Urban Vegetation: Towards Cooler, Biodiverse Cities of the Future

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Statement of Authentication

The work presented in this thesis is, to the best of my knowledge and belief, original except as acknowledged in the text. I hereby declare that I have not submitted this material, either in full or in part, for a degree at this or any other institution.

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List of Achievements

Publications

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Thesis Abstract

With more than half of the global population now living in cities and towns, urban expansion and climate change are key factors affecting the nature of human interactions with the environment. The replacement of natural ecosystems by built environment changes the microclimate and poses a significant threat to local and regional biodiversity. Urban vegetation is increasingly recognised as playing a key role in mitigating heat and supporting biodiverse cityscapes. Protecting, enhancing and appropriately managing urban green infrastructure are, therefore, key strategies for sustaining the ecosystem services that vegetation provides in cities across the globe.

This thesis presents my doctoral research programme that aims to increase the understanding of the role that vegetation plays in functional urban ecosystems. To do this, I have systematically evaluated: (1) relationships between tree canopy traits and associated sub-canopy cooling; (2) how planting context (e.g., parks or streets) influences the ability of trees to provide cooling benefits; (3) how canopy-associated cooling is influenced by ambient climatic conditions (solar radiation, vapor pressure deficit (VPD) and wind speed); and (4) how habitat/vegetation complexity influences invertebrate biodiversity, in urban settings across Greater Sydney.

Whilst there has been some research exploring relationships between vegetation and landscape traits and urban temperature profiles, to date no study has systematically evaluated how different planting contexts, such as parks, nature strips or pavement/tarmac settings, influence the ability of trees to provide cooling benefits in the local environment. To address this gap, my research identifies those traits that determine the cooling potential of urban trees, alongside the impacts of planting context on the ability of trees to reduce air and surface temperatures. The first data chapter presents the results of a study across Greater Sydney that

found tree shade reduced air and surface temperature by a maximum of 3.7 °C (mean 1.1 °C) and 45 °C (mean 27.4 °C), respectively. Air and surface temperatures were lower under trees in parks and nature strips compared to those planted in pavements or asphalt. I found that the extent of air and surface temperature reductions due to tree shade was greater with increasing leaf area index (LAI), and these relationships were stronger in asphalt and park contexts.

The magnitude and variability of tree-derived cooling benefits differ greatly among studies, likely reflecting differences in tree species' traits, urban characteristics and local climate conditions. The second data chapter presents findings from a systematic study focused on ten commonly occurring species in western Sydney. Here I found that tree species mitigate summertime urban heat more effectively in the morning (07:00 to 13:00), lowering sub-canopy air temperatures by an average of 1.75 - 3.02 °C, compared to full sun exposure, including during periods of high solar radiation and VPD. There were significant species differences in the extent of canopy cooling; those species with higher LAI and wider canopies were associated with the greatest morning temperature reductions. Individual tree cooling benefits (~4 °C, mean maximum temperature reduction) peaked at around 10:00 am. In the afternoon, the extent of canopy-associated cooling was, however, negligible. Indeed, sub-canopy temperatures were marginally warmer than the surrounding air, likely reflecting the effect of afternoon winds coming from the hot, desert-covered interior of the country and canopy retention of heat during the hottest part of the day. Summertime sub-canopy warming peaked at +3.8 °C around 18:00, continuing through much of the night. Overall, I found that the magnitude of the afternoon and night-time warming was strongly dependent on VPD and windspeed and less on tree species and traits.

In addition to the cooling benefits provided by trees, urban vegetation also provides other critical ecosystem services, including habitat and resources for a diverse range of vertebrate and invertebrate animal species. For example, an invertebrate-rich environment contributes to food security, nutrient cycling and pest control. Systematic explorations of the association between habitat/vegetation complexity and invertebrate biodiversity in urban areas are limited. My third data chapter therefore examined whether and how trees and shrubs differ in terms of the diversity, abundance and composition of associated invertebrate communities. It also explored whether increased structural complexity of vegetation at the plot-level (i.e., trees and shrubs growing together) was associated with enhanced diversity and/or abundance of invertebrates, relative to tree-only or shrub-only plantings. Key findings were that shrubs and tree-shrub mixtures were associated with greater abundance and diversity of invertebrate species and functional groups (in particular for detritivores, herbivores, predators, scavengers and pollinators) than trees. Furthermore, the abundance of invertebrates was strongly associated with the presence of flowers, plant volume and LAI for both trees and shrubs.

Overall, my thesis research provides valuable new insights into the extent of cooling benefits provided by a variety of tree species in urban areas throughout Greater Sydney, and how these are influenced by tree canopy traits and local environmental conditions. This information can be used in urban planning and management by informing species and trait selection to optimise urban heat mitigation. My findings also suggest that management of urban green infrastructure should focus on incorporating shrubs alongside trees to maximise their impacts on biological diversity – and the associated pollination, pest control and aesthetic benefits these bring - in urban areas. Taken together, my research advances the current understanding of the key role vegetation can play in mitigating heat and promoting biodiversity in urban areas, and the factors that influence the contribution of vegetation to urban liveability – a key challenge for the future.

CHAPTER ONE: General Introduction

1.1 Introduction

Greenspaces are essential elements in liveable, sustainable cities (United Nations, 2015; WHO., 2016). Urban greenspace has been defined in many ways in different disciplines with the most common definition referring to greenspace as being synonymous with nature and, more explicitly, any vegetated land adjoining an urban area (Taylor & Hochuli, 2017). It includes parks, gardens, rooftop gardens and vertical gardens as well as ditches, canals, rivers and riverbanks, and is therefore also sometimes referred to as the blue -green zone (De Haas et al., 2021). The recognised positive impacts of nature have prompted a growing interest in the various roles urban greenspaces can play in influencing the physical and biological characteristics of the environment (Arnold, 1993), as well as human health and psychological well-being (Sandifer et al., 2015).

Greenspaces in urban ecosystems, which include both natural and engineered areas, provide services to city residents despite the generally high cover of grey infrastructure in most cityscapes (Cameron & Blanusa, 2016; Elmqvist et al., 2015; Gómez-Baggethun et al., 2013). The vegetation and associated fauna in these green spaces provide a number of ecosystem services to urban residents, as recognised by major initiatives such as the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment. 2005). The benefits urban ecosystems can provide include, but are not limited to: (a) provisioning services which comprise the supply of fresh water and food; (b) regulatory services, such as climate regulation, pollination, water purification and erosion control; (c) regulatory services which encompass nutrient cycling, biodiversity support and habitat provision to support species and genetic diversity and (d) cultural services which encompass wellbeing, tourism and spirituality (Elmqvist et al., 2015; Elmqvist et al., 2016; Frey et al., 2018; Gómez-Muñoz et al., 2010; Pataki et al., 2011a). Many of these services are actually provided by vegetation and other associated taxa. The economic value of the ecosystem services inherent in urban greenspaces is also increasingly recognized. For example, building energy usage has been shown to be reduced as a result of urban vegetation lowering air temperatures; estimated savings associated with this are in the range of $1.3 – 2.9 billion annually for 97 cities in the USA, for example (McDonald et al. 2020). In total, existing global urban greenspace provides roughly $33 billion worth of services including biocontrol, pollination, climate regulation and soil formation (Clinton et al., 2018). Unfortunately, urban greenspaces are often designed with a single purpose in mind (e.g., aesthetic enjoyment or temperature reduction), without considering other potential co-benefits such as providing habitat and food resources for animals and invertebrates. Given the world's changing climate, as discussed in the following section, it is important to understand, and optimise, the multi-functionality of urban greenspace, particularly when the areas available for local councils to dedicate or zone as greenspace are impacted by so many factors. These include, but are not limited to traffic, compacted soil, belowground infrastructure and overhead power lines. This thesis focuses on two key ecosystem services provided by vegetation in urban areas - heat mitigation and support for urban biodiversity.

1.2 Climate extremes and urban warming

Global warming is recognised as one of the biggest challenges facing humans and ecosystems alike (Allen, 2018). Air temperatures have risen by 1.1 °C since pre-industrial times (IPCC, 2021a) and extreme heat events e.g., heatwaves, are increasingly common globally (Seneviratne et al., 2014). Heatwaves are characterized by maximum and minimum temperature that are unusually high over a three-day period compared to the local long-term climate and previous weather of that location (BOM, 2014). In the last two decades in particular, heatwaves have increased in frequency across large parts of Europe, Asia, Australia (Cowan et al., 2014; Russo et al., 2015) and north America (Cotlier & Jimenez, 2022). Heatwaves are estimated to have caused 489,075 annual average excess human deaths globally for 2000-2019 (Zhao et al., 2021).

Urban areas can be engines of economic prosperity; they are also often associated with adverse environmental conditions such as water and air pollution, and urban heat (Crutzen, 2004). The conversion of natural surfaces to built-up areas contributes to both economic prosperity and adverse environmental conditions as the surface energy balance changes, altering fluxes into, out of and storage of heat per-unit surface area. The latent heat flux – the exchange of energy from the Earth's surface to the atmosphere due to evapotranspiration - in urban areas is typically greatly reduced, relative to surrounding rural areas, due to lower vegetation cover (Oke et al., 2017). Buildings and impervious surfaces have low albedos — the fraction of solar radiation reflected by a surface (Taha, 1997) and therefore, capture and store higher amounts of incoming solar radiation than natural, pervious surfaces (Agathangelidis et al., 2019; Deilami et al., 2018; van Hove et al., 2015).

Urban modifications to the local climate are manifested in a variety of forms. The capacity of an area to evaporate/transpire water – which is strongly linked to the presence of water bodies and vegetation - to a large degree defines its capacity to cool (Duveiller et al., 2018). Urban areas are drier and warmer than surrounding regions due to lower rates of evaporative cooling, high cover of impermeable surfaces and reduced wind for surface roughness (IPCC, 2021b). The most studied feature of the local climate in urban contexts is the urban heat island (UHI) effect which is defined as the temperature difference between urban and surrounding rural areas (Oke, 1973, 1982). This the temperature difference is driven by the greater storage and re-radiation of energy, alongside the additional waste heat caused by human activities, for example, heat release from cars and air conditioning units in urban areas compared to surrounding peri-urban and rural areas (Memon et al., 2008; Maimaitiyiming et al., 2014; Oke et al., 1991). The relative lack of evapotranspiration cooling is a major contributor to UHI effect (Oke et al., 1991).

Manoli et al. (2019) estimated that, globally, urban areas are up to 2 °C warmer, on average, than suburban areas (Chrysoulakis et al., 2018; Grimmond et al., 2010). The temperature differences between rural and urban areas also depend on season (Chun and Guldmann, 2018), size of the city (Oke, 1973) and local climatic conditions such as humidity (Raj et al., 2020). Similar to other cities, Australian cities are experiencing UHI effects. For instance, during summer days UHI intensity in Sydney – which experiences humid subtropical conditions - could be as large as 10 °C (Santamouris et al., 2020; Sidiqui et al., 2016). The residents of Melbourne, Australia's second largest city, experience 3 to 4 °C of UHI effect (Jamie et al., 2020) while residents in Adelaide, which is located in a Mediterranean climate may experience air temperatures up to 6 °C higher than surrounding rural areas during winter (Soltani & Sharifi, 2017). For most cities, the UHI effects show diurnal variation which is more pronounced at night (Manoli et al., 2019). This thesis investigates heat mitigation in Western Sydney to assess if urban vegetation, in particular the extent to which trees in urban streets, impact diurnal temperature variation.

Around seven billion people will be living in urban areas by 2050 (Ritchie et al., 2018; UN., 2018). To accommodate this projected population, a large proportion of natural area will be converted to housing and other associated urban infrastructure. Many of these are typically dark in colour (low albedo), which will exacerbate urban warming and UHI effects. Studies support a linear (negative) relationship between the extent of vegetation cover and UHI effects on human health and wellbeing. For instance, in Australia, Coates et al. (2022) found that extreme heat accounted for at least 354 human deaths from 2000 to 2018; from these results they suggested a positive correlation between heatwave-related death, dwelling types e.g. number of storeys and urban design. Nicholls et al. (2008), in an earlier study, reported a 15-17% increase in mortality of elderly people on days with a mean daily temperature higher than 30 °C. Furthermore, Australia’s nationwide mortality rate increased by 2% during heatwaves in 2020, with urban areas that have lower vegetation cover associated with more deaths (Varghese et al., 2020). Given the relationships between urban heat and human health/mortality, it is important to quantify and understand the role that urban greenspace, including street trees, can play in reducing temperatures, especially in the context of rising global temperatures and associated heat exposure.

1.3 Urban vegetation and urban heat mitigation

Vegetation, in contrast to grey infrastructures such as concrete and asphalt, typically transforms a proportion of the incoming solar energy, via the process of photosynthesis. Vegetation also has a lower thermal mass, absorbs and stores less energy and re-releases less energy compared to paved and concrete surfaces (Bowler et al., 2010; Santamouris et al., 2017). By intercepting and attenuating direct solar radiation, trees also contribute to improving human thermal comfort outdoors (Coutts et al., 2016). Furthermore, transpiration can increase relative humidity, for example from values as low as 0.5% to 6.4% around a single tree (Gillner et al., 2015) and up to 30% for clusters of tree canopies (Souch & Souch, 1993). Motazedian et al. (2020) observed that parks with tree cover and regular irrigation had air temperatures up to 5 °C lower than surrounding streets. However, there are challenges to expanding green areas in many cities; these challenges relate to the high commercial value and conflicting demands for land, costs associated with maintaining greenspaces (Ignatieva & Hedblom, 2018; Pataki et al., 2021), and the poor soil quality of the available spaces for planting trees, shrubs and herbaceous vegetation.

The growing conditions for urban trees, and in particular access to essential resources i.e., water (Bartesaghi-Koc et al., 2020) and nutrients and soil conditions influence trees' capacity to regulate the microclimate. Urban trees, especially those in street settings, face high heat and radiation loads from surrounding paved and built-up surfaces (Oke, 1989). Urban soil is often drastically disturbed and modified by human activities, with impermeable surfaces such as concrete and asphalt replacing soil in many areas. Natural soil profiles are rarely present and most urban soil is highly disturbed and frequently dominated by subsoil (Jim, 1998; Yeakley, 2020). Urban soils, therefore, have high bulk density and being heavily compacted, lower water holding capacity (Mullaney et al., 2015a). Impervious surfaces also limit the amount of stormwater infiltrating the soil (Norton et al., 2015; Zhang et al., 2015), with low soil moisture levels having the potential to directly reduce the cooling capacity of trees (Konarska et al., 2016). Lower soil volume in streetscapes provides less space for roots to grow, which can lead to stunted growth and reduced overall health (Scharenbroch et al., 2017). Trees growing in streetscapes can also experience nutrient deficiencies, as there may not be enough soil to support a healthy microbial population to break down organic matter and release nutrients into the soil (Mónok et al., 2019; Zhao et al., 2012). Other disturbances (such as physical damage and vandalism) can further negatively impact the growth rate, canopy shape, vitality and physiological activity (e.g., transpiration rate) of urban vegetation (Mullaney et al., 2015b). Such confounding effects will negatively impact levels of evapotranspiration and the amount of shade cast, thereby influencing the cooling capacity of urban trees (Coutts et al., 2012).

1.4 Tree traits and cooling benefits

Tree shade in particular creates localized cool areas beneath canopies by intercepting solar radiation and minimising exposure of shaded horizontal and vertical surfaces to direct sunlight (Kong et al., 2017). Tree transpiration involves the passive loss of water freely in unstressed condition via the stomata in the trees' leaves. The aperture of plant stomata on the epidermis of leaf controls transpiration and depends on the amount of solar irradiance (Jarvis & McNaughton, 1986). Transpiration allows cooling of leaves under abundant water conditions, driving large latent fluxes of heat that result in localised cooling (Zou et al., 2019). Under stress conditions, such as during periods of drought and high temperatures, most plants increase their water use efficiency by closing stomata and thereby reduce their transpiration rate and increase sensible heat flux (Chen & Avissar, 1994). The vertical and horizontal structures of trees – crown depth and width respectively - reflect the overall leaf surface area and associated transpiration. These traits also explain the capacity of the canopy to trap air below, or mix with air above (Peters et al., 2013). Therefore, the interplay of canopy traits along with solar irradiance, temperature, humidity, wind speed and soil moisture availability, impact the amount of water released and thus the trees' transpirational cooling potential (Tuzet, 2011).

In addition to transpirational cooling, species' morphological traits can determine their ability to reduce air and surface temperature (Armson et al., 2012; Lin & Lin, 2010). Rahman et al. (2019), in their study of nature strip plantings, reported the denser canopy (LAI = 3.64 m^2m^{-2}) of temperate species like *Tilia cordata* (deciduous tree) provided higher air and surface cooling benefits (1.6 °C and 25.6 °C, respectively), compared to the sparse canopy (LAI = 2.61 m^2m^{-2}) of species like *Robinia pseudoacacia* (deciduous tree) (0.6 °C and 12.7 °C, respectively). Species with high LAI, like *Ulmus parviflora* (deciduous or semi-deciduous tree) and *Ficus macrocarpa* (evergreen tree), were significantly more effective than *Bischofia javanica* and *Cassia fistula* (deciduous tree, each with lower LAI) in reducing both air and surface temperatures. Helletsgruber et al. (2020) showed that tree height and the height of the crown base were negatively correlated with the surface cooling benefits of trees. They also reported stronger effects of these two traits than LAI which is in contrast with the meta-analysis of Rahman et al. (2020). Speak et al. (2020) also evaluated correlations between tree traits such as canopy width, leaf size, leaf arrangements and LAI, and surface cooling benefits of trees reporting that LAI and crown width have the strongest relationship with cooling. The majority of studies evaluating the role of vegetation in combatting urban heat have focused on small geographic areas and small sets of tree species. This limits the applicability of findings to other regions due to differences in species composition and climatic conditions.

Urban morphology and microclimate influence the capacity of trees' to provide cooling benefits. For example, street canyons and the direction of streets impact on the ability of trees to cool the microclimate (Chen et al., 2021). A study conducted in Melbourne, Australia which has a Mediterranean climate, observed that trees in wide streets with shallow canyons could reduce daytime air temperature up to 0.9 °C, compared to narrow streets with high canyons (Coutts et al., 2016). Indeed, trees with high canopy density can have a warming effect in deep canyon streets (Morakinyo et al., 2017). Street orientation, the amount of solar radiation and wind direction and speed have all been shown to influence trees' capacity to reduce local temperatures at the street level (Rahman et al., 2020; Sanusi and Livesley, 2020). Critically though, during summer days trees play a crucial role in mitigating urban warming. All these factors need to be considered when exploring how urban tree planting in Western Sydney can promote heat mitigation.

1.5 Climatic conditions and cooling benefits

A comparative modelling study of the cooling potential (i.e., temperature reduction ability) of multiple mitigation technologies reported median and maximum temperature reductions for individual trees of 0.6 °C and 3.5 °C, respectively (Santamouris et al., 2017). This large range in potential cooling benefits could be due to differences in trees' response to variation in both air temperature and soil moisture (Kong et al., 2017). Under stress conditions, such as exposure to droughts and extreme heat, trees tend to close stomata and stop transpiring to avoid leaf heat damage (Schymanski et al., 2013).

Furthermore, the cooling benefits provided by trees can be affected by local topography and meteorological conditions (e.g., cloud cover, solar irradiance, wind speed) (Clay et al., 2016; Fabrizi et al., 2010; Koomen & Diogo, 2017; Steeneveld et al., 2011). The negative correlations between wind speed and cloud cover, and urban warming have been reported from several studies (Kim & Baik, 2004; Oke, 1982). For instance, comparatively small areas of Adelaide, in South Australia, experienced an UHI effect within a large urban area, with the effect diminishing as wind speed increased beyond 3 m s^{-1} (Clay et al., 2016). The intensity of urban warming is smaller in coastal cities due to higher moisture and atmospheric circulation compared to inland cities (Kim et al., 2004; Zhou et al., 2019).

Satellite remote sensing is one of the most common methods used to characterize changes in urban landscapes over time and understand the resultant urban warming (Patino & Duque, 2013). Usually, this method is used for larger, regional or city-scale evaluations (List of studies in Kadhim et al., 2016). However, the urban heat balance is modulated by microclimate, surface smoothness, soil moisture, building heights, street canyons and vegetation at the small spatial scale (Coutts et al., 2016; Oke et al., 2017). For this reason, empirical, ground-level research is needed to evaluate the respective roles of these parameters. However, since such research requires a great deal of effort and time and is costly (Kadhim et al., 2016; Tuholske et al., 2021), there have been relatively few ground-based studies of this nature but see Aguiar et al., (2014); Speak et al., (2020).

Australia has experienced record-breaking temperatures in recent years, in line with other countries across the globe (Lewis et al., 2017). Temperatures for Western Sydney suburbs are predicted to reach a maximum of 50 °C by 2040 (Ossola & Lin, 2021), with both the maximum temperatures and the number of days expereincing these highs predicted to increase. More research is needed to understand the impacts of future climate change in Australia and the role that vegetation can play in mitigating urban heat. To date, however, there have been only a handful of studies focusing on urban warming in Australia or Greater Sydney in particular (Kaluarachichi et al., 2020; Madden et al., 2018; Pfautsch & Rouillard, 2019a, 2019b; Santamouris et al., 2019; Wujeska-Klause & Pfautsch, 2020). In contrast, in the USA (Hall et al., 2016; Ibsen et al., 2021; Pincetl et al., 2013), Hong Kong (Kong et al., 2017; Ng & Cheng, 2012) and China (Huang et al., 2020; Zeng & Dong, 2015; Zhang, 2020) there have been many studies focusing on urban heat and the associated role of urban vegetation. While there is a clear knowledge gap in understanding the interplay among urban warming, the use of trees with desirable tree traits and local climatic conditions to mitigate global warming, this is particularly the case for the southern hemisphere in general and the Western Sydney region in particular.

1.6 Urban vegetation and biodiversity

Urbanisation has likely caused the extinction of thousands of species throughout human history (McKinney, 2002). Cities have also been established in biodiversity hotspots, as these areas are rich in resources (McDonald et al., 2008). As cities expand human selection pressures determine the size, shape and quality of greenspaces by introducing new species assemblages that result in novel ecosystems (Kowarik et al., 2020; Lepczyk et al., 2017).

Urban managers and ecologists invest much effort in increasing urban greenspace, often motivated by the need for restoration and conservation of local biodiversity (Dearborn & Kark, 2010). Evidence suggests that while urbanisation is a threat for plants and animals, planning and the introduction of diversely vegetated and structured natural areas can help to promote habitat and food resources for animals (Ahrne et al., 2009; Ives et al., 2016; Kowarik et al., 2020). However, to support and promote a diversity of animals within urban environments, it is important to consider their requirements for habitat and food resources and how these can be met in heavily built-up environments.

The conservation of invertebrates in urban settings has become a priority (Ives et al., 2016; Mata et al., 2014) due to their roles in facilitating plant reproduction, nutrient cycling and urban food production (Theodorou et al., 2020b). Furthermore, invertebrate groups also play key ecological roles in pest control and as food sources for other taxa —especially insectivorous birds, reptiles and microbats (Jones & Paine, 2006). Insects rely upon access to adequate food and habitat resources to complete their life cycles (Schmitt & Burghardt, 2021). For example, bees, butterflies and moths that rely feed on nectar and pollen need access to flowering plant species (Ahrne et al., 2009), while ground nesting wasps and bees, need bare soil to build their nests (Braschler et al., 2020). Despite recognition of the important roles that invertebrates play in all ecosystems, relatively little is known about the capacity of urban greenspaces to support diverse insect communities (but see (but see Kadas, 2006; McKinney, 2008; Smith et al., 2006).

Vegetation is integral to urban greenspace (De Haas et al., 2021) and mixtures of different plant growth forms create complex vegetation structures. Most natural habitats are shaped by the vegetation complexity. This, in turn, influences the distributions of, and interactions among, animal species (Lawton, 1983). An increase in understory vegetation, for example, has been shown to be associated with increased presence of different taxa such as insects, bugs and birds, by up to 40% (Threlfall et al., 2017). A comparative study among golf courses, parks and gardens also showed that golf courses with multiple layers of vegetation support more insects than other parks and gardens (Mata et al., 2016). These studies indicate that areas with more complex vegetation provide greater habitat diversity to support more abundant and/or diverse invertebrate assemblages. This complexity is often overlooked by urban planners and practitioners who frequently prioritise easy management or aesthetics at the expense of providing more complex landscapes comprising mixtures of trees and shrubs as well as understories of flowering herbs and grasses.

This thesis includes an evaluation of the capacity of different vegetation types (namely tree and shrubs) to support invertebrate biodiversity. Well-planned greenspace that incorporates biodiversity into the design of multi-functional urban landscapes, will help conserve and protect invertebrate biodiversity and provide heat mitigation. A review of the literature on urban greenspace indicates there is limited understanding of how planting contexts (e.g., parks or street settings) Influence the ability of trees to provide cooling benefits. This is despite evidence that growing conditions restrict plant access to water and soil resources, potentially influencing the growth and physiology of urban trees (Gillner et al., 2015). Local climatic conditions, solar irradiance and vapour pressure also have the potential to mediate tree-related cooling benefits. However, these have not been explored in a systematic manner and may account for the wide range of tree cooling (or warming) reported in the literature. My research contributes to addressing these gaps by evaluating relationships between canopy traits and microclimate for a wide variety of tree species, in different planting contexts.

Several earlier studies have documented positive effects of vegetation structural complexity on invertebrate diversity in urban ecosystems (Nooten et al., 2018; Threlfall et al., 2017). Vegetation in these studies varied from native, remnant forest to private gardens and golf courses, covering a variety of vegetation types and species richness as well as a wide range of climatic conditions. This wide scope makes it challenging to disentangle the role of vegetation structure from other factors that influence invertebrate populations. My study therefore uses a common-garden approach that controls for soil and climate conditions, to evaluate the role of planting complexity on plot-level invertebrate abundance and diversity.

1.7 Thesis objectives

This thesis has three data chapters that address the research gaps identified above:

I. Chapter Two: Evaluates the canopy traits that determine the cooling potential of urban trees and the impacts of planting context on their ability to reduce local air and surface temperatures.

II. Chapter Three: Links species-specific effects of trees on urban heat mitigation and explores how canopy-associated cooling is influenced by ambient climatic conditions along with the tree traits.

III. Chapter Four: Assesses the role of vegetation structural complexity in supporting invertebrate biodiversity in urban areas.

Chapter Two details an empirical study undertaken across Greater Sydney during summer 2018-19. This quantifies canopy-associated reductions in air and surface temperatures for a large number of urban trees, spanning 61 species, growing in different planting contexts (i.e., street *versus* park, surrounded by grass *versus* asphalt). Here, I evaluate relationships between canopy traits and the extent of air and surface temperature reductions, and contrast these across park and street planting contexts. Chapter Three extends the work presented in chapter two and evaluates species-specific effects of ten tree species on urban heat mitigation in Richmond and Cranebrook - suburbs of Western Sydney - over the summer of 2019-2020. In this chapter I specifically evaluate relationships between species-level traits and sub-canopy temperature benefits to identify species-specific effects of trees on urban heat mitigation. Chapter Four evaluates invertebrate assemblages associated with trees and shrubs and the role of vegetation structure for enhancing biodiversity in urban areas. This chapter examines how the abundance and richness of invertebrate taxa and functional groups vary among shrubs and trees, growing singly and together, in a common garden experiment at Western Sydney University's Hawkesbury Campus, over a three-year period (from 2019 to 2021). Overall thesis findings in the context of heat mitigation and biodiversity in urban areas – and suggestions for future research in this area – are discussed in chapter five.

CHAPTER TWO: Tree crown traits and planting context contribute to reducing urban heat

Abstract

Urban warming is affecting many millions of city dwellers worldwide. Trees in urban areas are widely recognized for their ability to regulate air and surface temperature, although this benefit can be moderated by the planting setting (i.e., context) in which they are growing. Different settings can vary in the amount of resources available for trees such as soil and water. The current study evaluated the extent to which trees are able to reduce air and surface temperatures in urban settings across Greater Sydney. Summertime air and surface temperatures were measured directly in the shade of 470 individual trees of 23 deciduous and 40 evergreen species, planted in three contrasting contexts (parks, nature strips, asphalt) and compared these with temperatures in paired adjacent areas receiving full sunlight. Differences between shade and sunlit temperatures were evaluated against measured morphological traits - leaf area index (LAI), clear stem height, crown depth, height and diameter at breast height - for all trees. On average, tree shade reduced mean and maximum air temperatures by 1.1 °C and 3.7 °C, respectively. Temperatures of standardised reference surfaces (black and white tiles and artificial grass) in tree shade were up to 45 °C lower compared to full-sun exposure, and were also lower in parks and on nature strips compared to asphalt. The surface temperature of shaded natural grass was cooler compared to sunlit natural grass although this difference did not vary between nature strip and park contexts. The magnitude of air and surface temperature reductions due to tree shade was significantly related to increasing LAI and these relationships were stronger in asphalt and park contexts compared to nature strips. These findings can inform decisions made by urban practitioners and urban planners around the selection of tree traits to enhance cooling benefits as an important step towards more liveable and resilient cities.

2.1 Introduction

Globally, mean annual temperature has increased by 1.1°C over the last century (IPCC, 2021b) and climate models predict a further increase to 1.8-4.0° C warming by the end of this century (IPCC, 2014, 2018). Furthermore, models predict an increase in the magnitude and frequency of heatwaves with longer warm seasons (IPCC, 2021a). In recent decades, record-breaking air temperatures, particularly during heatwaves, have been linked with increased human mortality rates in urban areas (Singh et al., 2015; Yang et al., 2019). In addition to climate change, urban densification makes cities warmer than surrounding non-urban areas (Mohajerani et al., 2017; Santamouris, 2013). Given that more than half of the world's population now live in cities (Roberts, 2011), exposure to extreme heat undoubtedly will threaten human health in the coming years (Basara et al., 2010; Heaviside et al., 2016; Lowe, 2016; Tan et al., 2010). Human health and environmental concerns are therefore driving the need to develop sustainable options to mitigate urban heat.

Urban areas are mosaics of buildings, streets, green spaces and different types of artificial surfaces that often cover up to half of a city (Fuller & Gaston, 2009). The high proportion of impervious surfaces with different radiative, thermal, aerodynamic and hydraulic properties, modify albedo and thereby the surface energy balance within the urban landscape (Oke, 1982; Ziter et al., 2019). The albedo of dark surfaces is typically in the range of 0.05–0.2 (Akbari, 2009), indicating that less incoming radiation is reflected by dark surfaces compared to either white surfaces (albedo of 0.45-0.8) or natural grass (albedo of 0.25-0.3) (Akbari, 2009; Angstrom, 1925; Radhi et al., 2014). Unshaded artificial surfaces with low albedo absorb a higher percentage of solar radiation than natural surfaces, and the subsequent re-radiation of accumulated sensible heat increases ambient air temperatures in cities, compared to rural, more vegetated areas (Mohajerani et al., 2017). Several strategies have been proposed in recent decades to mitigate urban heat. These include cooling streets by installing reflective surfaces (Santamouris, 2014; Yang et al., 2015) along with nature-based solutions, such as planting trees, increasing vegetation cover and construction of green walls and roofs (Rahman et al., 2019; Speak et al., 2020; Wujeska-Klause et al., 2020).

Urban energy balance can be significantly regulated by the intrinsic properties of surfaces such as their albedo, thermal characteristics and moisture content (Oke, 1982). Black and low albedo surfaces dominate the built environment across cities throughout the world (Mohajerani et al., 2017; Radhi et al., 2014). Light-coloured surfaces are highly reflective compared to darker coloured surfaces (Mohajerani et al., 2017) and represent an alternative to reduce urban heat when incorporated into new developments (Radhi et al., 2014). Artificial grass is widely used in school playgrounds, outdoor gyms and children's play areas (Pfautsch et al., 2019b), and typically has much higher surface temperatures than natural surfaces (Charalambous et al., 2016).

Studies using remote sensing show that urban vegetation cover can influence temperature at local and regional levels (Adams & Smith, 2014). A study in Kalaburagi City, Karnataka State, India, found that urban areas were 5 to 9 °C warmer than suburban areas with higher vegetation cover (Kumar & Shekhar, 2015), while Manoli et al. (2019) found that, globally, urban areas are up to 2°C warmer than suburban areas. The ability of trees to reflect almost 10% of visible light and 50% of incoming infrared radiation, among other attributes, can have a significant impact on local microclimate (Kong et al., 2017). Built surfaces that are shaded by trees can be much cooler than those exposed to the sun (Akbari et al. 1997). The difference in surface temperature (ΔT) underneath a tree canopy, compared to an adjacent sunlit area, provides a proxy for tree shade benefits (Lin and Lin, 2010). Indeed, shade cast by trees has been reported to reduce surface temperatures by up to 27 °C (Kaluarachichi et al., 2020).

Trees differ considerably in their capacity to intercept solar radiation. This reflects difference in tree architecture, including height, canopy width, canopy depth, diameter at breast height (DBH), canopy density (Kong et al., 2017; Pataki et al., 2011a; Shashua-Bar et al., 2010) and clear stem height (Heisler, 1986; Kong et al., 2017), as well as shape, orientation, surface characteristics and phenologies (e.g., deciduous versus evergreen) of leaves (Qin et al., 2014). Previous research has shown that trees with wider crowns are more effective at reducing heat gain of shaded surfaces during sunny days (de Abreu-Harbich et al., 2015; Heisler, 1986; Kong et al., 2017). Indeed, a study by Sabrin et al. (2021) in Philadelphia, a city with a humid subtropical climate, reported cooling benefits of trees with wider canopies and also those with greater height and DBH.

The cooling benefits of trees, beyond the physical property of casting shade, are enhanced through evapotranspiration. Leaves release water via evapotranspiration - a process which uses energy from sunlight to convert liquid water to vapour; this process increases humidity in surrounding areas and results in increased latent heat flux and associated cooling (Kong et al., 2017). Tan et al. (2020) and Rahman et al. (2014) reported that daily latent heat flux can vary among tree species from 1369 to 2200 W m^{-2}, resulting in considerable inter-specific variation in air temperature reduction (Rahman et al., 2017; Shashua-Bar et al., 2011).

The capacity of trees to affect the local microclimate also depends on growing conditions which, in urban contexts, can be modified, especially in relation to soil physical properties such as bulk density, compaction and water holding capacity (Mullaney et al., 2015a). In addition, hydrological dynamics is quite complex in urban areas. This is due to engineered rainwater harvesting systems and reduced water infiltration owing to impervious surfaces and compacted soil in comparison to natural areas (Norton et al., 2015). Urban trees growing in parks and gardens typically have access to relatively higher volumes of soil, potentially greater access to water and nutrients, and lower soil compaction (Konarska et al., 2016) compared to trees planted in pavements or asphalt (e.g., in car parks) (Mullaney et al., 2015b). Due to differences in growing conditions and local planting decisions, both tree species and their associated traits can vary across planting contexts. Moreover, streets are exposed to more radiant heat from the surrounding asphalt and concrete which, especially when water availability is limited, can exacerbate temperature and water stress in trees (Gillner et al., 2015) potentially limiting growth and evapotranspiration. As a consequence, street trees in particular, may have a lower potential to reduce surrounding temperatures via evapotranspiration, compared to those growing in less restricted soils, or exposed to less re-radiated heat. Higher surface temperature reductions associated with canopy shade have indeed been observed for surfaces (e.g., asphalt and concrete) than for evaporative surfaces (e.g., grass) (Massetti et al., 2019; Speak et al., 2020). Higher surface temperature reductions have also been reported in dry Mediterranean climatic conditions, compared to wet subtropical climates (Rahman et al., 2020). However, given that air temperatures in built up areas are typically considerably warmer than in surrounding, more vegetated areas (Ziter et al., 2019), the potential for urban vegetation to reduce local air temperatures could also be greater in street contexts than in parks and other large green spaces. Understanding air and surface temperature differences between different planting contexts is therefore useful for urban planning and planting decisions (Souch et al., 1993).

The objectives of this chapter were to: (1) determine whether air and surface temperature reductions by trees differ in contrasting planting contexts (i.e., park, nature strip [street trees with grass] and asphalt [trees without grass and surrounded by asphalt or concrete pavers]) (Figure 2.1); and (2) how tree traits are correlated with air and surface temperature reductions and whether these relationships differ among planting contexts. I hypothesized that: (1) the air and surface cooling benefits provided by trees will be lower in asphalt and nature strip contexts compared to parks due to relatively higher heat absorption properties of surrounding areas compared to vegetated areas, lower rate of evapotranspiration and smaller size of the trees ; (2) the extent (e.g., canopy width, tree diameter at breast height [DBH] and height) and density (e.g., leaf area index [LAI], depth) of tree crowns will be positively associated with greater reductions in air and surface temperatures compared to unshaded surfaces, and these relationships will differ among planting contexts.

2.2 Materials and Methods

2.2.1 Sites and tree selection

This study was conducted in five suburbs – the city of Sydney, Leichhardt, Marrickville, Parramatta and Penrith within the urban landscape of Greater Sydney, New South Wales, Australia. Greater Sydney has a humid subtropical climate (Khan et al., 2021). The city of Sydney, Leichhardt and Marrickville are located at the east of the Sydney whereas Parramatta and Penrith are located in the western part of the city. In summer, the western Sydney region experiences mean maximum temperatures of 23 °C and a mean minimum of 12 °C and is a few degrees warmer than the central and eastern areas (BOM, 2019b). Months of late summer and early autumn have more rain on average (BOM, 2023). Western Sydney has a lower mean annual precipitation (e.g., 705 mm in Penrith) compared to the coastal region and Central Business District (1213 mm) (BOM, 2018).

Figure 2. 1 Examples of urban trees in three planting contexts; (a) asphalt, (b) park and (c) nature strip.

2.2.2 Tree traits

Individual trees were identified across three urban planting contexts: parks, nature strips (i.e., street trees surrounded by grass) and those surrounded by asphalt/concrete (i.e., car parks and pavements) (Figure 2.1). Mature, disease free, individual trees with distinct (i.e. separate) fully grown untrimmed canopies were chosen, with species randomly selected across planting contexts. A total of 490 individual trees of 63 species (23 deciduous and 40 evergreen) were measured across the five suburbs (Table 2.1). The number of trees measured varied among suburbs. Sixty trees were sampled from the City of Sydney, 64 from Leichhardt, 68 from Marrickville, 57 from Parramatta and 240 from Penrith.

Tree height, DBH, canopy depth, clear stem height (i.e., stem height from ground to first branch of the tree), canopy width and leaf area index (LAI) were measured for each individual tree. Tree height and crown depth (i.e., distance from top to bottom of the crown) were measured using a Haglöf laser meter (L400, Haglöf, Sweden) with a height resolution of 0.1 m, angle resolution of 0.1° and an accuracy of 0.1°. Crown width measurements were taken using a measuring tape as the length of two orthogonal axes parallel with road/path from edge to edge through the crown centre and then averaged. Diameter at breast height (DBH) was measured at a height of 1.3 m above ground. LAI is the area of leaves per unit ground area under each individual tree canopy and was measured using a plant canopy analyser (LAI-2200, Li-COR, USA) on overcast days to improve contrast. Subsequent data analysis used Li-COR FV2200 software, deploying an isolated canopy model and removing the 5th mask (68°) for LAI calculation. Four LAI measurements per tree were taken at 1 m height from the ground and with a 90° view cap on a fish-eye lens.

Table 2.1 List of 61 tree species included in the survey across Greater Sydney.

Species name	Family name	Mature height	LAI	Origin
Acer negundo	Sapindaceae	25	5.8	Deciduous
Araucaria bidwillii	Araucariaceae	30	8.44	Evergreen
Araucaria heterophylla	Araucariaceae	50	5.41	Evergreen
Brachychiton acerifolius	Malvaceae	35	3.61	Deciduous
Brachychiton populneus	Malvaceae	10	4.01	Evergreen
Callistemon viminalis	Myrtaceae	9	4.21	Evergreen
Carya illinoinensis	Juglandaceae	30	5.63	Deciduous
Casuarina cunninghamiana	Casuarinaceae	30	4.45	Evergreen
Casuarina glauca	Casuarinaceae	20	3.19	Evergreen
Casuarina littoralis	Casuarinaceae	20	4.29	Evergreen
Cedrus deodara	Pinaceae	30	4.86	Evergreen
Celtis occidentalis	Cannabaceae	15	3.87	Deciduous
Cinnamomum camphora	Lauraceae	40	8.2	Evergreen
Corymbia citriodora	Myrtaceae	40	2.38	Deciduous
Corymbia maculata	Myrtaceae	30	2.5	Deciduous
Cupressus sempervirens	Cupressaceae	35	9.2	Evergreen
Cupressus sp	Cupressaceae	30	4.59	Evergreen
Eucalyptus crebra	Myrtaceae	35	2.4	Evergreen
Eucalyptus melliodora	Myrtaceae	30	1.87	Evergreen
Eucalyptus microcorys	Myrtaceae	60	4.52	Evergreen
Eucalyptus moluccana	Myrtaceae	30	2.23	Evergreen
Eucalyptus punctata	Myrtaceae	35	2.33	Evergreen
Eucalyptus saligna	Myrtaceae	35	2.51	Evergreen
Eucalyptus sideroxylon	Myrtaceae	35	3.59	Evergreen
Eucalyptus tereticornis	Myrtaceae	50	1.69	Evergreen
Eucalyptus torelliana	Myrtaceae	30	3.07	Evergreen
Ficus macrocarpa	Moraceae	15	5.45	Evergreen
Ficus nitida	Moraceae	15	4.94	Evergreen

Ficus rubiginosa	Moraceae	30	4.86	Evergreen
Fraxinus angustifolia	Oleaceae	30	4.2	Evergreen
Fraxinus nigra	Oleaceae	30	3.32	Evergreen
Fraxinus pennsylvanica	Oleaceae	20	2.33	Deciduous
Gleditsia triacanthos	Fabaceae	20	4	Deciduous
Grevillea robusta	Proteaceae	30	3.49	Deciduous
Hymenosporum flavum	Pittosporaceae	25	1.29	Deciduous
Jacaranda mimosifolia	Bignoniaceae	20	6.03	Deciduous
Lagerstroemia indica	Lythraceae	6	3.94	Deciduous
Liquidambar styraciflua	Altingiaceae	40	5.17	Deciduous
Lophostemon confertus	Myrtaceae	15	6.18	Evergreen
Magnolia grandiflora	Magnoliaceae	35	4.73	Deciduous
Melaleuca bracteata	Myrtaceae	8	4.11	Evergreen
Melaleuca decora	Myrtaceae	7	5.65	Evergreen
Melaleuca quinquenervia	Myrtaceae	12	4.11	Evergreen
Melaleuca styphelioides	Myrtaceae	20	5.96	Evergreen
Melia azedarach	Meliaceae	15	3.94	Deciduous
Metasequoia glyptostroboides	Cupressaceae	24	2.34	Deciduous
Olea europaea	Oleaceae	10	6.03	Evergreen
Pinus radiata	Pinaceae	30	4.05	Evergreen
Platanus acerifolia	Platanaceae	30	5.55	Deciduous
Podocarpus sp	Podocarpaceae	40	4.25	Evergreen
Populus deltoides	Salicaceae	30	3.77	Deciduous
Quercus palustris	Fagaceae	22	4.93	Deciduous
Quercus robur	Fagaceae	40	6	Deciduous
Robinia pseudoacacia	Fabaceae	25	3.74	Deciduous
Sapium sebiferum	Euphorbiaceae	10	6.19	Deciduous
Schinus areira	Anacardiaceae	15	4.14	Evergreen
Sequoia sempervirens	Fabaceae	60	7.13	Evergreen
Styphnolobium japonicum	Fabaceae	15	2.71	Deciduous
Syzygium sp	Myrtaceae	30	1.65	Evergreen
Tristaniopsis laurina	Myrtaceae	15	6.12	Evergreen
Waterhousea floribunda	Myrtaceae	30	5.37	Evergreen

2.2.3 Air and surface temperature measurements

Air and surface temperature data collection were conducted during the austral summer from mid-January to the end of March 2019, between 10:00 am to 16:00 pm on sunny days when air temperature was >25°C. Air temperature and wind speed were measured simultaneously using two hand-held weather stations (Kestrel 2250, Australia; accuracy ± 0.5 °C) at a height of 1.5 m above ground, underneath the tree canopy and in an adjacent, sunlit position. The Kestrel temperature sensor (thermistor) was aspirated and shielded from direct sunlight while measurements were collected. From mid-January to the end of March 2019, daily maximum air temperatures were extracted from the nearest weather stations of the suburbs (Table 2.2).

Table 2.2 Location and details of nearest weather stations within surveyed LGAs.

Suburbs	Weather Station	Station no.	Location	Height (Above sea level)	Location
Parramatta	Parramatta North (Masons drive)	066124	-33.79 °S, 151.02°E	55 m	Central Sydney
Penrith	Penrith Lake AWS	067113	-33.72 °S, 150.68 °E	25 m	Western Sydney
Marrickville Leichhardt	Sydney Observatory Hill	066214	-33.86 °S, 151.20 °E	43 m	Central Sydney
Richmond	Richmond RAAF	067105	-33.60° S, 150.78° E	19 m	Western Sydney

Three artificial surfaces (hereafter, standardised surfaces) of contrasting albedo – black (albedo :0.02) and white (albedo: 0.69) ceramic tiles and artificial grass (albedo: 0.18) - each 20×15 cm in size, were selected as standardised surfaces. Tiles were 0.32 cm and artificial grass was 0.3 cm in thickness. In situ surfaces, particularly paved impervious surfaces of roads, pavements and carparks differed greatly in colour and in composition. The use of standardised surface therefore controls for emissivity (i.e., its effectiveness in emitting thermal radiation), a property which influences measurement of surface temperatures using infrared thermometers. The use of black versus white ceramic tiles provide material of identical composition (ceramic) but differing albedo to provide a contrast. Ceramic or porcelain tiles have comparatively high levels of thermal conductivity and therefore absorb heat from the surface below in addition to absorbing incoming solar radiation directly; this is acknowledged as a limitation of this study. Artificial surfaces - black tiles, white tiles and artificial grass, were not insulated and only one pair of artificial surfaces were used for taking the temperature measurements. However, the middle parts of the tiles were filled with plywood, which reduces the thermal conductivity of surfaces. It is likely to be a small signal relative to the warming received from above and consistent for the three different types of artificial surfaces. In addition to that, each time of taking temperature measurements, the use of the previous set of heated artificial surfaces was avoided. Temperatures of existing natural grass surfaces surrounding study trees were also measured. Obviously, the native grass surfaces were not presented in asphalt / pavement settings. Since most native surfaces, apart from grass, were not uniformly present across planting contexts, these were excluded from the study to avoid ambiguity.

To measure surface temperature reductions associated with tree shade, one set of each surface type (i.e., black and white tiles and artificial grass) was placed directly under the shade of the tree crown and a second set of each was simultaneously placed in an adjacent area outside of the influence of crown shade, in full sunlight. During the initial data collection, the surface temperatures were tested to ensure they were stable for a period of five minutes; the measurements were stable in the five-seven minute window, therefore the measurements were taken during this time window. Subsequently, temperatures of standardised surfaces were measured along with the temperature of natural grass (referred to as “natural grass”), wherever this was present, with both sun and shade exposure, using a hand-held infrared thermometer (835-T1, Testro, Australia; resolution ± 0.1°C, accuracy of ± 0.5°C).

2.2.4 Data Analysis

Air and surface temperature reductions (ΔT) were calculated as the differences in temperatures recorded under tree canopies and those in adjacent areas exposed to full sunlight. Pairwise *t*-tests were used to compare surface temperatures measured in full sunlight and under tree shade; this, along with calculations of ΔT were used to address the first research question. To determine whether temperature differences varied among planting contexts, surface temperatures (natural grass and standardized surfaces) measured under tree shade and in full sunlight, and ΔT, were analysed using linear mixed effect models (*lmer*) followed by ANOVA analysis to determine best-fit models. Tukey post-hoc tests were used to assess models' significance as differences between the pairs of means. Daily maximum air temperature data sourced from the BOM and planting context were considered as fixed effects, while sampling dates and tree species were considered as random effects. The *lme4* package was used to perform *lmer* analyses (Bates et al., 2015).

Relationships between ΔT of air and surface temperatures as response variables and individual tree traits (i.e., tree height, DBH, canopy depth, clear stem height, canopy width and LAI) as explanatory variables were analysed using *lmer* to understand how tree traits are correlated with air and surface temperatures. To address the second research question - whether the above relationships vary among planting contexts - planting contexts were used as explanatory variable in the models. Tree species and suburbs were specified as random effects for each of the models. No interaction terms were tested. For each *lmer* model only one trait was used as an explanatory variable. Where necessary ΔT values for air and surfaces were log or square root transformed to correct for data skew. All individuals (n = 470, from 61 species) were used in the analysis, apart from 20 trees (two species) which were excluded due to measurements being conducted during periods of windy and overcast weather (Table 2.2). The statistical software package R version 3.5.1 was used for all analyses and plotting (R Core Team, 2020). A *p* value of 0.05 was considered as significant for all analysis.

2.3 Results

2.3.1 Air and surface temperature

Maximum temperatures of 82.1 °C (under air temperature 39.7 °C) and 89.7 °C (air temperature 38.4 °C) in full sunlight were recorded for black tiles. For white tiles and artificial grass, maximum temperatures were 62.1 °C (under air temperature 38.2 °C) and 64 °C (air temperature 37.2 °C), respectively. Under tree shade, the surface temperatures of black tiles

(47.8 °C) and artificial grass (57.7 °C) were higher than white tiles (44.2 °C) and natural grass (41.1 °C) (Figure 2.2 (a). Air temperature was significantly lower when measured in the shade of trees compared to direct sunlight (t = - 33.1, $p < 0.001$), with a mean ΔT_{air} between sun and shade of 1.1 °C (± 0.7), and maximum values of 3.7 °C. Surface temperatures were significantly lower under tree shade compared to temperatures in sunlight for black surfaces (t = -99.1, $p < 0.001$), white surfaces (t = - 68.1, $p < 0.001$) and artificial grass (t = - 81.1, $p < 0.001$) (Figure 2.2 (a); Supplementary Table 2.1 & 2.2). The temperature reduction associated with canopy shade (ΔT) varied among standard surface types (F = 1015.5, $p < 0.0001$) (Figure 2.2 (b)). Mean ΔT of black surfaces (28.1 °C ± 6.1 °C; max = 46.1 °C) and artificial grass (27.9 ± 7.6 °C; max = 51.9 °C) was significantly higher than the mean ΔT of white surfaces (11.4 ± 3.7 °C; max = 25.9 °C, $p < 0.001$) (Figure 2.2 (b)).

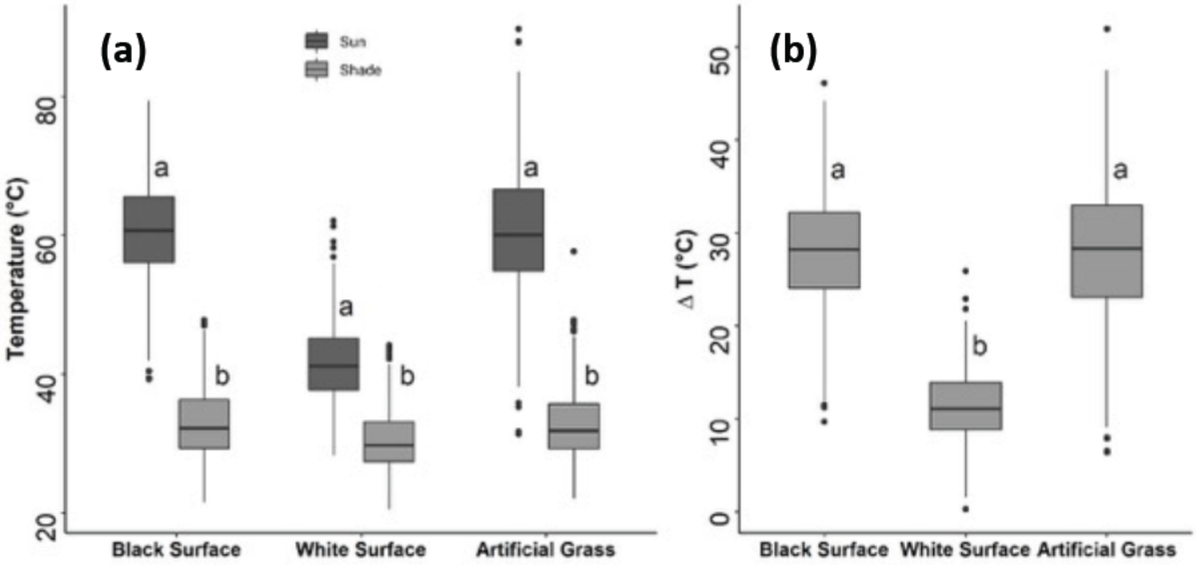


Figure 2.2 Box-whisker plots showing (a) pairwise comparison of temperatures measured on the standardized surfaces in direct sunlight (dark grey) and under the shade of tree crowns (light grey), and (b) temperature reduction (ΔT) in shade under individual tree crowns compared to adjacent unshaded surfaces in direct sunlight. Different letters indicate significant differences among surfaces at $p < 0.05$.

2.3.2 Comparison of planting contexts

Standardized surface temperatures varied across tree planting contexts, with the highest values recorded in direct sunlight for asphalt settings (black 61.2 ± 0.98 °C; white 43.6 ± 0.73 °C; artificial grass, 61.1± 1.10 °C), with lower values in parks (black 59.4 ± 0.91 °C; white 40.3 ± 0.68 °C; artificial grass 58.9 ± 1.00 °C) and nature strips (black 61.0 ± 0.90 °C; white 41.8 ± 0.68 °C; artificial grass 60.2 ± 1.00 °C).

Shaded standardized surface temperatures differed significantly (black, F= 12.5 , $p < 0.001$; white, F = 22.5, $p < 0.001$; artificial grass, F = 8.7, $p < 0.001$) among planting contexts, with consistently lower values in parks (black 32.1 ± 0.58 °C; white 29.4 ± 0.44 °C; artificial grass, 32.2 ± 0.55 °C) and nature strips (black 32.7 ± 0.58 °C; white 30.5 ± 0.44 °C; artificial grass, 32.9 ± 0.55 °C) than for measurements under trees surrounded by asphalt (black 34.6 ± 0.61 °C; white 31.8 ± 0.48 °C; artificial grass, 34.0 ± 0.59 °C) (Figure 2.3). However, ΔTs of standardized surfaces were not significantly different among planting contexts.

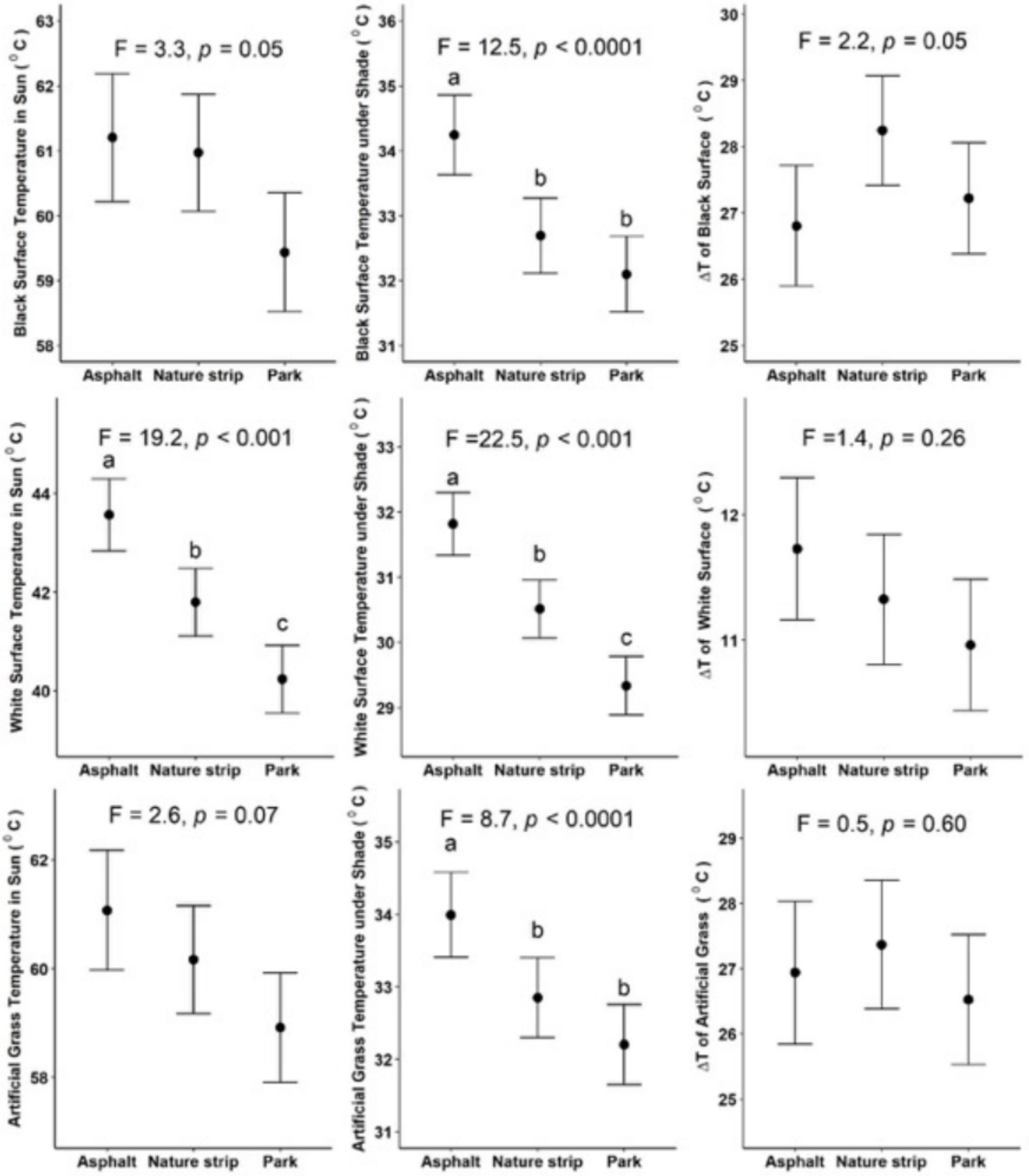


Figure 2.3 Surface temperatures of standardized surfaces in full sunlight (left, "in Sun"), under tree shade (middle, "under Shade") and temperature reduction under tree canopy compared to in sun (right "ΔT") in relation to planting context (asphalt, nature strip and park). Different letters indicate significant differences at $p < 0.05$. Values are given as means with standard errors and for all three surfaces the degree of freedom is 469. Analysis of variance (ANOVA) results of *lmer* models for each plot are presented in Supplementary Table 2.3.

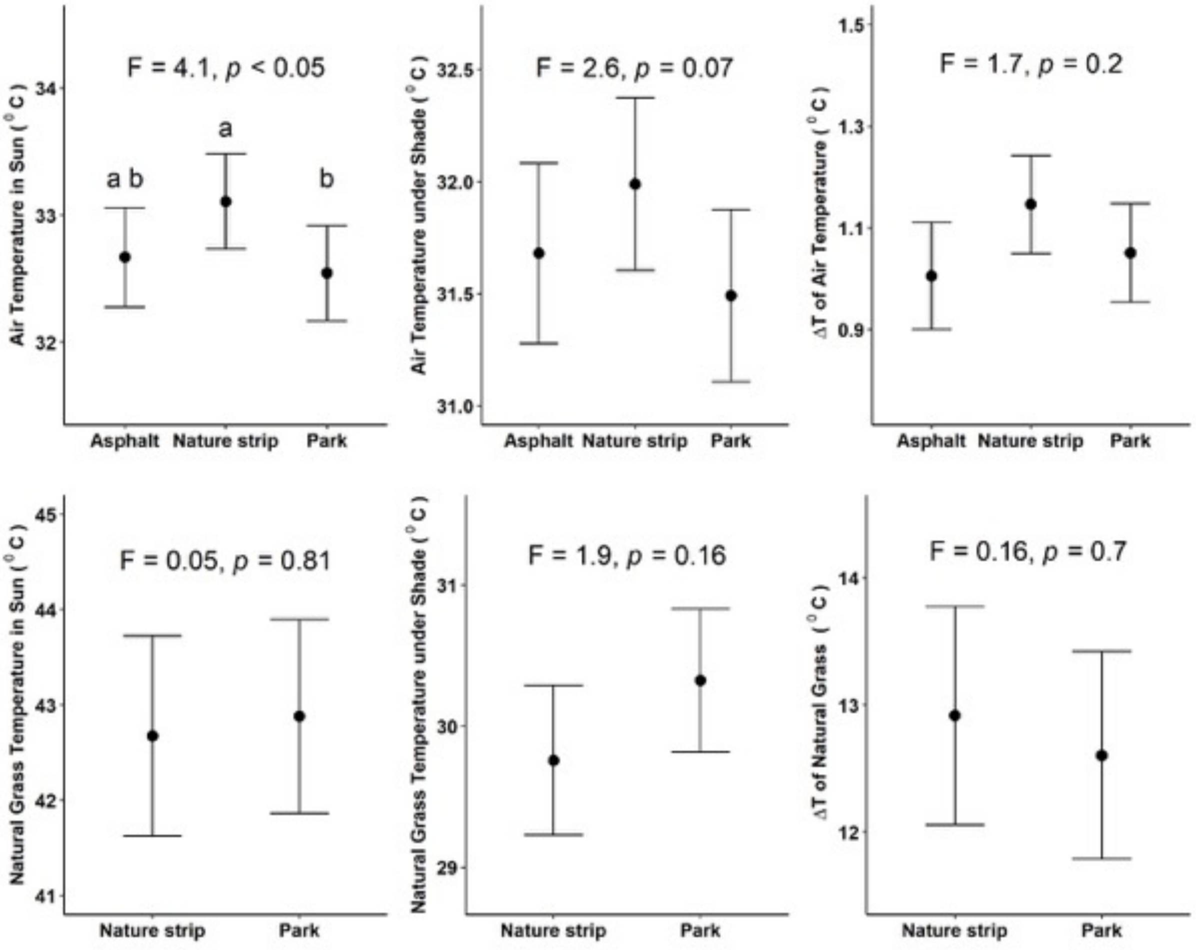


Figure 2.4 Temperatures of air and natural grass in full sunlight (left), under tree shade (middle), and temperature reductions under tree canopy compared to in sun (ΔT; right), among planting contexts. Different letters indicate significant differences at $p < 0.05$. Values are given as means with standard errors. The degree of freedom is 469 for air temperature and 274 for natural grass surfaces. For analysis of variance (ANOVA) results of *lmer* models see Supplementary Table 2.3.

Surface temperatures of natural grass in sunlight and shade, and ΔT, did not differ significantly between parks and nature strips (Figure 2.4). Mean air temperatures in direct sunlight were, however, higher in nature strips (33.1 ± 0.37 °C) than in parks (32.5 ± 0.37 °C) (Figure 2.4).

2.3.3 Canopy-associated temperature reduction and tree traits

The relationships of LAI with ΔT of air, black tiles and artificial grass differed significantly among planting contexts. Mean LAI in parks and nature strips was significantly higher than in asphalt settings (Supplementary Table 2.4). Park trees also had significantly deeper and wider crowns compared to trees in nature strips but did not differ from those in asphalt (Supplementary Table 2.4).

The overall air temperature reduction (ΔT) associated with tree shade increased with increasing LAI ($R^2 = 0.11$; $p = 0.02$). (Figure 2.5 (a); Supplementary Table 2.5), that was consistent among planting contexts. ΔT of air temperature did not show any response to changes in canopy width for all samples trees (Figure 2.6 (a)); however, it showed a positive relationship with canopy width for trees in park contexts ($R^2 = 0.35$; $p = 0.02$) (Figure 2.6 (b).

ΔT for black tiles ($R^2 = 0.20$; $p = 0.01$) and artificial grass ($R^2 = 0.25$, $p = 0.003$) were significantly, positively related to LAI (Figure 2.5 (b) & (c); Supplementary Table 2.5), with these relationships varying among planting contexts (Figure 2.7 (a) & (c); Supplementary Table 2.6). Positive relationships between LAI and ΔT for both black surfaces and artificial grass were significant on asphalt (black surface: $R^2 = 0.10$, $p = 0.02$; artificial grass: $R^2 = 0.28$, $p = 0.01$) and park (black surface: $R^2 = 0.22$; $p = 0.01$; artificial surface: $R^2 = 0.11$, $P = 0.01$) settings, but not on nature strips ($p > 0.05$) (Supplementary Table 2.6).

There was a significant context-dependency for the relationship between LAI, and ΔT of white surfaces, which was negative for nature strips ($R^2 = 0.33$, $P = 0.03$) and not significant for the other settings (Figure 2.7 (b)). In contrast, ΔT of white surfaces increase with increasing canopy depth ($R^2 = 0.33$, $P = 0.03$) and tree height ($R^2 = 0.33$, $P = 0.03$). None of the measured tree traits were significantly correlated with ΔT of natural grass (Supplementary Tables 2.5-2.8).

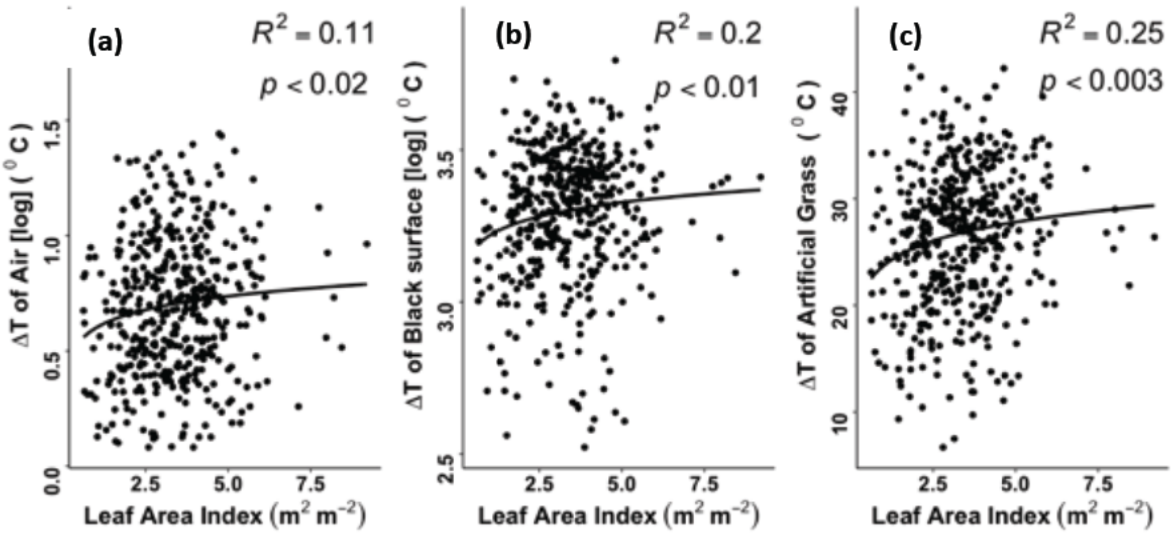


Figure 2.5 Relationships between leaf area index (LAI) and (a) log transformed ΔT for air temperature, (b) log transformed ΔT for black surfaces (tiles), (c) and ΔT for artificial grass, for all sampled trees. Noting R^2 values represent Rc^2, which is interpreted as the variance explained by both fixed and random factors. Details of the model parameters are summarised in Supplementary Table 2.5.

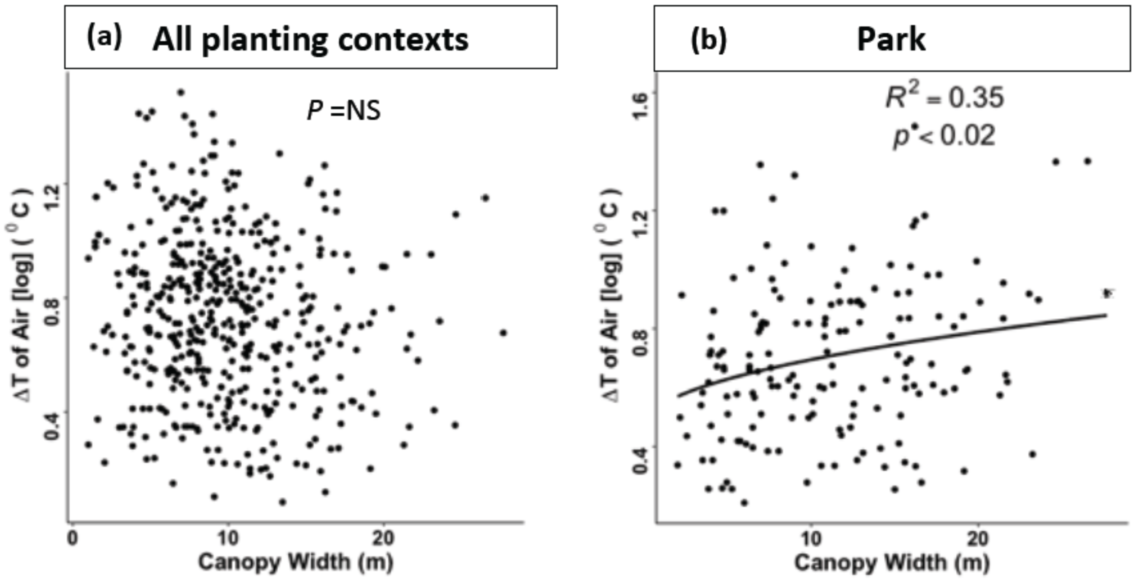


Figure 2.6 Relationships between ΔT of air and canopy width of all sampled trees in (a) three planting contexts and those in just (b) parks. Non-significant relationships are indicated as NS. Note R^2 values represent Rc^2 which is interpreted as the variance explained by both fixed and random factors. Details of the model parameters are summarised in Supplementary Table 2.7.

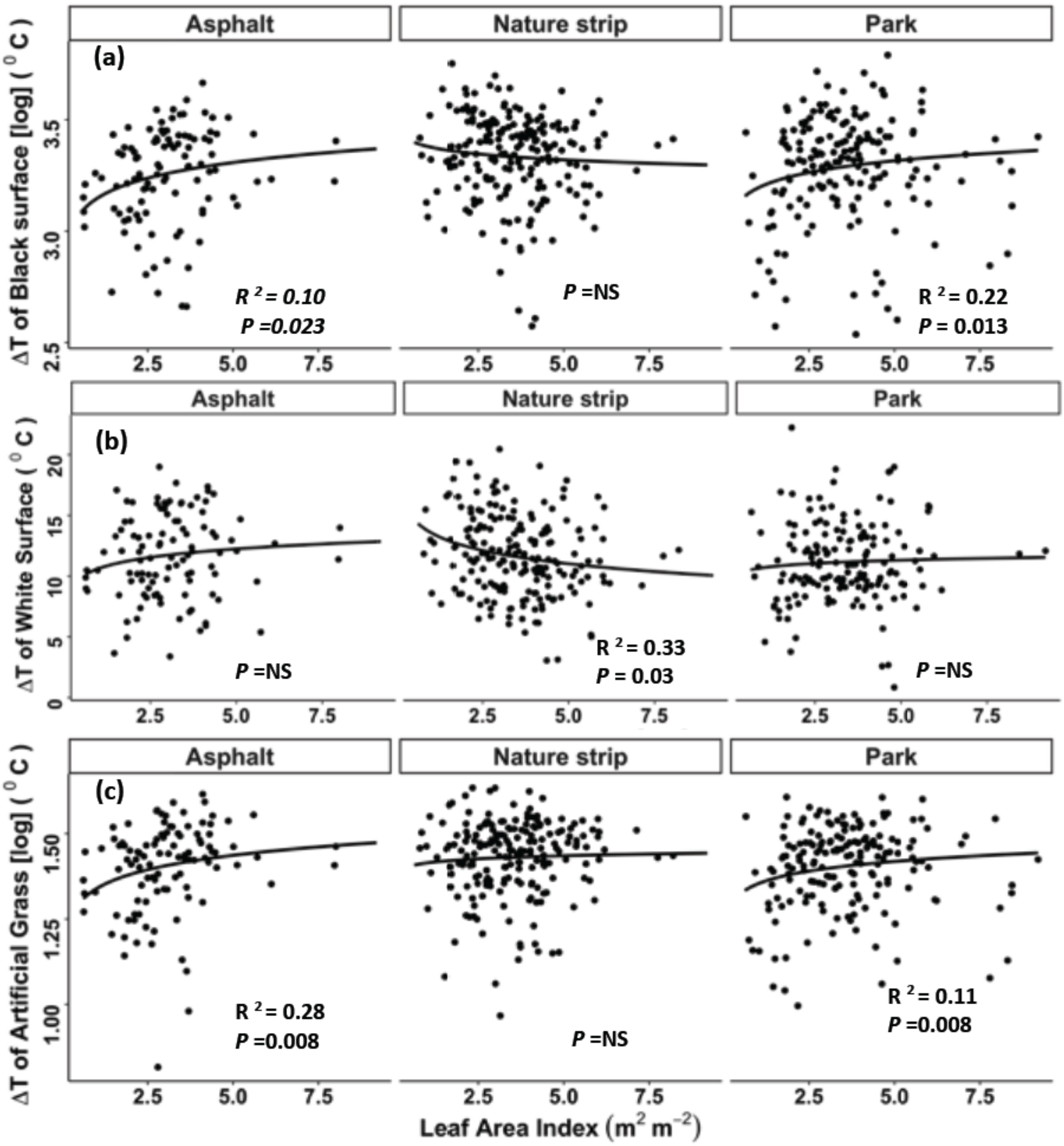


Figure 2.7 Relationships between ΔT of standardized surfaces ((a) black tile, (b) white tile and (c) artificial grass) and leaf area index (LAI) in three planting contexts (asphalt, nature strip, park) during summer 2018-2019. Note R^2 values represent Rc^2 which is interpreted as the variance explained by both fixed and random factors. R^2 and p values in each plot show the relationships between LAI and ΔT of the artificial surfaces for the three planting contexts individually. Non-significant relationships are indicated as NS. Details of the model parameters are summarised in Supplementary Table 2.6.

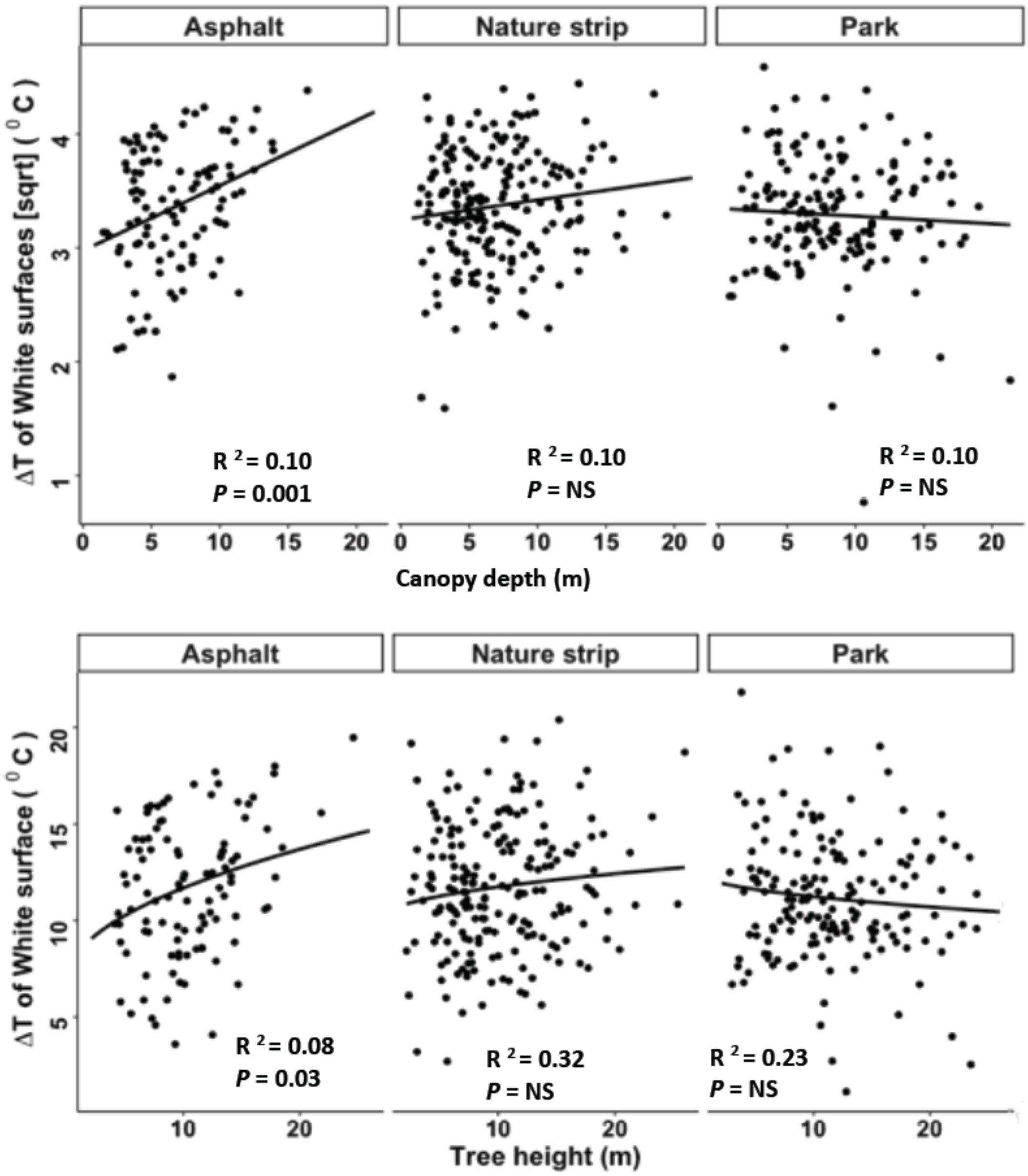


Figure 2. 8 Relationships between ΔT of white surfaces (tiles) and (a) canopy depth (R^2 = 0. = 23, P = 0.002) and (b) tree height (R^2 = 0.26, P = 0.01) in three planting contexts (asphalt, nature strip, park). Note R^2 values represent Rc^2 which is interpreted as the variance explained by both fixed and random factors. Non-significant relationships are indicated as NS. Details of the model parameters are summarised in Supplementary Table 2.7 & 2.8.

2.4 Discussion

This study demonstrated that planting context and tree crown traits can influence the magnitude of temperature reductions and cooling benefits provided by urban trees. Planting context affected the extent to which the temperatures of standard surfaces (tiles and artificial grass) were reduced by tree shade, with greater temperature reductions in parks and nature strips, compared to locations surrounded by asphalt. Air and surface temperature reductions were greater under trees with crowns that were more dense (high LAI), wide and deep, but the strength of these relationships varied among planting contexts.

2.4.1 Effects of trees on air and surface temperatures

This study provided evidence that urban trees can help to reduce air and surface temperatures under summertime conditions in SE Australia. Trees were able to reduce air temperatures by up to 3.7 °C (mean 1.1°C) compared to adjacent sunlit areas, which is similar to the reported cooling benefits of 1.5 °C (mean 0.9 °C) associated with trees in urban street canyons in Melbourne, Australia (Coutts et al., 2016). The findings of the current study are also in line with a study in Indiana, USA, a city with a humid tropical climate and summertime maximum daily temperatures of 28 to 30°C, where midday temperatures underneath tree canopies were 0.7–1.3°C lower compared to non-shaded areas (Souch et al., 1993).

Comparison of surface temperatures under tree shade and in full sunlight showed significant reductions for black and artificial grass surfaces (mean ΔT black = 28.1 °C and mean ΔT artificial grass = 27.9 °C), levels that are greater than previous studies. The study of Armson et al. (2012) in Manchester City, UK, recorded a shade-associated reduction in surface temperature of 19 °C (concrete) at mid-day, when maximum air temperatures were between 23.5 and 25 °C. In northern Italy, Speak et al. (2020) reported a similar reduction of 20.9 °C (asphalt) for measurements taken under a mean maximum ambient temperature of 28.3 °C. The air temperature ranges in both these studies were carried out were moderate compared to the current study. Aguiar et al. (2014) measured the effect of tree shading on asphalt surfaces on days between 25 to 30 °C in Wollongong – a city located to the south of Sydney. They reported a maximum reduction of 4.7 °C, which is substantially lower than the maximum of 46.1 °C found for black surfaces in the current study. However, a recent study in Greater Sydney reported canopy shade-associated reductions of 20.9 °C for asphalt surfaces under air temperature ranges that are much more in line with those observed in o study (Kaluarachichi et al., 2020).

Tree-associated cooling benefits for surrounding surfaces depend on the characteristics of the surface itself. Surface temperatures of natural grass in full sunlight and shade, and ΔT (mean 12.9 °C), were all lower than those of black tiles, artificial grass as well as white surfaces in the current study, although ΔT_{grass} was similar to the high albedo white surface (mean ΔT_{white} 11.4 °C). The ability of natural grass to reflect light, along with its evapotranspiration rate and water permeability, helps to maintain cooler temperatures compared to artificial surfaces (Rahman et al., 2020). However, it is worth noting that the cooling effect of grass may vary with different species, sward height, percentage cover and health/phenology (i.e. green versus brown/dead). Furthermore, the correlation between surface reflectivity and urban cooling has been demonstrated in numerous studies (e.g., Middel et al., 2020; Taha, 1997). However, non-reflective surfaces may result in greater heat absorption by nearby buildings, thereby incidentally resulting in lower pedestrian thermal comfort and/or the need for greater energy use to maintain suitable temperatures within buildings (Qin et al., 2016). The increased use of cooling energy can, of course, increase local air temperatures due to higher heat emissions from air conditioning systems (Sen & Khazanovich, 2021), highlighting the complex interplay between surface characteristics (including reflectance/albedo) and heat exposure at the local scale.

2.4.2 Air and surface temperature differences among planting contexts

Whilst urban trees can help to reduce surface and air temperatures, the contexts in which trees are planted may affect local-scale climate (Mullaney et al., 2015a). Many studies have investigated the effects of trees in either parks or streets independently (Colter et al., 2019; Park et al., 2019; Ren et al., 2021; Sanusi et al., 2016). Only a small number of studies have explicitly addressed how the planting context affects the ability of tree shade to modify air and surface temperature (Armson et al., 2012; Coronel et al., 2015; Shashua-Bar et al., 2011).

Incorporating natural grass alongside trees can produce greater surface cooling benefits in urban areas. In this chapter, temperatures of tree-shaded standardised surfaces (black and white tiles and artificial grass) were significantly lower in parks and nature strips than in asphalt contexts. This is in line with findings from an experiment in the hot humid climate of Tel Aviv, Israel, that reported that trees growing in grass settings are more effective than those in pavement (concrete) at reducing midday surface temperatures (Shashua-Bar et al., 2011). Vegetated surfaces evaporate available soil water and reflect solar irradiance, both of which play an important role in local energy balance (Gunawardena et al., 2017; Oke et al., 2017). It can also be assumed that street trees are pruned to reduce interference with overhead powerlines and for security reasons to prevent hazards as well as being trimmed by large vehicles such as trucks that go past when branches overhang roads; this may reduce the level of shading of the underlying surfaces.

The use of grass in nature strips may provide trees with greater rooting space and access to soil water than when they are planted into pavements or asphalt, with consequent benefits for tree growth and vitality (Mullaney et al., 2015a). However, grass and other understorey vegetation is also sensitive to water availability and incoming solar radiation (Broadbent et al., 2017). These factors may reduce the ability of grasses to transpire and stay green, although the shade provided by overstorey tree canopies could compensate for some of these issues (Gill et al., 2007).

Limited evidence was found to support the hypothesis that tree shading benefits for air temperature are higher in parks compared to asphalt and nature strips. In this study, coincidentally, some of the data on parks were collected during particularly hot days where maximum daily air temperatures were greater than 35 °C, which – if in excess of species' physiological thermal optima - may have limited the potential cooling benefits of trees (Gunderson et al., 2000).

2.4.3 Relationships between tree canopy traits and air/surface and minimum temperature reductions

Tree traits have remarkable effect on air and surface temperatures. Air temperatures were found to be influenced by the LAI and canopy width. Both LAI and canopy width were positively related to leaf surface area and, although it was not measured here, transpirational cooling is also generally linked to canopy-level leaf surface area (Foley et al., 2003; Grimmond & Oke, 1999).

An increase of each unit of LAI was associated with a 1.1 °C reduction in air temperature. However, it is noted that this cooling benefit only applies to the species included in the study, for the Sydney summertime conditions experienced during the study period and is derived from a fairly weak coefficient of correlation. Also, beyond an LAI of 2.5 – 3 $m^2 m^{-2}$, which typically represents full canopy cover, any further cooling benefits with increasing LAI (greater leaf density) are likely to be a result of higher canopy transpiration rates. However, this is speculative and therefore a topic that warrants further research.

Dense and wide canopies can intercept large amounts of incoming short-wave radiation via reflection (Gillner et al., 2015; Kong et al., 2017; Pataki et al., 2011a; Qin et al., 2014). A comparable study in Taipei in sub-tropical Taiwan by Lin & Lin, (2010) reported that air temperature decreased by 0.29 °C with each unit of LAI. Sanusi et al. (2017) linked plant area index (PAI), another index of canopy density which measures branches, twigs, flower and fruits, to air temperature and observed that air temperature decreases significantly with increasing PAI. It is reasonable to expect that incorporating all of the above-mentioned tree features, along with leaves, will enhance the effect of tree shade on air and surface temperature. Furthermore, Wang et al. (2021a) reported that canopy size (i.e., crown radius) had a strong positive effect on physiological equivalent temperature calculated by air temperature, relative humidity, wind speed and mean radiant temperature an index commonly used to measure a tree's cooling effect on humans, and which is highly correlated with air temperature (Mayer & Höppe, 1987). Therefore, the current study adds to a growing body of empirical data supporting the selection of trees with dense and wide canopies to help mitigate warming in urban areas. However, there is also a growing body of evidence suggesting that wide and dense canopies can impede air movement, trapping heat underneath them and thereby contributing to night-time urban warming (Bowler et al., 2010; Wang et al., 2021b; Wujeska-Klause et al., 2020).

Almost all tree traits except DBH and clear stem height influence surface temperature. Among the artificial surfaces, ΔT of black tiles and artificial grass increased with LAI for asphalt and park settings, which indicates that street trees influence local microclimate in a similar way to those in parks. However, these relationships were not evident in nature strips. This may reflect the relatively greater homogeneity of asphalt and park settings compared to the more variable nature strips which typically include a mixture of asphalt, bare ground and grass. In terms of surfaces, relationships between canopy-associated temperature reductions and tree traits were stronger for white surfaces than black tiles. It was observed that ΔT of white surfaces showed a positive interaction with canopy depth and tree height and as hypothesized, this was more evident for asphalt contexts. These results indicate that the temperature of highly reflective (light) surfaces in parks and nature strips can be managed – at least to some extent - by planting trees with desirable traits.

The findings from this study highlight the need to evaluate the extent of both tree and black (absorptive) surface cover in the design and management of urban spaces from a heat perspective. Although it not feasible to replace all dark or impervious surfaces with light and/or native materials, strategic targeting of areas to enhance levels of tree shade and the use of permeable surfaces can help to mitigate urban warming.

2.5 Conclusion

This study evaluated canopy-associated air and surface cooling benefits for a large number of urban trees with contrasting traits in three different planting contexts. The demonstrated context-dependency of air and surface temperatures advances existing understanding of tree cooling benefits in urban areas. It also highlights the complexity in relationships between canopy traits and temperatures across different parts of the urban landscape, especially given that some traits, such as LAI and canopy width, were higher in parks than trees were growing in asphalt. To mitigate urban heat, planting of trees with dense (high LAI) and wide canopies is recommended, particularly in areas with little tree cover and/or large areas of dark colored built environment which absorb high levels of solar radiation. The findings of this study can be used to support decisions by policy makers, urban practitioners and planners aiming to reduce urban heat and increasing the liveability of cityscapes across the globe.

Supplementary Tables

Supplementary Table 2.1 Temperature of black tiles, white tiles, artificial grass and natural grasses in sun and under tree shade.

Surface	Temperature in sun (°C)		Temperature under Shade (°C)	
	Mean ± SD	Max	Mean ± SD	Max
Black tiles	61.0 ± 7.4	82	32.9± 5	47.8
White tiles	41.82 ± 5.7	62.1	30.5±4.31	42
Artificial grass	60.4 ± 8.9	89.7	32.8±5.03	57.7
Natural grass	43.4 ± 8	64	30.4 ±4.4	41.1

Supplementary Table 2.2 Pairwise comparison of black tiles, white tiles, artificial grass, natural grass and air temperature in direct sunlight and under tree shade.

Surface/Air		t value	df	*p*
Black tiles	Shade vs Sun	-99.7	513	<0.001
White tiles	Shade vs Sun	-67.4	513	<0.001
Artificial grass	Shade vs Sun	-81.0	513	<0.001
Natural grass	Shade vs Sun	-37.8	306	<0.001
Air	Shade vs Sun	-33.1	513	<0.001

Supplementary Table 2.3 Analysis of variance (ANOVA) results of *lmer* models to test whether artificial surfaces and air temperatures varied among planting contexts in full sunlight, under tree shade and differences between the two (ΔT). Planting context and daily temperature were considered as fixed effects, and dates and tree species ID were random effects.

		Sum sq	**Mean sq**	**NumDF**	**DenDF**	**F-value**	***p* value**
	Black tiles						
Sun	Context	214.01	107.01	2	469	3.27	0.05
	Daily Temp*	927.81	927.81	1	19.11	28.50	**0.00**
Shade	Context	227.64	113.82	2	469	12.51	**0.00**
	Daily Temp	475.31	475.31	1	19.13	52.27	**0.00**
ΔT	Context	136.33	68.16	2	469	2.205	0.11
	Daily Temp	46.26	46.26	1	19.12	1.50	0.24
	White tiles						
Sun	Context	528.24	264.12	2	469	19.25	**0.00**
	Daily Temp	749.15	749.15	1	19.51	54.60	**0.00**
Shade	Context	298.98	149.49	2	469	22.45	**0.00**
	Daily Temp	561.55	561.55	1	19.51	84.35	**0.00**
ΔT	Context	29.38	14.70	2	469	1.35	0.26
	Daily Temp	35.56	35.56	1	19.51	3.27	0.09
	Artificial grass						
Sun	Context	246.19	123.09	2	469	2.57	0.08
	Daily Temp	2170.58	2170.58	1	19.48	45.25	**0.00**
Shade	Context	154.02	77.01	2	469	8.68	**0.00**
	Daily Temp	670.25	670.25	1	19.48	75.58	**0.00**
ΔT	Context	48.31	24.15	2	469	0.52	0.59
	Daily Temp	309.14	309.14	1	19.48	6.70	0.02
	Air temperature						
Sun	Context	20.54	10.27	2	469	3.84	**0.02**
	Daily Temp	338.75	338.75	1	19.47	126.76	**0.00**
Shade	Context	14.76	7.38	2	469	2.61	0.07
	Daily Temp	348.48	348.48	1	19.54	123.21	**0.00**
ΔT	Context	1.19	0.60	2	469	1.73	0.18
	Daily Temp	0.09	0.0915	1	19.55	0.27	0.61
	Natural grass						
Sun	Context	1.98	1.98	1	274	0.05	0.82
	Daily Temp	510.27	510.27	1	16.24	13.52	**0.00**
Shade	Context	12.89	12.89	1	274	1.91	0.17
	Daily Temp	308.16	308.16	1	14.75	45.78	**0.00**
ΔT	Context	4.62	4.62	1	274	0.16	0.69
	Daily Temp	13.26	13.26	1	16.26	0.45	0.51

*Daily maximum temperature

Supplementary Table 2.4 Pairwise comparison of tree traits between planting contexts. All trait values were log transformed.

Tree traits	Planting context	Diff	Lower value	Upper value	*p* value
Leaf area index	Nature strip-Asphalt	0.15	0.03	0.28	**0.01**
	Park-Asphalt	0.15	0.02	0.28	**0.02**
	Park-Nature strip	-0.01	-0.12	0.10	0.99
Tree height	Nature strip-Asphalt	-0.08	-0.22	0.06	0.36
	Park-Asphalt	0.03	-0.11	0.17	0.85
	Park-Nature strip	0.11	0.00	0.23	0.06
DBH	Nature strip-Asphalt	-0.10	-0.31	0.11	0.48
	Park-Asphalt	0.01	-0.20	0.23	0.99
	Park-Nature strip	0.12	-0.06	0.30	0.28
Canopy depth	Nature strip-Asphalt	0.00	-0.16	0.16	1.00
	Park-Asphalt	0.15	-0.02	0.31	0.09
	Park-Nature strip	0.14	0.01	0.28	**0.03**
Canopy width	Nature strip-Asphalt	-0.11	-0.26	0.04	0.20
	Park-Asphalt	0.09	-0.07	0.24	0.41
	Park-Nature strip	0.20	0.07	0.33	**0.00**
Clear stem height	Nature strip-Asphalt	-0.18	-0.35	0.00	**0.04**
	Park-Asphalt	-0.20	-0.38	-0.03	**0.02**
	Park-Nature strip	-0.03	-0.17	0.12	0.90

Supplementary Table 2.5 ANOVA results of *lmer* models for temperature reduction under tree canopies compared to in full sun (ΔT) for air, black tiles, white tiles, artificial grass and natural grass, and the canopy trait leaf area index (LAI) for the full dataset.

ΔT (°C)	Variable	Rc^2 / Rm^2	*p* value
Air	LAI	0.01/0.11	0.02
Black tiles	LAI	0.01/0.10	0.01
White tiles	LAI	0.01/0.20	NS
Artificial grass	LAI	0.02/0.25	0.003
Natural grass	LAI	0.01/0.05	NS

Supplementary Table 2.6 ANOVA results of lmer models for temperature reduction under tree canopies compared to in full sun (ΔT) for air, black tiles, white tiles, artificial grass and natural grasses, and the canopy trait, leaf area index (LAI) for all three planting contexts.

ΔT (°C)	**Variable**	**All planting context**		**Asphalt**		**Nature strip**		**Park**	
		X^2	***p* value**	**X^2**	***p* value**	**X^2**	***p* value**	**X^2**	***p* value**
Air	LAI	4.23	0.04						
Black tiles	LAI	5.80	0.02	5.40	0.02	0.20	NS	6.41	0.01
	Context	15.65	0.00						
	LAI X context	9.55	0.01						
White tiles	LAI	1.62	NS	2.52	NS	(-)4.39	0.04	0.02	NS
	Context	9.58	0.01						
	LAI X context	8.76	0.01						
Artificial grass	LAI	6.67	0.01	6.95	0.008	1.13	NS	6.97	0.01
	Context	6.22	0.04						
	LAI X context	4.11	NS						
Natural grass	LAI	2.44	NS						
	Context	0.19	NS						
	LAI X context	1.02	NS						

Supplementary Table 2.7 ANOVA results of *lmer* for temperature reduction under tree canopy compared to in full sun (ΔT) for air, black tiles, white tiles, artificial grass and natural grasses, and the tree traits crown width and crown depth for all three planting contexts.

ΔT (°C)	Variables	X^2	*p* value	Variables	X^2	*p* value
Air	Canopy width	1.5	NS	Canopy depth	1.04	NS
	Context	7.92	**0.02**	Context	6.73	NS
	Canopy width X context	10.72	**0.004**	Canopy depth X context	5.67	**0.06**
Black tiles	Canopy width	2.38	NS	Canopy depth	2.61	NS
	Context	8.90	**0.01**	Context	3.24	NS
	Canopy width X context	3.35	NS	Canopy depth X context	1.46	NS
White tiles	Canopy width	0.01	NS	Canopy depth	9.82	**0.001**
	Context	2.61	NS	Context	6.64	**0.04**
	Canopy width X context	8.33	**0.02**	Canopy depth X context	9.12	**0.01**
Artificial grass	Canopy width	0.39	NS	Canopy depth	1.35	NS
	Context	2.68	NS	Context	1.72	NS
	Canopy width X context	5.4	NS	Canopy depth X context	0.97	NS
Natural grass	Canopy width	0.39	NS	Canopy depth	1.04	NS
	Context	0.03	NS	Context	0.93	NS
	Canopy width X context	0.018	NS	Canopy depth X context	0.87	NS

Supplementary Table 2.8 ANOVA results of *lmer* for temperature reductions under tree canopy compared to in full sun (ΔT) for air, black tiles, white tiles, artificial grass and natural grass, and the tree traits DBH, clear stem height and tree height, for all three planting contexts.

ΔT (°C)	**Variables**	**X^2**	***p* value**	**Variables**	**X^2**	***p* value**	**Variables**	**X^2**	***p* value**
Air	DBH	0.17	NS	Clear stem height	1.01	NS	Tree height	0.07	NS
	Context	5.25	0.07	Context	4.62	NS	Context	**6.08**	**0.04**
	DBH x context	2.76	NS	Clear stem height X context	1.99	NS	Tree height X context	4.33	NS
Black tiles	DBH	0.52	NS	Clear stem height	0.01	NS	Tree height	1.25	NS
	Context	1.44	NS	Context	0.84	NS	Context	2.04	NS
	DBH x context	0.11	NS	Clear stem height X context	0.75	NS	Tree height X context	0.94	NS
White tiles	DBH	1.46	NS	Clear stem height	0.10	NS	Tree height	**6.12**	**0.01**
	Context	3.24	NS	Context	3.63	NS	Context	**6.95**	**0.03**
	DBH x context	5.55	0.06	Clear stem height X context	5.80	0.05	Tree height X context	**9.10**	**0.01**
Artificial grass	DBH	0.77	NS	Clear stem height	0.001	NS	Tree height	0.71	NS
	Context	0.82	NS	Context	0.91	NS	Context	1.54	NS
	DBH x context	0.43	NS	Clear stem height X context	0.34	NS	Tree height X context	0.86	NS
Natural grass	DBH	0.39	NS	Clear stem height	0.03	NS	Tree height	0.43	NS
	Context	0.03	NS	Context	0.48	NS	Context	0.67	NS
	DBH x context	0.02	NS	Clear stem height X context	0.35	NS	Tree height X context	0.60	NS

CHAPTER THREE: Tree traits and microclimatic conditions determine cooling benefits of urban trees

Abstract

Trees play a key role in mitigating urban heat by cooling the local environment. However, the extent to which trees can provide such temperature benefits is influenced by differences in species' traits and prevailing climatic conditions. This study evaluated the extent to which tree species can reduce canopy air temperature, relative to ambient conditions (termed "delta temperature", ΔT) and how ΔT relates to tree traits and microclimatic variables. Air temperature under the canopies of the ten most abundant street tree species was recorded for ten replicates per species within residential areas in Western Sydney, Australia, during summer 2019/2020. Tree trait measurements included tree height, specific leaf area, leaf dry matter content, leaf area index (LAI), crown width and the Huber value (the ratio of sap wood area to leaf area). Data from the nearest meteorological stations were sourced to calculate ΔT and extract climatic variables (vapor pressure deficit [VPD], solar irradiance and windspeed). Correlations between the magnitude of ΔT and microclimatic variables or tree species/traits vary among different parts of the day. In the morning, sub-canopy temperatures of individual trees were lower (mean maximum 3.9 °C) than ambient (i.e., negative ΔT), including during periods of high solar radiation and VPD. The extent of canopy-associated cooling was, however, smaller in the afternoon and at night-time. Trees with high LAI and wider canopies were associated with the greatest cooling benefit (i.e., more negative values of ΔT). Species differed significantly in their ΔT values with the highest daytime cooling observed for *Platanus × acerifolia* (mean 3.0 °C; mean 3.62 $m^2 m^{-2}$.) and the lowest for *Jacaranda mimosifolia* (mean 1.27 °C; mean LAI 1.75 $m^2 m^{-2}$). The findings of this study provide valuable information on how tree triats and microclimate influence potential cooling benefits that may aid planning decisions on the use of trees to mitigate heat in urban greenspace.

3.1 Introduction

Global mean air temperature has increased by 1 °C compared to pre-industrial times and is expected to reach +1.5 °C by 2050 (IPCC, 2021a). Climate models predict that global warming will be associated with more frequent, severe and intense extreme heat events (IPCC, 2021a; Meehl & Tebaldi, 2004). Within cities, increasingly extreme temperatures lead to challenging conditions for urban residents, particularly during heatwaves (Tuholske et al., 2021). Therefore, a greater understanding of mitigation strategies to reduce urban heat is necessary to improve local conditions for urban residents.

Urban areas are mosaics of buildings, streets and different types of impervious surfaces, along with green and blue infrastructure (Elmqvist et al., 2013). Impervious surfaces (such as concrete, asphalt and buildings) can cover up to 50% of urban areas, although this cover varies considerably across cities (Fuller et al., 2009). Urban areas also differ in their radiative, thermal, aerodynamic and moisture properties relative to surrounding peri-urban and natural areas, absorbing heat and re-radiating longwave radiation within the urban matrix, resulting in higher air and surface temperatures (Manoli et al., 2019; Oke, 1982). As a consequence, air and surface temperatures in urban and peri-urban areas are typically higher than in the surrounding rural landscape — a phenomenon known as the urban heat island effect (UHI) (Zhao et al., 2014).

Several urban heat mitigation strategies are available globally, which can modify the heat balance of urban areas. However, nature-based solutions (e.g., green roofs and walls, planting trees, presence of water bodies) are highlighted as sustainable, cost-effective ways of mitigating urban heat and improving the liveability of cities across the globe (Ossola et al., 2021). Nature-based solutions also offer other benefits such as increased biodiversity in urban areas (Bates et al., 2011; Kabisch et al., 2022; Threlfall et al., 2016). Trees are a key component of urban greenspace and have a demonstrated ability to reduce local air temperatures (Akbari et al., 2001; Bowler et al., 2010). A modelling study of four cities from different climate zones (Melbourne [Australia], Zurich [Switzerland], Phoenix [USA] and Singapore [Singapore]) reported that vegetation cover can decrease daytime maximum air temperatures by 3.1 to 5.8 °C (Meili et al., 2021). In contrast, in empirical studies, tree canopy has been associated with air temperature reductions of 2.8 °C in Campinas, Brazil (de Abreu-Harbich et al., 2015), 0.1 °C in Indiana, USA, (Souch et al., 1993) and 1.1 °C in Greater Sydney, Australia (This thesis, Chapter two). The extent to which tree crowns reduce air temperature varies substantially across the studies (Ibsen et al., 2021; Meili et al., 2021). From these studies, it is clear that the cooling benefits associated with urban trees vary across regions, likely due to the nature of the surrounding built-up areas, the vegetation cover, soil water availability, local microclimate and time of the day (Motazedian et al., 2020). Although, cooling benefits of trees are widely expected, there is evidence that trees can increase sub-canopy air temperature at night-time by reducing air circulation and trapping warm air within the canopy (Oke 1989; Wujeska-Klause et al., 2020). This nocturnal effect of tree canopies can intensify the urban warming phenomenon. Therefore, the nocturnal effect of tree canopies needs to be explored further.

Trees cool the surrounding area directly by blocking solar radiation and transpiring water in the atmosphere (Winbourne et al., 2020). Among tree traits, morphological traits, such as tree size, crown width, crown density, leaf dry matter content (LDMC), leaf area (LA) and specific leaf area (SLA) relate to the amount of shade cast and light reflectance as well as the transpiration rate (Fauset et al., 2018; Kong et al., 2017; Wang et al., 2019a). The huber value reflects a tree's transpirational capacity under optimal conditions when water is not limiting. Shaded surfaces absorb less solar irradiance therefore re-radiate less heat from the surfaces which therefore maintain cooler air temperatures underneath tree crowns (second chapter, this thesis). Both LDMC and SLA are negatively correlated with leaf thickness (Wilson et al., 1999) and thinner leaves can effectively cool down air temperature (Rahman et al., 2020), due to their thinner leaf boundary - a thin layer of still air that surrounds each leaf. The Huber value (i.e., the ratio of stem xylem cross-sectional area to leaf area supported by the respective stem segments) (Sellin et al., 2015) represents the amount of leaf area of a stem that can transpire water (de Abreu-Harbich et al., 2015) hence a tree's capacity for transpirational cooling (Tyree & Ewers, 1991).

On clear days, during daytime, plants open their stomata for photosynthesis and reduce leaf and ambient air temperature by transpiring water to the environment (Waring & Silvester, 1994). Transpiration reduces heat by increasing latent heat flux, as energy is used to evaporate water from leaves (Grossiord et al., 2020; Pataki et al., 2011b). Transpiration rate is regulated by solar irradiance, VPD (vapor pressure deficit) and windspeed (Grossiord et al., 2020), with stomatal opening generally greatest at biologically optimal solar irradiance, VPD and windspeed (Aphalo & Jarvis, 1993). High windspeed conditions disrupt the leaf boundary layer enhancing CO_2 and H_2O diffusion (Carvalho et al., 2015) resulting in increased transpiration. In many species, the opening of stomata at night has been observed (Chen et al., 2011; Konarska et al., 2016). Night-time plant transpiration can be up to 20% of daytime transpiration levels and is also positively correlated with VPD (Zeppel et al., 2010). As a result, night-time cooling due to plant transpiration can be increased under and surrounding tree crowns (Konarska et al., 2016).

The cooling benefits of urban trees - delta temperature (ΔT, the difference between ambient air temperature and sub-canopy air temperature), vary among species and depend on plant traits, tree structure and crown traits (de Abreu-Harbich et al., 2015; Rahman et al., 2020). Knowing to what degree different tree species and associated traits are likely to provide microclimatic benefits in summer conditions can improve urban liveability and thereby the wellbeing of urban residents. However, although some species can provide greater cooling benefits than others (Lin & Lin, 2010; Rahman et al., 2014), it is not feasible to evaluate all urban trees for any given city. Therefore, a traits-based approach that identifies those tree traits that are broadly associated with greater shade and evapotranspiration rates, and hence high cooling benefits, is a more feasible approach.

Given that cities in Western Sydney, Australia, have recently experienced record-breaking temperatures of 45-48.9 °C (BOM, 2020a; Ossola et al., 2021; Trancoso et al., 2020) and are predicted to reach 50 °C by 2040 (Lewis et al., 2017), this chapter assessed sub-canopy temperatures of 10 commonly planted tree species across urbanised areas in Western Sydney to address the following questions: (1) How do tree species differ in their ability to influence summertime air temperatures and which canopy traits are associated with greater cooling benefits? (2) How do relationships between species/canopy traits and sub-canopy air temperature differ during day and night-time? 3) How do climatic variables influence diurnal patterns in canopy-associated cooling in urban areas? It is hypothesized that (1) tree species with more extensive (e.g., crown width, height and DBH) and dense (e.g., leaf area index [LAI]) crowns, along with specific leaf traits (e.g., high leaf dry matter content [LDMC] and low specific leaf area [SLA]) are associated with higher rates of evapotranspiration and greater shading, and hence, will have lower sub-canopy air temperatures; (2) canopy-associated cooling will occur dominantly during sunlight hours, with limited overnight cooling; and (3) the combined effects of high VPD and high solar irradiance will partly reduce canopy cooling benefits by reducing transpirational cooling via stomata closure, while high wind speed will partly increase cooling benefits of trees via distributing the cool air.

3.2 Materials and methods

3.2.1 Study site

This study was carried out in Richmond and Cranebrook, two suburbs in Western Sydney, New South Wales, Australia (33°52'32.6" S, 151°12'33.1" E). Western Sydney has a humid subtropical climate with an annual mean maximum temperature of 24 °C, mean minimum temperature of 11.1 °C and mean annual rainfall of 741 mm (average between 1990-2020 Bureau of Meteorology, www.bom.gov.au). Rainfall is typically higher in late summer and early autumn months than in other periods (BOM, 2023). The 2019-2020 austral summer in which the study was carried out was Australia's warmest summer on record, with a mean maximum temperature 2.14°C warmer than the 1961–90 average (BOM, 2019a). The number of days per year over 35 °C in Western Sydney has increased from an average of 9.5 days per year in the 1970s to 15.4 in the past decade and is predicted to rise to 19 days per year by 2030 (CSIRO & BOM, 2015). At air temperatures above 35 °C, the ability of human body to cool itself is reduced, making it a common benchmark temperature for occupational health and safety experts, as well as academic and government researchers (Singh et al., 2015).

3.2.2 Data collection

3.2.2.1 Tree species selection

The ten most abundant street tree species in Richmond and Cranebrook were identified using urban tree inventory data sourced from the respective city councils. Ten replicates of each tree species were identified. Temperature loggers were installed at the base of each tree canopy. Vandalism resulted in the loss of one replicate of each of the tree species *Eucalyptus microcorys*, *Melaleuca quinquenervia*, *Sapium sebiferum* and *Ulmus parvifolia).* To standardize urban microclimatic conditions, individual street trees with separate canopies growing in grass verges were randomly selected within low-rise residential areas comprising a mix of impervious and vegetated surfaces (Figure 3.1). Specifically, species were selected with contrasting crown and leaf traits (Table 3.1). All trees were located in residential areas characterized by single-storey buildings (except five individuals of *Platanus acerifolia* that were situated along a street on a non-residential area (Wazeopedia, 2022). Trees were located within a 4 km radius of the respective weather stations (see details below).

3.2.2.2 Air temperature under tree crown data collection

Microclimatic data were collected from the 9th of December 2019 to the 28th of February 2020, during the austral summer. Sub-canopy temperatures were measured using automated, water-proof temperature loggers (Tempmate®-S1 V2, Imec Messtechnik, Heilbronn, Germany) installed at the base of tree canopies, at a height of 3 - 4 m aboveground. Loggers were enclosed in a custom-made shield to avoid exposure to direct sunlight (Figure 3.1 (c)). Holes were drilled into the top of the shield to allow ventilation of the logger. Loggers were programmed to record air temperature at 10-minute intervals for 110 days, with an accuracy of ±0.5 °C (-20 °C / +40°C), ± 1.0 °C outside of that range of temperature and a resolution of 0.1°C. The accuracy of the custom-built device and loggers were evaluated by comparison with data from the nearest weather stations (Wujeska-Klause et al., 2020).

Figure 3. 1 (a) Example of the urban morphology of the studied suburb, Cranebrook, (b) trees (*Eucalyptus microcorys* located on Arafura Avenue, Cranebrook), and (c) sensor and shielding affixed to a tree. The street map of Cranebrook shows that trees are growing on nature strips (verges) and at varying distances from houses. Images of (a) and (b) were sourced from google maps.

3.2.2.3 Climate data collection

Ambient climate data were sourced from the closest BOM weather stations, funding constraints precluded the deployment and installing project-specific weather stations in exposed conditions across the study area, as well as concern for the risk of vandalism for sensors that were visible to passers-by (i.e., not shielded by the canopies). Whilst it is recognised that this imposes some constraints in interpretation of canopy cooling benefits, it has been considered that the relatively small study area and availability of high temporal resolution data from two nearby BOM stations made this a viable alternative. Furthermore, the study of Pfautsch et al., (2023) showed a correlation of 0.98 between data from the BOM station at Cranebrook (Penrith Lake) and the nearby co-located weather station data for Richmond NSW. Therefore, ambient air temperature, humidity and wind speed data at 1-minute resolution were sourced from the meteorological stations at Richmond Royal Australian Air Force (RAAF) (station id: 067105, latitude -33.60, longitude 150.78 and height 19.0 m above sea level [asl]) and Penrith Lakes for Cranebrook (station id: 067113, latitude -33.72° S, longitude 150.68° E and height 24.7 m asl). Solar radiation (~ 1-minute resolution) data were sourced from a weather station at the Hawkesbury Campus of Western Sydney University at Richmond (latitude -33.62° S, longitude 150.75° E and height 20 m asl). Solar radiation was measured using a LI-200 pyranometer (wavelength 400 to 1100 nm range, sensitivity 75 μA per 1,000 W m^{-2}). All data matched the interval when sub-crown air temperatures were recorded. High R^2 value were yielded from the correlations between data from three BOM stations (EucFace, RAAF and Cranebrook) (Figure 3.2). The method of Snyder and Shaw (1984) was followed to calculate VPD from air temperature. At first saturated vapour pressure (SVP) given temperature and humidity were calculated using equation 1. Vapor pressure difference (VPD) is the difference between SVP and actual vapor pressure and was calculated with equation 2.

$$\text{SVP (Pascals)} = 610.7 * 10^{7.5T/(237.3+T)} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ (1)$$

$$\text{VPD} = (1 - (RH/100)) * SVP) \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ (2)$$

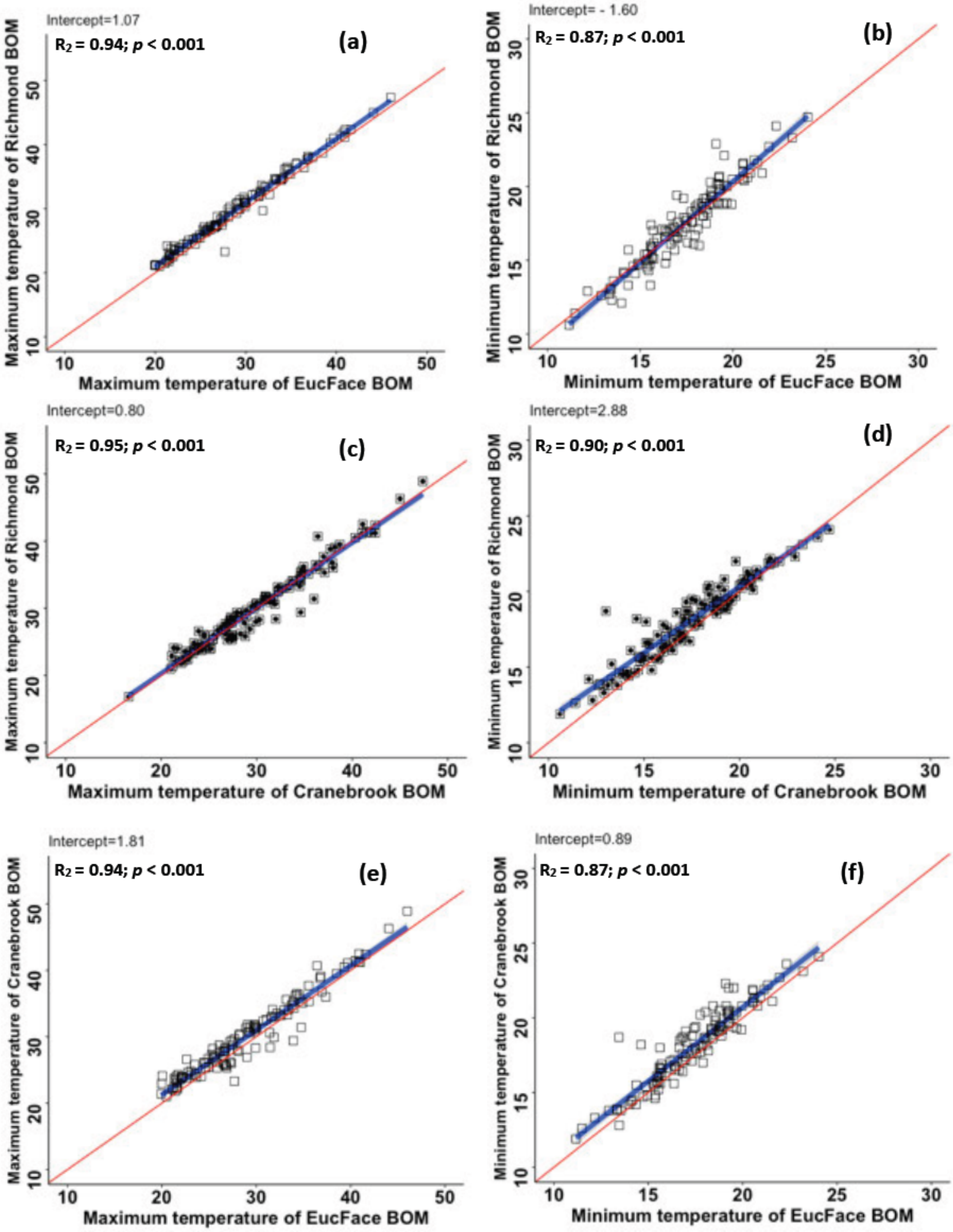


Figure 3. 2 Correlation between air temperatures recorded by three weather stations (EucFace, RAAF and Cranebrook); (a & b) maximum and minimum temperature of Richmond BOM vs maximum and minimum temperature of EucFace BOM; (c & d) maximum and minimum temperature of Richmond BOM vs maximum and minimum temperature of Cranebrook BOM; (e & f) maximum and minimum temperature of Cranebrook BOM vs maximum and minimum temperature of EucFace BOM.

3.2.2.4 Morphological trait measurements

Height, diameter at breast height (DBH; 1.3 m above ground), crown width and leaf area index (LAI) were measured for 96 individual trees. Tree height and crown depth were measured using a Haglöf laser meter (L400, Haglöf, Sweden) with a height resolution of 0.1 m, angle resolution of 0.1□□and accuracy of 0.1□. Crown width measurements were taken as the length of *x* and *y* orthogonal axes from edge to edge for the widest and narrowest points through the crown centre and then averaged. DBH was measured using a diameter tape at a height of 1.3 m.

Leaf area index (LAI) was measured in January and March 2020 for each individual tree using a plant canopy analyser (LAI-2200, Li-COR, USA). Four LAI measurements were taken at 1 m height from the ground and with a 90° view cap on a fish-eye lens. LAI is the ratio of the area of leaves to the area of the ground under the crown (Breda, 2003) and was measured on overcast days to improve contrast. LAI data were analysed using FV2200 software developed for the LAI-2200, deploying an isolated crown model.

One small branch (diameter range of 10 to 15 mm and length of 30 cm) was collected from six to eight replicates of each species from the north side of the mid-canopy of each tree to measure specific leaf area (SLA), leaf dry matter content (LDMC) and determine the Huber value (HV; the ratio of the sapwood cross-sectional area to the total supported leaf area) (Vinya et al., 2012). Excised samples were immediately placed in an insulated foam box.

To calculate the HV, fresh leaves of each branch were scanned and then the total projected leaf area of each branch was estimated using WinFOLIA Software (Regent Instrument Inc., Canada). Subsequently, the sapwood diameter with- and without bark, and the length of branches were measured using a digital calliper (ABS Digital calliper, CD-6-inch ASX, Japan) with an accuracy of ± 0.02 mm, resolution of 0.01 mm and a ruler, respectively, to calculate the sapwood area and I assumed that branches did not contain hardwood. To calculate leaf area, SLA and LDMC, three to five fully-expanded leaves per individual tree, excluding the petiole, were chosen. Fresh weight of the leaves was measured, before scanning their area and drying them in an oven for 48 hr at 70 °C after which their dry weight was determined. Specific leaf area (SLA) was calculated by dividing the leaf surface area by its dry mass (Pérez-Harguindeguy et al., 2013). Leaf dry matter content (LDMC) was determined by dividing the dry weight by fresh weight (Pérez-Harguindeguy et al., 2013).

3.2.3 Data analysis

To evaluate diurnal species-specific patterns in sub-canopy air temperature, days were split into morning (07:00 to 13:00), afternoon (15:00 to 18:00) and night-time (00:00 to 05:00) periods. Cloudy or hazy days (solar irradiance below 500 KW m^{-2}) resulting from the extensive bushfires occurring at the time were excluded from the dataset; this resulted in a subset of 37 days (from a total of 110) used for analyses.

To calculate delta temperature (ΔT), ambient air temperature was subtracted from sub-canopy air temperature of the nearest weather station (RAAF or Cranebrook). Negative ΔT values indicate sub-canopy temperatures were lower than ambient air temperature measured by the BOM weather station (i.e., cooling benefit), whereas positive ΔT values indicate warmer sub-canopy temperatures compared to ambient air temperature. Analyses were conducted in two phases, detailed below. All analyses and graphing were carried out using R version 3.5.1 (R Core Team, 2020). A p value of 0.05 was considered as significant for all analysis.

First, to understand the effect of tree species on ΔT, linear mixed effect models (*lmer*) were fitted for each part of the day separately, with tree species, solar irradiance and VPD considered as fixed effects. Dates and suburbs were specified as random effects. A similar model was used for the night-time data, except solar irradiance was excluded. The fitted models were used to evaluate differences among tree species using the function emmeans from the emmeans package (Russell et al., 2022). Letters displayed for pairwise comparisons were extracted using multcomp packages (Bretz et al., 2010).

Secondly, linear mixed effect models (*lmer*) were used to investigate relationships between ΔT, tree traits and climate variables. For analyses using daytime (morning and afternoon) measurements, temperature data, solar irradiance, windspeed, VPD, days since last rain and all tree traits (i.e., height, stem height, crown width, leaf areas, LAI, SLA, LDMC, HV) were specified as fixed effects. Date, suburbs and tree species were specified as random factors. For night-time analyses, the model excluded solar irradiance. The lme4 package was used for *lmer* analyses (Bates et al., 2015).

3.3 Results

Tree species used for this differed significantly in their traits - canopy width, DBH, tree height, Huber value, LDMC, LAI and SLA (Supplementary Figure 3.1, Table 3.1). Within-species differences are also evident from the boxplots of Supplementary Figure 3.1.

Table 3. 1 Average and standard deviation of the traits of the ten tree species selected in Richmond and Cranebrook, Greater Sydney, Australia. Numbers of individual trees are given in parenthesis. The ten species are ordered alphabetically in the table.

Species list	**Species abbreviation (number)**	**Family**	**Evergreen /Deciduous**	**Canopy Width**	**Height (m)**	**DBH (cm)**	**Huber value (cm^2 mm^{-2})**	**LDMC (mg g^{-1})**	**LAI (m^2 m^{-2})**	**SLA (cm^2 g^{-1})**
Callistemon viminalis Sol. ex Gaertn	Cal (10)	Myrtaceae	Evergreen	7.2 ±1	6.6 ±1.1	18.6 ±4.5	0.0052 ±0.002	516.2 ±50.0	2.74 ±0.7	58.4 ±15
Eucalyptus microcorys F. Muell	Euc (9)	Myrtaceae	Deciduous	13.1 ±1.9	13.6 ±1.9	50.1 ±14.9	0.0017 ±0.0005	493.0 ±22.1	2.9 ±0.5	73.2 ±7.6
Jacaranda mimosifolia D.Don	Jac (10)	Bignoniaceae	Deciduous	10.9 ±1.8	9.2 ±2	32.3 ±11.8	0.0016 ±0.001	384.1 ±39.1	3.42 ±0.5	144 ±25
Liquidambar styraciflua L.	Liq (10)	Altingiaceae	Deciduous	13.4 ±4.4	15.1 ±3.1	51.8 ±16	0.0022 ±0.001	389.0 ±0.2	4.05 ±0.7	121 ±17.9
Lophostemon confertus R.Br.	Loph (10)	Myrtaceae	Deciduous	10.3 ±1.5	11.2 ±1.8	44.9 ±13	0.002 ±0.001	437.5 ±80.0	4.23 ±0.8	69.7 ±13.8
Melaleuca quinquenervia (Cav.) S.T.Blake	Mel (9)	Myrtaceae	Evergreen	9.7 ±2	10.5 ±1.8	61 ±35.1	0.0026 ±0.0006	379.7 ±50.1	3.55 ±0.8	61.9 ±13.5
Platanus × acerifolia (Aiton) Willd.	Plat (10)	Platanaceae	Deciduous	16.2 ±3.8	16.4 ±2.5	56.4 ±38.4	0.0016 ±0.001	392.0 ±50.3	3.62 ±1	133 ±40.6
Pyrus calleryana Decne.	Pyr (10)	Rosaceae	Deciduous	7.5 ±1.3	8.1 ±3.4	21.3 ±6.5	0.003 ±0.0008	463.2 ±23.7	4.35 ±1.2	89.3 ±14.1
Sapium sebiferum (L.) Roxb.	Sap (9)	Euphorbiaceae	Deciduous	9.5 ±2	9.2 ±1.3	35.5 ±11.9	0.001 ±0.0003	378.0 ±43.0	3.44 ±0.2	156 ±53.8
Ulmus parvifolia Jacq.	Ulm (9)	Ulmaceae	Deciduous	11.5 ±2.7	10.0 ±1.6	29.9 ±7.4	0.003 ±0.002	408.0 ±39.5	3.92 ±1	90.1 ±8

3.3.1 Daytime and night-time delta temperature

The temperature differences between the sub-canopies of individual trees and species, and ambient air, showed a diurnal pattern (Figure 3.3 (a) & (b)). In summer 2019-2020, the mean maximum cooling benefit (i.e., the coolest sub-crown temperatures relative to ambient air) for individual trees and species were observed in the morning between 09:00 and 10:00 h (local time) with a mean maximum of 3.9 °C and 3.2 °C, respectively. Sub-crown temperatures were typically below ambient from 07:00 to 14:00 hr, with positive ΔT values recorded outside of this time interval (Figure 3.3 (a) & (b)). At 18:00 hr, mean maximum sub-canopy air temperature of individual trees and species were up to 3.8 °C and 2.9 °C warmer than the ambient temperature (Figure 3.3 (a) & (b)). A drop in positive ΔT values from late afternoon peaks to the early part of the night was observed, although night-time values remained positive (ie warmer sub-canopies than ambient air) until around 6-7 am. Mean maximum night-time warming under individual trees ranged from 0.6 °C to 2.45 °C, and 1.08 °C to 1.86 °C across the 10 tree species.

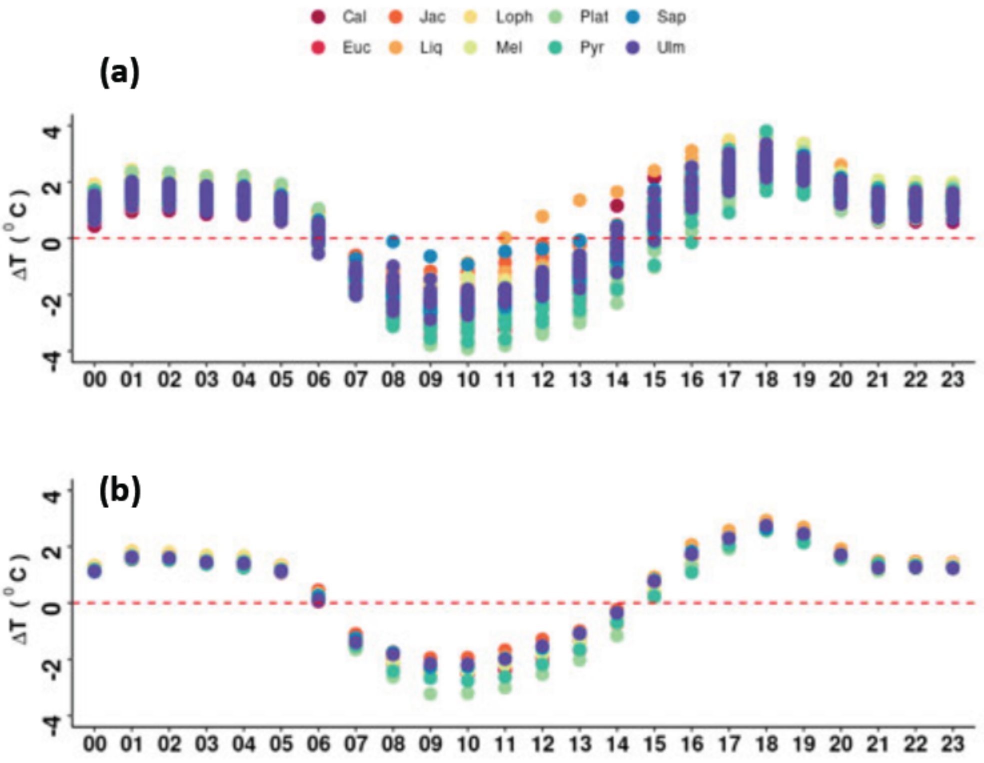


Figure 3. 3 Diurnal variations in ΔT on summer days for individual trees (n = 96) (a) and species average (n = 9-10) (b). The redlines indicate no difference between the canopy and ambient air temperatures. Circle colours correspond to tree species. For abbreviation of species names and number of trees for each species see Table 3.1.

3.3.2 Species differences

Delta T differed among species and across different times of the day (Supplementary Table 3.1). In the morning (07:00 to 13:00 h), *Platanus* × *acerifolia* (mean -3.02 °C) had the most negative ΔT (i.e., cooling benefit), followed by *Pyrus calleryana* (-2.58 °C), and with a maximum mean tree species difference of 1.27 °C (Figure 3.4 (a)). *Jacaranda mimosifolia* had the least negative ΔT in the morning, followed by *Liquiambar styraciflua, Sapium sebiferum and Ulmus parvifolia.*

Sub-canopy air temperatures for all species were warmer than ambient air temperatures in the afternoon (i.e., positive ΔT) and this pattern continued at night-time for some species (e.g., *Eucalyptus microcorys, J. mimosifolia* and *Lophostemon confertus*) (Figure 3.4 (b) &

(c)). Although the rank order of tree species' ΔT differed in the afternoon, compared to morning, *P. acerifolia* and *P. calleryana* had relatively cooler sub-canopies (although still warmer than ambient air temperature), while those under *L. styraciflua* were the warmest.

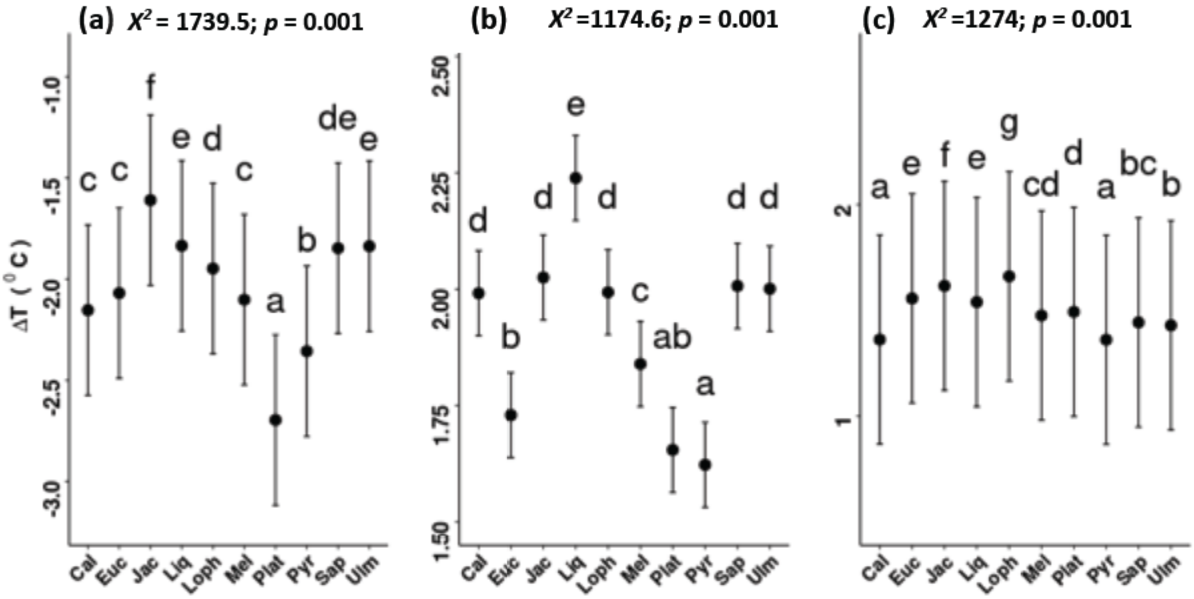


Figure 3. 4 Species differences in ΔT (a) in the morning (07:00 to 13:00), (b) in the afternoon (15:00 to 19:00) and (c) at night (01:00 to 05:00). Different letters indicate significant differences at $p < 0.05$. The black points indicate mean values (±SE). For abbreviation of species names and number of replicates see Table 3.1. Chi-square (X^2) and *p* values for each of the *lmer* models were given. Analysis of variance (ANOVA) results of *lmer* model for each plot see Supplementary Table 3.1.

At night-time (00:00 to 05:00), species differences in ΔT (average of 0.19 °C) were lower than during daytime, although sub-canopy temperatures in *Callistemon viminalis, P. calleryana, S. sebiferum and U. parvifolia* were slightly lower than those of other species (Figure 3.4 (c)).

3.3.3 Relationships between ΔT, climatic variables and tree traits

To evaluate the relationships between ΔT, tree traits and climatic variables, all data points (10 min resolution) for the 37 days, for all individuals (96 trees), were used for analysis. The *lmer* model for the morning (07:00 to 13:00 h) showed that, canopy-associated ΔT was affected by microclimatic variables and tree traits ($R^2 = 0.48$ $p < 0.001$).

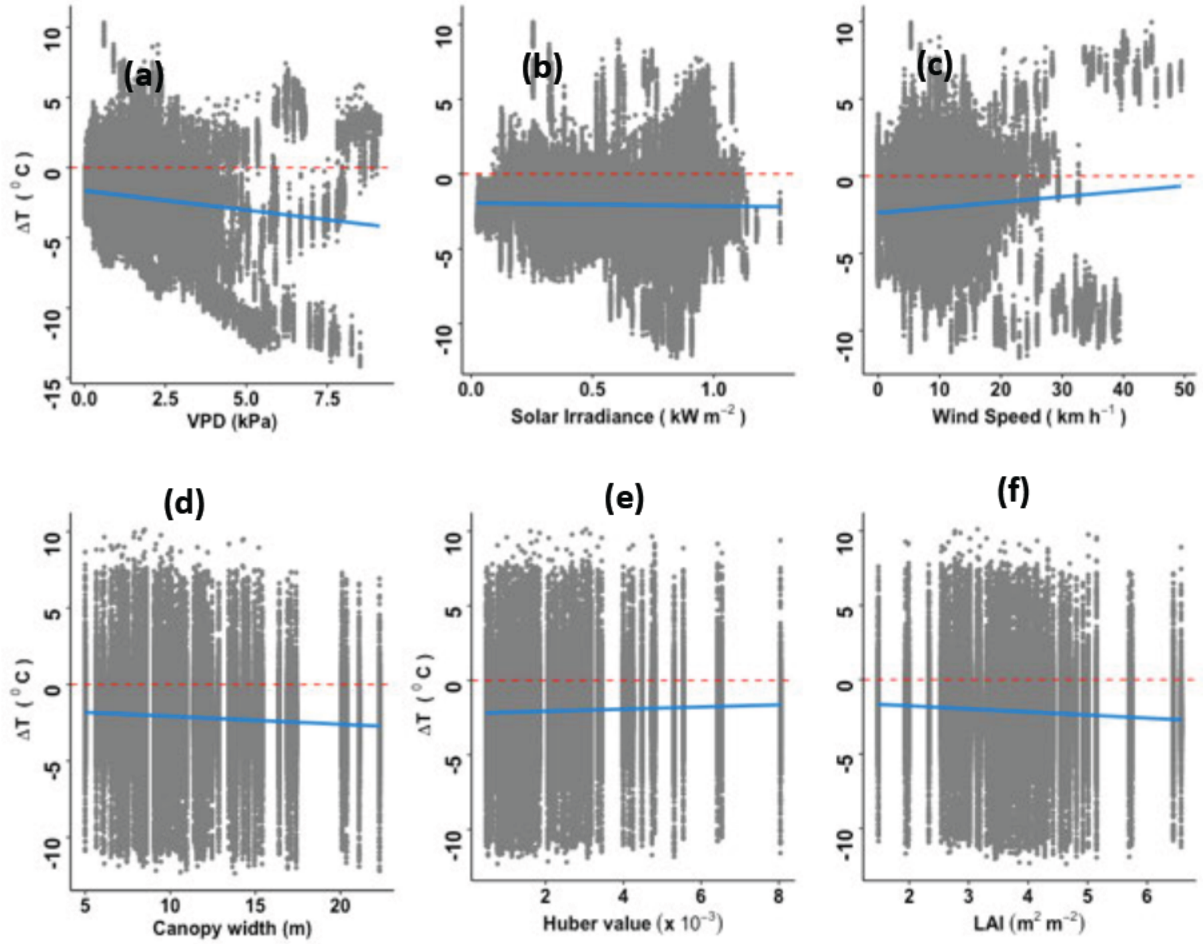


Figure 3. 5 Relationship between ΔT and (a) vapor pressure deficit (VPD), (b) solar irradiance, (c) wind speed, (d) canopy width, (e) Huber value and (f) leaf area index (LAI) in the morning (07:00 to 13:00 h). The redlines indicate no difference between the canopy and ambient air temperatures and blue lines indicate best-fit trend predictions. Total number of observations was 127822. Only significant relationships are shown. For detailed results see Supplemntary Table 3.2.

Delta T was negatively related with LAI and crown width (i.e., sub-canopy temperature was lower when tree crowns were wide and dense), but not with other traits (e.g., tree height, DBH and SLA (Supplementary Table 3.2). ΔT values became more negative (greater cooling benefits) with increasing VPD and windspeed (Figure 3.5). The opposite pattern was found between ΔT and solar irradiance. Among the evaluated climatic parameters and tree traits, VPD (t- value 23.10, $p < 0.001$) and LAI (t-value = 18.78, p value = 0.001) explained the greatest amount of variance in ΔT (Supplementary Table 3.2).

In the afternoon (15:00 to 19:00), ΔT was inversely related to VPD and solar irradiance (i.e., cooler sub-canopy temperatures with higher values of these climate variables) (Supplementary Table 3.3). The relationship between ΔT switched from positive to negative at a VPD value of ~3.5 kPa (Figure 3.6). There was also a significant positive relationship between ΔT and windspeed (i.e., atmospheric mixing). Among the climatic variables, VPD and windspeed explained the highest amounts of data variance. ΔT was significantly and negatively related to canopy width and LAI. Leaf area index (LAI) explained the greatest amount of data variance of all measured traits (Figure 3.6; Supplementary Table 3.3).

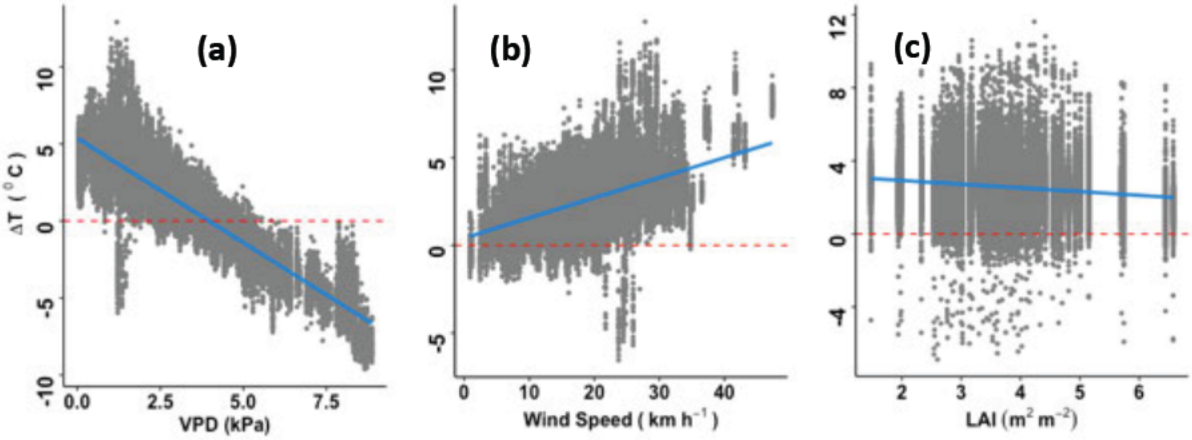


Figure 3.6 Relationship between ΔT and (a) vapor pressure deficit (VPD), (b) wind speed and (c) leaf area index (LAI) in the afternoon. The red lines indicate no difference between the sub canopy and ambient air temperatures and blue lines correspond to regression slopes. The number of observations was 127822. For detailed results see Supplemntary Table 3.3.

In the night-time, warming effects decreased with increasing VPD and windspeed ($R^2 = 0.93$, $p < 0.0001$). ΔT was significantly and negatively correlated to Huber value and canopy width (Figure 3.7). The tree canopy-associated warming effect increased with tree height and higher values of LDMC. Among the climatic parameters, climatic parameters especially VPD and among tree traits LAI explained more variances in ΔT (Figure 3.7; Supplementary Table 3. 4).

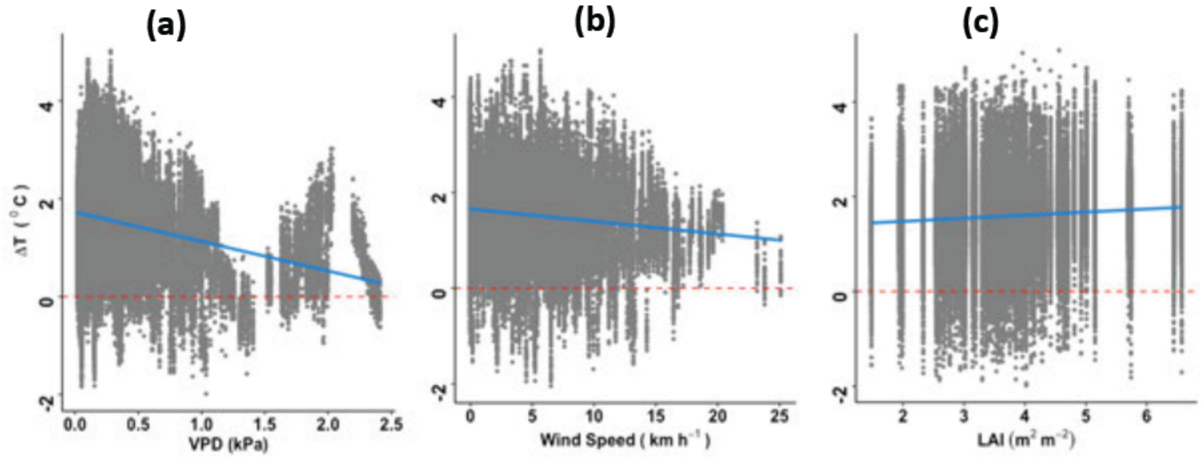


Figure 3.7 Relationship between ΔT and (a) vapor pressure deficit (VPD), (b) wind speed and (c) leaf area index night-time. Only the three strongest relationships with high chi-square (X^2) values are shown. The red lines indicate no difference between the sub canopy and ambient air temperatures and blue lines represent regression slopes. There was a total of 106560 observations. Only significant relationships are shown. For detailed results see Supplementary Table 3.4.

3.4 Discussion

A large dataset of sub-canopy air temperatures allowed this study to verify how common urban tree species in Western Sydney modify daytime and night-time microclimates in summer. Inclusion of local-climate variables along with tree species and trait information provided novel insight into the importance of species identity and the role local microclimatic parameters in the cooling benefits offered by urban trees.

Tree traits are known to influence cooling benefits in urban environments. The mean maximum 3.9 °C of air cooling in the morning recorded in this study is greater than the ~1°C cooling reported by Alonzo et al. (2021) in Washington, DC, USA. It is important to note, however, that tree species differed significantly in terms of their cooling / warming effects in the current study, with average species-level values varying two-fold. Among the 10 species studied, *P. acerifolia* (mean 3.02 °C; mean LAI 3.62 $m^2 m^{-2}$ and mean crown width 16.21 m) provided the greatest cooling benefits, followed by *P. calleryana* (mean 2.58 °C, mean LAI

4.35 m^2 m^{-2} and mean crown width 7.52 m) in the morning. At the other end of the scale, *J. mimosifolia* had the lowest ability to reduce air temperatures during summertime, with an average reduction of 1.5 °C. This species had a low LAI (mean 1.75 m^2 m^{-2}) and had no leaves during its flowering period from October to early December. However, when this species is in full leaf (mean LAI of 3.42 m^2 m^{-2} measured in February 2020), it can potentially provide much greater cooling benefits.

A number of studies have investigated the effect of LAI on sub-canopy air temperatures and showed that daytime cooling benefits improve with increasing LAI values (Kong et al., 2017; Sanusi et al., 2017; Wujeska-Klause et al., 2020). Similarly, wider crowns can potentially reflect a greater proportion of incoming solar radiation and provide a larger shaded area, resulting in greater cooling benefits (Shahidan et al., 2010). Pfautsch et al. (2019b) observed that air temperatures of streets with high canopy cover were, on average, 0.5 °C (max 2.1 °C) cooler than streets with low canopy cover, again highlighting the importance of canopy size.

Trees with large canopies cast shade over large areas, which can lead to a decrease in local air temperature (Rahman et al., 2020). Shaded areas absorb less solar irradiation, resulting in lower energy storage and re-emission in form of sensible heat, which can warm the surrounding air (Barišić et al., 2022; Rahman et al., 2017). Overall, the findings of relationships between cooling benefits and LAI and canopy width provide partial support for the first hypothesis of the current study and highlight the importance of shade cast (both in terms of the absolute amount and intensity of shade) for daytime cooling in urban streetscapes.

The observed interactions between canopy-associated cooling and solar irradiance across the day support observations by Meili et al. (2021); Motazedian et al. (2020); Shashua-Bar et al. (2011). Solar energy drives evapotranspiration in the morning, particularly from 07:00 to 10:00 hours (Granier et al., 1996), which may explain the increase in magnitude of morning cooling benefits with increasing solar irradiance in this study. High solar zenith angles in the morning can lead to large structural shading thereby slowing the warming of surfaces, and thereby increasing the cooling benefits of street trees (Yu et al., 2020).

Canopy-associated warming was observed in the afternoon and continued overnight, with average temperature increases of 1.19 °C (afternoon) and 1.53 °C (night), relative to ambient air. Among the ten tree species in the current study, *Liquidambar styraciflua* had the warmest afternoon sub-canopy temperatures (mean 2.43 °C) and *P. acerifolia* (mean 1.78°C) the lowest. Although positive afternoon ΔT values decreased with increasing LAI, again indicating that shade intensity can influence the local microclimate by reducing thermal loading of surrounding man-made materials, they were still mostly indicating higher sub-canopy temperatures than ambient air. The positive relationship between afternoon sub-canopy air temperatures and windspeed in this study highlights the important role of atmospheric mixing (Alcoforado et al., 2009; Oliveira et al., 2021). The transport of warm air masses from the hot, arid interior of the continent into suburban areas in the afternoon likely negated canopy-associated cooling, especially when combined with VPD and associated stomatal closure. This, combined with re-radiated heat from surrounding buildings likely plays an important role in our observation of afternoon sub-canopy warming. Several studies have reported that trees with high LAI can trap re-radiated heat in the evening and overnight (Taha et al., 1991; Souch & Souch, 1993; Aguiar et al., 2014; Kaluarachichi et al., 2020). For example, a study conducted in Washington DC, USA, found that the cooling benefits of trees located along streets were less than trees on grass, due to re-radiated heat from sub-canopy and surrounding surfaces, highlighting the importance of planting context (Alonzo et al., 2021). The surrounding environment, particularly street canyons and the direction of streets, can impact the ability of trees to provide cooling benefits (Chen et al., 2021). High tree density or closely planted trees might reduce air movement which can potentially trap heat within the canopy (Wujeska-Klause et al., 2020). The effects of canopy density on both trapping re-radiated heat (resulting in warming) and increasing the extent of sub-surface shading (thereby reducing re-radiation of sensible heat and reducing sub-canopy air temperatures) is a topic that warrants further study.

Canopy-associated warming can also be a response to sub-optimal water availability limiting transpiration. This, in turn, reduces latent heat flux and generates large amounts of sensible heat, thereby increasing local air temperature (Carvalho et al., 2015). I speculate that the extreme nature of the weather in the run up to and during the study period played a role in observed sub-canopy warming, particularly during the afternoon. From 2017 to 2019, the study sites experienced very low levels of rainfall, with 2019 having the lowest rainfall on record (BOM, 2020c). Furthermore, during the "black summer" bushfire season of 2019-2020, Richmond and Cranebrook experienced 10 days above 40 °C, representing a period of extreme heat (Marchin et al., 2022). During this period, Tabassum et al. (2021) found widespread evidence of tree canopy damage in Penrith, Western Sydney, while Marchin et al (2022) reported that around 60% of plant species they studied in Western Sydney experienced dieback due to the extremely hot and dry conditions prevailing during this time. While high temperatures and VPD during heatwaves are known to result in stomatal closure (Kaluarachichi et al., 2020), the lack of physiological measurements in our study does not allow us to conclude that stomatal closure and the associated reduction in latent heat loss was responsible for sub-canopy warming during afternoon periods – a potential "disservice" of urban trees. The higher afternoon sub-canopy temperatures can be interpreted as evidence of both the role of atmospheric mixing, discussed above, and the urban heat island effect, with high levels of afternoon re-radiated heat from surrounding built environment captured in our sub-canopy dataloggers.

In terms of the response of trees to water stress, tree species can differ widely in their stomatal (and thus transpirational) strategies, exhibiting both isohydric (maintaining leaf water potential by reducing midday stomatal conductance) and anisohydric (maintaining stomatal opening despite water limitation) responses (Klein & Niu, 2014). Anisohydric trees keep their stomata open for a longer time than isohydric species under sub-optimal water conditions and therefore continue transpiring water (Tardieu & Simonneau, 1998). There is currently insufficient information on species-level stomatal strategies or plant strategies to water deficit for the trees included in this study. Marchin et al., (2022) assessed plant physiological strategies for coping with heat and drought stress in 20 Australian native species (with none overlapping with the present study). They observed that stomata close before the hydraulic threshold is reached but that isohydric species can maintain high levels of stomatal conductance even after the turgor loss point. This implies that some species can still provide transpirational cooling under conditions of water scarcity, although this strategy may lead to hydraulic failure and, ultimately, death (Marchin et al., 2022; McDowell et al., 2022). There is a substantial need to improve our understanding of the trade-off between water conservation and plant transpirational cooling, both trees growing in natural environment and streets in particular.

Vapor pressure deficit (VPD) was posited to have an indirect, yet important influence on sub-canopy temperatures and ΔT across the day (Urban et al., 2017). Indeed, evapotranspiration can decrease air temperatures by 2.0 to 8.0 °C in vegetated areas (Taha, 1997). VPD typically increases rapidly in the morning and reaches a maximum rate by early afternoon, then declines toward dusk (O'grady et al., 1999). The effect of stomatal closure in the late morning leads to reduction in cooling (ΔT moving towards zero) before VPD reaches its highest levels. In this study, a warming effect was observed at 13:00, where ΔT became less negative while VPD still increased until 14:00 or 15:00 hours. In this study, VPD had consistent effects on morning and afternoon cooling (> 3.5 KPa), which is consistent with the hypothesis of positive impacts of VPD on the contribution of transpiration to canopy cooling.

Despite the common expectation that night-time stomatal closure results in minimal transpiration after dark, studies have shown that transpiration can occur after sunset in plants from a wide range of climates (Dawson et al., 2007; Konarska et al., 2016). Lindén et al. (2016) and Ibsen et al. (2021) observed a positive correlation between VPD and transpiration-driven, canopy-associated cooling at night. The findings of a negative correlation between night-time VPD and ΔT in the current study also points to the possibility of night-time transpiration, although, ΔT values were generally positive during the night-time. It can be speculated that, in conditions of low ambient rainfall – such as was the case during the months preceding this study - and thus low soil water availability, trees need to conserve water, thus, transpiration is unlikely to play a major role in canopy-associated cooling. In Sydney, street trees do not typically receive additional irrigation beyond the initial ~ 2 year establishment phase, even during prolonged dry spells (Esperon-Rodriguez et al., 2022). A more thorough investigation of whether VPD and water availability regulated night-time transpiration by street trees in a manner that is similar to daytime is, therefore, warranted.

Night-time canopy-associated warming of up to 0.4 °C has been reported by (Alonzo et al., 2021) in Washington DC, and also by (Wujeska-Klause et al., 2020) in a study in Parramatta, NSW, Australia, around 40 km far from the current study area. Taken together, night-time warming (and in our study also during summer afternoons) associated with street trees in heavily built-up areas suggests that canopies can hamper the transfer of re-radiated heat from built infrastructure into the atmosphere overnight. The percentage of canopy- and built surface cover are, however, known to affect the influence of vegetation on temperature in the built environment (Alonzo et al., 2021; Oke et al., 2017; Ziter et al., 2019), and the relative amount of built versus natural surface cover is clearly a key factor influencing urban heat dynamics.

Reducing the area of heat-absorbing dark and unshaded surfaces is critical for reducing night-time air temperatures during summer, given the amount of heat stored and subsequently re-radiated overnight (Memon et al., 2010). Indeed, the lack of difference in night-time temperature between streets with low and high canopy cover reported by Pfautsch et al. (2019b) strongly suggests that reducing the area of grey surface cover is the key to reducing night-time air temperatures in urban settings. The current study indicates that, in Western Sydney, trees play an important role in managing daytime temperatures and mitigating urban heat (particularly in the morning), but provide few, if any temperature benefits over night during the hot summer months. These results highlight the importance of developing a deeper understanding of the interactions between surface characteristics, local microclimate and tree cover for reducing urban warming and improving the liveability of cities during the night-time.

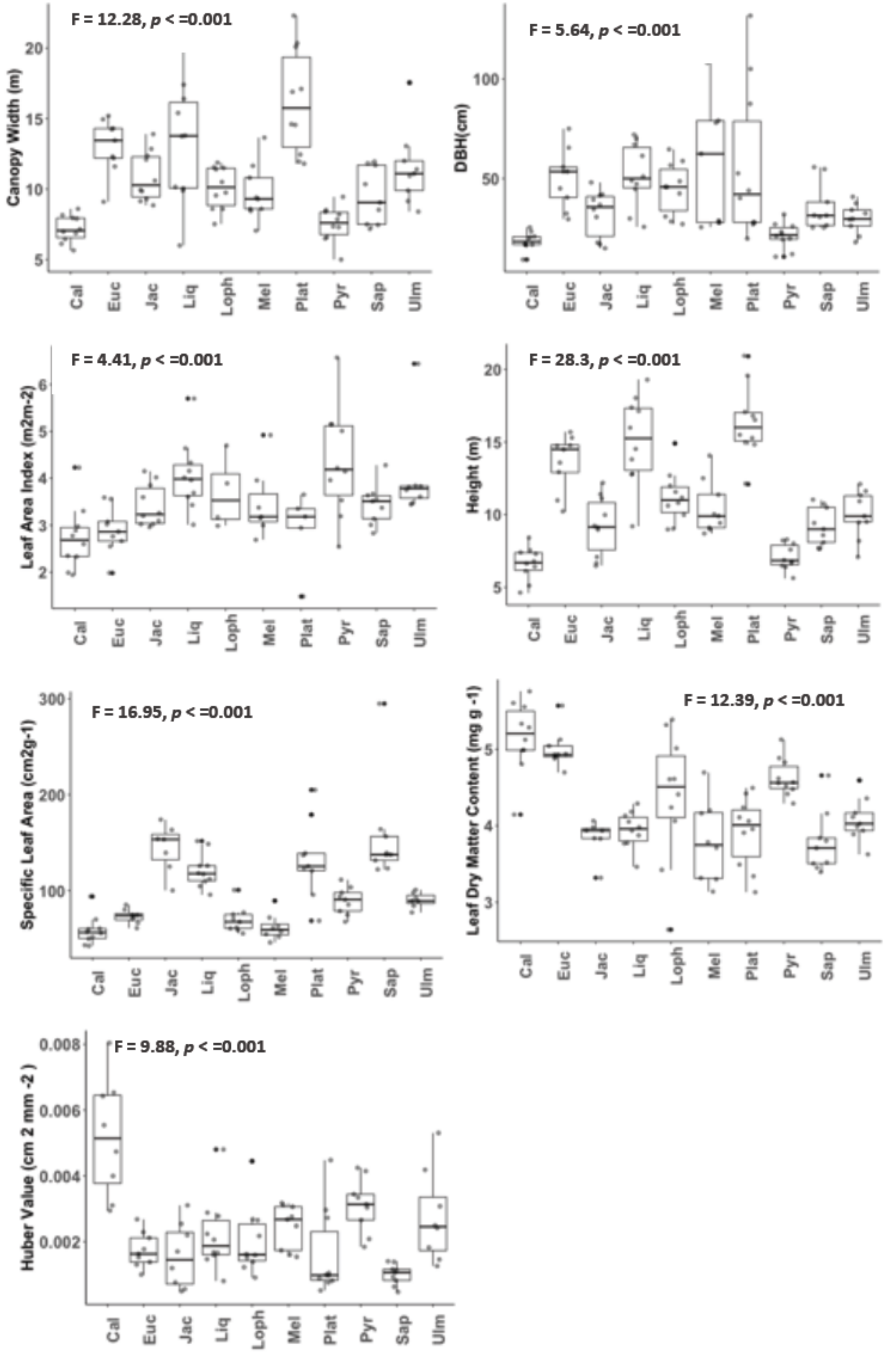
3.4.1 Study limitations

It is worth noting that while the selected 10 tree species varied in terms of their traits, they represented only a small proportion of the species planted and growing across Sydney. For urban practitioners, it is crucial to identify and plant tree species with traits that can help mitigate the adverse effects of urban warming; this study is useful for this purpose although caution should be taken in applying the results of this single site study across different areas. While the use of five to seven replicates to measure traits such as LDMC and SLA might not capture the full range of intra-species trait plasticity, it did allow us to broadly characterise species-specific trait differences.

3.4.2 Conclusion and Implications

This study focused on identifying the canopy and leaf traits associated with cooling urban air temperature and the influence of climatic variables on diurnal patterns in canopy-associated cooling. Trees with dense (high LAI) and wide canopies provided the greatest cooling benefits. Of the 10 species assessed here, *P. acerifolia* provided the highest cooling benefits in streetscapes across the studied suburbs. Species with similar traits may offer the best opportunity to reduce urban air temperatures, and this can potentially be factored into the design of future planting strategies. Studying the influence of climatic variables on diurnal patterns in canopy-associated cooling provides a deeper understanding of the potential for street trees to contribute to urban heat mitigation under different climate conditions. Trees provided the greatest level of summertime cooling in the morning, but with increasing solar irradiance and windspeed and a shift in wind direction in the afternoon, tree canopies were associated with sub-canopy warming in the afternoon and at night. Collectively, these findings reinforce the current perception that planting trees with wide, dense canopies in cities across the globe can help mitigate urban heat, but that the extent of associated cooling benefits depends on climatic conditions. Active irrigation and passive irrigation by reducing rainwater runoff in the city will help to increase soil moisture which keep the stomata open and allow trees to maintain high rates of transpiration. Other nature-based solutions (e.g., blue space, green walls and roofs) also need to be considered as part of a strategy to increasing evaporative cooling across the built environment, to provide relief from afternoon and night-time urban heat.

Supplementary Figure



Supplementary Figure 3.1 Within and among species differences in tree traits; (a) canopy width, (b) DBH, (c) leaf area index, (d) height, (e) specific leaf area, (f) leaf dry matter content and (g) Huber value. ANOVA results of the significant differences among species for respective traits are given in each plot.

Supplementary Table

Supplementary Table 3. 1 Analysis of variance (ANOVA) results of LMER models to test whether delta temperature (ΔT) varies among tree species in the morning, afternoon and night-time. Tree species, solar irradiance and VPD (vapor pressure deficient) were considered as fixed effects, and tree identity as a random effect.

Morning	**Sum Sq**	**Mean Sq**	**NumDF**	**df**	**F value**	***p***
Tree species	10346.4	1149.6	9.0	89022.9	210.8	< 0.001
Date	512687.7	13491.8	38.0	88662.2	2474.4	< 0.001
Afternoon						
Tree species	2852.6	317.0	9.0	90241.9	222.5	< 0.001
Date	134121.0	3529.5	38.0	90242.4	2477.3	< 0.001
Night						
Tree species	845.5	93.9	9.0	108007.3	138.4	< 0.001
Date	780279.4	20533.7	38.0	108007.3	30259.8	< 0.001

Supplementary Table 3. 2 Results of LMER models assessing the relationship between delta temperature (ΔT) in the morning and tree traits (leaf area index [LAI], tree height, crown width, specific leaf area [SLA], Huber value and leaf dry matter content [LMDC] and microclimatic variables (solar irradiance, VPD and wind speed).

	Estimate	**Std. Error**	**df**	**t value**	***p***
Fixed Effects					
(Intercept)	-0.360	0.456	51.71	-0.79	NS
Solar irradiance	-0.193	0.045	104400	-4.30	0.001
VPD	-0.276	0.012	103900	-23.10	0.001
Wind speed	0.035	0.002	94580	14.33	0.001
Canopy width	-0.052	0.005	95	-11.31	0.001
LDMC	-0.046	0.022	95	-2.05	0.04
LAI	-0.207	0.011	95	-18.78	0.001
Height	-0.009	0.005	95	-1.61	NS
SLA	-0.292	0.358	95	-0.82	NS
Huber value	0.072	0.009	95	7.90	0.001
Random Effects		**Variance**		**Std.Dev.**	
Tree Species		0.12		0.34	
Date		6.23		2.50	
Suburb		0.02		0.16	
Residual		5.20		2.28	
Rc^2 (Rm^2)		0.48 (0.12)			

Supplementary Table 3. 3 Results of LMER models assessing the relationship between delta temperature (ΔT) at afternoon and tree traits (leaf area index [LAI], tree height, crown width, specific leaf area [SLA], Huber value and leaf dry matter content [LMDC] and microclimatic variables (solar irradiance, VPD and wind speed).

	Estimate	**Std. Error**	**df**	**t value**	**Pr(>|t|)**
Fixed Effects					
(Intercept)	3.915	0.433	25.77	9.034	0.001
Solar irradiance	-0.283	0.029	75440	-9.612	0.001
VPD	-1.356	0.007	75430	-184.987	0.001
Wind speed	0.115	0.001	75390	119.621	0.001
Canopy width	-0.021	0.003	95	-8.159	0.001
LDMC	0.016	0.013	95	1.271	NS
LAI	-0.203	0.006	95	-33.054	0.001
Height	0.007	0.003	95	2.379	0.02
SLA	1.043	0.201	95	5.199	0.001
Huber value	0.014	0.005	95	2.647	0.01
Random Effects	**Variance**	**Std.Dev.**			
Tree Species	0.03	0.18			
Date	5.05	2.25			
Suburbs	0.12	0.35			
Residual	1.37	1.17			
Rc^2 (Rm^2)	0.92 (0.49)				

Supplementary Table 3. 4 Results of LMER models assessing the relationship between night-time ΔT and tree traits (leaf area index [LAI], tree height, crown width, specific leaf area [SLA], Huber value and leaf dry matter content [LMDC] and microclimatic variables (solar irradiance, VPD and wind speed).

	Estimate	**Std. Error**	**df**	**t value**	***p* - value**
Fixed Effects					
(Intercept)	1.214	0.493	39.14	2.464	0.02
VPD	-0.601	0.016	75370	-37.7	0.001
Wind speed	-0.026	0.001	75350	-23.74	0.001
Canopy width	-0.008	0.002	95	-4.48	0.001
LDMC	0.067	0.008	95	7.95	0.001
LAI	0.065	0.004	95	15.65	0.001
Height	0.015	0.002	95	7.431	0.001
SLA	0.502	0.135	95	3.710	0.001
Huber value	-0.037	0.003	95	-10.837	0.001
Random Effects		**Variance**	**Std.Dev.**		
Tree Species		0.01	0.10		
Date		8.27	2.87		
Suburbs		0.04	0.22		
Residual		0.61	0.78		
*R*c^{2} (*R*m^{2})		0.93 (0.03)			

CHAPTER FOUR: Shrubs and complex habitat structure help to boost invertebrate biodiversity in urban greenspaces

Abstract

In urban areas, trees are often planted individually in street verges and parks. Despite their important role in urban ecosystem functioning, isolated trees offer limited foraging, hibernation and nesting habitats for invertebrates, relative to the multi-layered, more structurally complex vegetation that is found in natural environments. One option for boosting biodiversity in urban areas is integrating shrubs alongside trees within the urban landscape. This would work to increase the structural complexity of vegetation and broaden the range of habitat available to support urban fauna. This chapter reports on a study investigating how the abundance and taxonomic richness of invertebrates (1) varies between individual shrubs and trees, and (2) differs across different types of vegetation structures, i.e., trees and shrubs planted alone or in combination. A common garden experiment, comprising systematic plantings of shrubs and trees was used to evaluate how canopy-associated invertebrate assemblages are influenced by vegetation type and structure. Canopy dwelling invertebrates were sampled using the branch beating method from November-2019 to January-2021. The overall abundance of invertebrates and that of specific functional group (e.g., herbivores, pollinators, detritivores) was greater on individual shrubs compared to individual trees. "Shrub only' and 'tree plus shrub" plantings also harbour more abundant and taxonomically rich invertebrate communities than 'tree only' plantings. Further analysis of the data suggests that planting shrub and tree species with specific attributes, such as wide and dense canopies and abundant flowers, can increase the associated taxonomic- and functional group diversity of invertebrate communities. Overall, the findings suggest that urban planning of greenspace should focus on incorporating shrubs alongside urban trees to maximise invertebrate diversity in urban landscapes.

4.1 Introduction

The previous chapters have highlighted the importance of trees for urban cooling. However, vegetation in cities also deliver other, equally important, functions and ecosystem services that can improve the quality of life in cities (Gómez-Baggethun et al., 2013). A wide range of ecosystem services rely on biodiversity, for example, invertebrate diversity in agroecosystems can enhance food production, pest control, and nutrient turnover (Baldock et al., 2015; Mata et al., 2021; Threlfall et al., 2015). Decomposition and nutrient turnover are vital ecosystem functions provided by invertebrates such as springtails and beetles (Marschalek & Deutschman, 2022; Ossola et al., 2016), while arthropods such as bees (Ahrne et al., 2009; Theodorou et al., 2020c) and many other flower-visiting insects play a key ecological role by providing pollination services (Klein et al., 2007; Rader et al., 2016). Furthermore, parasitoid insects constitute a highly diverse group of natural enemies in agricultural and urban habitats (Burks & Philpott, 2017; Corcos et al., 2019), contributing to pest control by regulating population sizes of insect herbivores. To thrive, these invertebrate functional groups need access to a wide range of resources, such as food, shelter and reproduction sites. However, land transformation has reduced the diversity and extent of available habitat, leading to associated declines in the diversity and abundance of many plant and animal species in urban areas (Seto et al., 2012). It is therefore a priority to develop strategies that incorporate biodiversity into urban areas to promote and preserve healthy and functional urban environments (Ives et al., 2016; Soanes & Lentini, 2019).

In most cities, 50 to 60 % of greenspace is dominated by standalone trees, planted individually with distinct, separate canopies which are traditionally maintained to an aesthetic standard that often offers relatively low habitat complexity (Aronson et al., 2017; Le Roux et al., 2014a). Habitat with complex vegetation structure (e.g., multi-layered canopies such as is provided by trees growing alongside shrubs) (McCoy et al., 1991) can potentially harbour high invertebrate species richness (Galle et al., 2017; Lassau et al., 2005). Remnant forests, golf courses, parks and gardens can provide structurally complex vegetation of high volume and species richness, and have been associated with high invertebrate biodiversity, although they typically occupy only a small proportion of urban green area (Threlfall et al., 2016). In addition, a flowering under- or mid-storey vegetation can increase the availability of nectar and pollen resources for a large range of invertebrate species, including pollinators (McCall & Irwin, 2006; Mody et al., 2020; Simpson & Neff, 1981; Wardhaugh et al., 2012). Mid-story vegetation such as shrubs act as a stepping-stone to connect tall trees with ground level vegetation and soil, facilitating movement of animals across the urban landscape (Lepczyk et al., 2017). Furthermore, shrubs can improve nutrient cycling by providing abundant leaf litter (Gómez-Aparicio et al., 2005).

In general, the vast majority of the urban biodiversity studies have focused on only a small sub-set of invertebrates (e.g., pollinators, ants, ground dwelling beetles) (Braschler et al., 2020; Matteson & Langellotto, 2010; Ossola et al., 2015; Uhey et al., 2020) despite there being many functional groups and diverse invertebrate taxa that play important roles in ecosystem functioning. Furthermore, given the important roles that climate and phenology play in determining both plant growth and invertebrate activity there is a need to evaluate biodiversity across multiple seasons, to capture the effects of changing temperature and life history stages (Grimbacher et al., 2018). The low number of studies covering multiple taxa and multiple seasons represents a knowledge gap and highlights the need for a more detailed understanding of tree/shrub-invertebrate interactions and dynamics to inform future planning strategies aimed at effectively promoting biodiversity in urban areas.

In this study a common garden experiment comprising systematic plantings of shrubs and trees, was used to address the following questions: (1) How do the abundance and richness of invertebrate communities vary between individual shrubs and trees? (2) How do the abundance and richness of invertebrates differ among different vegetation structural types, i.e., tree-only, shrub-only and shrub plus tree treatments. It is hypothesised that: (1) shrubs will support a greater abundance and richness of invertebrates than trees whether they are growing on their own or alongside trees in the same plot; and (2) plots with more structurally complex vegetation (i.e., 'tree plus shrubs' treatment) have a greater volume of vegetation to support higher abundance and richness of invertebrates compared to less complex plantings (i.e., 'tree only' or 'shrub only' treatments).

4.2 Materials and Methods

4.2.1 Study site

In order to understand how complexity of urban vegetation can positively influence insect abundance and diversity, we established a model planting comprising trees and shrubs, growing on their own or together, in the peri-urban setting of the Hawkesbury campus of Western Sydney University, in Richmond, Australia (latitude 37° 49' 22.8072" S and longitude 144° 59' 52.8036" E). The study was conducted during summer and spring across three years, namely summer 2019/20, spring 2020 and summer 2020/21; these periods were chosen as being times of high invertebrate activity, when access to the field site was permitted under Covid-19 travel restrictions. The weather conditions at the study site during survey periods are summarized in Table 4.1.

Table 4. 1 Climate at the study site, and weather conditions during survey periods (2019-2021) .

	1960-90 Average[1]	Summer 2019/20 [2]	Spring 2020 [3]	Summer 2020/21[4]
Mean maximum temperature (°C)	23.7	31.4	25.2	27.9
Mean minimum temperature (°C)	17.5	18.2	15.9	17.0
Total rainfall (mm)	906.3	320	174.6	268

[1](BOM, 2021a); [2] (BOM, 2020b); [3](BOM, 2022); [4]((BOM, 2021b)

4.2.2 Data collection

4.2.2.1 Study design

Four tree species (eight individuals per species) and four shrub species (32 individuals per species) were used for the experiment (Figure 4.1; Table 4.2). The experimental site covers an area of ~3900 m^2 and is composed of 48 plots, each 4 m^2. The dominance of the Myrtaceae family (one tree species and three shrub species) in the experiment setup reflects its dominance in urban landscapes in Sydney.

Trees (in 45 litre bags) and shrubs (in 140 ml pots) were sourced from the local plant nursery; all tree stock conformed to the Australian Standard AS2303 (Standards Australia AS 2303, S.A., 2018). Trees and shrubs were planted in the ground in the end of October 2018. Individual trees were planted into 60 cm deep and wide holes with the addition of a slow-release fertiliser mixed into the soil around the root ball (NPK 21.8:0.7:7.2, Osmocote Fertiliser; Scotts Australia). Each plot has an area of 2.25 m^2 and is in a grass matrix, resembling nature strip of park settings. Woodchip mulch was applied, to a depth of 100 mm, around each tree and shrub after planting. Although all individuals of a species were of the same age, differences in tree height were recorded at the time of planting (average heights: *Lophostemon confertus* 1.87 m ± 0.15; *Elaeocarpus reticulatus*, 1.79 m ± 0.49; *Lagerstroemia indica* 1.62 m ± 0.94 and *Liriodendron tulipifera* 1.03 m ± 0.11; n = 8 individuals of each species). Shrub species also showed differences in height at the time of planting (average heights: *Baeckea virgata* 39.53 cm ± 8.39; *Melaleuca citrina* 41.06 ± 8.06; *Melaleuca* *thymifolia* 26.84 ± 5.67 *and Westringia fruticose* 13.75 ± 3.75; n = 8 individuals of each species).

All plots were watered individually using 2 × 8 L h^{-1} drippers. During the first four weeks of establishments, plants were irrigated every two days. Beyond this period, plants were irrigated every seven days until October 2019, when this was increased to every four days until the end of April 2020 due to the dry conditions in the study area during summer 2019− 2020. Irrigation then returned to a weekly schedule at the end of summer 2020. On irrigation days, water was supplied over a 40- minute period (ca. 10.7 L day^{-1}).

The experiment comprised three levels of vegetation structure, namely 'tree only' (1 individual of 1 species), 'shrub only '(4 individuals, 1 each of 4 species) and 'trees plus shrubs' (5 individuals - 1 tree species, 4 shrub species). The 'tree only' treatment comprised a single individual of each tree species (each replicated four times). Similarly, the 'shrub only' treatment was composed of four individuals, one of each of the four species, replicated four times (Figure 4.1). The 'tree plus shrub' treatment was composed of four individual shrubs - one of each species - and one individual tree, repeated for each of the four tree species (i.e., 1 tree + 4 shrubs in a plot); each of these was replicated four times. Throughout the experiment period plant canopies remained distinct from each other.

4.2.2.2 Invertebrate sampling

Invertebrate samples were collected from individual plants using the branch-beating method, which targets less-mobile taxa. Samples were collected on two white trays of 30 cm length and 45 cm width. Trays were placed under the plants and branches were shaken for 10 seconds, by two people, on opposite sides of the plant. Trays had 150 ml of water containing a drop of detergent in them, to prevent invertebrates from escaping. Individuals collected from both trays were combined and transferred to 70% ethanol-filled containers for storage and subsequent identification. Sampling was conducted only under optimal weather conditions (i.e., days with clear sky, 1–1.4 m s^{-1} wind speed and warm temperatures, ≥ 24 °C) when insects were likely to be more are active. Invertebrate sample collection was performed on 2-3 consecutive days, from 09:00 to 15:00, for each sampling round (Table 4.1). To minimise the effect of surrounding vegetation, spaces between the sampling plots were mown one week before every sampling round (Figure 4.2).

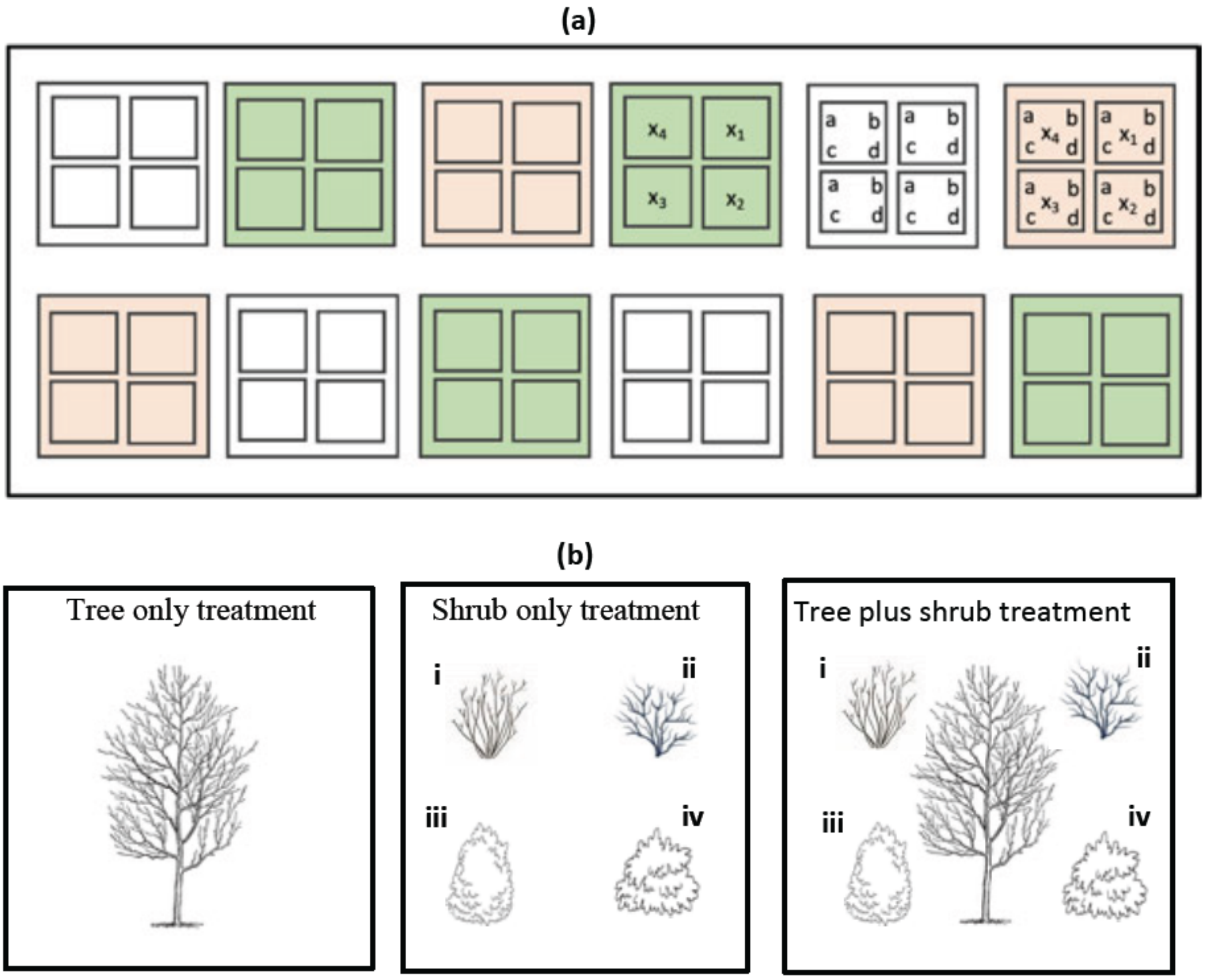


Figure 4.1 Study design of this research. The site consists with (a) 12 blocks of three vegetation structure treatments - 'tree only'; green colour, 'shrub only'; white colour and 'tree plus shrub'; orange colour. Each block has four plots of 4 m^2 area. (b) plots of 'tree only' treatment has one of the four tree species (x_1, x_2, x_3, x_4), 'shrub only' has all four shrub species (a, b, c, d) and 'tree plus shrub' has one tree species and all four shrub species. Species names and phenology are given in Table 4.2.

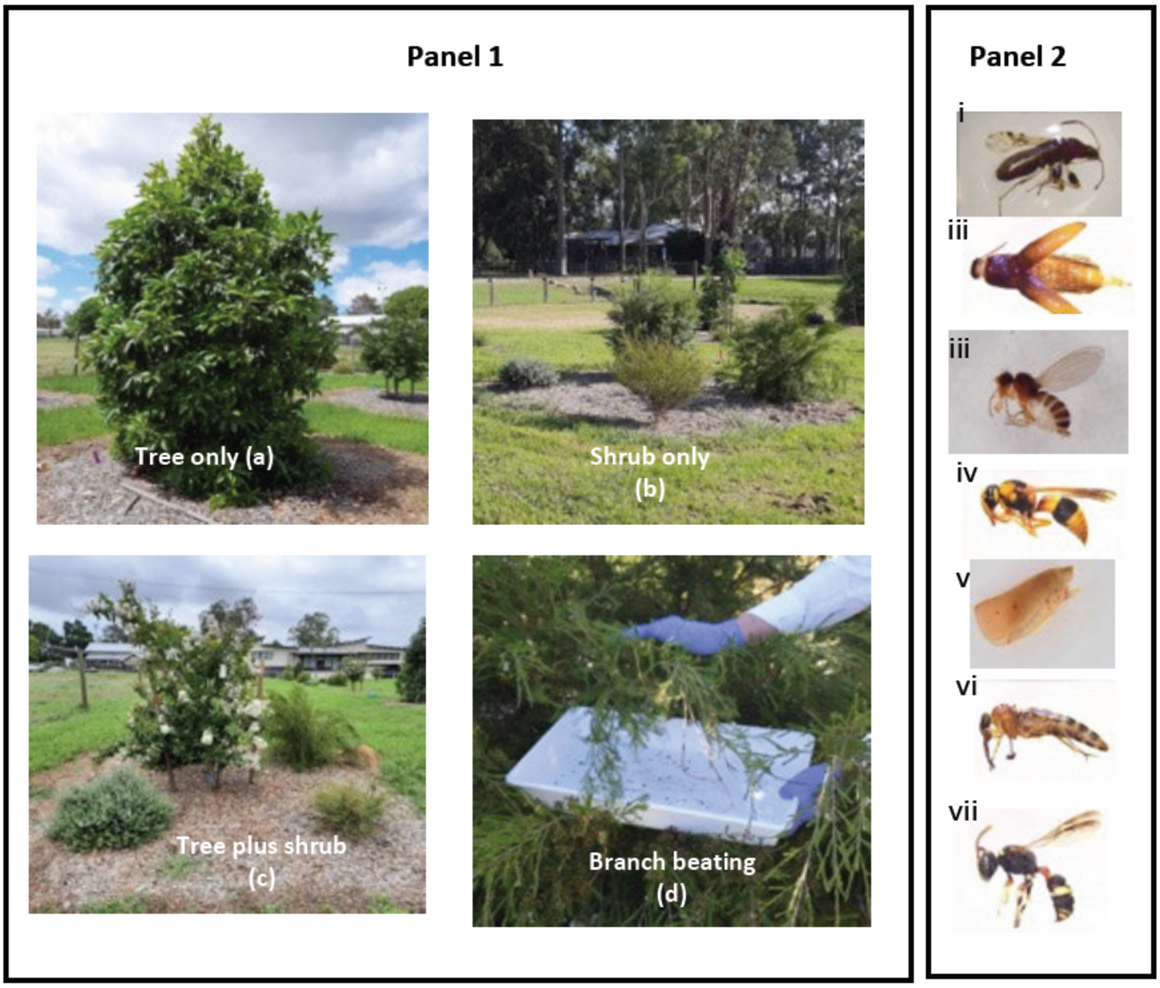


Figure 4.2 Panel 1 shows the experimental design, with examples of 'shrub only', 'tree only' and 'tree plus shrub' treatment plots (a, b & c) and invertebrate sample collection (d). Panel 2 shows example photographs taken in early summer 2020/21, after two years of site establishment. Examples of insects collected: Rhyparochromidae (Hemiptera); ii) Elateridae (Coleoptera), iii) Chloropidae (Diptera), iv) Vespoidae (Hymenoptera), v) Flatidae (Hemiptera), vi) Asillidae (Diptera), vii) Crabronidae (Hymenoptera).

4.2.2.3 Invertebrate identification

Invertebrate identification was performed in two stages. First, ants, spiders and springtails were identified into morphospecies and separated from the samples due to their high abundance and counted. Following this, the rest of the samples were identified to family level using identification keys (Triplehorn et al., 2005). Immature stages were also considered in the identification process, and those that were not identifiable were grouped as larvae/nymphs/pupae. Subsequently, invertebrates were categorized into seven functional groups: detritivores, herbivores, parasitoids, pollinators, predators, sap suckers and scavengers (Supplementary Table 4.1). All specimens were stored in the Entomology Lab of the Hawkesbury Institute for the Environment, Western Sydney University, Richmond, NSW, Australia.

Table 4. 2 List of tree and shrub species used in this study and origin of each species (N = native to Australia, E = exotic). The presence of flowers for each species during the surveying times is marked with “x”. Mean and standard error of the plant height (cm) in January 2021 are given.

Species	**Family (Origin)**	**DEC 2019**	**JAN 2020**	**FEB 2020**	**SEP 2020**	**NOV 2020**	**DEC 2020**	**JAN 2021**	**Height (cm) (mean + se)**
		Summer 2019/2020			Spring 2020		Summer 2020/2021		
Tree species									
Elaeocarpus reticulatus	Elaeocarpaceae (N)	x				x	x		347 (25.4)
Lagerstroemia indica	Lythraceae (E)	x	x	x		x	x	x	478 (21.9)
Liriodendron tulipifera	Magnoliaceae (E)								309 (51.5)
Lophostemon confertus	Myrtaceae (N)	x				x			253 (8.5)
Shrub species									
Baeckea virgata	Myrtaceae (N)	x	x				x	x	183 (2.6)
Melaleuca citrina	Myrtaceae (N)				x	x	x		148 (2.7)
Melaleuca thymifolia	Myrtaceae (N)					x	x	x	63.7 (2.9)
Westringia fruticosa	Lamiaceae (N)				x		x	x	98.9 (4.7)

4.2.2.4 Vegetation data collection

Plant traits i.e., plant height (cm), canopy width (m), canopy depth (m) and leaf area index (LAI) (m^2m^{-2}) were measured during each sampling season. Tree height and crown depth were measured using a Haglöf laser meter (L400, Haglöf, Sweden) with a height resolution of 0.01 m. Crown width measurements were taken as the length of *x* and *y* orthogonal axes from edge to edge through the crown centre and then averaged. Leaf area index (LAI) was measured for each individual plant using a plant canopy analyser (LAI-2200, Li-COR, USA). LAI is the ratio of the area of leaves to the area of the ground under the crown (Kumar and Kaushik, 2005) and was measured on overcast days. Four LAI measurements were performed from four corners of each plant at 1 m height above ground for trees and 10 cm height for shrubs, and with a 90° view cap on a fish-eye lens. LAI data were analysed using FV2200 software developed for LAI-2200, deploying an isolated crown model and removing the 5th mask (68°) to remove the effects of surrounding built-up areas and these four measurements were averaged for each plant. The presence of flowers for each plant was documented during sampling rounds. To estimate plant canopy volume (v), all individual plants were considered as a cylinder (Franceschi et al., 2022). Equation (1) was used to calculate plant canopy volume, where h and r denote height and canopy width, respectively. Leaf area per plant (LA) was calculated using equation (2), where LAI and PCA denote leaf area index and projected crown area (PCA), respectively. The projected crown area (PCA; m^2) was calculated as an ellipse using crown diameters in the wider and narrower directions (Li et al., 2017). Canopy volume and LA across all plants within a plot were summed to obtain plot-level values.

$$V = \pi r^2 h \qquad \text{..................} \ (1)$$

$$LA = LAI / PCA \qquad \text{..................} \ (2)$$

4.2.3 Data analysis

Invertebrate data were used to calculate the abundance and taxonomic richness of invertebrate communities and their associated functional group. Data analyses were conducted first to evaluate differences between plant types (i.e., trees versus shrubs) and then to evaluate differences among the different vegetation structures (i.e., ‘shrub only’, ‘tree only’, ‘tree plus shrub’).

To assess sampling effort and compare species richness for both plant types and vegetation structures across the three sampling seasons, invertebrate species accumulation curves were plotted (Supplementary Figure 4.2). Function “specaccum” which is a sample-based rarefaction method from the Vegan package was used to produce species accumulation curves for each season (Oksanen et al., 2022).

To compare the abundance and richness of invertebrates (as response variables) between trees and shrubs, data were analysed using Fit Generalized Linear Mixed Models with the *glmerPQL* function from the MASS package (Ripley et al. 2013). Plant type (whether tree or shrub) and sampling round were used as explanatory variables and plot ID was considered as a random effect in this model (Table 4.3).

To evaluate differences in invertebrate abundance (as the response variable) for vegetation structure treatments (shrubs only, trees only and tree plus shrubs) data were analysed using the lmer function from lme4 package (Bates et al., 2015); data were log-transformed to ensure normality of errors. To compare invertebrate richness for vegetation structure treatments glmer.nb function from lme4 package were used to analyse the data (Bates et al., 2015). Tree species ID accounted for <1% of the model variance, and was therefore not retained in the final models. For both models, sampling round (summer 2019/20; spring 2020 and summer 2020/21) was used as a fixed effect and plot ID as a random effect (Table 4.3).

To evaluate relationships between plant traits and the abundance and taxonomic richness of invertebrates, both overall and within functional groups, data for individual shrubs/trees and vegetation structure treatments (i.e., ‘shrub only’, ‘tree only’, ‘tree plus shrub’) were analysed using the *lmer* function in R. Each model included plot level variables (i.e., treatment, plant volume (sum of the canopy volume of all plants present in each plot), LAI and the number of plants in flower) as fixed effects, and sampling round and plot ID as random effects. Models were fitted using the *lmer* and *glmer.nb* functions in the lme4 package (Bates et al., 2015). Canopy width and height were not included in the models since both traits were highly correlated (r = 0.91 and 0.81, respectively) with plant volume. Residual plots were inspected to check model fits. For pairwise comparison, post-hoc Tukey tests were conducted using the *Multcomp* package (Hothorn et al., 2008). All analyses were performed using R version 3.2.2 (R Core Team, 2021) and statistical significance was considered at $p \leq 0.05$.

Table 4.3 Models used to analyse invertebrate data

	Function	**Response variable**	**Explanatory variable**	**Random effect**
Question 1	How does the abundance and richness of invertebrate communities vary between individual shrubs and trees and plant structure?			
Plant types (trees and shrubs)	*glmerPQL*	Invertebrate abundance	Plant type and sampling round	Plot ID
	lmer	Species richness		
	lmer	Abundance of invertebrates of each functional group		
Plant structure (‘shrub only’, ‘tree only’, ‘tree plus shrub’)	lmer	Invertebrate abundance	Vegetation structure and sampling round	Plot ID
	glmer.nb	Species richness		
	lmer	Abundance of invertebrates of each functional group		
Question 2	What factors explain differences in invertebrate communities between plant types and among different structures?			
Plant types (Trees, shrubs and overall)	lmer	Invertebrate abundance	Volume, LAI and presence of flower	Plot ID
	lmer	Species richness		
Plant structure (‘shrub only’, ‘tree only’, ‘tree plus shrub’ and overall)	lmer	Invertebrate abundance	Volume, LAI and No. of flower plant	Plot ID
	lmer	Species richness		

4.3 Results

4.3.1 Invertebrates associated with individual shrubs/trees

Invertebrate abundance varied significantly between individual shrub and tree species (p = 0.01) (Figure 4.3 (a); Supplementary Table 4.2 & 4.3). Abundance varied from 0 to 979 individuals on individual shrubs (mean 95.72, ± 4.53 standard error) and from 0 to 362 individuals (53.85 ± 5.05) on trees.

Invertebrate richness at the individual shrub/tree level also varied significantly between shrubs and trees ((p = 0.01). Taxonomic richness ranged from 0 to 19 taxa for shrubs (7 ± 0.28) and 0 to 16 taxa for trees (9.26 ± 0.23) across all sampling rounds (Figure 4.3 (b)).

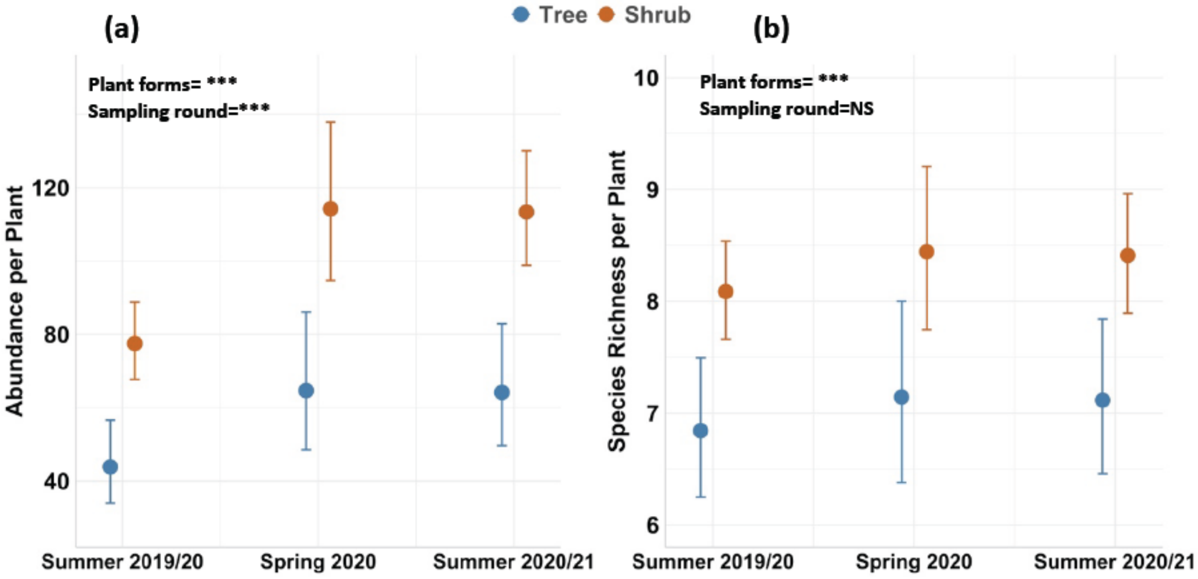


Figure 4.3 Pairwise comparisons of abundance (a) and species richness (b) of invertebrates between shrubs and trees across sampling periods – summer 2019/20, spring 2020 and summer 2020/21. Circles and error bars depict mean abundance and standard errors. Supplementary Table 4.2 summarises ANOVA results of the *lmer* model for each figure. Plot' ID was included as a random factor in models.

4.3.2 Invertebrates associated with vegetation structure treatments

There were similar patterns among the vegetation structure treatments (p = 0.01) and across sampling rounds (Figure 4.4 (b)). Both 'shrub only' and 'tree plus shrub' treatments were associated with significantly higher numbers of invertebrates than the 'tree only' treatment (Figure 4.4 (a); Supplementary Table 4.2 & 4.3).

In contrast, invertebrate richness varied up to two-fold among vegetation structure treatments across all sampling rounds (Figure 4.4 (b), Supplementary Table 4.2). Both 'shrub only' and 'tree plus shrub' plots were associated with higher invertebrate richness than tree-only plots, and taxonomic richness varied significantly (p = 0.01) across all sampling rounds with greater richness observed in spring 2020 compared to the summer sampling rounds (Figure 4.4 (b)).

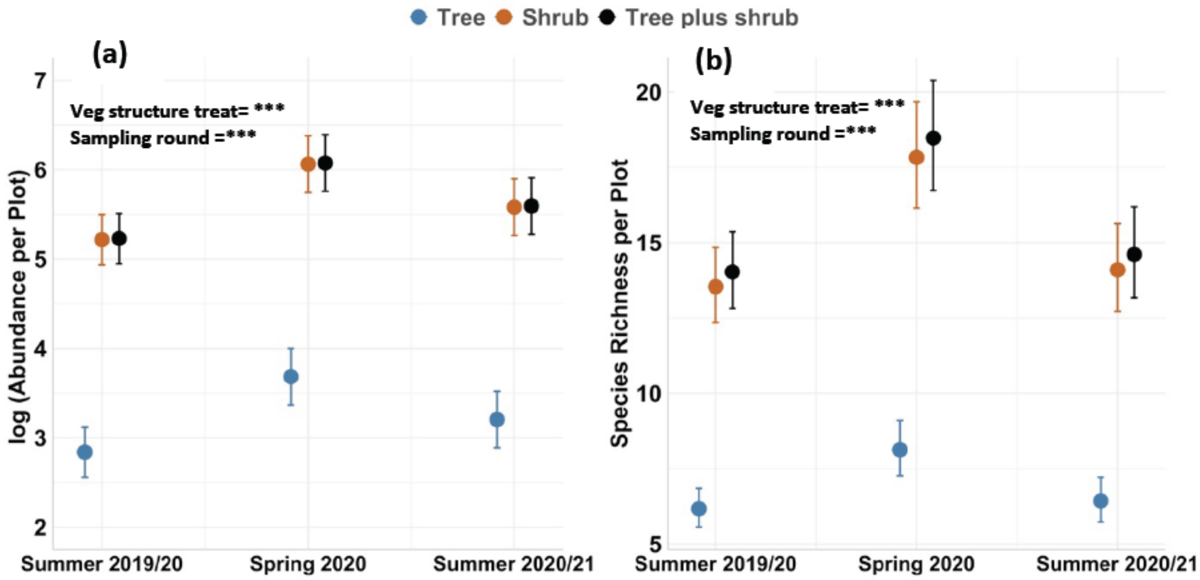


Figure 4.4 Pairwise comparisons of the abundance (a) and species richness (b) of invertebrates at the plot-level among vegetation structure treatments (i.e., 'tree only', 'shrub only' and 'tree plus shrub' treatments) across sampling rounds. Circles and error bars depict means and standard errors. Supplementary Table 4.2 summarises ANOVA results of the *lmer* model for each figure. Plot ID was included as a random effect.

4.3.3 Relationships between plant traits and invertebrates for individual shrubs/trees and vegetation structure treatments

4.3.3.1 Individual shrubs/trees

For individual shrubs, invertebrate abundance and taxonomic richness were significantly positively associated with canopy volume, LAI and the presence of flowers, whereas for individual trees, there was a strong association only with canopy volume (Table 4.4). Pooling the data for individual shrubs and trees, canopy volume and LAI were positively associated with invertebrate abundance and richness overall (Table 4.4). A positive correlation between plant height and invertebrate richness was evident for shrubs, but not for trees for the overall dataset. It is important to note that canopy width and height were not included in the full models, due to their strong correlation with plant volume, but it is apparent that all three traits contribute to differences in invertebrate abundance and richness between individual shrubs and trees and across vegetation structure treatments.

Table 4.4 ANOVA results of the three *lmer* models for individual shrubs, trees and the combined dataset (trees and shrubs) summarising chi-square (X^2) values for each of the explanatory variables (canopy volume, LA (leaf area per plant) and presence of flowers), for invertebrate abundance and taxonomic richness data. Plot ID and sampling round were included as random factors in models.

Response variable	**Explanatory variables**	**Abundance (log)**			**Richness (sqrt)**	
		df	X^2	***p* value**	X^2	***p* value**
Shrubs (n=128)	Volume	1	116.0	0.001	138.3	0.001
	LA	1	4.06	0.04	6.92	0.01
	Presence of flowers	1	48.5	0.001	49.4	0.001
Trees (n=32)	Volume	1	0.26	ns	0.38	ns
	LA	1	0.09	ns	7.44	0.01
	Presence of flowers	1	16.0	0.001	2.08	ns
Overall (individual trees and shrubs combined) (n=160)	Volume	1	35.8	0.001	8.45	0.003
	LA	1	1.31	ns	10.6	0.001
	Presence of flowers	1	47.3	0.001	28.9	0.001

4.3.3.2 Vegetation structure treatments

There were significant associations between invertebrate abundance and the total plot-level canopy volume and the number of flowering plants in both 'shrub only' and 'tree plus shrub' treatments. This was also the case for the overall dataset (i.e., all three vegetation structure treatments together) (Table 4.5). In the 'tree only' treatment, however, invertebrate abundance and richness were only significantly (positively) correlated with the number of individual flowering trees.

Table 4.5 Results of four *lmer* models summarising chi-square (X^2) values for each model variable; total volume, total LA (leaf area per plant) and number of flowering plants), for invertebrate abundance and taxonomic richness for vegetation structure treatments (tree only, shrub only, tree plus shrub, and overall (all three treatments combined)), including significance for each variable of the models. Plot ID and sampling rounds were included as random factors in models.

Vegetation structure treatments	**Explanatory variables**	**Abundance (log)**			**Richness(log)**	
		df	X^2	***p* value**	X^2	***p* value**
Tree only Treatment (n=16)	Total volume	1	0.11	ns	1.11	ns
	Total LA	1	0.03	ns	1.18	ns
	No. of flowering plants	1	25.85	0.001	20.48	0.001
Shrub only Treatment (n=16)	Total volume	1	174.62	0.001	150.28	0.001
	Total LA	1	2.98	ns	2.52	ns
	No. of flowering plants	3	23.61	0.001	2.51	ns
Tree plus shrub Treatment (n=16)	Total volume	1	18.84	0.001	18.01	0.001
	Total LA	1	6.6	0.01	9.02	0.002
	No. of flowering plants	4	4.20	0.04	0.01	ns
Overall (all treatments combined) (n=48)	Total volume	1	27.01	0.001	32.43	0.001
	Total LA	1	0.05	ns	2.52	ns
	No. of flowering plants	4	153.55	0.001	48.34	0.001

4.3.4 Invertebrate functional group differences between individual shrubs/trees and among vegetation structures

Herbivores and predators were the most abundant groups associated with both individual shrubs and trees, and all vegetation structure treatments, while scavengers were the scarcest functional group across the full dataset (Supplementary Table 4.4). Overall, except for parasitoids, the abundance of the different functional groups was greater in individual shrubs compared to individual trees, and in 'shrub only' and 'tree plus shrub' vegetation treatments compared to tree only plots (Figures 4.5 & 4.6; Supplementary Table 4.4, 4.5 & 4.6). Detritivores, herbivores, parasitoids and sap suckers were more abundant in spring 2020 compared to both summer sampling rounds for individual shrubs/trees. Abundance of predators and scavengers was greater in summer 2020/2021 than in preceding sampling periods for both individual shrubs/trees and the different vegetation structures. In contrast, for both individual shrubs/trees and vegetation structure treatments, pollinators were more abundant in summer 2019/20 (Figures 4.5 & 4.6; Supplementary Tables 4.5 & 4.6).

4.3.5 Invertebrate functional groups for individual shrubs/trees and vegetation structure treatments

For individual shrubs and trees, the numbers of detritivores, pollinators, herbivores and parasitoids were significantly positively associated with plant volume and the presence of flowers. Plant volume accounted for the highest proportion of the data variance for herbivores and parasitoids. The numbers of predators, sap suckers and scavengers were significantly related to all measured plant traits (Supplementary Table 4.7).

For vegetation structure treatments, the number of flowering plants was the most important explanatory variable for differences in all invertebrate functional groups except pollinators and parasitoids. Total leaf area and total canopy volume were important explanatory variables for the number of pollinators and parasitoids, respectively (Supplementary Table 4.8).

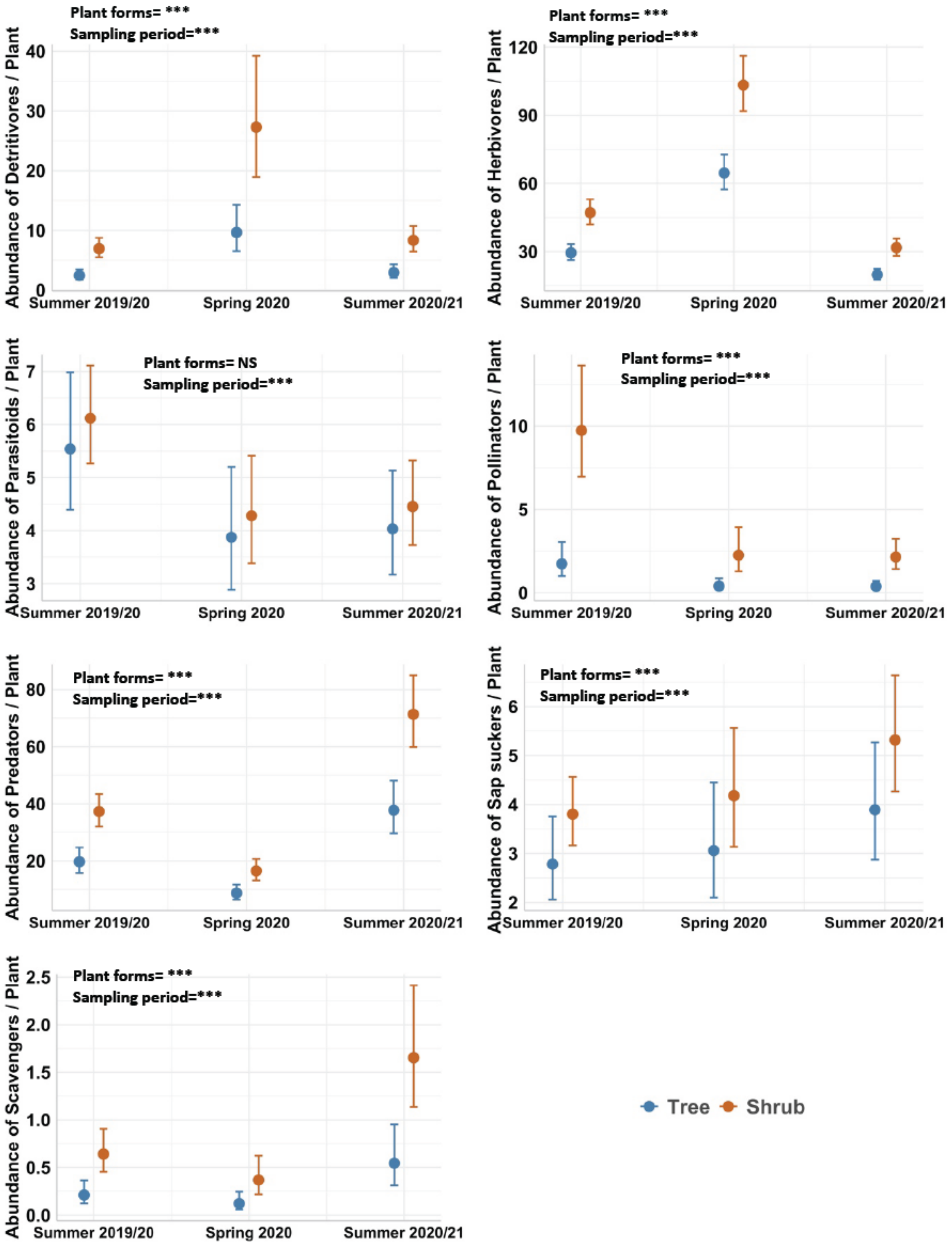


Figure 4.5 Plots show pairwise comparison of abundance of seven invertebrate functional groups between individual shrubs and trees across sampling periods – summer 2019/20, spring 2020 and summer 2020/21. Circles and error bars depict means and standard errors, respectively. See supplementary Table 4.5 for ANOVA results of the model for each plot. Plot ID was included as a random factor in models. Asterisks represent significance of the variables: *** $P < 0.001$.

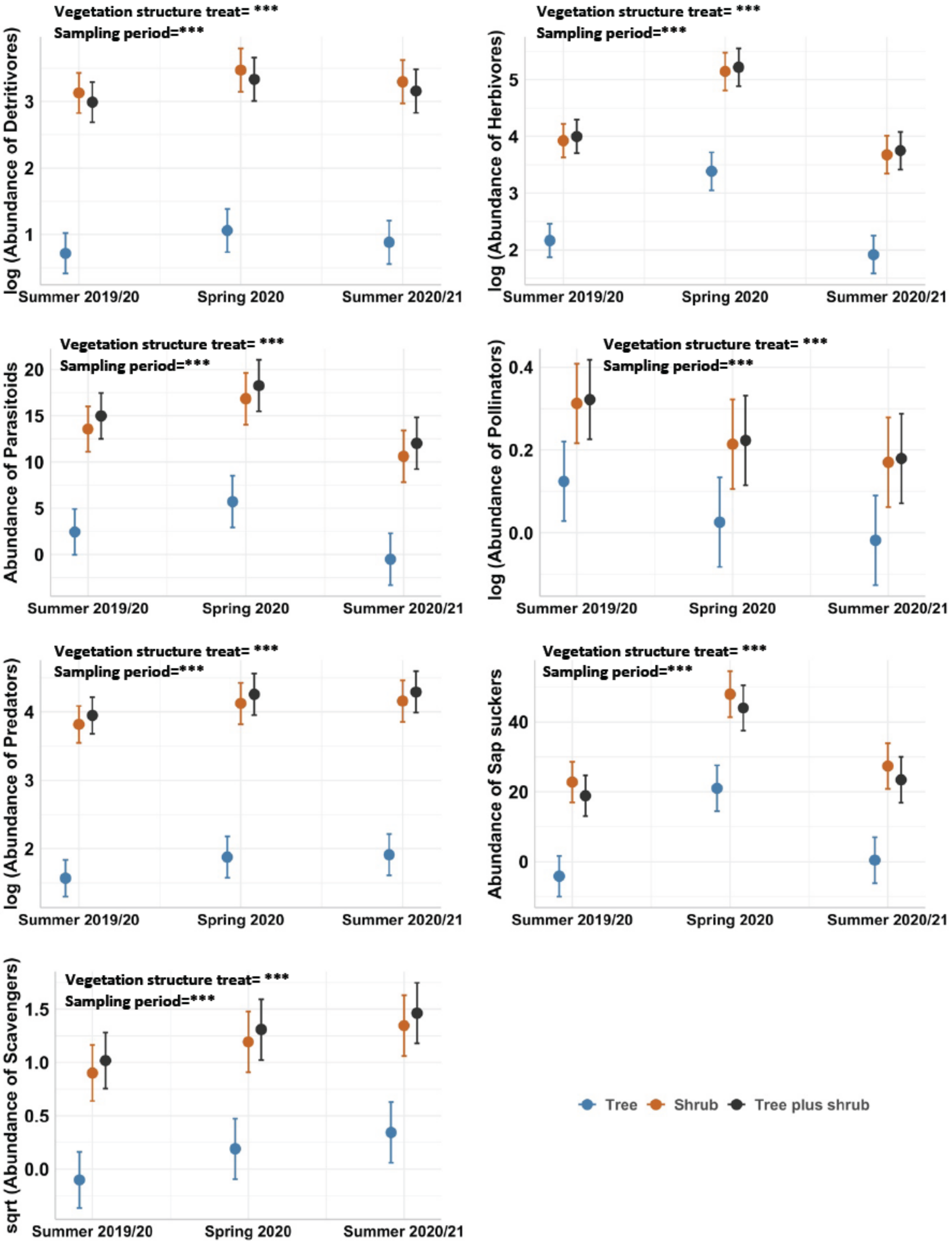


Figure 4.6 Plots show pairwise comparison of abundance of seven invertebrate functional groups among vegetation structure treatments at the plot level across sampling periods – summer 2019/20, spring 2020 and summer 2020/21. Circles and error bars depict means and standard errors, respectively. See supplementary Table 4.6 for ANOVA results of the *lmer* model for each plot. Plot ID was included as a random factor in models. Asterisk represent significance of the variables: *** P < 0.001.

4.4 Discussion

This study shows that invertebrate abundance and richness tend to be greater in shrubs, both when planted alone and in combination with trees, and lowest in trees growing on their own. This highlights the importance of shrubs in promoting invertebrate biodiversity in urban landscapes. Furthermore, plant species with wide and dense canopies and those that were in flower harboured more diverse invertebrate communities in terms of taxonomic and functional groupings. The key findings of this study are that an urban greening strategy that includes complementary plantings of both flowering shrubs and trees increases biodiversity in urban areas, by providing both a greater volume and diversity of vegetation and associated resources in a given area. Such an approach thus has the potential to increase biodiversity services associated with green areas across urban landscapes.

4.4.1 Invertebrates were mostly associated with shrubs

Shrubs were associated with more abundant invertebrate communities overall, compared to trees (Supplementary Figure 4.1), a difference that was also evident within functional groups (apart from parasitoids). Among functional groups, herbivores were the most abundant group followed by predators, with scavenger being the least abundant group. A study in the Californian desert, USA, by Zuliani et al. (2021) reported that an increase in shrub-density was associated with a higher abundance of invertebrates, particularly driven by herbivorous groups (e.g., moths, trichopterans, thrips). Insect herbivores are a key part of terrestrial food webs and influence numerous ecosystem services such as primary production and aesthetic value (Schmitt et al., 2021; Schowalter, 2012). Shrubs were also associated with a large number of detritivores (mean difference of nearly 65%) which facilitate the decomposition of organic matter and return essential nutrients to the system. Although all functional groups were associated with both trees and shrub, the inclusion of shrubs significantly increased the abundance of all invertebrates' groupings.

Compared to trees, the studied shrubs were considerably smaller in height and volume (Supplementary Figure 4.3 & 4.4, Table 4.2) but were still associated with greater invertebrate numbers. The shrubs used in this study all had more than two stems and comparatively more branches than trees which indicates that they have more shoot architecture (Gotmark et al., 2016; Ulyshen, 2011). Furthermore, the relative proximity of shrub foliage to the ground, compared to trees, implies close interaction among soil, leaf litter and shrub canopy layers (Gotmark et al., 2016). These characteristics of shrubs may provide greater shelter from harsh climates by buffering temperatures and wind (Zuliani et al., 2021) as well as providing a greater physical connection – and free movement - between soil- and canopy-dwelling invertebrates. However, it should be noted that the relatively young age of trees (1-3 years) using in this study means that, although their canopies were still more distant from the ground than shrubs, stem heights were relatively small compared to mature trees.

At the plot level, the 'shrub only' and 'tree plus shrub' treatments increased the aggregated invertebrate abundance compared to the 'tree only' treatment, as would be expected from increasing the number of individuals from one (tree only) to four (shrub only) or five (tree plus shrub). These results are in line with Mata et al. (2021), who reported the importance of mid-story vegetation (consisting of shrubs and graminoid) for sustaining high insect biodiversity in urban parks in Melbourne, Australia. The most complex vegetation structure – the 'tree plus shrub' treatment - was not associated with a greater number of invertebrates per plot than the 'shrub only' treatment, even though the latter was significantly greater than tree-only plots. The results of the current study did not support the hypothesis that greater structural complexity can supports higher invertebrate diversity and abundance. This is not consistent with Galle et al. (2017) who found that the number of beetles and bugs increased with increasing vegetation complexity. A plant architecture manipulation study by Grof-Tisza et al. (2017) also overserved that invertebrate abundance and species richness were highly corelated with the density of branches rather than with the height and width of vegetation. Whereas, in case of the current study, the inclusion of tall and wide trees alongside shrubs did not bring any remarkable increase in invertebrate abundance and richness. Larval life stages of invertebrates which are dependent on host plants can be affected even by simple vegetation structure (Bernays & Chapman, 2014). In urban areas, tree planting, including a focus on native trees, has received considerably more attention than other vegetation types (Turner-Skoff & Cavender, 2019). However, the habitat and resource requirements of some invertebrate taxa cannot be fully achieved by planting trees alone (Le Roux et al., 2014a).

4.4.2 Plant traits, flowering and invertebrate diversity

Canopy volume of shrubs was positively correlated with invertebrate abundance and richness for individual shrubs and the tree-shrub plus tree mixed plots, suggesting that greenspace with higher vegetation complexity will enhance the diversity of invertebrate communities. This finding is consistent with Delgado de la Flor et al. (2020), who studied the importance of local and landscape-scale features in the city of Cleveland, Ohio, USA on spider community assembly, showing a positive corelation between spider community diversity and plant height and biomass.

The current study found the greatest abundance, taxonomic and functional group richness of invertebrates in both trees and shrubs during their flowering periods. In the ‘shrub only’ and ‘tree plus shrub’ treatments, the four shrub species had flowers at least three (out of seven) of the surveying times and showed a succession in flower emergence (overlapping flowering phenology). Flower-rich urban greenspaces attract a higher number of pollinators (e.g., bees, beetles and flies) than florally-depauperate areas (Popic et al., 2013; Theodorou et al., 2020a). Among the four tree species, *Lagerstroemia indica* flowered profusely and attracted large numbers of honeybees, native bees and wasps, although flower-visitors were not specifically included in this study. Wardhaugh et al. (2012) explored the interaction between flowers and invertebrates in an Australian tropical forest and observed that the microhabitat of flowers attracted a greater proportion of invertebrates than tree canopies without flowers. Flowers are nutritious resources and strong determinants of invertebrate (e.g., pollinators, herbivores) abundance in ecosystems (Schoch et al., 2022). Therefore, tree or shrub species that flower profusely, and/or over longer periods can attract and support a higher level of invertebrate biodiversity in urban settings. The potential for spill-over pollination benefits for urban gardens, community gardens and remnant native land within the urban matrix are also clear, with benefits that go beyond the simple aesthetics of more biodiverse streetscapes (Baldock et al., 2019).

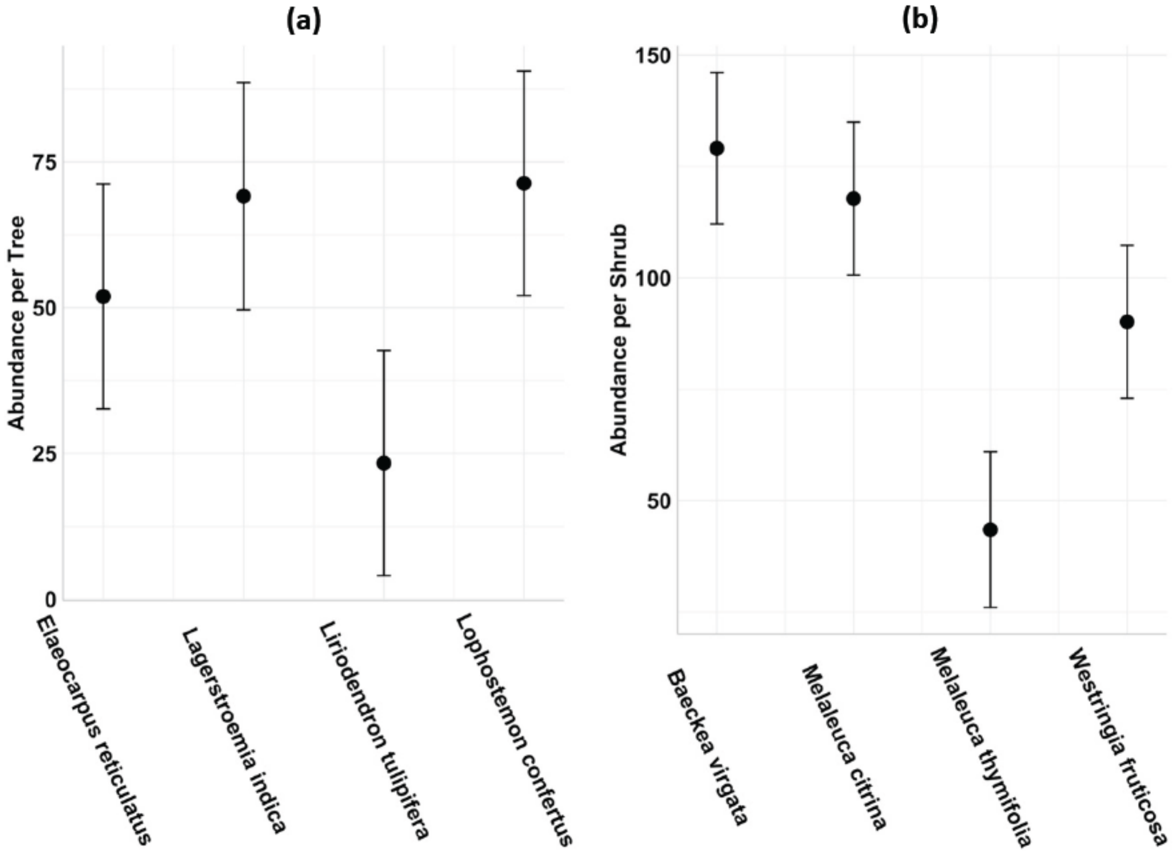
4.4.3 Study limitations

We acknowledge that the experiment was conducted in a suburban setting, which we are extrapolating to urban settings as our objective was to explore whether plant forms and vegetation structure vary in terms of invertebrate abundance and the functional groups they support. Although invertebrate abundance and diversity will likely differ between such settings, we believe that treatment-related differences are still informative and likely to apply, in broad terms, across a variety of contexts. Further, as we focused on the entire community of invertebrates, rather than just a representative taxonomic group, community-level responses provide an overview of responses that are arguably less-context-dependent than narrower focussed studies. Invertebrate assemblages may be slightly different in winter due to their dormancy, high mortality rate and/or migration than in summer and spring. Due to favorable weather conditions and availability of resources than the other season, invertebrates are comparatively more abundant and active during spring and summer. Therefore, sampling in summer and spring would be more effective in evaluating the entire community of invertebrates. The presence of tall trees could potentially attract birds to study plots, with negative effects on associated invertebrate abundance and richness (via predation and dislodgement). However, we observed that shrubs were also frequented by visiting birds with, for example *M. citrina* attracting rainbow parakeets, noisy mynas and common mynas, at numbers similar to (or above) visitors to study trees, such that treatment-associated differences in bird visitation was unlikely when considered across the whole study period. The use of presence/absence data for flowering might not be as effective as measurements of floral abundance or size, for evaluating the importance of flowers for invertebrate communities. Inclusion of floral traits such as flower density, size and the type of reward for pollinators can provide more detailed insights into the drivers of invertebrate abundance and diversity. This topic warrants further study.

4.4.4 Implication for urban greenspace management

This study highlights the need for a more refined urban planting strategy that incorporates not just plant species aesthetics and cooling benefits, but also the food, shelter, oviposition and nesting site requirements of target invertebrate groups when selecting trees and understory species for urban areas. Shrubs can support higher overall levels of invertebrate biodiversity, as well as greater abundance and diversity within key functional groups, such as detritivores, predators and herbivores. High invertebrate abundance, in turn, attracts and provisions associated organisms, thereby enhancing the abundance and diversity of other taxonomic groups over and above the existing trees. More structurally complex plantings can be easily adopted in parks, roundabouts and urban gardens. In streetscapes, trees and shrubs may be more restricted by concrete walls and/or pavements, and visibility considerations may constrain planting options. Where shrubs can be planted alongside already established trees, the additional floral trait diversity and microhabitat complexity are likely to support greater numbers and more diverse invertebrate communities. Maintaining shrubs in urban areas does, however, have challenges, including water scarcity, competition with trees for soil resources, air pollution, vandalism and the urban heat island effect. This three-year experiment did, however, demonstrate that planting shrubs along with trees can generate positive outcomes for invertebrates and thus overall biological diversity in suburban areas. Given that urban greening strategies focus strongly on trees, it is important to recognise the benefits shrubs bring to urban landscapes and, ideally, to broaden planting strategies and policy to recognise the role that shrubs can play in sustainable urban planning.

Supplementary Figure

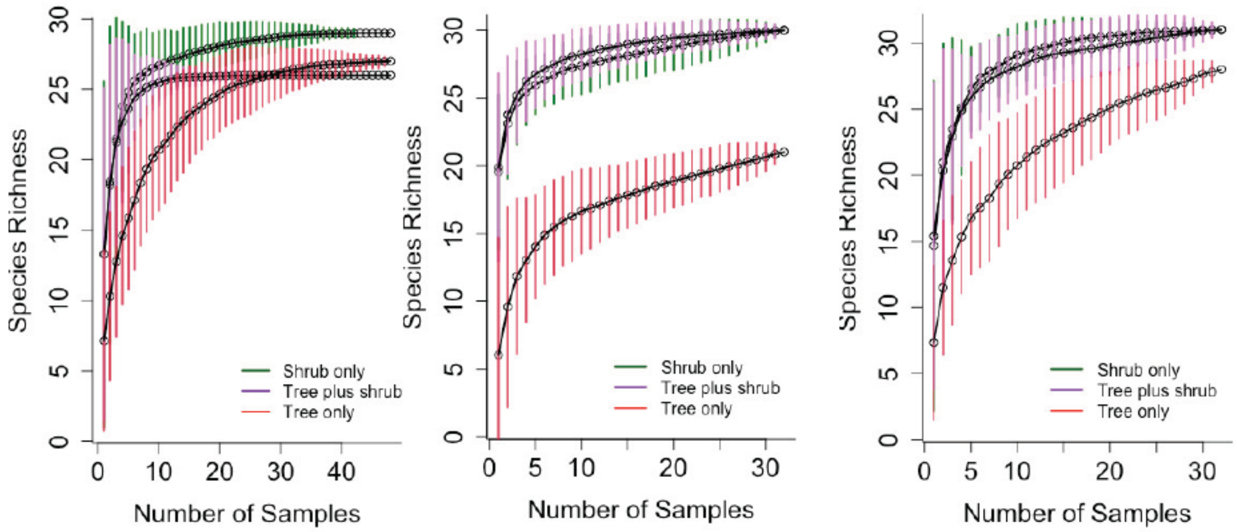


Supplementary Figure 4.1 Plots show pairwise comparisons of abundance of invertebrates among (a) Tree and (b) Shrub species across all sampling rounds. Circles and error bars depict means and standard errors.

Vegetation Structure

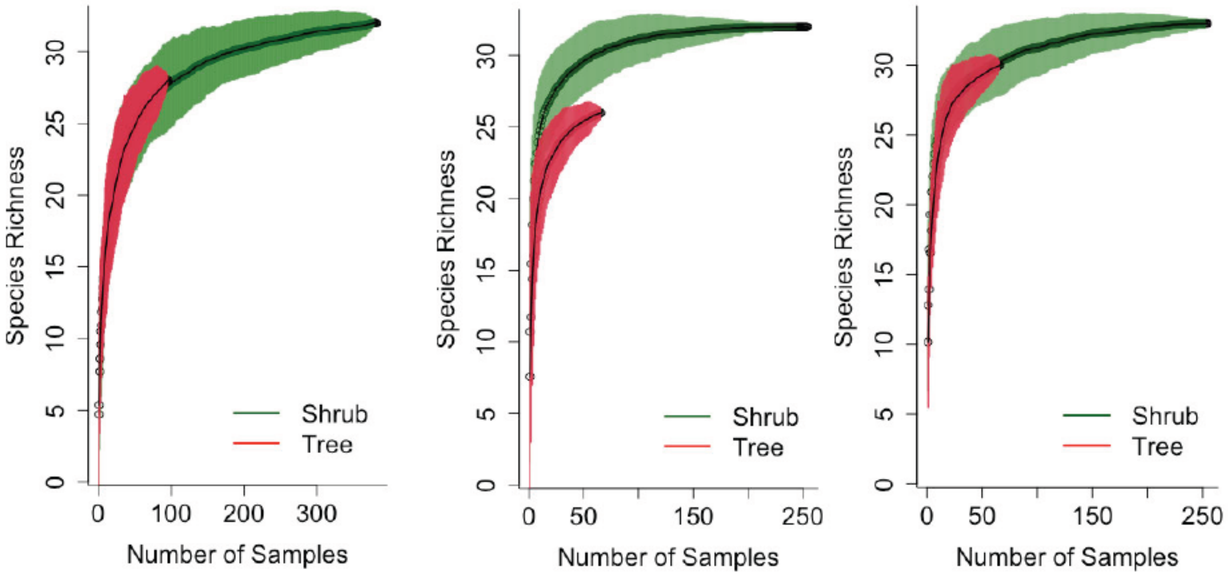
Summer 2019/20 **Spring 2020** **Summer 2020/21**

(a) **(b)** **(c)**

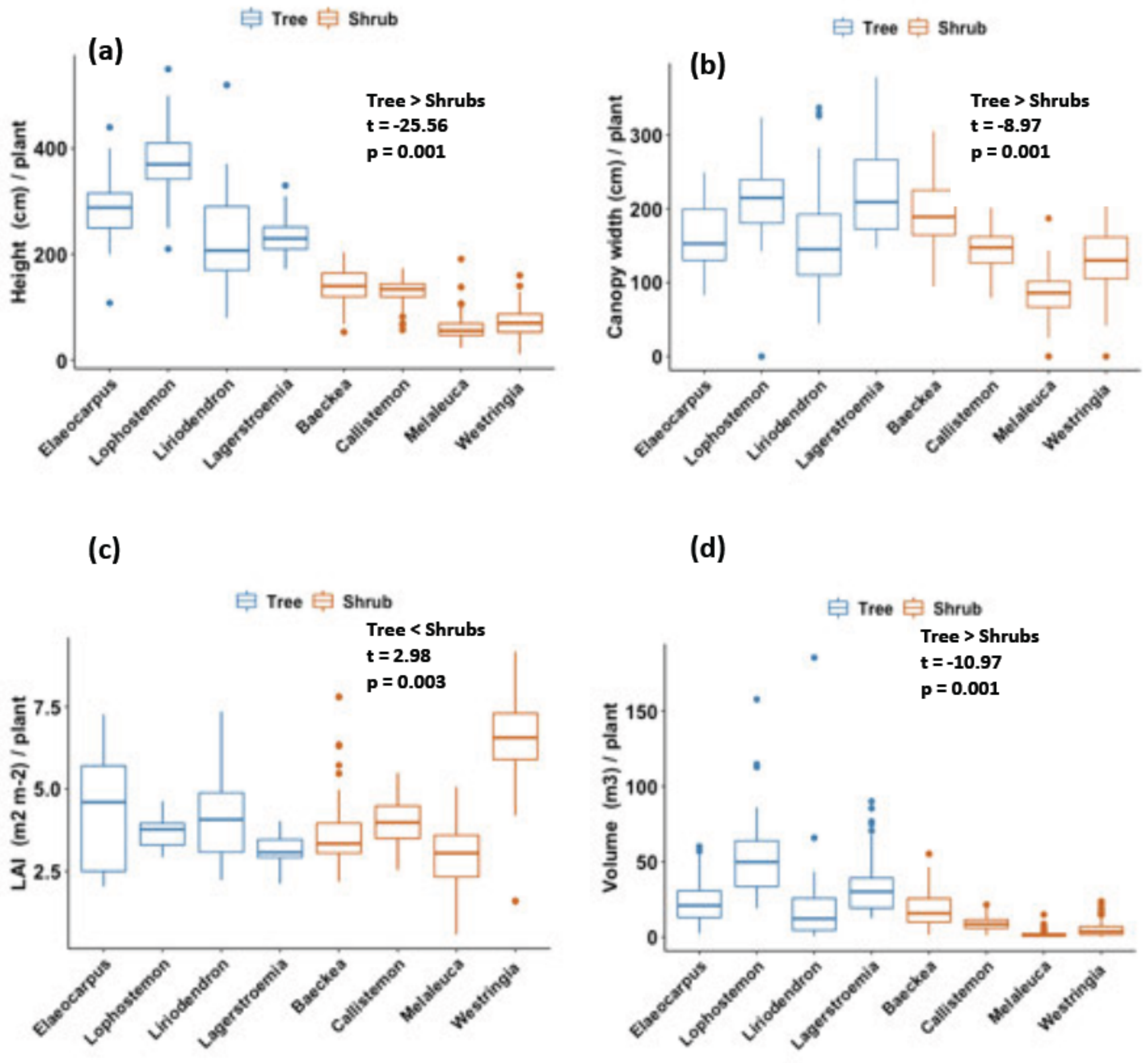


Plant Form

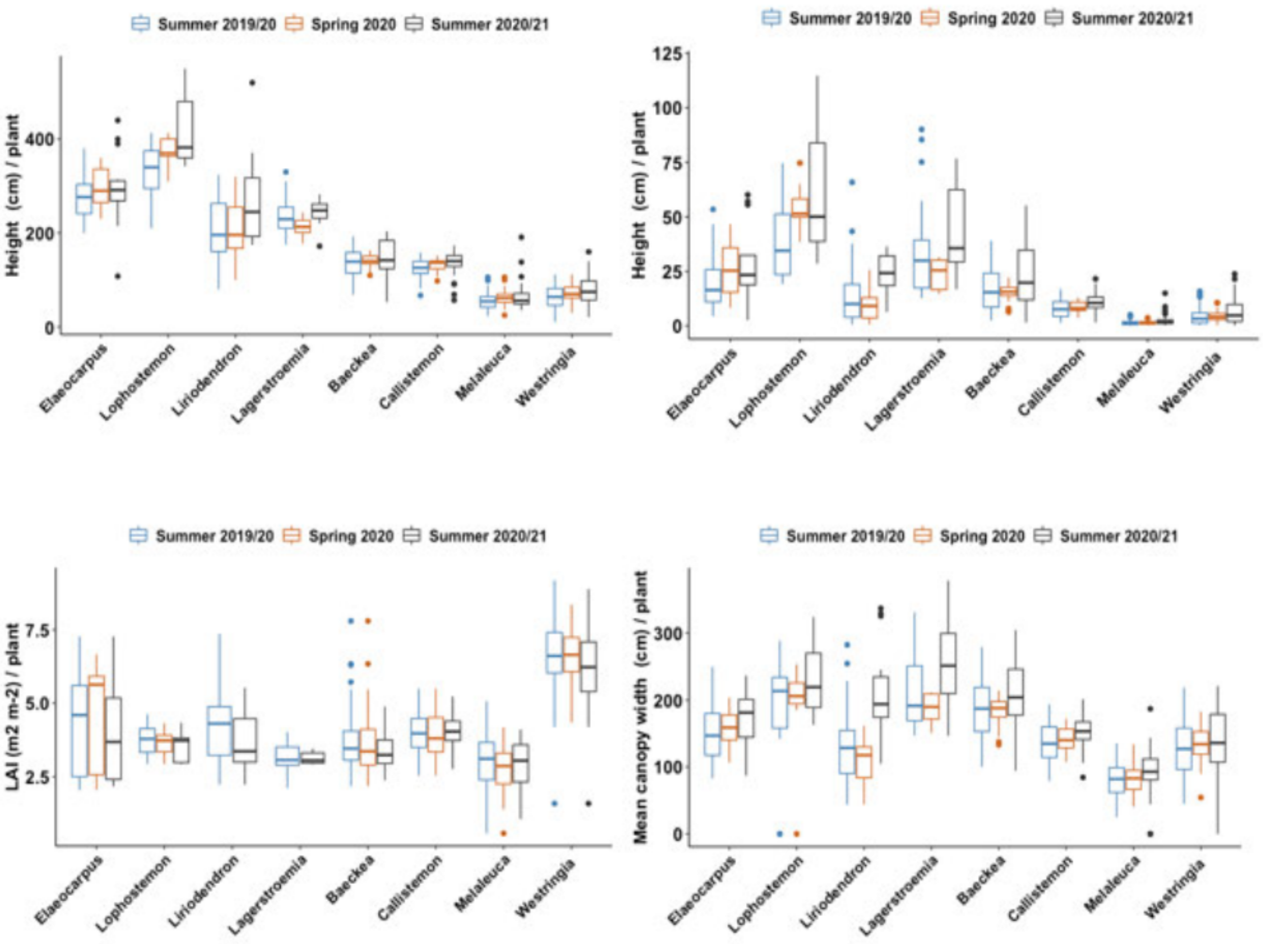
(d) **(e)** **(f)**



Supplementary Figure 4.2 Species accumulation curves for all three vegetation structures – 'Tree only','Tree plus Shrub' and 'Shrub only' (a-c) and plant forms- shrub and tree (d-e) of summer 2019/20, spring 2020 and summer 2020/21.



Supplementary Figure 4.3 Plots show comparisons of height (a), canopy width (b), leaf area index (LAI) (c) and volume (d) among individual shrubs and trees species. t-test results showed that plant traits vary significantly between trees and shrubs.



Supplementary Figure 4.4 Plots show comparisons of height (a), canopy width (b), leaf area index (LAI) (c) and volume (d) among shrubs and trees species across sampling rounds.

Supplementary Table

Supplementary Table 4.1 Invertebrate functional group categorisations

Functional groups	Definition	Invertebrate groups/families	Order
Detritivores	A group of organisms that feeds on detritus or organic waste.	Formicidae Trichoptera	Hymenoptera Lepidoptera
Herbivores	A group of invertebrates that feeds on plant parts	Nymph/ larva of Moths/ Trichoptera Seed wasp Fig wasp Agromyzidae Acrididae Heleomyzidae Argidae Midge Fungus gnat Symphyta Chrysomelidae Chrysomelidae:Alticinae Culicidae Cynipidae Chloropidae Curculionidae Cerambycidae Psocoptera Prostigmata Noctuidae Terebrantia Tubulifera Cricket	Hymenoptera Lepidoptera Coleoptera Heteroptera Orthoptera Thysanoptera Homoptera
Parasitoids	A group of invertebrates that parasitize other arthropods by depositing eggs in the pupae, larva, or eggs of their host.	Bethylidae Brachonidae Chalcididae Sub-Chalcidoidea Ichneumonidea Encyrtidae Eurytomidae Gastropidae Ichneumonidae Megaspilidae Mymaridae Platygastridae Pteromalidae Perilampidae Scelioninae Proctotrupidae Torymidae Tachinidae	Hymenoptera Diptera

Pollinators	A group of invertebrates that transfers pollen from the anther of one flower to the stigma of another, thereby helps plants to reproduce	Honeybees Aculeata Anthicidae Cleridae Colletidae Halictidae Nitidulidae Mordellidae Scarabaeidae Syrphidae	Hymenoptera Coleoptera
Predators	A group of organisms that usually feeds on other invertebrates	Spider Anthocoridae Asilidae Cantharidae Coccinellidae Hemerobidae Mutillidae Melyridae Monotomidae Mesostigmata Mantispidae Nabidae Pompilidae Psychodidae Psocoptera Reduviidae Staphylinidae Tiphiidae Vespidae	Araneae Diptera Coleoptera Hemiptera Heteroptera Hymenoptera Arachnid Neoproptera
Sap suckers	Invertebrates that feed on the sugary sap produced in foliage and transported in the soft phloem tissue beneath the bark.	Aleyrodidae Cicadas Cicadelidae Coreidae Delphacidae Flatidae Geocoridae Gerridae Hemiptera Hemerobidae Lygaeidae Miridae Pentatomidae Psyllidae Rhyparochromidae Scutelleridae Terebrantia Tingidae	Hemiptera Heteroptera Homoptera Thysanoptera
Scavengers	Invertebrates that consume dead organisms.	Blattodae Cucujidae Hydrophilidae Muscidae	Coleoptera Diptera

Supplementary Table 4.2 Results for *lmer* models indicating chi-square (X^2) values of each of the model variables for individual shrubs/trees and vegetation structure treatments ('tree only', 'shrub only' and 'tree plus shrub').

	df	**Abundance**		**Richness**	
		X^2	*p* **value**	X^2	*p* **value**
Individual shrubs and trees	1	19.83	0.01	12.50	0.001
Sampling rounds	2	20.36	0.001	1.33	NS
Vegetation structure treatments	2	263.97	0.001	214.20	0.001
Sampling rounds	2	24.78	0.001	36.79	0.001

Supplementary Table 4.3 Mean and standard error (in parenthesis) values of invertebrates sampled in the plant forms and vegetation structure treatments in different sampling periods.

Sampling periods	**Plant forms (mean ± se)**		**Vegetation structure treatments (mean ± se)**		
	Tree	**Shrub**	**Tree only**	**Shrub only**	**Tree plus shrub**
Summer 2019-2020	42 (4.96)	78 (6.07)	39 (5.97)	320 (49.7)	347 (55.4)
Spring 2020	65 (19.1)	114 (11)	68 (13.8)	649 (47)	692 (62.6)
Summer 2020-2021	67 (9.05)	113 (8.30)	61(11.2)	470 (76.9)	500 (85.2)

Supplementary Table 4.4 Mean and standard error (in parenthesis) values for invertebrate abundance per plant and per plot for the respective functional groups.

Functional groups	Individual shrubs/trees		Vegetation structure (plot level)		
	Tree	**Shrub**	**Tree only**	**Shrub only**	**Tree plus shrub**
Detritivores	6.97 (2.17)	10.8 (0.88)	5.72 (2.53)	43.87 (4.71)	46.4 (8.07)
Herbivores	34.0 (3.12)	54.7 (4.03)	20.5 (2.11)	156.5 (15.70)	149.2 (14.50)
Parasitoids	4.77 (0.44)	5.44 (3.44)	2.53 (0.25)	13.65 (1.23)	15.07 (1.35)
Pollinators	1.11 (0.28)	5.91 (1.01)	0.09 (0.03)	0.79 (0.34)	0.45 (0.08)
Predators	26.1 (3.04)	47.0 (2.72)	10.5 (1.42)	96.3 (9.36)	123.9 (13.60)
Sap suckers	3.55 (0.61)	4.48 (0.29)	4.37 (0.80)	31.3 (3.41)	27.4 (2.97)
Scavengers	0.4 (0.12)	1.07 (0.10)	0.18 (0.08)	2.65 (0.41)	2.91 (0.44)

Supplementary Table 4.5 Results of *lmer* models indicating chi-square (X^2) values of each of the model variables for invertebrate functional groups associated with individual trees and shrubs. Plant type (trees or shrubs) and sampling round - summer 2019/20; spring 2020 and summer 2020/21, were specified as fixed effects and plot ID as a random effect.

Functional groups	Variables	df	X^2	*p* value
Abundance of detritivores per individual plant	Individual shrubs / trees	1	35.5	0.001
	Sampling rounds	2	59.1	0.001
Abundance of herbivores per individual plant	Individual shrubs / trees	1	8.27	0.001
	Sampling rounds	2	60.6	0.001
Abundance of parasitoids per individual plant	Individual shrubs / trees	1	2.36	ns
	Sampling rounds	2	145.1	0.001
Abundance of pollinators per individual plant	Individual shrubs / trees	1	231.7	0.001
	Sampling rounds	2	1561.7	0.001
Abundance of predators per individual plant	Individual shrubs / trees	1	30.5	0.001
	Sampling rounds	2	124.9	0.001
Abundance of sap suckers per individuals plant	Individual shrubs / trees	1	4.29	0.03
	Sampling rounds	2	7.20	0.02
Abundance of scavengers per individuals plant	Individual shrubs / trees	1	17.5	0.001
	Sampling rounds	2	35.8	0.001

Supplementary Table 4.6 Results of *lmer* models indicating chi-square (X^2) values of each of the model variables for abundance of invertebrate functional groups per plot (response variable) for vegetation structure treatments. For all seven models, vegetation structure treatments and sampling round - summer 2019/20; spring 2020 and summer 2020/21, were specified as fixed effects and plot ID as a random effect..

Functional groups	**Variables**	**df**	X^2	***p* value**
Abundance of detritivores per plot	Vegetation structure treatments	2	188.9	0.001
	Sampling rounds	2	6.27	0.04
Abundance of herbivores per plot	Vegetation structure treatments	2	138.6	0.001
	Sampling rounds	2	70.7	0.001
Abundance of parasitoids per plot	Vegetation structure treatments	2	86.1	0.001
	Sampling rounds	2	15.2	0.001
Abundance of pollinators per plot	Vegetation structure treatments	2	15.1	0.001
	Sampling rounds	2	6.96	0.03
Abundance of predators per plot	Vegetation structure treatments	2	277.4	0.001
	Sampling rounds	2	6.05	0.04
Abundance of sap suckers per plot	Vegetation structure treatments	2	69.9	0.001
	Sampling rounds	2	56.7	0.001
Abundance of scavengers per plot	Vegetation structure treatments	2	52.6	0.001
	Sampling rounds	2	14.1	0.001

Supplementary Table 4.7 ANOVA results of *lmer* indicating chi-square (X^2) values of each of the model variables for abundance of invertebrate functional groups for combined datasets of individual shrubs and trees, including significance for each explanatory variables namely volume, LA (leaf area per plant) and presence of flower. Plot ID and sampling rounds were included as random effects in the models.

Functional groups (Response variables)	**Explanatory variables**	***df***	X^2	***p* value**
Abundance of detritivores per individual plant	Volume	1	4.90	0.02
	LA	1	21.5	0.001
	Presence of flowers	1	74.1	0.001
			$R^2m = 0.06; R^2c = 0.21$	
Abundance of pollinators per individual plant	Volume	1	90.7	ns
	LA	1	0.10	ns
	Presence of flowers	1	66.7	ns
			$R^2m = 0.14; R^2c = 0.20$	
Abundance of herbivores per individual plant	Volume	1	172.1	0.001
	LA	1	9.69	0.002
	Presence of flowers	1	104.7	0.001
			$R^2m = 0.19; R^2c = 0.66$	
Abundance of parasitoids per individual plant	Volume	1	80.2	0.001
	LA	1	15.2	0.001
	Presence of flowers	1	35.1	0.001
			$R^2m = 0.22; R^2c = 0.24$	
Abundance of predators per individual plant	Volume	1	53.3	0.001
	LA	1	106.9	0.001
	Presence of flowers	1	98.2	0.001
			$R^2m = 0.23; R^2c = 0.43$	
Abundance of sap suckers per individual plant	Volume	1	59.4	0.001
	LA	1	33.7	0.001
	Presence of flowers	1	59.2	0.001
			$R^2m = 0.17; R^2c = 0.53$	
Abundance of scavengers per individual plant	Volume	1	0.41	ns
	LA	1	94.7	0.001
	Presence of flowers	1	58.3	0.001
			$R^2m = 0.13; R^2c = 0.50$	

Supplementary Table 4.8 Results of *lmer* indicating chi-square (X^2) values of each of the seven model variables for abundance of invertebrate functional groups for vegetation structure treatments (plot level), including significance of explanatory variables (total volume, total LA (leaf area per plant) and no. of flowering plants) for each model. Plot ID and sampling rounds were included as random effects in the models.

Functional Group (Response variable)	**Explanatory variable**	***df***	X^2	***p* value**
Abundance of detritivores	Total volume	1	0.25	ns
	Total LA	1	37.7	0.001
	No. of flowering plants	1	85.0	0.001
			$R^2c = 0.35; R^2m=0.41$	
Abundance of pollinators	Total volume	1	0.02	ns
	Total LA	1	10.6	0.001
	No. of flowering plants	1	10.6	0.001
			$R^2c = 0.13; R^2m=0.21$	
Abundance of herbivores	Total volume	1	15.9	0.01
	Total LA	1	0.00	ns
	No. of flowering plants	1	35.3	0.001
			$R^2c = 0.21; R^2m=0.30$	
Abundance of parasitoids	Total volume	1	14.3	0.001
	Total LA	1	9.49	0.001
	No. of flowering plants	1	22.3	0.002
			$R^2c = 0.20; R^2m=0.20$	
Abundance of predators	Total volume	1	7.77	0.001
	Total LA	1	2.13	ns
	No. of flowering plants	1	93.8	0.001
			$R^2c = 0.33; R^2m=0.33$	
Abundance of sap suckers	Total volume	1	12.6	0.001
	Total LA	1	4.18	0.04
	No. of flowering plants	1	77.6	0.001
			$R^2c = 0.35; R^2m=0.44$	
Abundance of scavengers	Total volume	1	6.47	0.01
	Total LA	1	8.14	0.001
	No. of flowering plants	1	45.2	0.001
			$R^2c = 0.22; R^2m=0.25$	

CHAPTER FIVE: Thesis discussion

The overarching aim of this thesis is to increase understanding of the capacity for trees to mitigate heat and the biotic (e.g., canopy traits) and abiotic (e.g., location of the study sites, VPD) controls on this, along with their role in supporting urban biodiversity. This concluding chapter summarises results presented in the previous chapters, synthesises key findings across chapters and identifies avenues for future research.

City dwellers are increasingly exposed to extreme heat, frequently at levels that are historically unprecedented (Tuholske et al., 2021). Worldwide, the current daytime average maximum difference between urban and rural land surface temperature is 2 °C (Manoli et al., 2019). Many countries have undertaken mass tree plantation programs in an approach to adapt to rising temperatures and mitigate heat in urban landscapes. Examples of these include the million-trees planting program in Los Angeles (Pincetl et al., 2013), the 5 million trees planting program in Greater Sydney, (NSW, 2018) and the billion-trees planting program in Pakistan (IUCN, 2017). However, as it has been pointed out in the previous chapters that the ability of trees to reduce temperatures is influenced by the urban heat balance. Any change in the proportion of impervious surfaces, the size and density of buildings, wind speed, or vegetation cover significantly changes the heat balance in urban areas.

Replacement of vegetation with man-made surfaces such as asphalt with low albedo and high heat storage capability, increases urban air and surfaces temperatures (Brandani et al., 2016; Coutts et al., 2007; Mohajerani et al., 2017; Synnefa et al., 2011). The results of chapter two have shown that black and artificial grass surfaces absorb a high percentage of solar radiation, store and subsequently re-radiate heat, thereby warming the surrounding micro-climate. On 40 °C days the surface temperature of the black tiles in this study reached a staggering 82.1 °C. While my study shows the benefits of high albedo for reducing surface temperatures in urban settings, the use of black or low albedo surfaces in the form of roads and roofs is increasing across the world (Ouyang et al., 2022). In cites in the USA, pavements, roads and parking lots cover one third or more of the land surface area (Gilbert et al., 2017). For developing nations, large increases in road length are projected to improve regional connectivity (Miguel-Chinchilla et al., 2019), but given their heat absorbing materials and colour this can potentially increase local and even regional surface and air temperatures. Additionally, in recent decades, artificial grass has become popular in gardens, sports fields, schools and early learning centres (Madden et al., 2018), replacing natural grass and other types of vegetation. These changes will alter the heat balance of urban areas across the globe, with implications for the health and well-being of city-dwellers and associated energy consumption.

A study in Los Angeles by Gilbert et al. (2017), showed that an increase in albedo of 0.1 can reduce air temperature by 0.92 °C. A very similar finding (0.1 increase in albedo and a decrease of 0.92 °C for air temperature) was also reported in the review by Santamouris (2014). Furthermore, a simulation study by Zahra and Umberto (2019) reported a 2- 4 °C surface temperature reduction associated with high reflectance surfaces in Toronto, Canada during heatwave days. The instantaneous measurements of (ΔT) cooling benefits in chapter two of this thesis demonstrated that sun-exposed white surfaces have a mean temperature that is on average 18.8 °C (range 30 °C to 3.3 °C) lower than sun-exposed black surfaces. By coupling tree shade with light coloured surfaces (e.g., white) and/or natural grass, this research shows that surface temperatures in urban areas can be substantially reduced. This highlights the importance of considering the type and especially colour of materials used in the built environment, and the crucial role that tree shade can play in reducing surface, as well as air temperatures.

The amount of vegetation cover and impervious surfaces in the local environment have a direct influence on surface temperatures (Connors et al., 2012; Vo & Hu, 2021; Xiao et al., 2007). The high temperature recorded for artificial surfaces in asphalt contexts in chapter two was also in line with these results. Shaded black surfaces and artificial grass were significantly warmer even under trees planted in pavements/asphalt relative to those in nature strips and parks (Chapter Two). My explanation for this finding is that in asphalt contexts, black tiles and artificial grass absorbed and stored more heat while they were exposed to the sun compared to when these surfaces were placed in nature strips and parks contexts and took longer to re-radiate that heat in the shade. Tree species composition was quite different in parks compared to asphalt or nature strip contexts, with the former having trees with generally higher LAI. Furthermore, the impact of LAI on air and surface temperatures has been shown to vary depending on the characteristics of the surrounding landscape (Chapter Two) and Kaluarachichi et al. (2020). These findings confirm the benefits of having pervious and natural surfaces which store less heat and may also have greater soil water availability (Oke, 1989; Oke et al., 2017). This also highlights the importance of large green spaces such as parks in urban areas as they can act as a cooling oasis on hot summer days (Cheng et al., 2015). The conflicting demands for space in urban areas does, however, mean that many cities have limited amounts of greenspace (Arku et al., 2016). Therefore, understanding the role that vegetation and paved surfaces play in moderating air temperature within existing parks can help achieve optimal cooling outcomes for users and residents.

In Greater Sydney, NSW, Australia, trees were able to reduce summertime air temperatures by up to 3.8 °C (mean 1.1 °C Chapter Two; and mean 3.8 °C in the morning, Chapter Three) compared to adjacent sunlit areas. The difference between the mean temperature reduction benefits offered by trees between the chapters lies in the differences in study sites. Chapter Two covered a large geographic area ranging from inland to coastal locations, and used one-off, instantaneous measurements. Chapter Three, on the other hand focussed on a smaller, inland geographic area, and used data loggers to continuously monitor air temperatures. The review by Turner-Skoff et al. (2019) of daytime tree cooling benefits for 94 metropolitan cities worldwide, showed that trees can provide average cooling benefits of 1.9 °C with a mean maximum of 4 °C. The large cooling range indicates that appropriately selected urban vegetation has clear potential to influence air temperatures and therefore improve human thermal comfort.

A particular feature of my work was the strong diurnal pattern in how trees affect sub-canopy air temperature in Australian urban areas, demonstrated in chapter three. In the morning, trees provided mean maximum cooling benefits of up to 3.8 °C, whereas during the afternoon and at night re-radiated heat appeared to be trapped in the canopy, making local temperatures higher under tree crowns than outside. This result aligns with previous research by Wujeska-Klause et al. (2020), who also reported that tree crowns in streetscapes can trap warmed air masses at night and when windspeed is low. A longitudinal study of Kensington Gardens in London, UK, by Doick et al. (2014), on the other hand, reported night-time warming associated with tree crowns on the majority of nights, although with cooling observed on a small number of occasions. While the long-term daily average maximum temperatures of the study sites (15.8 °C in London, UK and 23.4 °C in Sydney, Australia) were quite different, both studies reported night-time warming effects of tree crowns in urban areas. In contrast, the study of Ziter et al. (2019) study in Madison, Wisconsin in the USA reported only a limited effect of tree canopies on nightime air temperature, while that of Shashua-Bar and Hoffman (2000) reported no significant canopy-related cooling at midnight in Tel Aviv, a city with a Mediterranean climate. The lack of studies from humid tropical climates that evaluated the effect of tree canopies on nighttime temperatures in urban contexts limits comparison with my study. Nonetheless, the pattern of warming or limited cooling benefits from different climate zones suggests that tree canopy cover alone is not sufficient to mitigate urban heat at nighttime. Other mitigation strategies such as the use of water bodies (blue space) or the reduction in grey surface cover will likely need to be incorporated into urban planning to minimise urban heat retention, especially during the warmest, summer months.

My study explored the microclimatic drivers of these contrasting tree-associated night-time warming and cooling effects and identified that VPD and wind speed were negatively associated with ΔT. These relationships also differed according to species identity and associated traits, influencing the extent of cooling benefits trees offers. As mentioned in previous chapters, transpiration rates increase with VPD up to a certain threshold. The whole tree transpiration measurement study by Chen et al. (2011) showed that urban trees have the capacity to limit transpiration under high VPD conditions to avoid unsustainable water loss under conditions where soil water availability is limiting. Similarly, a common garden study by Esperon-Rodriguez et al. (2021) in Western Sydney reported stomatal conductance decreased dramatically with air temperature and VPD during heatwave periods. The suburbs in my study are around 70 km from the coast, at the foot of the Blue Mountains. Due to their geographic position, the suburbs of Richmond and Cranebrook receive warm desert winds from central Australia in the afternoon and can experience particularly high afternoon summertime temperatures and high VPD. It is notable that transpiration rates of shaded leaves have been show to increase with increasing VPD (Konarska et al., 2016) whereas, sunlit leaves of some species showed decreased rates of transpiration with increasing VPD. The cooling effects of urban trees and associated green spaces are therefore influenced not only by tree traits (chapters Two and Three), but also by multiple environmental parameters specific to each location, particularly wind speed and direction, and VPD.

Impervious surfaces, in addition to reducing albedo, increase surface runoff, reduce rainwater infiltration into the ground and thus soil moisture content, and modify the water balance (Bounoua et al., 2015; Strohbach et al., 2019). Water availability affects several tree physiological processes such as evapotranspiration and thus not only the local microclimate (via evaporation) but also the cooling potential of urban trees (Konarska et al., 2016). If soil water is accessible to trees with deep roots during heatwaves, it can be used to maintain transpiration and support their associated cooling function (Drake et al., 2018). The studied years for chapters Two (2018-19) and Three (2019-2020) were exceptionally hot and experienced several heatwaves along with meteorological droughts (BOM, 2019b, 2020a). Increasing air temperatures and repeated droughts can challenge the survival and functioning of urban vegetation and make it difficult to sustain the benefits they are intended to provide (Cregg & Dix, 2001). Urban practitioners need to consider the need to irrigate urban trees, especially in streetscapes, during prolonged periods without rain and when excessive heat is forecast. Numerous studies have explored the potential of stormwater harvesting to supply street trees and other green infrastructure (e.g. storm gardens), using bio-retention and porous materials on the surfaces. These approaches can improve the water balance in urban areas and by providing additional water to urban trees, which is especially beneficial during hot and dry periods (Coutts et al., 2012; Nazarpour et al., 2023).

While my heat-related research has focussed on urban trees, it is also crucial to evaluate the combined effects of trees with other greenspace such as green walls and rain gardens, as well as blue spaces such as lakes and streams, for the studied suburbs. Green walls help to reflect solar radiation so that the building surfaces absorb less heat and provide comfortable indoor environments for residents. Blue spaces which include oceans, lakes, rivers, streams and wetlands, have been recognized for their ability to cool the local area, via evaporative cooling (Hu et al., 2020). So, there is a pressing need to understand the role that existing blue-and green- spaces play in urban cooling, while simultaneously exploring options for increasing green infrastructure, including green walls and roofs, to reduce the impact of extreme heat episodes in urban environments.

Among the studied ten tree species in Chapter Three, five were native species and the rest originated from outside Australia. Of those, the exotic species *Platanus × acerifolia* and *Pyrus calleryana* provided the greatest cooling benefits. In the field of urban planning and urban ecology, studies have argued for the inclusion of native species, and often discouraged the use of exotic species (Anderson & Minor, 2021; Kendle & Rose, 2000). Chapter Three demonstrated that, at least for the studied sites, exotic tree species were performing well in terms of modulating local micro-climate in a beneficial way. Therefore, exotic species may represent good options in urban selection proving that they are not invasive and can adapt to local micro and macro climates. However, irrespective of tree species identity, the results of Chapters Two and Three showed that LAI and canopy width were the traits that explained most of the variance in tree cooling benefits for air, black tiles and artificial grass surfaces temperatures. For white tiles, tree height and canopy depth were positively associated with cooling benefits. In chapter three, species differed significantly in their cooling benefits (i.e., ΔT values) with the highest daytime cooling observed for *Platanus × acerifolia* (3.0 °C; mean LAI 3.62 $m^2 m^{-2}$) and the lowest for *Jacaranda mimosifolia* (2.58 °C; mean LAI 1.75 $m^2 m^{-2}$). Of the native species investigated in this study, *Melaleuca quinquenervia* and *Lophostemon confertus* were associated with moderate levels of cooling, and both have relatively high LAI and canopy widths. For urban planning, aesthetics and the availability of plant material are key considerations, often resulting in streetscapes that are dominated by exotic trees including evergreen and deciduous species (White et al., 2005). However, it is crucial to select trees that are matched to local conditions, in particular native species, where such plantings are aimed at mitigating heat and increasing thermal comfort. To this end, I recommend that urban practitioners and planners should focus on natives with high LAI and wide canopies, such as *M. quinquenervia* and *L. confertus* which occur throughout the Sydney basin, although we do not have explicit data on cooling benefits associated with these particular species.

A recent study has shown that tree canopy covers of greater than 90% in an area can exert a cooling influence that extends up to 90 m beyond the canopies themselves (Ziter et al., 2019). According to an Australian urban tree assessment by Jacobs et al. (2014) the Western Sydney local government area has 20 - 30% of vegetation cover. Achieving 60 to 80% green cover is needed to lower air temperatures by 2 to 3 °C at the city level, although this is unlikely to be feasible for many cities that currently have very low vegetation cover or no trees at all (Pataki et al., 2021). However, even in the absence of high city-level vegetation cover, air temperature reductions associated with tree canopy cover at the street or park scale can still be beneficial since tree cover can increase human thermal comfort for pedestrians and residents (Erlwein & Pauleit, 2021; Li et al., 2018; Speak & Salbitano, 2022). In addition, both empirical and modelling studies have shown that appropriate placement of trees can reduce energy demands for indoor cooling (Rouhollahi et al., 2022; Santamouris et al., 2015; Santamouris et al., 2017), which feeds into issues of urban livability, especially in the context of climate change.

Urban greenspaces are multifunctional, benefiting society by providing multiple ecosystem services, such as conserving biodiversity, improving soil infiltration, carbon sequestration and cultural services along with heat mitigation (Beer et al., 2003; Lovell, 2010; Lovell & Taylor, 2013; Russo et al., 2021). Conservation of biodiversity in urban areas has received much attention mainly due to its ability to influence the well-being of urban residents and the liveability of cities (Keeler et al., 2019). The effectiveness of urban greenspaces in providing habitat for diverse fauna, including invertebrates, is strongly influenced by the type and amount of vegetation (Threlfall et al., 2017). Indeed, greater complexity in vegetation structure is associated with a greater variety of habitats and resources which in turn can help to support greater diversity and abundance of invertebrates, many of which perform important ecological services for society (Kovalenko et al., 2011; Rutten et al., 2015). Chapter Four demonstrated how the abundance and richness of invertebrates can vary among different types of vegetation structures in a common garden experiment in Richmond, Australia, and the role of vegetation structural complexity in supporting functionally diverse invertebrate communities.

Whilst 'tree only' planting strategies can support a diverse array of wildlife in urban areas (Alvey, 2006; Le Roux et al., 2014b), findings from Chapter Four showed that the diversity and abundance of invertebrates can be increased by planting shrubs on their own or in close proximitsy to trees. This finding does not therefore support the chapter's hypothesis that the diversity and abundance of invertebrates would be highest in more structurally complex (i.e tree plus shrub) plantings, relative to tree-only or shrub-only plantings. Noting that shrubs provide the biggest biodiversity gains, these should be included either on their own or alongside trees, in urban settings. Trees may, however, support more diverse invertebrate assemblages as they get bigger and more structurally complex over time. Invertebrate biodiversity brings a wide range of benefits and contributes to many different ecosystem services, such as food production via pollination and pest control, improving the quality of soil via decomposition and nutrient cycling, and maintenance of wildlife species (Losey & Vaughan, 2006). These services are in high demand in urban areas, not least because 20-30% of the world's food is produced within urban landscapes (Armar-Klemesu, 2000). Furthermore, taxonomic and functional group diversity can provide stability to these important ecosystem services in the face of changing or fluctuating environmental conditions (such as temperature extremes) (Garcia-Palacios et al., 2018; Gross et al., 2014; Srivastava & Vellend, 2005). Although the results from Chapter Four were from an experimental planting that was only three years old, differences in structure between trees and shrubs were nonetheless already clear. With maturity these structural differences – and associated differences in invertebrate communities - may change, especially as trees gain in height and canopies develop. Despite the relative young age of plantings, my research suggests that selecting plant species with wide and dense canopies and abundant flowers can support more diverse invertebrate communities for individual trees and shrubs, as well as for more complex tree-shrub plantings. Combined tree and shrub plantings can be tailored to most urban situations to promote and sustain invertebrate diversity, which in turn contributes to more functional, resilient and diverse urban landscapes, as well as enhancing the aesthetic qualities and human connection to urban greenspace (Wang et al., 2019b).

Very few studies have explored the combined effects of trees growing with shrubs and/or grasses in the context of heat mitigation. My second chapter results showed that trees growing in parks and nature strips that had grassy undergrowth provided greater cooling benefits than trees growing in asphalt surrounding. Cheung and Jim (2019) studied the effect of landscape parameters such as the cover of trees, shrubs and turf and the presence of water bodies in Hong Kong, and reported that a 50% increase in trees and shrubs can reduce daytime air temperature by 0.3 °C and 0.2 °C, respectively. Although Cheung and Jim (2019) did not explore the combined effect of trees and shrubs, their study nevertheless indicates that shrubs can contribute to the cooling effect of vegetation in urban settings. Another study in the tropical, humid city of Singapore reported that, compared to trees (- 0.6 °C), trees with a shrub understorey (-0.9 °C) provide higher cooling benefits (Richards et al., 2020). Therefore, inclusion of shrubs in greenspaces could contribute to heat mitigation as well as supporting a high level of biodiversity. However, it is also important to note that shrubs could potentially reduce air flow and thus trap heat during summer nights. The multiple co-benefits of more varied (and structurally diverse) urban plantings is an interesting topic for further research.

Given that there is always a high demand for land in urban areas, only a limited amount can be used as greenspace. Therefore, it is important that greenspace is planned in a way that can maximize biodiversity conservation and heat mitigation, as well as fulfilling the needs of the multiple stakeholders that use these spaces. The empirical evidence from this thesis shows that urban vegetation can serve the dual roles of ameliorating microclimates and providing habitat for invertebrates and associated species. This research also highlights the value of plant structural complexity and the important role that co-occurring shrubs and trees play in supporting urban biodiversity and improving ecosystem function. Understanding the role of urban vegetation in providing heat mitigation benefits, connectivity to (semi-)rural greenspace and healthy ecosystem functioning is crucial for increasing the livability of cities and promoting the coexistence of nature and urban citizens into the future.

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