Renaturing the nature strip:
Spatial, environmental and social drivers of road verge extent, composition and resident gardening behaviour

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Abstract

In this thesis I ask *What spatial, environmental and social drivers underpin road verge extent, distribution and vegetation?* I investigated road verges across 47 neighbourhoods in Melbourne, Australia, quantifying their extent and distribution and the extent and distribution of the verge gardening undertaken by residents, and I surveyed residents on their beliefs regarding the road verge and verge gardening, and characterised the flora of the road verge understorey.

Road easement green space constituted 7.0% of land cover and a high 36.7% of all public green space. The percentage of the road easement that was green space was positively correlated with date of neighbourhood development, footpath absence, social disadvantage and parcel size. Streets with a greater percentage of road easement green space were associated with residential parcels that had a greater percentage of yard (i.e. garden).

Verge gardening was common, occurring in almost a quarter (22.1%) of verges and in almost every block in every neighbourhood. I investigated two types of verge gardening, resident-planting of understorey and resident-planting of street trees. The absence of footpaths was a major driver of both. Properties with no adjacent footpath were 5.27 times more likely to have understorey verge gardening, and 2.06 times more likely to have resident-planted streets trees, than those with a footpath. Tree cut-outs (also called tree pits) were a second major driver of understorey verge gardening, 1.75 times more likely to be gardened than standard verges. Local roads were 3.74 times more likely to have understorey verge gardening than major roads. Age of street was negatively correlated with understorey verge gardening. Verges without the presence of street trees planted by local government were 1.33 times more likely to have understorey verge gardening than those with local government street trees. Social contagion was also present, with the presence of verge gardening in a neighbouring property increasing the likelihood of verge gardening by 9%.

By surveying residents, I identified cultural background, gardening enthusiasm, sense of community and level of education as significant factors differentiating respondents who planted verge understorey, who planted street trees and who did not verge garden. Normative beliefs were the main cognitive construct affecting verge gardening behaviour, with verge gardeners less likely, compared to residents who didn't verge garden, to be constrained by others' perceived disapproval of verge gardening. In particular, residents were constrained by their perceptions of local government attitudes, much more so than their perceptions of neighbours' attitudes or housemates' attitudes. Sense of community, beliefs regarding the benefits of verge gardening, and feelings for nature also had significant, but less direct, effects than normative beliefs.

Floral surveys identified 150 species, of which 82.7% were exotic, with native species mostly introduced through verge gardening. Species richness, abundance and composition were mostly driven in part by residents' verge gardening behaviour, mowing frequency, rainfall, soil compaction and canopy openness, but much variation remained unexplained and was likely to be due to stochastic factors such as degree and frequency of disturbance. Seven vegetation communities were identified, distinguished by the presence of garden plants, rhizomatous turfgrasses, and the relative proportions of three dominant grasses.

The extent of the road verge, combined with its often city-wide distribution, makes the road verge a green space component of fundamental importance to our urban ecosystems. Its varying distribution and extent across neighbourhoods means its significance also varies across the urban area. Verge gardening increased the overall species richness of verges, doubled the number of native species, and introduced structural complexity, suggesting that verge gardening can significantly contribute to quality and complexity of urban greening through the summed effect of the many small acts of citizen greening. Verge gardening promotes further verge gardening in a positive feedback loop. The influence of footpaths, road type and tree cut-outs shows that urban design can encourage this resident greening of public space. Municipal authorities are well-positioned to lead change, through reframing policy and outreach in order to positively frame verge gardening as an acceptable practice, by increasing plantings in the verges they maintain, and by promoting alternative low-mow practices that reduce the normative position of the well-manicured lawn. Planners, landscape architects, urban foresters, engineers and ecologists should work together to reimagine the ecological and greening roles of existing and future road easements. The potential for road easement green space to provide for the biodiversity, ecosystem function and human amenity now being demanded from urban green spaces is much greater than previously thought.

Declaration

This is to certify that:

a) The thesis comprises only my original work towards the degree of the Doctor of Philosophy except where indicated in the Preface;

b) Due acknowledgement has been made in the text to all other material used;

c) The thesis is fewer than 100,000 words in length, exclusive of tables, maps, bibliographies and appendices.

Signed**:**

(Adrian Marshall)

Melbourne, 4 March 2020

Preface

Thesis with Publication

This is a 'thesis with publication', i.e. a thesis that includes 'in progress' or published material (The University of Melbourne, 2019). In this thesis with publication, Chapters 3–6 are presented in format for publication in peer-reviewed journals, each with their own detailed Introduction, Methodology, Results, Discussion and Conclusion. Consequently, there is some repetition, especially within Methodology sections. Chapters 3–6 reflect the style guidelines of the journals they are intended for (Table 1) and include my supervisors as co-authors. This thesis with publication is submitted in conjunction with the requisite *Declaration of thesis with publication*, and *Co-author authorisations*, which document myself to be the primary author, responsible for the planning, execution and preparation of the work for publication. Chapters 3 and 4, which have been published, are presented in their accepted format. This thesis with publication also includes the requisite Literature Review that details the research question and places it in an appropriate context. No third-party editorial assistance was provided in preparation of this thesis.

Table i: Thesis chapters, target journals and acceptance status.

Chapter	Title	Journal and status
3	From little things: More than a third of public green space is road verge	Published by *Urban Forestry and Urban Greening* on 2 August 2019
4	Footpaths, tree cut-outs and social contagion drive citizen greening in the road verge	Published by *Urban Forestry and Urban Greening* on 6 August 2019
5	Mowers and growers: road verge gardening is strongly influenced by social norms	Accepted for publication in *Landscape and Urban Planning* on 27 February 2020
6	Urban road verge vegetation is driven by verge gardening, mowing, environmental, climactic and stochastic factors	Not yet submitted for publication

Author contributions

I undertook all conceptualisation of the studies undertaken, development of methods, planning, fieldwork, analysis of data, and development of figures and tables. I wrote all sections of all drafts, with Dr Margaret Grose and A/Prof. Nick Williams editing and assisting with development of ideas.

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To the staff and PhD cohorts of both the Melbourne School of Design and the School of Ecosystem and Forest Sciences at the University of Melbourne. To Dr Sue Finch and Dr Ian Gordon from the Statistical Consulting Centre. To Jillian Walliss for sage advice. To Dr Peter May, for his understanding of local government tree planting palettes; to John Raynor, Dr Sue Murphy, John Delpratt, Dr Sasha Andriwicz and Prof. Roger Cousens for plant identification; to Ben Smith and A/Prof. Craig Nitschke for loan of hemispherical photography equipment and assistance with Gap Light Analyser software; and to Dr Amy Shaw for advice on survey design, Scott Campbell for graphic design oversight, and Brett Lockwood for survey proofreading. Thanks, too, to the staff of the ten local government areas who provided GIS and street tree data, and to all the residents who took the time to answer my survey. To Marg Ryan, for her support. And to members of UrBEC who provided a collegial background of intellectual rigour.

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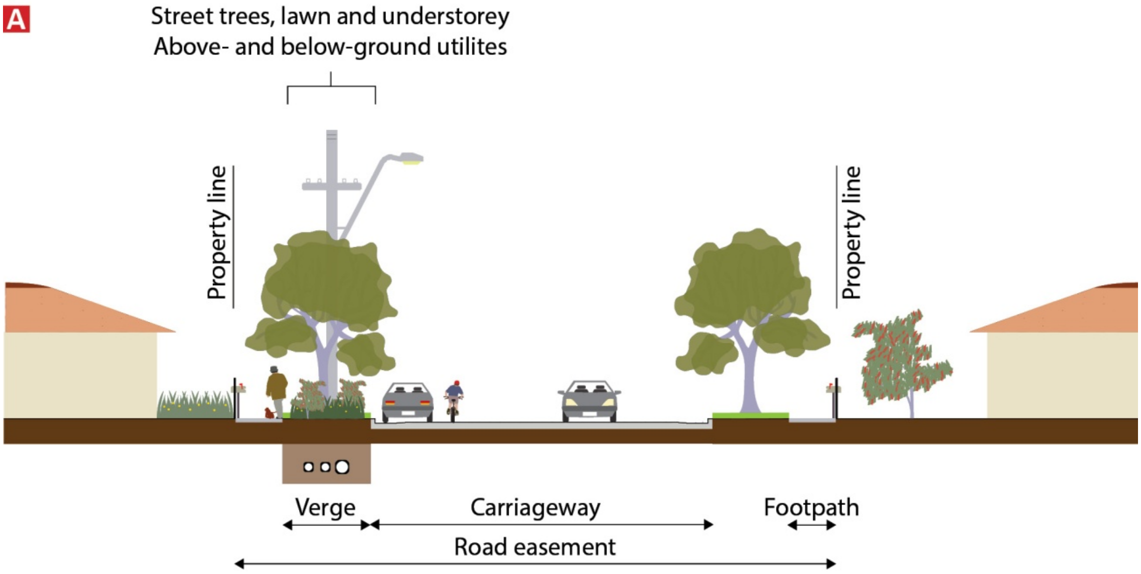
Chapter 1
Introduction and significance

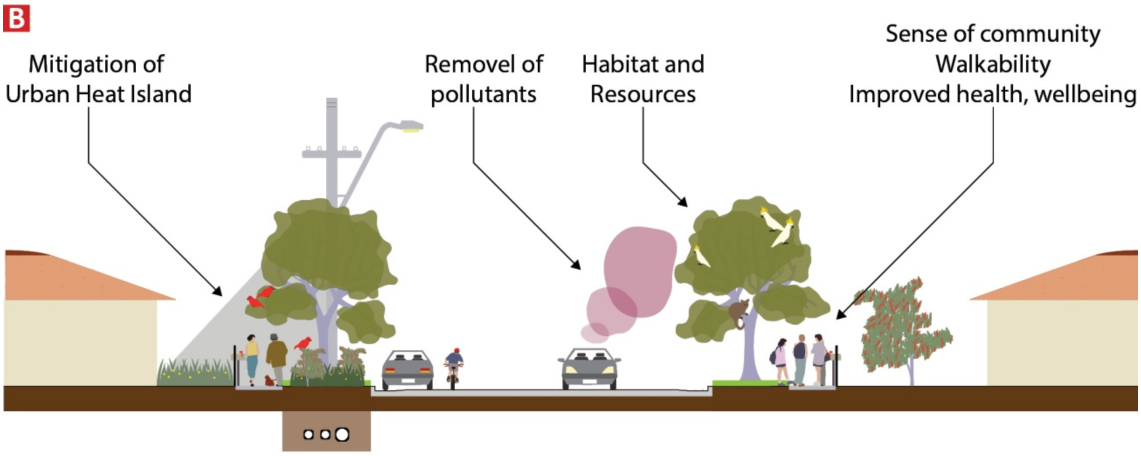
There are increasing calls to maximise the biodiversity, ecosystem function and human amenity benefits that flow from urban green space (Standish, Hobbs and Miller, 2013; Haaland and van den Bosch, 2015; Shanahan *et al*., 2015). Increasing population densities are putting greater pressure on existing biodiversity and urban green spaces (Fuller and Gaston, 2009; Haaland and van den Bosch, 2015) and urban areas need to become more resilient to meet the challenges of climate change (Fünfgeld and McEvoy, 2012).

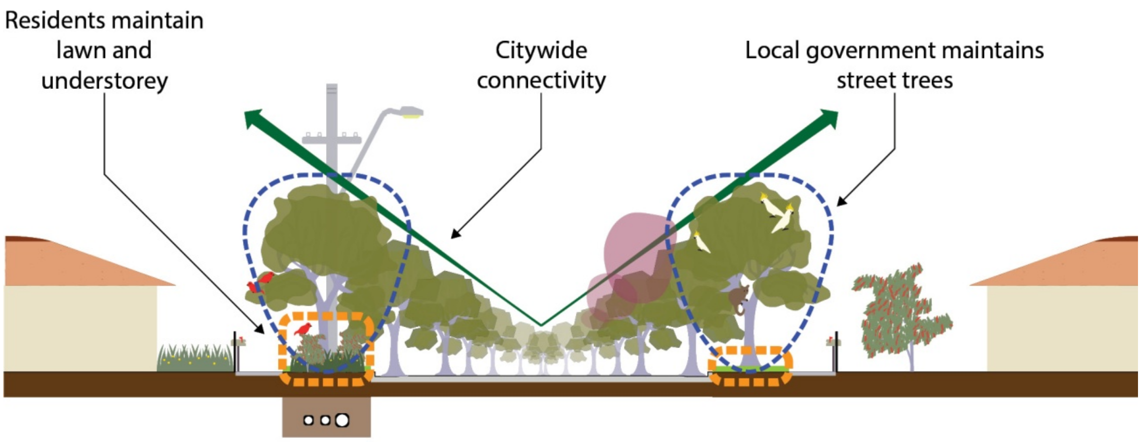
Road verges (also termed the nature strip in Australia and, variously, tree lawn, parking strip or sidewalk buffer in North America) are a component of the ubiquitous and complex type of urban green space that is the road easement, which also includes the carriageway, footpaths (sidewalks) and utilities (Figure 1A).

Road verges are likely to be large in combined extent (Richards *et al.*, 1984). That extent, together with their often broad distribution within cities, means they are likely to play an important role in urban ecosystem processes. For instance, street greenery provides habitat and resources for fauna (Schaffers, Raemakers and Sýkora, 2012), provides connectivity between patches of larger habitat such as parks (Oprea *et al*. 2009), regulates temperature (Sanusi *et al.*, 2015); mitigates stormwater runoff (Armson, Stringer and Ennos, 2013); and mitigates air pollution (Leonard, McArthur and Hochuli, 2016). Road verges can include remnant vegetation, including endangered species and vegetation communities (Lorimer 2006; McDougall 1987). Road verges also contribute to human wellbeing. For instance, street greenery improves human health through reduction in stress and improvements in social cohesion (De Vries *et al.*, 2013), and reduces early childhood asthma (Lovasi *et al.*, 2008) and depression (Taylor *et al.*, 2015). Street greenery improves the walkability of neighbourhoods (Lu, Sarkar and Xiao, 2018), which can reduce obesity and promote physical health (Creatore *et al.*, 2016). (Figure 1B.)

Figure 1.1 (Following page): **A:** Elements of the road easement. The road easement extends from property line to property line and includes the carriageway, footpaths and verges. The road verge is the location for street greenery, including street trees, lawn and understorey, as well as above-ground and below-ground utilities. **B:** Benefits of street greenery include mitigation of the Urban Heat Island, removal of air pollution, and provision of habitat and resources for a wide range of species. Street greenery also creates a sense of community, and improves the walkability of a neighbourhood, with consequent benefits to human health and wellbeing, such as decreased obesity. **C:** The management of the road verge in Australia is generally shared between residents and local government, with residents responsible for maintaining the lawn and any understorey, and local government responsible for the street trees. Because street greenery is ubiquitous and often has broad distribution within cities, it provides connectivity between patches of larger habitat, such as parks and river corridors.


A
Street trees, lawn and understorey
Above- and below-ground utilites
Property line
Property line
Verge
Carriageway
Footpath
Road easement


B
Mitigation of
Urban Heat Island
Removel of
pollutants
Habitat and
Resources
Sense of community
Walkability
Improved health, wellbeing


Residents maintain
lawn and
understorey
Citywide
connectivity
Local government maintains
street trees


A
B
C
D
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K
L

Figure 1.2 (Preceeding page). Examples of road verges and verge gardening found in in Melbourne, Australia. A: Emphasising sustainability through recycled materials. B: A standard or typical road verge with the verge bounded by the kerb, footpath and driveways. C: A verge where the absence of footpath and no fence combined to allow an uninterrupted extension of a front yard to the kerb. D: Elliston Estate, Rosanna, Melbourne, where a sense of community and shared landscape was created in part through the deliberate absence of footpaths. E: A tree cut-out with understorey planted by residents. F: Verge gardening where the grass has been replaced by gravel. G: A very wide verge providing space for future widening of the carriageway. H: A long uninterrupted verge beside the back and side fences of the properties of an inward-facing housing estate. I: A well-vegetated outstand for traffic calming. J: A road closure transforms the verge into a small local park. K: Planter boxes make a statement in a streetscape with only tree cut-outs and small front yards to provide greenery. L: With the footpath moved slightly towards the kerb, a verge space between the footpath and the fenceline makes itself more appealing to garden than the verge between footpath and kerb. All photos by author except for D, copyright Victorian Heritage Database.

Road verges have been generally overlooked in assessments of urban greenspace (O'Sullivan *et al.*, 2017), and there are many gaps in the peer-reviewed literature. The extent of the road verge has only been the subject of one study (Richards *et al.*, 1984), hence their relative contribution to urban greenspace compared to other major components of urban greenspace such as parks and yards is unknown. The distribution of the road verge has not been studied. Little is known about people's beliefs about and perceptions of this complex public/private space, and how those beliefs and perceptions affect their management of road verges (Weber, Kowarik and Säumel, 2014). The social, demographic and urban form drivers of verge gardening are unstudied, and it is not known how common this behaviour is or what we can do to promote it, though there is substantial parallel literature on front and back yard gardening. Though street trees have been the subject of much research, the understorey and lawn (or mown amenity grass) flora of road verges has only been investigated as part of one study in New Zealand (Stewart *et al.*, 2009).

This thesis therefore seeks answers to the question *What spatial, environmental and social drivers underpin road verge extent, distribution and vegetation?* In doing so, it seeks to fill important gaps in our understanding of the extent and distribution of road verges, the drivers of the citizen-led greening activity that is verge gardening, residents' beliefs regarding road verges and their management, and the consequent composition of the flora of the road verge understorey, which is driven in part by residents' verge gardening and maintenance activities. By investigating road verges across 47 neighbourhoods in the sprawling, suburban city of

Melbourne, Australia, this thesis aims to collect data essential for developing policies to effectively manage road verges as a significant greenspace component in a complex socio-ecological urban environment. This research is transdisciplinary, positioned across landscape architecture, urban design, urban ecology and environmental psychology to provide insights not possible through investigations restricted to a single lens.

Aims

This thesis sought to answer the question: *What spatial, environmental and social drivers underpin road verge extent, distribution and vegetation?*

Specifically, this thesis aimed to:

- Quantify the extent and distribution of road verges, and to investigate the relationships extent and distribution have with urban form, land use, date of neighbourhood development and demographic characteristics at the neighbourhood scale (Chapter 3);
- Quantify the extent of other public green space and the extent and distribution of private residential yards to compare these major green space categories with road easement green space (Chapter 3);
- Quantify the extent and distribution of verge gardening, and investigate the relationship that urban form and demographic characteristics at the neighbourhood, street and property scales have on that extent and distribution (Chapter 4);
- Test for the presence of spatial contagion (the increased likelihood of verge gardening between neighbours) (Chapter 4);
- Investigate the relationships between resident's beliefs regarding road verges and verge gardening, and demographic data and the neighbourhood, street and property scales (Chapter 5);
- Characterise the diversity, abundance and floristic composition of road verge understorey, and to investigate the influences on that diversity, abundance and composition of climatic and other environmental factors, and residents' management and gardening practices (Chapter 6); and
- Articulate pathways through which we may increase the biodiversity, ecosystem function and human amenity benefits flowing from road verges (Chapter 7).

Methodology

Because this is a thesis with publication, each data chapter (chapters 3-6) includes a detailed methodology. This section therefore provides only an abbreviated overview of the study methodology.

The study area

The study area was the western and northern residential suburbs of Melbourne, Australia, which occur in the Victorian Volcanic Plain (VVP) bioregion, an area with distinct soils and native flora to the rest of Melbourne. Greater Melbourne (latitude -37.814, longitude 144.96332) is located in south-eastern Australia, has a population of 5 million people (Australian Bureau of Statistics, 2018) and is the capital of the state of the State of Victoria (Figures 1.3).

The Australian Bureau of Statistics (ABS) geographical unit 'Statistical Area 1' (SA1) (Australian Bureau of Statistics, 2016) was chosen as a study unit because SA1s are generally homogenous in street layout, housing stock, development date, recognise urban historical and landform boundaries, and their use allows direct comparison with existing ABS demographic data. SA1s have an average population of approximately 400 people. Forty-seven SA1s were randomly selected that were at least partially zoned residential and entirely within both the Urban Growth Boundary of Greater Metropolitan Melbourne and also the Victorian Volcanic Plain (Figure 1.4). These were predominantly residential, but also included other land uses such as schools, commercial strips, parks, sports fields and churches. The term 'neighbourhood' is hereafter used to refer to an SA1.

Vegetated road verges occur across the city, although in suburbs built prior to World War 1 road verges are less common, with tree cut-outs (rectangles usually 1–4 m^2 cut from the footpath) the most usual way of providing green space within the streetscape. Usually road verges are 0.5-2 metres wide though they can be far wider, e.g. if major gas lines are present underground or if land is set aside for future road widening. The road easement includes greenspace other than road verges, e.g. medians, traffic islands, roundabouts, outstands to calm traffic, and planters. Figure 4 and Appendix 1 provide examples.

Both residents and government authorities manage Melbourne's road verges. Residents maintain the lawn, usually by mowing, and understorey, and government authorities look after the trees. Verge gardening is regulated by local government guidelines, but these vary greatly. For instance, plantings may or may not be allowed depending on if they are native, woody, weedy, or edible, or simply not allowed at all. Residents' verge gardening often contravenes these regulations.

Figure 1.3: Melbourne is the southernmost state capital of mainland Australia.

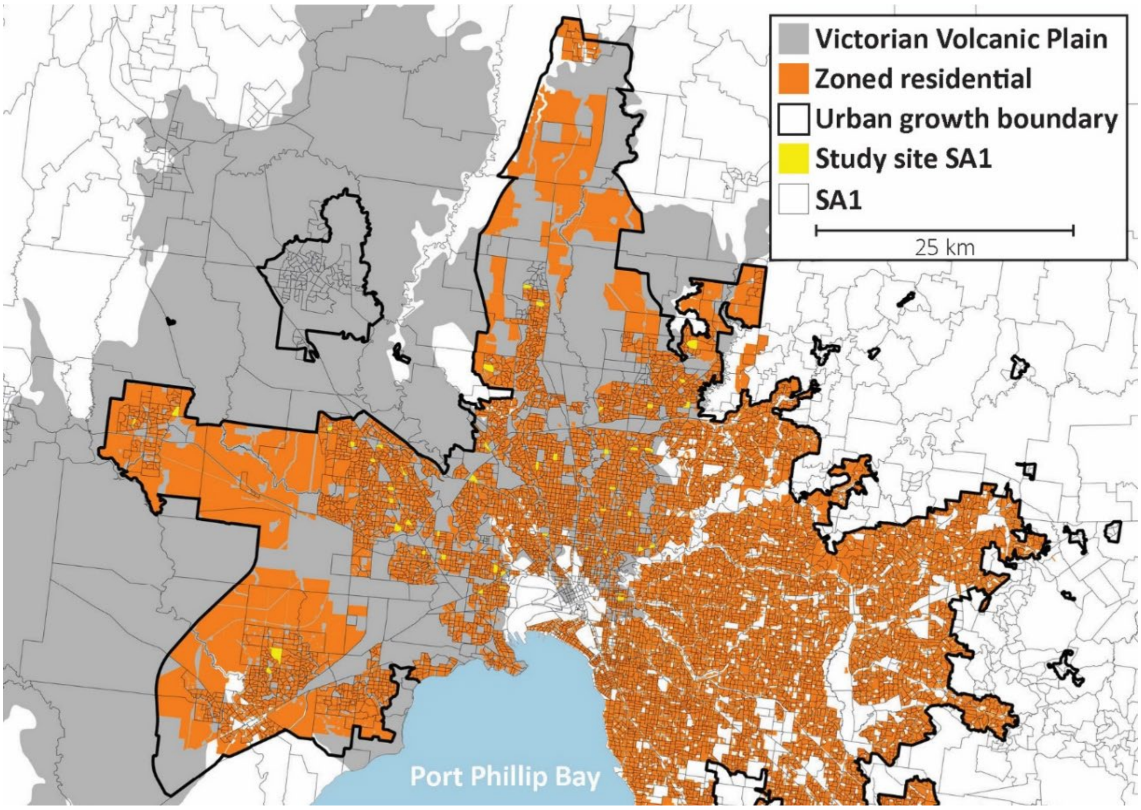

Figure 1.4: The relationship between the Urban Growth Boundary, Victorian Volcanic Plain, urban land zoned residential and the 47 neighbourhoods (SA1s) comprising the study area in Melbourne, Australia.

Data collection and analysis

Chapter 3 used GIS data relating to ground-surface permeability to water, as well as transport and cadastral data, in combination with aerial photography, to identify and quantify the extent of three types of green space – road easement greenspace, other public greenspace and private greenspace. It used multivariate regression to test for correlations between those greenspace types and a range of spatial and social factors including development date, residential parcel size, the Index of Relative Social Disadvantage, footpath, cul-de-sac, road type, and road density.

Chapter 4 used a physical field survey to collect property-level data for 5151 single-residence properties in the study area, including the presence of two type of verge gardening – resident-planting of understorey and resident-planting of street trees. That data was combined with data from Chapter 3 to investigate the extent of verge gardening. Because data was nested at the property, street and neighbourhood level, Generalised Linear Mixed Modelling (GLMM) was used to test the distribution of verge gardening for correlation with urban form and demographic factors.

Chapter 5 took an environmental psychology approach to develop print and on-line questionnaires. Salient issues were identified through the literature and an elicitation study. The questionnaire included questions on demographics, knowledge of the law, verge gardening and maintenance practices. It included established psychometric scales – the Nature Relatedness Scale and the Sense of Community Scale – and question sets related to particular aspects of residents' beliefs, such as their normative and behavioural beliefs. It used a range of methods to maximise survey response, including attention to physical design and hand-delivering to all 5151 properties. Data was analysed using standard statistical tests such as Chi-squared tests, as well as through Structural Equation Modelling.

Chapter 6 used a quadrat-based flora survey of a 158-verge subset of the road verges of the respondents to Chapter 5's questionnaire. Verges were stratified by time since development, and 536 quadrats investigated. ANOVA, all subsets regression and ordination through principle coordinate analysis (PCO) were used to compare species diversity, abundance and composition with a range of social and environmental factors.

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Chapter 2
The road verge in the urban socio-ecological system: A literature review

Introduction: The purpose of this literature review

This chapter reviews the literature relevant to the potential of road verges to contribute to urban biodiversity, ecosystem function, and human amenity. The significance of urbanisation and its effects is discussed, the road verge framed as part of an urban socio-ecological system, and the extent, distribution and benefits provided by urban green space and the road verge in particular is examined. A discussion of the spatial structure and vegetation composition of urban green space follows, including the flora of lawns, a major component of road verge vegetation. Also discussed is human interactions with urban green space, the role of perceptions, gardening practices and their motivations. Together, these areas of research provide a framework for a theoretical positioning of the role of the road verge in the urban socio-ecological system and its consequent potential to contribute to urban biodiversity, ecosystem function, and human amenity. Finally, gaps in the literature that have led to the research undertaken as part of this thesis are summarised.

The effects of urbanisation

The world is rapidly urbanising. Most population growth is occurring in urban areas (United Nations, 2014), 55% of people live in cities (United Nations, 2014), and 3% of earth's surface is now urban (Seto, Sánchez-Rodríguez and Fragkias, 2010; Zhou *et al*., 2015). Among the many consequences of this urbanisation is a reduction in vegetation cover, increase in impermeable surfaces, alterations to hydrology, increased pollution and waste, fragmentation, and changes to species composition (Parris, 2016). These effects are exacerbated by the fact that urbanisation often occurs in areas of high biodiversity (Ives *et al*., 2016; Threlfall and Kendal, 2018). The world's cities are home to and refuge for many threatened species (e.g. Schwartz *et al*. 2002; Rebelo *et al*. 2011) and urbanisation leads to local losses in that native biodiversity (Kühn, Brandl and Klotz, 2004; Seto, Guneralp and Hutyra, 2012). Nevertheless, it is important to recognise that cities meet many of our collective needs in a very efficient manner, and the "footprint" of human civilisation would be much greater if there were no cities at all (Grimm *et al*., 2008; Wu, 2010).

Impermeable surfaces increase as urbanisation increases (Angel *et al.*, 2005), causing changes in local climate such as the urban heat island effect. They also alter urban hydrology, causing loss of aquatic environment, especially wetlands and associated habitat and resources, increased runoff, flooding, scouring or creeks, loss of base water flow, lack of aquifer recharge, and a range of other effects together described as 'urban stream syndrome' (Walsh *et al.*, 2005).

Green space becomes increasingly fragmented, patches of green space become smaller, and the patch ratio of interior to edge increases creating heightened edge effects such as weed invasion, temperature and wind change, and changes to light and noise levels (Cadenasso, Pickett and Schwarz, 2007), with consequent effects. For instance, invasion by exotic plant species can alter the physical structure of native vegetation communities, causing the loss of habitat for flora and fauna (Hadden, 1995), completely replace vegetation communities (Williams, McDonnell and Seager, 2005), alter biogeochemical cycles, disturbance regimes and create difficulties for the management of urban green spaces (Strayer *et al.*, 2006). Fragmentation caused by urbanisation affects species' populations dynamics through altered landscape connectivity, changed predation, access to nutrients, with consequent effects of population genetics (Munshi-South, 2012).

Urbanisation causes distinct changes in biodiversity and species abundance and composition. Urbanisation is associated with an increased proportion of exotic flora species (McDonnell and Hahs, 2008; Yan *et al.*, 2019), and the use of exotic plants within cities is considered to be the greatest movement of species globally (Niinemets and Penuelas, 2008). However, the increased species richness of flora does not equate with increased phylogenetic richness (Knapp *et al.*, 2012), meaning that introduced species – generally ornamental species in domestic yards (i.e. gardens) – tend to come from a narrow range of families compared to more natural ecosystems.

Globally, the rural edge within urban areas often has greater habitat diversity, vegetation diversity and higher plant species richness (Pyšek, 1995; Luck and Smallbone, 2010), and floral species richness varies across all types of urban green space, for instance residential yards, golf courses, remnant vegetation and urban parks all have different species richness and distinct species composition (Threlfall, Ossola, *et al.*, 2016). Urbanisation selects for certain species functional traits, for instance the method of seed dispersal changes, seed weight tends to decrease, annuals tend to increase and perennials decrease, specific leaf area increases, and there is an increase in self-pollinated species (Johnson and Swan, 2014). Clonal plants are favoured because they have a reliable reproduction method and are successful at dispersing over short distances, while in Australia, studies have shown that annual plants are highly tolerant of anthropogenic disturbance and can complete their lifecycle in a favourable temporal niche (Zeeman, 2017). The effects of urbanisation on vegetation cover vary as development proceeds and according to the biome (Wu, Liang and Li, 2019), for instance increasing cover in less-productive biomes such as shrubland until a certain level of urbanisation is reached after which cover declines, while in tropical biomes urbanisation is associated with decreased cover.

Fauna species richness and diversity decline in urban systems, with urbanisation favouring species that are 'urban adaptors' – generalist rather than specialist species (McIntyre *et al.*, 2001). Mammal populations are particularly affected by urban fragmentation (FitzGibbon, Putland and Goldizen, 2007). Bird populations change, with increases in species that nest in cavities, or are granivores or omnivores rather than insectivores, along with increases in total bird biomass (White *et al.*, 2005; Chace and Walsh, 2006). These effects are not necessarily linear. For instance intermediate levels of urbanisation seem to support the greatest bird species richness, perhaps because at intermediate urbanisation the diversity of edge habitats in the environment is at its greatest (Marzluff, 2008). Arthropods also increase in abundance, their numbers no longer limited by resources but by bird predation (Faeth *et al.*, 2005). In general, predation pressure decreases in urbanised areas, changing species compositions (Béla, Magura and Lövei, 2018).

For both flora and fauna, rapid land use change means species that are rapid colonisers are favoured (Alberti *et al.*, 2003). Generally, cities are considered to be homogenising influences on global biodiversity (McKinney, 2008). Light pollution associated with urbanisation affects many taxa including humans, influencing foraging, physiology, reproduction, movement and communication (Swaddle *et al.*, 2015).

Why is it important to consider road verges as part of a socio-ecological system?

The road verge provides a significant exemplar of the inextricable link between human actions and ecological outcomes in urban systems. A socio-ecological system is a 'bio-geo-physical unit and its associated social actors and institutions.' (Glaser *et al.*, 2008). In urban ecology, cities are viewed as complex, dynamically and adaptively changing, heterogeneous socio-ecological systems, in which humans and ecosystems are linked at multiple scales (Grimm *et al.*, 2000; Pickett *et al.*, 2001; Pickett, Cadenasso and Grove, 2005). As early as 1984, Sanders (1984) proposed that urban morphology, natural factors, and human management systems determined the distribution of the urban forest, and in 1990 McDonell and Pickett (1990) discussed how humans cause change along urbanisation gradients including changes in disturbance regimes, biota, landscape structure, physiological stresses such as air pollution, and cultural, economic and political factors. Alberti *et al* (2003) have argued that attempts to study humans and ecology in urban systems separately are reductionist and that integrative approaches are essential. Hunter and Luck (2015) have argued for a new typology of urban green space that combines ecological and social values in a way that aids cross-disciplinary studies.

The biodiversity, ecosystem function and human amenity contributions of road verges are a product many factors, including land-use, development history, climate, geology, demographics, resident behaviour, policy and law, standards, cultural symbolism, disturbance processes and management actions. The social and ecological benefits provided by road verges need to be seen in comparison to, and in conjunction with, those provided by other urban green space types. Only then can we begin to discuss the best ways to maximise the roles road verges can play within an urban socio-ecological system.

History

Research is generally lacking on the complex history of the road verge and this thesis does not attempt to fill that gap in our knowledge. However, it is likely that the road verge as a distinctive feature of suburban development was first formalised in the Garden City Movement (Howard, 1898). Howard's cooperatively owned, planned cities were intended for social revolution, with environmental values going hand-in hand with social values. By attempting to make a park-like environment, Garden Cities established the road verge as a distinctive element of the streetscape. Some of the ideas of the Garden Cities and Garden Suburbs movements were exported to Australia prior to the First World War. The first mention of a 'nature strip', the distinctive Australian term for a road verge, occurs in the Australian literature as a 'natural strip' and it is made by Charles Reade (1880–1933), the prominent advocate of garden cities, speaking on his 1914–1915 Australasian Town Planning Tour ('How to make Albury a Garden City: Scientific lecture by Mr Reads: Beautifying Western Hill, the Cemetery, and Lagoons,' 1915). The exact term 'nature strip' first appears 12 years later, in a description of Linden Court, a small development along garden city lines in St Kilda (*Morning Bulletin (Rockhampton, Qld. : 1878 - 1954)*, 1927). In Australia, the earliest of such planned nature strips can probably be found in Daceyville, Sydney, construction of which started in 1912 (Freestone, 2000). From then on, the nature strip became a ubiquitous element of the residential housing landscape.

The presence of road verges thus depends on the age of the neighbourhood. In the European context this is especially important, given the greater age of urban areas compared to the USA and Australia. Road verges are rare in city areas dating from the nineteenth century or earlier.

Until 1941 in Australia, all road verges were the responsibility of council to maintain. The Mercury in Tasmania carries the first mention of residents being responsible for their road verges (*The Mercury (Hobart, Tas. : 1860 - 1954)*, 1941). This devolution of responsibility from council to resident continued through the 1950s. Unlike domestic gardens, and despite being public land, road verges are generally managed by both residents and government authorities in Australia, Britain, Canada and the USA, though the distribution of responsibility varies from jurisdiction to jurisdiction locally and internationally. For instance, in Australia, it is generally the case that residents are required to maintain the lawn (which they do generally by mowing) and any understorey, while municipal authorities maintain the trees. In many cities in the USA, e.g. Portland, Oregon (Donovan and Butry, 2010), residents are required to maintain both the understorey and the trees, and in Europe the verge is often managed only by the municipality, though more complex public–private relationships also exist in more recent suburban developments.

In Australia, any gardening by residents in the road verge is subject to municipal government guidelines. These guidelines vary enormously; for instance, plantings may be not permitted at all, or may be restricted by height, woodiness, weediness, edibility, or provenance (indigenous, native, exotic). Residents who go beyond mowing of the road verge and expand their activities to gardening in the road verge often do so in contravention of municipal government guidelines and the motivations for doing this are unclear.

Planning

Planning controls that regulate elements of developments, such as the dimensions of the carriageway (road surface), sidewalks, crossovers (driveways crossing from the property line to the kerb), and the positioning of utilities, have changed over time, suggesting that the distribution of road easement green space might vary with the date of urban development across a city (Austroads 2009; Standards Australia 2013). And traffic calming strategies have also changed the form of the road easement, introducing roundabouts, median strips, chicanes and vegetated outstands (Van Schagen, 2003).

Cul-de-sacs are a common and often defining feature of modern suburbia that are also likely to influence the extent and distribution of urban green space. The housing estate style characterised by cul-de-sacs, which is known as 'Radburn' style after the design of a 1929 estate in Radburn, New Jersey, was designed to a) fit more parcels of land on topographically irregular development areas, b) minimise road length per residential parcel of land thus saving costs on roads and utilities, and c) do so by using a non-hierarchical road system where there are many low-traffic local roads and only a few high-traffic collector roads (Stein, 1957). A number of consequences for urban green space follow from this. Firstly, housing parcels tend to be irregular in shape, which has been shown to decrease the amount of vegetation in a development (Nielsen and Jensen, 2015). Secondly, the amount of road verge should tend to be less because a greater proportion of the frontage of a cul-de-sac property will be devoted to driveway (Stone, 2004). Thirdly, the proportions of major to minor roads between Radburn-style developments and grid-style developments will vary and the width of these roads and their easements are also likely to influence the distribution of road verge. Fourthly, because cul-de-sac developments are better able to respond to variations in topography, they allow development to occur in places where otherwise development would not, such as at the steep edges of ravines, thus intruding into land that might well otherwise be set aside as nature reserve (Gordon and Tamminga, 2002).

The distribution of urban green space is also influenced by the ability of authorities to acquire land. In new developments on the urban fringe the acquisition of land is often a legislated necessity, e.g. in Melbourne, Australia, 10% of the unencumbered land available for new development must be set aside as public open space (VPA, 2017), though there are generally few specifications as to what sort of land should be set aside, e.g. in terms of its conservation significance. Similarly, planning regulations often specify the maximum proportion of a land parcel that can be covered in impermeable surfaces, and set the proportions of the elements within the road easement, thus influencing the amount of road verge. In already developed urban areas, where acquisition of land is affected by both the availability of land and by its cost, which is often much higher than land on the urban fringe (Boulton, Dedekorkut-Howes and Byrne, 2018), the quality and size of newly acquired green space will be very different.

The benefits, extent and distribution of urban green space

Benefits

One of the most influential frameworks used in recent decades for discussing the benefits of urban greenspace is the concept of ecosystem services. Ecosystem services first came to prominence in the United Nations' Millennium Ecosystem Assessment (2005), and are considered benefits to humans derived from ecosystems (Berghöfer *et al.*, 2011). Ecosystem services are generally classified into four groups: supporting, provisioning, regulating and cultural. Supporting services underlie the other services and include the creation of soil, nutrient recycling, primary production, pollination and habitat provision, and biodiversity is considered essential for supporting services. Provisioning services include the production of food, raw material and energy. Regulating services include climate regulation, purification, waste decomposition and pest and disease control. Cultural services include spiritual, aesthetic, recreational, therapeutic and educational services.

The benefits urban green space provides vary according to the type of urban green space. For instance, in terms of habitat provision, large areas of semi-natural habitat can provide space for species sensitive to the effects of urbanisation (Beninde, Veith and Hochkirch, 2015; Lepczyk *et al.*, 2017); ornamental ponds can provide important habitat for a range of aquatic species and their habitat provision is largely independent of size (Oertli *et al.*, 2002); and industrial wastelands can support considerable species diversity because of the range of micro-habitats present and varying levels of management and succession (Harrison and Davies, 2002; Kattwinkel, Biedermann and Kleyer, 2011; Fischer *et al.*, 2013; Bonthoux *et al.*, 2014; Hunter, 2014; Mathey *et al.*, 2015). In the case of road verges, the lawn (or mown amenity grass) and understorey of the road verge provide habitat and resources for many species including arthropods (Leonard, McArthur and Hochuli, 2018), nematodes (Cheng *et al.*, 2008), ants (Pećarević, Danoff-Burg and Dunn, 2010) and soil microbes (Beard and Green, 1994), and urban road verges can harbour critically endangered species (Stuwe, 1986; Kirkpatrick, McDougall and Hyde, 1995).

The urban forest, which provides a broad range of benefits, is to a significant extent composed of street trees. For instance, in Chicago street trees account for 24% of the urban forest by leaf surface area (McPherson *et al.*, 1997), in Canberra street trees provide 20% more crown area than park trees (Brack, 2002), and in Santa Monica, 87% of public trees are street trees (Xiao and McPherson, 2002).

Urban green space provides resources for pollinators. Depending on the extent of development, urban areas can support bee populations better than natural environments (Tonietto *et al.*, 2011; Carper *et al.*, 2014; Larson, Kesheimer and Potter, 2014), with bee abundance higher and species richness unchanged. This is likely to be due to greater floral resources present in suburbanised urban areas as well as the range of habitats available for nesting. Butterflies similarly find support through urban green space, their abundance and diversity positively correlated with yard space and negatively correlated with canopy cover (Matteson and Langellotto, 2010), Roadside vegetation has also been found to be important butterfly habitat (Saarinen *et al.*, 2005; Schaffers, Raemakers and Sýkora, 2012). In addition, insecticide use may be lower in urban areas than in surrounding agricultural areas that have seen massive declines in insect numbers (Hallmann *et al.*, 2017). Road verges are therefore likely to be an important resource for pollinators, though factors such as their different frequencies of mowing, and their proportional extent of canopy, compared to other forms of urban greenspace such as yards and parks, will influence their relative importance in providing resources for pollinators.

The urban heat island (UHI) is a phenomenon whereby temperatures a higher in the city compared to surrounding areas, and it is caused by hard surfaces that trap heat (Gillner *et al*., 2015). Increased urban temperatures affect human health, with an Australian study showing the number of heat-related deaths greater than all other natural hazards combined (Coates *et al*., 2014). Urban green space helps mitigate such harmful consequences of UHI, and the influence of street trees is significant (Norton, Evans and Warren, 2016). Moreover, the efficiency of street trees in providing cooling effects is influenced by street orientation (Sanusi *et al*., 2015). Urban green space has more indirect effects on climate regulation too. For instance, trees and other woody plants provide carbon sequestration, helping to alleviate the effects of climate change (Roy, Byrne and Pickering, 2012) while lawn is a net emitter of greenhouse gasses (Gu *et al*., 2015).

Trees and roadside vegetation purify air, trapping particulate pollutants (Vailshery, Jaganmohan and Nagendra, 2013), with leaf size, total leaf area and orientation important traits in their efficiency (Leonard, McArthur and Hochuli, 2016). The benefits of trees in this regard are not limited to roadside locations (Beckett, Freer-Smith and Taylor, 1998). No literature could be found specifically comparing the benefits of street greenery to other forms of greenery, but it is likely that, compared to other public vegetation, the vegetation within the road easement provides proportionally more air filtering than park or yard greenery.

Integrated water management approaches to urban stormwater can significantly improve stream health by reducing flows, infiltrating water into soils, recharging aquifers, and distributing points of inflow into streams, leading to increased ecosystem service provision (Walsh *et al.*, 2005). Trees and other vegetation retard flows by capturing a significant portion of rainfall (Armson, Stringer and Ennos, 2013). Vegetation allows soil to infiltrate water by the action of roots loosening soil and creating pathways for water movement (Burgess *et al.*, 2001). Stormwater treatment occurs at a range of scales within urban green spaces, for instance through raingardens located within the road verge, to large sedimentation ponds built to service urban developments of many thousands of homes. The urban green spaces associated with each scale of water management each contribute unique ecosystem services. For instance, raingardens are part of the everyday engagement of nature that road verges provide (Weber, Kowarik and Säumel, 2014), swales and other drainage lines provide nutrient recycling, biodiversity, landscape quality, and air cooling and filtering (Säumel, Weber and Kowarik, 2015), and engineered ponds for water treatment and flood control provide distinct biodiversity benefits and parkland areas (Sun *et al.*, 2018).

Management practices that increase leaf litter and understorey are likely to promote greater decomposition processes (Ossola *et al.*, 2016). A study in Neuchâtel, Switzerland, showed the diversities of the common detritivores earthworms and enchytraeids were positively (earthworm) and negatively (enchytraeid) influenced by soil age (i.e. the date of local urban development). Moreover, gardening practices influence the rate of decomposition in urban soils and below-ground diversity (Tresch *et al.*, 2018), as does the application of mulch (Byrne, Bruns and Kim, 2008). Road verges are often gardened (Hunter and Brown, 2012), and the type and extent of those gardening practices – as well as the age of urban development – will influence the decomposition benefits provided by this green space type.

Verges as a constituent of urban green space

Residential yards, parks and road verges are the main components of urban green space in terms of area, though the significance of road verges is less clear than that of residential yards and parks, because only one study could be found that measured the extent of the road verge. That study was in Syracuse, New York, and calculated road verges to be 7% of total land use, compared to parks at 9% and residential yards at 16% (Richards *et al.*, 1984). Residential yards have been the subject of numerous studies. In Stockholm 16% of land is yard (Colding *et al*. 2006), in five cities in the United Kingdom it is 21.8–26.8% (Loram *et al*. 2007), in Dunedin, New Zealand, 36% (Mathieu *et al*. 2007);, and in Leipzig, Germany 12% (Haase, Jänicke and Wellmann, 2019). The differences between these figures is likely to reflect complex interactions between each city's historic mixes of parcel size and shape, housing size and style, population density and sprawl. These differences are probably also influenced by varying definitions of what the extent of the urban area is, for instance the built-up area of a city is usually different to the political boundary of a city, and boundaries can be predicated on varying projections of future growth. As a proportion of urban green space, yards often account for over 50% of total greenery (in Sheffied, UK, Gaston *et al.*, 2005; in Dunedin, New Zealand, Mathieu, Freeman and Aryal, 2007; in Brisbane, Australia, Shanahan *et al.*, 2014; in Sydney, Australia, Lin, Meyers and Barnett, 2015), and parks generally occupy less area than residential yards. For instance, in Melbourne, 10.7% of total land use was found to be urban green space consisting of protected areas, natural and semi-natural areas, parklands and organised recreation areas (VEAC, 2017), a figure much lower than the 50+ per cent suggested above.

The extent of urban greenspace varies between different cities. Globally, more populated cities tend to have proportionally less urban green space (Fuller and Gaston, 2009; Dobbs, Nitschke and Kendal, 2017; Richards, Passy and Oh, 2017). Urban green space is more fragmented, and green cover is least, at the city centre. Cities with greater levels of social inequality are associated with more fragmented urban green space (Dobbs, Nitschke and Kendal, 2017). There is proportionally more urban green space in cities with higher per capital GDP compared to poorer cities (Richards, Passy and Oh, 2017). However, as cities grow larger over time, access to nearby urban green space increases (Huang *et al.*, 2017).

The spatial distribution of urban green spaces

Just as different types of urban green space provide varying benefits in terms of biodiversity, ecosystem function and human amenity, so they also have different spatial extents and distributions. For instance, road verges are each individually small, but they are many in number and are found across the city. In comparison, large parks are few in number, and only found at a few locations, while rail lines and high voltage electricity transmission lines are linear green space features that are largely continuous. It is easy to see that in each of these examples the spatial distribution and extent may influence the benefits provided by that resource. For instance, rail corridors might allow species to move long distances relatively unimpeded, large parks might provide habitat for species that tend to avoid humans, and road verges, because of their ubiquity, might provide everyday contact with nature for humans (Pyle, 2003; Miller, 2005; Standish, Hobbs and Miller, 2013; Weber, Kowarik and Säumel, 2014).

Small parcels, such as road verges, can be seen as part of the "long tail" of urban green space. The long tail is a statistical term popularised by Chris Anderson in *Wired* magazine (Anderson, 2004) in a discussion of Amazon's publishing market. The 'short head' of sales are the bestsellers that sell in large numbers, and the 'long tail' are the many less well-selling publications. Interestingly, the long tail provides as much revenue as the short head. In terms of urban green space, large open spaces are the short head, and the road verges, vacant land, and many small, less formally recognised, patches are the long tail. Richards' *et al* study (1984) in Syracuse, New York, that showed road verge accounted for 7% of land use and parks 9%, demonstrates that the long tail of urban green space is a major contributor to total urban green space in the same way as less well-selling publications are to Amazon's book revenue.

The idea of the long tail as a tool for urban greenspace design has been noted (Grose, 2017). However, the substantial extent of the long tail is not always the case. For instance, Gaston *et al* (2013) showed that in one regional area of the U.K., the top 5% of green spaces in size account for approximately 80% of total urban green space. As they pointed out, “the spatial distribution of patches of different sizes tends to be highly variable amongst different urban areas.” They also note that large green spaces are generally found in the outer areas of a city. Neighbourhood age also influences the distribution of urban green space. As noted above, yards are the main contributor to urban green space. Housing style has been associated with extent of yard vegetation (Ossola *et al.*, 2019), indicating that the distribution of urban green space will vary with the age of housing development across a city. Neighbourhood age also affects the distribution of tree canopy (Lowry, Baker and Ramsey, 2012), and historic census data better explains urban forest cover than present-day census data (Bigsby, McHale and Hess, 2014). This is not simply because older neighbourhoods tend to have more mature and hence larger trees. For instance, street density is positively correlated with tree canopy, but this relationship diminishes with age. Similarly, the positive association of income with vegetation – i.e. the ‘luxury effect’ (Hope *et al.*, 2003) – is not present in newer neighbourhoods but becomes increasingly pronounced as neighbourhoods age (Lowry, Baker and Ramsey, 2012). Because street density is correlated with the distribution of canopy in new developments but not in older neighbourhoods, the benefits associated with canopy vary between newly developed neighbourhoods and older neighbourhoods (Lowry, Baker and Ramsey, 2012). Lin *et al* (2017) found house age and yard size were positively correlated with vegetation cover, and people with a greater ‘nature relatedness’ (as measured on a psychometric scale) and less socio-economic disadvantage also had greater vegetation cover.

Similarly, planning controls that regulate elements of developments, such as the dimensions of the carriageway (road surface), sidewalks, crossovers (driveways crossing from the property line to the kerb), and the positioning of utilities, have changed over time, suggesting that the distribution of road easement green space might vary with the date of urban development across a city (Austroads 2009; Standards Australia 2013). And traffic calming strategies have also changed the form of the road easement, introducing roundabouts, median strips, chicanes and vegetated outstands (Van Schagen, 2003).

Cul-de-sacs are a common and often defining feature of modern suburbia that are also likely to influence the extent and distribution of urban green space. The ‘Radburn’ housing estate style, which is named after the design of a 1929 estate in Radburn, New Jersey, is characterised by cul-de-sacs and winding roads. It was designed to fit more parcels of land on topographically irregular development areas, minimise road length per residential parcel of land thus saving costs on roads and utilities, and to do so by using a non-hierarchical road system where there are many low-traffic local roads and only a few high-traffic collector roads (Stein, 1957). A number of consequences for urban green space follow from this. Housing parcels tend to be irregular in shape, which has been shown to decrease the amount of vegetation in a development (Nielsen and Jensen, 2015). The amount of road verge should tend to be less because a greater proportion of the frontage of a cul-de-sac property will be devoted to driveway (Stone, 2004). The proportions of major to minor roads between Radburn-style developments and grid-style developments will vary and the width of these roads and their easements are also likely to influence the distribution of road verge. And because cul-de-sac developments are better able to respond to variations in topography, they allow development to occur in places where otherwise development would not, such as at the steep edges of ravines, thus intruding into land that might well otherwise be set aside as nature reserve (Gordon and Tamminga, 2002).

The distribution of urban green space is also influenced by the ability of authorities to acquire land. In new developments on the urban fringe the acquisition of land is often a legislated necessity, e.g. in Melbourne, Australia, 10% of the unencumbered land available for new development must be set aside as public open space (VPA, 2017), though there are generally few specifications as to what sort of land should be set aside, e.g. in terms of its conservation significance. Similarly, planning regulations often specify the maximum proportion of a land parcel that can be covered in impermeable surfaces, and set the proportions of the elements within the road easement, thus influencing the amount of road verge. In already developed urban areas, where acquisition of land is affected by both the availability of land and by its cost, which is often much higher than land on the urban fringe (Boulton, Dedekorkut-Howes and Byrne, 2018), the quality and size of newly acquired green space will be very different.

Residential development companies purchase land as large areas and develop residential estates within which hundreds or thousands of houses are constructed simultaneously. This means residential areas grow development-by-development rather than house-by-house and that roads and their patches (road verges) and houses and their patches (yards) get built together as part of the one process so we may expect to see these vary together in their distribution and extent. Variations in approach exist between development companies, which lead to differing urban forms and resident demographic patterns (Coiacetto, 2007). Urban in-fill, where old areas are demolished and rebuilt as new residential developments, places older residential developments adjacent to newer developments. Together, these produce an urban landscape that is a mosaic and far less homogenous than the simplistic idea of 'suburbia' suggests. It is important then to recognise the housing development as a significant unit when considering strategies for maximising biodiversity, ecosystem function and human amenity within residential areas.

At any one time, there is a considerable portion of urban land that is undeveloped and present as an significant component of urban greenspace. The distribution and extent of this land is changes continuously as part of the 'churn' or turnover of cities as buildings are torn down and land is redeveloped or awaits development, e.g. on the urban fringe. Vacant land that is unmanaged undergoes succession, e.g. where ruderal species give way to shrubs which then become overtopped by young trees. The rate of turnover in vacant land maintains a range of stages of succession and hence a diversity of habitats, increasing biodiversity. One study estimated that for maximum conservation value, brownfields should remain undeveloped for 15 years (Kattwinkel, Biedermann and Kleyer, 2011). Vacant parcels of land are a dynamic mosaic in which each parcel of land acts as a temporary conservation zone.

Redevelopment of existing urban land promotes trees loss (Guo, Morgenroth and Conway, 2018). At the same time, in shrinking cities, lack of development, or an increase in demolition rather than construction, is associated with greater levels of urban greening (Endsley, Brown and Bruch, 2018).

The distribution of urban green space is also a product of human and biophysical legacies, including the pre-settlement biome, colonial attitudes, trends in park design, changes to urban form, demographic change, historic disturbances (e.g. fire), all acting at a broad range of scales (Roman *et al.*, 2018). For instance, Dobbs *et al* (2017) have shown in the inner areas of hilly cities green space is located upslope because settlement tends to occur in valleys.

Patches, connectivity and road verges

Road verges are patches of habitat that can contribute to the connectivity of the urban environment. In general terms, connectivity is often seen as a product of patches habitat, stepping stones and corridors. Patches are conceived as being connected by stepping stones and corridors, and connectivity between patches is affected by the distance and barriers between patches. Stepping stones and corridors create connectivity between patches of habitat (e.g. urban green space), making it more permeable, allowing species to move to new locations if local resources are insufficient, improving opportunities for outbreeding and providing resilience against local extinctions. In urban ecology, the land between patches (e.g.

houses, the carriageway, footpaths) is collectively termed the matrix. This idea, that the environment consists of patches of habitat within an inhospitable environment, has been central to conservation biology for several decades and comes from the theory of island biogeography (MacArthur and Wilson, 1967; Levins, 1970).

However, this characterisation is not without its problems. The distinction between what constitutes a patch of habitat and what does not is a fraught one, especially in highly fragmented urban systems (Prugh *et al.*, 2008). For instance, many species rely on specific habitat *features*, such as old trees, rather than on *types* of landscape (Franklin *et al.*, 2009), and what one species may perceive as habitat may be inhospitable to another species. For instance, in a study of five bat species in Waco, Texas, the distribution of two species was best described by vegetation, two species by water and one species by buildings (Li and Wilkins, 2014).

Moreover, the quality of the matrix is of fundamental importance, affecting connectivity and its permeability (Baum *et al.*, 2004). Driscoll *et al* (2013) argue that the matrix needs to be understood as non-homogenous and not static, that species disperse through the matrix at different rates and dependent on varying temporal scales, and that species may adapt to the matrix or change their response to the matrix over time. In a broad study of the distribution of urban biodiversity in derelict sites within the city of Birmingham, U.K., the researchers concluded that for butterflies and carabid beetles the quality of patch was more important than its location, possibly implying that the quality of the matrix between sites was sufficient for dispersal (Angold *et al.*, 2006). Their modelling suggested small and medium sized mammals may be more dependent on stepping stones or corridors for dispersal compared to butterflies and beetles. No evidence was found for flora requiring corridors for dispersal. Corridor planning, therefore, needs to be taxon-specific.

The effects of fragmentation on species add further complexity to the narrative of patches, connectivity and how the landscape provides resources and connectivity. The fragmentation of a landscape is in many ways the opposite of the connectivity of a landscape. Fragmentation is generally characterised in the ecological literature as a negative characteristic, leading to the isolation and general decline of populations (Villard and Metzger, 2014; Hanski, 2015). A recent review (Fahrig, 2017) showed that one of the main reasons for this mischaracterisation is a failing to disentangle the effects of fragmentation and habitat loss. For instance, if a road is put through a forest, both fragmentation and habitat loss occur, but a decline in species numbers is attributed to the fragmentation when in fact it is an effect of the habitat loss. Overall, Fahrig's review showed fragmentation had positive effects in a landscape by a) increasing connectivity by increasing the number of edges that species tend to move along and by reducing the distance between patches, b) increasing diversity in the landscape, spatial heterogeneity and the number of ecotones (edge landscapes), c) making predator/prey relationships more stable by allowing more locations for refuge and more complex relationships between patches, d) reducing overall risk to populations by spreading that risk across patches, e) in the same way reducing competition between individuals, and f) creating areas of complementary landscape, e.g. a landscape for resources and a landscape for habitat. 'Fragmentation' is an inherent quality of landscapes that are mosaics. While the urban landscape is generally characterised, negatively, as highly fragmented, it is important to recognise that fragmentation can have positive aspects too. Moreover, the road verge is a highly discontinuous type of green space, divided by driveways and each block separated by road from the next. The negative associations with fragmentation inhibit new ways of thinking that may allow us to reconceptualise the road verge as a green space that spans the city, providing connectivity and habitat.

There is also considerable debate over how much the size of a patches matters. Conventional ecological thinking is that larger patches tend to provide more benefits than the equivalent summed size of smaller patches (Gaston, Ávila-Jiménez and Edmondson, 2013). This has implications for management. For instance, one British study has suggested that the size of a single yard is insufficient for successful maintenance of viable populations for many species and that community engagement through 'wildlife-friendly' gardening schemes should be used to encourage the residents of groups of properties to all promote the same habitat qualities together, effectively increasing patch size (Goddard, Dougill and Benton, 2010). Against that position, Kendal *et al* (2010) have argued that individual landowners benefit urban ecosystem health by creating greater diversity of niches through individual actions while as the same time being positively engaged in greening activities. Moreover, recent research is showing that when it comes to patch size, big is not always better. For instance, the significance of small reserves for biodiversity conservation has been underestimated (Kendal *et al.*, 2017), with small reserves in Melbourne, Australia, having greater species to area ratios, though less rare species per unit area. Similarly, a study on the distribution of threatened species in Berlin showed that the distribution of rare species was not influenced by patch size in remnant patches of vegetation, and that many threatened species were found exclusively in non-remnant patches (Planchuelo, von Der Lippe and Kowarik, 2019). perhaps most convincingly, recent research on the Habitat Area Hypothesis (Watling et al., 2020) shows that the quality of a patch is more related to the amount of habitat around the patch than to either the size of the patch itself or to measures of its degree of isolation.

These results have implications for road verges, most importantly that they cannot be dismissed as simply patches that are too small to be worth considering in terms of the habitat they provide. moreover, it is possible that the benefits this highly fragmented greenspace type provide are as significant as those of larger patches.

Road verges in the context of land sharing of land sparing

The varying qualities of, and relationships between, urban green spaces begs the question of how, ideally urban green space should be apportioned. Green space in a city can be provided at a range of scales, for instance along a continuum from many small to a few large areas of urban green space. In ‘land sharing’ models of urban development, green space is more distributed across yards, road verges and small parks. In 'land sparing' models, green space is more concentrated in large parks (Lin and Fuller, 2013) that may even be beyond the boundaries of the city. There is considerable debate about which of these models is most beneficial.

Land sharing models require greater multifunctionality from urban green space because those spaces need to meet the multiple needs of humans and other species (Grimm *et al.*, 2008). Road verges are a clear example of this: being the location of underground utilities, weekly rubbish collection, needing to be traversed for access to parked vehicles, as well as providing habitat, water infiltration and so forth. At the same time, land sharing elements create a set of benefits unavailable in land sparing models, for instance street trees reduce the urban heat island, and yards provide opportunities for growing vegetables. Many argue that small everyday encounters with nature, such as are provided by road verges, are just as important as the experiences of large reserves when it comes to awakening biophilia and preventing the extinction of experience (Pyle, 2003; Miller, 2005; Standish, Hobbs and Miller, 2013). Moreover, one study identified that people living in urban land-sharing environments visit urban green space more often than those in urban land-sparing environments, suggesting that extinction of experience will be higher in land-sparing models. And, as noted above, Fahrig (2017) suggests that since fragmentation is generally beneficial, and that hence land sharing development should be preferred to land sparing.

Against such arguments, land sparing models provide habitat that is less modified by anthropogenic factors, meaning urban sensitive species (i.e. species that respond poorly to urbanisation) can be retained alongside urban adaptors (Villaseñor *et al.*, 2017). Proponents argue that well-planned cities, for instance with corridors for species movement, can achieve such outcomes, e.g. (Sandström, Angelstam and Mikusiński, 2006). Moreover, it is argued, the green space available through land sparing is better in terms of supplying a range of ecosystem services, e.g. water infiltration (Stott *et al.*, 2018). There is also some evidence to suggest that greater connection to nature is achieved in wilder landscapes (Chawla, 1999; Wells and Lekies, 2006) and when the activity is purposeful (Vorkinn and Riese, 2001), meaning that land sparing landscapes may provide a better quality nature experience with more lasting impacts. Road verges and residential yards would be minimised in land sparing models, leading to a poorer day-to-day experience for residents, but with the potential for more profound experiences possible for residents who make the effort to travel some distance for experience of nature.

Much depends on the quality of the landscapes. For instance, large reserves will tend to have more understorey, compared with, say, road verges, which are mostly trees and lawn. This however doesn't need to be the case: it is quite possible to have significantly more understorey on road verges. Small improvements in the quality of the urban forest may move optimal outcomes away from land sparing and towards land sharing (Collas *et al.*, 2017).

The land sharing/sparing debate is complicated by the realities of development occurring across a range of scales from the single property to the street to the housing development (Garrard *et al.*, 2018). Planning processes (Norton, Evans and Warren, 2016; Hansen *et al.*, 2019), as well as the complexities of species response to urbanisation (e.g. Caryl *et al*. 2016) further complicate matters. In practice, optimal solutions may lie between sparing and sharing (Geschke *et al.*, 2018). Finally, there is a larger debate playing out about the relationship between urban spatial development and urban sustainability into which the land sparing/sharing debate has some input only, with issues such as greater spatial access to infrastructure such as hospitals and schools, and carbon footprints of traffic systems perhaps having greater influence (Silva, Oliveira and Leal, 2017; Khan and Zaman, 2018).

The structure and composition of urban green space vegetation

The structure, taxonomic composition and age of the vegetation comprising urban green space has significant influence on ecosystem function, biodiversity and human amenity. This in turn will have implications for understanding the contribution that road verges do and can make in the urban context.

Habitat structure underlies community structure, providing resources and mediating species interactions (Bell, McCoy and Mushinsky, 1991). Within the urban context, a recent meta-analysis shows that overall vegetation structure, as well as structure at the tree, shrub and herb scales, has been shown to have positive effects on biodiversity in urban green spaces (Beninde, Veith and Hochkirch, 2015), with insects, birds and mammals responding in distinct ways. Vegetation's structural complexity has been shown to influence below-ground species communities and biochemical and hydrological processes (Ossola, Hahs and Livesley, 2015; Ossola *et al.*, 2016). Urban areas often lack elements of habitat structure important to species, such as old trees with hollows (Le Roux *et al.*, 2014), which Threfall *et al* have shown is linked to bat activity (Threlfall, Williams, *et al.*, 2016). However, urban areas have artificial analogues to natural habitat elements (Lundholm and Richardson, 2010); a well-known example is that tall buildings can function as high cliffs for peregrine falcons (Luniak, 2004).

The proportion of native to exotic species can have significant effects of community composition. White *et al* (2005) showed that in Melbourne, Australia, native streetscapes supported more native bird species, and insectivorous and nectivorous species, when compared to exotic streetscapes. Other Australian studies have shown an associations between the proportion of native plant species and the species richness of bats; and between increased quantity of understorey vegetation and bird species richness (Threlfall, Ossola, *et al.*, 2016; Threlfall, Williams, *et al.*, 2016). Australian native pollinators significantly prefer native flora to exotic flora (French, Major and Hely, 2005; Batley and Hogendoorn, 2009). This is not necessarily the case in all countries, for instance Salisbury *et al*. (2015) showed that in the United Kingdom exotic species can prolong the time important floral resources are available for native pollinators.

Lack of management also alters species composition by allowing spontaneous vegetation to develop over time. Spontaneous vegetation can have considerable biodiversity benefits (Bonthoux *et al.*, 2014). In one study, semi-natural urban forest, lawn and urban spontaneous vegetation were compared. Spontaneous vegetation had the highest plant species diversity and diversity and numbers of arthropods, and had similar carbon capture and biomass to lawn (Robinson and Lundholm, 2012). Because spontaneous vegetation occurs of its own accord, and is always appropriate to site conditions, it may work as a cheap means of adding some extra biodiversity into the urban context. Kuhn describes research to transform spontaneous vegetation communities by the addition of other species for design purposes (Kühn, 2006). Similar research has been undertaken in Sheffield, UK (Tylecote and Dunnett, 2012) and elsewhere (e.g. Landschaftspark Duisburg Nord by Latz and Partner), with brownfield sites becoming minimally managed parks presenting a strong sense of wildness and celebrating historical forces of change and "decay". While spontaneous vegetation can also be culturally associated with dereliction, evidence now suggests that, at least in Europe, the public now recognise its biodiversity benefits (Weber, Kowarik and Säumel, 2014; Fischer *et al.*, 2018). In the United Kingdom, too, public perceptions are changing, with the messier aesthetic associated with native meadow planting now becoming more acceptable (Hoyle *et al.*, 2017) and perceived as being of increased quality compared to mown amenity grass (Southon *et al.*, 2017a).

Lawn and the road verge

Road verges are usually dominated by lawn (mown amenity grass). Lawn is a ubiquitous element of global urban landscapes and has been characterised as a major urban biotope (Müller, 1990; Stewart *et al.*, 2009). It is estimated 23% of urban areas in the U.S. are lawn and lawn is the largest irrigated crop produced by area (Robbins and Birkenholtz, 2003; Milesi *et al.*, 2005). Lawns are usually sown with seed mix, in some cases with fertiliser added, or are laid as turf, and are then subject to colonisation and succession.

Lawn has been characterised as a sort of green desert (Haeg *et al.*, 2008), devoid of biodiversity, low in environmental benefits, but this characterisation may be an oversimplification. For instance, lawns can improve soils, water recharge, and reduce urban heat island effects (Beard and Green, 1994) and the vegetation of lawn is rarely a monoculture or restricted to a minimal blend of turfgrasses, though biodiversity is linked to the extent of management (Grewal, 2012). Understanding the species composition of lawn may be important for several reasons. First, because of its extent, lawn plays a large role within existing urban ecosystems, especially in terms of connectivity (Robbins and Birkenholtz, 2003). Second, common lawn weed species are significant resources for pollinators (Carper *et al.*, 2014; Larson, Kesheimer and Potter, 2014). Research on lawn has focussed on domestic yards and public parks, and has been undertaken mostly in America, Europe and New Zealand. In a German study of lawns, Müller (1990) argues that lawns are a relatively young type of plant community that is still evolving.

The species composition of road verges has only been the subject of one investigation in the international literature (Stewart *et al.*, 2009), and then only as a component of a broader study of the urban lawns of Christchurch, New Zealand, in which they found 127 species. In other lawn studies, Wheeler *et al*. (2017) found 353 species across domestic lawns of 7 cities in the U.S., in Paris 79 species were found across domestic and public lawns (Bertoncini *et al.*, 2012), while a British study (Thompson *et al.*, 2004) found 159 species in domestic lawns. In all these studies, turfgrass species contributed comparatively little to total species richness. Thompson *et al*. (2004) showed lawn species accumulation curves to be more similar to natural grasslands than to cultivated yards, with lawn composition mainly “typical” lawn species with additional impermanent community species that are “very unlikely to attain reproductive maturity”. It is important to note, however, that the phylogenetic diversity of yard floras is significantly less than that of natural systems (Knapp *et al.*, 2012), and that lawns are likely to be poorly diverse compared to natural grasslands. Urban road verges can harbour critically endangered species (Stuwe, 1986; Kirkpatrick, McDougall and Hyde, 1995) and within Melbourne some indigenous species are more common on roadsides than elsewhere (Lorimer, Pers. comm. 2017).

Management, climate and environmental factors affect species composition and abundance. Stewart *et al*. (2009) found that lawn care factors such as frequency of mowing, removal of clippings, use of fertiliser and herbicides affected species composition more than social factors, and they identified 7 distinct communities within domestic and park lawns and road verges of Christchurch, New Zealand. Leaving lawn clippings in situ after mowing can positively affect sward growth (Knot *et al.*, 2017). Wheeler *et al*. (2017) showed turfgrasses species varied across domestic lawns of the U.S. more than the non-turfgrass species. Thompson *et al*. (2004) found climate and trampling to more significant influences in species composition across Sheffield, but not environmental or management factors. The floral composition of domestic lawns has been shown to be related to household income, education, gender, fertiliser use, degree of irrigation and a number of sociocultural constructs such as prestige and pride – see Martini *et al*. (2013) for an overview.

Management affects non-floral taxa as well. Microbial biomass reduces with increased application of fertilisers and pesticides (Cheng *et al.*, 2008). Mowing height, frequency and timing are significant factors, variations of which can allow species to flower and seed depending on species characteristics (Rupprecht *et al.*, 2015). Low-mow management regimes have been successful in terms of promoting butterflies (Valtonen and Saarinen, 2005) and birdlife (Mason *et al.*, 2007), as well as reducing costs associated with the maintenance (O’Sullivan *et al.*, 2017). Semi-natural grasslands, such as lawn, can support pollinators that in turn support species beyond the grassland site (Jakobsson and Ågren, 2014).

Grass free lawns have been the subject of some research. For instance, Smith and Fellows (2015) investigated grass-free lawns to try to find an aesthetic, biodiversity-friendly alternative to ecologically impoverished, species poor and intensively managed lawns, and they concluded native British species performed best in terms of floral production, that mowing frequency and height were important factors, and that grass-free lawns needed a substantially different management approach to standard lawns.

Back yards and front yards

Being an extension of the domestic space, road verge vegetation may be related to yard vegetation. Vegetation is different in front yards compared to back yards. In Australia, back yards are more likely to be used for recreation and vegetable growing (Timms, 2006; Hall, 2010). Back yards in have proportionally more tree cover in Boston, USA (Ossola *et al.*, 2019) and tree cover increases proportionally with yard size in Michigan, USA (Nassauer *et al.*, 2014). Larger yards and larger parcels have larger trees (Ossola *et al.*, 2019). Back yards in Chicago, USA, have more complex vegetation structure and resources for wildlife with front yards having more lawn (Belaire, Westphal and Minor, 2016). In a multi-city study in the USA, back yards contain both more cultivated and more spontaneous species than front yards, though their lawns may not differ (Locke *et al.*, 2018). The increased species diversity of back yards may be related to their larger size, with their diversity equal once size is controlled for (Daniels and Kirkpatrick, 2006; Locke *et al.*, 2018). Back yards in Australia and Puerto Rico contain more food taxa and front yards more showy taxa, ornamentals, small shrubs and screening taxa (Daniels and Kirkpatrick, 2006; Vila-Ruiz *et al.*, 2014). Daniels and Kirkpatrick (2006) found that yards were highly variable in their composition and style and stressed that as a consequence simplifications as to the drivers of vegetation composition should be avoided. Indeed, in one study in Hobart, Australia, residents were found to be actively making their front yards look different to each other (Kirkpatrick, Daniels and Davison, 2009).

A Danish study found that parcel shape influences the number of trees a yard can support, with rectangular yards supporting more trees than irregular parcels, and the tallest trees tending to be located furthest away from housing (Nielsen and Jensen, 2015). The extent of tree canopy also varies: larger yards, yards in more socially advantaged areas, and yards in older neighbourhoods, all have a higher proportion of tree cover (van Heezik *et al.*, 2013; Lin *et al.*, 2017), and in many cases, they are the main location of trees (Shanahan *et al.*, 2014).

Urban agriculture in the road verge

Urban agriculture, broadly the organised growing of food plants within cities, occurs across a variety of green space types: for instance, road verges, vacant lots, public housing land (Van Veenhuizen, 2014; Dobernig and Stagl, 2015). In road verges in Melbourne the practice is likely to be generally an extension of domestic gardening, and is likely to be mostly confined to herbs and to fruit and nut trees, though globally the range of plants is likely to be far greater and the practices more varied (FAO, 2019). Road verge food sources include uncultivated species as well as cultivated species, for instance many wild, common and spontaneous species are edible (Grubb and Raser-Rowland, 2012; McLain *et al.*, 2014). The urban forest (including road verges and street trees) can also be a source of timber and wood waste (USDA, 2002) as well as non-timber products (Jahnige, 2004). Road verges have been proposed as the location of biofuel production (O'Sullivan *et al.*, 2017).

Urban agriculture may be limited by the extent of contamination (Olszowy, Torr and Imray, 1995). For instance, in Balmain, Sydney, over two thirds (68%) of soils sampled exceeded national guidelines (Environmental Health Unit, 1988). Similar levels of contamination have been found by other studies: see Laidlaw & Taylor (2011) for a comprehensive overview. Lead presence is highest adjacent to roads, decreasing exponentially away from roads, and proportional to the quantity of traffic over time (Laidlaw & Taylor 2011).

The role of common flora in the road verge

The species found in the road verge are generally common. The role of common species has been generally overlooked. Gaston (2011) argues that ecology has essentially been driven by the study of rare species; that common species – i.e. non-rare species – are fundamental to ecosystem processes, in particular because of their abundance; that conventional conservation measures are not appropriate for common species but that urban greenspace planning can be; and that changes to common species abundance are poorly observed but highly significant. Common species are responsible for the majority of coverage in road verges (Thompson *et al.*, 2004; Stewart *et al.*, 2009; Bertoncini *et al.*, 2012; Wheeler *et al.*, 2017), so the role common species play in the socio-economic system is important. Many of the common species of road verges are 'weedy' species, in that they can grow in a wide range of environmental conditions, can survive mowing well, and are good colonisers of disturbed soils.

The road verge as novel ecosystem

Novel ecosystems are ecosystems in which species occur in new compositions that have not previously occurred and which are the product of human agency (Hobbs *et al.*, 2006). They have crossed thresholds whereby it is practically impossible to restore those ecosystems to their former natural states (Perring, Standish and Hobbs, 2013). New ecosystems are constantly occurring – throughout the evolution of the Earth species have displaced species and habitat has changed. The primary differences today are the pace of change (Handel, 2015), and the human cause.

Most urban ecosystems are novel. Road verges mostly are because their flora are dominated by turfgrass cultivars, bred for the specific characteristics that make a good lawn, and a mix of other spontaneous species (Thompson *et al.*, 2004; Stewart *et al.*, 2009; Bertoncini *et al.*, 2012; Wheeler *et al.*, 2017), all maintained in a state of interrupted succession by regular maintenance. Recognising that road verge communities are novel is an important step in seeing their potential for several reasons. First, many lawn species are commonly referred to as weeds: describing this community as, instead, novel allows discussion to move away from the pejorative term “weedy”, or other terms such as degraded or altered, and begins a conversation about traits and the opportunities these species can provide. Second, using the term novel frees us from any attempts at restoration to some pre-settlement ideal. Third, the description embraces human agency, and accepts the reality of the extent to which anthropogenic change has occurred (Pearce, 2015), and forces us to look at social, cultural and economic factors and not just traditional ecological factors, when thinking about these systems (Pickett *et al.*, 2001; Grimm *et al.*, 2008). Light (2013) argues that novel ecosystems may well provide new and different benefits to humans, can have higher diversity, may have different management needs, can be used to increase resilience to significant changes such as climate change, can enhance the conservation effectiveness in large-scale conservation projects, and, as uncontrolled experiments, have much to teach us about the way vegetation can contribute to urban socio-ecological systems. While a powerful concept, some argue that promoting idea of novel ecosystems will lead to less attempts to restore degraded systems, or to lower standards of restoration (Aronson *et al.*, 2014).

Humans and urban green space

Who we are – our backgrounds, beliefs and values – affect the way we act towards urban green space. Those actions, in turn, transform our urban green space. And urban green space influences who we are, what we value, and how we act. Urban environments are driven by human concerns. ‘‘The biodiversity that matters’’ is a phrase that Harrison and Davies (2002) coined to describe the “places that urban residents value because of social concerns associated with greenspaces and semi-natural areas such as access to greenspace, aesthetics, and opportunities for contact with nature" (Nilon, 2011). In making this phrasing, Nilon was emphasising that biodiversity in cities should not be measured according to the same rules and scales that biodiversity in more natural systems is, that not just remnant green spaces are important because in cities different social and cultural factors weigh-in on how we judge biodiversity.

For humans, the presence of urban green space directly influences physical and mental health. Tree-lined streets increase physical activity (Vich, Marquet and Miralles-Guasch, 2019), providing physical and mental health benefits, as do urban green spaces in general (Lee and Maheswaran, 2011; Konijnendijk *et al.*, 2013). These benefits include reducing obesity (Lachowycz and Jones, 2011) reducing heart disease and diabetes (Maas *et al.*, 2008; Tamosiunas *et al.*, 2014). Urban green space is an effective public health tool (Sugiyama *et al.*, 2018), and the many physical health benefits are well summarised in a review of the literature on nature contact and health by Frumkin *et al*. (2017). Despite much work on the health and wellbeing benefits of greenspace, and the streetscape, Chalmin-Pui *et al.* (2019) note that much remains to be researched with regard to the front yard and its health and well-being benefits.

Street greenery promotes mental health (Bratman, Hamilton and Daily, 2012), for instance by reducing depression (Taylor *et al.*, 2015). The quantity and quality of green space affects wellbeing outcomes (De Vries *et al.*, 2013; Shanahan *et al.*, 2016). Exposure to nature restores mental capacity (Kaplan, 1995; Lee *et al.*, 2015). Urban green space even promotes altruism (Guéguen and Stefan, 2016). Mehta (2008), building on Alfonzo (2005) and Southworth (2005), identifies, safety, comfort, sensory pleasure and sense of belonging as being directly related to street greening and the additional experience of private plantings adjacent to the street. Shelter from sun, wind and rain have all been shown to improve the walkability of streets, and tree and shrub planting can contribute significantly in this regard (Bosselmann *et al.*, 1984). It should be noted that studies of health benefits of urban green space have at times been inconsistent in their findings, possibly due to varying methodologies, urban morphologies, definitions of urban green space (Zhang, Tan and Diehl, 2017).

Sociologists have long emphasized the significance of the symbolic dimension of shared experiences of people in a neighbourhood, e.g. Jacobs (1961), and streets and their vegetation are a vital part of that. Thus, our sense of community is closely related to urban green space. Urban green space creates more cohesive neighbourhoods (Groenewegen *et al.*, 2012), improves social support and decreases loneliness (Maas *et al.*, 2009), parks promote social interactions (Peters, Elands and Buijs, 2010), and social support increases with the quality of those local parks (Kaźmierczak, 2013). Participation in community gardens improve perceptions of community (Booth *et al.*, 2018). At the same time, vegetation reduces crime (Kuo and Sullivan, 2001).

Gardening

Gardening is a pertinent, and well-studied, lens through which the complexities of the human–nature interaction can be seen, and it is also an activity undertaken in the road verge.

Gardening is a complex activity undertaken for many reasons. In a summary of the literature, Kiesling and Manning (2010) identified creative expression and personal meaning, health promotion, production, skill-building, knowledge enhancement, feelings of connection to nature, perceived social benefits, and expression of faith through caring, as key motivations for gardening. Residential yards, especially front yards, are public expressions of residents' beliefs and values, e.g. conforming to social norms (Larsen and Harlan, 2006), the virtue of order and tidiness (Butler-Bowdon, 2001), or the encultured value of the local indigenous ecology (Seddon and Nossal, 1998). Lawns can reflect pride of ownership, industriousness, and family values (Jackson, 1987) and 'suburban respectability' (Feagan and Ripmeester, 1999). Yards can also reflect residents' complex environmental concerns (Kirkpatrick *et al*. 2012; Mustafa *et al*. 2010). Attitudes to native species, and the influence of yard norms (i.e. perceived social expectations), were found to affect residents' preferences for degree of habitat in front yards in a study in Perth, Australia (Kurz and Baudains, 2012). Goddard *et al*. (2013) demonstrated that yard characteristics, management intensity, householder demographics, wider environmental activity and landscape context all influence the occurrence of wildlife-friendly gardening in Leeds, UK. Personal well-being, and an ethical position, motivated residents to garden for wildlife, and they were influenced by social norms. However, "neighbour-mimicry" allowed adoption of new wildlife-friendly practices. Such beliefs and values and their benefits manifest differently in back yards and front yards (Daniels and Kirkpatrick, 2006; Hurd and White, 2006; Larsen and Harlan, 2006; Chalmin-Pui *et al.*, 2019). Numerous studies have found that gardening behaviour is correlated with age, gender, income, class, education, property value and housing age (Cook, Hall and Larson, 2012) as well as rental status (Kirkpatrick, Daniels and Zagorski, 2007). For instance, in Phoenix, Arizona, Yabiku *et al*. (2008) found men preferred xeriscape-style yards, which they attributed to the gendered role of domestic duties and a desire in men to reduce lawn maintenance, e.g. mowing. Housing age affects gardening behaviour because it is positively correlated with cover of vegetation (Grove *et al.*, 2006). Social contagion (i.e. adopting similar gardening practices as others nearby) is well documented (Askew and McGuirk, 2004; Kirkpatrick, Daniels and Zagorski, 2007; Nassauer, Wang and Dayrell, 2009; Kirkpatrick, Daniels and Davison, 2011; Goddard, Dougill and Benton, 2013), further emphasising the strong social links between humans and vegetation.

Gardening often extends to the road verge, and the way residents manage and care for the road verge also reflects their beliefs and values, though these beliefs, values and the actions arising from them may be different to those expressed in either front yards or back yards because of the gardening in the road verge takes place on public land. In that public context, gardening can become community oriented, or can become guerrilla gardening. Cultivators of food in public are motivated by complex personal desires as well as socioecological issues, with the activity identity-forming, allowing contact with nature, and acting to collectively promote societal change (Guitart, Pickering and Byrne, 2012; Finley, 2013). Residents who choose to garden their verge may do so seeking to make cities more sustainable by providing food security, community engagement, and by reducing food miles (Van Veenhuizen, 2014). Verge gardeners may seek to activate public space by gardening where gardening is not authorised, emphasising issues of sustainability, the waste of urban systems, and the arbitrariness of authority (Reynolds, 2014). Guerrilla gardening can be a deliberate means of reshaping urban governance and creating socially innovative urban green space policy (Spijker and Parra, 2018), and Crane *et al*. (2013) have noted that such unexpected interventions can open-up the sort of disruptive possibilities that are essential for ongoing considerations of sustainability. Against that, the road verge can be a place where "residents colonize the land to make it an expression of their own yards and lifestyles, and a measure of their taste and social status… a neat lawn with trees was a sign of moral probity, domestic peace and good citizenship" (Hogan, 2003), and parking on the road verge seen by others as a sign of the resident's lack of civic character (Baumgartner, 1989).

Differences between yard gardening behaviour and verge gardening may also vary from culture to culture because attitudes to urban public space may vary between cultures. For instance, Europeans tend to live in apartment blocks with shared internal courtyards, Americans rarely fence the front or even sides of suburban blocks, while Australians have received the acute territoriality of the English urban yard and have strong domestic property boundaries (Seddon and Nossal, 1998).

In addition, all these underlying drivers of verge gardening are influenced by the road verge's generally harsher gardening environment, being subject to greater trampling, higher compaction, higher pollution levels and increased nutrients from animal waste (Gilbert, 1989), and lack of irrigation and profound disturbance for access to utilities (Gilbert, 1989; van der Ree, Grilo and Smith, 2015).

Perceptions of nature

The way we perceive nature is rarely simple. For instance, perceived biodiversity is not the same as actual biodiversity. Constructed stormwater facilities that visually mimic natural wetlands are perceived as functioning as well as natural wetlands when that it not necessarily the case (Rooney *et al.*, 2015; Sievers *et al.*, 2018). And in another study, perceived biodiversity rather than actual biodiversity was a better predictor (though the effect was weak) of the improvements in stress and mood gained from park visitation (Schebella *et al.*, 2019), with naturalness, vegetation cover, and structural heterogeneity having the most significant effects of the factors tested; and natural parks had a greater effect than sports parks, pocket parks or community parks. At the same time, many studies have shown “neat” vegetation is generally preferred over “wild” or “messy” vegetation (e.g. Nassauer 1995), suggesting that want people prefer isn’t necessarily what bestows the greatest benefits to them. Gobster and Nassauer argue that landscape aesthetics can bridge the social and the ecological, (Gobster *et al.*, 2007), that by aligning these and possibly creating an ecological aesthetic we may find a way to practically increase both the aesthetics and ecological qualities of our urban green spaces. For instance, there is considerable room for the acceptance of biodiverse roadside plantings (Weber, Kowarik and Säumel, 2014; Fischer *et al.*, 2018). Survey respondents approved of wild vegetation, though they preferred planted and maintained vegetation, and showed “a surprisingly high awareness” (Weber, Kowarik and Säumel, 2014) of the benefits that ecosystems can provide, and many considered neatness less important than economic and ecological attributes. Meadows in the United Kingdom have similarly been shown to be able to bridge the gap between the ecological and the aesthetic (Hoyle *et al.*, 2017; Southon *et al.*, 2017b; Norton *et al.*, 2019).

Urban green space provides connection to nature. Many authors consider connection to nature to be fundamental to human wellbeing (Gobster *et al.*, 2007; Louv, 2008; Säumel, Weber and Kowarik, 2015; Taylor and Hochuli, 2015). Biophilia is a term coined by E.O. Wilson as “the innately emotional affiliation of human beings to other living organisms” (1984). The term is used to promote a wide range of urban design and planning strategies, e.g. (Beatley, 2011; Marshall and Williams, 2019) , seen as necessary to create sustainable and resilient cities. Greater interaction with nature is correlated with greater knowledge of nature, and greater awareness can lead to change to a richer urban nature experience (McDaniel and Alley, 2005; Straka, Kendal and van der Ree, 2016). Moreover, perceiving good things about nature has been shown to improve connectedness to nature (Richardson and Sheffield, 2017). Experience of urban nature is important for the conservation and promotion of natural values within our urban environment (Fuller *et al.*, 2007). The particular qualities of nature experienced can vary with the type of urban green space. For instance, informal green spaces such as vacant lots and railway verges, have many characteristics not associated with more formal green space: ambiguous ownership (de Solà-Morales, 1995), a blurring of traditional binaries such as public/private, controlled/neglected, nature/culture (Barron and Mariani, 2013; Foster, 2014), lack of supervision (Rupprecht, Byrne and Lo, 2015), and a sense of wildness (Jorgensen and Tylecote, 2007; Weber, Kowarik and Säumel, 2014). Road verges share some of the qualities of informal green space, such as ambiguity of ownership. For many urban residents, especially those without yards, road verge vegetation is the nature they have most contact with on a day-to-day basis, and such routine contact increases the likelihood of residents implementing pro-biodiversity practices (Prévot *et al.*, 2018).

The connection between values, such as those ascribed to the importance of a connection to nature, and beliefs and actions is the subject of much theorising (Ives and Kendal, 2014). This study uses social psychology theories relating to values, beliefs and norms to better understand residents' perceptions of road verges and verge gardening and how those perceptions relate to action. Both the Theory of Planned Behaviour (Ajzen, 1991) and Value–Belief–Norm theory (Stern *et al.*, 1999) are built on the idea of a cognitive hierarchy, where values are considered more stable and are fewer in number, while the attitudes or beliefs that more directly precede action are more changeable and numerous (Rokeach, 1973). In these theories, values are deeply held constructs that underpin the judgements people make of their environment and, whilst stable, are able to change in response to major life events or to a changing world (Bardi *et al.*, 2009). As an example, the importance of having a significant relationship with nature is a value relevant to this study. Beliefs, in these theories, are less overarching than values, 'facts as people perceive them' (Dietz, Fitzgerald and Shwom, 2005), and more subject to change. An example might be 'verge gardening is good for the environment'. Social context influences the way an individual acts through their 'normative' beliefs. These normative beliefs are an individual's beliefs that an action (for instance verge gardening) is not approved of by another individual, community or institution, e.g. 'I shouldn't verge garden because it is against the law'. The stronger an individual's normative beliefs, the more likely they are to conform to others' expectations. In the context of this study, mowing the lawn of the verge is a normative behaviour: not only do by-laws mandate it, but there are also deeper social forces behind the well-tended lawn (Ignatieva *et al.*, 2015), for instance lawn can signify good morals (Hogan, 2003) and community mindedness (Carrico, Fraser and Bazuin, 2013). In contrast, verge gardening tends to be non-normative, in that it is often undertaken despite municipal authority restrictions, and it is – at least on the surface – against community expectations in so much as most verges are not gardened.

Environmental justice

The uneven distribution of urban green space and the varying benefits of different types of urban green space mean that not all segments of the human population are equally served by urban green space. A number of studies have shown a correlation between measures of social disadvantage (e.g. income, education or more complex indices such as the Index of Relative Social Disadvantage that weights a range of demographic factors, see Australian Bureau of Statistics 2015) and reduced access to urban green space or access to reduced quality of urban green space (see Dobbs *et al*. 2018 for a review). Income, education and land use were first identified as driving factors in plant diversity in Hope *et al*.'s Arizona study (2003). Grove *et al*. identified the now popular concept of the "ecology for prestige" or the "luxury effect" (Grove *et al*., 2004), in which the authors suggest that types of residential vegetation reflect residents' lifestyle choices, including the desire to outwardly reinforce the prestige their community associates with various types of vegetation, for instance a well-maintained lawn. Several reasons have been identified for the association between income and vegetation quantity and quality (Mitchell *et al*., 2016): higher incomes allow more investment in vegetation and its management (Pham *et al*., 2012), greater income allows purchase of larger properties and in older neighbourhoods with mature vegetation (Lowry, Baker and Ramsey, 2012), and greater social power means greater influence on authorities that provide urban green space (Heynen, Perkins and Roy, 2006), as well as legacy effects. Arid climates exacerbate the luxury effect, perhaps because the use of water is costly in such environments, while tropical climates diminish it because "water (and temperature) is not a limiting factor and that the income gap between neighbourhoods is less exaggerated than in other studies where the luxury effect has been supported" (Leong, Dunn and Trautwein, 2018).

The interactions between demographics and urban green space extent, distribution and quality are complex. Socio-economic factors affect people's needs from and desires for urban green space, and the extent and types of benefits they receive (de la Barrera *et al.*, 2016; Wilkerson *et al.*, 2018). For instance, one study in Berlin found older peri-urban residents wanted urban green space to provide experiences of nature, while younger inner-urban residents wanted urban green space to provide opportunities for social interaction (Riechers, Barkmann and Tscharntke, 2018). Management decisions to green disadvantaged areas can have unintended consequences, increasing gentrification rather than reducing disadvantage (Wolch, Byrne and Newell, 2014)

Road verges contribute to this uneven distribution of benefits. Kendal *et al*. (2012) observed education level to correlate to urban tree cover and that inequality was more pronounced in streetscapes than in residential yards, suggesting governance was more important than resident behaviours in explaining street tree canopy cover. Their results supported similar findings by Kirkpatrick *et al*. (2011). Pham *et al*. found street tree canopy cover to be the product of urban form and demographics (Pham *et al.*, 2017). In Salt Lake City, older neighbourhoods tend to have greater street tree species diversity than newer neighbourhoods (Avolio *et al.*, 2018).

Examining road verges

There is a significant body of literature that suggests the road verge's importance within a complex urban socioecological system. Road verges are common. They are likely to have significant influences on the benefits flowing from urban green space because of their considerable spatial extent and because of the complex interactions between humans and the nature provided by road verges. However, beyond this theoretical framing, little data exists on the road verge. In particular, only one study has estimated its extent. No studies have considered its distribution. Analyses of the floristic composition of the lawn of the road verge have been restricted to a single study. The gardening activities that take place in the road verge, the motivations behind them, and their extent, are also understudied despite their potential to substantially contribute to urban green space. To manage road verges for greater biodiversity, ecosystem function and human amenity will require data on the extent, distribution and qualities of road verges as well as other green space types, in particular residential yards and parks. However, trade-offs will inevitably be part of that management, because many actions are taxon-specific, and human amenity benefits do not necessarily coincide with biodiversity and ecosystem function benefits. Without the data on which to base an understanding of these complexities, we cannot effectively frame strategies that work to maximise the benefits from this unique urban green space type.

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Chapter 3
From little things: More than a third of public green space is road verge

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Original article

From little things: More than a third of public green space is road verge

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ABSTRACT

Road easements are ubiquitous in urban landscapes but the green space they provide has been largely ignored, even though it represents a significant proportion of total urban green space. In this study, the extent and spatial distribution of green space within road easements are compared with private residential gardens and other public green space in Melbourne, Australia. Ground-surface permeability to water data and aerial photography were used to identify and quantify the extent of green space within 47 randomly selected neighbourhoods. Road easement green space constituted 7.0% of land cover and a surprisingly high 36.7% of public green space. The percentage of the road easement that was green space was positively correlated with date of neighbourhood development, residential parcel size, footpath absence and social disadvantage. The percentage of residential parcels that was private garden was positively correlated with residential parcel size, but negatively correlated with road density, and positively correlated with date of neighbourhood development as date increased to the 1950s and then negatively correlated for the following decades. Streets with a greater percentage of road easement green space were associated with residential parcels that had a greater percentage of garden. The potential for road easement green space to provide for the biodiversity, ecosystem function and human amenity now being demanded from urban green spaces is much greater than previously thought. Planners, urban foresters, landscape architects, engineers and ecologists need to work together to reconsider the ecological and greening roles of existing road easements and consider this in future road designs. Our data suggests that the roles that residents, municipal governments and road authorities can play will vary across developments, with each presenting specific opportunities to improve the benefits flowing from road easement green space.

1. Introduction

Road easements are a ubiquitous component of urban form that fulfil many functions, including the transport of goods, pedestrian, cycle and vehicle movement, the collection of storm water and carrying overhead and underground utilities. They also contribute to the urban green space network because vegetation is common within the road easement. Road verges (also called the nature strip in Australia and, variously, tree lawn, parking strip, boulevard or sidewalk buffer in North America), along with roundabouts, outstands, median strips and traffic islands, are often-vegetated elements of the road easement. We use the term 'road easement green space' to describe the road verge together with these additional elements. Road easement green space is usually mown grass, perhaps with understorey, and often includes street trees.

Road easement green space provides many benefits, well summarised by Säumel et al. (2015), including regulating temperature (Gillner et al., 2015; Sanusi et al., 2015); mitigating stormwater runoff (Breen et al., 2004; Armson et al., 2013); and mitigating air pollution (Vailshery et al., 2013; Leonard et al., 2016). It can include remnant vegetation, including endangered species and vegetation communities (McDougall, 1987; Lorimer et al., 1997; Lorimer, 2006), and can include numerous distinct vegetation communities (Cilliers and Bredenkamp, 2000). The road easement network can be important in providing connectivity between patches of larger habitat (e.g. parks) for a range of fauna (Fernández-Juricic and Jokimäki, 2001; Oprea et al., 2009; Munshi-South, 2012) and habitat for many taxa including arthropods (Schaffers et al., 2012) and moths and butterflies (Saarinen et al., 2005). The human health benefits of road easement green space include promoting mental health (Bratman et al., 2012), reducing depression (Taylor et al., 2015) and reducing early childhood asthma (Lovasi et al., 2008). Tree and shrub plantings improve walkability (Bosselmann et al., 1984; Vich et al., 2019), which has numerous health benefits, such as reducing obesity (Lachowycz and Jones, 2011), all well summarised in Frumkin et al. (2017). Street greenery also improves social cohesion (Jacobs, 1961; De Vries et al., 2013) and is

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positively associated with safety, comfort, sensory pleasure and sense of belonging (Alfonzo, 2005; Southworth, 2005; Mehta, 2008).

Despite its narrow and fragmented nature, road easement green space's contribution to urban green space is likely to be substantial. We could find only one study that quantified its extent and one other study from which extent could be estimated. Measurements of road verge from a single neighbourhood-scale study in Seattle suggest road verge comprised approximately 4.7% of total land use once total site area was also calculated (Murphy, 2012). In a larger study across 10 census districts in Syracuse, New York, Richards et al. (1984) showed that road verge accounted for 7% of total land use, compared to 9% for parks and 16% for residential gardens. In terms of spatial extent, road verges contributed almost as much as parks.

There are increasing calls to maximise the biodiversity, ecosystem function and human amenity benefits that flow from urban green space (Standish et al., 2013; Haaland and van den Bosch, 2015; Shanahan et al., 2015). Cities tend to occur in biodiversity hotspots (Seto et al., 2012; Threlfall and Kendal, 2018) (Threlfall and Kendal, 2018) and are home to many threatened species (e.g. Ives et al., 2016; Schwartz et al., 2002; Rebelo et al., 2011). Globally, most population growth is occurring in urban areas, 55% of people live in cities (United Nations, 2014) and 3% of earth's surface is now urban (Seto et al., 2010; Zhou et al., 2015). Increasing population densities put greater pressure on existing biodiversity and urban green spaces (Fuller and Gaston, 2009; Haaland and van den Bosch, 2015), so improvements to the ecological function of existing spaces are necessary just to maintain the *status quo*. Moreover, urban areas need to become more resilient to meet the challenges of climate change (Fünfgeld and McEvoy, 2012).

There are many ways design interventions on road verges can do this. Water sensitive urban design (WSUD), in particular the use of swales and raingardens to improve stormwater retention and infiltration, is a well-accepted approach that is nevertheless still underutilised (Radcliffe, 2018). WSUD on roads is often limited to small interventions rather than being implemented at the watershed scale, though notable exceptions exist, such as research in the Little Stringybark Creek catchment in Melbourne's east (Walsh et al., 2015). Urban forest programmes, such as New York's Million Trees programme (McPhearson, 2011), are larger-scale projects that recognise the many benefits of trees, though urban forest extent continues to decline (Nowak and Greenfield, 2018). In Melbourne all local governments have street tree planting programmes but similar programmes for verge understorey planting are lacking, with only 21% of residential properties in ten local government areas having understorey on the road verge, and that understorey almost entirely the result of resident plantings (Marshall *et al.*, unpublished data). Beyond interventions such as these, broader conceptions of the way the road easement could be reconfigured to maximise the benefits of its green space while recognising the complexity of the trade-offs generally remain unbuilt.

To begin to realise the potential of road easement green space to contribute to this urgent urban greening agenda we need to answer a number of research questions. Firstly, what is the extent of road easement green space's contribution to total urban green space? Secondly, how does it compare to common green space types such as parks and residential gardens? And thirdly, what are the drivers of the extent and distribution of road easement green space?

There are numerous potential drivers of road easement green space and residential garden distribution. Green space provision in cities can vary over time (Haaland and van den Bosch, 2015). Planning controls that regulate elements of the road easement, such as the dimensions of the carriageway (road surface), sidewalks, crossovers (driveways crossing from the property line to the kerb), and the positioning of utilities, have changed over time, suggesting that the distribution of road easement green space might vary with both the date of urban development across a city and the type of road (e.g. major, collector, local) (Van Schagen, 2003; Austroads, 2009; Standards Australia, 2013). Street width, length and age are correlated with canopy cover (Pham et al., 2017). The spatial arrangement of roads, e.g. grid or Radburn-style featuring cul-de-sacs, may also have implications for the extent and distribution of road green space and residential gardens because Radburn-style estates (named after the design of a 1929 estate in Radburn, New Jersey) intentionally minimise road length per residential parcel of land (Stein, 1957), and their different road hierarchies mean different proportions of collector (few in Radburn-style) to local roads. Radburn-style properties have been shown to have less canopy because of the often-irregular shape of their land parcels (Nielsen and Jensen, 2015). Canopy cover is also correlated with street density and neighbourhood age (Lowry et al., 2012), extent of vegetation is correlated with residential parcel size (Lin et al., 2017), and Stone (2004) has shown that the length of residential parcel street frontage is correlated with the proportion of residential land that is garden. Ossola et al. (2019) found residential architectural style to be correlated with extent of both front and back garden canopy. Income, education and other measures of social disadvantage have been associated with the extent of residential garden greenery (Hope et al., 2003; Grove et al., 2006), as has population density (Nesbitt et al., 2019).

To quantify the contribution of road easement green space we mapped its extent and distribution over 47 neighbourhoods in Melbourne, Australia, and compared that to the extent of total urban green space and the extent and distribution of residential garden space. We then investigated the potential relationships between the patterns observed and a range of factors including neighbourhood age, road type, road density and social disadvantage. Only with such data can we begin to craft policy to maximise the benefits flowing from this unique green space type, and to redesign our street and suburbs accordingly.

2. Methods

2.1. The study area

The study area is the western and northern residential suburbs of Melbourne, Australia, which occur in the Victorian Volcanic Plain (VVP) bioregion, an area with distinct soils and native flora and fauna to the rest of Melbourne, mostly flat, and that prior to settlement was predominantly grassland or grassy woodland (DELWP, 2019). Melbourne's historical climate is wet and temperate (maritime) with rainfall even year-round (Köppen classification Cfb) but climate change projections suggest it will be increasingly Mediterranean with hot and dry summers (Webb and Hennessy, 2015). Mean annual rainfall is 650 mm, with the west of Melbourne drier than the east. Greater Melbourne (latitude −37.814, longitude 144.96332) is located in south-eastern Australia and has a population of 5 million people (Australian Bureau of Statistics, 2018). It is relatively young, founded in 1835. The 1851 gold rush sparked a building boom that lasted until the 1890s' depression (Keesing, 1971). Inner-city housing built during this time tends to be terraces of small cottages or two-storey houses on small parcels (Barrett, 1971), with the presence of higher density urban infill projects markedly increasing from the early 1990s. Melbourne's growth stalled between the 1890s and the 1920s, after which income, population and suburban subdivision again increased, a trend which continued after World War 2. Housing from the 1890s to 1920s is mostly detached or semi-detached, and on larger parcels compared to earlier development. The post-World War 2 period saw rapid population growth and spatial expansion, an influx of migrants and a substantial increase in the role of cars (Alexander, 2000). Post-war housing is mostly single storey detached dwellings, and middle ring suburbs are home to the 'Australian Dream' of quarter acre blocks (0.1 ha) (Davison et al., 1995; Timms, 2006), though in some areas many are now subdivided, and in general these suburbs have experienced economic decline in recent decades. In the outer suburbs, residential parcels are generally smaller than in middle ring suburbs, though larger than inner-city parcels. Houses are almost always detached, and mostly single storey. Australia has the second largest new home house size in the world after the USA (CommSec, 2017; Demographia, 2018). Melbourne's suburban sprawl continues largely unabated: Melbourne is ranked the world's 32nd largest city by area but 955th by population density (Demographia, 2017). The older parts of Melbourne are now gentrified (Stimson, 1982; Baum and Gleeson, 2010).

Vegetated road verges occur across the city, although in suburbs built prior to World War 1 road verges are less common, with tree cut-outs (rectangles usually 1–4 m^2 cut from the footpath) the most usual way of providing green space within the streetscape. Usually road verges are between half a metre and two metres in width though, in some instances, road verges can be dozens of metres wide, especially when land has been set aside for future road duplication or for major underground infrastructure such as gas lines sewerage and potable water. Road verges are generally divided by vehicle crossovers, although in older suburbs developed before cars became common road verges can run the length of the street. Long, uninterrupted road verges can also be found along the outer edges of inward-facing housing estates that typically have only a single road into the estate and the outer edges of the estate comprised of the side and rear fences of individual properties. Road verges are also not the only greenspace with the road easement, and the variety of additional forms attests to the complexity of this type of public space. For instance, medians that divide lanes of traffic, traffic islands that provide safety for pedestrians crossing busy roads, roundabouts, outstands that narrow the carriageway to calm traffic, road closures designed to increase public green space, planters and pots placed on the footpath, and walkways connecting adjacent cul-de-sacs, are all forms of greenspace present within the road easement (Fig. 1).

Residents in Melbourne are required to manage the understorey and lawn of road verges associated with their property, and public authorities (e.g. municipal governments or the state road authority) are responsible for the maintenance of trees. In practice there are exceptions, when road easement green space is not easily accessible to residents or lies beyond the road verge. This publicly managed road easement green space includes vegetation associated with roundabouts, easements adjacent to residential land but not easily accessible by those residents, the edges of freeways, and the road easements associated with parks, schools and commercial land. The distinction between publicly managed and resident-managed road easement green space is important because variations in management are likely to differentially impact biodiversity and road easement green space benefits.

The Australian Bureau of Statistics geographical unit 'Statistical Area 1' (SA1) (Australian Bureau of Statistics, 2016) was chosen as a study unit because SA1s are generally homogenous in street layout, housing stock, development date, recognise urban historical and landform boundaries, and their use allows direct comparison with existing Australian Bureau of Statistics demographic data. SA1s have an average population of approximately 400 people. Forty-seven SA1s were randomly selected that were at least partially zoned residential and within both the Urban Growth Boundary of Greater Metropolitan Melbourne and also the Victorian Volcanic Plain. These were predominantly residential, but also included other land uses such as schools, commercial strips, parks, sports fields and churches. The term 'neighbourhood' is hereafter used to refer to an SA1.

2.2. Mapping

QGIS (QGIS Development Team, 2015) was used to map the 47 neighbourhoods of the study area, and to overlay Victorian Government spatial property data and the Melbourne Water Impermeability Layer (MWIL), a binary permeable/impermeable vector dataset for Melbourne developed from LIDAR and infrared data with a 0.5 m resolution (Joshphar, 2009). All permeable land within the road easement was considered road easement green space. To investigate the effects of residential land and its urban form without the complicating presence of non-residential land uses such as schools, parks and commercial strips, we separately mapped residential parcels plus their associated road easements (together hereafter called residential areas) to non-residential land. Within residential areas green space was divided into road easement green space and residential garden. Road easement green space was further divided into publicly managed and resident-managed road easement green space. Colour-coded maps were imported into Adobe Photoshop CC (Adobe Systems, 2017) at a resolution of 1 pixel = 10 cm × 10 cm (i.e. finer than either the original resolution of the aerial imagery or MWIL). Pixel counts were then made by colour to determine the extent of each mapped category (Appendix A).

To examine the distribution of green space, we chose four measures: 1) the extent of public green space other than road easement green space (e.g. parks, sports fields, railways easements, creek corridors and otherwise vacant land); 2) the percentage of road easement that was green space; 3) the percentage of residential parcels that was garden; and 4) the percentage of residential road easement green space that was resident-managed. We did not examine the variation in the extent of public green space between neighbourhoods, but rather confined our examination of public green space to its summed extent across all study sites. Similarly, we did not examine variation in the proportion of the road easement green space that was resident-managed, but rather confined our examination to overall proportion across all study sites

To investigate the influences of urban form and neighbourhood demographics on the percentage of road easement that was green space and the percentage of residential parcels that was garden, we used the following eight explanatory factors: date of neighbourhood development; road density, measured as the percentage of the residential area that was road; the Index of Relative Social Disadvantage (IRSD); the cul-de-sac percentage; the local road percentage; the footpath percentage; average residential parcel size; and population density.

The Australian Bureau of Statistics (2011) provided population and Index of Relative Social Disadvantage data for each neighbourhood. Population density was considered a factor worth exploring because it can point to neighbourhoods where individuals have access to greater or less green space by green space type. The date a street was developed was determined from the first date the street name appeared in historic street directories. The date of neighbourhood development was calculated by assigning a date of development to each street, counting the number of dwellings within that street, and then making a weighted average of all dates of street development within a neighbourhood. Strata title data was obtained through the aerial imagery database Nearmap (Nearmap, 2016) and used to establish the number of dwellings in each neighbourhood.

To distinguish cul-de-sac from grid-based development, a simple measure of the extent of cul-de-sacs (the 'cul-de-sac percentage') within a neighbourhood was used. Aerial photography (Nearmap, 2016) was inspected, the number of properties in cul-de-sacs counted and divided by the total number of properties. Similarly, a footpath percentage, measuring the percentage of properties with an adjacent footpath, and a local road percentage, measuring the percentage of properties on local roads, were calculated, because we hypothesised that both these factors were likely to influence the distribution of green space. To establish if a road was local, reference was made to the Victorian Government's VicMap Transport dataset (DELWP, 2017).

2.3. Data analysis

The 47 neighbourhoods captured considerable diversity, with all factors measured showing marked variation, some evenly distributed (e.g. date of neighbourhood development), others skewed (e.g. footpath percentage) and others with significant outliers (e.g. residential parcel size) (Appendix B). To meet the assumptions of statistical analysis, the following data transformations were undertaken: the percentage of residential parcels that was garden was log transformed to reduce the skewing effect of the very large residential parcel sizes present in several neighbourhoods; footpath percentage was converted to binary data

Fig. 1. Resident-managed (A and B) and publicly managed (C–J) road easement green space. A, tree cut-out; B, standard road verge; C, giant road verge set aside for future road duplication; D, roundabouts and median strips; E, development-edge land adjacent to continuous back and side fences; F, land adjacent to freeway soundwall; G, path between cul-de-sacs; H, bulge at intersection for traffic calming; I, road closure; J, median tree cut-outs. All images from the study area: Google Street View 2017.

(all values below 96% became 0, the rest 1); cul-de-sac percentage was converted to binary data (all non-zero values became 1); and local road percentage was converted to three categories (low < 75%, 75% ≤ medium < 96%, high ≥ 96%). For ease of interpreting the relative influences of each factor in subsequent multivariate analysis, all explanatory non-binary numerical factors were then standardised by subtracting the mean and dividing by the standard deviation. We then tested these eight factors for correlation (Appendix C). We found a very high degree of correlation between population density and residential parcel size ($r = -0.902$), and so removed population density from further analysis. We removed population density rather than residential parcel size because population density may vary over time while residential parcel size is fixed. Four factors had notable correlations with date of development, but we did not remove these factors from analysis because we were interested in how these urban form features might contribute in their own right to the urban green space distribution we were analysing.

We used Minitab 18 (Minitab, 2010) to perform multivariate linear regressions to test the seven remaining explanatory factors for correlation with the percentage of road easement that was green space and the percentage of residential parcels that was garden. We also used linear regression to test if these two green space measures were correlated.

We produced scatterplots to visualise all correlations for factors found to be significant in multivariate regression. For factors that we had transformed to binary or categorical data (to meet the statistical requirements of multivariate linear regression), we used untransformed data. We fitted linear and quadratic regression lines to the data, examining residuals and comparing r^2 values to determine which fitted line was appropriate. For ease of interpretation we used unstandardised data in these univariate tests rather than the standardised data used in the multivariate regressions.

Because neighbourhood area varied greatly, from 3.8 ha to 66.6 ha (Appendix B), we have reported our neighbourhood level results in terms of the percentages of neighbourhood area occupied by various land uses rather than by area.

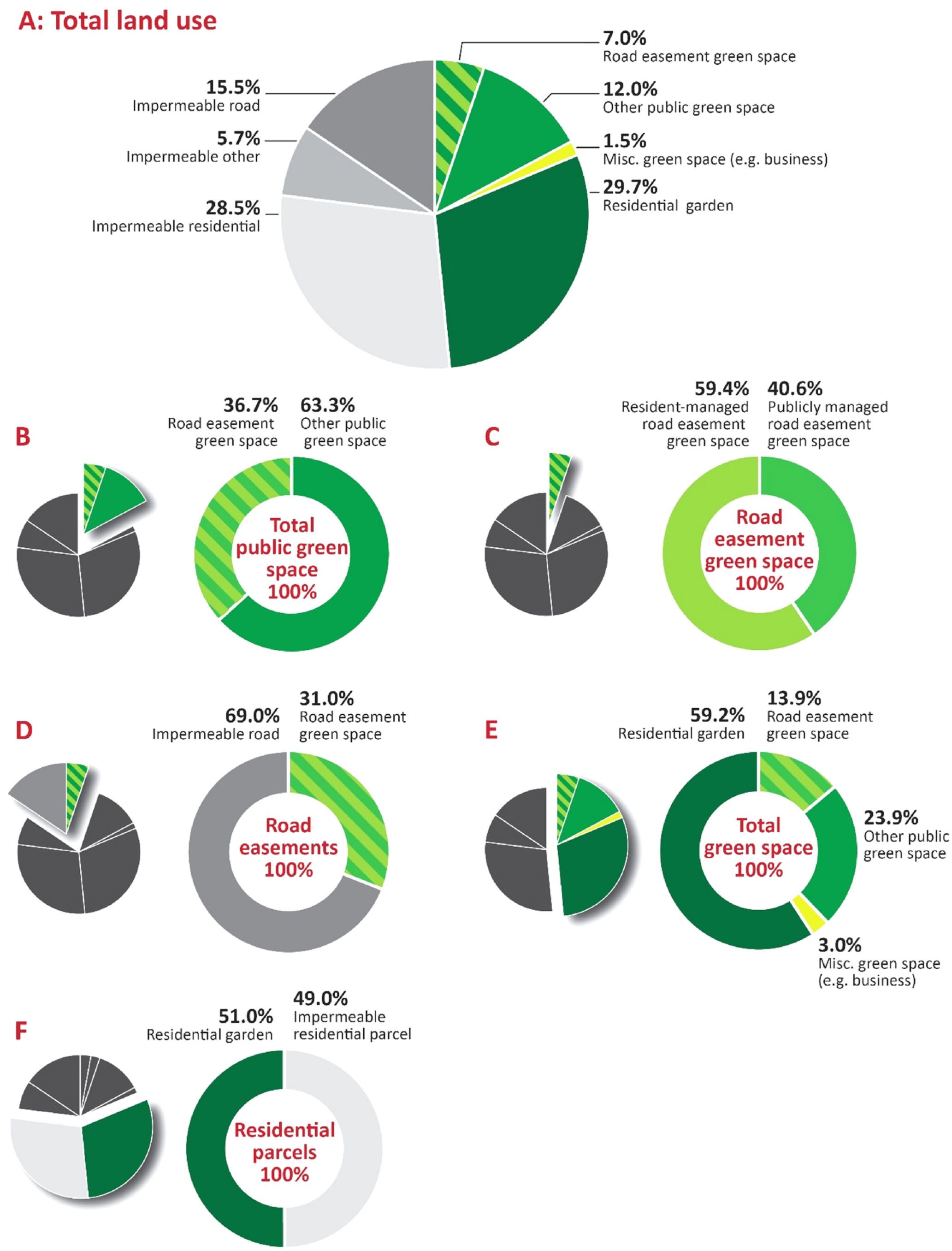

Fig. 2. Percentages of land use for 47 neighbourhoods in Melbourne, Australia (A), which is then further broken down to show the relative percentages of land use that comprise total public green space (B), road easement green space (C), road easements (D), total green space (E) and residential parcels (F).

3. Results

The 47 neighbourhoods analysed totalled 781.2 ha. This included 392.5 ha (50.2% of total study area) of green space, 592.6 ha (75.9% of total study area) of residential area and 175.9 ha (22.5% of total study area) road area. Road easement green space contributed more than a third (37%) to total public green space (Fig. 2B), and it was comprised of 59% resident-managed green space and 41% publicly managed green space (Fig. 2C). Road easements were almost a third (31%) road easement green space (Fig. 2D). The greatest contributor to total green space was residential garden (59%), with road easement green space contributing 14% and other public green space 24% (Fig. 2E). Residential parcels were divided almost evenly between garden (51%) and impermeable land (49%) (Fig. 2F). Further data on the study area is shown in Appendix B. We identified numerous types of road easement green space, beyond publicly managed and resident-managed road easement green space, as part of the mapping process (Fig. 1).

The percentage of road easement that was green space was significantly greater in neighbourhoods that were more recently developed, had greater social disadvantage (lower IRSD), larger residential parcels, and that had fewer footpaths (Fig. 3). Residential parcel size had the strongest influence (highest coefficient in multivariate regression), followed by footpath presence, date of neighbourhood development, and Index of Relative Social Disadvantage (Table 1). The model fitted well (r^2 = 80.0%, r^2 (pred) = 67.5%) (Table 1).

The percentage of residential parcels that was garden was significantly greater in neighbourhoods which were developed in the decades immediately following World War 2, which had lower road

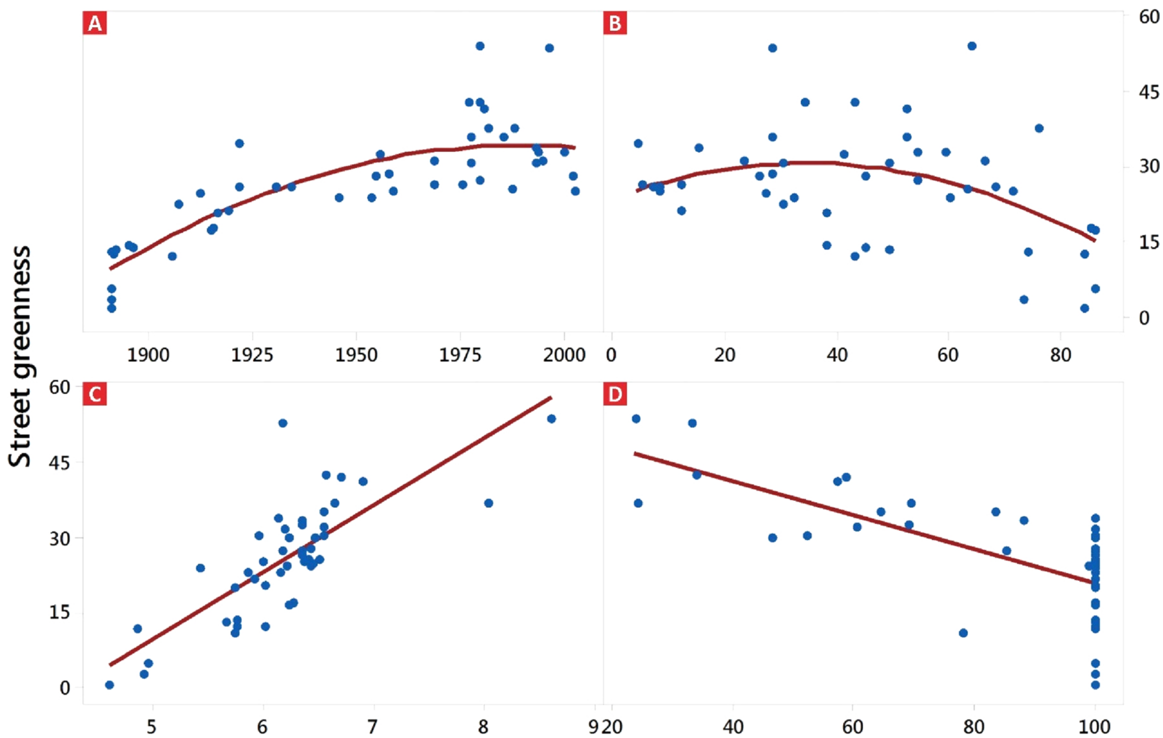

Fig. 3. The percentage of road easement that was green space plotted against four factors found to be significantly correlated at the 0.05 level: A, date of neighbourhood development (r^2 = 67.0%); B, Index of Relative Social Disadvantage (IRSD) (r^2 = 17.5%); C, residential parcel size (m^2, log transformed) (r^2 = 61.2%); and D, footpath percentage (the percentage of properties that have footpaths) (r^2 = 49.2%); for 47 neighbourhoods in Melbourne, Australia. Red lines are fitted regression lines (linear or quadratic).

density, and which had larger residential parcels (Fig. 4). While the multivariate regression indicated that the local road percentage was significant, its r^2 value in the univariate analysis was very low (r^2 = 0.5%). Residential parcel size had the strongest influence on the percentage of residential parcels that was garden, followed by age of neighbourhood development and road density (Table 1). The model fitted well (r^2 = 82.5%, r^2 (pred) = 71.9%) (Table 1).

The percentage of road easement that was green space and the percentage of residential parcels that was garden were positively correlated (p < 0.001, r^2 = 40.5%, Fig. 5).

4. Discussion

4.1. The significance of road easement green space

We found road easement green space accounted for 7.0% of total land use within our study area, other public green space accounted for 12%, and residential green space accounted for 29.7%. This means that the percentage of public green space that was road easement green space was a remarkable 36.7%–more than a third. This substantial contribution of road easement green space to total public green space suggests that the capacity of road easement green space to sustain or increase urban biodiversity, ecosystem function and amenity has been significantly underestimated. Our results for road easement green space match those of Richards et al. (1984) (7%), and are higher than those

Table 1
Results of multivariate linear regressions modelling the effect of 7 factors on percentage of road easement that is green space (street greenness) and percentage of residential parcels that is garden (residential parcel greenness), for 47 neighbourhoods in Melbourne, Australia. R^2 values are derived from univariate plots, are shown for both linear and quadratic fitted regression lines, and are shown only for factors significant at the 0.05 level.

	Street greenness				Residential parcel greenness			
Factor	Coef.	*P*-Value	R^2 (linear)	R^2 (quadratic)	Coef.	*P*-Value	R^2 (linear)	R^2 (quadratic)
Development date	4.32	0.009	61.5	67.0	−2.88	0.048	17.5	37.5
Road density	0.13	0.903			−2.042	0.047	39.5	44.9
IRSD	−2.493	0.008	8.3	17.5	−0.17	0.834		
Residential parcel size	5.17	0.001	61.2	63.9	10.43	<0.001	73.2	73.8
Footpath			49.2	49.3				
Present	−5.04	0.05			−0.33	0.886		
Cul-de-sac (0.1)								
Present	−2.02	0.422			0.93	0.682		
Local road							0.5	2.7
Medium	−1.87	0.455			−5.73	0.015		
High	−0.5	0.843			−4.92	0.036		
	S	R^2	R^2 (adj)	R^2 (pred)	*S*	R^2	R^2 (adj)	R^2 (pred)
Model Summary	5.60	80.0	75.7	67.5	5.08	82.5	78.8	71.9

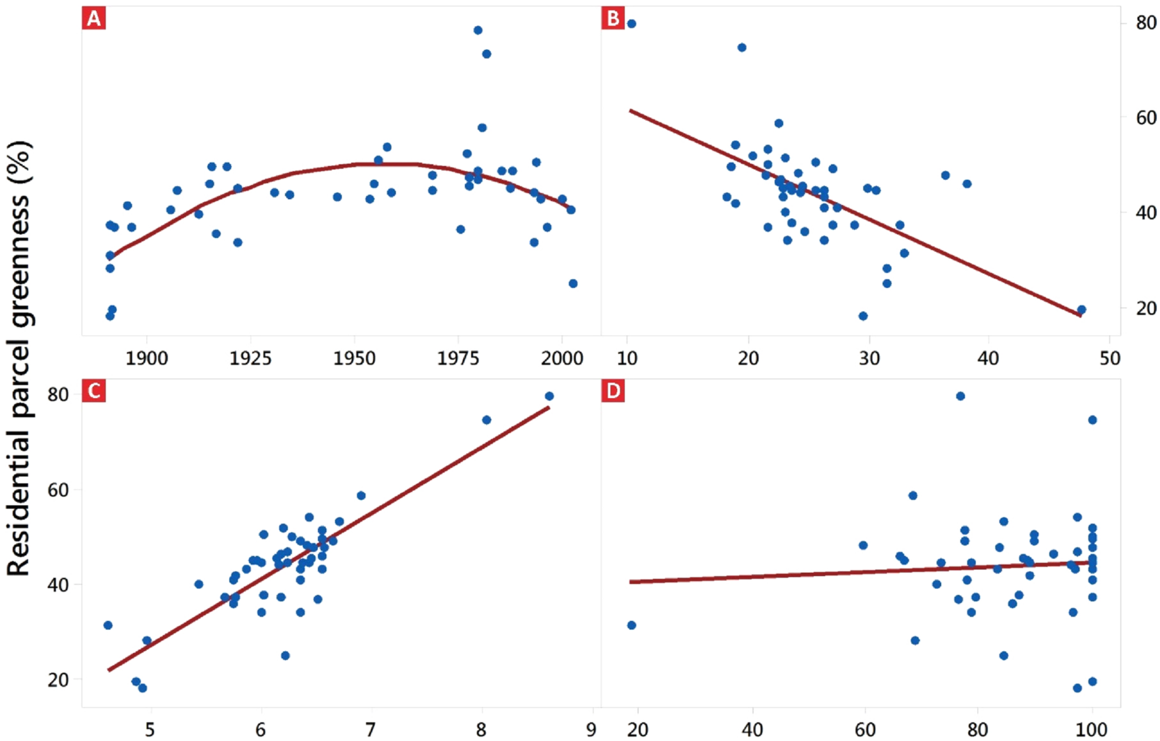

Fig. 4. The percentage of residential parcels that was garden plotted against four factors found to be significantly correlated at the 0.05 level: A, date of neighbourhood development (r^2 = 37.5%); B, road density (m^2/ha) (r^2 = 39.5%); C, residential parcel size (m^2, log transformed) (r^2 = 73.2%); and D, local road percentage (the percentage of properties that are on local roads) (r^2 = 0.5%); for 47 neighbourhoods in Melbourne, Australia. Red lines are fitted regression lines (linear or quadratic).

we derived from Murphy (2012) (4.7%). Our research, in conjunction with these other studies, shows that road easement green space must now be recognised as an overlooked but vital component of our urban green space.

We measured the extent of all public green space apart from road easement to be 12%, which closely matches that measured in a 2017 study quantifying Melbourne's public land (VEAC, 2017), which found that 11.0% of Melbourne was public green space (the study did not include road easements), giving confidence to our results.

The percentage of total land use that is residential garden in Melbourne (29.7%) was substantially higher than Syracuse (16%; Richards et al., 1984) and Stockholm (16%; Colding et al., 2006), slightly higher than in cities in the United Kingdom (21.8–26.8%; Loram et al., 2007), but less than Dunedin, New Zealand (36%; Mathieu

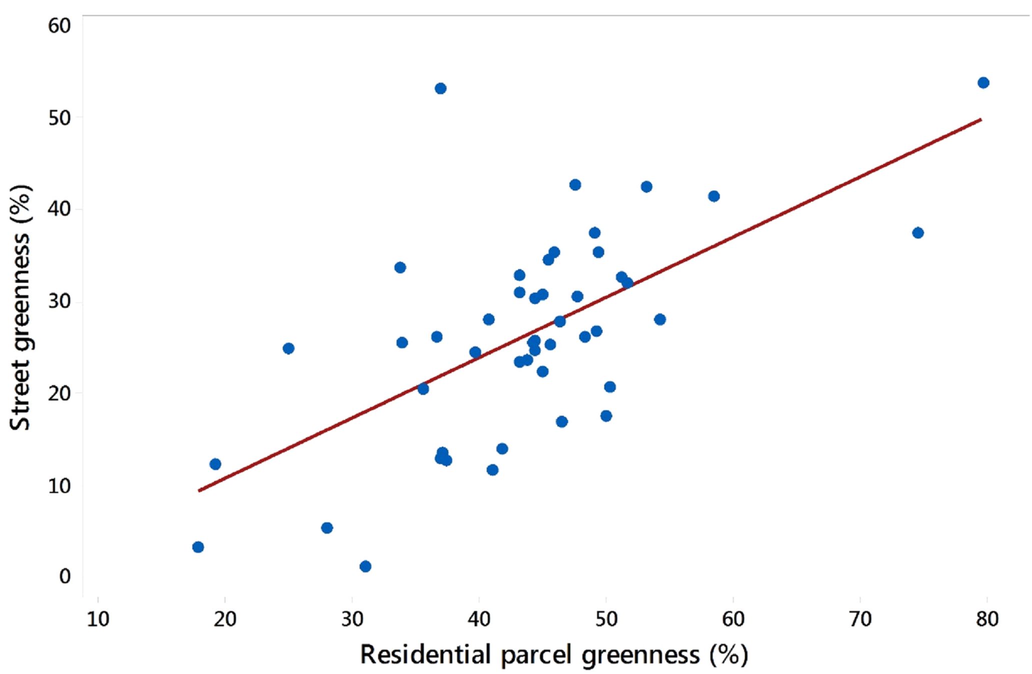

Fig. 5. The percentage of road easement green space plotted against the proportion of residential parcel that is garden, for 47 neighbourhoods in Melbourne, Australia. Red line is fitted linear regression line (r^2 = 40.5%).

et al., 2007). The differences between these figures likely reflect complex interactions between each city's historic mixes of parcel size and shape, housing size and style, population density and sprawl.

In Australia and many other developed nations, new residential development occurs when large development companies purchase substantial tracts of land (e.g. 10–1000 ha) and develop residential housing estates within which hundreds or thousands of houses are constructed (Goodman et al., 2010) typically within 1–3 years in staged land releases. This means residential areas grow development-by-development rather than house-by-house and that roads and houses effectively get built together. Consequently, it is not surprising that we found a correlation between the percentage of the road easement that was green space and the percentage of residential parcels that was garden. Developments are targeted to the state of the housing market (Ball, 1984) and to specific lifestyles, lifestages and price ranges, which lead to differing urban forms and resident demographic patterns (Galster, 1996; Megbolugbe et al., 1996). Urban in-fill, where old areas are demolished and rebuilt as new residential developments, places older residential developments adjacent to newer developments. Together, these produce an urban landscape that is a mosaic and far less homogenous than the simplistic idea of 'suburbia' suggests (Coiacetto, 2007). It is important then to recognise the residential housing subdivision as a significant unit when considering strategies for maximising biodiversity, ecosystem function and human amenity within cities.

4.2. *Factors influencing the distribution of green space*

4.2.1. *Date of neighbourhood development*

It is not surprising that date of neighbourhood development was positively correlated with the percentage of the road easement that was green space because road design has evolved over time. New elements such as roundabouts and chicanes have been introduced (Van Schagen, 2003), and standards developed for setbacks from traffic (e.g. Austroads, 2009) and the location of utilities (Standards Australia, 2013), while older neighbourhoods in Melbourne tend to have tree cutouts rather than continuous verges. Residential parcel frontage, subject to changing planning controls and development styles, also influences the percentage of road easement that is green space, because the vehicle crossover will occupy proportionally less of the road easement adjacent to a property that has a greater street frontage.

The percentage of residential parcels that was garden also correlated with date of neighbourhood development, with the percentage of residential parcels that was garden rising to a peak in neighbourhoods developed in the late 1950s then declining after that decade. These post-World War 2 neighbourhoods are the neighbourhoods of the 'Australian dream' and the quarter acre block (Davison et al., 1995; Timms, 2006), where usually modestly-sized houses sit on relatively large blocks. The decline, since the 1950s, of the percentage of residential parcels that was garden (due to increasing house sizes as well as shrinking block sizes) has had the effect of pushing the responsibility for the provision of urban green space more and more into the public realm (Hall, 2010). Fortunately, our data show that the percentage of the road easement that is green space has been increasing over time, providing greater capacity to offset the ongoing shrinking of residential garden size. Land managers may wish to focus greater effort on using the road easement to provide for biodiversity, ecosystem function and human amenity outcomes in neighbourhoods where the percentage of residential parcels that is vegetated is low.

Developments featuring cul-de-sacs are a distinctive urban form which were first introduced into Melbourne just prior to World War 1 and rose to considerable popularity after World War 2 (Freestone and Nichols, 2013). They are likely to have a complex effect on the extent and distribution of road easement green space. In practice, lower traffic volumes in cul-de-sac streets often mean reduced street width and vegetated verges instead of footpaths (Southworth and Ben-Joseph, 2004). In comparison, grid-style developments have more evenly distributed traffic flows and hence streets are often wider and generally have footpaths on both sides of the road (Stein, 1957). The differing mixes of major and minor roads in cul-de-sac- and grid-style developments further influence the percentage of road easement green space. Cul-de-sacs require wide turning circles (AustRoads, 2016), which increase the percentage of impermeable surface in the road easement. Housing parcels on turning circles have narrow frontages, causing an increased percentage of the land between the street frontage and the kerb to be occupied by crossovers compared to grid-based housing. Given all these influences on green space provision, we were surprised that cul-de-sac percentage was not significantly correlated with the percentage of road easement green space or the percentage of residential parcels that was garden. It may be that in this study the various effects on green space provision at the neighbourhood level cancelled each other out. Differences may be more obvious at the street level scale. Further investigation is required.

4.2.2. *Footpaths*

The percentage of road easement that was green space was correlated with the absence of footpaths, which makes intuitive sense. If the width of the road easement and carriageway remain constant, then neighbourhoods with less footpath will have more verge. There are also other influencing factors; for instance, streets without footpaths tend to be low-traffic volume streets and lane widths may be less for such streets (Southworth and Ben-Joseph, 2004; Austroads, 2009).

4.2.3. *Residential parcel size*

It is also intuitive that the size of residential parcels correlates with the percentage of residential parcels that was garden. Residential parcel size constrains house footprint when residential parcel size is small, and as residential parcel size increases building footprints increase but eventually level out to a maximum size, meaning the percentage of residential parcels that was garden will then steadily increase. Moreover, residential parcel size affects whether a smaller footprint two-storey dwelling of a larger footprint single storey dwelling is built, in turn influencing the proportion of the residential parcel that is garden (Stone, 2004).

4.2.4. *Index of relative social disadvantage*

The Index of Relative Social Disadvantage was an important factor predicting the distribution of the percentage of road easement that was green space. Neighbourhoods with the lowest percentage of road easement green space tended to be the least disadvantaged neighbourhoods, which in our study area were generally the gentrified older inner-city neighbourhoods. These had a high proportion of streets that only had cut-outs for trees within paved footpaths rather than strips of grassed green space within the easement, and consequently had less road easement green space. The neighbourhoods that had the lowest percentage of road easement green space were also the neighbourhoods with the lowest percentage of vegetation within residential parcels, compounding the lack of total green space in these neighbourhoods. It is important to emphasise here that road easement green space in the context of this paper refers to a ground-surface area of permeable road easement, not the extent of tree canopy. In practice, inner-city neighbourhood streets can appear quite green because of the age of the street trees and consequent size of their canopies, whereas at ground level there may be very little green space evident.

The environmental injustice associated with the uneven distribution of public green space has been well documented (e.g. Jenerette et al., 2011; Kirkpatrick et al., 2011). In the scientific literature, a higher percentage of vegetation is usually associated with greater social advantage, e.g. the ‘ecology of prestige’ (Hope et al., 2003; Grove et al., 2006), while in this study a higher percentage of vegetation is associated with greater social disadvantage. This illustrates that cities are not all the same. Australian cities have not experienced ‘white flight’ in the way American cities have (Kruse, 2013; Sadler and Lafreniere, 2017), and older inner-urban areas with small block sizes and little green space in the road easement are often gentrified. The demographics are complex in Melbourne: for instance, outer suburban populations in Australian cities can be economically well-off but educationally poor (Kendal et al., 2012). Middle ring neighbourhoods examined in this study tend to have large residential gardens because of post-World War 2 prosperity but are now in economic decline (Newton, 2006). We consider that care must be taken to understand the cultural complexities that can underlie concepts such as ecology of prestige, and not assume what happens in one country applies to another.

4.2.5. Road density

Our findings of a negative correlation between road density and the percentage of residential parcels that was garden match those of a study of urban green space in Sheffield, U.K., in which Davies et al. (2008) found housing density (and hence percentage of residential parcels that was garden) to be related to road density and that both influenced the extent of urban green space. Residential parcels tend to be smaller in more road-dense neighbourhoods.

4.3. Urban connectivity

Road easement green space and residential garden combine to connect more traditional forms of open space, e.g. parks and waterways, with significant implications for urban ecological processes (Hope et al., 2003; Gardiner et al., 2013; Bonthoux et al., 2014). Many calls have been made to incorporate residential garden, residents, and local green action groups into a more general approach to promoting urban biodiversity (Colding et al., 2006; Goddard et al., 2010; Beumer and Martens, 2015; Shaw et al., 2017). Our research demonstrates that road easement green space is also a significant green space component that could be integrated into larger-scale urban green space network management to provide connectivity for biodiversity.

4.4. Implications for planning, design and management

The complex variation in the percentage of road easement green space and residential green space across neighbourhoods has significant implications for green space planning, design and management. Planners, urban designers, landscape architects, urban foresters, engineers and ecologists need to work together to reconsider the potential biodiversity, ecosystem function and human amenity benefits flowing from road easements and should become more actively involved in the road design process. The roles that residents, municipal governments and road authorities can play will vary as the mix of resident-managed road easement green space and publicly managed road easement green space shifts across old and new developments, with each combination presenting its own specific opportunities to improve the benefits flowing from road easement green space.

Moreover, different categories of road easement green space (Fig. 1) each have potential to provide contributions to biodiversity, ecosystem function and amenity that may be realised through urban design. For example, roundabouts often have a strong visual presence in the road easement, tend to be away from overhead power lines, and are often associated with entry points to distinct localities. Dall’Ara et al. (2019) used these qualities, along with site-specific understanding of the local ecology, to masterplan the development of a series of roundabouts across an Italian city, deploying rain gardens, tree groves and understorey display presented in various levels of wildness to meet multiple mobility, social, budgetary and ecological objectives. Much of the publicly managed road easement green space in Melbourne was associated with the outer edges of inward-facing residential developments. These outer edges comprise verge adjacent to property rear or side fences uninterrupted by driveways and usually occur along major roads (Fig. 1E). The relatively large patch size and little pedestrian access of this type of road easement green space may make it particularly suitable for specific management actions to promote biodiversity, ecosystem function and human amenity.

Road easement green space is a significant resource and 59.4% of it is managed by residents, which in turn means 21.8% of total public green space is managed by residents. Recognising the substantial role of residents in managing this resource will create opportunities for new pathways for improved urban greening. By encouraging change in the way residents manage their road easement green space we may develop a ‘resource by small gardening actions’ with ‘the positive cumulative outcome of individual garden owners adopting pro-environmental gardening practices’ (Dewaelheyns et al., 2016).

5. Conclusion

This investigation showed the surprising extent of road easement green space, and it suggests that the road easement's capacity to contribute to greening cities has been undervalued. While Richards et al's 1984 paper reported similar extents, the significance of that result appears to have been overlooked, perhaps overshadowed by research (including Richards et al., 1984) quantifying the extent of residential gardens. Only recently, O'Sullivan et al. (2017) have noted their surprise ‘that verges are often excluded from studies assessing the value of urban green space, especially as they can support considerable biodiversity’. Now it is time to recognise that road easement green space is a ubiquitous and significant type of urban green space in its own right. In our Melbourne study, road easement green space constituted 36.7% of total public green space, and many cities in developed nations are likely to exhibit similar patterns. The importance of road easement green space as an urban green space type able to contribute to biodiversity, ecosystem function and human amenity will likely grow given the trends of increasing percentage of road easement that is green space and decreasing percentage of residential parcels that are vegetated, and the ongoing growth and densification of urban areas (United Nations, 2015; Gardiner et al., 2018).

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Appendix A. Mapping

Fig. 6

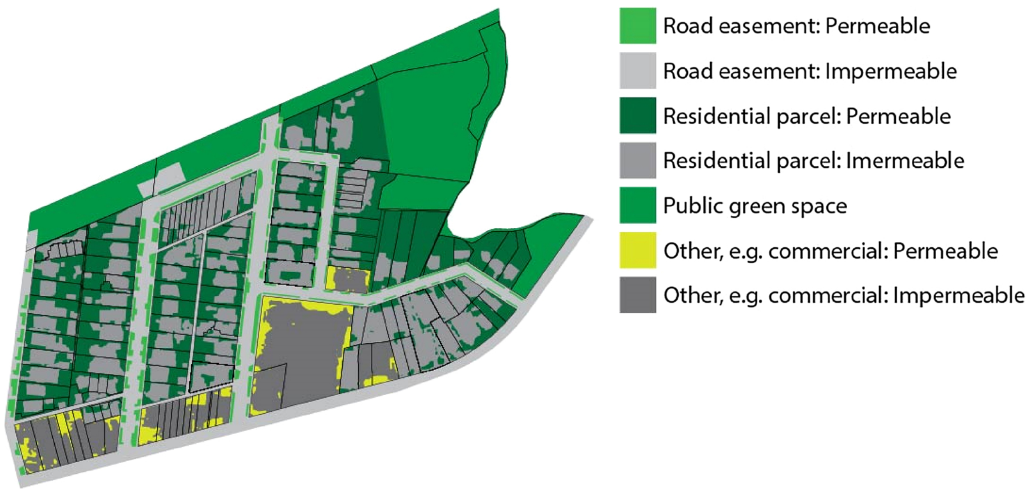

Fig. 6. Sample neighbourhood mapping showing land use categories used to determine extents of road easement green space, residential gardens and public green space.

Appendix B. Data showing nature of the study area

Fig. 7

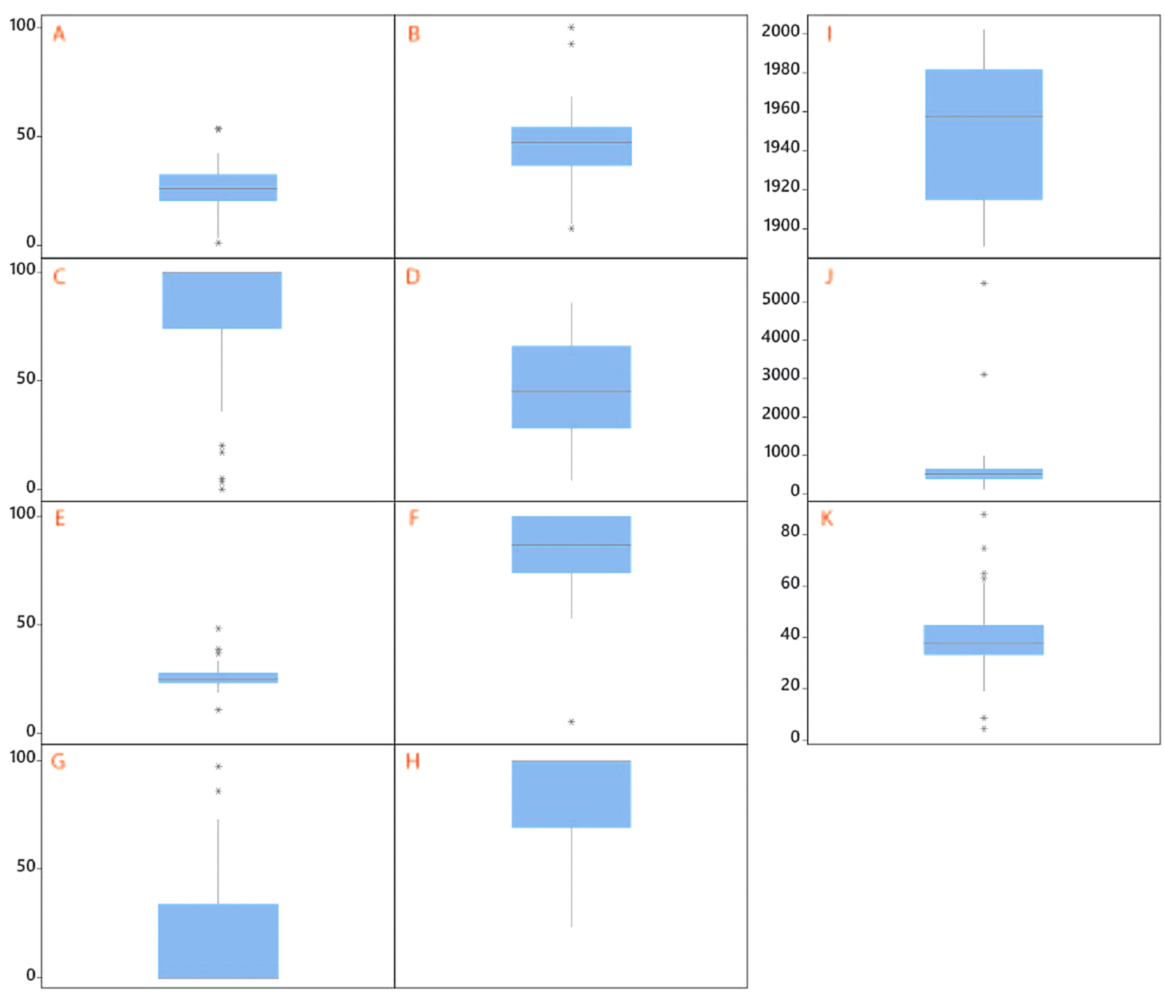

Fig. 7. Boxplots showing the variation in distribution, the interquartile range and outliers for the three green space categories (A–C) and eight factors tested for correlation (D–K) for the residential areas of 47 neighbourhoods in Melbourne, Australia: A, the percentage of road easement that was green space; B, the percentage of residential parcels that was garden; C, the percentage of road easement green space that was resident-managed; D, Index of Relative Social Disadvantage (IRSD); E, road density; F, local road percentage; G, cul-de-sac percentage; H, footpath percentage; date of neighbourhood development; J, residential parcel size; and K, population density. A–H are measured as percentages, I as date, J in m^2 and K as persons per hectare.

Appendix C. Correlations between explanatory factors

Table 2

Table 2
Pearson correlations and p-values for correlations between eight factors investigated as drivers of the percentage of road easement that was green space and the percentage of residential parcels that was garden, for 47 neighbourhoods in Melbourne, Australia.

	Development date	Road density	IRSD	Residential parcel size	Population density	Footpath	Cul-de-sac
Road density	−0.152						
	0.307						
IRSD	−0.160	0.110					
	0.283	0.462					
Residential parcel size	0.664	−0.552	−0.097				
	<0.001	<0.001	0.515				
Population density	−0.610	0.494	0.236	−0.902			
	<0.001	<0.001	0.110	<0.001			
Footpath	−0.621	0.097	−0.070	−0.521	0.353		
	<0.001	0.516	0.639	<0.001	0.015		
Cul-de-sac	0.708	−0.149	−0.216	0.420	−0.395	−0.554	
	<0.001	0.317	0.145	0.003	0.006	<0.001	
Local road	0.153	−0.111	0.139	0.146	−0.066	−0.227	0.066
	0.304	0.456	0.351	0.326	0.657	0.126	0.659

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Chapter 4
Footpaths, tree cut-outs and social contagion drive citizen greening in the road verge

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Footpaths, tree cut-outs and social contagion drive citizen greening in the road verge

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ABSTRACT

Verge gardening is a citizen-led form of public urban greening where residents plant and maintain understorey and trees in the road easement, and it has the potential to significantly increase the diversity and complexity of street greenery which is, most commonly, mown grass and monocultures of street trees. In Melbourne, Australia, two forms of verge gardening – understorey planted by residents and street trees planted by residents– were investigated. The distribution of verge gardens was tested for correlation with a range of urban form and social factors that included the proportion of garden in residential parcels, the proportion of the street set aside as road verge, age of street development, the presence of footpaths, tree cut-outs (tree pits) and social disadvantage. By better understanding the factors associated with verge gardening, we hope planners, urban designers and land managers will be able to better incorporate verge gardening into urban greening strategies and so maximise the biodiversity, ecosystem function and human amenity benefits associated with this significant public activity. We found that verge gardening was common, occurring in almost a quarter (22.1%) of verges and was strongly associated with verges without footpaths, tree cut-outs rather than continuous verges, and verges on local roads rather than collector roads. Neighbourhoods with proportionally more garden had more verge gardening. With the exception of the oldest streets in our study, streets that were newer had more verge gardening than those that were older. The presence of street trees planted by local government was associated with less verge gardening. Verge gardening was less common in neighbourhoods with greater social disadvantage, and spatial contagion increased the likelihood of neighbours having verge gardens. Local government can facilitate verge gardening through engagement, education, adopting alternative street types, using tree cut-outs to better strategic advantage, and targeting areas of social disadvantage.

1. Introduction

Greenspace within the road easement can account for over a third (36.7%) of a city's public greenspace (Marshall et al., 2019; Richards et al., 1984), with most of that greenspace occurring as road verge. The large contribution of the road verge (variously called the nature strip in Australia, and sidewalk buffer, parking strip, tree lawn, or boulevard in the USA) to public greenspace means it has considerable, but generally overlooked, potential to contribute to biodiversity, ecosystem function and human amenity in urban landscapes. Typically, the vegetation of this ubiquitous form of public greenspace is lawn, often with monocultures of street trees planted by local government. However, residents often undertake their own urban greening by planting street trees or by creating gardens on the verge. For instance, in Ann Arbor, Michigan, resident-planted understorey was present in the road verges of 11% of properties (Hunter and Brown, 2012). Regulations regarding the maintenance of, and permitted plantings within, the road verge vary from country to country and city to city. For instance, in Australia, street trees are always the responsibility of a government authority, while in many cities in the USA (*e.g.* Portland, Oregon; Donovan and Butry, 2010) the resident must maintain a street tree adjacent to their property; and in Canberra, Australia, residents are permitted to grow vegetables in the road verge (Lawson, 2016), while in the municipality of Monash, Australia, they are not (City of Monash, 2014). Verge gardening practices are likely to be widespread internationally (Guitart et al., 2012; Hadden, 2014; Cogger and Brown, 2016; African Conservation Trust, 2017; Yamamoto, 2017; Natural Habitats, 2018), but this has not been quantified.

Verge gardening can increase the benefits that flow from urban greenspace. For instance, by replacing lawn and spontaneous weedy

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vegetation with more complex understorey and introducing a suite of native species, verge gardening can improve habitat and resources for a range of fauna species (Threlfall et al., 2016) and can improve faunal connectivity between patches of larger habitat (*e.g.* parks) (Fernández-Juricic and Jokimäki, 2001; Oprea et al., 2009; Munshi-South, 2012). By increasing the extent and diversity of street greenery, verge gardening makes cities more liveable (Säumel et al., 2015) by regulating temperature (Leuzinger et al., 2010; Gillner et al., 2015; Sanusi et al., 2015), mitigating stormwater runoff (Breen et al., 2004; Armson et al., 2013), mitigating air pollution (Vailshery et al., 2013), improving human health through reduction in stress and improvements in social cohesion (De Vries et al., 2013), reducing early childhood asthma (Lovasi et al., 2008) and reducing depression (Taylor et al., 2015). For many urban residents street greenery is also likely to be the nature they have most contact with on a day-to-day basis, and such routine contact may increase the likelihood of residents implementing pro-biodiversity practices (Prévot et al., 2018).

To develop pathways to greater urban greening that realise the many benefits of verge gardening we need to know the extent of verge gardening and its drivers. However, only one study quantifying the extent of verge gardening exists (Hunter and Brown, 2012) and no specific research exists on the drivers of verge gardening, though in the absence of this information, research investigating residential gardening and the social drivers of gardening behaviour may lead to a greater understanding of verge gardening (*e.g.* Cook et al., 2012).

1.1. Residential gardening behaviour

Residential gardening behaviour is influenced by residents' values, beliefs, attitudes and norms (Nielson and Smith, 2005; Larson et al., 2011), social ideals (Feagan and Ripmeester, 1999), environmental concerns (Kiesling and Manning, 2010; Kirkpatrick et al., 2012) and symbolic identity (Mustafa et al., 2010). Income, education, property value, gender and home ownership have all been linked to gardening behaviour (Zagorski et al., 2004; Grove et al., 2006; Larsen and Harlan, 2006; Zhou et al., 2009; Kendal et al., 2012), as has population density (Nesbitt et al., 2019). Residents may be motivated to verge garden as a way of providing household food, community engagement, and reducing food miles (Van Veenhuizen, 2014). As a form of guerrilla gardening, residents may be motivated to verge garden as a way of activating public space by gardening where gardening is not authorised, emphasising sustainability, the waste of urban systems, the arbitrariness of authority, and promoting access to land for those who have none (Finley, 2013; Reynolds, 2014) (Fig. 1A). Verge gardening can also be a community action where, by introducing vegetation into the street or by meticulously mowing and weeding the verge lawn, residents act to beautify the neighbourhood (Butler-Bowdon, 2001; Yamamoto, 2017).

Gardening practices are influenced by the amount of garden available. Yard size affects both tree planting (Ossola et al., 2019) and the proportion of the yard that is vegetated (Lin et al., 2017), and irregular yard shapes produced by cul-de-sac style developments reduce tree planting (Nielsen and Jensen, 2015), as does building from fence to fence (Hall, 2010).

1.2. Urban form and verge gardening

The prevalence of verge gardening may also be influenced by the form of the road verge because road verges are not uniform. While a typical verge in Melbourne is a continuous strip of green bordered by the kerb on one side and the footpath on the other (Fig. 1B), many streets do not have footpaths, and in these cases the verge can appear to be the continuation of the front yard (Figs. 1C and D) all the way to the kerb. Conversely, some streets are nearly all footpath between the property line and the kerb and have tree cut-outs rather than typical verges – a tree cut-out (sometimes called a tree pit) being usually an area of 1–4 m^2 cut from the footpath or occasionally the carriageway (the part of the road easement for vehicles) and planted with a street tree (Fig. 1E). Each of these verge configurations may affect the likelihood of a resident verge gardening.

Streets vary in other ways that may affect the likelihood of verge gardening. Traffic volume and speed impact the way residents use their streets, as does the layout of the broader road network, with cul-de-sacs adversely affecting the walkability of a neighbourhood (Giles-Corti et al., 2011). Cul-de-sacs have been associated with creating greater sense of community than through-roads, and more use of the street as recreational public space by residents (Hochschild, 2013), suggesting that verge gardening might be more common in cul-de-sacs than through-roads. In new neighbourhoods, street density is positively correlated with canopy cover, but this relationship declines as neighbourhoods age (Lowry et al., 2012), suggesting possible correlations between neighbourhood age, canopy cover and verge gardening. The presence of existing vegetation in the verge, such as street trees planted by the local government, and the absence of groundcover (sometimes replaced by gravels, Fig. 1F), may also influence the likelihood of resident verge gardening.

Road verges are not uniformly distributed across cities and it may be that verge gardening is also influenced by the same factors that drive this unequal distribution. We previously identified factors associated with the proportion of the road easement that was road verge (Marshall et al., 2019) and found that it was positively correlated with the percentage of residential parcels that was garden, the degree of social disadvantage, residential parcel size, and development date. Spatial and social contagion (where residents' social interactions drive similarity in garden style) are well documented in residential gardening in the UK, USA and Australia (*e.g.* Askew and McGuirk, 2004; Goddard et al., 2013: Kirkpatrick et al., 2007; Kirkpatrick et al., 2011; Nassauer et al., 2009). Spatial contagion may also influence verge garden distribution, *i.e.* verge gardening may tend to encourage further verge gardening in neighbouring properties. Hunter and Brown (2012) found that verge gardens tended to cluster, and that clustering was strongest at the neighbour level.

1.3. This study

We investigated the extent and distribution of verge gardening in the sprawling, suburban city of Melbourne, Australia. We examined the distributions of two forms of verge gardening (understorey planted by residents and street trees planted by residents) for correlation with a range of factors that may influence road verge extent and distribution. These factors included the proportion of residential parcel that was garden, the proportion of the street that was road verge, indicators of social disadvantage, the presence of local government street trees, street age, road density, the type of road verge, the presence of footpaths, and road type. In particular, we sought to answer the following questions. First, what percentage of residential properties had verge gardening? Second, at the neighbourhood level, what urban form and social factors were associated with the distribution of verge gardening? For instance, we hypothesised that verge gardening be more common in neighbourhoods with less yard than in neighbourhoods with more yard, as residents turn to public space to make up for a lack of private greenspace. And finally, was spatial contagion evident, *i.e.* was the likelihood of one property having verge gardening greater if a neighbouring property had verge gardening – and if it was present, did it correlate with any of the factors potentially associated with verge gardening? Greater understanding of the factors associated with verge gardening will enable planners, urban designers, landscape architects, road engineers, ecologists, social scientists and land managers to better incorporate this significant public activity into urban greening strategies.

Fig. 1. Examples of verge gardening found in the study area in Melbourne, Australia. A: Guerrilla gardening, emphasising sustainability through recycled materials. **B:** A standard or typical road verge with the verge bounded by the kerb, footpath and driveways. **C:** A verge where the absence of footpath and no fence combined to allow an uninterrupted extension of a front garden to the kerb. **D:** Elliston Estate, Melbourne, where a sense of community and shared landscape was created in part through the deliberate absence of footpaths. **E:** A tree cut-out with understorey planted by residents. **F:** Verge gardening where the grass has been replaced by gravel. **G:** A very wide verge providing space for future widening of the carriageway. **H**: A long uninterrupted verge beside the back and side fences of the properties of an inward-facing housing estate. **I:** A well-vegetated outstand for traffic calming. **J:** A road closure transforms the verge into a small local park. **K:** Planter boxes make a statement in a streetscape with only tree cut-outs and small front yards to provide greenery. **L:** With the footpath moved slightly towards the kerb, a verge space between the footpath and the fenceline makes itself more appealing to garden than the verge between footpath and kerb. All photos by author.

2. Methods

2.1. The study area

Melbourne, (founded 1835, latitude 37.814, longitude 144.96332) has a population of 5 million (Australian Bureau of Statistics, 2018) and is the second largest city in Australia. Within the urban growth boundary of Melbourne, we restricted our study area to: 1) residential areas, because internationally they are a dominant urban land use type (*e.g.* Pauleit and Duhme, 2000); and 2) to neighbourhoods that were within the Victorian Volcanic Plain bioregion, because the soils of the Victorian Volcanic Plain are distinctive and affect gardening conditions. Most of Melbourne's north and west falls within the Victorian Volcanic Plain (Fig. A1).

Melbourne's inner suburbs have mostly small blocks and terraced single- or double-storey housing, along with higher density urban infill. After World War 2, housing in now Melbourne's middle ring exemplified the 'Australian Dream' (Timms, 2006) of a single-storey detached house on a quarter-acre (0.1 ha) block. Outer suburbs sprawl unabated, with Melbourne being the world's 32nd largest city by area but 955th by population density (Demographia, 2017).

Road verges, usually 0.5–2 m wide, are common across all Melbourne, though in areas developed pre-World War 1 they are less so, with tree cut-outs more common. In some cases, road verges can be very wide, *e.g.* 20 m, when land is set aside for road duplication or for major underground infrastructure (Fig. 1G). When crossovers are absent, road verges run the length of the street, as occurs in older streets developed before the car became popular. Inward-facing housing estates in the middle and outer suburbs frequently have long, uninterrupted road verges along their outer edges, corresponding to the side and rear fences of individual properties (Fig. 1H).

Greenspace within the road easement is not limited to road verges. Medians separating traffic lanes, roundabouts, traffic calming outstands that narrow the carriageway (Fig. 1I), areas of green obtained by closing roads (Fig. 1J), planters on the footpath (Fig. 1K), and connections between cul-de-sacs, are all forms of greenspace found within the road easement.

Melbourne's road verges are managed by both residents and government authorities, with residents required to maintain the lawn, which they do generally by mowing, and any understorey, while municipal authorities maintain the trees. Local government guidelines that regulate verge gardening vary enormously. For instance, plantings may be not permitted at all, or may be restricted by height, woodiness, weediness, edibility or provenance (indigenous, native, exotic). Residents who verge garden often contravene these regulations.

2.2. Data collection

The geographic unit for data collection was the Australian Bureau of Statistics (ABS) 'Statistical Area 1' (SA1). SA1s are useful units of study because they capture urban form well, being generally homogenous in development age, housing stock and street layout, with internal connectedness and their boundaries respond to landform and historical boundaries (Australian Bureau of Statistics, 2016). They also facilitate the use of ABS demographic data. SA1s have an average population size of approximately 400 people and are hereafter referred to as 'neighbourhoods'. Of the 4589 neighbourhoods entirely within the UGB and VVP that overlapped land zoned residential, we randomly chose 50. Of those, we excluded 3 because they were still in development or were high-density residential. The remaining 47 were mostly residential, though commercial strips, churches, schools, hospitals and sports fields were also present. In total, the 47 neighbourhoods contained 5151 residential properties in 10 local government areas which were the survey 'population' for this study.

Property-level data collected during the physical survey included two measures of verge gardening: 1) understorey planted by residents; and 2) street trees planted by residents. Other data collected included: the presence of street trees planted by local governments; verges where grassed surfaces had been replaced by non-grass surfaces (*i.e.* gravels); and footpath presence. Each property's road verge was categorised as standard (*i.e.* a continuous verge only interrupted by a vehicular crossover), tree cut-out, or none. When tree cut-outs were too small around the base of the tree to permit understorey plantings or had compacted toppings that made understorey planting impossible, the adjacent property was logged as having no road verge.

Trees present on verges were categorised as planted by residents or by local government based on species, location and context. Each local government was contacted, and expert opinion sought on species likely to be planted by them. Trees that occurred consistently in a street were considered planted by local government and those outside typical local government species palettes were considered planted by residents. Eucalypts occurring in streets otherwise planted with exotic trees were considered planted by residents, although large eucalypts were classified as planted by local government. All conifers and fruit trees, except for olives (*Olea europaea*), were considered planted by residents. Olive trees were considered planted by residents when they were not so numerous in the street as to suggest they were part of a local government planting program. Reference was also made to the characteristics of the garden of the property adjacent to the verge, with trees closely fitting the garden's palette generally classified as planted by residents. Location was also considered, with trees planted too close to street trees planted by local government, poles, street corners, or otherwise being in odd locations such as at the edge of a wide verge, being considered planted by residents. The presence of multiple trees within a property's road verge indicated some were probably planted by residents. Trees with pipes for truck watering were considered planted by local government. Other indicators included staking style and the presence of protective cuffs to contain mulch. Some trees considered planted by residents may have established spontaneously, but the number of these was likely to be low.

Street-level data for all properties was also collected. These included: road class, and whether the street was a cul-de-sac. Road class was determined by reference to the Victorian Government's VicMap Transport dataset (Department of Environment, Land, Water and Planning, 2017). These data were combined to produced four Road Type categories: major roads (combining road classes 2 and 3); collector roads (road class 4); local roads (road class 5); and local cul-de-sac roads (road class 5). Some properties were located on service roads of Class 2 or Class 3 roads (*i.e.* subsidiary roads parallel to the main road and providing residents access). These properties were classified as being on Class 4 roads to account for the relative protection afforded by the service road. Each street was assigned a date of development category by using historic street directories (1850–1917; 1918–1947; 1948–1957; 1958–1967; 1968–1977; 1978–1987; 1988–1997; and 1998–2008) (Collins' street directory, 1922–1963; Melway, 1964–2009; Morgans' street directory, 1917–1944; Moulton's street directory, 1912–1916) and Sands and McDougall (1860–1911) records of property ownership.

Neighbourhood-level data on the percentage of the residential parcel that was garden, road density, and the Index of Relative Social Disadvantage (IRSD) percentile was obtained from a companion study of the same neighbourhoods (Marshall et al., 2019). The Index of Relative Social Disadvantage is a general socio-economic index developed by the Australian Bureau of Statistics index that 'summarises a range of information about the economic and social conditions of people and households within an area' (Australian Bureau of Statistics, 2015) including income, employment status, education, rental costs, marital status and English language skills, and that more accurately captures the complexity of social disadvantage than single-factor measures (Australian Bureau of Statistics, 2013).

Table 1
Generalised Linear Mixed Model (GLMM) testing for ten factors predicting the presence of two types of verge gardening – resident-planting of understorey, and resident-planting of street trees – in 47 neighbourhoods and 5151 properties in Melbourne, Australia. Level refers to the three levels of nesting of the explanatory factors.

Factor	Level	Understorey P-value	Tree P-value
Index of Relative Social Disadvantage	Neighbourhood	< 0.001	0.529
The proportion of the road easement that was verge	Neighbourhood	0.574	0.280
The proportion of garden in residential parcels	Neighbourhood	0.160	0.003
Road density	Neighbourhood	0.248	< 0.001
Road type	Street	< 0.001	0.007
Date of street development	Street	< 0.001	< 0.001
Footpath presence/absence	Property	< 0.001	< 0.001
Grass presence/absence	Property	0.093	0.798
Road verge type (tree cut-out or standard)	Property	0.021	0.025
Local government tree presence/absence	Property	0.003	< 0.001

2.3. Analyses

Data were modelled in Genstat 18 (VSN International, 2011) using Generalized Linear Mixed Models (GLMMs), and nested at the property, street and neighbourhood scales (Table 1). We tested two measures of verge gardening – understorey planted by residents and street trees planted by residents – against ten explanatory factors (Table 1).

To test for spatial contagion, we used the join count method of measuring spatial connectivity in lattices (Cliff and Ord, 1981; Schabenberger and Gotway, 2005, Appendix B). To eliminate confounding urban form factors, we identified continuous sets of properties in the same street and with the same footpath presence/absence. Because our study excluded properties on strata titles, non-residential buildings and vacant lots, often a street block had multiple such property sets. Because we were comparing pairs of neighbours with verge gardening, our analysis could only use property sets with more than one property with verge gardening and without all properties with verge gardening. These sets (n = 139 for understorey verge gardening, n = 136 for resident-planting of street trees) contained between one and 37 properties. In a set, each property was given a value B if verge gardening was present or A if absent. Two properties were 'joined' if they were neighbours. By comparing the observed number of BB joins with the expected number of BB joins given the number of B properties within a set of length *n*, denoted E(BB), we calculated if the set had more BB joins than expected (Fig. B1). If the observed number of BB joins was greater than expected, the set supported the proposition that spatial contagion was present. We calculated an average over all property sets. If the average was greater than 1.0 then social contagion was present. Variance was calculated using a Monte Carlo method, and p values derived using a two-sided test.

To investigate factors that may have influenced the presence of spatial contagion, a nested mixed effects model was run in Minitab 18 for understorey verge gardening and resident-planting of street trees. In each case, the response variable was the ratio BB/E(BB), weighted by the inverse of its variance for each property set, which was tested against the ten explanatory factors discussed above.

3. Results

3.1. The extent of verge gardening

Verge gardening was associated with 1138 (22.1%) of the 5151 properties examined. Of these 1138 properties, 769 (67.6%) had understorey planted by residents, 642 (56.4%) had street trees planted by

Table 2
Odds ratios (and, for ease of interpretation, their inverses below) for eight factors significantly correlated with the likelihood of understorey planted by residents or street trees planted by residents in 47 neighbourhoods in Melbourne, Australia. Odds ratios for road type are shown compared to major roads; for street date, compared to 1850–1917; footpath presence is compared to footpath absence, standard road verge is shown compared to tree cut-out, and local government tree presence is shown compared to absence. The Index of Relative Social Disadvantage (IRSD), the percentage of residential parcels that was garden (residential parcel greenness) and road density are factors are continuous, and odds ratios are for a 10% increase in their measure.

	IRSD (per 10%)	Residential parcel greenness (per 10%)	Road density (per 10%)	Footpath present compared to absent	Standard road verge compared to tree cut-out	Council tree present compared to absent	Road type: Major	Collector	Local	Cul-de-sac	Date of street development: 1850–1917	1918–1947	1948–1957	1958–1967	1968–1977	1978–1987	1988–1997	1998–2008
Understorey	1.11			0.19	0.57	0.75	1	1.32	3.74	3.56	1	0.39	0.59	0.42	0.57	0.87	0.80	1.07
	0.90			5.27	1.75	1.33	1	0.76	0.27	0.28	1	2.54	1.70	2.39	1.76	1.15	1.26	0.93
Tree		1.24	1.44	0.48	4.92	0.25	1	3.02	3.20	3.87	1	1.78	0.89	0.81	0.68	1.38	0.82	0.69
		0.80	0.69	2.06	0.20	3.98	1	0.33	0.31	0.26	1	0.56	1.12	1.23	1.47	0.73	1.22	1.44

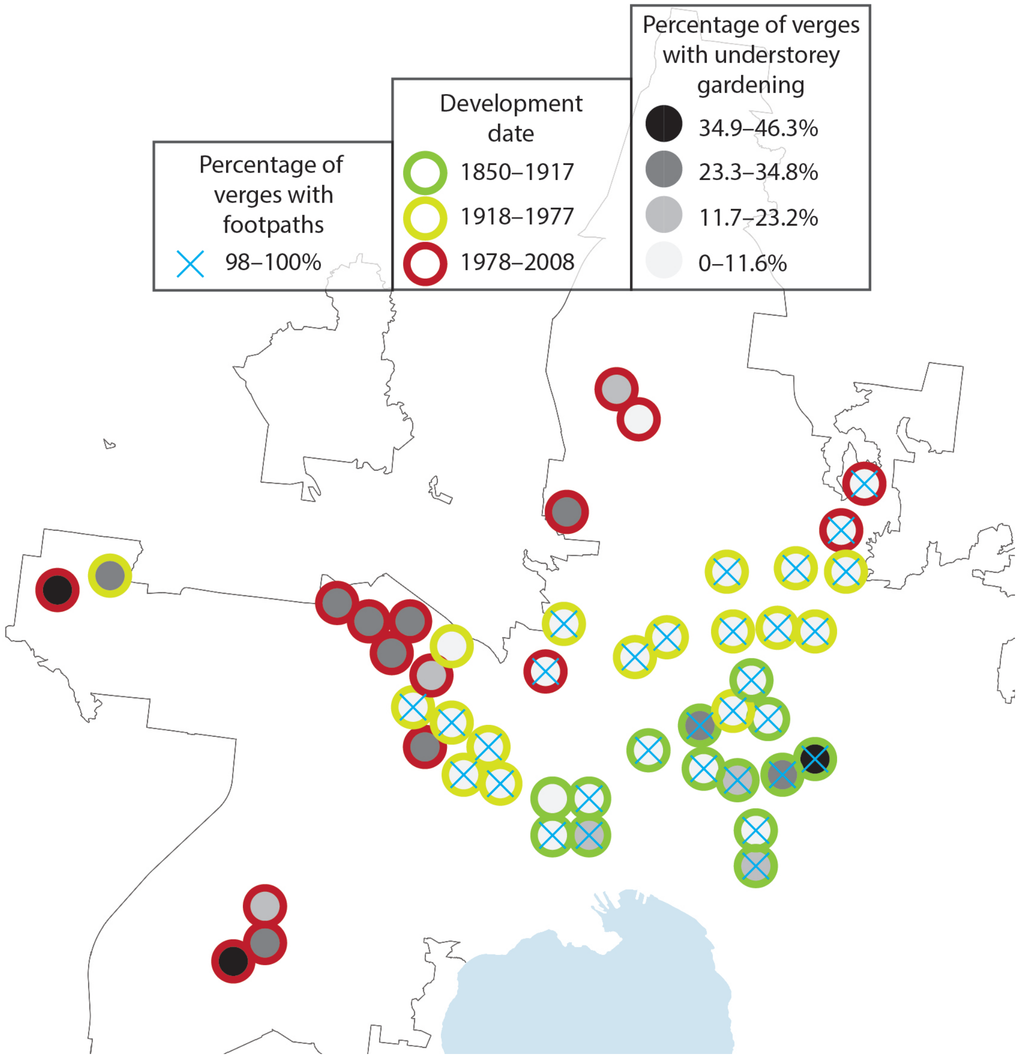

Fig. 2. The percentage of residential properties with understorey verge gardening, and with footpaths, and date of development, for 47 neighbourhoods in Melbourne, Australia. The grey line shows Melbourne's Urban Growth Boundary.

residents, and 273 (24.0%) had both. 16.1% of properties with street trees had street trees planted by residents. The frequencies of verge gardening, street trees and the urban form elements mapped are shown in Appendix C.

3.2. Footpath presence drives distribution of understorey verge gardening and resident-planting of street trees

The absence of footpaths was a major driver of verge gardening. Properties with no adjacent footpath were 5.27 times more likely to have understorey verge gardening and 2.06 times more likely to have resident-planted streets tree than those with a footpath (Table 2). In this study, road verges without footpaths only occurred in streets developed after 1977, and the percentage of understorey verge gardening and resident-planting of street trees was much higher in those neighbourhoods (Figs. 2 and 3). Moreover, neighbourhoods built after 1977 with only streets with footpaths had low percentages of understorey verge gardening and resident-planting of street trees.

3.3. Tree cut-outs drive distribution of understorey verge gardening

High percentages of understorey verge gardening were present in neighbourhoods developed prior to 1918 (Fig. 2) despite all properties in these neighbourhoods having footpaths. These early neighbourhoods often have tree cut-outs rather than standard road verges, and properties with tree cut-outs were 1.75 times more likely to have understorey verge gardening than properties with standard verges (Table 2).

3.4. Additional factors influencing the distribution of understorey verge gardening

Verges without a street tree planted by local government were 1.33 times more likely to have understorey verge gardening than those with a street tree planted by local government. Local roads were the most likely to have understorey verge gardening, 3.74 times more likely than major roads, and cul-de-sacs were 3.56 times more likely. The most recently constructed streets (1998–2008) were the most likely to have understorey verge gardening, 1.07 times that of the oldest streets (1850–1917), while streets built 1918–1947 were least likely to have understorey verge gardening. Understorey verge gardening was significantly correlated with the Index of Relative Social Disadvantage, with a 10% increase in social advantage associated with an 11% increase in the likelihood of understorey verge gardening. (Tables 1 and 2.)

3.5. Additional factors influencing the distribution of resident-planting of street trees

Street trees planted by residents were significantly and positively

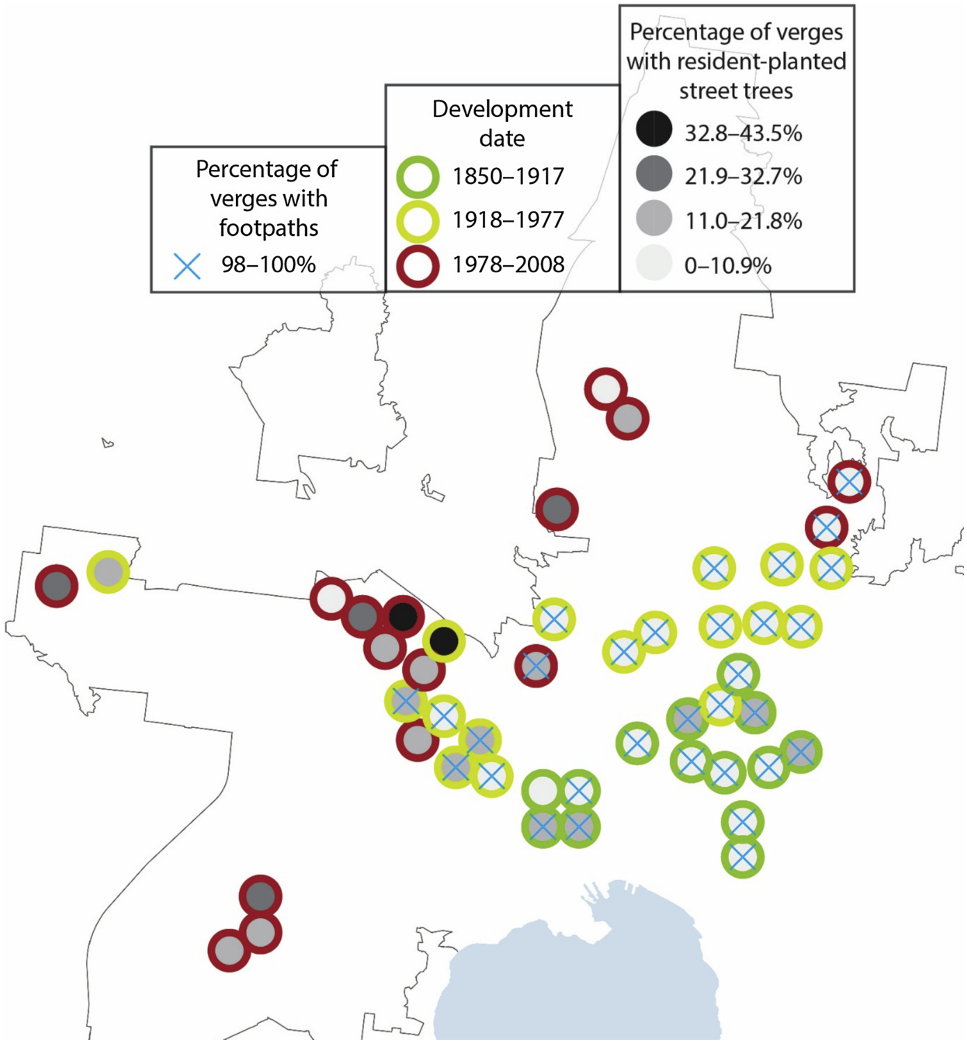

Fig. 3. The percentage of residential properties with resident-planted street trees, and with footpaths, and date of development, for 47 neighbourhoods in Melbourne, Australia. The grey line shows Melbourne's Urban Growth Boundary.

correlated with the percentage of the residential parcels that was garden, and a 10% increase in that percentage was associated with a 24% increase in the likelihood of resident-planting of street trees. Road density was also significant, with a 10% increase associated with a 44% increase in the likelihood of resident-planting of street trees. Verges without footpaths were 2.06 times more likely to have street trees planted by residents than verges with footpaths. Verges without a street tree planted by local government were 3.98 times more likely to have street trees planted by residents than those with a street tree planted by local government. Cul-de-sacs were the most likely to have verges with street trees planted by residents, 3.87 times more likely than major roads. Local roads and collector roads were also far more likely than major roads to have resident-planted street trees. Streets in the second oldest category of streets (1918–1947) were the most likely to have street trees planted by residents, 1.78 times that of the oldest streets (1850–1917), while streets in the 1968–1977 category and the newest streets (1998–2008) were the least likely to have street trees planted by residents. (Tables 1,2.)

3.6. The presence of spatial contagion

Significant spatial contagion was present for understorey planted by residents ($p = 0.032$), but not for street trees planted by residents ($p = 0.269$). The ratio BB/E(BB) for understorey planted by residents was 1.09, indicating that we observed 9% more verges with understorey planted by residents than expected. Testing of factors for correlation with spatial contagion showed no significant correlation at the 0.05 level for any factor.

4. Discussion

In Melbourne, residents' desire to green public space is considerable, evidenced by the fact that verge gardening is common, being present in 22% of properties and on almost any block in any neighbourhood. In the USA, a study in Ann Arbor, Michigan, found verge gardening present in 11% of properties (Hunter and Brown, 2012), though that was likely to be an underestimate of residential verge gardening because it didn't include street trees planted by residents and did include non-residential properties.

4.1. Neighbourhood design influences verge gardening

Our research shows that the design of a neighbourhood influences the way residents engage with verge gardening. Two factors – the absence of footpaths and the presence of tree cut-outs – together explained much of the unequal distribution of verge gardening across Melbourne.

Residents without footpaths were more than five times as likely to understorey verge garden as residents with footpaths. The lack of a strong border between private space (front yard) and public space (the verge) likely meant residents treated the public space as their own.

Removing footpaths has been used to good effect in historic estate designs, such as Elliston Estate in Melbourne, to emphasize the natural character of the landscape and to promote a sense of shared communal space (Heritage Council of Victoria, 1999, Fig. 1E). Shared streets – called 'home zones' in the UK and 'woonerfs' (literally 'living yard') in the Netherlands – are a contemporary articulation of similar principles (Department of Transport, 2005; Ben-Joseph, 2007; National Association of City Transportation Officials, 2013). They emphasise the entire road easement as a place for pedestrian use, with vehicles no longer given priority, or at least vehicle speeds reduced, and with the kerb often removed to diminish any physical indication of the division between vehicle-priority and pedestrian-priority space. Home zones have numerous social benefits beyond freeing-up more space for urban greening, such as creating safe play areas (Biddulph, 2010). Such alternative street designs will be most effective in areas with low traffic volume, such as cul-de-sacs.

Another implication of the positive influence on verge gardening of footpath absence is that, where footpaths are considered desirable, street greening can be encouraged by moving the footpath towards the kerb (Fig. 1L). The area of verge created between the footpath and the property line will be more likely to be gardened because the yard will appear to extend beyond the property line, just as if no footpath was present though to a lesser extent. Moving the footpath would bring pedestrians closer to traffic, which may be undesirable for safety reasons (Austroads, 2009), but many road verges are wider than necessary for acceptable pedestrian safety. Retrofitting could be undertaken in a targeted way when roads are undergoing upgrades.

Tree cut-outs were the other major factor influencing the distribution of understorey verge gardening, being much more likely to be gardened than standard verges. The reasons for this are unclear and need to be the subject of further research. Possible explanations include: the small size of tree cut-outs makes them appealing to garden (as observed in community gardens, Dennis and James, 2016); tall weeds common in cut-outs prompt gardening; reduced trampling in cut-outs makes them appear more suitable to garden; cut-outs are perceived as less tied to particular households than standard road verges; and gardening in cut-outs, generally in streets with small fenced front yards, relieves the grey ground plane of the street. Streets with tree cut-outs are more likely to have more understorey planted by residents and more resident engagement in public greening, but they also have more impermeable surface than streets with standard verges.

Residents were also more likely to verge garden on local roads and cul-de-sacs than busy roads, possibly because of the increased sense of community in these streets (Hochschild, 2013) or because the volume and speed of traffic on major roads creates a less enjoyable environment in which to verge garden. Public authorities considering increasing vegetation in the road verge may wish to concentrate their efforts on busier roads where residents are less likely to verge garden, while encouraging resident-led greening on more local roads.

4.2. Social factors influence the likelihood of verge gardening

The presence of spatial contagion suggests that verge gardening can be encouraged through social pathways. Nassauer et al (2009) emphasised in a study on the effect of social norms on garden design that 'This is an open door for ecological design innovation whether by developers, government rules or incentives, or neighbours working together to improve the ecosystem services and perceived value of their homes.' In our study, verge gardening occurred despite local government generally being opposed to the practice (*e.g.* City of Whittlesea, 2015; Shire of Melton, 2019), with published local government guidelines to verge gardening emphasising the punitive consequences of non-compliance with by-laws and restricting verge gardening behaviour often without clear reasoning, for example by forbidding plants that are, variously, weedy, woody, non-native, vegetables, or of a certain size. The language of these guidelines emphasises risk minimisation (*e.g.* no trip hazards), and functionality (*e.g.* space for car doors to swing open), while failing to mention benefits (*e.g.* increased sociability, provision of habitat, and improved mental well-being, Säumel et al., 2015). Negative attitudes to non-lawn alternatives are not restricted to local government. In gated communities run by private entities, strictly enforced guidelines control the appearance of non-residential land (Dowling et al., 2010). Encouragement by local government, rather than opposition, could promote substantial beneficial change through the many small gardening actions of residents. Such a change in attitude from local government does appear to be occurring, *e.g.* verge gardening is to be allowed in Canberra, Australia (Lawson, 2016). And planting and maintenance guidelines can have positive benefits for biodiversity, for instance in the USA homeowners associations have been correlated with increased presence of birds (Lerman et al., 2012).

The uneven spread of greenspace in a city in relation to income has been well researched (Jenerette et al., 2011; Kirkpatrick et al., 2011; Kendal et al., 2012). We found that verge understorey gardening was more likely in more advantaged suburbs than disadvantaged suburbs. This may reflect an 'ecology of prestige' (Hope et al., 2003; Grove et al., 2006), *i.e.* that more advantaged members of society have greater resources, in either time or money, to facilitate verge gardening. In comparison, the likelihood of resident-planting of street trees showed no evidence of the influence of disadvantage. This difference might be explained by street trees requiring less resources to maintain than understorey, so the planting of street trees by residents may be less restricted by lack of resources than the planting of understorey. Alternatively, disadvantaged residents may invest scarce resources in planting street trees to achieve a desirable type of street greenery; the high proportion of fruit trees observed suggests residents can value street tree planting that has productive and cultural benefits.

We did not find that residents turned to public space to fulfil gardening needs when their own properties had little space for garden, as we had hypothesised. Rather, properties with a greater proportion of garden were significantly more likely to have verges with resident-planted street trees. It may be that the residents of properties with more garden enjoy having trees, and so are more likely to plant trees in the verge. Or it might be that the they felt that the typical local government policy of one street tree per house was insufficient for a larger house and garden. Conversely, people who build houses or other impervious surfaces over most of their property may not value greenery and so are less likely to plant trees in the verge. Further research is needed.

Street trees planted by local government reduced the likelihood of both understorey verge gardening and resident-planting of street trees. This may mean residents acted to create street greenery when none was present, with tree planting a priority over understorey planting. But it also suggests that residents might green their verge up to a certain point beyond which many consider no further action necessary, *e.g.* one tree is enough, and there is no need for any understorey or further trees.

5. Conclusion

Verge gardening is a citizen-led form of greening public space that is common, being present in 22% of residential properties in our study area and occurring on almost any block in any neighbourhood, and it is a means of improving the biodiversity, ecosystem function and human amenity benefits that flow from urban greenspace (Standish et al., 2013; Haaland and van den Bosch, 2015; Shanahan et al., 2015). By verge gardening, residents can increase the quantity, diversity and complexity of vegetation, which is known to increase the diversity of a range of faunal taxa (Wichmann et al., 2004; Garden et al., 2010;

Threlfall et al., 2017), although this is not universal (Leonard et al., 2018). Consequently, the collective effects of many resident actions could substantially increase the extent and diversity of urban greenery (Goddard et al., 2010) and the many benefits provided by the road verge. Being citizen-led means such change could occur at little financial cost to local government, and local government can facilitate verge gardening through education, providing advice, and by reorienting existing guidelines to encourage residents to verge garden. The social contagion we observed means that actions which promote verge gardening may create a positive feedback loop. Other pathways to greater greening of the road verge include removing or repositioning footpaths and installing more tree cut-outs. Ecologists, social researchers and built environment professionals need to work with road engineers to reimagine the road environment and implement change. Our research provides insight into the complexities of verge gardening and the opportunities to be gained from the unique public greenspace that is the road verge.

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Appendix A. The study site

See Fig. A1

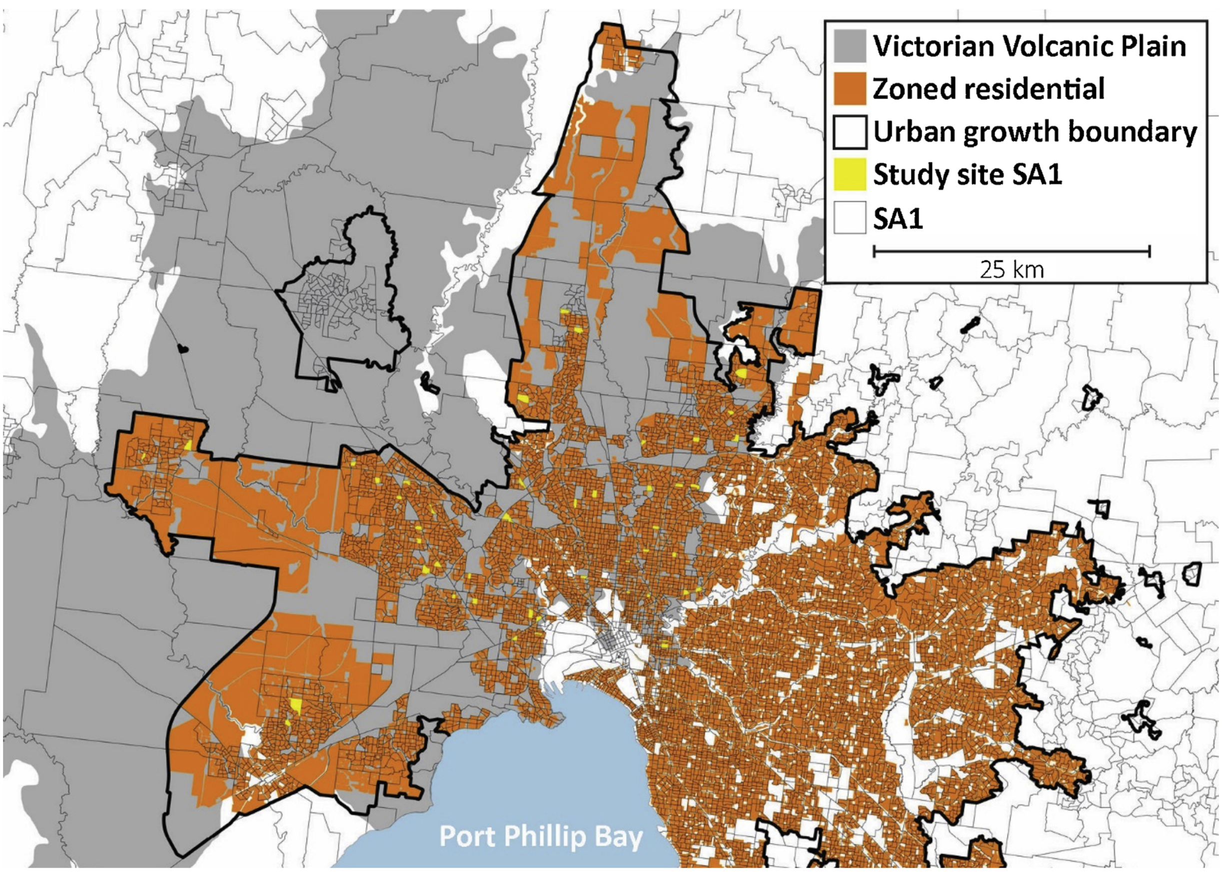

Fig. A1. The relationship between the Urban Growth Boundary, Victorian Volcanic Plain, urban land zoned residential and the 47 neighbourhoods (SA1s) used in the study in Melbourne, Australia.

Appendix B. Analysis of join counts

Denote k = the number of properties in a set of consecutive properties (however defined), and m = the number of properties with the feature of interest on the verges in this set.

We are interested in whether the number of *adjacent* properties with the feature in sets is more than expected, based on an assumption of no social contagion. Adjacent properties with the feature are called 'joins'.

Usable sets for this analysis must have $2 \leq m \leq k$-1. This means they cannot have 0, 1 or k properties with the feature. The reason may not be obvious. It is important to understand that we are analysing 'local' contagion: contagion within sets. Clearly, if there are areas with many sets where $k = m$ (all properties in a set grow plants on their road verges), and other areas where almost all sets have no properties with growth ($k = 0$), this is

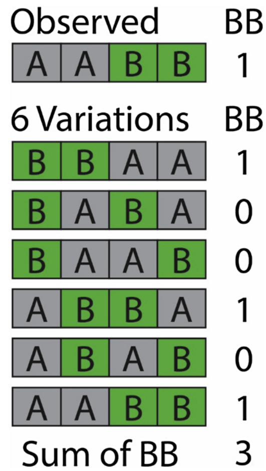

Fig. B1. Example of a continuous set of four properties, two with verge gardening (green, labelled B) and two without (grey, labelled A). The number of BB joins (joins between properties with verge gardening) observed is 1. To calculate the expected number of BB joins, we look at all variations of two properties with verge gardening and two properties without. There are six possible variations. For each we count the number of BB joins. Then we average those results. In this case the sum of the number of BB joins for the six possible variations is 3, so E(BB) = 3/6 = 0.5, *i.e.* on average the expected number of BB joins is 0.5. The ratio of the observed number of BB joins to the expected number of BB joins, BB/E(BB), is therefore 1/0.5 = 2.0. The number of BB joins we observe (1) is twice as high as the number we expect (0.5), supporting the proposition that spatial contagion is present.

an indication of a difference in practice between areas, on a broader scale.

When a set is considered, however, we want to compare the observed number of adjacent properties with the expected number, given the number of properties with the feature, *within the set*. If 0, 1 or k properties have the feature, the observed and expected numbers of joins are necessarily the same. For 0 or 1, there cannot be a join, and none will be observed: the observed and expected numbers of joins are both zero. If all k properties have the feature ($m = k$) then the number of joins is k-1, and (again) the observed and expected numbers of joins are necessarily the same. Hence such sets cannot shed light on local social contagion.

For all other cases, we develop theory for a single set. The variable of interest is denoted by BB in the literature, arising from a generic binary descriptor of black (B) and white (W) (Cliff and Ord, 1981; Schabenberger and Gotway, 2005).

Firstly, consider the expected number of joins for a set with k properties and m with the feature, assuming no social contagion. This can be derived from first principles, using indicator functions. The number of potential joins is k-1. At each of these potential joins, the probability of an actual join is the probability that one of the properties has the feature, multiplied by the conditional probability that the other one does, given that first has the feature. The chance of the first property having the feature is m/k. The chance of the second having the feature, given that the first has it, is $(m\text{-}1)/(k\text{-}1)$. Therefore, the probability that they both have the feature is $[m(m\text{-}1)]/[k(k\text{-}1)]$.

This is the probability for one potential join site. We use the additive mathematical property of expectations: the expected number of joins in the set is the sum of these probabilities, over the number of potential join sites, k-1.

Putting this together: the expected number of joins, based on no social contagion and hence random positions of the m properties with the feature, is $[m \times (m\text{-}1)/k]$. Fig. A2 gives an example.

The variance of BB, under the null hypothesis of no social contagion, is much more challenging. This is because the joins are not statistically independent. It is technically very difficult to derive the variance explicitly. A satisfactory alternative is to obtain the variance by simulating.

For each set separately, for a given m and k, we produced 100 randomly arranged properties; that is each of the 100 simulated sets had the same value of m and k. For this sample, we calculated the number of joins in each of the 100 simulated sets. Then we obtained the mean and variance of the sample. This is an estimate of the true mean and variance.

The sets themselves can be assumed to be statistically independent of each other. Hence the expected value and variance of the total number of joins can be obtained by summing the values for each set.

In these calculations, we used the *actual* expected values, since they can be obtained, and the *estimated* variances, from the simulations.

As a check on the reliability of the simulations, we examined the differences between the simulated expected values and the true means. They were very close. For understorey planted by residents, the average difference was 0.000; zero accurate to three decimal places. The middle half of the distribution was between –0.03 and +0.04. The minimum and maximum differences were −0.17 and +0.17 respectively. This is reassuring evidence that the simulations were performing as designed.

The BB values, their expected values and variance are often small. For small k, BB may be zero. A normal approximation at the individual set level is not reasonable. However, due to the central limit theorem, the sum of BB values will be well approximated by a normal distribution; there are over 100 sets in each case. The BB values are not identically distributed, but this is not critical for the approximate normality; what does matter is that is that there are many of them, and none dominates the sum.

Appendix C. The extents of verge gardening and urban form features in 47 neighbourhoods in Melbourne, Australia

See Table C1

Table C1
Number of properties within the study area, listed by feature, with percentage of total number of properties and, in brackets, percentage of those properties within their subgroup.

	Amount (n)	%
Total	5151	100
Verge gardening	1138	22.1
Verge gardening consisting of understorey planted by residents only	769	14.9 (67.6)
Verge gardening consisting of street trees planted by residents only	642	8.0 (36.4)
Verge gardening consisting of both understorey planted by residents and street trees planted by residents	273	5.3 (24.0)
Properties with street trees (planted by residents or local government)	3982	69.6
Properties with street trees that are planted by local government	3583	63.1 (90.0)
Properties with street trees that are planted by residents	642	11.2 (16.1)
Properties with both street trees planted by local government and street trees planted by residents	243	4.2 (6.1)
Street trees (planted by residents or local government) and understorey planting	599	11.6
Tree cut-outs	155	3.0
Standard verges	4996	97.0
Major roads	198	3.8
Collector roads	452	8.7
Local roads	3695	71.7
Cul-de-sacs	806	15.6
With footpaths	4308	83.6
Without footpaths	843	16.4
With grass verges	4980	96.7
Without grass verges	171	3.3
On streets developed 1850–1917	990	19.2
On streets developed 1918–1947	631	12.3
On streets developed 1948–1957	399	7.7
On streets developed 1958–1967	477	9.3
On streets developed 1968–1977	556	10.8
On streets developed 1978–1987	805	15.6
On streets developed 1988–1997	554	10.8
On streets developed 1998–2008	739	14.3

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Chapter 5
Mowers and growers: road verge gardening is strongly influenced by social norms

Abstract

Road verge gardening is a practice undertaken by residents, which can contribute to the quantity, diversity and structural complexity of street greenery. By understanding the social drivers of verge gardening and influencing behaviours, we can increase benefits from biodiversity, ecosystem function and human amenity. We surveyed residents across Melbourne, Australia, and captured demographic data and beliefs regarding road verge gardening. We used structural equation modelling to test causal models of cognitive constructs underlying verge. We identified cultural background, gardening enthusiasm and level of education as significant factors differentiating respondents who planted verge understorey, planted street trees or did not verge garden. Normative beliefs were the main cognitive construct affecting resident behaviour, with those who did not verge garden more likely to think that others would disapprove of them verge gardening compared to residents who did. Sense of community, beliefs regarding the benefits of verge gardening, and feelings for nature had significant, but less direct, effects. The possibility of changing normative beliefs might provide a significant pathway to promoting verge gardening. Municipal authorities should reorient policy to encourage verge gardening and increase plantings in the verges they maintain. The normative position of the well-manicured lawn can be challenged by municipal authorities adopting low-mow practices.

Introduction

The vegetation of the road verge is usually structurally simple and typically comprises lawn and street trees. It accounts for a large proportion of public urban greenspace in many cities in developed nations, for example 36.7% in Melbourne Australia (Marshall *et al*., in press) and 43.8% in Syracuse, New York (Richards *et al*. 1984). Resident verge gardening is quite common, associated with 22.1% of properties in Melbourne, Australia (Marshall *et al*., unpublished data) and 11% of properties in Ann Arbor, Michigan (Hunter and Brown, 2012). Verge gardening can make road verge vegetation more complex by introducing a new suite of species and replacing lawn (or mown amenity grass) and low-growing spontaneous vegetation with more structurally and floristically diverse understorey. This may improve the biodiversity, ecosystem function and human amenity values of the road verge (Threlfall *et al.*, 2017).

Despite its potential to significantly improve both the ecological and the social qualities of urban green space, the social drivers of verge gardening are not well understood. However, extensive research has been undertaken investigating residential gardening behaviour, as well as guerrilla gardening and community gardening, both of which may occur on road verges. Much of this research is likely to be relevant to understanding resident beliefs and attitudes towards verge gardening

In a review of the residential gardening literature, Kiesling and Manning (2010) identified creative expression and personal meaning, health promotion, production, skill-building, knowledge enhancement, feelings of connection to nature, perceived social benefits, and expression of faith through caring, as key motivations for gardening. Residential yards (i.e. gardens), especially front yards, are public expressions of residents' beliefs, e.g. the importance of conforming to social norms (Larsen and Harlan, 2006), the virtue of order and tidiness (Butler-Bowdon, 2001), and the encultured value of the local indigenous ecology (Seddon and Nossal, 1998), and yards can reflect residents' complex environmental concerns (Kirkpatrick et al. 2012; Mustafa et al. 2010). Lawns can reflect pride of ownership, industriousness and family values (Jackson, 1987) and suburban respectability (Feagan and Ripmeester, 1999). Neatly mown verges can be a symbol of moral probity (Hogan, 2003), and parking on the road verge a sign of the resident's lack of civic character (Baumgartner, 1989). Gardening manifests differently in back yards and front yards (Visscher, Nassauer and Marshall, 2016). Front yards act "as unique buffer zones that connect the home to the outside world, providing services both to residents and passers-by, while simultaneously separating the private from the public realms" (Chalmin-Pui et al., 2019). Front yards may be more formal and have more screening plants and back yards more food plants (Daniels and Kirkpatrick, 2006); may be designed to use less water than back yards (Spinti, St. Hilaire and VanLeeuwen, 2004); may be more social class-specific in landscaping typology than back yards, which may tend to reflect individual preferences more than front yards (Larsen and Harlan, 2006), and may be less treed than back yards (Ossola et al., 2019). Given such differences, It is likely that gardening manifests differently in the road verge as well. Numerous studies have found that gardening behaviour is correlated with age, gender, income, class, education, property value and housing age (see Cook et al. 2012 for a review) as well as rental status (Kirkpatrick, Daniels and Zagorski, 2007).

Unlike domestic yards, road verges are managed by both residents and government authorities. The distribution of responsibility varies from jurisdiction to jurisdiction locally and internationally, but generally in Australia residents are required to maintain the lawn, which they do generally by mowing, and any understorey, while municipal authorities maintain the trees. Any gardening by residents in the road verge is subject to municipal government guidelines. These guidelines vary enormously, for instance plantings may be not permitted at all, or may be restricted by height, woodiness, weediness, edibility, or provenance (indigenous, native, exotic). The residents who go beyond simple mowing of the road verge and expand their activities to gardening in the road verge often do so in contravention of these municipal government guidelines (Marshall *et al*., unpublished data).

Because verge gardening happens on public land, it can be a public statement of beliefs, for instance the importance of helping the environment, or the importance of being in a community. Verge gardeners can seek to activate public space by gardening where gardening is not authorised, emphasising issues of sustainability, the waste of urban systems, and the arbitrariness of authority (Reynolds, 2014). Cultivators of food in public can be motivated by socioecological issues, with the activity acting to collectively promote societal change. Residents who choose to garden their verge can do so seeking to provide food security, reduce food miles, and to create community engagement (Van Veenhuizen, 2014). Guerrilla gardening can be a deliberate means of reshaping urban governance and creating socially innovative urban green space policy (Spijker and Parra, 2018); and Crane *et al*. (2013) have noted that such unexpected interventions can open-up the sort of possibilities essential for ongoing discussions of sustainability.

To better understand how residents' beliefs might influence their perceptions of road verges and verge gardening, we looked to social psychology theories that considered the relationships between values, beliefs and action. Both the Theory of Planned Behaviour (Ajzen, 1991) and Value–Belief–Norm theory (Stern *et al.*, 1999) are theories built on the idea of a cognitive hierarchy, where values are considered more stable and are fewer in number, while the attitudes or beliefs that more directly precede action are more changeable and numerous (Rokeach, 1973; Ives and Kendal, 2014). According to these theories, values are deeply held constructs that underpin the judgements people make of their environment and, whilst stable, are able to change in response to major life events or to a changing world (Bardi *et al.*, 2009). Beliefs, in these theories, are less overarching than values, ‘facts as people perceive them’ (Dietz, Fitzgerald and Shwom, 2005), and more subject to change. An example might be ‘verge gardening is good for the environment’.

Social context influences the way an individual acts through their normative beliefs. Normative beliefs are an individual’s beliefs that an action (for instance verge gardening) is not approved of by another individual, community or institution, e.g. ‘I shouldn’t verge garden because it is against the law’. The stronger an individual’s normative beliefs, the more likely they are to conform to others’ expectations. In the context of this study, mowing the lawn of the road verge is a normative behaviour: not only do by-laws mandate it, but there are also deeper social forces behind the well-tended lawn (Ignatieva *et al.*, 2015), for instance lawn can signify good morals (Hogan, 2003) and community mindedness (Carrico, Fraser and Bazuin, 2013). In contrast, verge gardening tends to be non-normative, in that it is often undertaken despite municipal authority restrictions, and it is – at least on the surface – against community expectations in so much as most verges are not gardened.

The barriers to verge gardening are not just municipal authority by-laws and social pressures. The environment of the road verge may also be a barrier to verge gardening, in that road verges are generally a harsher environment for plants than domestic yards (Gilbert, 1989), likely to be subject to greater trampling, higher pollution levels, higher compaction, increased nutrients from animal waste, lack of irrigation and disturbance due to the need for access to utilities (van der Ree, Grilo and Smith, 2015).

To investigate the psychological constructs influencing residents’ verge gardening behaviour we proposed a relationship between connection to nature, sense of community, normative beliefs, behavioural beliefs regarding the benefits of verge gardening, barriers to the uptake of verge gardening, and the observed behaviour that is verge gardening (Figure 1). In this proposed relationship, we considered connection to nature to be situated higher in the hierarchy between actions and values than the other constructs measured in this study because: A) many have theorised that connection to nature is a fundamental construct underlying pro-environmental behaviours (e.g. Fromm, 1964; Wilson, 1984); B) it is theorised as a value-based attitude (Brügger, Kaiser and Roczen, 2011); and C) connection to nature is a relatively stable personal orientation (Nisbet, Zelenski and Murphy, 2008). Connection to nature is therefore situated in this hierarchy above other, more mutable, beliefs. We proposed that normative beliefs be considered to have two aspects, the strength of those beliefs and a motivation to comply: that is, one might strongly believe that one’s neighbour disapproves of verge gardening, but one may not care much about one’s neighbour’s feelings. Similarly, behavioural beliefs and beliefs regarding the barriers to verge gardening are considered to have two aspects: for example, one may believe the harshness of the street environment to be a considerable barrier to verge gardening, but may choose to disregard or to try to overcome it. These dual distinctions are commonly used to investigate beliefs (e.g. Francis *et al.*, 2004).

Specifically, in this study we wished to answer the following questions: 1) How do residents' beliefs and values affect their gardening and maintenance of the road verge? 2) What are the barriers to greater uptake of verge gardening? And 3) are there differences between residents who verge garden and those who do not? We used questionnaires to obtain data on residents' demographics, verge maintenance and gardening practices, and their values and beliefs on a broad cross-section of factors that may influence their perceptions of road verges, verge maintenance and verge gardening including underlying values such as nature relatedness not directly related to verge gardening. By understanding verge gardening in more detail, we hope to contribute to policies that encourage this behaviour, thereby increasing the benefits flowing from street greenery.

Methods

Site selection

The study area was the western and northern residential suburbs of Melbourne, Australia, that lie within the city's 2016 urban growth boundary (UGB), and which occur on the distinctive soils of the Victorian Volcanic Plain (VVP) bioregion. Greater Melbourne (latitude -37.814, longitude 144.96332) is located in south-eastern Australia, is the capital city of the State of Victoria, and has a multicultural population of five million people (Australian Bureau of Statistics, 2018). It is a relatively young city, founded in 1835. Suburban sprawl is prominent: Melbourne is ranked the world's 32nd largest city by area but 955th by population density (Demographia, 2017). The inner city is mostly gentrified (Stimson, 1982; Baum and Gleeson, 2010) and housing tends to be small terraced cottages or two-storey houses on small parcels (Barrett, 1971), with higher density urban infill. Middle-ring suburbs in the study area were mostly developed after World War 2 in a period that saw rapid population growth and spatial expansion, with an influx of migrants (Burnley, 1975) and a substantial increase in the role of cars (Alexander, 2000). These suburbs are home to the 'Australian Dream' of quarter acre blocks (0.1 ha) (Davison, Dingle and O'Hanlon, 1995; Timms, 2006), though in some areas many are now subdivided. Parcel sizes in the outer suburbs are generally smaller than in middle ring suburbs, though larger than inner-city properties, and housing is almost entirely detached, single storey, and large by international standards (CommSec, 2017; Demographia, 2018).

The Australian Bureau of Statistics (ABS) geographical unit 'Statistical Area 1' (SA1) was chosen as the geographical data collection unit for this study because SA1s are generally homogenous in street layout, housing stock, development age, recognise urban historical and landform boundaries, have internal connectedness, and their use allows direct comparison with existing ABS demographic data. They have an average population of approximately 400 people (Australian Bureau of Statistics, 2016). Forty-seven SA1s were randomly selected from the 4589 SA1s entirely within the UGB and VVP that overlapped land zoned residential. These included 5151 single-title residential properties located across 10 municipal government areas, and were predominantly residential, but also included other land uses such as schools, commercial strips, sports fields and churches. The term 'neighbourhood' is hereafter used to refer to an SA1.

Survey development

To assess issues salient to resident verge gardening, we first conducted an elicitation study, for which interviewees were chosen from peers, and by doorknocking a set of properties that: 1) were in a geographic range similar to the study sites; 2) had a range of the urban form factors known to influence verge gardening (Marshall *et al.*, in press); 3) included properties both with and without verge gardening; and 4) had various levels of verge lawn maintenance. These mostly short interviews (n = 27, duration between two and 97 minutes) were transcribed and analysed in NVIVO (QSR, 2015). Factors identified for further investigation included being a renter or owner, time of occupancy, number of cars owned by household, whether the property was new when the respondent moved in, and whether a design professional was used in the design or planting of the front yard. Further questions were developed relating to the demographic factors identified in the literature as being of significance, such as income, education, age, enthusiasm for gardening, gender and ethnicity. Additional questions were included to assess the change in the elements comprising the road verge, as well as mowing frequency, maintenance time and cost, who the maintainer was, and knowledge of municipal government guidelines, and an open-ended question asked, 'Is there anything else you would like to say?'. Multiple survey drafts were trialled on peers and householders known to the researchers and feedback incorporated. Survey questions are reproduced in full in Appendix A.

Measuring beliefs

We measured eight cognitive constructs: two regarding behavioural beliefs about the value of verge gardening (question sets K and L in Appendix A), two regarding barriers to verge gardening (question sets M and N in Appendix A), two regarding normative beliefs (question sets O and P in Appendix A), sense of community (question set Q in Appendix A) and connection to nature (question set R in Appendix A) (see Table 1 for a summary). We used a seven-point Likert scale for all cognitive questions. To measure the role played by respondents' sense of community, we used six questions from the widely used Sense of Community 2 scale (Chavis, Lee and Acosta, 2008) that related to influence within the community and shared emotional connection within the community (question set Q in Appendix A). To measure values associated with a connection to nature, we used the NR6 (Nisbet and Zelenski, 2013), which is the abbreviated version of the nature-relatedness scale (Nisbet, Zelenski and Murphy, 2008). Psychologists have developed numerous means of quantifying a person's connection to nature, for instance through affect (emotions and feelings for or of nature), cognition (knowledge) and behaviour (experiences in nature, actions) (Tam, 2013). Of the many instruments put forward to measure connection to nature, the multidimensional nature-relatedness scale, and its abbreviated scale the NR6, is considered the most reliable (Tam, 2013; Whitburn, Linklater and Abrahamse, 2019).

Final design and delivery

A professional editor, a designer and a statistician were consulted to establish the final content and physical form of the survey. To maximise response rates we followed several recommendations from the tailored design method of Dilman *et al*. (2014). The survey was hand delivered to all 5151 single-residence properties with 'To the resident' handwritten on the envelope; a covering letter and reply-paid envelope were included, all clearly marked with the University of Melbourne logo to signal the survey was not from municipal government. A financial incentive was offered (win one of three $100 vouchers). Each survey carried a code that identified the address to which it was delivered. Three weeks after delivery, all households from which no response was received were letterboxed with a postcard providing a link to an online survey identical to the print survey but with one additional question, 'What is your address?'. Four weeks later, the online survey was closed, and further postal responses received securely shredded.

Data analysis

We investigated two types of verge gardening: resident-planting of verge understorey and resident-planting of street trees.

Non-cognitive questions were tested using chi-squared tests to determine if responses from properties with resident-planted verge understorey or resident-planted street trees differed significantly from those with no verge gardening. To score responses to our sense of community and nature-relatedness scales, we summed individual responses (scored as 1–7) to the questions in each question set, thus respondents scored between 6 and 42 for each. We then used one-way ANOVAs to test for differences in those scores between respondents with resident-planted verge understorey, resident-planted street trees and those with no verge gardening.

To make a causal model of the relative effects and interactions between verge gardening behaviour and residents' beliefs and attitudes (represented by the eight cognitive constructs the survey measured), we used structural equation modelling (SEM). The use of SEM is common in the social sciences because it focuses on determining though imputation the relationships between unobserved variables (the eight cognitive constructs in our study) (Rahman, Shah and Rasli, 2015). The SEM process requires factor analysis and confirmatory factor analysis prior to final causal modelling. Factor analysis in SPSS 23 (IBM Corp., 2015) tested the internal consistency of the eight cognitive constructs, i.e. their ability to each measure a single underlying concept. Confirmatory factor analysis was conducted in AMOS 23 (IBM Corp., 2015). We then tested for common method bias, i.e. the possibility that our method of study was influencing the results of our study. To do a common method bias test we compared the unconstrained common method factor model to the fully constrained common method factor model.

Results

Of 5151 properties surveyed, 917 valid print and on-line surveys were returned, giving a 17.9% response rate. Online response was poor ($n = 60$). 330 responses included additional comments, totalling approximately 10,000 words. Of the 917 responses, 606 were answered by the person in the household who mostly maintained the road verge, and we then further analysed those responses. Survey questions and summarised responses are shown in Appendix A.

To test if the respondents to our questionnaire were representative of the general population, we compared data for the 917 respondents to our survey with data from the 5151 properties to which the survey was delivered. We compared: resident-planting of verge understorey, resident-planting of street trees, presence of footpath, presence of trees, type of road verge and gender. In each case, except for verge gardening, the difference in responses was not significant ($p < 0.05$, 1 proportion test, Minitab 18). A significantly higher percentage of people with resident-planted verge understorey answered the survey (17.6%) than would be expected given the proportion of resident-planted verge understorey in the total surveyed properties (14.9%, $p = 0.036$).

Most respondents managed their verges themselves (63.7%), though neighbours were involved in managing 7.8% of verges, and gardening services were used for 8.2% of verges (question B). Some verges were not maintained by anyone (2.3%). Verges were mown on average 9.6 times per year (a conservative estimate obtained by taking the lower endpoints of the mowing frequency categories, question C), with 39.8% of verges being mown 16 or more times a year. Beyond mowing, maintenance was minimal, with verges taking on average 3.3 minutes to manage per week (using the lower endpoint of the maintenance time categories, question D). Costs associated with maintenance came to $22.90/year (using the lower endpoint of the cost categories, question E).

Overall, respondents reported that verges had more grass, shrubs and trees compared to when they moved in; that groundcovers and native plants had remained constant; and gravels had reduced (question set F).

Most respondents understood that property owners were responsible for maintaining their road verges (74.4%, question H1), but some did not know (13.4%) or believed that they were not responsible (11.2%). Most respondents correctly answered that their municipal authority had guidelines about maintaining the road verge (82.7%, question H2), with 15.8% not knowing. Very few respondents (7.4%) thought their verge unlikely to meet municipal government guidelines (question I1). Many (41.5%) thought they would get in trouble from their municipal authority if they verge gardened (question I2). In many cases respondents had difficulty knowing which land they should be maintaining, with 17.0% responding that is was unclear which land was their responsibility to manage (question J1). A surprising 56.6% of respondents saw the verge as part of their yard (question J2). Approximately 20% of properties in the study area have no adjacent footpath – the property line is immediately adjacent to the verge – and respondents with properties with no footpath were more likely (72.6%) to think that of the verge as part of their yard than those with footpaths (54.1%) ($p = 0.001$, chi-squared test for association).

Comparing verge gardeners to verge non-gardeners

We investigated two types of verge gardening: resident-planting of verge understorey and resident-planting of street trees.

Compared to respondents with no verge understorey, respondents who had planted verge understorey: were more enthusiastic gardeners; more often looked after their verges themselves; were more likely to never mow their verge; generally mowed less often; spent more non-mowing time looking after their verge; were more likely to spend $1–$150, and less likely to spend $0 or $150+, on maintenance; were more likely to think their verge failed to meet municipal authority guidelines; were more likely to think that they would not get in trouble from their municipal authority for verge gardening; and were more likely to see the verge as part of their yard (Table 2).

Compared to respondents with no verge gardening, respondents with resident-planted street trees: were more enthusiastic gardeners; more often looked after their verges themselves; spent more non-mowing time looking after their verge; were more likely to think their verge failed to meet municipal authority guidelines; were more likely to see the verge as part of their yard; were more likely to have 4+ cars and less likely to have 0 or 1 cars; were more likely to speak a language other than English at home; and were more likely to have year 10 or below education and were less likely to have a Bachelor degree (Table 2).

Scores for sense of community were significantly higher for respondents with resident-planted verge understorey compared to those with no verge gardening ($p = 0.016$), but no significant difference was detected between respondents with resident-planted street trees and those with no verge gardening ($p = 0.143$). Scores for nature relatedness were not significantly different between those with resident-planted verge understorey and those with no verge gardening ($p = 0.074$), or between those with resident-planted street trees and those with no verge gardening ($p = 0.560$).

Beliefs

Most respondents had positive beliefs regarding verge gardening (question sets K and L, Appendix A), believing it made the street better (67.3%, question K1), helped nature (70.5%, question K2), made their house and yard look better (56.2%, question K7) and provided opportunities to socialise (55.8%, question K8). Almost all valued tidiness (89.8%, question L3) and few saw road verges as messy (6.6%, question K3). The potential for verge gardening to hinder car access was noted by 40.7% (question K4). Most did not think verge gardening caused trip hazards (61.8%, question K5). Most respondents did not agree that keeping all the road verges looking the same would make the street look better (58.7%, question K6) or that to do so was desirable (57.6%, question L6).

Of respondents who agreed that others thought that they should not garden in the road verge, most (54.8%) answered that their municipal authority thought that they should not verge garden, with far fewer (17.8%) thinking neighbours disapproved of verge gardening, or that housemates disapproved (27.4%) (question set M).

Respondents reported that car parking on road verges would make it harder to garden on the road verge (69.2%, question O2) but few thought it was a barrier to verge gardening (38.2%, question P2). The presence of dog faeces was recognised by many (66.8%, question O6). Respondents reported that plants got damaged in the verge environment (61.6%, question O1), but this was also not a great barrier to verge gardening (34.5%, question P1). Traffic was seen by some as making it unpleasant to verge garden (38.2%, question O5) but was not a barrier (22.7%, question P5). Lack of gardening knowledge wasn't generally seen as a hindrance (34.4% question O3, 24.1% question P3). Gardening in the road verge was generally not seen as a public activity (28.2% question O7, 19.9% question P7).

Structural equation modelling

Factor analysis extracted eight latent variables corresponding to each of the eight question sets developed to test values and beliefs (maximum likelihood method, Promax rotation, Table A1). The extraction process was sound: a Kaiser-Meyer-Olkin (KMO) Test, which measures the sampling adequacy for each factor and for the model as a whole, gave KMO = 0.834 (good, close to 1.0) and $p < 0.001$ (good, significant). All communalities were above 0.3 (indicating questions worked well as sets of questions) except for question R1 in the nature relatedness question set which was 0.201, but which we decided to retain because it was part of an established and independently verified scale. The eight-factor model explained 54.1% of variance in question responses. It had less than 4.0% non-redundant residuals. Cronbach's alphas, which are a measure of the internal consistency of the set of responses to the identified factors, were good (> 0.70), though the construct measuring the strength of barriers to verge gardening was slightly less (0.653, Table A1). Convergent validity, which measures the extent to which constructs that should be related are actually related, was evidenced by all questions loading above 0.5, except for question R1 which loaded at 0.410. The absence of strong (i.e. above 3.0) cross-loadings (which measure the extent to which factors relate to each other) was evidence of discriminant validity, which is the degree to which factors that are not supposed to be related are actually not related. Discriminant validity was further evidenced by there being no values in the factor correlation matrix above 0.7 (Table A2), i.e. the factors are not strongly correlated and are each measuring their own construct.

In the confirmatory factor analysis, good model fit was achieved by excluding four questions in question set L, and all of question set O (Figure A1). Goodness of fit in SEM is typically assessed by using a range of indices. For our model: the average of the observed variables' loadings onto each latent variable were all above 0.7; CMIN/DF was good at 2.253 (i.e. < 3.0, close to 1.0); CFI = 0.943, which though less than 0.95 was good given the complexity of model and the chi-squared value of 856; PCLOSE = 0.966 (good, above 0.5); RMSEA = 0.046 (good, < 0.05); SRMR = 0.50 (good, significant). We then performed a successful validity check: convergent validity was evidenced by average variance extracted (AVE) values all being greater than 0.5; reliability was evidenced by composite reliability (CR) values all being greater than 0.7; and discriminant validity was evidenced by the square root of the AVE being greater than any inter-factor correlation (Table A3).

To rectify structural problems when testing for common method bias, it was necessary to remove question set K because it was preventing the common method bias test from running (minimisation was not being achieved). We also removed question P1 to achieve better model fit (Figure A2). The common method bias test was then run successfully, and a chi-squared test revealed significant differences in the chi-squared values and the degrees of freedom between the unconstrained and fully constrained models ($p < 0.001$), indicating the model factors had significant shared variance, i.e. common method bias was present. Because common method bias existed, we had to retain the common method factor, but rather than include it in the causal structural equation model we imputed new values for the six remaining cognitive constructs that incorporated the common variance.

To achieve good model fit in our causal model (the initial model fitted too well, CFI = 1.000, DF = 1), we deleted variances and co-variances, step-by-step, starting with the lowest loadings, until CFI < 1.000. This resulted in a model with good fit (CMIN = 5.186, DF = 5, CMIN/DF = 1.237, CFI = 0.997, PCLOSE = 0.852, RMSEA = 0.020, SRMR = 0.0223). Consequently, our causal model of resident-planting of verge understorey was based on five cognitive constructs relating to: 1) consequences of behavioural beliefs; 2) strength of normative beliefs; 3) motivation to comply with normative beliefs; 4) sense of community; and 5) nature relatedness (Figure 2). The causal model was not a full model that included the observed factors because we had common method bias. A second model was similarly developed to explain resident-planting of street trees (Figure 3, CMIN = 6.649, DF = 5, CMIN/DF = 1.330, CFI = 0.995, PCLOSE = 0.826, RMSEA = 0.023, SRMR = 0.0239).

The process of achieving good model fit in the causal models required the removal of three cognitive constructs from the models: the strength of barriers to verge gardening, the power of the barriers to verge gardening to influence verge gardening behaviour, and the strength of behavioural beliefs regarding verge gardening. This does not impact the validity of the final models, but means we cannot discuss the influence of these constructs on verge gardening behaviour, and further studies may be needed to investigate these influences.

Discussion

Attitudes to verge gardening in Melbourne are generally positive; residents mostly appreciate the many benefits that can flow from verge gardening and they have a reasonable understanding of their responsibilities to maintain the verge.

The greatest barrier to the uptake of verge gardening was the strength of residents' normative beliefs. Residents who did not verge garden were more likely to think that others would disapprove of them verge gardening than were residents who did verge garden. That constraint on their actions came more from their perceptions of the municipal authority than of their neighbours or housemates. In contrast, verge gardeners were less constrained in their actions. The other cognitive constructs included in our models – sense of community, consequences of behavioural beliefs, perceived barriers to verge gardening, and nature relatedness – were less influential than normative beliefs, and while they still had significant influence, in our modelling that influence operated by affecting normative beliefs, which in turn influenced verge gardening behaviour. If we wish to promote verge gardening, these results suggest that strategies to reduce the normative constraints on verge gardening are more likely to be successful than strategies that aim to increase positive beliefs about the benefits of verge gardening or to promote nature relatedness and sense of community.

One significant pathway to effecting change will be to change the role of municipal authorities from that of opposition to verge gardening to enabling of verge gardening. Our investigation showed that municipal guidelines tend to emphasise the punitive consequences of non-compliance with by-laws and thus restrict verge gardening behaviour. Their language emphasises risk minimisation (e.g. no trip hazards), and functionality (e.g. space for car doors to swing open), while failing to mention the many benefits of verge gardening such as increased sociability, provision of habitat, and improved mental well-being (Säumel, Weber and Kowarik, 2015). Our study found that residents were very aware of these benefits. In comments made in response to the final survey question 'Is there anything else you would like to say?', numerous people emphasised that gardening in the road verge can promote community, biodiversity and sustainability. This was expressed through: enjoyment at seeing others' contributions to the streetscape; the presence of people in the street; a place to chat with neighbours; the play of children and dogs; community environmental action by coordinated creation of bird and bee habitat; and linking houses and their yards to the wider streetscape. Many residents emphasised the potential of community gardens with vegetables and fruit. Community-positive responses were not restricted to gardening in the road verge: mowing a neighbour's verge as a gesture of community mindedness, and having simple well-maintained lawn road verges were also seen as positive. Moreover, the disjunction between the benefits of verge gardening and municipal authority restrictions on verge gardening generated considerable community resentment. The most common attitude to emerge was frustration or a sense of injustice with the fact residents are required to maintain road verges but feel they have no say in what they can do on the road verge: e.g. 'If local council considers the space is ours to upkeep, then they should not have a voice or opinion about its condition or use by the gardener. We are doing them a favour' and 'I would like to do more with our road verge but feel threatened by council if I was to do so.' Reframing policy to embrace residents' pro-environmental and pro-community behaviour would not only promote verge gardening and community values, but also remove a considerable source of conflict between residents and policy makers.

Municipal authorities should also investigate further strategies for demonstrating a positive attitude to verge gardening. For instance, they could make greater efforts to plant the many road verges they directly manage and implement other green street strategies. They could provide residents with incentives such as provision of plants, mulch and soil. Campaigns to 'Make your street a better place' may succeed in the same way programs like Melbourne-based Backyard Biodiversity have (Shaw, 2014). Interestingly, a large proportion (60.8%) of residents considered the verge to be part of their yard, with verge gardeners more likely (73.0%) than non-verge gardeners (56.3%) to think so, suggesting a 'treat it like your yard' communication program could also be an effective tool for change. A considerable proportion of respondents used gardening services (8.2%), which suggests that these paid gardeners might also be able to become effective advocates for verge gardening.

Another means of making verge gardening more normative is to make the well-mown lawn less normative. Lawns carry considerable symbolic weight, and have been linked to colonial history, power, class and wealth (Ignatieva *et al.*, 2015), good morals (Hogan, 2003), and community mindedness (Carrico, Fraser and Bazuin, 2013). These associations are often at odds with the environmental effects of maintaining lawn in a pristine state (Robbins and Birkenholtz, 2003; Askew and McGuirk, 2004; Gu *et al.*, 2015). To make the well-mown lawn less the single acceptable form of a road verge, alternative viewpoints need to be heard or made visible (Reckwitz, 2002). For example, municipal authorities could adopt low-mow maintenance regimes. Low-mow maintenance allows grasses time to set seed, providing additional resources and habitat and promotes biodiversity (Rupprecht *et al.*, 2015), promoting butterflies (Valtonen and Saarinen, 2005) and birdlife (Mason *et al.*, 2007), as well as reducing costs associated with the maintenance (O'Sullivan *et al.*, 2017). Even reducing mowing frequency from once per week to once every three weeks can increase the number of lawn flowers available for pollinators by 250% (Lerman *et al.*, 2018). Municipal authorities are in a position to lead by example here, and the practice is increasingly being adopted internationally , e.g. in Hyde Park, London (Greater London Authority, 2008). However, improved communication may be necessary for public acceptance (Lucey and Barton, 2011). Lawn alternatives can also be considered in this context. For instance, research in the United kingdom has shown that meadow-style plantings on municipal land can be effective replacements for lawn, requiring little maintenance, having distinct biodiversity benefits, such as improving communities of arthropods and soil microbes, while being positively perceived by the general public (Hoyle *et al.*, 2017; Southon *et al.*, 2017; Norton *et al.*, 2019).

Mowing differentiated verge gardeners and those who didn't verge garden, with verge gardeners mowing less. Verge gardeners were also more enthusiastic gardeners, and gardening enthusiasm has been linked in other studies (e.g. Goddard *et al*. 2013) to wildlife-friendly gardening practices.

We also found differences between the two groups of verge gardeners that we studied – those who planted understorey and those who planted street trees. First, residents who planted street trees were more likely to speak a language other than English at home and had a different educational profile. This is likely to be in part the influence of post-World War 2 migration, when many mostly Mediterranean migrants from poor agricultural backgrounds and with low levels of formal education came to Australia (Burnley, 1975). These migrants introduced new gardening styles, often transforming their front yards into vegetable plots and planting fruit trees (Timms, 2006; Hall, 2010). Second, and apparent in the SEM modelling, was that resident-planting of understorey was directly influenced by sense of community while resident-planting of street trees was not. It may be that understorey plantings required more regular care, with consequent greater engagement with others and a strengthening of sense of community (Booth *et al.*, 2018). Alternatively, residents who were community-minded found planting understorey in the road verge a means of expressing that community-mindedness.

Increasing verge gardening is a small public intervention by a resident, a niche behaviour that can pave the way for larger changes (Geels and Schot, 2007), creating levers that can transform society (Everard, Reed and Kenter, 2016). Verge gardening encourages others to verge garden (Marshall *et al*., in press; Hunter & Brown 2012). Thus creating communities of verge gardeners is likely to create positive feedback (Nassauer, Wang and Dayrell, 2009; Harris *et al.*, 2013; Jachimowicz *et al.*, 2018) that further stimulates change. Moreover, the practice of verge gardening makes verge gardening more normative, which our study suggests will further increase verge gardening, providing another positive feedback loop.

It is worthwhile noting the extent of the collective labour residents undertake in maintaining their road verges. In Melbourne there are approximately 1.2 million private dwellings (.idCommunity, 2019), which each spend at a conservative estimate $22.90/year on verge maintenance, making a total expenditure of $27 million, which is more than a quarter of the annual budget for the parks authority for the entire State of Victoria (Parks Victoria, 2018). The financial scale of verge maintenance gives a sense of the collective change that could be achieved through the many small acts of individual gardening that residents undertake.

Conclusion

Verge gardening may be able to become a vehicle for significant change in the urban environment. Road verges are a major component of our urban green space (Marshall *et al*., in press; Richards *et al.*, 1984), meaning changes away from lawn and street trees may have considerable impact on urban ecological systems. Verge gardening is already fairly common (Marshall *et al*., in press), suggesting that with the right strategies to enable and promote verge gardening it may be able to become widespread. It is a public expression of individuals' beliefs and values (Kiesling and Manning, 2010), which means it may be able to contribute to a greater societal change.

Residents' perception of municipal authorities being opposed to verge gardening appears to be a major barrier to increasing the prevalence of this citizen greening activity that provides biodiversity, ecosystem function and human amenity benefits. Municipal authorities can reverse this by embracing verge gardening and its benefits, revisiting their communications with residents, and leading by example through the adoption of low-mow lawn maintenance. These actions would have the additional benefit of removing a considerable source of frustration between residents and policy makers (Ives and Kendal, 2014). The many small actions of residents have the potential to sum to substantial beneficial change.

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Figures

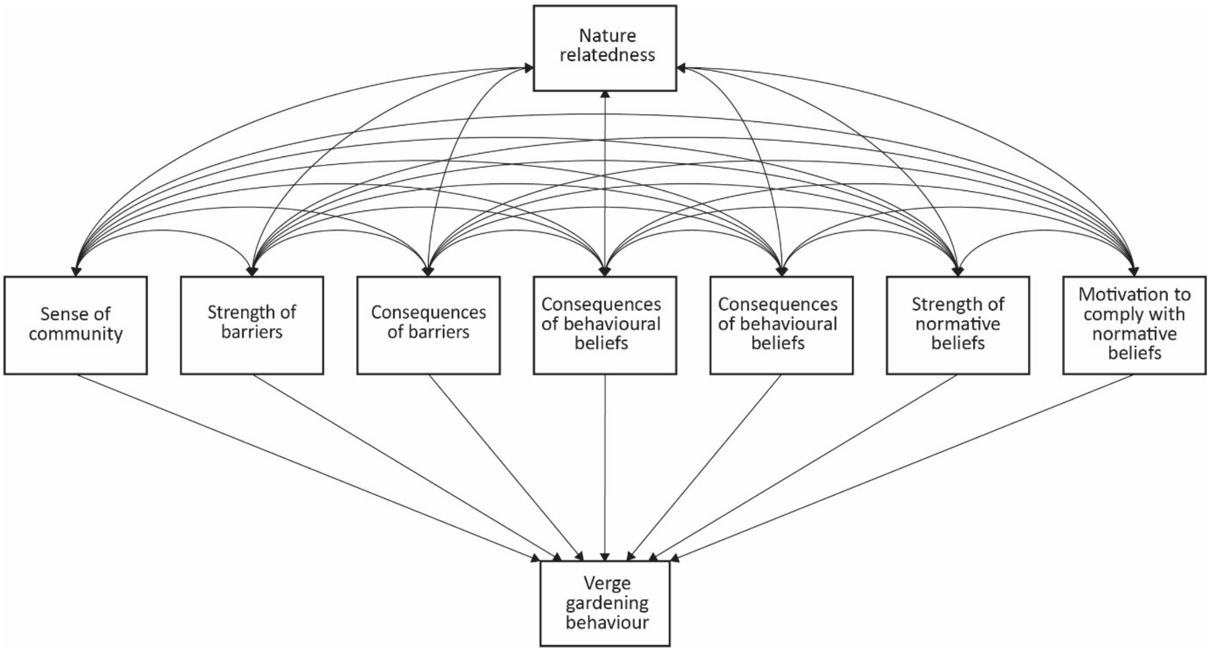


Figure 1: Causal model illustrating the proposed relationships between eight cognitive constructs and verge gardening. Vertical position reflects each construct's position in a cognitive hierarchy from value-based attitude (top, nature relatedness) to behaviour (bottom, observed behaviour, verge gardening). All predictor factors co-vary, shown by double-headed arrows. Single-headed arrows show the relationship between the predictor factors and the outcome behaviour.

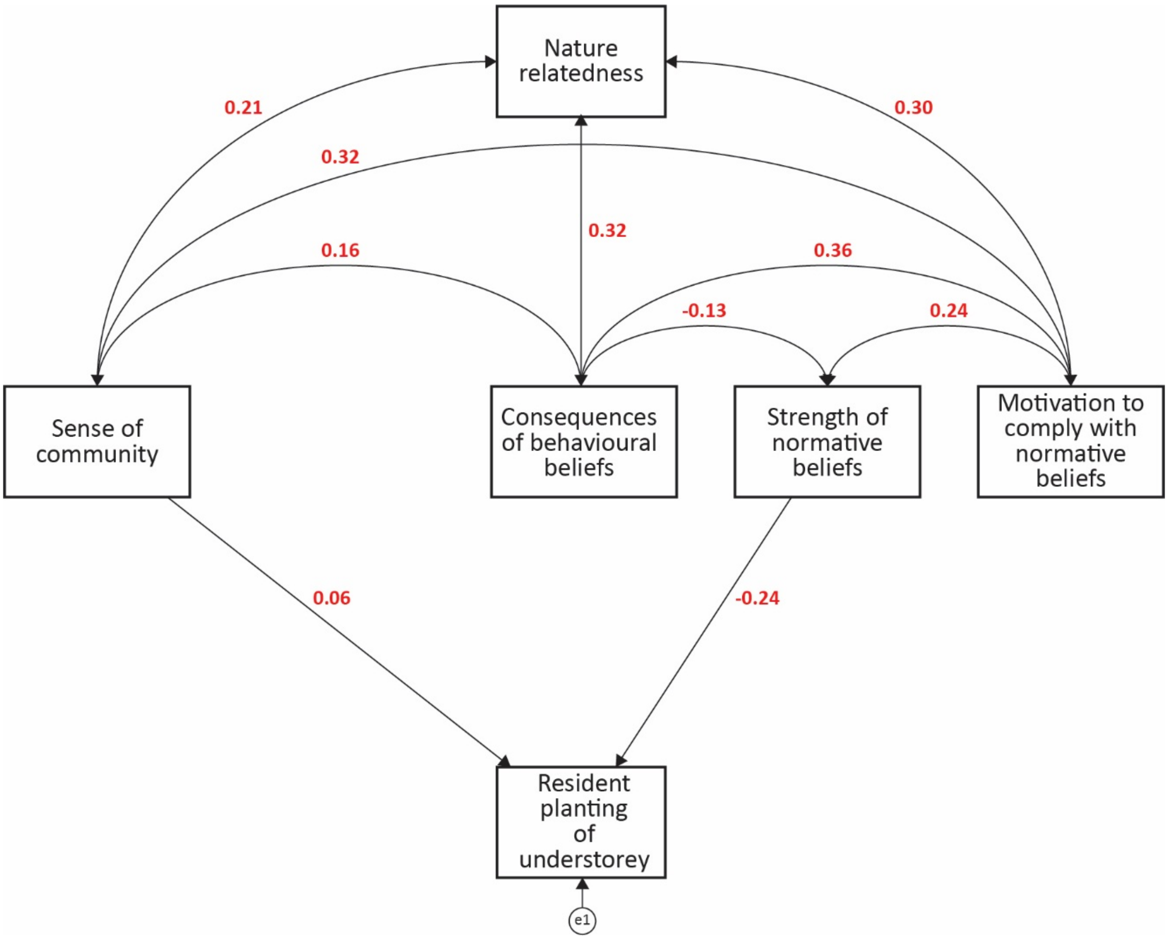


Figure 2: Causal model for resident-planting of verge understorey, with good fit, showing the relationship between five cognitive constructs and resident-planting of verge understorey for data from 606 responses to a survey of verge gardening behaviour across 47 neighbourhoods in Melbourne, Australia. Red numbers are Pearson correlation values. Vertical position reflects each construct's position in a cognitive hierarchy from value-based attitude (top, nature relatedness) to behaviour (bottom, resident-planting of verge understorey). All predictor factors co-vary, shown by double-headed arrows. Single-headed arrows show the relationship between the predictor factors and the outcome behaviour. E1 is a measure of omitted cause or error.

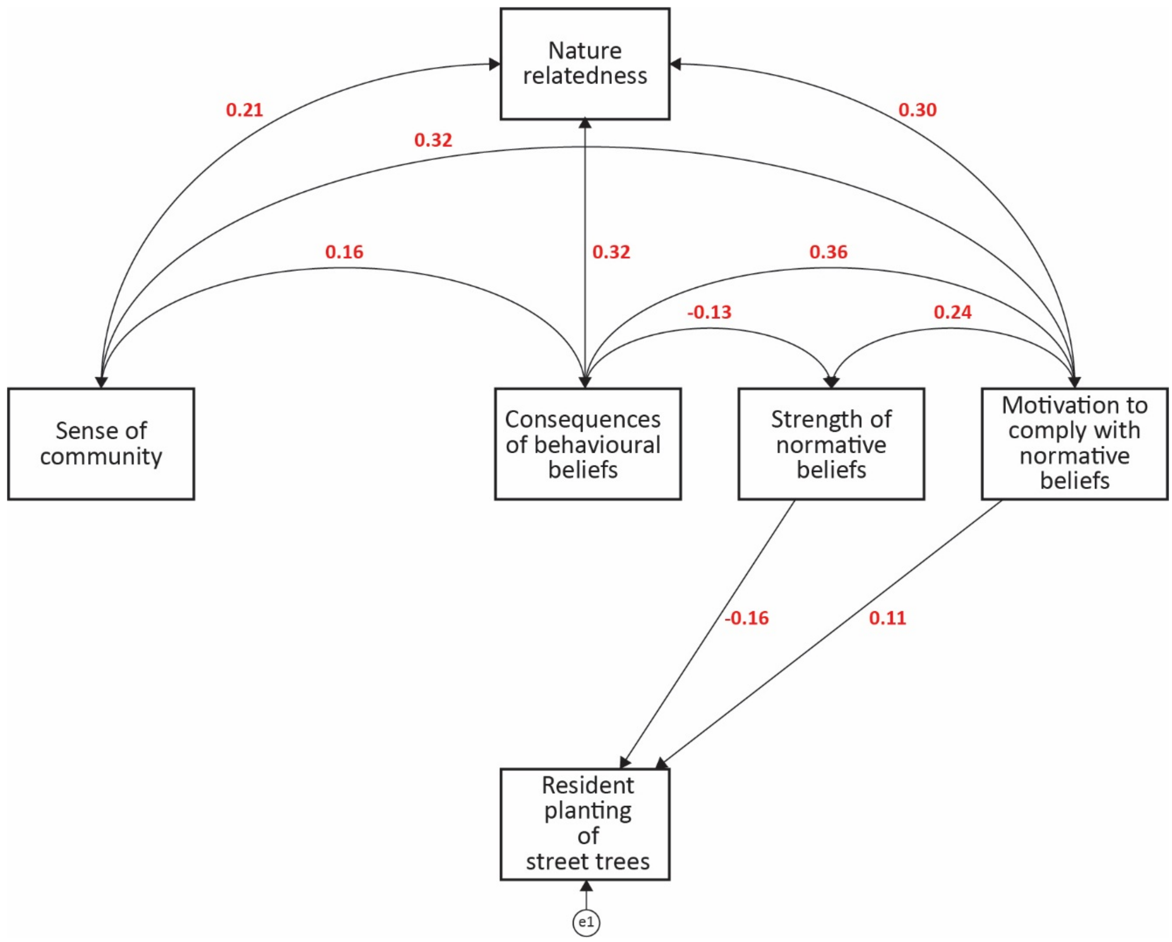


Figure 3: Causal model for resident-planting of street trees, with good fit, showing the relationship between five cognitive constructs and the observed behaviour for data from 606 responses to a survey of verge gardening behaviour across 47 neighbourhoods in Melbourne, Australia. Red numbers are Pearson correlation values. Vertical position reflects each construct's position in a cognitive hierarchy from value-based attitude (top, nature relatedness) to behaviour (bottom, resident-planting of street trees). All predictor factors co-vary, shown by double-headed arrows. Single-headed arrows show the relationship between the predictor factors and the outcome behaviour. E1 is a measure of omitted causes or error.

Tables

Table 1: Eight cognitive constructs reflecting values and beliefs associated with verge gardening used in the survey delivered to 5151 households across 47 neighbourhoods in Melbourne, Australia. Each construct was investigated through a set of 3–8 questions, identified by labels K–R.

Construct	Question Set	Example survey question
Strength of behavioural beliefs regarding verge gardening	K	'Gardening in the road verge provides opportunities to chat with neighbours [Strongly disagree … Strongly agree]'
Consequences of behavioural beliefs regarding verge gardening	L	'Having the opportunity to chat with neighbours is [Very undesirable … Very desirable]'
Strength of normative beliefs about verge gardening	M	'My neighbours think I should not garden in the road verge [Strongly disagree … Strongly agree]'
Motivation to comply with normative beliefs about verge gardening	N	'It matters to me that the people I share my house with approve of how I look after my road verge [Strongly disagree … Strongly agree]'
The strength of barriers to verge gardening	O	'You have to know about gardening to look after plants on the road verge [Very unlikely … Very likely]'
The power of the barriers to verge gardening to influence verge gardening behaviour	P	'I would be more likely to garden in the road verge if there was less traffic in my street [Very unlikely … Very likely]'
Sense of community	Q	'I feel hopeful about the future of this community [Strongly disagree … Strongly agree]'
Nature relatedness	R	'I always think about how my action affect the environment [Strongly disagree … Strongly agree]'

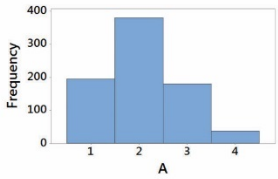
Table 2: Significant differences between both respondents with resident-planted verge understorey and resident-planted street trees and respondents with no verge gardening from 606 responses to a survey of verge gardening behaviour across 47 neighbourhoods in Melbourne, Australia.

Question	P-value	Respondents with resident-planted verge understorey compared to respondents with no verge gardening	P-value	Respondents with resident-planted street trees compared to respondents with no verge gardening
A	0.035	More enthusiastic gardeners	0.003	More enthusiastic gardeners
B	0.028	More likely to manage verge themselves	0.047	More likely to manage verge themselves
C	< 0.001	Much more likely to never mow their verge, and generally likely to mow less often		
D	< 0.001	More likely to spend more time maintaining verge	< 0.001	More likely to spend more time maintaining verge
E	0.038	More likely to spend $1–$150, and less likely to spend $0 or $150+	0.008	More likely to spend $1–$100, and less likely to spend $0
F1	<0.001	More likely to have less grass		
F2	<0.001	More likely to have more shrubs	<0.001	More likely to have more shrubs
F3	<0.001	More likely to have more ground covers	0.011	More likely to have more ground covers
F4			<0.001	More likely to have more trees
F5	<0.001	More likely to have more native plants	<0.001	More likely to have more native plants
F6	<0.001	More likely to have more gravels		
I1	< 0.001	More likely to think their verge fails to meet municipal authority guidelines	0.017	More likely to think their verge fails to meet municipal authority guidelines
I2	0.026	More likely to think that they will not get in trouble from their municipal authority for verge gardening		
I3	< 0.001	More likely to think that they will garden in the verge in the next year	< 0.001	More likely to think that they will garden in the verge in the next year
J2	< 0.001	More likely to see the verge as part of their garden	0.005	More likely to see the verge as part of their garden
S3			0.030	More likely to have 4+ cars and less likely to have 0 or 1 cars
S4			0.004	More likely to speak a language other than English at home
U			0.050	More likely to have year 10 or below education and are less likely to have a Bachelor degree

Appendix 5A: survey questions and results

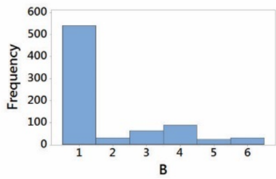
A Which one of the following best describes you?

- ○ Gardening enthusiast
- ○ Make-an-effort gardener (I make some effort to make the garden attractive)
- ○ Gardener out of necessity (I only do what is absolutely necessary to keep the garden tidy)
- ○ Non-gardener (I do not do any gardening)



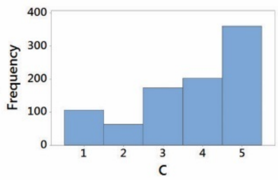
B Who looks after your road verge?
(Tick as many boxes as you like)

- ○ You ○ A neighbour ○ A gardening service
- ○ Another person in your house ○ No one
- ○ Other



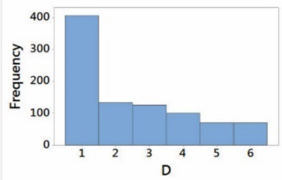
C How many times a year does your road verge get mown?

- ○ Never ○ 1–5 ○ 6–10
- ○ 11–15 ○ Over 15



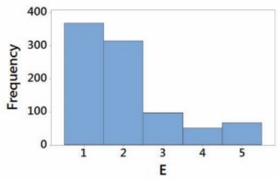
D Apart from mowing, how many minutes a week do you spend looking after your road verge?

- ○ None ○ 1–2 ○ 3–5
- ○ 6–10 ○ 11–15 ○ Over 16



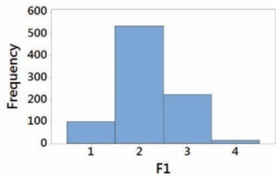
E How much money do you spend a year to maintain your road verge?

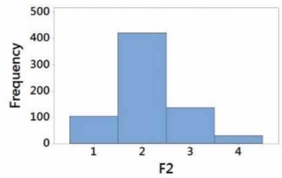
- ○ None ○ $50 or less ○ $51–$100
- ○ $101–$150 ○ Over $150

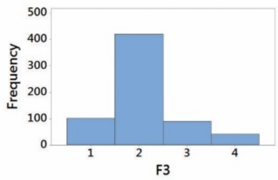


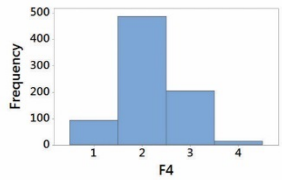
F Compared to when you moved in, does your road verge now have less of more of the following?

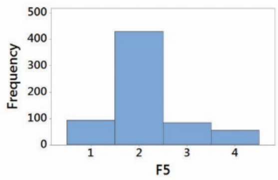
		LESS	ABOUT THE SAME	MORE	DON'T KNOW
F1	Mown grass	○	○	○	○
F2	Shrubs or bushes	○	○	○	○
F3	Ground covers	○	○	○	○
F4	Trees	○	○	○	○
F5	Native plants	○	○	○	○
F6	Gravel, artificial lawn or toppings	○	○	○	○

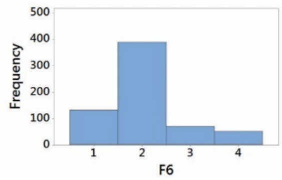






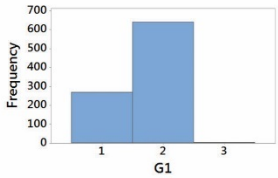






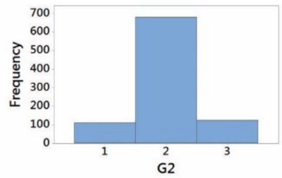
G1 Was your residence new when you moved in?

- ○ Yes ○ No ○ Don't know



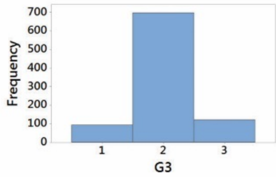
G2 Was a professional landscaper or designer used to design the front garden?

- ○ Yes ○ No ○ Don't know



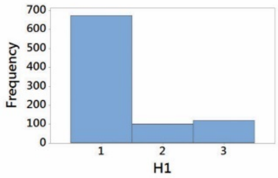
G3 Was a professional landscaper or designer used to plant your front garden?

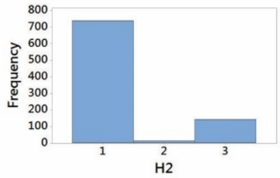
- ○ Yes ○ No ○ Don't know



H Please indicate if you think the following statements are true:

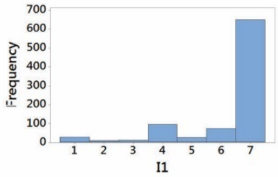
	TRUE	NOT TRUE	DON'T KNOW
H1 Property owners are by law responsible for maintaining their nature strip (apart from street trees)	○	○	○
H2 Council has guidelines about what I can and cannot do to my nature strip	○	○	○

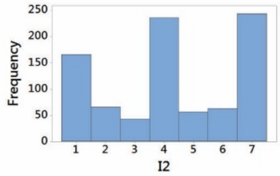


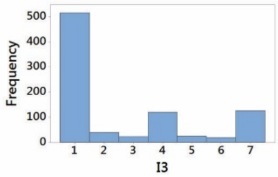


I Please indicate how likely you think the following are:

	VERY UNLIKELY			NEUTRAL			VERY LIKELY
I1 My nature strip meets council guidelines	○	○	○	○	○	○	○
I2 I will get in trouble from council if I garden in my nature strip	○	○	○	○	○	○	○
I3 In the next year, I will do some gardening in the nature strip	○	○	○	○	○	○	○

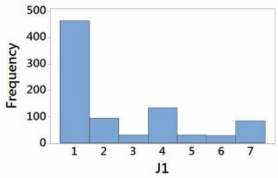


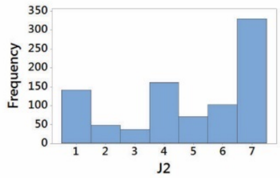




J Please indicate how much you agree with the following:

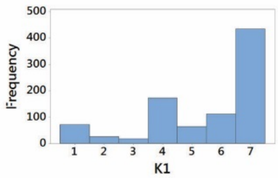
	STRONGLY DISAGREE			NEUTRAL			STRONGLY AGREE
J1 It is unclear which land is my responsibility to look after and which is my neighbours' or council's land	○	○	○	○	○	○	○
J2 I see the nature strip as part of my garden	○	○	○	○	○	○	○

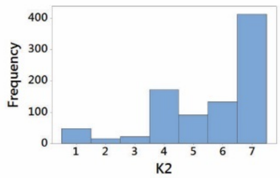


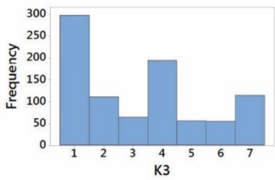


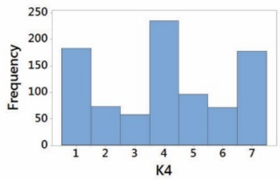
K Please indicate how much you agree with the following:

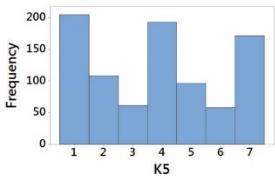
	STRONGLY DISAGREE			NEUTRAL			STRONGLY AGREE
K1 Gardening in the nature strip makes the street a better place	○	○	○	○	○	○	○
K2 Gardening in the nature strip helps nature and the environment	○	○	○	○	○	○	○
K3 Gardening in the nature strip makes the nature strip messy	○	○	○	○	○	○	○
K4 Plants in the nature strip make it difficult to get to and from the car	○	○	○	○	○	○	○
K5 People might trip or hurt themselves if there are plants in the nature strip	○	○	○	○	○	○	○
K6 The street would look best if all the nature strips looked the same	○	○	○	○	○	○	○
K7 Gardening in the nature strip makes my house and garden look better	○	○	○	○	○	○	○
K8 Gardening in the nature strip provides opportunities to chat with neighbours	○	○	○	○	○	○	○

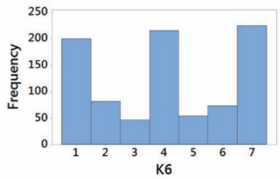


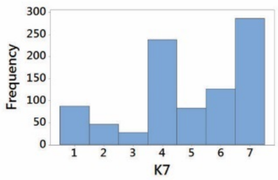


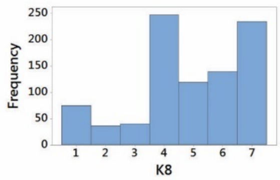






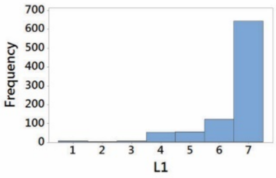


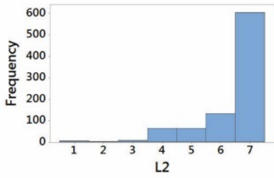


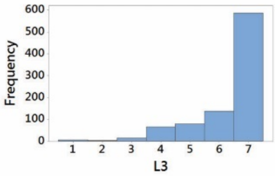


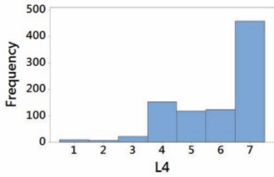
L Please indicate how desirable the following are:

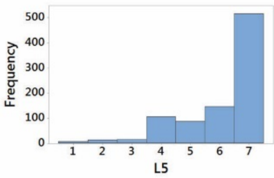
	VERY UNDESIRABLE			NEUTRAL			VERY DESIRABLE
L1 Making the street a better place is…	○	○	○	○	○	○	○
L2 Helping nature and the environment is…	○	○	○	○	○	○	○
L3 Making the nature strip look tidy is…	○	○	○	○	○	○	○
L4 Getting to the car as easily as possible is…	○	○	○	○	○	○	○
L5 Ensuring the nature strip is a safe place to walk is…	○	○	○	○	○	○	○
L6 Keeping the street's nature strips all looking the same is…	○	○	○	○	○	○	○
L7 Making my house and garden look better is…	○	○	○	○	○	○	○
L8 Having the opportunity to chat with neighbours is…	○	○	○	○	○	○	○

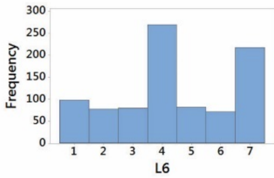


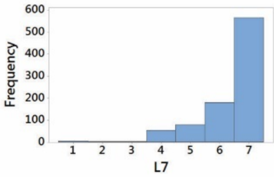


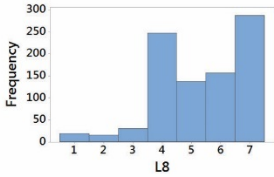






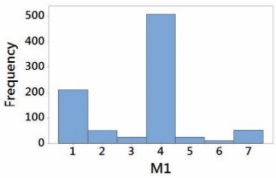


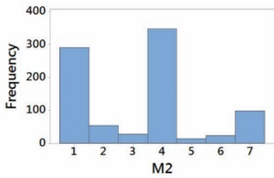


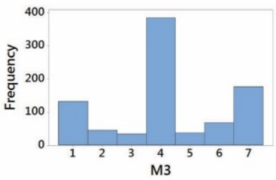


M Please indicate how much you agree with the following:

	STRONGLY DISAGREE			NEUTRAL			STRONGLY AGREE
M1 My neighbours think I should not garden in the nature strip	○	○	○	○	○	○	○
M2 The people I share my house with think I should not garden in the nature strip	○	○	○	○	○	○	○
M3 Council thinks I should not garden in the nature strip	○	○	○	○	○	○	○

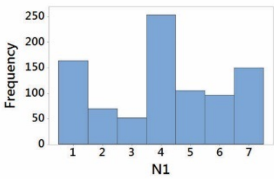


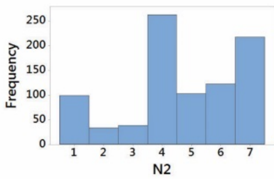


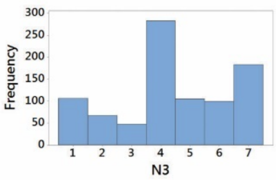


N Please indicate how much you agree with the following:

	STRONGLY DISAGREE			NEUTRAL			STRONGLY AGREE
N1 It matters to me that my neighbours approve of how I look after my nature strip	○	○	○	○	○	○	○
N2 It matters to me that people I share my house with approve of how I look after my nature strip	○	○	○	○	○	○	○
N3 It matters to me that council approves of how I look after my nature strip	○	○	○	○	○	○	○

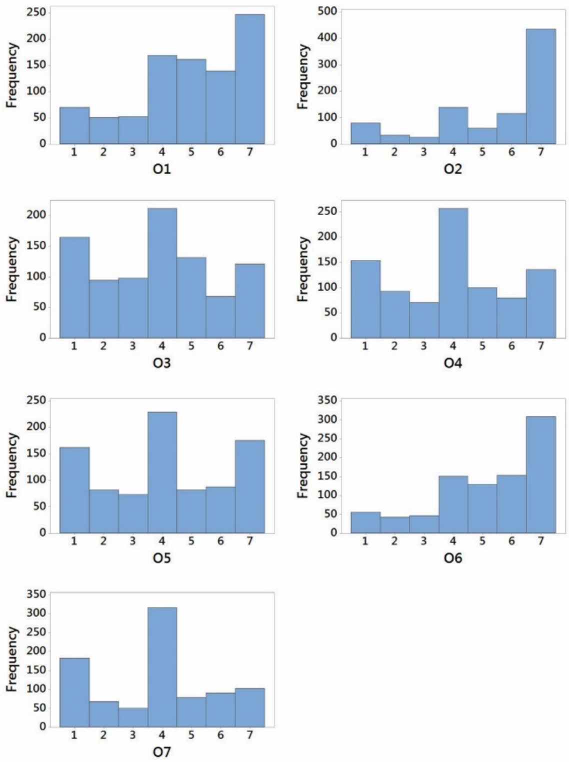






0 Please indicate how likely you think the following are:

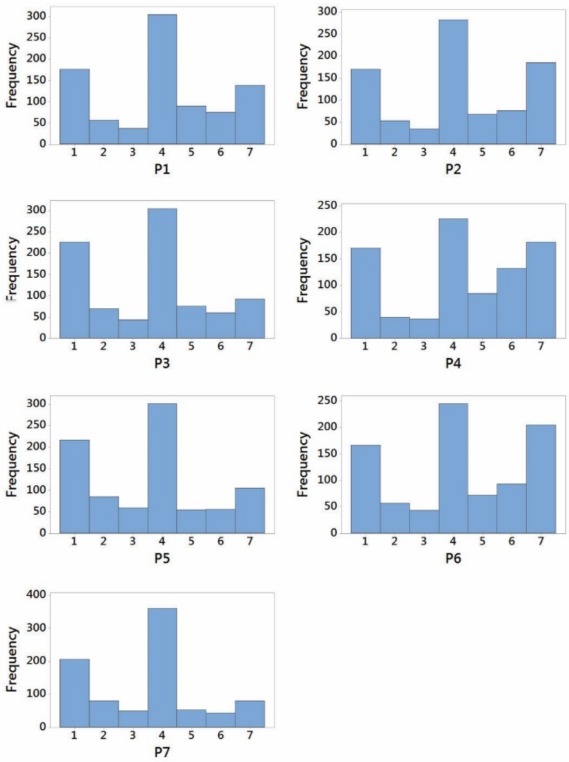
		VERY UNLIKELY			NEUTRAL			VERY LIKELY
01	Plants get damaged in the nature strip	○	○	○	○	○	○	○
02	Car parking on the nature strip makes it hard to garden on the nature strip	○	○	○	○	○	○	○
03	You have to know about gardening to look after plants on the nature strip	○	○	○	○	○	○	○
04	It takes too much time to garden in the nature strip	○	○	○	○	○	○	○
05	Too much traffic makes it unpleasant to garden in the nature strip	○	○	○	○	○	○	○
06	Nature strips have dog poo on them	○	○	○	○	○	○	○
07	Gardening in the nature strip is a public activity	○	○	○	○	○	○	○



P Please indicate how likely you think the following are:

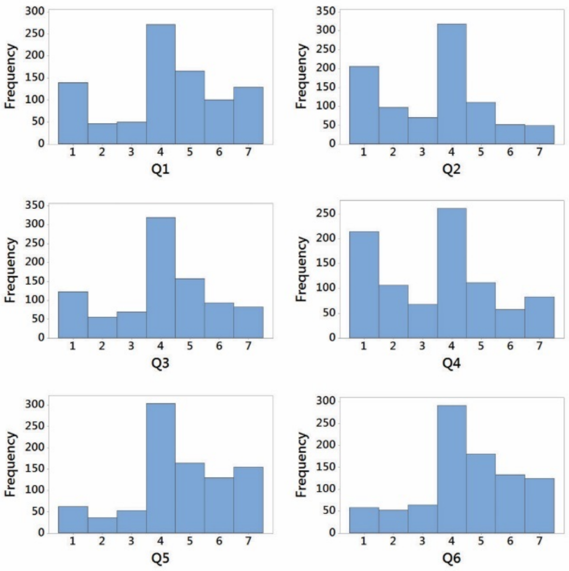
I would be more likely to garden in the nature strip if:

		VERY UNLIKELY			NEUTRAL			VERY LIKELY
P1	People damaged plants in the nature strip less	○	○	○	○	○	○	○
P2	People parked less on the nature strip	○	○	○	○	○	○	○
P3	I knew more about gardening	○	○	○	○	○	○	○
P4	I had more time	○	○	○	○	○	○	○
P5	There was less traffic in my street	○	○	○	○	○	○	○
P6	Dogs didn't go the toilet there	○	○	○	○	○	○	○
P7	It was a less public activity	○	○	○	○	○	○	○



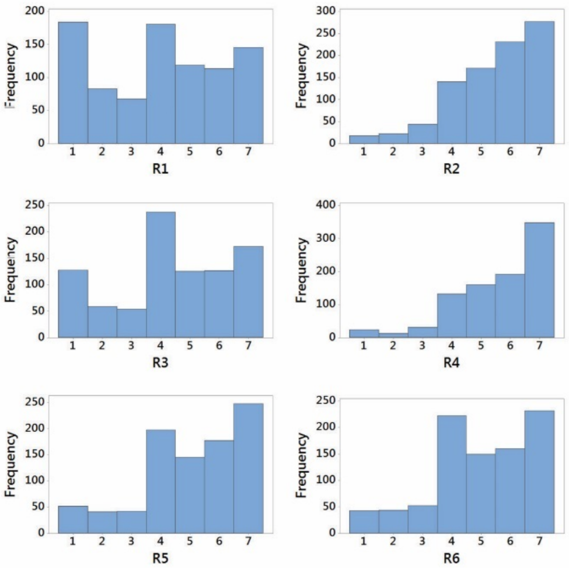
Q Please indicate how much you agree with the following:

	STRONGLY DISAGREE			NEUTRAL			STRONGLY AGREE
Q1 I care about what other community members think of me	○	○	○	○	○	○	○
Q2 I have influence over what this community is like	○	○	○	○	○	○	○
Q3 If there is a problem in this community, members can get it solved	○	○	○	○	○	○	○
Q4 Members of this community have shared important events together, such as holidays, celebrations, or disasters	○	○	○	○	○	○	○
Q5 I feel hopeful about the future of this community	○	○	○	○	○	○	○
Q6 Members of this community care about each other	○	○	○	○	○	○	○



R Please indicate how much you agree with the following:

	STRONGLY DISAGREE			NEUTRAL			STRONGLY AGREE
R1 My ideal vacation spot would be a remote, wilderness area	○	○	○	○	○	○	○
R2 I always think about how my actions affect the environment	○	○	○	○	○	○	○
R3 My connection to nature and the environment is a part of my spirituality	○	○	○	○	○	○	○
R4 I take notice of wildlife wherever I am	○	○	○	○	○	○	○
R5 My relationship to nature is an important part of who I am	○	○	○	○	○	○	○
R6 I feel very connected to all living things and the earth	○	○	○	○	○	○	○



S1 What is your age?

- ○ 20 or less
- ○ 21–30
- ○ 31–40
- ○ 41–50
- ○ 51–60
- ○ Over 60

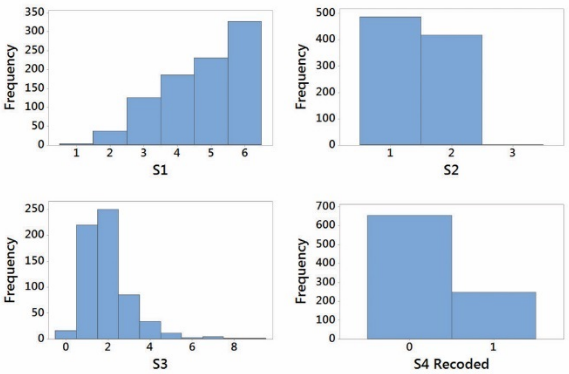
S2 What is your gender?

- ○ Female
- ○ Male
- ○ Other

S3 How many cars are owned in the household?

S4 Do you speak a language other than English at home?

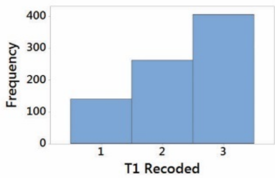
- ○ No, English only
- ○ Yes, Arabic
- ○ Yes, Cantonese
- ○ Yes, Croatian
- ○ Yes, Filipino/Tagalog
- ○ Yes, Greek
- ○ Yes, Hindi
- ○ Yes, Macedonian
- ○ Yes, Maltese
- ○ Yes, Mandarin
- ○ Yes, Punjabi
- ○ Yes, Spanish
- ○ Yes, Turkish
- ○ Yes, Vietnamese
- ○ Yes, other (please specify)



S5 If you were born overseas, please indicate how many years you have lived in Australia?

Results for S5 not shown.

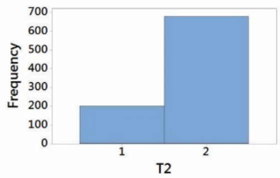
T1 **How many years have you lived in this street?**



T2 **Are you a tenant/renter?**

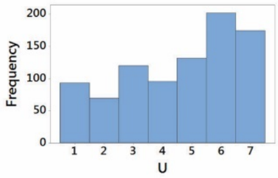
- ○ Yes ○ No

Results of T1 recoded to three categories: <4, 4–16, >16 years.



U **What is the highest level of education you have completed?**

- ○ Year 10 or below
- ○ Year 11
- ○ Year 12
- ○ Trade certificate, e.g. VET/TAFE
- ○ Diploma/Advanced Diploma
- ○ Bachelor or Graduate Certificate/ Diploma
- ○ Postgraduate



V **Is there anything else you would like to say?**

Results of V not shown.

Appendix 5B: Factor analysis and confirmatory factor analysis

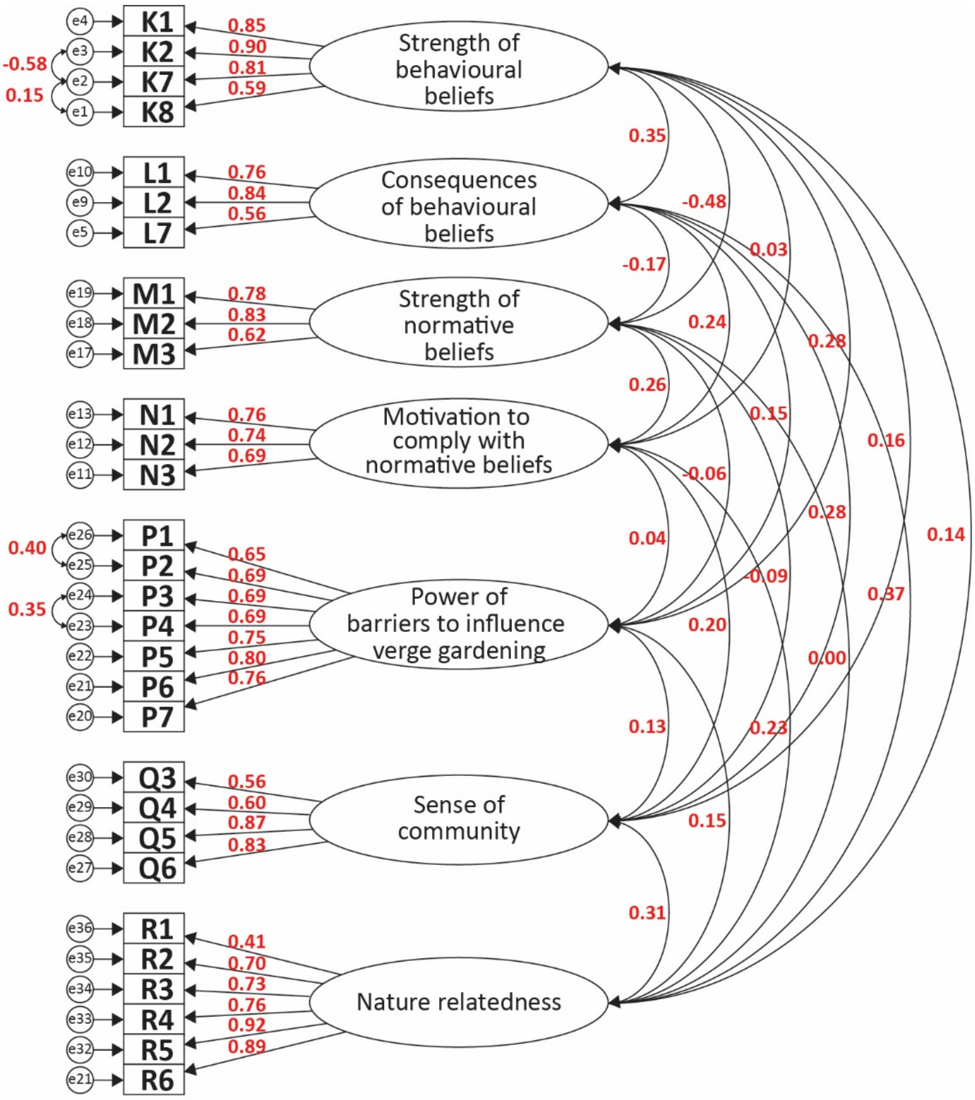


Figure A1: Confirmatory Factor Analysis model with good model fit for seven cognitive constructs (latent variables) extracted from 30 survey questions (observed variables) using data from 606 responses to a survey of verge gardening behaviour in Melbourne, Australia. Red numbers are Pearson correlation values. K1–R6 each represent one of the 30 survey questions included in the confirmatory factor analysis. Omitted causes or errors are represented by e1–e30. Single-headed arrows show cognitive constructs predict question responses, and double-headed arrows show co=varying relationships between cognitive constructs.

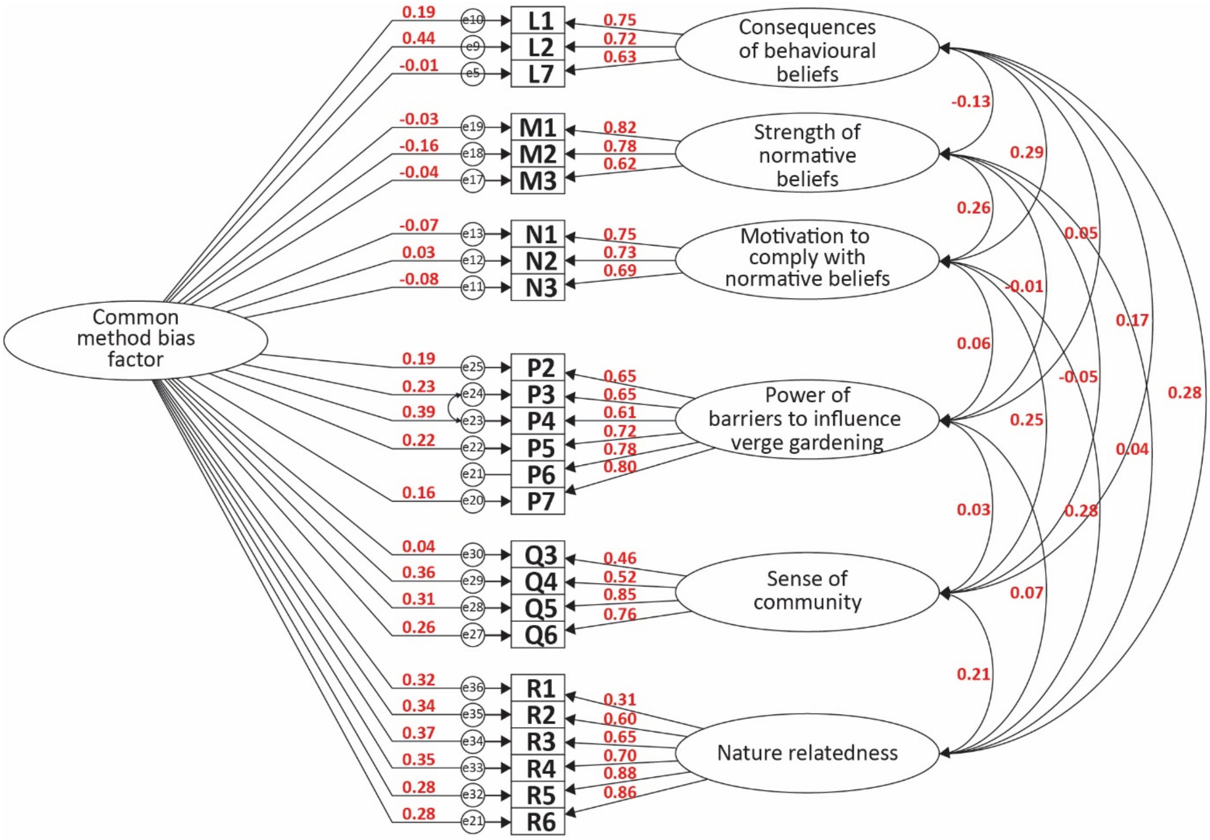


Figure A2: Confirmatory factor analysis model following corrections for common method bias, showing six cognitive constructs (latent variables) extracted from 25 observed variables using data from 606 responses to a survey of verge gardening behaviour in Melbourne, Australia. Red numbers are Pearson correlation values. L1–R6 each represent one of the 25 survey questions kept in this confirmatory factor analysis. Omitted causes or errors are represented by e1–e30. Single-headed arrows show that both cognitive constructs and the common method bias factor predict question responses, and double-headed arrows show relationships co-varying between cognitive constructs.

Table A1: Pattern matrix for eight latent variables (cognitive constructs) extracted from 606 respondents' answers to 48 survey questions (observed factors) relating to verge gardening in Melbourne, Australia. No loadings above 0.3 are shown. Values for Chronbach's alpha shown in brackets. Latent factors correspond to: 1) strength of behavioural beliefs; 2) consequences of behavioural beliefs; 3) strength of normative beliefs; 4) motivation to comply with normative beliefs; 5) the strength of barriers to verge gardening; 6) the power of barriers to verge gardening to influence behaviour; 7) sense of community; and 8) nature relatedness.

	Latent factor							
Observed factor	1 (0.856)	2 (0.789)	3 (0.779)	4 (0.770)	5 (0.653)	6 (0.889)	7 (0.803)	8 (0.864)
K1			0.929					
K2			0.943					
K7			0.639					
K8			0.558					
L1				0.639				
L2				0.561				
L3				0.759				
L4				0.628				
L5				0.588				
L7				0.622				
M1						0.814		
M2						0.719		
M3						0.699		
N1							0.743	
N2							0.746	
N3							0.669	
O3								0.645
O4								0.693
O5								0.505
P1	0.699							
P2	0.782							
P3	0.719							
P4	0.663							
P5	0.728							
P6	0.768							
P7	0.751							
Q3					0.539			
Q4					0.623			
Q5					0.857			
Q6					0.865			
R1		0.410						
R2		0.650						
R3		0.731						
R4		0.749						
R5		0.961						
R6		0.914						

Table A2: Factor correlation matrix for eight latent variables extracted from 606 respondents' answers to 48 survey questions relating to verge gardening in Melbourne, Australia.

Factor	1	2	3	4	5	6	7
2	0.166						
3	0.304	0.254					
4	0.079	0.233	0.121				
5	0.163	0.357	0.260	0.138			
6	-0.063	0.002	-0.469	-0.019	-0.091		
7	0.032	0.230	0.040	0.273	0.180	0.202	
8	0.080	-0.121	-0.457	-0.057	-0.088	0.500	0.100

Table A3: Confirmatory factor analysis checks for model validity for model relating six latent variables (cognitive constructs) and showing: composite reliability (CR), average variance extracted (AVE), maximum shared variance (MSV), maximum reliability (MaxR(H)), and including the square root of the AVE on the diagonal in bold above correlations in the correlation matrix between the six latent variables.

	CR	AVE	MSV	MaxR(H)	Sense of Community	Consequences of behavioural beliefs	Motivation to comply with normative beliefs	Strength of normative beliefs	Power of barriers to influence verge gardening	Nature Relatedness
Sense of Community	0.81	0.53	0.10	0.86	**0.73**					
Consequences of behavioural beliefs	0.78	0.63	0.14	0.78	0.28	**0.80**				
Motivation to comply with normative beliefs	0.77	0.53	0.06	0.77	0.20	0.40	**0.73**			
Strength of normative beliefs	0.79	0.56	0.06	0.81	-0.09	-0.16	0.24	**0.75**		
Power of barriers to influence verge gardening	0.88	0.52	0.02	0.89	0.13	0.14	0.04	-0.05	**0.72**	
Nature Relatedness	0.88	0.56	0.14	0.93	0.31	0.37	0.23	0.00	0.15	**0.75**

Chapter 6
Urban road verge vegetation is driven by verge gardening, mowing, environmental, climactic and stochastic factors

Abstract

Road verges are a major type of urban greenspace and their considerable extent, ubiquity and often broad -wide distribution within cities means that understanding the drivers of road verge species diversity, cover and composition may inform urban biodiversity and greening strategies. We surveyed the flora of road verges across 39 neighbourhoods in Melbourne, a sprawling temperate city. We identified 150 species, of which 82.7% were exotic, with native species mostly present because of residents' verge gardening behaviour. ANOVA and principle coordinate analysis (PCO) ordination revealed that species richness, abundance and composition were driven by residents' verge gardening behaviour, mowing frequency, rainfall, soil compaction and canopy openness, though the factors we investigated explained only 20.2% of variation in species richness, 34.9% in cover and 17.4% in community composition. Levels of disturbance are high in road verges, and stochastic processes may explain much of the observed variation. There is significant potential to increase native species richness, structural complexity and overall ecosystem function by increasing the incidence of verge gardening, and decreasing mowing frequency.

Introduction

Road verges are a form of public greenspace that typically consists of lawn, gardened areas and street trees (Marshall, Grose and Williams, 2019a). Because of their considerable extent in many cities in the developed world, where they can comprise more than a third of all public greenspace (Richards *et al.*, 1984; Marshall, Grose and Williams, 2019b), and their ubiquity and often broad distribution within cities, road verges are likely to play a substantial role in urban ecosystem processes. They are also likely to be the place where urban residents most interact with nature. Road verge lawn (or mown amenity grass) and understorey contribute to urban cooling (Robinson and Lundholm, 2012; Wang *et al.*, 2016), mitigate stormwater runoff (Mueller and Thompson, 2009; Carmen, Hunt and Anderson, 2014; Beck, McHale and Hess, 2016), provide connectivity between larger green patches (Söderström and Hedblom, 2007), remove airborne pollutants (Leonard, McArthur and Hochuli, 2016), and contribute to human sense of wellbeing and health (Säumel, Weber and Kowarik, 2015). Sometimes urban road verges support critically endangered plant species (Stuwe, 1986; Kirkpatrick, McDougall and Hyde, 1995), and road verge lawn and understorey provides habitat and resources for many species including bees (Larson, Kesheimer and Potter, 2014), butterflies (Schaffers, Raemakers and Sýkora, 2012), arthropods (Leonard, McArthur and Hochuli, 2018), nematodes (Cheng *et al.*, 2008), ants (Pećarević, Danoff-Burg and Dunn, 2010) and soil microbes (Beard and Green, 1994). Despite such evidence, urban road verge understorey and lawn species composition and abundance, and the factors that may influence that composition and abundance, remain largely uncharacterised.

Several studies have examined the species composition of urban lawns. For instance, Wheeler *et al*. (2017) found 353 species across domestic lawns of seven cities in the USA. Turfgrass species contribute comparatively little to lawn species richness, with the majority of species generally being spontaneous vegetation (Thompson *et al.*, 2004; Stewart *et al.*, 2009; Bertoncini *et al.*, 2012; Wheeler *et al.*, 2017). Species-area curves are more similar to natural grasslands than to cultivated yards (i.e. gardens, Thompson *et al.*, 2004), although lawns tend to be more species poor than natural grasslands and the phylogenetic diversity of yard floras is significantly less than that of natural systems, i.e. they come from a narrow range of families (Knapp *et al.*, 2012).

The floristic composition of lawn has been shown to be associated with climatic and environmental factors. Wheeler *et al*. (2017) found mean annual rainfall had a small positive correlation with species richness over a continental-scale gradient, while Pearse *et al*. (2016) found that as regional-scale aridity increased, species richness of both cultivated and spontaneous patches increased, possibly due to irrigation. Müller (1990) associated varying lawn communities across Germany with a rainfall and altitude gradients, noted that lawns are generally located on urban soils that have been altered through the deposition of humus, and concluded water balance in the soil determines lawn community composition rather than other soil qualities. Thompson *et al*. (2004) found altitude drove species composition differences in Sheffield. Soil factors (sand/loam) were found by Stewart *et al*. (2009) to be important in determining lawn communities in Christchurch, New Zealand.

The species composition and abundance of the lawn of road verges is likely to be different to that of domestic yards not only because of varying maintenance practices but because the road verge generally has harsher conditions (Gilbert, 1989; van der Ree, Grilo and Smith, 2015): it is likely to be subject to greater trampling, lack of irrigation, higher pollution levels, higher compaction, increased nutrients from animal waste, occasional profound disturbance, and more irregular maintenance regimes. Moreover, the road verge differs from lawn in public parks because patches are smaller, managed by residents rather than public authorities or their contractors, trees are generally present, and use is different. Our observation suggests the lawn of road verges is less likely to be laid as turf than the lawn of domestic yards. It is likely that the maintenance of road verge lawns is far less input-intensive than residential lawn maintenance.

Only one study has studied the floristic composition of road verge lawns (Stewart *et al.*, 2009), as part of a broader study of private, street and park lawns in Christchurch, New Zealand. The authors found lawns in general to be species rich, with 127 species. Of the seven plant communities identified, road verge lawn was most associated with 'coarse textured public lawns' and 'patchy public lawns', which the authors suggest may indicate less maintenance, shadier conditions and considerable trampling – compared to 'fine textured' private lawn. The authors also found the great majority of species to be exotic, unlike other studies which were conducted the United States and Europe.

Both domestic and public lawns are generally managed for social rather than ecological reasons, for instance to keep the grassed area tidy and to meet social norms (Davison, Dingle and O'Hanlon, 1995; Nassauer, Wang and Dayrell, 2009). Mowing is the most common form of lawn maintenance, with weeding by either hand or herbicide, fertiliser application and irrigation also undertaken (Robbins and Birkenholtz, 2003). A lush green lawn can be a source of pride (Robbins, 2012). Maintenance practices such as fertiliser application have been associated with social and demographic factors such as household income, education, gender and prestige: see Martini *et al*. (2013) for an overview. Frequent mowing promotes increased cover (Rudolph *et al.*, 2017) and filters species intolerant of such disturbance, and infrequent mowing promotes functional and phylogenetic diversity (Chollet *et al.*, 2018). Mowing regimes that allow lawn species to flower and to set seed, such as low-mow management regimes, have been successful in providing resources for, and increasing the presence of, bees (Lerman *et al.*, 2018), butterflies (Valtonen and Saarinen, 2005), bugs (Helden and Leather, 2004) and birdlife (Mason *et al.*, 2007). Moreover, there is increasing awareness of these benefits, with the general public now increasingly preferring biodiversity to simple green urban space (Southon *et al.*, 2017; Fischer *et al.*, 2018; Norton *et al.*, 2019).

Gardening (as opposed to lawn maintenance) also occurs in road verges, with two studies identifying verge gardening being associated with 11% of properties in Ann Arbor, Michigan (Hunter and Brown, 2012) and 22% in Melbourne, Australia (Marshall, Grose and Williams, 2019a). The species gardeners introduce into the road verge have not been characterised.

This study examined the lawn and understorey flora of road verges in the sprawling, suburban city of Melbourne, Australia, in order to answer the question *What management, climatic and environmental factors influence species richness, cover and community composition?* By doing so, we hope to gain an understanding that can be used to improve the management of this unique urban greenspace type for biodiversity, ecosystem function and human wellbeing.

Methods

The study area

The study area consisted of 39 neighbourhoods chosen at random from a set of neighbourhoods all of which were at least partly zoned residential, within the urban growth boundary (UGB) and within the Victorian Volcanic Plain (VVP) bioregion in Melbourne, Australia (population 5 million; Australian Bureau of Statistics 2018). The soils and native flora of the VVP are distinctive and affect gardening conditions. The dominant vegetation across the study area that would have existed prior to European settlement is classified as Plains Grassland of the Victorian Volcanic Plain (DELWP, 2004). We chose residential neighbourhoods because internationally they are the dominant urban land use (e.g. Pauleit & Duhme, 2000). Road verges are present on most streets in Melbourne. Melbourne's climate is wet and temperate (maritime) with rainfall even year-round (Köppen classification Cfb). Mean annual rainfall is 650 mm, with the west of Melbourne drier than the east. A survey of lawn seed suppliers and a major hardware retailer indicated road verges in Melbourne were usually sown with a mix containing a combination of *Cynodon dactylon* (Couch grass), *Lolium* sp. (Ryegrass), *Pennisetum clandestinum* (Kikuyu), *Festuca rubra* (Red Fescue), *Poa pratensis* (Kentucky bluegrass) and *Stenotaphrum secundatum* (Buffalo Grass), with *Cynodon dactylon* the most common (McKays Grass Seeds, 2019; Bunnings hardware stores).

Data collection

Floristic surveys

Floristic surveys were conducted at 158 road verges selected from a set of 917 verges used in a companion study (Marshall *et al.*, unpublished data). We restricted selection of verges to those with a continuous strip of verge bordered by crossovers (driveways), kerb and footpath because tree cut-outs and footpath absence have both been linked to substantially higher incidences of verge gardening compared to standard verges bordered by crossovers, kerb and footpath (Marshall, Grose and Williams, 2019a). We stratified verge selection by time since development, with verges chosen equally from eight age categories (1850–1917; 1918–1947; 1948–1957; 1958–1967; 1968–1977; 1978–1987; 1988–1997; and 1998–2008), because we considered that time since development might have a significant influence on floristic composition. Surveys were conducted during Spring, between 21 September and 1 November 2017, to increase the likelihood of encountering and identifying all species present, particularly annual species.

Species were categorised as turfgrass, non-turfgrass graminoid, woody plant, forb and succulent. The numerous, commonly sown and functionally similar, commercial cultivars of rhizomatous turfgrass species such as *Cynodon dactylon* (Couch grass), *Pennisetum clandestinum* (Kikuyu), *Festuca rubra* (Red Fescue), and *Stenotaphrum secundatum* (Buffalo Grass), were conflated to one single category, rhizhomatous turfgrasses, to avoid time-consuming identification of species difficult to distinguish when mown, and together with *Lolium* sp. (Ryegrass) were classified as turfgrasses. Species other than turfgrasses were identified as garden plants if it appeared that they had been deliberately planted in the road verge. A spontaneous species was considered to be any species not a turfgrass and not a garden plant.

A pilot study was conducted with 0.25 m^2 quadrats on three road verges of varying ages, and all vascular plants were identified to determine species–area accumulation curves, with results suggesting one quadrat per 8 m^2 would include the majority of species on each road verge. Because initial observations indicated considerable variation in species composition along the road verges, especially when trees were present, the length of each road verge was divided into sections, and quadrats placed randomly within each section. In each quadrat, we recorded the presence and estimated percentage cover of all vascular flora, the degree of soil compaction, and overhead canopy openness. The degree of soil compaction was recorded at four points per quadrat using a cone penetrometer, with the depth reached given an applied pressure of 2000 pascals recorded, and results averaged for each quadrat. Canopy cover above each quadrat was obtained by true-colour hemispherical lens photography, with images processed using Gap Light Analyzer (Fraser, Canham and Lertzman, 1999).

Resident socio-demographics and road verge maintenance practices

Data on residents and their road verge maintenance practices were obtained from a questionnaire to which the residents of all properties associated with the study sites had previously responded (Marshall *et al.*, unpublished data). This data included: A) mowing frequency per year (never, 1–5, 6–10, 11–15, Over 15); B) maintenance duration (minutes per week resident spent on non-mowing verge maintenance [none, 1–2, 3–5, 6–10, 11–15, Over 15]); C) maintenance cost [none, $50 or less, $51–$100, $101–$150, over $150]; D) level of enthusiasm for gardening; E) age; F) gender; H) if respondent was a renter or home owner; I) if respondent spoke a language other than English at home; and J) number of years lived in street.

Additional social and environmental data

We also collected data on a range of social and environmental factors that might influence species richness, coverage and community composition. At the neighbourhood scale, these were: 90-day rainfall and mean average rainfall, obtained from Silo Climate Data (Department of Environment and Science, 2018); and the Index of Relative Social Disadvantage (IRSD), a measure of social disadvantage developed by the Australian Bureau of Statistics (Australian Bureau of Statistics, 2015). We used rainfall in previous 90 days as a factor because it has been associated with the growth of annual species in native grasslands in Melbourne (Zeeman *et al.*, 2017). At the street scale we collected data on: the street's date of development, determined by using historic street directories (Collins' street directory, 1922–1963; Melway, 1964–2009; Morgans' street directory, 1917–1944; Moulton's street directory, 1912–1916) and Sands and McDougall (1860–1911) records of property ownership, with street development date categories being 1850–1917, 1918–1947, 1948–1957, 1958–1967, 1968–1977, 1978–1987, 1988–1997 and 1998–2008; and the type of road (i.e. major roads, collector roads, local through roads, local dead-end roads and local cul-de-sac roads) determined by reference to the Victorian Government's VicMap Transport dataset (Department of Environment, Land, Water and Planning, 2017).

Analysis

To investigate the influence of social and environmental factors on species richness and cover, we used all subsets regression in Genstat (VSN International, 2011) to identify the best models predicting species richness and cover. We used species richness as a measure rather than species diversity because we hypothesised road verge floristic communities would be very uneven in their species distribution and we wished to investigate the influence of environmental and social factors on that unevenness in our study. The 16 factors tested in the models were: gardening presence, 90-day rainfall, mean annual rainfall, Index of Relative Social Disadvantage, mowing frequency, canopy openness, date of street development, soil compaction, gender, verge area, road type, maintenance duration, years lived in street, age, rental status, and language spoken at home. All subsets regression minimises the effects of multicollinearity between factors (Graham, 2003). For each response variable we selected the most parsimonious set of factors with an AIC within 2 of the lowest AIC. Multivariate linear regressions was used to determine significant factors and an overall measure of the total amount of variance explained (r^2). To compare the relative influences of each factor, univariate regressions were used to establish the amount of variation explained by each variable. Scatterplots were used to visualise the specific contributions of continuous factors, and for categorical factors we used interval plots with Tukey Pairwise Groupings generated as part of the ANOVA routine in Minitab (Minitab, 2010).

We used ordination techniques to test for patterns in species composition and abundance, using Primer 6 (Clarke and Gorley, 2016) to generate a Bray-Curtis dissimilarity matrix. Prior to generating the matrix, we fourth root transformed species abundance data to reduce the influence of dominant species and to allow patterns in species composition and abundance be more readily discerned in the less abundant species. The dissimilarity matrix was analysed using non-metric multidimensional scaling (NMDS), sequentially removing outliers from the data cloud and stopping before the NMDS stress was too high for reasonable analysis (i.e. while stress < 0.20). We then compared the NMDS data cloud with one generated from the same matrix through Principal Coordinates Analysis (PCO) because, while NMDS is the more common method for ordination, PCO can in some instances reveal patterns in data clouds that NMDS cannot (Anderson, Gorley and Clarke, 2008; Chariton, Pettigrove and Baird, 2016). Visual inspection showed the PCO revealed much clearer clustering of the data into distinct floristic communities compared to the NMDS (Figure 6C.1). To further investigate the clustering pattern revealed in the PCO-transformed data, a new resemblance matrix was generated using data from the first three principle axes of the PCO results. The CLUSTER routine could then be used on that new resemblance matrix to create a hierarchical cluster analysis dendrogram of similarity between the road verge communities that further revealed the clearer clustering present in the PCO data. We selected a similarity level at which to cut the dendrogram by matching as closely as possible the number of floristic communities visually apparent in the PCO (six clusters were readily visible, our cut of the dendrogram at 40% distance generated seven clusters). The validity of the clustering process was confirmed through a 1-way ANOSIM test on the similarities between and within clusters. To characterise the seven identified floristic communities, we combined: a) the results of one-way ANOVAS that tested for differences in variation in each environmental factor of significance against the seven floristic communities; and b) the species identified in a one-way SIMPER as contributing most to floristic community differentiation.

To identify environmental variables that covaried with community structure, we generated a Euclidean distance dissimilarity matrix from the environmental data and tested it against the species abundance Bray-Curtis dissimilarity matrix using the DistLM function (Best selection procedure, AICc selection criteria). DistLM also allows a fitted model to be visualised through distance-based redundancy analysis (dbRDA). Prior to running DistLM, those categorical explanatory factors that couldn't be treated as ordinal into multiple binary factors were divided and treated as sets within the DistLM analysis. AICc (a refinement of AIC) was used rather than AIC because there were less than 40 samples per explanatory factor (Burnham and Anderson, 2003). From the DistLM results we selected the most parsimonious set of factors within two of the set of factors identified as having the lowest AICc. To visualise the effects of these environmental factors identified through DistLM, a canonical analysis of principal coordinates (CAP) was performed on the species abundance resemblance matrix to generate constrained ordinations for each environmental factor. To identify changes in species composition due to varying categories of each environmental variable, SIMPER analysis was used to provide within-category species similarities and between-category species dissimilarities. SIMPER emphasises abundance in its analysis of similarity and dissimilarity, rather than providing indicator species (i.e. species distinctive to a specific category).

Primer was used to create Chao 1 species accumulation curves for cultivated species and spontaneous species.

Not all environmental data was available for all quadrats because in some instances respondents did not answer all survey questions or answered on behalf of the resident who maintained the road verge. Some of the above analyses (e.g. ordination) required a data set without missing values, so these were performed on a reduced data set of 406 quadrats rather than the full data set (after outliers removed) of 536 quadrats. Prior to analysis, soil compaction was log transformed and area of road verge was square root transformed to reduce skewness. We tested for collinearity between environmental factors and found some significant multicollinearity (Table 6.A1) but not enough to warrant the exclusion of any factors.

Results

Species richness

A total of 588 quadrats were surveyed. One hundred and twenty seven unique species were identified, and an additional 23 sets of specimens identified to genus level (Appendix 6B). Of this total 150 taxa, 26 (17.3%) were native species with the rest exotic to Australia. 37 (24.7%) were garden plants, 29 (19.3%) were non-turfgrass graminoids, 82 (54.7%) were forbs, 30 (20.0%) were woody plants and 7 (4.7%) were succulents. Of the 26 native species, half (13) were garden plants. All succulents were garden plants.

The factors we investigated accounted for a relatively small (20.2%) of total species richness variation. Mean annual rainfall had a positive correlation with species richness ($p < 0.001$, $r^2 = 5.2\%$, Table 6.1). Soil compaction had a negative correlation with species richness ($p = 0.049$, $r^2 = 2.7\%$, Table 6.1). Road type correlated with species richness, with major roads and local cul-de-sacs having higher species diversity than local dead-end streets ($p < 0.001$, $r^2 = 3.4\%$, Table 6.1, Figure 6.1). Maintenance duration had a correlation with species richness, with longer periods of non-mowing maintenance associated with reduced species richness ($p = 0.002$, $r^2 = 2.3\%$, Table 6.1, Figure 6.1). Gardening had a significant but negligible correlation with species richness ($p = 0.033$, $r^2 = 0.8\%$, Table 6.1). Species accumulation curves showed the expected asymptotic curve for all spontaneous species, but a more linear relationship for gardened species (Figure 6.2).

Cover

Total cover consisted of 54.2% turfgrasses, 16.0% non-turfgrass graminoids, 27.5% forbs, 1.8% woody plants and 0.4% succulents. Garden plants contributed 3.1% of total cover. Native plants contributed 2.3% of total cover. Quadrat species richness had a mean of 5.6 (SD = 2.6) and cover of mean 90.5% (SD = 33.2%).

The rhizomatous turfgrasses were the most common species by coverage, followed by *Lolium* sp., *Poa annua*, *Trifolium dubium*, *Trifolium subterraneum*, *Plantago lanceolata*, *Bromus catharticus*, *Agapanthus* sp., *Ehrharta erecta* and *Trifolium repens*. The rhizomatous turfgrasses were the most common species by frequency of distribution, followed by *Trifolium dubium*, *Poa annua*, *Lolium* sp., *Plantago lanceolata*, *Taraxacum* sp., *Hypochaeris radicata*, *Oxalis* sp., *Trifolium subterraneum*, and *Cerastium vulgare*. Forty two species occurred in 2% or more of all quadrats. 67 (44.7%) species occurred in only one quadrat. A few species, particularly garden plants, were rare but locally abundant, e.g. *Pelargonium* x *domesticum*, but this was also true of some spontaneous species also, e.g. Stellaria media and the native grass *Rytidosperma setacea*.

The factors we investigated accounted for 34.9% of total cover variation. Mowing frequency had a significant, positive correlation with cover, with quadrats mowed six or more times a year having more cover than those mowed five or less times a year ($p < 0.001$, r^2 = 21.7%, Table 6.1, Figure 6.1). Canopy openness had a significant, positive correlation with cover ($p < 0.001$, r^2 = 17.5%, Table 6.1). 90-day rainfall had a significant, but much weaker negative correlation with cover ($p = 0.001$, r^2 = 3.6%, Table 6.1). Mean annual rainfall had a significant but negligible correlation with cover ($p < 0.001$, r^2 = 0.1%, Table 6.1), as did soil compaction ($p < 0.001$, r^2 = 0.2%, Table 6.1) and verge area ($p = 0.022$, r^2 = 0.1%, Table 6.1). A strong negative correlation existed between the cover of rhizomatous turfgrasses and the combined cover of other species (r^2 = 40.6%).

Community composition

After removing outliers (which were mostly quadrats with no species or single, rare species), the NMDS had a stress of 0.23 in two dimensions and a stress of 0.17 in three dimensions, and a degree of clustering was evident (Figure 6C.1). Principal coordinate analysis (PCO) showed a visually greater degree of clustering, with six floristic communities apparent (Figure 6C.1), with the first three axes accounting for 42.1% of fitted variation.

The dendrogram generated from the PCO data identified seven floristic communities when cut at 40% similarity. An ANOSIM test of clustering showed this clustering was significant (p = 0.001), with a high global R of 0.833, indicating 83.3% of samples correctly placed within clusters. The seven distinct floristic communities were distinguished in the first instance by the presence or absence of gardening and by the presence or absence of rhizomatous turfgrasses.

When rhizomatous turfgrasses were present, floristic communities were further distinguished by the presence or absence of turfgrass species *Lolium* sp. and *Poa annua*, as well as *Trifolium dubium*. When rhizomatous turfgrasses were absent, floristic communities were further distinguished by the dominance or co-dominance of *Taraxacum* sp., *Ehrharta erecta*, *Lolium* sp., *Bromus catharticus* and *Trifolium dubium*. The environmental factors that best modelled community composition also had significant correlations with the seven floristic communities (Figure 6.3), as did species richness and cover (Figure 6.3). These relationships are summarised in Table 6.2.

The set of environmental factors that best modelled quadrat species composition had 10 factors: gardening, 90-day rainfall, mean annual rainfall, Index of Relative Social Disadvantage, mowing frequency, canopy openness, date of street development, soil compaction, gender and road type. These factors explained 14.9% of total variation. Marginal tests showing the univariate strength of effect of these factors are shown in Table 6.1, and by this measure gardening was the most significant factor, followed by 90-day rainfall, mean annual rainfall, the Index of Relative Social Disadvantage, mowing frequency, canopy openness, street date, soil compaction, gender, and road type. A dbRDA plot with environmental factors overlaid is presented in Figure 6C.2, and with a minimum threshold for correlation set at 0.3 reduces this set of factors to seven. Even though only 14.9% of total variation in community composition is explained, constrained canonical analysis of principal coordinates (CAP) ordinations show the distinct effects of these factors on community composition (Figures 6C.3A–F; cultivation is a binary outcomes and cannot be usefully shown through CAP and is shown instead through an unconstrained dbRDA, Figure 6C.3G).

Gardened quadrats had lower abundances of rhizomatous turfgrasses, *Poa annua, Trifolium* spp., *Lolium* sp., *Plantago lanceolata* and *Hypochaeris radicata*, but higher abundances of *Taraxacum* sp. and *Ehrharta erecta*. Gardened quadrats included species not found in ungardened quadrats, most notably *Agapanthus* sp. and all succulents. Gardened quadrats had very low similarity between quadrats.

Differences in mowing frequency did not affect species richness. However, similarity between species composition of quadrats within the same mowing category increased from the least mown to the most frequently mown category, i.e. quadrats with less mowing were more varied in their species composition than quadrats with more frequent mowing, though overall there was no difference in species richness between mowing categories. More frequently mown quadrats had more cover. Mowing favoured rhizomatous turfgrasses. The dominance of rhizomatous turfgrasses (and the increased total cover) perhaps then influenced the abundance of a range of small species such as *Polycarpon tetraphyllum* and *Cerastium vulgare*, species which had less opportunity to find interstitial spaces in quadrats dominated by the thickly growing, rhizomatous turfgrass species. Frequency of mowing also influenced the abundance of other grass species, with infrequent mowing favouring *Lolium* sp., *Poa annua*, *Bromus catharticus* and *Hordeum* sp., and lack of mowing favouring *Ehrharta erecta*. Mowing increased the abundance of flatweeds (e.g. *Hypochaeris radicata*, *Plantago lanceolata*, *Arctotheca calendula*), but *Taraxacum* sp. was favoured by infrequent mowing rather than frequent mowing. Mowing also increased the abundance of all *Trifolium* spp. recorded. Mowing-intolerant species such as *Sonchus oleraceus* became more abundant as mowing frequency decreased. Garden plants also became more frequent as mowing frequency decreased.

Soil compaction had a linear relationship with street age, and was higher in newly developed streets than older streets.

Discussion

The road verge as a whole included a substantial number of species (150), and the majority of those were exotic (82.7%). Unlike lawn studies from America and Europe, which found the majority of species to be native, our Melbourne results showed that of the 113 lawn flora in our study only 11.5% were native, a result similar to Stewart *et al*. (2009), who found only 13% of species to be native in Christchurch, New Zealand.

The 113 lawn species found is comparable with other studies: Stewart *et al*. (2009) found 127 species; Wheeler *et al*. (2017) found 63–152 species across domestic lawns of seven cities in the USA; in Paris, 79 species were found across domestic and public lawns (Bertoncini *et al.*, 2012); Knapp *et al*. (2012) found 233 spontaneous flora in Minnesota yards, while a British study (Thompson *et al.*, 2004) found 159 species including bryophytes in domestic lawns. Species richness (for lawn and gardened plants combined) was 5.6 per 1 m x 1 m quadrat with a standard deviation of 2.6 and a range of 0–15, comparable to Thompson *et al*. (2004), who found from 4 to 20 species per entire lawn, and Bertoncini *et al*. (2012), who found from 1 to 24 species with a mean of 3.5 per 30 cm x 30 cm quadrat. We found lower forb species richness (56.7% of total species) than either Stewart *et al*. (2009), who found 74.0%, or Bertoncini *et al*. (2012), who found 83%. Our lower proportion of forbs is due in part to our inclusion of gardened plants, many of which were woody or other graminoids. Our species accumulation curves for gardened plants compared to spontaneous species were similar to those of Thompson *et al*. (2004).

Many of the species identified in this study as most common either by total abundance or frequency of occurrence at a quadrat level were also reported as most common by other studies (*Cynodon dactylon*, *Lolium perenne*, *Poa annua*, *Trifolium repens*, *Taraxacum* sp., *Plantago lanceolata*, *Oxalis stricta*, *Bellis perennis*) (Müller, 1990; Thompson *et al.*, 2004; Cheng *et al.*, 2008; Ignatieva and Stewart, 2009; Bertoncini *et al.*, 2012; Wheeler *et al.*, 2017). This finding supports the proposition that lawn vegetation is generally homogenous internationally (Stewart *et al.*, 2009; Knapp *et al.*, 2012; Wheeler *et al.*, 2017). However, verge gardening introduced less common species, including many natives, and so served to add regional diversity to the lawn of the road verge.

The range of factors we examined explained only 20.2% of variation in species richness, 34.9% in cover and 17.4% in community composition. These low figures may be due in part to our omission of potential explanatory factors. Surrounding land use, previous land use, soil type, temperature, household income, altitude, fertilisation rates, irrigation rates, lawn care behaviours such as weeding and the removal of cuttings, and the presence of leaf litter, have been found in other studies to influence richness, abundance and composition (Müller, 1990; Thompson *et al.*, 2004; Briggs *et al.*, 2006; Williams *et al.*, 2006; Stewart *et al.*, 2009; Bertoncini *et al.*, 2012; Wheeler *et al.*, 2017; Roman *et al.*, 2018). It is likely that data on these additional factors would have increased the variation explained, but it was unavailable. However: fertilisation and irrigation of road verges is uncommon in Melbourne; the Index of Relative Social Disadvantage we used incorporates household income; our study was confined to the Victorian Volcanic Plain, which has a consistent soil type and very little variation in altitude; and the effects of temperature have been shown to be small (Wheeler *et al.*, 2017). A significant portion of the unexplained variation may be due to stochastic processes. For instance, road verges are a highly disturbed environment (Gilbert, 1989; van der Ree, Grilo and Smith, 2015), subject to trampling, excavation for access to underground utilities, deposition of debris from construction, excess nutrients from dog and cat urine and faeces, exposure to pollution, and weekly cycles of domestic waste removal. Maintenance is also highly variable, changing as property owners or renters move.

Of the native species observed, all those that were garden plants were common in nurseries. The other, spontaneous, native species were all typical species of Melbourne's pre-settlement grassland communities (DELWP, 2019). They included five species of *Rytidosperma* (Wallaby Grass); a single example of *Convolvulus angustissimus*; a small unidentified sedge (likely to be native but lack of flowers made confirmation difficult) that had considerable extent in the quadrats in which it was present; the prostrate, hardy and vigorous *Einadia* *nutans*, often located beneath trees, and which provides food for butterflies and their larvae (Greening Australia, 2019); and three small forbs, *Veronica gracilis*, *Cotula australis* and *Dichondra repens*. *Veronica gracilis* is considered well suited to mown lawns because it spreads rhizomatously and won't be shaded-out by larger plants (Australian Plants Society Keilor Plains Group, 2012). *Cotula australis* is native to Australia and New Zealand but is a common weed found in many countries. It roots at the nodes of its stems (Agriculture Victoria, 2019; Atlas of Living Australia, 2019), and its flowers provide nectar for butterflies (Shire of Yarra Ranges, 2019). *Dichondra repens* is a groundcover able to withstand light traffic and is now used in horticulture as a lawn alternative, as it has a low, creeping habit. It is also native to New Zealand and was one of the few native forbs identified by Stewart *et al*. (2008), as was *Cotula australis*. These three low-growing forbs all reproduce vegetatively, and their size and means of reproduction adapt them to the mown environment.

In addition to native species, much of the species diversity in our study came from low-growing species such as *Lotus* sp., *Trifolium* sp., *Medicago* sp., *Galenia* sp., *Galium* sp., *Stellaria* sp., *Polycarpon* sp., *Lepidium* sp., *Gamochaeta* sp. and *Cerastium* sp., many of which produce attractive flowers that may be beneficial to pollinating insects. Mowing frequency strongly influences the abundance of such floral resources. Lerman *et al*. (2018) showed that changing mowing frequency from once per week to once per three weeks (the 'lazy lawnmower') can increase the number of flowers by 250%. We found that increased mowing frequency increased the total vegetation cover of road verges but also the cover of *Trifolium* spp., which are one of the main providers of floral resources within the road verge (Larson, Kesheimer and Potter, 2014), possibly because mowing removed taller species that would otherwise overshadow the *Trifolium* spp.. To maximise *Trifolium* spp. flowering, the best mowing regime may be one that is frequent when flowers are not either developing or present, and less frequent during their flowering phase. The cutting height that the mower is set to will also be an important consideration (Lerman *et al.*, 2018).

As with other studies (Thompson *et al.*, 2004; Stewart *et al.*, 2009; Wheeler *et al.*, 2017), this study shows road verge lawn vegetation is dominated by the species common in locally available seed mixes. This study has shown that there are opportunites for many other species, with non-sown, spontaneous species accounting for almost half of total cover. Though we currently know little about the specific ecological roles of many verge flora, with further research it should be possible to develop new seed mixes that improve the biodiversity and ecosystem benefits of the road verge, for instance by introducing a greater number of native flowing herbs as a resource for pollinators.

Gardening in the road verge increased species diversity, especially native species diversity, and it introduced life forms otherwise not present, e.g. succulents. By increasing the quantity, diversity and complexity of vegetation, it is likely to have increased the diversity of a range of faunal taxa (Wichmann *et al.*, 2004; Garden, McAlpine and Possingham, 2010; Threlfall *et al.*, 2017), although the effect is not universal (Leonard, McArthur and Hochuli, 2018). Greater plant species richness has also been linked to greater carbon storage and improved pollination (Isbell *et al.*, 2017; Chen *et al.*, 2018). Gardening improves soil litter decomposition rates, water holding capacity and nutrient supply when compared to lawn (Tresch *et al.*, 2019) and reduces soil compaction, thus improving the verge's capacity to infiltrate water and mitigate the urban stream syndrome (Walsh *et al.*, 2005). Given road verges can account for over a third of public greenspace (Richards *et al.*, 1984; Marshall, Grose and Williams, 2019b), gardening the road verge clearly has considerable potential to provide a range of positive biodiversity and ecosystem health benefits.

Conclusion

Verge gardening has significant potential to improve the biodiversity and ecosystem function of the road verge because it is the main means by which native species are introduced into this type of public space, which otherwise consists of mostly exotic species. Garden plants increased species diversity and added structure and biomass, potentially providing better resources and habitat for many species (Threlfall *et al.*, 2016). Gardening also decompacted soils and is known to increase soil health (Tresch *et al.*, 2018). Altering mowing frequency may be a useful strategy for increasing the benefits that lawn species provide to pollinators, but the relationships between mowing frequency, cover and flower abundance appear complex and require additional research. Because varying management strategies variously filter species for specific traits, a broad variety of management practices is likely to most provide species richness at a city-scale (Bertoncini *et al.*, 2012; Norton *et al.*, 2019). Several native species appear suitable for inclusion in novel seed mixes that would increase the benefits provided by road verges, though again further research is needed.

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Figures

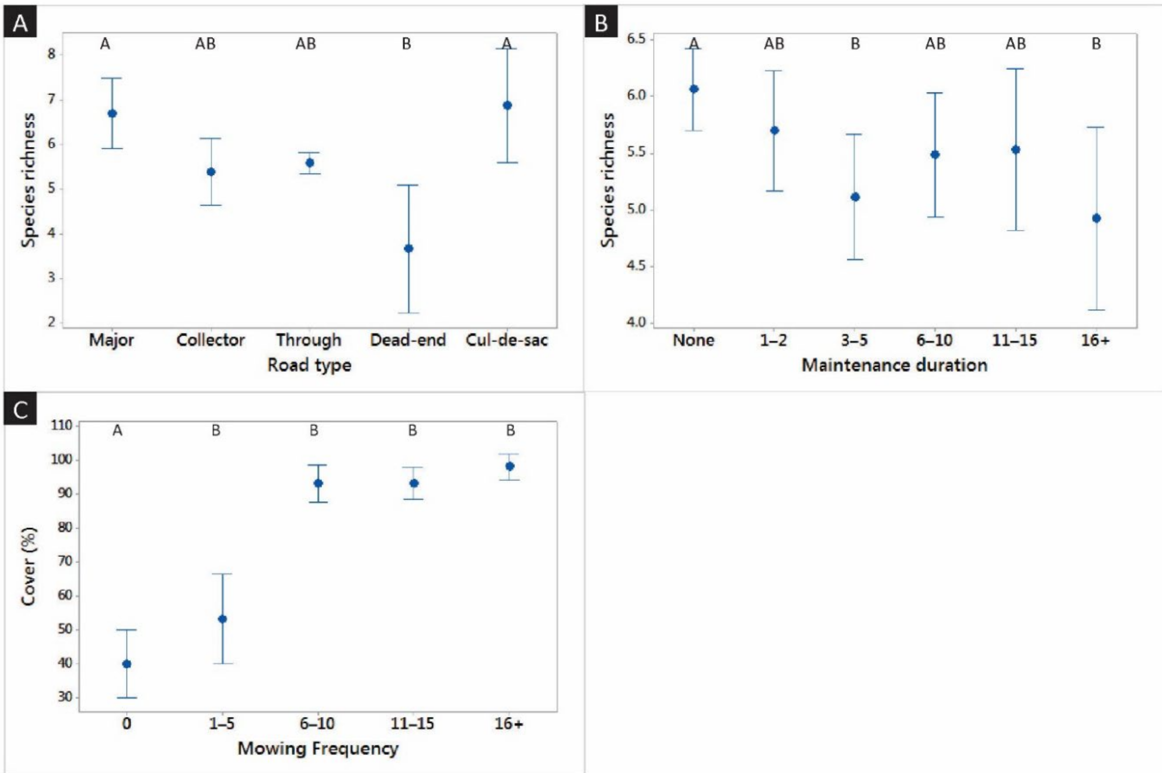

Figure 6.1: Interval plots of A) species richness of the verges of five categories of road; B) species richness for six maintenance durations (minutes per week); and C) cover for five categories of mowing frequency per year; for 536 quadrats across 39 neighbourhoods in Melbourne, Australia. Horizontal lines show median, first and third quartiles, asterisks are outliers. Lettered labels represent indicate Tukey Pairwise Comparisons at 95% confidence (80% for maintenance duration).

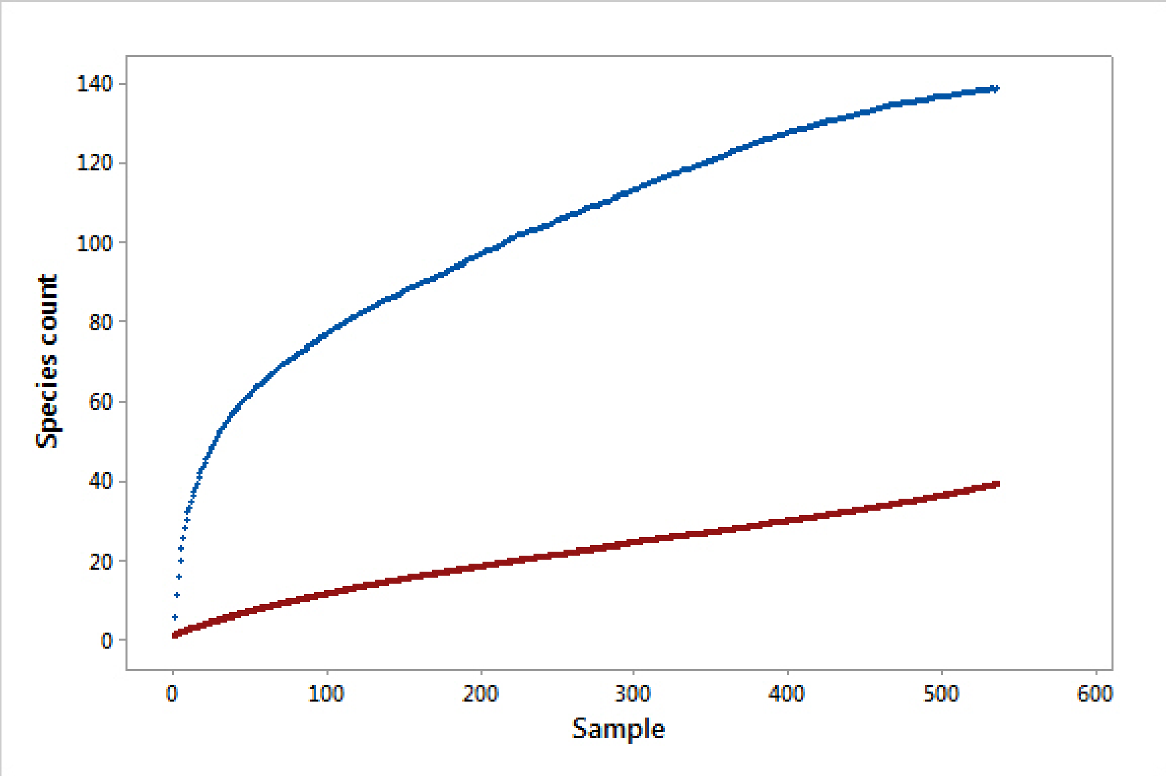

Figure 6.2: Chao 1 species accumulation curves for gardened (red line) and non-gardened (blue line) and gardened species for 536 road verge quadrats across 39 neighbourhoods in Melbourne, Australia.

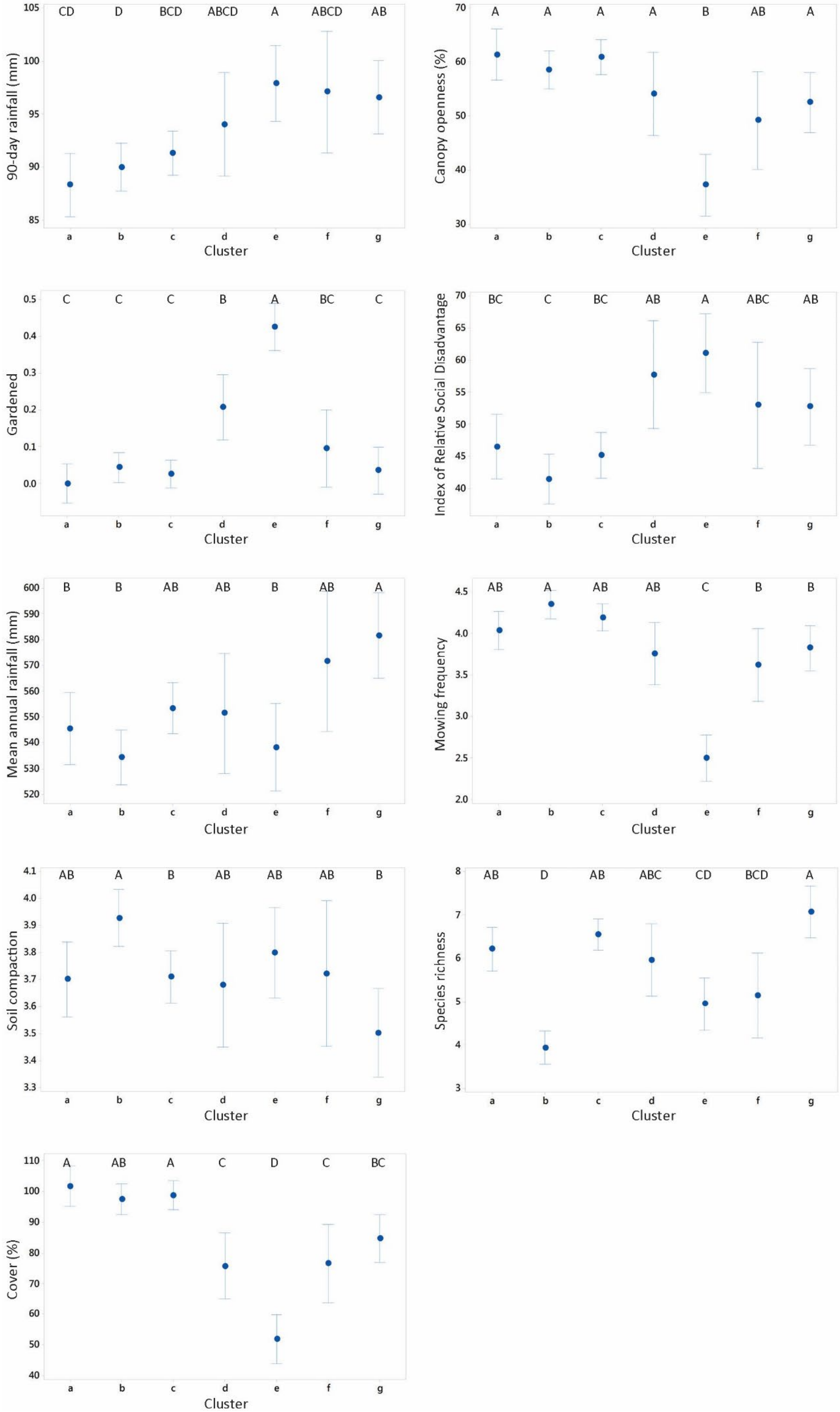
CD D BCD ABCD A ABCD AB
90-day rainfall (mm)
Cluster
A A A A B AB A
Canopy openness (%)
Cluster
C C C B A BC C
Gardened
Cluster
BC C BC AB A ABC AB
Index of Relative Social Disadvantage
Cluster
B B AB AB B AB A
Mean annual rainfall (mm)
Cluster
AB A AB AB C B B
Mowing frequency
Cluster
AB A B AB AB AB B
Soil compaction
Cluster
AB D AB ABC CD BCD A
Species richness
Cluster
A AB A C D C BC
Cover (%)
Cluster

Figure 6.3: Interval plots for seven environmental factors, and species richness and cover, showing their relationship to seven floristic communities identified from community composition data for 536 road verge quadrats across 39 neighbourhoods in Melbourne, Australia. Dots show mean values, whiskers show 95% confidence intervals and capital lettered labels indicate Tukey Pairwise Comparisons at a 95% confidence levels.

Tables

Table 6.1: Factors identified as influencing species richness, cover and community composition for 536 road verge quadrats across 39 neighbourhoods in Melbourne, Australia. Individual r2 values are derived from univariate tests and allow comparison of the relative strength of influence of individual factors.

	Species richness		Cover		Community composition
Factor	P-Value	r^2	P-Value	r^2	r^2
Mean annual rainfall	<0.001	5.2	<0.001	0.1	2.7
Soil compaction	0.049	2.7	<0.001	0.2	1.6
Road type	<0.001	3.4			0.6
Maintenance duration	0.002	2.3			
Gender	0.163				1.0
Gardening	0.033	0.8			3.3
IRSD	0.113				2.7
Canopy openness			<0.001	17.5	2.1
Mowing frequency			<0.001	21.7	2.6
90-Day rainfall			0.001	3.6	2.8
Verge area			0.022	0.1	
Total r^2 (multivariate)		20.2		34.9	14.9

Table 6.2: Identifying features of vegetation communities (clusters) in 536 road verge quadrats across 39 neighbourhoods in Melbourne, Australia.

Cluster	General description	Characteristic Species	Species absent	Species richness	Cover	Gardening	Mowing	90-day rainfall	Canopy	Soil compaction	Mean annual rainfall	IRSD
A	Full cover of lawn comprising rhizomatous turfgrasses and *Lolium* sp. but with *Poa annua* absent, tending to occur in lower rainfall areas	Rhizomatous turfgrasses, *Lolium* sp.	*Poa annua*		Full	No		Lower			Lower	More disadvantaged
B	Full cover of lawn comprising rhizomatous turfgrasses and *Poa annua* but with *Lolium* sp. absent, generally species poor, tending to occur on relatively uncompacted soils and in lower rainfall areas	Rhizomatous turfgrasses, *Poa annua*	*Lolium* sp. *Trifolium dubium*	Poor	Full	No		Lower		Less	Lower	More disadvantaged
C	Full cover of lawn comprising rhizomatous turfgrasses, *Lolium* sp. and *Poa annua*, with significant presence of *Trifolium dubium*	Rhizomatous turfgrasses, *Trifolium dubium* *Poa annua*			Full	No		Lower				More disadvantaged
D	Gardened verges without rhizomatous turfgrasses and *Lolium* sp. but well-covered by *Poa annua* and *Trifolium dubium*	*Poa annua* *Trifolium dubium*	Rhizomatous turfgrasses, *Lolium* sp.		Well	Yes						Less disadvantaged
E	Gardened verges, rarely mown, shady, species poor, with little cover	*Taraxacum* sp. *Ehrharta erecta*	Rhizomatous turfgrasses, *Trifolium dubium*	Poor	Sparse	Yes	Rarely	Higher	Shaded		Lower	Less disadvantaged
F	Gardened verges, well-covered with lawn comprising *Lolium* sp. but with rhizomatous turfgrasses and *Poa annua* absent, species poor	*Lolium* sp. *Bromus catharticus*	Rhizomatous turfgrasses, *Poa annua*	Poor	Well	Yes		Higher				Less disadvantaged
G	Ungardened, well-covered and species rich without lawn of rhizomatous turfgrasses, *Lolium* sp. or *Poa annua*, on compacted soils	*Trifolium dubium* *Lolium* sp.	Rhizomatous turfgrasses	Rich	Well	No		Higher		More	Higher	Less disadvantaged

Appendix 6A: Collinearity between factors

Table 6.A1: Pearson correlation between factors investigated as possible drivers of species richness, abundance and community composition or the road verge flora of 406 road verge quadrats across 39 neighbourhoods in Melbourne, Australia.

	Area	IRDS	Street date	Road type	Soil compaction	Canopy openness	90-Day rainfall	Mean annual rainfall	Cultivated	Mowing Frequency	Maintenance duration	Maintenance cost	Age	Gender	Language	Years in street
IRDS	-0.01															
Street date	0.22	-0.04														
Road type	-0.18	0.05	0.10													
Soil compaction	0.04	0.02	-0.20	-0.03												
Canopy openness	0.27	-0.21	0.42	-0.31	-0.19											
90-Day rainfall	-0.36	0.32	-0.50	0.15	0.07	-0.35										
Mean annual rainfall	-0.33	0.13	-0.25	0.24	-0.02	-0.22	0.71									
Cultivated	0.01	0.21	-0.09	0.01	0.19	-0.21	0.20	0.06								
Mowing Frequency	0.20	-0.33	0.21	-0.05	-0.07	0.29	-0.29	-0.11	-0.45							
Maintenance duration	0.13	-0.03	-0.08	-0.05	0.02	-0.06	0.06	-0.04	0.06	-0.02						
Maintenance cost	0.23	-0.23	0.00	-0.10	0.06	0.14	-0.17	-0.19	-0.02	0.18	0.36					
Age	-0.18	-0.03	-0.07	-0.07	-0.01	-0.09	0.11	0.17	0.07	-0.01	0.18	0.00				
Gender	0.08	-0.01	-0.07	-0.04	-0.05	-0.04	0.02	0.08	-0.06	0.08	-0.04	0.25	0.16			
Language	0.06	-0.13	-0.01	-0.21	-0.08	0.12	-0.04	-0.01	-0.08	0.07	0.09	0.21	-0.07	0.32		
Years in street	-0.15	-0.06	0.04	-0.04	-0.03	-0.03	-0.04	0.02	0.03	0.13	-0.02	-0.01	0.31	-0.01	-0.16	
Rental status	0.08	0.02	0.13	-0.13	-0.02	0.08	-0.19	-0.25	-0.02	0.02	0.06	0.06	0.07	0.12	-0.01	0.20

Appendix 6B: Species list

Table 6.B1: All vascular plant species identified, their nativeness, status as a garden plant, and life form, for 536 quadrats across 39 neighbourhoods in Melbourne, Australia.

Species	Native	Garden plant	Life form
Acer sp.	No	No	Woody plant
Acetosella vulgaris	No	No	Forb
Aeonium atropurpureum	No	Yes	Succulent
Aeonium haworthii	No	Yes	Succulent
Agapanthus sp.	No	Yes	Other graminoid
Aira sp.	No	No	Other graminoid
Allium sp.	No	No	Forb
Aphanes arvensis	No	No	Forb
Arctotheca calendula	No	No	Forb
Atriplex semibaccata	Yes	Yes	Forb
Banksia petiolaris	Yes	Yes	Forb
Bellis perennis	No	No	Forb
Brassica sp.	No	No	Forb
Bromus catharticus	No	No	Other graminoid
Bromus hordaceus	No	No	Other graminoid
Capsella bursa-pastoris	No	No	Forb
Cardimine hirsuta	No	No	Forb
Carex apressa	Yes	Yes	Other graminoid
Carpobrotus sp.	No	Yes	Succulent
Cerastium glomeratum	No	No	Forb
Cerastium vulgare	No	No	Forb
Chlorophytum comosum	No	Yes	Woody plant
Choisya ternata	No	Yes	Woody plant
Clematis microphylla	Yes	Yes	Woody plant
Convolvulus angustissimus	Yes	No	Forb
Convolvulus arvensis	No	No	Forb
Conyza sp.	No	No	Forb
Coprosma repens	No	No	Woody plant
Correa alba	Yes	Yes	Woody plant
Correa glabra	Yes	Yes	Woody plant
Cotoneaster pannosus	No	No	Woody plant
Cotula australis	Yes	No	Forb
Cotyledon orbiculata macrantha	No	Yes	Succulent
Crassula tetragona	No	No	Woody plant

Species	Native	Garden plant	Life form
Cynosurus cristatus	No	No	Other graminoid
Dactylis glomerata	No	No	Other graminoid
Dianella tasmanica	Yes	Yes	Other graminoid
Dichondra repens	Yes	No	Forb
Dietes sp.	No	Yes	Forb
Dodonea viscosa	Yes	Yes	Woody plant
Echium candicans	No	No	Woody plant
Ehrharta erecta	No	No	Other graminoid
Ehrharta longiflora	No	No	Other graminoid
Einadia nutans	Yes	No	Woody plant
Epilobium sp.	No	No	Forb
Erigeron karvinskianus	No	No	Forb
Erodium cicutarium	No	No	Forb
Erodium malacoides	No	No	Forb
Erodium mocshatum	No	Yes	Forb
Escheveria sp.	No	Yes	Succulent
Euphorbia charachias	No	No	Woody plant
Euphorbia peplus	No	Yes	Woody plant
Euphorbia prostrata	No	No	Woody plant
Festuca arundinacea	No	No	Other graminoid
Fraxinus angustifolia	No	No	Woody plant
Fumaria muralis	No	No	Forb
Galenia pubescens	No	No	Forb
Galium aparine	No	No	Forb
Galium divaricatum	No	No	Forb
Galium murale	No	No	Forb
Gamochaeta sp.	No	Yes	Forb
Gazania sp.	No	No	Forb
Geranium molle	No	No	Woody plant
Helminthotheca echiodes	No	No	Forb
Hordeum sp.	No	No	Other graminoid
Hypochaeris glabra	No	Yes	Forb
Hypochaeris radicata	No	Yes	Forb
Iris germanica	No	No	Forb
Juncus pallidus	Yes	Yes	Other graminoid
Lactuca saligna	No	Yes	Forb
Lampranthus sp.	No	No	Succulent
Lavandula dentata	No	No	Woody plant
Lepidium virginicum	No	Yes	Forb

Species	Native	Garden plant	Life form
Lolium sp.	No	No	Turfgrass
Lomandra longifolia	Yes	Yes	Other graminoid
Lotus angustissimus	No	No	Forb
Lotus corniculatus	No	No	Forb
Lotus creticus	No	No	Forb
Lysimachia arvensis	No	No	Forb
Malva sp.	No	No	Forb
Medicago arabica	No	No	Forb
Medicago lupina	No	No	Forb
Medicago minima	No	No	Forb
Medicago orbicularis	No	No	Forb
Medicago polymorpha	No	Yes	Forb
Meliotus indicus	No	No	Forb
Mesembranthemum sp.	No	Yes	Succulent
Modiola caroliniana	No	Yes	Forb
Myoporum parvifolium	Yes	Yes	Woody plant
Nandina domestica	No	No	Other graminoid
Nasella neissiana	No	Yes	Other graminoid
Nasella trichotoma	No	No	Other graminoid
Osteospermum cv	No	No	Woody plant
Oxalis sp.	No	Yes	Forb
Paspalum dilatatum	No	Yes	Other graminoid
Pelargonium x domesticum	No	No	Woody plant
Pelargonium x hortorum	No	No	Woody plant
Plantago coronopus	No	No	Forb
Plantago lanceolata	No	No	Forb
Plectranthus neochilus	No	Yes	Forb
Poa annua	No	No	Other graminoid
Poa labillardieri	Yes	Yes	Other graminoid
Polycarpon tetraphyllum	No	No	Forb
Polygonum aviculara	No	No	Forb
Prunus cerasifera	No	Yes	Woody plant
Ranunculus muricatris	No	No	Forb
Rhagodia spinescens	Yes	Yes	Woody plant
Romulea sp.	No	No	Forb
Rumex sp.	No	No	Forb
Rytidosperma caespitosa	Yes	No	Other graminoid
Rytidosperma penicillatum	Yes	No	Other graminoid
Rytidosperma racemosa	Yes	No	Other graminoid

Species	Native	Garden plant	Life form
Rytidosperma setacea	Yes	No	Other graminoid
Rytidosperma tenuior	Yes	No	Other graminoid
Sagina apetala	No	Yes	Forb
Salvia verbenacia	No	No	Woody plant
Santolina rosmarinifolia	No	No	Woody plant
Schinus molle	No	No	Woody plant
Secale cereale	No	No	Other graminoid
Sedge (unidentified)	N/A	No	Other graminoid
Solanum nigrum	No	No	Forb
Solvia sessilis	No	No	Forb
Sonchus asper	No	No	Forb
Sonchus oleraceus	No	No	Forb
Sporobolus indicus	No	No	Other graminoid
Stellaria media	No	Yes	Forb
Taraxacum sp.	No	No	Forb
Tradescantia pallida	No	No	Woody plant
Tragopogon sp.	No	No	Forb
Trifolium dubium	No	No	Forb
Trifolium fragiferum	No	No	Forb
Trifoilium cernum	No	No	Forb
Trifoilium pratense	No	No	Forb
Trifolium glommereratum	No	No	Forb
Trifolium micranthum	No	No	Forb
Trifolium repens	No	No	Forb
Trifolium subterraneum	No	No	Forb
Trifolium suffocatum	No	No	Forb
Trifolium tomentosum	No	No	Forb
Turfgrass spececies, being *Cynodon dactylon, Pennisetum clandestinum, Festuca rubra, Stenotaphrum secundatum*	Yes	No	Turfgrass
Ulex europaeus	No	No	Woody plant
Veronica arvensis	No	No	Forb
Veronica gracilis	Yes	No	Forb
Veronica hederifolia	No	No	Forb
Veronica persica	No	Yes	Forb
Vicia sativa	No	No	Forb
Viola odorata	No	Yes	Forb
Vulpia sp.	No	Yes	Other graminoid
Westringia fruticosa	Yes	Yes	Woody plant
Xerochrysum palustre	No	Yes	Forb

Appendix 6C: Ordination

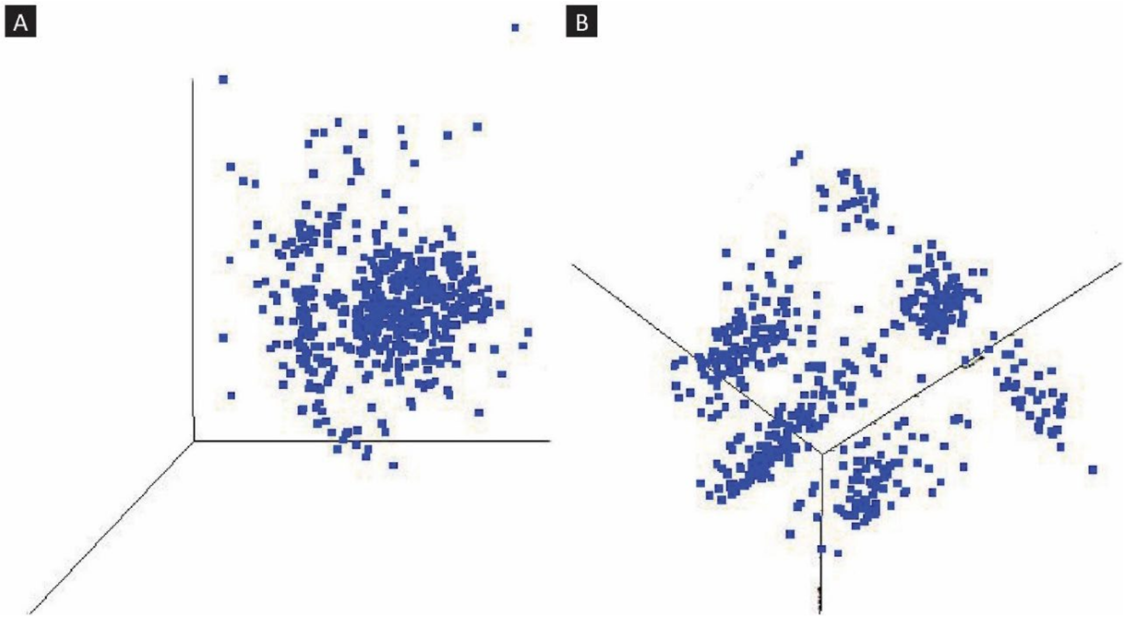

Figure 6C.1: A) NMDS ordination (stress = 0.17) and B) principal coordinate analysis (PCO) ordination of species cover–abundance data; both rotated in three dimensions to maximise visual clustering of the data cloud, for 406 road verge quadrats across 39 neighbourhoods in Melbourne, Australia. Clustering is more clearly expressed in PCO ordination compared to NMDS ordination.

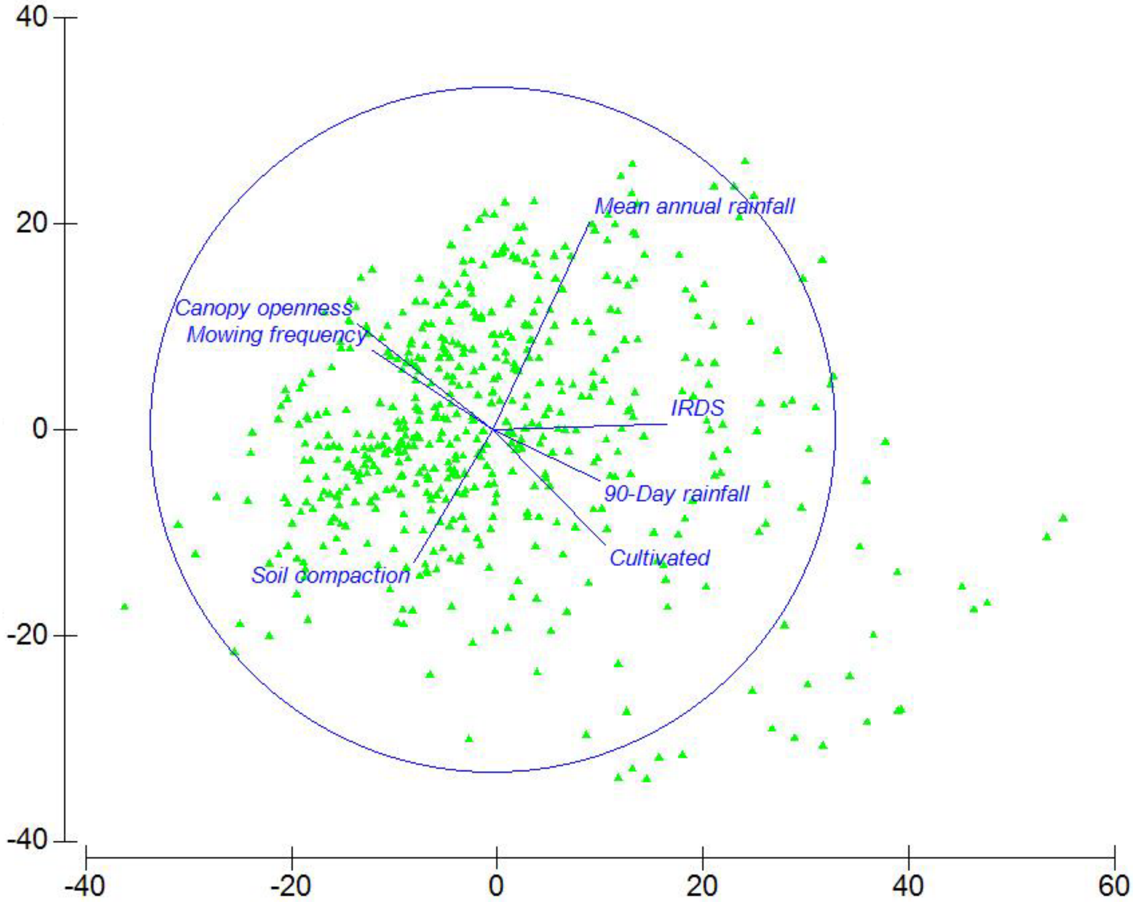

Figure 6C.2: dbRDA plot of species community composition for 406 road verge quadrats across 39 neighbourhoods in Melbourne, Australia. Vectors show the seven factors most correlated (correlation threshold set at 0.3) for the first two dbRDA axes. The first axis explains 37.9% of fitted variation, the second 24.5% of fitted variation.

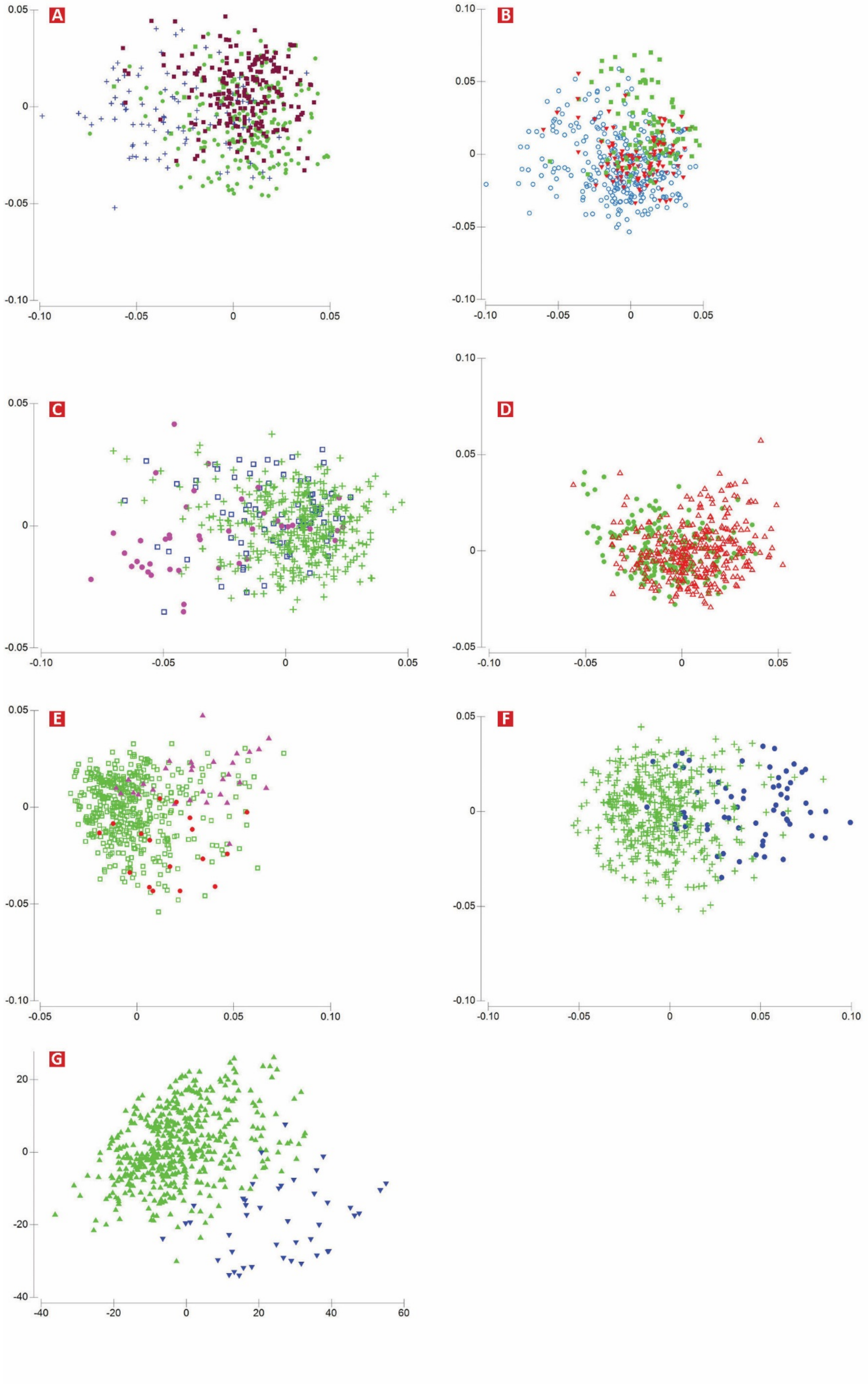
A
B
C
D
E
F
G
0.05
0
-0.05
-0.10
0.10
-0.10
-0.05
0
0.05
0.10
20
0
-20
-40
40
60

Figure 6C.3: Canonical analysis of principal coordinates (CAP) comparing community composition for six environmental factors influencing community composition for 406 road verge quadrats across 39 neighbourhoods in Melbourne, Australia. A) 90-day rainfall: lower (brown squares), intermediate (green circles) and higher (blue crosses) levels. B) Mean annual rainfall: lower (green squares), intermediate (red triangles) and higher (blue circles) levels. C) Canopy openness: lower (pink circles), intermediate (blue squares) and higher (green crosses) levels. D) Soil compaction: lower (green circles) and higher (red triangles) levels. E) Mowing: never mown (pink triangles), infrequently mown (red circles) and frequently mown (green squares). F) Index of Relative Social Disadvantage (IRSD): lower (green crosses) and highest (blue circles) levels. G) dbRDA ordination comparing community composition for quadrats containing gardened species (blue) and quadrats without gardened species (green).

Chapter 7
Conclusions

The aim of this thesis was to answer the question *What spatial, environmental and social drivers underpin road verge extent, distribution and vegetation?* In particular it sought to characterise the extent and distribution of road verges and the verge gardening undertaken by residents, investigate residents' beliefs regarding verge gardening and verge maintenance, and to characterise the flora of the road verge understorey and lawn. By filling these gaps in our understanding of road verges, this thesis has contributed knowledge important to developing better strategies to maximise the benefits that urban greening can provide.

The foremost conclusion of this thesis is that road verges have much greater potential than previously thought to provide benefits such as the provision of habitat and resources, the interception and infiltration of rainfall, mitigating the urban heat island effect, filtering pollution, contributing to sense of community, promoting walkability and improving human health and wellbeing. Road verges (together with other areas of vegetation within the road easement defined as street greenery) constituted 7.0% of total land use, and a remarkable 36.7% of public green space (Chapter 3). This is a proportion comparable to the contribution of the parks and reserves that are traditionally seen as public green space. This considerable extent occurs because road easements are an ever-present and defining structure of many cities, occupying almost a quarter (22.4% in this study, Chapter 4) of all urban land. Despite its surprising extent, the road verge is often overlooked in urban greenspace surveys (VEAC, 2011; O'Sullivan *et al.*, 2017). It should be noted that this study did not examine all road verge within the city, and these figures are extrapolated from sites distributed in residential areas that did not include the central core of the city. Nevertheless, the extent of the road verge, combined with its broad distribution within cities, makes the road verge a green space component of fundamental importance to our urban ecosystems.

Road verges were not evenly distributed across the neighbourhoods of this study. How 'green' a neighbourhood's streets were, i.e. the proportion of the road easement that was greenspace, was correlated with several factors. Older suburbs in the inner-city had streets that were less green than newer suburbs on the urban fringe, although older suburbs often appear greener because they have older, larger trees with more canopy. Neighbourhoods with greater social disadvantage had greener streets than more socially advantaged neighbourhoods, as did neighbourhoods with greater residential parcel size, and neighbourhoods with less footpaths. Moreover, the types of road verges were also unevenly distributed, with tree-cut outs common in older suburbs but rare in newer suburbs, and streets without footpaths only common from 1977 onwards. This uneven distribution of verge means that the potential benefits of road verges will vary by neighbourhood across the urban area. In addition, streets with proportionally more verge were associated with residential land with proportionally more yard (i.e. garden), thus reinforcing the uneven distribution of the benefits derived from urban green space.

Verge gardening by residents was common, occurring in 22.1% of verges and in almost every block of every neighbourhood. Verge gardening increased the overall species richness of verges, dramatically increased the number of native species, and introduced structural complexity, though the total cover of resident-planted understorey was only 3.1% of all road verge cover. The way a neighbourhood was designed influenced the likelihood of verge gardening, with properties with no adjacent footpath 5.3 times more likely to have understorey verge gardening and 2.1 times more likely to have a resident-planted street tree than those with a footpath. Properties with tree cut-outs were 1.8 times more likely to have understorey verge gardening than properties with standard verges. Residents were more likely to verge garden on local roads and cul-de-sacs rather than busy roads, and in neighbourhoods where a higher proportion of residential parcel was yard. Urban design can encourage resident greening of public space, and verge gardening has considerable potential to contribute to urban greening. Urban design professionals need to work with ecologists, community development and public health experts and environmental engineers to reimagine the road verge in order to maximise the many benefits that it can provide.

It should be noted that verge gardening produces disbenefits as well as benefits. For instance, it can cause obstructions to movement, impair sightlines and introduce invasive species. These disbenefits are likely to be less than the benefits of verge gardening and can be managed to minimise their effects.

Verge gardening also promoted verge gardening. Contagion between neighbours, most likely to be social in nature, increased the likelihood of understorey verge gardening by 9%. This positive feedback loop means that this common practice might be able to become a widespread practice given the right conditions.

The main constraint preventing greater uptake of verge gardening was residents' perception that local governments thought that they should not verge garden. Verge gardening occurred in a context where local governments were generally opposed to the practice, with published guidelines to verge gardening emphasising negative aspects (e.g. punitive consequences) while ignoring benefits (e.g. provision of habitat, improved mental well-being). Encouragement by municipal authorities could facilitate substantial beneficial change to occur through the many small gardening actions of residents. Moreover, by becoming more common, verge gardening would, *ipso facto*, be seen as more normative – less ‘against the rules’ – further freeing residents from their perceived constraints. Sense of community was also positively associated with verge gardening, emphasising the fact that there are social benefits as well as ecological benefits to be gained by encouraging verge gardening. Public space was being managed by private citizens acting to further green the streetscape, often out of community-mindedness, or to act sustainably.

The lawns (or mown amenity grass) of road verges were dominated by rhizomatous turfgrasses but were far from being monocultures, with 150 species and seven distinct floral communities identified. Mowing frequency, verge gardening, and environmental factors such as rainfall and shading were significant factors determining species richness, abundance and community composition, but much variation remained unexplained, suggesting other factors, such as frequency and severity of disturbance, and previous land use, may be important. Road verge vegetation has the capacity to be a laboratory for research into sustainable lawns that are able to thrive without irrigation or fertilisation and in an environment tougher than a typical domestic or park lawn.

This thesis has provided a range of findings that can assist in the development of urban greening strategies that include the road verge as a significant type of urban greenspace. Its findings have implications at a range of scales, which are summarised in Table 7.1. Residents’ verge gardening may be able to become a vehicle for significant change in the urban environment. Verge gardening is already quite common, and this thesis has shed light on how it may be able to become more widespread. By understanding the road verge in greater detail, we can begin to reimagine our streets and improve the management the road verge to increase the biodiversity, ecosystem function and human amenity benefits that this unique urban greenspace has so much potential for delivering.

Table 7.1: Recommendations to improve the biodiversity, ecosystem function and human amenity benefits of the road verge across scales.

City

- Recognise the road verge as forming a city-wide park comprising many small patches
- Include minimum verge areas in planning-level street design requirements
- Include road verge vegetation in considerations of environmental justice
- Develop planning guidelines for new developments that minimise footpaths in low traffic-volume streets and embrace new green street forms

Local government

- Revise verge gardening guidelines to encourage verge gardening
- Adopt low-mow strategies in local government-maintained verges
- Increase plantings in local government-managed road verges
- Link street greening strategies to annual road maintenance calendar
- Work with broad range of professionals to develop greener streets strategies
- Establish understorey planting programs for all tree cut-outs
- Enlarge tree cut-outs where appropriate

Neighbourhood

- Recognise the residential estate as a scale at which significant urban characteristics are set: public planting palette, distribution of parks, road pattern, hierarchy and specifications, lot size and shape, location of utilities

Street

- Retrofit streets where possible, increasing verge area and capacity for plantings. Target low-traffic volume streets such as cul-de-sacs. Include street greening strategies in annual road maintenance programs

Property

- Recognise that verge gardening adds native species, and improves understorey vegetation’s extent and structure
- Support verge planting through programs similar to Backyard Biodiversity programs

References

Kendal, D., Williams, N. S. G. and Williams, K. J. H. (2012) 'Drivers of diversity and tree cover in gardens, parks and streetscapes in an Australian city', *Urban Forestry & Urban Greening*, 11(3), pp. 257–265. doi: 10.1016/j.ufug.2012.03.005.

O'Sullivan, O. S. *et al.* (2017) 'Optimising UK urban road verge contributions to biodiversity and ecosystem services with cost-effective management', *Journal of Environmental Management*, 191, pp. 162–171. doi: 10.1016/j.jenvman.2016.12.062.

VEAC (2011) *Metropolitan Melbourne Investigation*, *Assessment*. Melbourne: VEAC.

Erratum

On page 78, reference is made to Kirkpatrick *et al* 2011. This is in error, and should be replaced by Kendal, Williams and Williams, 2012.

Appendix A: Road easement greenspace – a visual overview

The following images are not intended to be definitive or exhaustive, but rather to provide some broad indicative context regarding the range of greenspace within the road easement, and its varying qualities, in Melbourne, Australia.

Urban form

Figure A1: Urban form. A) With no footpath and no front fence, the lawn of the front yard extends unbroken to the kerb. B) A resident has claimed all the land to the kerb, the letterbox an outpost of their domain. C) A very wide verge possibly due to land being set aside for future road duplication. D) A vast area of road verge set aside above a major water pipeline. E) An island of vegetation in a Cul-de-sac turnaround. F) A bike-only road closure provides an opportunity for plantings.

Urban form

Figure A2: Urban form. A) The wide verges of a 'bush boulevard', as this road type is called by the local government, allow an outer suburb to maintain its 'bush' character. B) Long property edges can create large areas of verge that are a burden for residents to maintain. C) A substantial planting rises up the batter to the sound walls of a major road edge. D) A road closure transforms the verge into a local park. E) This estate was designed with very large medians to create a sense of space and greenery. F) The land in the foreground is public land, but without a strong marker of the residential property line its ownership appears ambiguous.

Urban form

Figure A3: Urban form. A) An unusual combination of laneway and street separated by road verge. B) An outstand calms traffic and provides a location for street greening in a street with little vegetation. C) It can difficult to determine which resident or government authority is responsible for the maintenance of various parts of the landscape. D) A well-planted outstand divides vehicles from bikes at an intersection. E) The long fenced edges of suburban developments make large areas of underutilised verge. F) Another example of verge associated with the long, fenced edge of a development.

Urban form

Figure A4: Urban form. A) This dead-end road is designed like an English “mews”, with a wide median providing ample green space. B) A footpath moved away from the property line creates a space in which residents are encouraged to garden. C) As part of a road entry treatment to a major suburban shopping area, substantial swales in the road verge contribute to water management. D) Large scale water management As part of a new gateway to Melbourne’s airport.

Tree cut-outs

Figure A5: Tree cut-outs. A) Tree cut-outs can support large plantings. B) All the tree cut-outs in this street have been planted by a single resident with the same clumping succulent. C) A mirror bush, *Coprosa repens*, probably self-seeded and now well-tended, grows in a tree cut-out. D) Some tree cut-outs have no room for any additional plantings. E) Compacted clay-based fine gravels are used by local government authorities as a way of preventing weed growth in tree cut-outs.

Plantings

Figure A6: plantings. A) A rare blast of colour on a major road. B) A groundcover planting along the long length of a residential block. C) A bush-style native planting leaves room for access to parked cars. D) native groundcovers provide tough and low-maintenance cover. E) A median supports a dense and colourful display of Agapanthus. F) Dense, indigenous plantings are used by the developer of this estate to create local identity.

Plantings

Figure A7: Plantings. A) Resident-planted street trees provide a house with distinctive presence. B) Daisies are a popular and colourful species for verge planting. C) Many succulents flower well. D) Even small spaces can support generous amounts of understorey. E) It's not unusual for a resident to make a path from the front door to the kerb when there is no footpath present. F) Weed mat controls and tidies.

Plantings

Figure A8: Plantings. A) A road verge with spontaneous plants living a full life-cycle. B) The road verge is here used to provide a buffer planting of indigenous species beside a patch of remnant grassland. C) This road verge has probably never been sown with turf grasses and presents a different character to a typical lawn. D) A close-up of a typical lawn reveals a number of small flowering herbs. E) Tree roots add texture to the ground plane. F) Car tyres recycled into planters put a sustainability message into the public domain.

Plantings

Figure A9: Plantings. A) A planted wheelbarrow adds colour outside a cafe and is easily moved indoors when the cafe is closed. B) A dense understorey planting and resident-planted street tree opposite a park and creek with significant environmental values. C) Adding life to the street through topiary. D) Carefully maintained ivy on a cautionary sign. E) It is not unusual for residents to replace lawn with gravels.

Waste management

Figure A10: Waste management. A) The road verge is the location of bins for weekly rubbish removal services. B) During the annual 'hard rubbish' collection, materials stored on the verge can have considerable impact on plantings and soils.

Parking

Figure A11: Parking. A) Parking on the verge causes erosion and compaction of soils. B) A carefully placed rock, painted white, to stop cars damaging the corner of an immaculate lawn. C) Rocks placed to stop truck parking and protect street trees in an industrial zone.

Food production

Figure A12: Food production. A) A resident’s planter boxes introduce green into a streetscape with only tree cut-outs and small front yards. B) A bed of yams ready for harvest. C) Road verge fruit trees netted to protect the crop.

Community

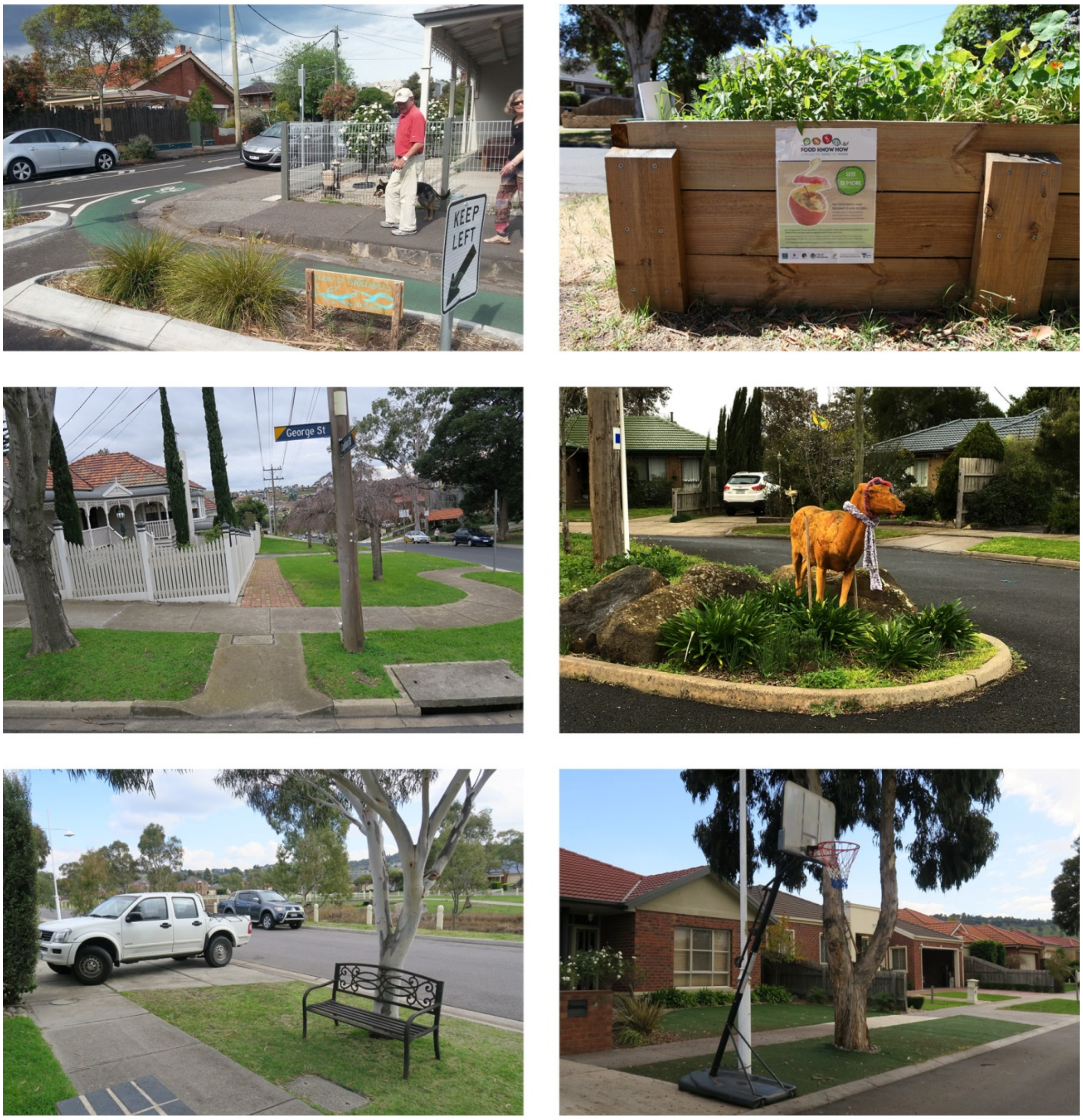

Figure A13: Community. A) A provocative sign "Glew St Community Gardens" in a small traffic island, accompanied by some resident-planted native plants. B) This planter is signed 'I heart Rossmoyne' and is part of a community action to create community in a suburban street (Rossmoyne St). C) A Brick footpath has been constructed by a resident to provide a low-maintenance connection where none had been previously provided. D) Humorous street art In the turnaround of a cul-de-sac. E) Bench seat installed in the road verge (and chained up against theft). F) Basketball hoop and artificial turf.

Community

Figure A14: Community. A) A median strip, heavily planted by a single resident. B) Pretend grave, R.I.P. "Lucky", in the road verge, Halloween humour. C) Owners who do not clean up after their dog often provoke residents' anger.

Maintenance

Figure A15: Maintenance. A) Most verges are mown. B) Freshly laid turf, a less common approach to creating lawn than using seed mix. C) Some residents use herbicide to prevent road verge vegetation growing out onto the footpath. D) Here a resident is laboriously using a trowel to keep edges away from the footpath.

Water sensitive urban design (WSUD)

Figure A16: WSUD. A) Street trees located in rain gardens that are planted with sedges for uptake of heavy metals from the road surface. B) A road verge swale for infiltration and detention of water, with ornamental rocks and plantings.