatic conditions are more benign (e.g. protected slopes and gulleys). Furthermore, microclimate can alter species' ecophysiology and thus their ability to cope with climate change (Barradas, Ruiz-Cordova, & Esperón-Rodríguez, 2016). Whenever possible, a more detailed climate assessment considering local microclimate should be used to develop urban planning and management actions.

Thirdly, in regards to the climate data, our baseline (1960–1990) might not accurately represent the current climate conditions. Climate has warmed appreciably over the last three decades (IPCC, 2018), with recent record-breaking summer temperatures testifying to the increasingly extreme conditions being experienced across the continent (Australian Bureau of Statistics, 2018; <www.bom.gov.au>). Additionally, as we pointed out, data from WorldClim do not capture extremes or heatwave events, but rather averages across the 1960–1990 baseline data. Furthermore, our climate vulnerability assessment focused on three environmental variables, *MTWM PWQ*, and *PET*. These variables quantify the extent of potential climate stress during the summer season when heatwave events can be particularly challenging for Australian species (McAlpine et al., 2009; Perkins-Kirkpatrick & Gibson, 2017). Nevertheless, other climate variables can be used to assess species vulnerability, such as the precipitation of the driest quarter or month, and for regions in higher latitudes using the minimum temperature of the coldest month may help identify species at risk from cold temperature anomalies (Jenerette et al., 2016; O'Donnell & Ignizio, 2012). Furthermore, the inclusion of an aridity or moisture index could be used to assess the availability of precipitation for plant growth and can be indicative of changes in rainfall seasonality when comparing different time periods (Stadler, 1998; Thornthwaite, 1948).

Finally, our work does not assess nor consider the adaptive capacity of species to climate change. Species differ in their capacity to cope with climate variability (Esperón-Rodríguez & Barradas, 2015; Jump & Penuelas, 2005), through adaptation (e.g. environmental and genotypic changes) or the use of avoidance and resistance strategies (e.g. adjustments in physiological traits in response to stress) (Lenoir, Gégout, Marquet, Ruffray, & Brisse, 2008; Pellegrini et al., 2017). Our results indicate that ~53% of the species currently planted in SUAs in Australia exceed the margins of their realized climate niches. This result indicates that there may be biological or environmental factors (e.g. local adaptation), or human interventions (e.g. irrigation) facilitating survival and prevalence of these species, although being planted in an area does not necessarily mean that a species is performing well in that location. Also, many native and exotic species are known to be able to tolerate climate conditions beyond those characteristic of their native ranges (Beaumont et al., 2009; Booth, 2017; Simberloff, 2000). Hence, it is possible that this approach may overestimate the vulnerability of species planted beyond their native range, particularly for those species whose non-native occurrence records are unknown or incomplete. Ultimately, a more detailed study on species provenance or varieties may help to elucidate the mechanisms by which species are able to inhabit regions at the limits of their natural realized climate niche and the extent to which this approximates the fundamental niche.

## 4.3 | Future urban forests

We highlight the need to develop tree inventories with detailed information on species' presence and performance in urban areas. Such information can serve as the basis for development of urban forest management plans (Crown, Greer, Gift, & Watt, 2018; Östberg, Wiström, & Randrup, 2018; Palacio-Prieto et al., 2000). Our findings provide new opportunities for species selection and management as climate changes. For future plantings, climate-vulnerable species may be replaced with non-vulnerable species. Developing climate assessments for urban species, for example via the use of species distribution models (Booth et al., 2014; Booth, Nix, Hutchinson, & Jovanic, 1988; Guisan & Thuiller, 2005), represent a powerful tool to increase our confidence in climate-resilient species selection. Selection of species that are identified as unlikely to be vulnerable to current climate in SUAs that have similar climates might be a viable option for expanding the current palette of species within a given city, although consideration must be given to minimize potential risks associated with introducing novel species. Also, we caution that relying only on a small pallet of popular, climate-tolerant species for urban greening can reduce diversity and ecosystem functioning (Wei & Huang, 2015). Finally, vulnerable species may require more intensive management (e.g. maintenance, watering) and in extreme cases, replacement, if they cannot cope with the ever-increasing pace of climate change, including increased exposure to pests and disease (Tubby & Webber, 2010).

## 5 | CONCLUSIONS

Australia's urban forests vary in composition, number of tree species, and potential vulnerability to extreme climate. High proportions of heat vulnerable species were found in warmer SUAs, indicating those trees are potentially more at risk from future increases in temperature. The impacts of climate change will be variable and depend on the resilience and adaptive capacity of individual species, as well as management actions (e.g. irrigation) to ameliorate impacts. Our results highlight species that are broadly planted in urban areas across the continent and identify those that may be vulnerable to heat and/or moisture stress, as they are cultivated in areas approaching the limits of environmental tolerances inferred from their natural and planted climate envelope. Future urban planning should aim to incorporate species that are well-adapted to projected conditions, and our results can be used to inform this objective.

### ACKNOWLEDGMENTS

We thank Rachael Gallagher, Kathryn Fuller and Edward Mifsud for their assistance in the data collection. We are also grateful to Leigh Staas, Michelle Leishman, David Ellsworth, and Alessandro Ossola for feedback on this work. Numerous people contributed to the collection of data used in this study, and we acknowledge their efforts: Fiona Ambrosino, Corey Andrews, Sarah Arnold, Hayden Baird, June Bendzulla, Grant Bilton, Bryan Bourke, Duri Bradshaw, Stephen Damon, Ben de Klepper, Dannielle Denning, Andrew Dickson, Gabriela Eiris, Tamryn Elks, Doug Foster, Dawn Grant, Nathan Kay, Melanie Kelly, Stewart Harris, Audrey Heaton, Nigel Hobden, Greg Hollis, Rachel Hughes, Jemaile Irvine, Brett Jeffery, Vanessa John, Tim Johnson, Bridget Jupe, Melanie Kelly, Amy Layton, Hugh Leckie, Tracey Lee, Jodie Loveridge, Jane Lloyd, Jodie Loveridge, Tania MacLeod, John McKinney, Melissa McManus, Rae McPherson, Carol Meikle, Michael Miggiani, John Milkins, David Mort, John Murray, Francene O'Connor, Will Oldfield, David Parham, Simon Roberts, Daniel Robins, Tracey Rorie, Jamie Smith, Russell Smith, Rob Stevenson, Kathryn Stolk, Maureen Summersides, David Sutton, Karen Sweeney, Natasha Szczyglowska, Adele Taylor, Michael Taylor, Pieter Taylor, Simon Trill, Cameron Tuck, John Turnbull, Jessica Volkanovski, Melinda Walker, Leah Warburton, David Wheeler, and Sally Whitelaw. This work was funded by the Hort Frontiers Green Cities Fund, part of the Hort Frontiers strategic partnership initiative developed by Hort Innovation, with co-investment from Macquarie University, Western Sydney University and the NSW Office of Environment and Heritage and contributions from the Australian Government.

### CONFLICT OF INTERESTS

The authors declare that they have no conflict of interest to disclose.

### AUTHOR CONTRIBUTIONS

MER: design of the research; performance of the research; data analysis, collection, and interpretation; writing the manuscript; MGT: performance of the research; data interpretation; writing the manuscript; SP: performance of the research; data interpretation; writing the manuscript; LB: design of the research; performance of the research; data interpretation; writing the manuscript; HB: performance of the research; data collection; DCR: performance of the research; data analysis and interpretation; PR: design of the research; performance of the research; data analysis and interpretation; writing the manuscript.

### ORCID

*Manuel Esperon-Rodriguez* https://orcid.org/0000-0003-3649-2134

### REFERENCES

Alvey, A. A. (2006). Promoting and preserving biodiversity in the urban forest. *Urban Forestry & Urban Greening*, *5*, 195–201. https://doi.org/10.1016/j.ufug.2006.09.003

Anderegg, W. R. L., Flint, A., Huang, C.-Y., Flint, L., Berry, J. A., Davis, F. W., …Field, C. B. (2015). Tree mortality predicted from drought-induced vascular damage. *Nature Geoscience*, *8*, 367–371. https://doi.org/10.1038/ngeo2400

ASGS. (2011). *Australian Statistical Geography Standard. Volume 4 - Significant Urban Areas, Urban Centres and Localities, Section of State (cat no. 1270.0.55.004)*. Canberra, Australia: Australian Bureau of Statistics.

Balick, M. J., & Cox, P. A. (1996). *Plants, people, and culture: The science of ethnobotany* (p. 228). New York, NY: Scientific American Library.

Ballinas, M., & Barradas, V. L. (2016). The urban tree as a tool to mitigate the urban heat island in Mexico City: A simple phenomenological model. *Journal of Environmental Quality*, *45*, 157–166. https://doi.org/10.2134/jeq2015.01.0056

Barradas, V. L., Ruiz-Cordova, J. P., & Esperón-Rodríguez, M. (2016). Microclimatology and ecophysiology of the urban vegetation of a city with tropical climate modified by altitude in Mexico. *Botanical Sciences*, *94*, 775–786. https://doi.org/10.17129/botsci.627

Beaumont, L. J., Gallagher, R. V., Thuiller, W., Downey, P. O., Leishman, M. R., & Hughes, L. (2009). Different climatic envelopes among invasive populations may lead to underestimations of current and future biological invasions. *Diversity and Distributions*, *15*, 409–420. https://doi.org/10.1111/j.1472-4642.2008.00547.x

Bolin, B., & Cook, R. B. (1983). *The major biogeochemical cycles and their interactions. SCOPE Report (Book 21)*. Hoboken, NY, USA: John Wiley & Sons.

Booth, T. H. (2017). Assessing species climatic requirements beyond the realized niche: Some lessons mainly from tree species distribution modelling. *Climatic Change*, *145*(3–4), 259–271. https://doi.org/10.1007/s10584-017-2107-9

Booth, T. H., Nix, H. A., Busby, J. R., & Hutchinson, M. F. (2014). BIOCLIM: The first species distribution modelling package, its early applications and relevance to most current MAXENT studies. *Diversity and Distributions*, *20*(1), 1–9.

Booth, T. H., Nix, H. A., Hutchinson, M. F., & Jovanic, T. (1988). Niche analysis and tree species introduction. *Forest Ecology and Management*, *23*(1), 47–59. https://doi.org/10.1016/0378-1127(88)90013-8

Busby, J. R. (1988). *Potential impacts of climate change on Australia's flora and fauna*. Melbourne, Australia: Commonwealth Scientific and Industrial Research Organisation.

Cayuela, L., Stein, A., & Oksanen, J. (2017). Taxonomic standardization of plant species names. Package 'taxonstand'. CRAN: https://cran.r-project.org/web/packages/Taxonstand/Taxonstand.pdf

Chamberlain, S., & Szocs, E. (2013). taxize - taxonomic search and retrieval in R. F1000 Research, 2:191. Retrieved from http://f1000research.com/articles/2-191/v2

Corburn, J. (2009). Cities, climate change and urban heat island mitigation: Localising global environmental science. *Urban Studies*, *46*, 413–427. https://doi.org/10.1177/0042098008099361

Crown, C. A., Greer, B. Z., Gift, D. M., & Watt, F. S. (2018). Every tree counts: Reflections on NYC's third volunteer street tree inventory. *Arboriculture & Urban Forestry*, *44*, 49–58.

De Sherbinin, A., Schiller, A., & Pulsipher, A. (2007). The vulnerability of global cities to climate hazards. *Environment and Urbanization*, *19*, 39–64. https://doi.org/10.1177/0956247807076725

Esperón-Rodríguez, M., & Barradas, V. (2015). Ecophysiological vulnerability to climate change: Water stress responses in four tree species from the central mountain region of Veracruz, Mexico. *Regional Environmental Change*, *15*, 93–108. https://doi.org/10.1007/s10113-014-0624-x

Feng, X., Vico, G., & Porporato, A. (2012). On the effects of seasonality on soil water balance and plant growth. *Water Resources Research*, *48*, W05543. https://doi.org/10.1029/2011WR011263

Germann-Chiari, C., & Seeland, K. (2004). Are urban green spaces optimally distributed to act as places for social integration? Results of a geographical information system (GIS) approach for urban forestry research. *Forest Policy and Economics*, *6*, 3–13. https://doi.org/10.1016/S1389-9341(02)00067-9

Gerstenberg, T., & Hofmann, M. (2016). Perception and preference of trees: A psychological contribution to tree species selection in urban areas. *Urban Forestry & Urban Greening*, *15*, 103–111. https://doi.org/10.1016/j.ufug.2015.12.004

Guisan, A., & Thuiller, W. (2005). Predicting species distribution: Offering more than simple habitat models. *Ecology Letters*, *8*, 993–1009. https://doi.org/10.1111/j.1461-0248.2005.00792.x

Haque, M., Nipperess, D. A., Gallagher, R. V., & Beaumont, L. J. (2017). How well documented is Australia's flora? Understanding spatial bias in vouchered plant specimens. *Austral Ecology*, *42*, 690–699. https://doi.org/10.1111/aec.12487

IPCC. (2014). *Climate Change 2014: impacts, adaptation, and vulnerability. Part a: global and sectoral aspects. Contribution of working group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, UK and New York, USA: Cambridge University Press.

IPCC. (2018). *Global warming of 1.5°C*. Intergovernmental Panel on Climate Change. Working Group I Technical Support Unit, Switzerland.

Jenerette, G. D., Clarke, L. W., Avolio, M. L., Pataki, D. E., Gillespie, T. W., Pincetl, S., …Alonzo, M. (2016). Climate tolerances and trait choices shape continental patterns of urban tree biodiversity. *Global Ecology and Biogeography*, *25*, 1367–1376. https://doi.org/10.1111/geb.12499

Jump, A. S., & Penuelas, J. (2005). Running to stand still: Adaptation and the response of plants to rapid climate change. *Ecology Letters*, *8*, 1010–1020. https://doi.org/10.1111/j.1461-0248.2005.00796.x

Kendal, D., Dobbs, C., Gallagher, R., Beaumont, L., Baumann, J., Williams, N., & Livesley, S. (2018). A global comparison of the climatic niches of urban and native tree populations. *Global Ecology and Biogeography*, *27*, 629–637. https://doi.org/10.1111/geb.12728

Kendal, D., Farra, A., Plan, L., Threlfall, C. G., Bush, J., & Baumann, J. (2017).Risks to Australia's urban forest from climate change and urban heat. The Clean Air and Urban Landscapes Hub. National Environmental Science Programme, Australia.

King, A. D., & Karoly, D. J. (2017). Climate extremes in Europe at 1.5 and 2 degrees of global warming. *Environmental Research Letters*, *12*, 114031. https://doi.org/10.1088/1748-9326/aa8e2c

Klockow, P. A., Vogel, J. G., Edgar, C. B., & Moore, G. W. (2018). Lagged mortality among tree species four years after an exceptional drought in east Texas. *Ecosphere*, *9*(10), e02455. https://doi.org/10.1002/ecs2.2455

Kumar, K., Kozak, J., Hundal, L., Cox, A., Zhang, H., & Granato, T. (2016). In-situ infiltration performance of different permeable pavements in an employee used parking lot – A four-year study. *Journal of Environmental Management*, *167*, 8–14.

Larcher, W., Kainmüller, C., & Wagner, J. (2010). Survival types of high mountain plants under extreme temperatures. *Flora-Morphology, Distribution, Functional Ecology of Plants*, *205*, 3–18. https://doi.org/10.1016/j.flora.2008.12.005

Legendre, P., & Legendre, L. F. (2012). *Numerical ecology*. Oxford, UK: Elsevier.

Lenoir, J., Gégout, J.-C., Marquet, P., DeRuffray, P., & Brisse, H. (2008). A significant upward shift in plant species optimum elevation during the 20th century. *Science*, *320*, 1768–1771. https://doi.org/10.1126/science.1156831

Lohr, V. I. (2007). Benefits of nature: What we are learning about why people respond to nature. *Journal of Physiological Anthropology*, *26*, 83–85. https://doi.org/10.2114/jpa2.26.83

Martínez, B., Arenas, F., Trilla, A., Viejo, R. M., & Carreño, F. (2015). Combining physiological threshold knowledge to species distribution models is key to improving forecasts of the future niche for macroalgae. *Global Change Biology*, *21*(4), 1422–1433. https://doi.org/10.1111/gcb.12655

McAlpine, C., Syktus, J., Ryan, J., Deo, R., McKeon, G., McGowan, H., & Phinn, S. (2009). A continent under stress: Interactions, feedbacks and risks associated with impact of modified land cover on Australia's climate. *Global Change Biology*, *15*, 2206–2223. https://doi.org/10.1111/j.1365-2486.2009.01939.x

McPherson, E. G., Berry, A. M., &vanDoorn, N. S. (2018). Performance testing to identify climate-ready trees. *Urban Forestry & Urban Greening*, *29*, 28–39. https://doi.org/10.1016/j.ufug.2017.09.003

Melillo, J. M., Field, C. B., & Moldan, B. (2003). *Interactions of the major biogeochemical cycles: Global change and human impacts*. *Scope Report 61*. Washington, DC: Island Press.

Moles, A. T., Perkins, S. E., Laffan, S. W., Flores-Moreno, H., Awasthy, M., Tindall, M. L., … Bonser, S. P.(2014). Which is a better predictor of plant traits: Temperature or precipitation?*Journal of Vegetation Science*, *25*, 1167–1180.

Norton, B. A., Coutts, A. M., Livesley, S. J., Harris, R. J., Hunter, A. M., & Williams, N. S. (2015). Planning for cooler cities: A framework to prioritise green infrastructure to mitigate high temperatures in urban landscapes. *Landscape and Urban Planning*, *134*, 127–138. https://doi.org/10.1016/j.landurbplan.2014.10.018

Nowak, D. J., Hoehn, R., & Crane, D. E. (2007). Oxygen production by urban trees in the United States. *Arboriculture & Urban Forestry*, *33*, 220–226.

O'Donnell, M. S., & Ignizio, D. A. (2012). *Bioclimatic predictors for supporting ecological applications in the conterminous United States*. Reston, VA: US Geological Survey Data Series 691.

Oksanen, J., Blanchet, F. G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., … Wagner, H. (2019) Vegan. Community Ecology Package. Retrieved from, https://cran.r-project.orghttps://github.com/vegandevs/vegan

Oliveira, S., Andrade, H., & Vaz, T. (2011). The cooling effect of green spaces as a contribution to the mitigation of urban heat: A case study in Lisbon. *Building and Environment*, *46*, 2186–2194. https://doi.org/10.1016/j.buildenv.2011.04.034

Östberg, J., Wiström, B., & Randrup, T. B. (2018). The state and use of municipal tree inventories in Swedish municipalities–results from a national survey. *Urban Ecosystems*, *21*, 467–477. https://doi.org/10.1007/s11252-018-0732-3

Palacio-Prieto, J., Bocco, G., Velasquez, A., Mas, J. F., Takaki-Takaki, F., Victoria, A., … Gonzalez, F. (2000). Technical Note: Current situation of forest resources in Mexico: Results of the 2000 National Forest Inventory. *Investigaciones Geograficas Boletin del Instituto de Geografia*, *43*, 183–203.

Pellegrini, A. F. A., Anderegg, W. R. L., Paine, C. E. T., Hoffmann, W. A., Kartzinel, T., Rabin, S. S., …Pacala, S. W. (2017). Convergence of bark investment according to fire and climate structures ecosystem vulnerability to future change. *Ecology Letters*, *20*, 307–316. https://doi.org/10.1111/ele.12725

Perkins-Kirkpatrick, S., & Gibson, P. (2017). Changes in regional heatwave characteristics as a function of increasing global temperature. *Scientific Reports*, *7*, 12256. https://doi.org/10.1038/s41598-017-12520-2

Quintans, D. (2018) taxizehelper: Convenience functions for R OpenSci's 'taxize' Package. R package version 0.3.0. Retrieved from https://github.com/DesiQuintans/taxizehelper. https://doi.org/10.5281/zenodo.1410632

R Core Team. (2018). *R: A language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing.

Ramage, B. S., Roman, L. A., & Dukes, J. S. (2013). Relationships between urban tree communities and the biomes in which they reside. *Applied Vegetation Science*, *16*, 8–20. https://doi.org/10.1111/j.1654-109X.2012.01205.x

Roman, L. A., Walker, L. A., Martineau, C. M., Muffly, D. J., MacQueen, S. A., & Harris, W. (2015). Stewardship matters: Case studies in establishment success of urban trees. *Urban Forestry & Urban Greening*, *14*, 1174–1182. https://doi.org/10.1016/j.ufug.2015.11.001

Sæbø, A., Benedikz, T., & Randrup, T. B. (2003). Selection of trees for urban forestry in the Nordic countries. *Urban Forestry & Urban Greening*, *2*, 101–114. https://doi.org/10.1078/1618-8667-00027

Sahin, S. (2012). An aridity index defined by precipitation and specific humidity. *Journal of Hydrology*, *444*, 199–208. https://doi.org/10.1016/j.jhydrol.2012.04.019

Simberloff, D. (2000). Global climate change and introduced species in United States forests. *Science of the Total Environment*, *262*, 253–261. https://doi.org/10.1016/S0048-9697(00)00527-1

Song, X. P., Tan, P. Y., Edwards, P., & Richards, D. (2017). The economic benefits and costs of trees in urban forest stewardship: A systematic review. *Urban Forestry & Urban Greening*, 29162–29170.

Stadler, S. J. (1998). Aridity indices. In: *Encyclopedia of Hydrology and Lakes* (pp. 78–83). Dordrecht, Netherlands: Springer. https://doi.org/10.1007/1-4020-4497-6_20

Stern, H., DeHoedt, G., & Ernst, J. (2000). Objective classification of Australian climates. *Australian Meteorological Magazine*, *49*(2), 87–96.

Stuart-Haëntjens, E., De Boeck, H. J., Lemoine, N. P., Mänd, P., Kröel-Dulay, G., Schmidt, I. K., …Smith, M. D. (2018). Mean annual precipitation predicts primary production resistance and resilience to extreme drought. *Science of the Total Environment*, *636*, 360–366. https://doi.org/10.1016/j.scitotenv.2018.04.290

Thornthwaite, C. W. (1948). An approach toward a rational classification of climate. *Geographical Review*, *38*, 55–94. https://doi.org/10.2307/210739

Tubby, K., & Webber, J. (2010). Pests and diseases threatening urban trees under a changing climate. *Forestry: An International Journal of Forest Research*, *83*, 451–459. https://doi.org/10.1093/forestry/cpq027

Tzoulas, K., Korpela, K., Venn, S., Yli-Pelkonen, V., Kaźmierczak, A., Niemela, J., & James, P. (2007). Promoting ecosystem and human health in urban areas using Green Infrastructure: A literature review. *Landscape and Urban Planning*, 81167–81178. https://doi.org/10.1016/j.landurbplan.2007.02.001

Vogt, J., Gillner, S., Hofmann, M., Tharang, A., Dettmann, S., Gerstenberg, T., …Roloff, A. (2017). Citree: A database supporting tree selection for urban areas in temperate climate. *Landscape and Urban Planning*, 15714–15725. https://doi.org/10.1016/j.landurbplan.2016.06.005

Wei, H., & Huang, Z. (2015). From experience-oriented to quantity-based: A method for landscape plant selection and configuration in urban built-up areas. *Journal of Sustainable Forestry*, 34698–34719. https://doi.org/10.1080/10549811.2015.1033555

Williams, N. S. G., Schwartz, M. W., Vesk, P. A., McCarthy, M. A., Hahs, A. K., Clemants, S. E., …McDonnell, M. J. (2009). A conceptual framework for predicting the effects of urban environments on floras. *Journal of Ecology*, 974–979. https://doi.org/10.1111/j.1365-2745.2008.01460.x

Woodward, F. I., & Williams, B. (1987). Climate and plant distribution at global and local scales. *Vegetatio*, *69*(1–3), 189–197. https://doi.org/10.1007/BF00038700

Zomer, R. J., Bossio, D. A., Trabucco, A., Yuanjie, L., Gupta, D. C., & Singh, V. P. (2007). *Trees and water: Smallholder agroforestry on irrigated lands in Northern India*. Colombo, Sri Lanka: IWMI Research Report 122.

## SUPPORTING INFORMATION

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

**How to cite this article:** Esperon-Rodriguez M, Power SA, Tjoelker MG, et al. Assessing the vulnerability of Australia's urban forests to climate extremes. *Plants, People, Planet*. 2019;00:1–11. https://doi.org/10.1002/ppp3.10064