A Multi-Dimensional Analysis of Urban Agriculture

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Thesis abstract

Immediate action is required to address food insecurity, climate change, and sustainable urban development. Growing food in the city, a practice known as urban agriculture, is commonly presented as a solution to these interconnected challenges. Sensitively applied, urban agriculture can contribute to 12 of the 17 United Nations Sustainable Development Goals (SDGs), which provide a framework to guide action towards sustainable and equitable growth. I argue this point in a scoping review, which examines urban agriculture's contribution towards the SDGs across two themes: People and Planet. I bring to light synergies between Goals and explore potentially perverse outcomes. In doing so, I outline the multifarious realised and prospective benefits of urban agriculture. Although urban agriculture is widely acknowledged as advantageous, there remains a lack of critical analysis that quantifies the plausibility, practicability, and probable outcomes of its widespread implementation. Specifically, low-tech urban farming conducted on residential properties has been largely overlooked and inadequately researched. Using a case study of Adelaide, Australia, to explore these themes, this thesis contains four original research chapters.

First, a land use classification that combines high resolution hyperspectral imagery with LiDAR data, is used to model the potential for food production on residential properties. The results illustrate that self-sufficiency through backyard vegetable production is a plausible reality. Using yields that reflect an achievable best-case scenario for home gardeners, just 23% of lawn area in a typical residential block would be required to meet the recommended vegetable intake of residents. Building on from the previous work,

the second paper expands the land use classification to half a million homes and includes rainwater harvesting in self-sufficiency estimates. Results indicate that 65% of residential properties contain enough available land to provide dietary self-sufficiency of vegetables, while capturing and storing adequate rainwater for irrigation, even in the modelled dry year scenario. The third research chapter investigates the correlation between self-sufficiency potential and financial stress indicators that lead to food insecurity to examine the opportunities and limitations of specific suburbs within Adelaide. The final paper examines the environmental impacts of residential vegetable production, using a life cycle approach to establish the global warming potential (GWP) of backyard vegetable farming and residential lawn. The model incorporates DeNitrification DeComposition (DNDC) modelling software to establish soil trace gas emission, as well as the embodied carbon equivalents of inputs. Importantly, this work uses observational data on weather, irrigation, yields, and crop type to simulate the reality of a typical home gardener. The study shows that urban agriculture can reduce greenhouse gas emissions associated with household vegetable consumption.

This thesis provides valuable new insights into the potential for food production to occur within residential properties and contribute to multiple dimensions of sustainable development. In addition to quantitative modelling, issues that intersect with social sustainability permeate throughout the works. Similarly, considerations for urban planners and policy makers are presented. Together, this work makes a compelling case, highlighting the potential contributions of urban agriculture towards sustainable development.

Thesis Declaration

I certify that this work contains no material which has been accepted for the award of any other degree or diploma in my name in any university or other tertiary institution and, to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference has been made in the text.

I acknowledge that similar research was included in my Honours thesis submitted to the University of Adelaide in 2019. In the Honours thesis, I developed a model to classify and quantify residential lawn in Adelaide using remotely sensed data. Additionally, I conducted basic modelling of productivity and water usage for vegetable crops. Despite the similarities to my honours work, there are significant differences. Every model parameter has been recalculated, the study site has changed, and I have significantly advanced the sophistication of the models themselves. I emphasise that all the work presented in the thesis was conducted during my PhD candidature. The honours work served as a rudimentary proof of concept and method development for certain ideas.

In addition, I certify that no part of this work will, in the future, be used in a submission in my name for any other degree or diploma in any university or other tertiary institution without the prior approval of the University of Adelaide and where applicable, any partner institution responsible for the joint award of this degree.

I acknowledge the support I have received for my research through the provision of an Australian Government Research Training Program Scholarship.

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Introduction

Outline

This thesis is comprised of a review article (found within Chapter 1), and four original research articles (Chapters 2-5). At the commencement of this project a 'classical' literature review, leading to a set of discrete aims, was drafted. As this project evolved, the aims and objectives of the project were refined and modified, and so the original literature review became increasingly less relevant to the work presented herein. Nevertheless, there were aspects of the original literature review that remained relevant, and were used to help develop a scoping review, which outlines urban agriculture's role in advancing the United Nations Sustainable Development Goals (SDGs). Furthermore, during my candidature I have written four research articles (two published, two under review) arising from the work I undertook, each of which contains text from my original literature review. Therefore, rather than presenting the original literature review at the start of this thesis, as is the norm, I have opted to start this thesis with this review paper titled *The Case for Urban Agriculture*. This review provides an overview of the topic of urban agriculture at a global scale. The paper was written in collaboration with my colleague, with whom I share joint first authorship, and outlines the significant importance of urban agriculture, and its relevance in addressing pressing interdisciplinary issues.

The scope of the review article *The Case for Urban Agriculture* is much broader than the scope of the remaining chapters, which form the main body of the thesis. Therefore, following on from the global review article, further introductory text will provide context for the subsequent sections. These later chapters have been published (Chapters 2 and 3), or have been submitted for publication (Chapters 4 and 5). These four closely related

chapters are a deep exploration of the feasibility and practicability of low-tech residential urban agriculture via a case study of Adelaide, Australia. These chapters, which build on one another in a modular fashion, explore resource availability and environmental sustainability. Critical quantitative analysis, and a mixed method modelling approach are used. As noted above, each research article contains its own literature review, which will not be repeated in this introduction, to avoid redundancy. Finally, the thesis concludes with a discussion in which I synthesise emerging themes, highlight the contribution to scientific body of knowledge, and offer suggestions for future research.

Despite the above, I present my original un-amended literature review, which was prepared in July 2020, as an Appendix (A) to this thesis for the sake of completeness (rather than information).

The case for Urban Agriculture: Opportunities for sustainable development

The following is an unabridged review article prepared in the style of a review article for publication in Nature Sustainability. The reference list is at the end of Chapter 1.

Statement of Authorship

Title of Paper	The case for urban agriculture: Opportunities for sustainable development
Publication Status	☐ Published ☐ Accepted for Publication ☐ Submitted for Publication ☑ Unpublished and Unsubmitted w ork w ritten in manuscript style
Publication Details	A Presubmission Inquiry entitled "How urban agriculture can help achieve the Sustainable Development Goals" has been submitted to Nature Sustainability

Principal Author

Name of Principal Author (Candidate)	Isobel Hume		
Contribution to the Paper	Literature review Interpretation Writing - original draft		
Overall percentage (%)	40%		
Certification:	This paper reports on original research I conducted during the period of my Higher Degree by Research candidature and is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in this thesis. I am the primary author of this paper.		
Signature		Date	22/05/2023

Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

i. the candidate's stated contribution to the publication is accurate (as detailed above);

ii. permission is granted for the candidate in include the publication in the thesis; and

iii. the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

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Abstract

The United Nations Sustainable Development Goals (SDGs) are 17 interconnected goals designed to guide action towards a sustainable and equitable future. These goals represent a call to action for all countries and highlight the most pressing challenges and opportunities of today. Among these challenges, urban agriculture has been recognised as a potential solution with multifarious benefits.

This scoping review examines the role of urban agriculture in addressing the SDGs Specifically, we examine relevant overarching goals and targets as they relate to two general themes: *People* and *Planet*. This review uses three specific goals: SDG 2 Zero Hunger, SDG 11 Sustainable Cities and Communities, and SDG 12 Responsible Consumption and Production, synthesising topics that span the multiple SDGs. We bring to light potential synergies between targets of the Goals and make the case that urban agriculture can contribute to 12 of the 17 Goals. Simultaneously, we discuss potential perverse outcomes. While the review does not cover all aspects of urban agriculture in relation to the SDGs, it demonstrates that the realised and potential contributions of urban agriculture to urban systems are significant.

Overall, the sensitive application of urban agriculture can help achieve sustainable development as defined by the SDGs. Encouraging more research to support safe and sustainable urban food production can further these positive impacts on both *People* and the *Planet*.

Introduction

We live in challenging, and rapidly changing times. Food production will need to double by 2050 if we are to achieve global food security (Searchinger et al., 2019). The climate is changing, biodiversity is being lost at an unprecedented rate (IPCC, 2022; Ritchie et al., 2022), and much of the world's arable land is degraded or under threat (IPCC, 2019). There is also tremendous inequity among the world's population, both within and among countries and regions (UNEP, 2020; World Bank, 2022). Moreover, rampant urbanisation has dramatically shifted the human population, with more than half the world's population living in city or urban areas; a number projected to reach 68% by 2050 (UN DESA, 2019). The recent COVID-19 pandemic has highlighted the fragility, but also the resistance and resilience, of the global community and systems. It is issues such as these, that lie at the heart of the United Nations (UN) Sustainable Development Goals (SDGs), designed to achieve a sustainable and equitable future – both an urgent call for action, and a shared blueprint for peace and prosperity for *People* and the *Planet* (UN, 2015). Here we explore the role of urban agriculture – the practice of farming within and around cities and densely populated areas (De Bon et al., 2010; Gómez-Villarino et al., 2021; Hume et al., 2021; Orsini et al., 2013; Salomon et al.,2020; van Veenhuizen & Danso, 2007) – in addressing the SDGs.

Under the current projections, by 2050 global food production will need to increase by more than 50 percent to feed nearly 10 billion people – using fewer inputs on less land (Searchinger et al., 2019). With most of this growth in human population projected to

occur in cities, we contend that urban agriculture has a central role to play in achieving the SDGs. Urban food production can occur in, around, under, and on top of urban infrastructure; on private and public land; for consumption, distribution, and sale. In the Global South, urban agriculture typically occurs on undeveloped land within the city boundary whereas in the Global North it is more likely to occur in post-industrial landscapes (Azunre et al., 2019). In addition to providing income and nutrition, urban agriculture reconnects people with their food supply, builds resilient communities, improves community engagement, and has positive environmental outcomes, including mitigating against the effects of climate change (Kingsley et al., 2021).

Although broad interlinks and interdependencies between each of the 17 SDGs, including urban (and peri-urban) agriculture, have been made (Kroll et al., 2019; Nicholls et al., 2020a), it is our contention that the strongest and deepest connections between urban agriculture are with three specific goals: SDG 2 (Zero Hunger), SDG 11 (Sustainable Cities and Communities), and SDG 12 (Responsible Consumption and Production) (Figure 1). For this reason, we primarily focus on the role of urban agriculture in addressing these particular SDGs, through the themes of *People* and *Planet*. Further, the potential for urban agriculture to address targets within these and other SDGs, are also captured herein. In this high-level scoping review, the term *urban agriculture* may be used in reference to any food production occurring within urban spaces. Weidner, Yang, and Hamm (2019) have highlighted the risks of attributing the benefits of one urban agricultural practice to the whole spectrum of activities. However, we intentionally keep our definition broad, and stress that no singular farming practice will be a panacea for all urban issues. Instead, we aim to examine the mosaic of urban farming practices and

their contributions towards SDGs. Taxonomic classifications of urban agricultural practices exist elsewhere in the literature (Goldstein et al., 2016).

Figure 1 A demonstration of the relationship between urban agriculture and the UN Sustainable Development Goals. This review focuses on SDG 2 (Zero Hunger), SDG 11 (Sustainable Cities and Communities) and SDG 12 (Responsible Consumption and Production). See main text for further explanation.

SDG2: Zero Hunger / End hunger, achieve food security and improved nutrition and promote sustainable agriculture

An estimated 700 million people globally went hungry in 2020 (Sachs et al., 2021). Whilst the majority of these people were in impoverished urban communities in the Global South, the risk of food insecurity is also elevated in the Global North, especially during times of stress (Azunre et al., 2019). Given the core focus of agriculture is being the production of food (and associated products including fibre and fuel), and knowing that a growing, urbanised population will continue to require access to self-sustaining food, the relationship between SDG2 (Zero Hunger) and urban agriculture is palpable (Figure 2).

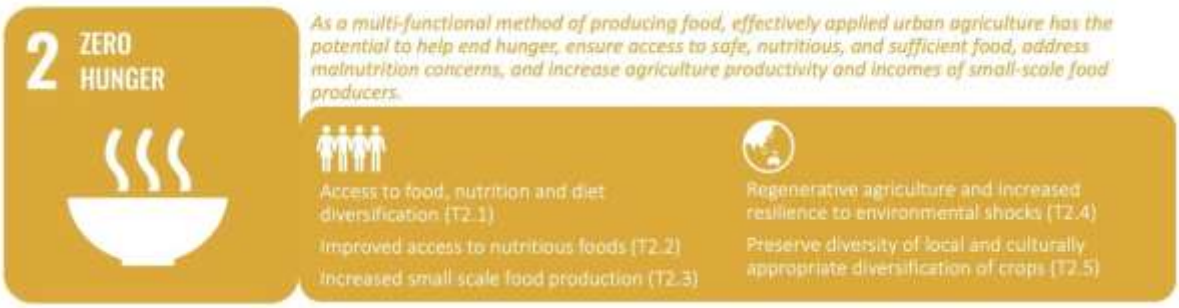

Figure 2 Urban agriculture applied effectively has the potential to help achieve a multitude of Targets (T) within SDG 2 (Zero Hunger). See Appendix B to read the full-text Targets.

People

Food security is having secure access to enough safe and nutritious food to meet growth and development needs and dietary preferences for an active and healthy life (FAO, 1996; IPCC, 2014). This complex concept encompasses food availability, access, and utilisation (FAO, 2006; Lerner & Eakin, 2011). Food insecurity exists on a spectrum

ranging from mild to severe, with uncertainty regarding the ability to obtain food at one end of the spectrum, and running out of food at the other (I. FAO,2020). As a mechanism for attaining food security, urban agriculture counteracts global dependencies and supply chain vulnerabilities by providing localised control of, and access to, food production capabilities (Langemeyer et al., 2021; Zezza & Tasciotti, 2010).

The urban poor spend a large majority of their income on food, and as such are particularly impacted by price shocks in food systems (Zezza & Tasciotti, 2010; Barthel et al., 2010; Langemeyer et al., 2021). The production of food close to, and by, consumers can help to build greater urban resilience (Loker & Francis, 2020). This is especially true in times of stress (Mougeot, 2000) (e.g., wars and the recent COVID-19 pandemic), where local and global food supply chains are disrupted (Rattan Lal, 2020; Loker & Francis, 2020; Davis, Downs, & Gephart, 2021; Yan, Liu, Liu, & Zhang, 2022). When confronted with global food supply issues, growing food locally is a time-tested strategy as a remedy to reduce hunger, undernourishment and malnutrition (Galhena et al., 2013). For example, one in three food insecure Australians during 2020 had never previously experienced food insecurity prior to the COVID-19 pandemic (Foodbank Australia, 2021), and there was a significant increase in urban farming activity during this time, with urban farming making a substantial difference to lower income communities (Donati & Rose, 2020) (Targets 2.1, 2.2).

The association between urban agriculture and improved diets is important in achieving food security and maintaining health. Insufficient nutrition is particularly detrimental to

vulnerable members of society such as pregnant women, babies and children, and other at-risk groups. Fluctuations in food prices at the local level, often playing out on a global scale, pose a significant risk to health, as nutrient dense foods are sacrificed for cheaper calorie dense food, particularly for the urban poor (Orsini et al., 2013; Zezza & Tasciotti, 2010). Engaging in urban agriculture not only provides more nutrient dense food (i.e., fruits and vegetable) it can also contribute to dietary diversification (Targets 2.1, 2.2).

In the Global North, poor diet is the leading risk-factor associated with death by non-communicable disease (FAO,2020). There has been a resurgence of food insecurity where poor, urban neighbourhoods have few fresh grocery stores, and more convenience stores and fast-food eateries serving less nutritious food (Besthorn, 2013). Urban agriculture has the potential to remedy food insecurity and reduce premature mortality from non-communicable diseases by improving household and community access to fresh produce (Opitz et al., 2016). In the Global South, engagement with urban agriculture has also been found to increase dietary diversity and calorie availability (Zezza & Tasciotti, 2010). (Target 2.1). Importantly, urban agriculture can improve nutritional literacy and drive changes towards more sustainable diets (Puigdueta et al., 2021). Taken together, the health and well-being benefits associated with urban agriculture contribute to positive outcomes around reducing hunger, undernourishment, malnutrition, and premature mortality (Targets 2.1, 2.2).

In addition to producing food for consumption, urban agriculture favours social inclusion and empowerment. Urban agriculture fosters economic inclusion through employment creation (Target 2.3), from the farmers through to stakeholders along the value chain,

ultimately becoming a major means of livelihood, particularly in the Global South (Azunre et al., 2019; World Bank Group, 2021). In contrast, the provision of ecological functions and support for social factors including recreation and leisure, have been identified as primary motivators of engagement in urban agriculture in the Global North (Azunre et al., 2019; Pearson, Pearson, & Pearson, 2010).

Urban agriculture has a role to play in safeguarding and enhancing the economic inclusion of women in diverse settings (Target 2.3). For example, while all genders are actively involved in urban agriculture, women in the Global South comprise a substantial portion of urban agriculture practitioners (Veenhuizen, 2006; Orsini et al., 2013). Urban agriculture, as a typically gendered activity, can potentially aggravate gender inequality by perpetuating the income gap between women and men and increasing the burden of work on women (Azunre et al., 2019). However, it can also provide opportunities for power redistribution within the family unit by providing women farmers with an independent income stream (Unicef, 2023). Although gender inequality is not restricted to urban agriculture, gender equality issues and factors need to be recognised and addressed, so that women can realise the full positive effects of urban agriculture.

Planet

Urban agriculture has a role to play in sustainably increasing food production. By utilising vacant or underutilised land within urban spaces (for example, (Hume et al., 2021)), urban agriculture can increase total food production without additional land clearing. In-ground urban gardens can produce higher yields that typical commercial farms, contributing to increased land use efficiency (Payen et al., 2022; Taylor, 2020;

McDougall et al., 2019). The productivity of small-scale urban farming is commonly attributed to dense cropping and the ability to tend to the garden by hand. However, it has also been theorised that literature is skewed towards highly controlled field experiments, with yields that may not reflect real-life scenarios (Payen et al., 2022). Gardener experience, education, and motivations all influence crop yields (CoDyre et al., 2015; McDougall et al., 2019; Pollard et al., 2018), and not all people growing food through urban agriculture are focused upon maximising their productive output (Donati & Rose, 2020).

High-tech urban agriculture such as indoor vertical farming, can operate with extreme resource efficiency, further increase yields per unit of land, and are thought to deliver food at competitive prices (Mir et al., 2022). Compared to in-ground urban farming, vertical farming frees land for other urban activities such as housing, services, and amenities (Al-Kodmany, 2018). It has been estimated that a single 30-storey building with a 2.02 ha (5 ac) footprint could produce a crop yield equivalent to 971.2 ha (2400 ac) (Despommier, 2011), while also allowing for year-round crop-production regardless of climate of weather. However, large-scale indoor farms require significant capital investment, and the high energy requirements of indoor farming can also result in increased global warming potential of crops ((Dorr et al., 2021); see also SDG 11 below). Nevertheless, indoor farms may have a role in shoring up food supply in times of acute stress. For example, immediately after the 2012 ‘Hurricane Sandy’, an indoor urban farm was the only fresh food supplier in New York City (USA) (Al-Kodmany, 2018; Velsey, 2012).

Healthy soils are the cornerstone of sustainable and productive farming systems. While traditional farming practices have led to widespread land degradation (Mbow et al., 2019), with careful management urban agriculture can maintain and enhance soil health (Salomon & Cavagnaro, 2022). Soil carbon (C), a key indicator of soil health, has been found to be between 3% and 10% total C in urban agricultural soils – approximately five times greater than in conventional agriculture (Dewaelheyns et al., 2013). Similarly, in a study of urban agricultural soils (allotment gardens in Great Britain (UK)), it was found that they contribute a disproportionate 0.05–0.14% of the country's total organic carbon but cover only 0.0006% of its total area (Dobson et al., 2021). High plant diversity, and the addition of composts in urban agriculture, both contribute to the increased C in urban soils (Salomon & Cavagnaro, 2022; Salomon et al., 2022a, 2022b).

Sub-optimal management by farmers and gardeners can degrade soil health. Irrigation schedules, fertiliser application, pesticide use, and mechanical disturbances are some of key factors that will affect soil quality (Salomon & Cavagnaro, 2022). In a survey of community (food) gardens in Adelaide, Australia, Salomon et al. (2020) found that 'overenthusiastic' application of phosphorous (P) by urban gardeners was causing nutritional imbalances. In the same study concentrations of (potentially toxic) heavy metals were higher in soils from community gardens situated on land with a prior history of industrial land use compared to those without. However, such risks can be mitigated through the utilisation of raised garden beds and root barriers to prevent contact with the underlying soil (Salomon & Cavagnaro, 2022). As with all farming systems, education and policy making are important drivers for sustainable production of safe and nutritious food.

Urban agriculture has gained attention for its ability to support biodiversity and ecosystem services for increasingly urbanised populations, particularly through its ability to safeguard food production and protect biodiversity (Philpott et al., 2020). In addition to increasing the amount and diversity of plants in urban spaces, urban gardeners themselves play an important role in the preservation of genetic diversity, local varieties and breeds. This is achieved through seed-selection and seed-saving practices, including the retention and sharing of traditional knowledge (Rural and Regional Affairs and Transport References Committee), thereby supporting Target 2.5. Moreover, the typically high diversity of plant and animal species found within urban food gardens encourage ecosystem services such as pollination and seed dispersal by bees and other pollinators, and pest control (Lin et al., 2015; Nicholls et al., 2020a). Conversely, replacement of native vegetation with introduced crops species could have adverse biodiversity and biosecurity outcomes (Matteson et al., 2008; Taylor & Lovell, 2014). However, it is likely that an increase in plant cover in typically denuded urban areas, be it facilitated by urban agriculture or not, will increase pollinator populations through enhanced habitat provision (Matteson et al., 2008). It is also worth noting that urban agriculture soils have been found to support complex and active microbial communities that provide valuable ecosystem services, and further contribute to urban biodiversity (Salomon et al., 2022a).

Conclusion

Agricultural activity and associated income are directly recognised by the UN as important in achieving the agreed aim of SDG 2 (Zero Hunger) (Target 2.3). Increasing

the share of self-sufficiency though local food production helps to protect the most vulnerable members of society, while addressing issues of food insecurity (Target 2.1). Approximately 80 percent of the extremely poor live in rural areas, with most relying upon agriculture for their livelihoods (World Bank Group, 2021). As these people move to the world's cities, their skills in farming may help facilitate uptake of urban agriculture, and their transition to a food secure urban life. However, providing the individual with the sovereignty to choose how their food is grown, where it comes from, and its suitability is critical in asserting localised rights to food. As our cities grow, preserving equitable access to land and resources will be key to achieving both food security and sovereignty. Thus, while not directly mentioned in SDG 2, the concept of food sovereignty (Clendenning et al., 2015; Campesina, 2021; Wittman, 2011) is pertinent. We contend that evidence-based and intentionally designed urban agriculture that considers the specific needs of *People* and *Planet* – including strengthening environmental integrity - can be an effective strategy to achieving SDG 2 (Zero Hunger).

SDG11: Sustainable Cities and Communities / Make cities and human settlements inclusive, safe, resilient and sustainable

SDG 11: Sustainable Cities and Communities responds to the advent of the 'urban millennium', where for the first time in history, the global urban population has outnumbered the rural population. Rampant urbanisation is projected to continue which will exert pressure on water supplies, the living environment, and public health (UN Habitat, 2022). As such, urban issues intersect many of the SDGs. However, this goal addresses sustainable development from a truly urban perspective. The sustainable

cities paradigm commonly considers the tripartite relationship of social, economic, and environmental dimensions of sustainability. Our climate, ecosystems, biodiversity, and human society are being increasingly recognised as interdependent, 'coupled systems' whose interactions form the basis of emerging and predicted risks, as well as future opportunities (IPCC, 2022). As urban populations grow and cities expand, it becomes more important to recognise and value the interconnecting 'webs of nature' (Beilin & Hunter, 2011). Urban agriculture sites are recognised as important community hubs that provide diverse individual and community benefits, beyond food production alone. As such, urban agriculture can advance developments towards SDG 11 (Sustainable Cities and Communities) (Figure 3).

Figure 3 Urban agriculture applied effectively has the potential to help achieve a multitude of Targets (T) within SDG 11 (Sustainable Cities and Communities). See Appendix B to read the full-text Targets.

People

Beyond its food security benefits, urban agriculture is an important component of green infrastructure that simultaneously supports biodiversity, ecosystem services, and health and well-being benefits for its users and consumers (Philpott et al., 2020). Further, the benefits to garden users, whether in public or private settings, are not limited to material aspects. For example, the aesthetic quality of urban agricultural spaces was

identified as providing a cultural service in a study focused upon Detroit (USA), where conversion of vacant lots from dumping grounds to urban food gardens replaced 'blight with places of beauty' (Newell et al., 2022). In turn, these sites became safer, and reduced the prevalence of rubbish and risky litter, thereby encouraging existing and new users to experience these multifunctional green spaces (Target 11.7).

Community gardens also offer a social-ecological refuge. To those escaping isolation, they provide opportunities to build social connectedness and trust, and facilitate participation, skills development and knowledge-building, within a community of practice (Barthel et al., 2014; Clarke et al., 2019). During the COVID-19 pandemic, the importance of urban gardens as inclusive sanctuaries was particularly realised. The non-material (sensory and emotional) aspects of both public and private urban gardens were identified as providing therapeutic benefits during this crisis, providing people the experience of feeling attuned and connected to the natural world (Marsh et al., 2021). These, and other co-benefits of improved physical and mental health, are possible through engagement with urban agriculture, by providing all ages and abilities with opportunities for exercise, stress-release, meaningful interactions with nature, social connectedness, and a sense of purpose and worth (e.g., Feola et al. (2020); Langemeyer et al. (2021); Philpott et al. (2020)). Ultimately, space assigned to publicly accessible urban agriculture contributes to the volume of open green space available to a community, which presents a multitude of benefits (Targets 11.3, 11.7, 11.a).

Planet

Urban agriculture provides both *in situ* and *ex situ* benefits which reduce the per-capita environmental impact of cities (Goldstein et al., 2016) (Target 11.6). *In situ* benefits include increased urban biodiversity, reduced stormwater run-off, and improved soil and air quality at the site of production. *Ex situ* benefits include carbon sequestration and reduced greenhouse gas emissions (i.e., reduced food miles), and reduced ecological footprint (i.e., encroachment on farmland). In recent years, Life Cycle Assessments (LCAs) of urban agriculture have become more prevalent, providing valuable insight into the environmental claims made about urban agriculture. A recent meta-analysis found that in 60% of published LCA comparisons, the global warming potential of *urban* agriculture (kg CO_2e kg produce $^{-1}$) was reportedly lower than that of conventional agriculture (Dorr et al., 2021). For example, in Santa Barbara County (USA), soil-based residential food gardens were found to reduce emissions by 2.10 kg of CO_2e per kg of vegetables when compared to commercial farms (Cleveland et al., 2017). The modelled gardens replaced urban lawn and used household greywater and composted organic waste to support agricultural production (Cleveland et al., 2017). The avoidance of environmentally harmful processes associated with waste management was a crucial feature of urban agriculture that contributed to its overall sustainability.

Simple urban production methods generally result in lower environmental impacts compared to both conventional agricultural systems and high-tech indoor growing methods (Dorr et al., 2021). For instance, in their meta-analysis, (Dorr et al., 2021) found that while open air, soil-based production of leafy greens consistently resulted in lower environmental impacts compared to conventional farming methods, indoor urban

production of leafy greens produced the opposite effect. Similarly, research that examined different crop types and production methods of urban agriculture in London (UK) found that while urban production reduced a crop's global warming potential by an average of 2.23 kg CO_2e per kg of fresh produce, growing strawberries locally in polytunnels produced 12% more greenhouse gas emissions than importing strawberries from Spain (Kulak et al., 2013). However, the sensitive application of technological solutions can deliver favourable outcomes. For example, in Barcelona (Spain), urban production of tomatoes in rooftop greenhouses was found to have reduced the carbon footprint by 33-62% (Sanyé-Mengual et al.,2015).

In addition to producing food, rooftop urban agriculture can mitigate climate change by lowering energy consumption in buildings. For example, in Madrid (Spain), the implementation of rooftop urban agriculture has been shown to save up to 25% of the energy required for cooling during summer (Saiz et al., 2006). Across various climates, highly vegetated green roofs have been shown to out-perform other cool-roof strategies, such as painting roofs white (Sailor et al., 2012). These findings support the case for urban agriculture as an innovative and effective approach to reduce urban energy consumption. However, not all roof tops have been designed to carry the additional load of a rooftop garden, and so appropriate planning, and risk assessment, is required.

One of the unique features of urban agriculture is that it enhances connections of the consumer, both spatially and conceptually, to the food they eat. Significantly, involvement in urban agriculture may influence citizens' food choices towards low

carbon diets. Participation in community gardens can increase pro-environmental identity, and progress into pro-environmental behaviours (Puigdueta et al., 2021; Tharrey et al., 2019). Similarly, a reduction in the consumption of animal-based products is one of the most impactful behavioural changes associated with urban agriculture, resulting in a significant reduction in greenhouse gas emissions (Clune et al., 2017). Meat and dairy are the primary drivers of global warming impacts in urban food systems (Goldstein et al., 2017). Therefore, small changes in food habits towards more vegetable-based diets have a substantial impact on food-related greenhouse gas emissions. This was observed by Puigdueta et al. (2021) in a longitudinal study of urban agriculture participants in Madrid (Spain), where the food related carbon footprint was 12% lower in urban gardeners than in the control, owing primarily to decreased meat consumption. The avoided greenhouse gas emissions equated to 205 kg CO_2 e yr^{-1} per capita. Puigdueta et al. (2021) concluded that urban gardens are effective tools for mobilising positive behavioural change, with promising implications for climate change mitigation policies in cities.

While urban vegetation may be strategically employed to improve air quality (Abhijith et al., 2017), the capture of pollutants in edible plants poses a human health risk (Ortolo, 2017). To minimise the risk to human health, edible crops should be grown at a reasonable distance from roads or behind barriers, and food grown near motor ways should be thoroughly washed to remove particulate pollution (Antisari et al., 2015). These risk factors highlight the need for careful planning and placement of urban agriculture sites in the larger urban context.

Conclusion

Mass migration to the cities continues to drive land consumption that ultimately results in cropland loss and threatens food security (Bren d'Amour et al., 2017; Salomon et al., 2022a). With thoughtful planning, if space for urban agriculture is preserved in new and existing cities, there is potential to help offset some loss of (rural) production while improving the cityscape in a safe and inclusive manner. Overall, the literature supports the case for urban agriculture to reduce the environmental impact of urban food consumption, thus reducing the per capita environmental impact of cities. The literature also highlights the importance of responsible practices in achieving environmental gains. Policy makers, community members, and other stakeholders must understand the nuanced complexities of urban food production and prioritise sustainable methods. Applied sensitively, urban agriculture can help to achieve SDG 11 (Sustainable Cities and Communities).

SDG12: Responsible Consumption and Production / Ensure sustainable consumption and production patterns

Goal 12 is to ensure sustainable consumption and production patterns. Changes in the way societies produce and consume are vital for achieving global sustainable development. While the goal refers to all material goods, there are specific indicators related to food, which inextricably tie urban agriculture to this goal. Furthermore, cities, and their associated agricultural regions, typically consume and dispose of nutrients in a linear fashion, which leads to global resource depletion and pollution (Cordell & White, 2011). Thus, the production of food in urban areas creates opportunities for circular

metabolism of water, nutrients, organic matter, which would otherwise be lost to waste streams. In this way, Urban agriculture can contribute towards Responsible Consumption and Production (Figure 4).

Figure 4 Urban agriculture applied effectively has the potential to help achieve a multitude of Targets (T) within SDG 12 (Responsible Consumption and Production). See Appendix B to read the full-text Targets.

People

Education is key to achieving sustainable development and lifestyles that work in harmony *with* nature, rather than against or without regard for it (Target 12.8). There is inherent learning in sites of urban agriculture, both as educational venues and through nature exposure (Matteson et al., 2008). For example, community gardeners in a London (UK) study revealed they became cognisant of the energy and resources embodied in purchased and homegrown foods, via their engagement with food production. In turn, this exposed them the true value of food, increased their food literacy, and encouraged pro-environmental food consumption (Kim, 2017). School gardens have also been associated with increased vegetable intake (Kafeero et al., 2020; Lohr et al., 2020), and improved attitude towards, recognition of, and a willingness to try new vegetables (Ratcliffe et al., 2009). Fischer et al. (2019) advocate for extending schools' critical environmental education beyond simple kitchen gardens to encouraging the concept of

"biodiverse edible schools" - linking environmental education, food production, and biodiversity conservation, to encourage students' understanding of healthy food and urban environments and strengthen the bonds between children and nature. In doing so, patterns of responsible consumption and production can be developed early on.

Planet

Environmentally sustainable production and consumption is about doing more with less. Urban agriculture is a powerful tool for sustainable production and consumption, particularly in relation to food waste (Target 12.3), nutrient and organic matter recovery from urban waste streams (Target 12.2), and the efficient use of key natural resources, namely water (Target 12.5).

Globally, one third of the food that is produced is lost or wasted, with *food waste* occurring at the retail or consumption stages, and *food loss* occurring between harvest and retail (Chaboud & Daviron, 2017; FAO, 2019). In total, around 14% of food produced is lost at various stages along the food supply chain, including: on the farm, post-harvest; during transport, storage, and distribution; and during processing and packaging (FAO, 2019). Bringing consumers and food producers closer together, eliminates many of the processes that result in food loss (Goldstein et al., 2016; Minten et al., 2016), thereby helping to retain food within the supply chain. Where food is scarce, combatting food loss through urban agriculture increases availability and accessibility of food (FAO, 2019), and can counteract vulnerabilities in the food supply chain (Langemeyer et al., 2021).

The issue of food waste is particularly acute in high-income countries, driven by excess buying, poor planning, and lack of knowledge on how to prepare leftovers (FAO, 2019). Food and green waste make up 44% of solid waste globally, and 53% and 57% of total solid waste in middle- and low-income countries respectively (Kaza et al., 2018). Community gardening and urban gardening can shape the internal motivations that lead to behavioural change around food consumption and food waste (Kim, 2017). Moreover, on-site reuse of waste prevents food from entering the waste stream (Zorpas et al., 2018; Zorpas & Lasaridi, 2013). Food scraps, yard waste, and paper can all be composted to recover nutrients and organic matter which can sustain food production (Vázquez & Soto, 2017; Wielemaker et al., 2019). However, only 5.5% of global waste is composted, leaving a significant gap between green waste generated and that which is recycled (Kaza et al., 2018). Composting at the community or household level is of particular value in locations without centralised green waste or kitchen waste collection bins. Decomposition of green waste in landfill causes methane production via anerobic decomposition of organic matter (Hellebrand, 1998), and home composting avoids indirect greenhouse gas emissions by eliminating collection and transport of green waste products (Lu et al., 2020). In Madrid (Spain), urban agriculture was found to increase the amount of food waste composted by over 40%, compared to the control group (non-urban gardeners) whose composting activities increased by a modest 4.5% (Puigdueta et al., 2021).

Urban agriculture also plays a crucial role in creating a sink, or destination, for recycled the products. This can be observed through two case studies conducted in Sydney (Australia) and Mumbai (India), which demonstrate the potential for urban agriculture to

consume waste and transform it into a valuable resource. In Sydney (Australia), McDougall et al. (2020) analysed the supply and demand of organic matter within a self-sufficiency model of urban food production. In all scenarios modelled, 100% of the un-recovered organic material in the local waste stream would be consumed in the establishment of urban agriculture. Agarwal et al., (2021) quantified the potential demand for composted organic matter in Mumbai (India), one of the world's mega cities. The case study of 1,177 multilevel apartments could cumulatively save 198 tonnes of wet waste from landfill each year by utilising household compost for rooftop farming. These findings highlight the capacity to consume and transform organic waste into a valuable resource, even when arable land is limited (Agarwal et al., 2021). However, nutrients coming from waste steams must be carefully managed to avoid imbalances and nutrient pollution (Salomon et al., 2020; Salomon & Cavagnaro 2022, Salomon et al., 2022; Shrestha et al., 2020).

Urban wastewater containing nutrients can be recycled through urban farming (de Graaff et al., 2010; Rufí-Salís et al., 2020; Wielemaker et al., 2019). Black water (i.e., containing human waste) contains large amounts of nitrogen and phosphorus (de Graaff et al., 2010). In a case study in Rotterdam (Netherlands), household waste (blackwater and kitchen waste) was shown to provide enough P, N and organic matter to fertilise all of the arable land within the city (including suitable rooftops) (Wielemaker et al., 2018). Similarly, in Barcelona (Spain), wastewater treatment could recover between 5 and 30 times the amount of P required to fertilize the city's associated agricultural area (Rufí-Salís et al., 2020). Urine is another N rich product readily available in urban settings. Urine fertilisers, collected from waterless urinals have been shown to replenish

micronutrients in the soil and caused less soil acidification than synthetic fertilisers (Chrispim et al., 2017), providing further synergies with a range of SDGs. In these examples, the presence of urban farms provides avenues for the use and recycling of urban wastes. However, the cost of high-tech solutions such as urine separating toilets can limit access for low-income cities (Azunre et al., 2019), and there are important safety considerations around the use of such waste streams in food production.

Water is vital in agricultural production but scarcely available in many parts of the world. Proximity to urban infrastructure allows urban agriculture to use harvested rainwater for irrigation. While urban agriculture is not the only urban activity to stand to benefit from roof-harvested rainwater, it's use in irrigation requires less treatment and infrastructure than other uses within the home. Moreover, access to water has been identified as a key challenge in the realisation of urban agriculture (Amos et al., 2018). In a modelling study of the self-sufficiency potential of urban agricultures in Sydney (Australia), it was found that not only could Sydney grow 15 % of its entire vegetable supply using vacant land, but harvested rainwater satisfied the irrigation requirements of all production scenarios modelled (McDougall et al., 2020). Similarly, Hume et al. (2022) found that a typical single-family household in in Adelaide (Australia) could capture and store enough water to sustain growing their recommended vegetable intake, even in a projected dry rainfall year. In Rome (Italy) up to one third of the existing 2,631 urban vegetable gardens could be irrigated using harvested roof-top rainwater from nearby buildings (Lupia et al., 2017). Even in areas of relatively high rainfall, where crops are not reliant on irrigation, integrating rainwater harvesting and storage systems were found to increase yields of urban agriculture two-fold compared

to rainfed production (Amos et al., 2021). And in arid regions, where water is scarce, rainwater harvesting can lower potable water use, save energy, and reduce greenhouse gas emissions (de Sá Silva et al., 2022), highlighting co-benefits with other environmentally related SDGs. Efficient water management strategies in urban, and indeed any production system, will be vital for attaining benefits relating to food security, economic inclusion, and health and wellbeing. Although, urban agriculture presents opportunities to increase the efficient use of natural resources, while providing co-benefits to other SDGs, this must be balanced against the risks (e.g., to human and soil health) of using waste streams as agricultural inputs (Buscaroli et al., 2021; Salomon & Cavagnaro, 2022).

Conclusion

In clear support of SDG 12 (Responsible Consumption and Production), urban agriculture has the potential to aid the adoption and maintenance of sustainable production and usage patterns with important implications for avoiding urban consumption that exacerbates climate change, biodiversity loss, and pollution. Localised production minimizes food losses that occur during the transportation, storage, and distribution of food, and can optimise resource use. Engagement with urban agriculture can provide critical cultural services of nature connection and appreciation, helping to awaken and foster environmental stewardship through lived experiences (Camps-Calvet et al., 2016; Langemeyer et al., 2018). Urban agriculture's cultivation of nature experiences, learning, and education can provide a foundation for understanding and supporting sustainable practice and objectives (Langemeyer et al., 2018), and overall aiding the attainment of SDG 12 (Responsible Consumption and Production).

Conclusion

There is a risk of overstating urban agriculture's benefits – it is not a panacea – however, urban agriculture clearly has a role to play in achieving the SDGs. We conclude that the sensitive application of urban agriculture can help with the attainment of the UN Sustainable Development Goals (SDGs). Given the breadth and depth of the SDGs this review is by no means exhaustive. Thus, while our emphasis here has been on SDG 2 (Zero Hunger), SDG 11 (Sustainable Cities and Communities), and SDG 12 (Responsible Consumption and Production), there are multifarious interactions among all 17 of the SDGs at all levels that need to be considered (Kroll et al., 2019). Urban agriculture has a clear role to play in addressing the SDGs in both the Global South and North. While urban agriculture in each region faces unique challenges, there are commonalities too. In undertaking this review it became evident that urban agriculture can have both beneficial and perverse outcomes, and so thoughtful planning and policy, education, and sensitive application is needed for urban agriculture to realise its full potential. For the full benefits of urban agriculture for both *People* and *Planet* to be achieved, we need a strong evidence base. Notably, much of the published literature that critically analyses the environmental impact of urban agriculture is concentrated in economically developed countries leaving a significant knowledge gap in the Global South.

Taken together, these examples highlight the complex trade-offs that need to be considered in order to provide a realistic interpretation of urban agriculture's proclaimed benefits.

The time has come to view urban agriculture through the lens of a serious farming system.

Background and context to succeeding research articles

It is undeniable that urban agriculture presents opportunities for sustainable development. Now, time has come for critical analysis of urban agriculture to evaluate resource availability, practicality, and realistic environmental outcomes. Critical to understanding the viability of wide-spread urban agriculture, is the intersection of available land and agricultural inputs in the urban environment. Understanding the scale at which urban agriculture can operate is also vital in establishing contributions to sustainability targets. For example, meaningful contributions to food security will only be made if the urban area is able to physically support food production. This is also true for other facets of sustainable development, including water regulation, and climate change. Despite the growing body of urban agriculture literature, there remain large knowledge gaps pertaining to low-tech farming carried out by individuals and households (Dorr et al., 2021; El Khateeb et al., 2023; Kirkpatrick & Davison, 2018).

Over the next four chapters (2-5), I present four original research articles, that critically examine resource availability, self-sufficiency potential and environmental sustainability of back yard farming using Adelaide (Australia), as a case study. The entire body of work examines low-tech vegetable production occurring on residential land, replacing what is currently lawn. Sensibly applied, urban agriculture can alleviate the effects of food insecurity, foster good health and wellbeing, and promote social and economic inclusion (Nicholls et al., 2020b) also see: The Case for Urban Agriculture (above)). As these outcomes will primarily benefit vulnerable groups, it is also important to consider how the resources, essential to engage in urban agriculture, are distributed throughout the

urban population. As was outlined on *Page 12,* relevant literature reviews are contained within each chapter of this thesis. Nonetheless, the subsequent paragraphs of the introduction provide relevant context and sufficient background information which lead to the project aims.

A striking observation that has emerged form researching urban agriculture is the broad diversity of practices that exist. As Weidner, Yang, and Hamm (2019) have noted, an issue in urban agriculture advocacy is that the benefits of one practice are often prescribed to the whole spectrum of urban farming activities. From the outside, this can look like a wilfully optimistic representation of urban agriculture, which can ultimately draw criticism towards the movement. In the Global North, which Australia belongs to, community-led urban agriculture is often studied for its contribution to social dimensions of urban sustainability (Azunre et al., 2019), whereas environmental assessments focus on highly controlled environments, or innovative high-tech solutions (Dorr et al., 2021). This leaves large knowledge gaps pertaining the productivity and environmental outcomes of low-tech home gardening, and other community-led practices.

There is sound academic rational for studying urban vegetable production. For example, producing vegetables is commonly regarded as the most efficient use of resources as they are highly perishable and more economically valuable than staple crops (Weidner et al., 2019). Vegetables are also an important component of dietary diversity, which is a cornerstone of food security and nutrition. This rational is outlined in more detail in Chapters 2-5. Furthermore, and perhaps most significantly, front and back-yard

vegetable production is the way most people in Australia engage in urban agriculture, with 97% of urban growers growing vegetable crops (Donati & Rose, 2020).

Adelaide, capital of the state of South Australia, is a relatively low-density city, which indicates good food producing capacities (Martellozzo et al., 2014), while it's mild, Mediterranean climate allows for year-round cropping of vegetables (Pollard et al., 2018). Like many places in the economically developed Global North, Adelaide is characterised by sprawling residential suburbs and private residential yards. Lawn, which is emblematic of many modern urban environments, exists in abundance in Adelaide's public and residential greenspace (Holt, 2021). Lawn is also arguably the urban land use most easily converted to in-ground vegetable production. While residential lawns have social, cultural, and aesthetic significance and are typically considered "natural" in the western world (Ignatieva et al., 2015), they can also be easily converted to vegetable garden beds with little capital investment, and no legislative barriers. Despite the potential for urban food production to occur on privately held land, such practices have been overlooked and understudied in the broader context of urban agricultural research (Kirkpatrick & Davison, 2018).

Remote Sensing as a tool

Integral to this project is the ability to classify urban land use at high spatial resolutions so that individual blocks can be parametrised for their self-sufficiency potential. Remote sensing of urban areas has applications in decision-making and planning, disaster response, and sustainable development, making it the subject of much research (Gong,

et al., 2019). Notably, remote sensing allows large expanses of landcover to be classified accurately and efficiently. Spectral reflectance can inform indices such as the Normalised Difference Vegetation Index (NDVI) (Curran & Steven, 1983), which delineates pixels containing photosynthetic material in urban (and indeed other) spaces. Supervised and unsupervised land use classifications, which classify pixels based on their spectral similarities or differences, can also be useful in classifying urban land use (Richards, 2013). However, spectrally similar yet functionally different cover types (e.g., lawn and trees, or bitumen and rooftops) present unique challenges for classifying land cover in urban spaces (Gong et al., 2019; Wetherley et al., 2017). Digital Surface Models, derived from Light Detection And Ranging (LiDAR) data, provides textural information describing variance in elevation amongst adjacent pixels (Mathews & Nghiem, 2021), and can be used alone, or in conjunction with spectral reflectance to inform urban land use classifications.

While remote sensing, sometimes referred to as Earth Observation (EO), is the practice of collecting information about the earth's surface, Geographic Information Systems (GIS), integrate descriptive information with location data. In urban land use and planning, GIS can incorporate remotely sensed data, and its derived land-use classifications, with other datasets (Kumar et al., 2023). To this end, GIS has proven a valuable tool in studying urban food production across various domains (e.g., Kazemi & Hosseinpour, 2022; McDougall et al., 2020; Saha & Eckelman, 2017; Wang et al., 2022).

The COVID of it all

This project commenced in February of 2020. While the research plan, structure, and aims were all developed before the impacts of the COVID-19 pandemic were understood, and certainly before academics could report on them, *the COVID of it all* has highlighted the value of urban agriculture (Kirkpatrick & Davison, 2018; Music et al., 2021). However, history tells us that urban agriculture becomes increasingly important during times of stress. For example, during WWII the US, UK, and AUS governments encouraged citizens to seed *Victory Gardens,* and grow their own food to support the war effort (Gowdy-Wygant, 2013; Music et al., 2021). The COVID-19 pandemic is not the sole reason that urban agriculture is worth studying, but it certainly reignited public engagement and academic interest (Donati & Rose, 2020; Lal, 2020; Mullins, Charlebois, Finch, & Music, 2021). There will be future shocks to our food systems, most likely from climate change, and local food production may help us to absorb these shocks.

Research Articles

The aims of each research paper, which build on from one another in a modular fashion, are as follows (Figure 5):

1. Paper 1 (Chapter 2): *Self- Sufficiency though urban agriculture: Nice Idea or Plausible Reality* (published research article). This paper aims to examine the spatial limitations on residential farming. i.e., how much potentially productive residential land there is, and how much food can be grown on that land. This work has been published as: Hume, IV, Summers, DM & Cavagnaro, TR 2021, 'Self-sufficiency through Urban Agriculture: Nice idea or

plausible reality?', *Sustainable Cities and Society*, vol. 68, 2021/02/09/, p. 102770.

2. Paper 2 (Chapter 3): *Lawn with a side salad: Rainwater Harvesting for Self-sufficiency though urban agriculture* (published research article). This research investigates the potential for rainwater harvesting to irrigate vegetables at a single property, considering potential barriers such as economic viability and land use changes over time. This work has been published as: Hume, IV, Summers, DM & Cavagnaro, TR 2022, 'Lawn with a side salad: Rainwater harvesting for self-sufficiency through urban agriculture', *Sustainable Cities and Society*, vol. 87, 2022/12/01/, p. 104249.
3. Paper 3 (Chapter 4): *Backyard farming: Who benefits and who misses out? A Geospatial Analysis* (submitted for publication). After establishing the self-sufficiency potential of homes in Adelaide, I examine how these resources are distributed throughout the population and explore potential policy implications.

   This work has been submitted for publication and is under review in the Journal of Sustainable Cities and Society.
4. Paper 4 (Chapter 5): *Is the grass always greener? Comparing the global warming potential of backyard farming and lawn* (submitted for publication). Here, using a mixed methods life cycle approach, I examine the global warming potential of the system which was modelled in earlier chapters. This work has been submitted for publication and is under review in the Journal of Cleaner Production.

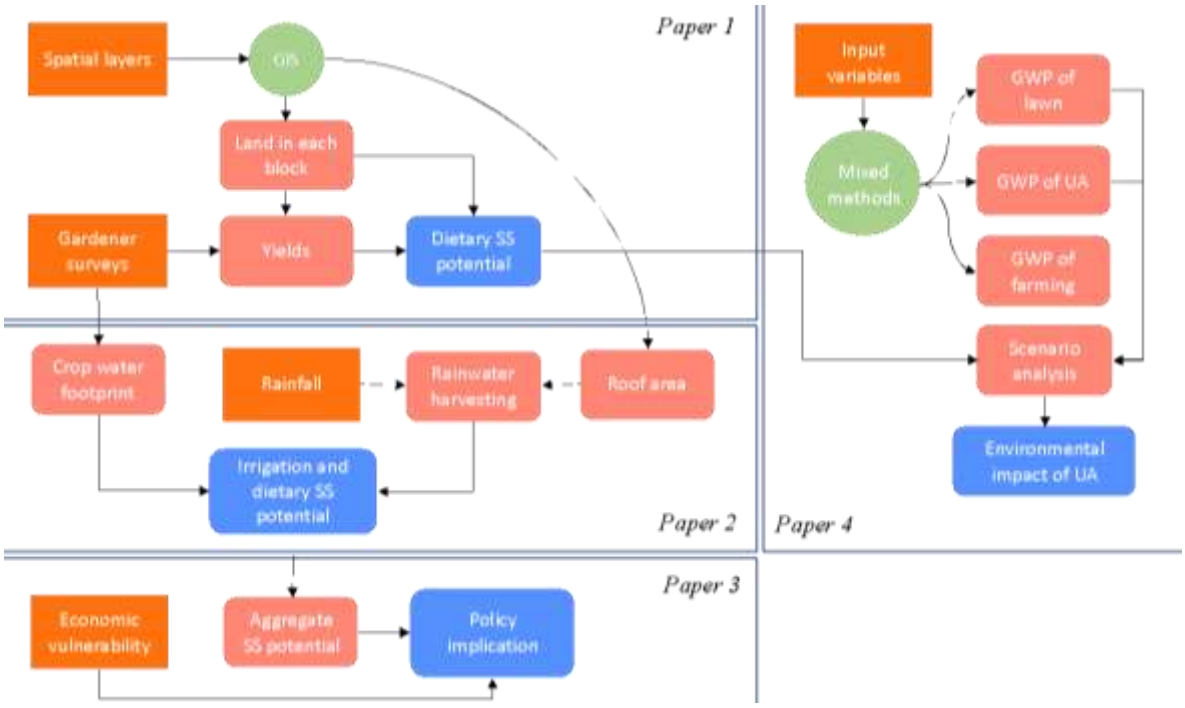

Figure 5 schematic diagram of the research chapters comprising this thesis. Orange blocks are input datasets, pink blocks are derived variables, green spheres are processes, and blue blocks represent outcomes. Figure is highly simplified, denoting

The fifth academic paper included in this thesis is the review article, *The Case for Urban Agriculture.* The manuscript has been prepared for submission, and pitched to the journal *Nature: Sustainability.* At the time of writing, the pre-submission inquiry is under review.

Additional works

In addition to the written academic works, the findings of this PhD have been shared with diverse audiences in formal and informal settings:

- I was invited to speak at the Urban Food Network's Stories of Local Food event (Adelaide, 2021).

- I presented my work at the Gawler Historic Centre, followed by a panel discussion on local food production in Gawler and Adelaide more broadly (Gawler, 2021).
- I presented at the University of Adelaide School of Agriculture Food and Wine Postgraduate Symposium, where I was awarded the Sally and Andrew Smith Award for most outstanding Presentation, as well as the People's Choice Award (Adelaide, 2021).
- The work was also distilled in a 1-minute video for the Visualise Your Thesis Competition, for which I was the University winner and represented the University of Adelaide at the international stage of the competition (Online, 2021) - *https://figshare.com/articles/presentation/Self-sufficiency_through_Urban_Agriculture/16583867*.
- I shared the findings of my research with COSMOS Magazine in a video interview for a piece on the role of urban agriculture in fighting climate change (Online, 2021).
- I entered the 3 Minute Thesis Competition, and was a University Finalist, taking home the People's Choice Award and School's Choice award (2022) - *https://lnkd.in/gPwe9wMA*.
- I spoke at Symposium for Graduate Student Work on Food, Agriculture, and the Environment, hosted by the Working Group on the History of Food, Agriculture, and Sustainability (Canada) (Online, 2022).
- I co-authored an article published in *The Conversation,* titled *Fennel looking a bit feeble? Growing enough veggies to feed yourself depends on these 3 things* (2022) (Appendix C).

- I was profiled by *Aerometrex* for the MetroMap Research Data Hub, which involved at long format video interview (Online, 2023) - *https://vimeo.com/812268424*.

I was also fortunate enough to attend four conferences and present at three:

- I attended the Water Sensitive Cities Conference (Adelaide, 2021).
- I travelled to Nottingham, UK for Total Food 2022 hosted by Nottingham University, where I presented a Poster (Appendix D) (2022).
- I presented a traditional talk at Spatial Information Day (Adelaide, 2022).
- Finally, I presented a technical seminar at the Advancing Earth Observation Forum in Brisbane, Australia (2022).

The findings of this research have been reported on by several media outlets including:

- The Advertiser (print and online, 2021),
- ABC Adelaide (live on-air interview, 2022), and
- Adelaide Radio (live on-air interview, 2022).

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Chapter 2
Self- Sufficiency through urban agriculture: Nice Idea or Plausible Reality

Chapter 2 contains the PDF of the published research article. The supplement to the work is found in Appendix E.

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Self-sufficiency through urban agriculture: Nice idea or plausible reality?☆

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ABSTRACT

With more than half of the world's population residing in cities, there has been intense interest in urban agriculture's contribution to sustainability and self-sufficiency. This paper provides new insights into the potential for food production to occur within residential properties. High-resolution optical imagery was employed to quantify residential land immediately available for vegetable production (i.e. residential lawn) in a low-density city. Adelaide (Australia) was used as a case study. Productivity was estimated according to three empirical (low, medium and high) and one commercial yield scenario. Under high and medium yield scenarios, lawn in a typical block exceeded the space required to achieve household self-sufficiency of vegetables. Under the high-yield scenario, 23 % of lawn area would be required to meet the recommended vegetable intake of residents, while under the medium yield scenario, 72 % would be required. This study demonstrates the plausible reality of self-sufficiency through backyard vegetable production. Other resources that might limit production, such as labour, water and nutrients, are explored in the discussion. Urban agriculture is shown to be particularly worthwhile in low density cities; contrasting previous research performed under different urban conditions. However, in this case study, potentially productive land has decreased over time.

1. Introduction

Engineering sustainable food systems for a growing population will be one of the greatest challenges of the coming decades (Bren d'Amour et al., 2017; Godfray et al., 2010). Urban agriculture, the practice of growing food within the urban boundary, creates the opportunity to safeguard access to healthy, fresh produce for urban dwellers. Historically, societies have leant on urban farming during times of stress: Governments employed urban agriculture to increase food security during each of the World Wars (Mok et al., 2014). In 2020, the COVID-19 pandemic accentuated existing issues of food insecurity, and once again highlighted the importance of local food production (Lal, 2020). With the majority of the global population now residing in cities (United Nations, 2018), and climate change threatening global productivity long after the pandemic subsides (Ray et al., 2019), it seems urban agriculture has never been more relevant. Yet there remain unanswered questions regarding urban agriculture's ability to provide meaningful amounts of food.

Urban agriculture can also be used as a tool to mitigate against further environmental degradation (Weidner, Yang, & Hamm, 2019). It's proximity to other urban processes creates opportunities to capture and reuse urban waste products as agricultural inputs. This can help *close the loop* on urban metabolism and contribute towards sustainability targets (Paiho et al., 2021; Wielemaker, Weijma, & Zeeman, 2018), while simultaneously reducing the environmental burden of food production. However, there are concerns that the academic discourse on urban agriculture tends to take an advocacy standpoint (Neilson & Rickards, 2017; Weidner et al., 2019) rather than critically analyse it's feasibility on a broad scale. While self-sufficiency would have economic and social interactions, this paper examines the spatial constraints on food production to understand the physical limitations of self-sufficiency.

'Urban agriculture' is an all-encompassing term, which may include allotments or rooftop gardens, building-integrated greenhouses and complex indoor 'plant factories' (Mok et al., 2014; Weidner et al., 2019). Despite some efforts, there is currently no universal taxonomy for classifying urban agricultural practices (Goldstein, Hauschild, Fernandez, and Birkved (2016)). As a result, benefits offered by a specific growing

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practice are often attributed to the whole spectrum, which can lead to exaggerated claims and ultimately draw criticism (Weidner et al., 2019). Here we use the term 'urban agriculture' to refer to soil-based, open-air production of fruits and vegetables occurring within urban residencies.

In suburbanised cities around the world, residential yards are a prominent feature of the urban landscape; this implies a significant potential for agriculture to occur within residential properties. Yet residential food gardens tend to be overlooked and understudied in the context of urban agriculture (Kirkpatrick & Davison, 2018; Taylor & Lovell, 2014). Focusing on nutrient-dense foods (i.e. vegetables), rather than calorie-dense foods (e.g. grains) is a more efficient use of the land available for urban agriculture, as vegetables are high value, perishable crops (Martellozzo et al., 2014; Safi, Dossa, & Buerkert, 2011; Weidner et al., 2019). A recent literature review by Weidner et al. (2019) concluded that partial self-sufficiency, focusing selectively on horticultural crops, will optimise the benefits of urban agriculture. The authors stated that 'nutrients and nutrient security should be the focus [of urban agriculture]' and that 'only partial self-sufficiency is required and desired' (Weidner et al., 2019 p. 1651). With these justifications, this paper takes a novel approach of land use classification via remote sensing to model residential land and investigate the potential for dietary self-sufficiency through urban agriculture. Adelaide (Australia) will be used as a case study.

Remote sensing allows large areas of land to be classified accurately and efficiently. Despite advancements of emerging technologies in high-resolution sensors, the fine spectral and spatial scale at which functionally different surfaces operate within urban environments make them the most challenging areas for remote sensing analysis (Gong, Liu, & Huang, 2019). Spectral classifications of urban vegetation can be particularly problematic due to the similar spectral properties of lawn and trees (Wetherley, Roberts, & McFadden, 2017). Textural data, such as LiDAR, is useful in distinguishing between spectrally similar, but structurally distinct surfaces, such as trees and lawn, and may be combined with spectral information to improve classification of urban vegetation (Anderson et al., 2018; Bartesaghi-Koc, Osmond, & Peters, 2019).

Remote sensing and GIS have been used to varying degrees to model self-sufficiency in the past. A global analysis of self-sufficiency potential revealed that, in Australia, just 10 % of the total urban extent would be required to meet the population's vegetable requirements (Martellozzo et al., 2014). As this study did not classify different land use within the urban boundary (e.g. parks, buildings, private gardens), the location and distribution of potentially productive land parcels were not identified. To establish the relationship between residential food gardens and self-sufficiency potential, a more sophisticated method of quantifying urban features (e.g., buildings, vegetation) and land ownership (public/private) is required.

McDougall, Rader, and Kristiansen (2020) used GIS to classify different land use in Sydney (Australia) and investigate the self-sufficiency potential through urban agriculture. Sydney's entire vegetable supply could be produced through urban agriculture if all street verges; miscellaneous blocks (e.g. vacant blocks); and 25 % of total yard space were employed (McDougall et al., 2020). However, this study did not investigate residential properties as a stand-alone resource, and 'yards' were considered to be the total space of the residential block minus the building footprint. As the spatial analysis did not discriminate between different ground level surfaces within properties, impervious surfaces, such as driveways, were not excluded from the analysis.

Richardson and Moskal (2016) successfully used spectral data to perform a remote sensing land use classification and identify residential lawn to be reclaimed for urban agriculture in Seattle (USA). While the approach taken was successful, the story of self-sufficiency was less encouraging; converting residential lawn into urban food gardens would feed less than 1% of Seattle's population on an entirely vegetarian diet, which included nuts and legumes (Richardson & Moskal, 2016). However, it has been established that partial self-sufficiency with a focus on nutrient-dense foods should be the focus of urban agricultural development (Weidner et al., 2019). Furthermore, Seattle has a city-wide population of 3429 people km^{-2} (Richardson and Moskal (2016), and is one of the most densely populated cities in the United States, with some neighbourhoods reaching 14,644 people km^{-2} (Statistical Atlas, 2018). Taken together, it is unclear what the value of residential yards is when it comes to self-sufficiency in lower density, sprawling cities.

In addition to identifying available land, self-sufficiency models must include some area-based yield estimate. Yet, identifying empirical trends in urban agriculture productivity is challenging. As urban food gardens are often multifunctional, yield estimates vary both within and between studies (Algert, Baameur, & Renvall, 2014; Gittleman, Jordan, & Brelsford, 2012; McDougall, Kristiansen, & Rader, 2019). Consequently, when building productivity models, some research has utilised commercial crop yield data in favour of highly variable yield estimates from urban agricultural sites (Clinton et al., 2018; Richardson & Moskal, 2013; Saha & Eckelman, 2017). Despite their variability, urban food gardens have been shown to produce considerably higher yields than their commercial counterparts (CoDyre, Fraser, & Landman, 2015; McDougall et al., 2019; Zaimuddin & Mercer, 2014). Therefore, commercial yields provide a robust yet conservative estimate of productivity.

Here, we present the results of a study in which we quantified the amount of residential lawn space available for vegetable production in a low-density city. A remote sensing classification was used to identify residential lawn, and different classification methods, employing both high-resolution optical imagery and LiDAR data, are compared. Finally, the notion of self-sufficiency was explored by modelling open-air, soil-based vegetable production occurring within residential properties.

1.1. Setting the scene: Adelaide as a case study

This investigation focuses on Adelaide, South Australia, as a case study to model self-sufficiency in a low-density city. Adelaide is a sprawling city. The greater region covers 3260 km^2 (1259 sq mi) and is home to 1.3 million people – approx. 400 people km^{-2}. However, the population is concentrated into smaller areas, with suburbs housing up to 3000 people km^{-2} (Australian Bureau of Statistics, 2020b). Apartments make up only 8% of residential dwellings; 75 % of dwellings are free-standing houses and a further 17 % are semi-detached or terrace houses (Australian Bureau of Statistics, 2020a).

Adelaide is situated in a productive agricultural landscape (Mok et al., 2014). The temperate Mediterranean climate results in a maximum average temperature of 22.5 °C, and a minimum of 12.3 °C, with an annual average rainfall of 547.1 mm (Bureau of Meteorology, 2020). Much of Adelaide's urban fringe is protected from development to preserve peri-urban agriculture (SA Planning Portal, 2019). However, there remains no legislation or policy surrounding food production within Adelaide's metropolitan areas. The spatial resolution at which land use in the greater Adelaide region is currently classified does not allow the recognition of food production within the metropolitan zone (Australian Bureau of Agricultural and Resource Economics and Sciences, 2017). However, both private and public urban agriculture do occur (Kitchen Garden Initiative, 2012; Pollard, Ward, & Roetman, 2018)

2. Methods

2.1. Location and datasets

2.1.1. Study sites

Three representative locations in the greater Adelaide region were used in this study, from here referred to as the Central, Northern and Southern extents (Fig. 1). The availability of high-resolution spectral

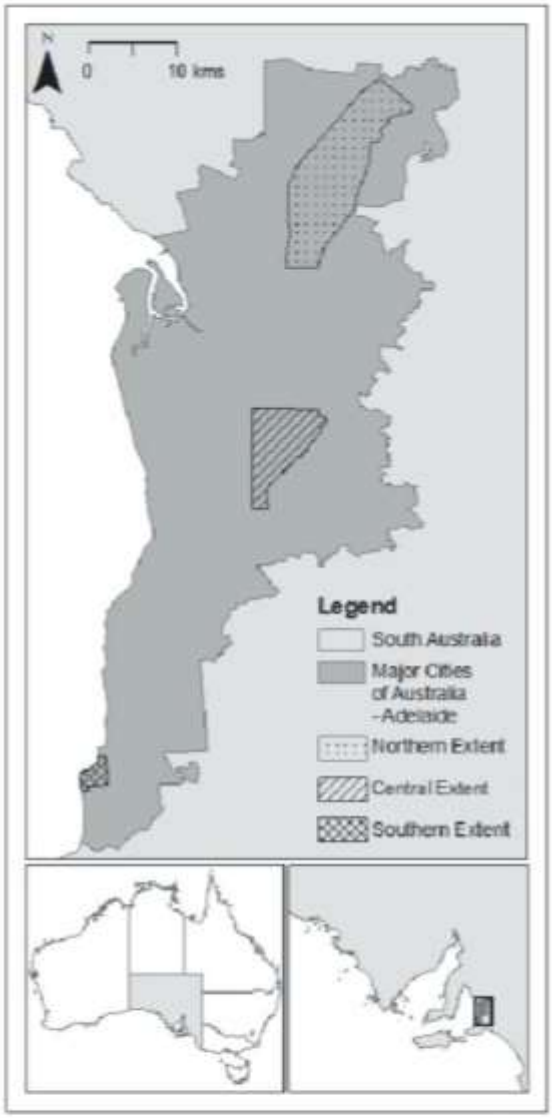

Fig. 1. The location of the three study sites (Northern, Central, and Southern extents) within Adelaide's boundary – as defined by the ABS (Remoteness Area classification: 'Major cities of Australia' (dark grey)). The state of South Australia, of which Adelaide is the capital, is shown in light grey.

data required to carry out this research (see Section 2.1.2.) determined the boundaries of each site. Not only do the three sites represent the geographical spread of Adelaide, but each has a unique settlement history. The Northern extent covers a large area extending from the council area of Elizabeth to Gawler. Gawler was South Australia's first regional settlement and was historically distinctly separate from Adelaide (Town of Gawler, 2017). The Southern extent is a smaller site over the Aldinga council area. Rather than a residential area, Aldinga was primarily a holiday destination in the early 20th century. However, due to population expansion through the second half of the 20th century (Robinson & Liu, 2015), Gawler now marks the northern boundary of the greater Adelaide region, and Aldinga nears the south. In contrast, the Central extent covers the eastern suburbs adjacent to the Central Business District (CBD), which are some of Adelaide's oldest suburbs. In 2019, the average median house price in the Central extent was AUD 859,438, compared to AUD 300,594, and AUD 434,219 for the Northern and Southern extents, respectively. The average population densities of the three study sites are between 1,800–1,900 people km^{-2} but reach over 2900 people km^{-2} in some neighbourhoods (Australian Bureau of Statistics, 2020b).

2.1.2. Remote sensing data

At all three locations, four-band optical imagery was acquired for January of 2011, and additionally at the Northern extent for February of 2018. Imagery was provided by Mapland, containing red, green, blue & near-infrared (NIR) bands captured on a Leica ADS80 Digital Optical Sensor with a spatial resolution of 50 cm. All input layers for each study site were georeferenced and resampled to standardise their cell size. One LiDAR data set was also sourced; a Digital Surface Model (DSM) covering the Northern extent and captured in 2018. Data was delivered as point cloud data in a LiDAR Data Exchange File (LAS) from Mapland with an average point density of 17.45 m^{-2}.

2.1.3. Cadastre data

Cadastral data was sourced from the Department of Planning, Transport and Infrastructure, South Australia for the greater Adelaide region for 2017 (to be used with 2018 imagery) and 2011. The Department of Planning, Transport and Infrastructure attributed land ownership as either residential or non-residential. Residential blocks were then screened by location to ensure they lay within an *urban centre (UCL)* (Australian Bureau of Statistics, 2017). *Urban centres* are defined by population and dwelling densities. Here, their boundaries have been used to exclude residencies that lie within the *Major cities of Australia* boundary, but are located on the peri-urban fringe (Fig. 1). Cadastral data was reprojected to align with the optical imagery.

2.2. Remote sensing classification

To determine the most suitable classification technique, two methods were compared during a preliminary study. *Method 1* was a relatively straightforward spectral classification, while *Method 2* was a more sophisticated method that incorporated both LiDAR and spectral data. As the preliminary study required both LiDAR and spectral data, the study site was determined by the availability of these data. Therefore, preliminary study occurred over the Northern extent using data captured in 2018 (Fig. 1). The workflow of the remote sensing classification is summarised in Fig. 2 and discussed in detail in sections 2.2.2 to 2.2.5. Image processing and data analysis were undertaken using Python 2.7 (Python Software Foundation) and the ArcMap 10.5.1 (ESRI, 2011) scripting tool Arcpy within Python 2.7.

2.2.1. Method 1

The four-band spectral imagery was subject to a maximum likelihood classification to assign pixels into one of five classes: Lawn, Trees, Sealed Surface, Shadow, or Pool. The maximum likelihood classification is a supervised classification, meaning that training areas were used to separate the spectral domain into regions that can be associated with classes of ground cover (Richards, 2013). The classification uses Bayes' theorem of decision making to assign each pixel in the image to the training class to which it has the highest probability of being a member (ESRI, 2019). To train each classification, polygons were drawn in ArcMap over five discrete land uses: Lawn, Trees, Sealed Surfaces, Shadows, and Pools, before the maximum likelihood classification tool (*MLClassify*) was applied.

2.2.2. Method 2

Point cloud data was pre-classified in accordance with the American Society for Photogrammetry and Remote Sensing (ASPRS)(ESRI, 2016). Therefore, each point was reassigned to the class 'unassigned', before

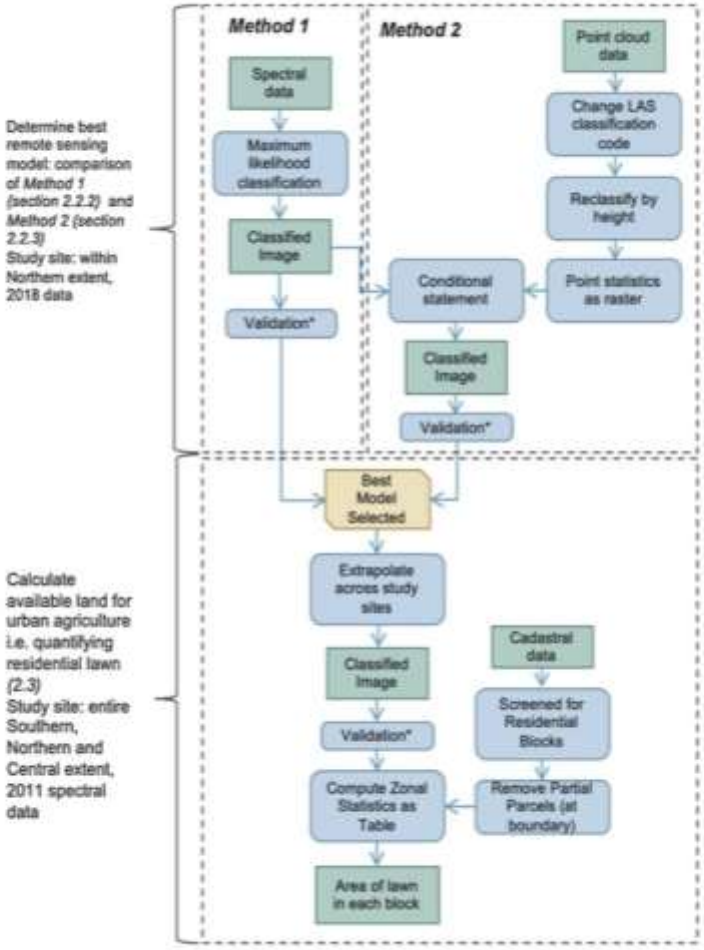

Fig. 2. Workflow of the remote sensing land use classification and calculation of area of lawn in each block.

being reclassified by height. The *LAS Point Statistics as Raster* tool was applied to extract the data into a raster conveying elevation with a pixel size of 0.5 m. Pixels that were classified as vegetation, i.e. Lawn or Trees, (from *Method 1*) *and* less than 30 cm from the ground were reclassified as lawn using a conditional statement *(con)*.

2.2.3. *Validation*

A total of 500 assessment points were created with Arcpy, employing the *create assessment points* tool, which more than satisfies the commonly-cited recommendation of 50 points for each class (Congalton, 1991). Points were stratified through the five classes so that the number of assessment points reflected the class size. Ground truthing of each point occurred through cross-referencing to high-resolution Google Earth imagery.

Error matrices were created using Arcpy, detailing the overall accuracy, producer's accuracy, user's accuracy and kappa coefficient of each method (Foody, 2002).

2.2.4. *Selecting the most successful model*

A successful model is one that arrives at the desired solution with little complexity. Therefore, the model's succinctness was considered along with the results from the accuracy assessment. *Method 1* and *Method 2* were subject to an accuracy assessment which resulted in an error matrix (Table 1). Despite *Method 1* resulting in a 5-class classification, Table 1 displays just two classes for comparison between *Method 1* and *Method 2*. The kappa coefficient is used to determine the overall accuracy of each model - it is a derived measure of accuracy taking into account the probability of a pixel being allocated to the correct class purely by chance (Foody, 2002). The use of LiDAR in urban classification (*Method 2*) is resource-intensive. Yet, its addition did not improve the results of the spectral classification (*Method 1*) (Table 1).

An anticipated advantage of the 2018 dataset over the 2011 dataset was that the inclusion of LiDAR would improve the classification results (relevant 2011 LiDAR data not available). On the other hand, the 2011 dataset covers a significantly larger spatial area than the 2018 dataset. Given the demonstrated redundancy of the LiDAR data, the decision was made to forgo using the more recent data in order to drastically increase

Table 1
Confusion matrices for Method 1 and Method 2. AU and PA stand for user's accuracy and producer's accuracy, respectively.

METHOD 1				
	Reference data			
Classified data	Lawn	Non-Lawn	Total	UA
Lawn	153	10	163	0.94
Non-Lawn	105	240	345	0.70
Total	258	250	508	
PA	0.59	0.96		
Overall accuracy = 0.77, Kappa = 0.55				
METHOD 2				
	Reference data			
Classified data	Lawn	Non-Lawn	Total	UA
Lawn	167	26	193	0.87
Non-Lawn	91	224	315	0.71
Total	258	250	508	
PA	0.65	0.90		
Overall accuracy = 0.77, Kappa = 0.54				

the breadth of the spatial analysis. To account for changes in land use patterns since the date of capture, cadastral and ABS data were used to identify trends and project expected results into 2020.

2.3. Quantifying residential lawn

The spectral classification outlined in Fig. 2 *(Method 1)* was performed in each of the three locations using imagery from 2011. The *Raster Domain* tool generated a footprint of each study site. *Select Layer by Location* was then used to select blocks wholly within the study sites' domain. Blocks less than 1 m and larger than 1 Ha were also removed in order to exclude errors in the record, large residential blocks containing multiple residencies (e.g. retirement villages), or large blocks edging on the peri-urban fringe. Finally, *Compute Zonal Statistics as Table* calculated area of lawn in each block.

The area of lawn in each block had a unimodal, negatively skewed distribution. Transformation of the data using Box-Cox did not normalise the data (the probability that the transformed data came from a normally distributed population (Shapiro & Wilk, 1965) was; $p = 2.91e^{-170}$, $p = 2.9e^{-170}$, and $p = 2.76e^{-51}$ for Central, South and North respectively). Therefore, the maximum bin vale of each dataset of the histogram (Fig. 4) was used as a measure of central tendency. The maximum bin values result in a lower number than the transformed mean, or median and will therefore result in a conservative estimate of self-sufficiency potential predicted for most residents. From this point on, the phrase 'a typical block' will be used in reference to maximum bin values (Fig. 4).

2.3.1. Projecting calculations to 2020

Cadastral data was analysed in ArcMap and Python to investigate trends in residential parcel size over time. The median block size in the study area decreased from 697 m^2in 2011 to 680 m^2in 2018, a 2.4 % decrease over six years. Assuming this trend continues, the median block size in the study area would be 20 m^2 smaller in 2020 than it was in 2011. During this time, the average size of a new house being built in Adelaide has remained relatively stable (reduction of 0.1 % P.A. over the last 15 years (Australian Bureau of Statistics, 2020c)). This does exclude extensions on existing properties. However, on the assumption that house size remains unchanged while block sizes are decreasing, there is 20 m^2less *yard* space in 2020 than there was in 2011.

A conservative assumption would be that all of this lost yard space occurs on land that would be available to urban agriculture (lawn). This calculation is used to estimate available lawn space in 2020 based on the 2011 imagery.

2.4. Calculating self-sufficiency

Self-sufficiency was measured as the capacity of private vegetable gardens to supply the total recommended vegetable intake of residents. The average dwelling in the Greater Adelaide region houses 2.5 people (Australian Bureau of Statistics, 2016), and each person requires five serves of vegetables each day (Dietitians Association of Australia, 2019). Therefore, yields (kg m^{-2} yr^{-1}), are converted to the number of serves m^{-2} yr^{-1}. Yields used in this study come from two different datasets. The first data set is published empirical data from polyculture vegetable gardens in Adelaide (Section 2.4.2). The second dataset comes from commercial yields published by the ABS (Section 2.4.3).

2.4.1. Empirical yields

Yield data was collected by citizen scientists involved in the South Australian Edible Gardens Project (Csortan, Ward, & Roetman, 2020; Csortan, Ward, & Roetman, 2020). The project's raw data was made publicly available by Csortan et al. (2020b) and is used in this study. Data were collected from 34 South Australian home gardens from September 2016 to June 2018 (not all participants collected data for the entire period). Of the 34 participants, eight were located outside of Adelaide. for sites located outside of Adelaide, the post/zip code was cross-referenced with the boundaries of Urban Centres and Localities (Australian Bureau of Statistics, 2017) and South Australian Climate Zones (Australian Building Codes Board, 2020). This was done to ensure the conditions under which the produce was grown matched the conditions expected in Adelaide urban gardens. Each of the Edible Gardens was made up of one or more *areas*. Participants recorded the size (m^2) and crop type (e.g. vegetable, fruit, eggs, honey etc.) grown in each *area*, as well as yields (kg) harvested. For this study, only crops labelled as "Vegetables" or "Vegetables & Herbs mixed" were included, and yields were converted into kg m^{-2} $year^{-1}$ (Csortan et al., 2020b).

The resulting dataset contains 34 yield estimates ranging from 0.24 kg m^{-2} $year^{-1}$ to 42.46 kg m^{-2} $year^{-1}$. A test for outliers (±1.5*IQR) detected three abnormally high yield values. In the interests of taking a conservative approach, all outliers were omitted.

The final dataset contains yields from 31 gardens. The minimum yield was 0.24 kg m^{-2} $year^{-1}$, mean was 5.08 kg m^{-2} $year^{-1}$, and maximum was 16.07 kg m^{-2} $year^{-1}$. These values will be used to represent a 'low yield' (Low_E), 'medium yield' ($Medium_E$)', 'and 'high yield' ($High_E$) scenario respectively. Importantly, these yields are similar to other yields reported for soil-based urban agriculture sites in Australia (McDougall et al., 2019).

2.4.2. Commercial yields

Commercial yields were used to create a hypothetical food garden that mimics a privately managed polyculture with crop rotations (Table 2). Yield data for fresh market vegetables grown outdoors in South Australia was collected from the Australian Bureau of Statistics (2019). The serving size of each vegetable was calculated from the FoodData Central online database (US Department of Agriculture, 2019). Finally, crops were grouped on planting season (spring, autumn, or all year round). The total number of serves grown per square meter was calculated as the average of [(spring + autumn) + year-round

Table 2
Data on crops included in the hypothetical garden using commercial yields. Yield data collected from the Australian Bureau of Statistics (2019), serving size from FoodData Central online database (US Department of Agriculture, 2019). Crops have been classified according to planting season: spring/ summer ('spring'), autumn/winter ('autumn'), and all year round ('all year').

	Yield (kg m^{-2})	Planting	One serve (g)	Serve m^2
Tomato (Fresh market - outdoor)	2.87	spring	123	23.33
Sweet corn	0.50	spring	72.5	6.90
Pumpkin	2.35	spring	122.5	19.18
Potato (Fresh market)	3.91	spring	68	57.50
Pea	0.09	autumn	80	1.07
Cauliflower	1.93	autumn	62	31.13
Brussel sprouts	2.58	autumn	78	33.12
Broccoli	0.79	all year	78	10.17
Carrot	6.20	all year	78	79.49
Lettuce	0.96	all year	35	27.46

crops]. Equal amount of land be allocated to each crop. In this way, a 1 m^2 plot would have 0.5 m^2 dedicated to year-round crops, and the other 0.5 m^2 dedicated to spring crops for 6 months of the year, and autumn crops for the other 6 months of the year. Given that urban food gardens produce higher yields than their commercial counterparts (CoDyre et al., 2015; McDougall et al., 2019; Zainuddin & Mercer, 2014), the commercial yields will provide a robust, yet conservative estimate of total output potential.

$$\text{serve per } m^2 = \frac{\text{serve per } m^2((\text{spring} + \text{autumn}) + \text{all year})}{2}$$

3. Results

3.1. Spectral classification

There is no universal standard for interpreting Kappa coefficients, however, a commonly cited guide states that values < 0 indicate no agreement, 0–0.20 indicates slight agreement, 0.21–0.40 indicates fair agreement, 0.41–0.60 moderate agreement, 0.61–0.80 indicates substantial agreement, and 0.81–1 is almost perfect (Landis & Koch, 1977). According to this guide, the classification of the Central extent has substantial agreement, while the Southern and Northern extents both have almost perfect agreement. The ability of the kappa coefficient to succinctly summarise the performance of classification across all classes omits information about interclass confusion. Conversely, producer's accuracy quantifies pixels classified as Lawn in the reference image (ground truth) but not by the model, i.e. false negative. While user's accuracy quantifies the proportion of pixels that have been classified as Lawn by the model but contain other cover types in the reference data i.e. false positives. There was some confusion between the Lawn and Pool classifications in the southern extent, but Lawn was most commonly confused with Trees and Sealed Surfaces across all study sites (Table 3).

To further illustrate instances of misclassification, a small area within the southern extent is shown in Fig. 3; first as an unclassified true colour image (Fig. 3*ii-b.*), and then overlaid with the boundary of residential blocks, and pixels classified as residential lawn (Fig. 3*ii-a.*). Fig. 3*a.* highlights an area in which lawn pixels have been omitted from the model, and incorrectly classified as 'tree' pixels. While Fig. 3*b* illustrates an area in which 'sealed surface' (a driveway) has been misclassified as lawn.

Table 3
Confusions matrices for remote sensing classification performed at each study site. AU and PA stand for user's accuracy and producer's accuracy respectively.

CENTRAL EXTENT 2011

	Reference data						
Classified data	Lawn	Tree	Sealed Surface	Shadow	Pool	Total	UA
Lawn	86	44	12	0	0	142	0.61
Tree	3	79	1	0	0	83	0.95
Sealed Surface	9	10	214	2	0	235	0.91
Shadow	0	14	2	25	0	41	0.61
Pool	0	0	1	0	9	10	0.90
Total	98	147	230	27	9	511	
PA	0.88	0.54	0.93	0.93	1.00		
Overall accuracy = 0.81 Kappa = 0.72							

NORTHERN EXTENT 2011

	Reference data						
Classified data	Lawn	Tree	Sealed Surface	Shadow	Pool	Total	UA
Lawn	166	7	6	0	0	179	0.93
Tree	13	79	1	1	1	95	0.83
Sealed Surface	8	2	186	14	0	210	0.89
Shadow	0	0	0	15	1	16	0.94
Pool	0	0	1	9	0	10	0.00
Total	187	88	194	39	2	510	
PA	0.89	0.90	0.96	0.38	0.00		
Overall accuracy = 0.87 Kappa = 0.82							

SOUTHERN EXTENT 2011

	Reference data						
Classified data	Lawn	Tree	Sealed Surface	Shadow	Pool	Total	UA
Lawn	137	14	11	0	0	162	0.85
Tree	10	56	0	0	0	66	0.85
Sealed Surface	9	7	241	4	0	261	0.92
Shadow	0	0	0	10	0	10	1.00
Pool	2	2	5	0	1	10	0.10
Total	158	79	257	14	1	509	
PA	0.87	0.71	0.94	0.71	1.00		
Overall accuracy = 0.87 Kappa = 0.80							

3.2. Quantification of residential lawn

The area of lawn in each block was calculated for each of the three study sites. The most frequent bin values from the resulting histogram have been used as a measure of central tendency, and signify the amount of lawn in a *typical* block for 2011 (section 2.3). In 2011 the amount of lawn found in a typical residential block was 127 m^2, 102 m^2, and 117 m^2 for Central, North and South respectively, and are represented by the three vertical lines in Fig. 4. Adjusting these values according to land use changes over time (section 2.3.1), the amount of lawn found in a typical residential block in 2020 is 107 m^2, 82 m^2, and 97 m^2 for Central, North and South respectively. These values were used to perform all further calculations.

Fig. 3. A subsection form the southern extent displaying residential lawn as classified by the spectral classification and boundaries of residential blocks. Lawn was classified with a user's accuracy of 0.85 and producer's accuracy of 0.87. (a.) highlights an area where lawn has been under classified, while (b.) highlights an area where lawn has been over classified.

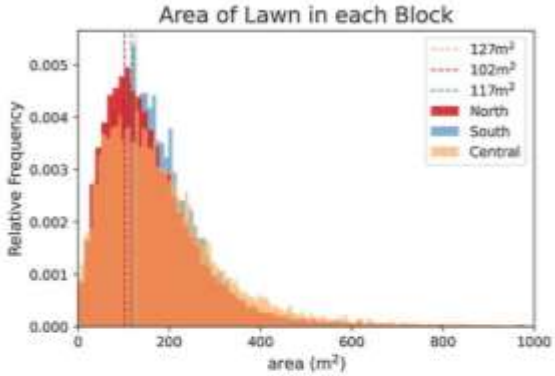

Fig. 4. Histogram quantifying the amount of Lawn in each residential block (m^2) within the three study sites. The horizontal lines depict the most frequent (highest) bin values. Central bin maximum is 126.94 (max = 4434.25, n = 28,260), Northern bin maximum is 102.20 (max = 6356.5, North n = 7185), and Southern bin maximum is 117.29 (max = 6289.25, South n = 4610). X axis max set to 1000.

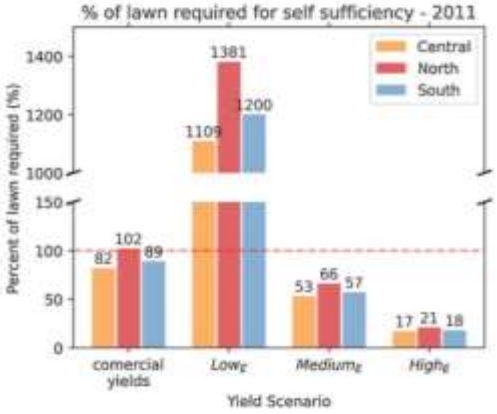

Fig. 5. The proportion (%) of available land within a typical residential block that would be required for self-sufficiency of vegetables (2.5 people per household). Available land is defined as land classified as residential lawn from 2011 spectral imagery. Values greater that 100 % indicate that space is limiting. Variables are: yield scenario (Commercial yield, LowE, MediumE, HighE); and study site (Central extent, Northern extent, Southern extent).

3.3. Self-sufficiency

The space required to grow the recommended vegetable intake of an average household (2.5 people) depends on the yield that can be achieved. Under commercial yields, 104 m^2 of land is required. The Low_E, $Medium_E$, and $High_E$ yield scenarios would require 1407 m^2, 67 m^2, and 21 m^2 of land respectively.

The proportion of available land required for self-sufficiency is dependent on both the yield achieved, and the amount of land available. This metric has been calculated for a typical residential block in 2011 (Fig. 5), and 2020 (Fig. 6), based on spectral imagery and land use changes experienced between 2011 and 2020. Values of >100 % indicate that residential lawn space exceeds the minimum spatial requirements for self-sufficiency. For example, under commercial yields, 104 m^2 of land is required to supply the household with its recommended vegetable intake. In 2011, a typical block in the central extent contains 127 m^2 of lawn space (Fig. 4). Therefore 82 % of lawn space

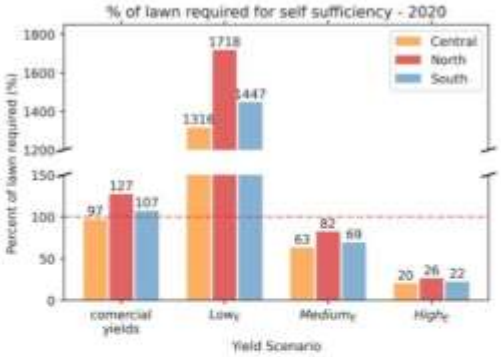

Fig. 6. The proportion (%) of available land within a typical residential block that would be required for self-sufficiency of vegetables (2.5 people per household). Calculations of available land have been projected to 2020 based on spectral imagery and trends in land use. Values greater that 100 % indicate that space is limiting. Variables are: yield scenario (Commercial yield, LowE, MediumE, HighE); and study site (Central extent, Northern extent, Southern extent).

would be required for self-sufficiency if yields are similar to commercial farms (Fig. 5). However, in 2020, a typical block in the central extent is expected to have 107 m^2 of lawn (*Section 2.3.1*), and so 97 % of lawn space would be required for self-sufficiency (Fig. 6).

In 2011, under all yield scenarios but Low_E, lawn in a typical block provides more than enough space supply the household's recommended intake (Fig. 5). The only exception being that under commercial yields a typical block in the northern extent would require 102 % of lawn to achieve self-sufficiency; however, the average across all three study sites is 91 %. The averages across study sites for Low_E, $Medium_E$, and $High_E$ yield scenarios are 1,230 % 59 % and 19 % respectively.

Changes in land use since the date of image capture in 2011 result in less residential lawn space available 2020 (*Section 2.3.1*). Consequently, with-out changing the yield scenarios, a higher proportion of available land would be required to achieve self- sufficiency. Specifically, 1,493 %, 72 % and 23 % of the available land in a typical block would be required for self-sufficiency for Low_E, $Medium_E$, and $High_E$ yield scenarios respectively (Fig. 6). Under commercial yields 111 % of available land would be required.

The previous result examined the experience of *a typical residential block* in 2020. Next, we consider the proportion of properties that meet the minimum spatial requirements for self-sufficiency (Fig. 7). The total amount of lawn in each block has been adjusted for land use change expected in 2020 (as per Fig. 6). Each subplot in Fig. 7 demonstrates a different yield scenario. The dashed vertical red line represents the cut-off point below which properties do not meet the minimum spatial requirements for self-sufficiency. Under commercial yields, 76 % of houses meet the minimum spatial requirements for self-sufficiency. Under $High_E$, $Medium_E$ and Low_E, the proportion of properties that meet the spatial requirements are 93 %, 69 % and <0.5 % respectively.

4. Discussion

The remote sensing classification was successful in quantifying available land for urban agriculture and revealed the significant food production potential of residential properties. Under medium and high yield scenarios ($Medium_E$ and $High_E$), lawn space in a typical block provides more than enough space to supply the household's recommended vegetable intake (Fig. 6). However, under $Medium_E$ approximately 30 % of properties did not meet the minimum spatial requirements for self-sufficiency, while under Low_E, less than 1% of properties met the spatial requirements (Fig. 7). Therefore, wide-spread

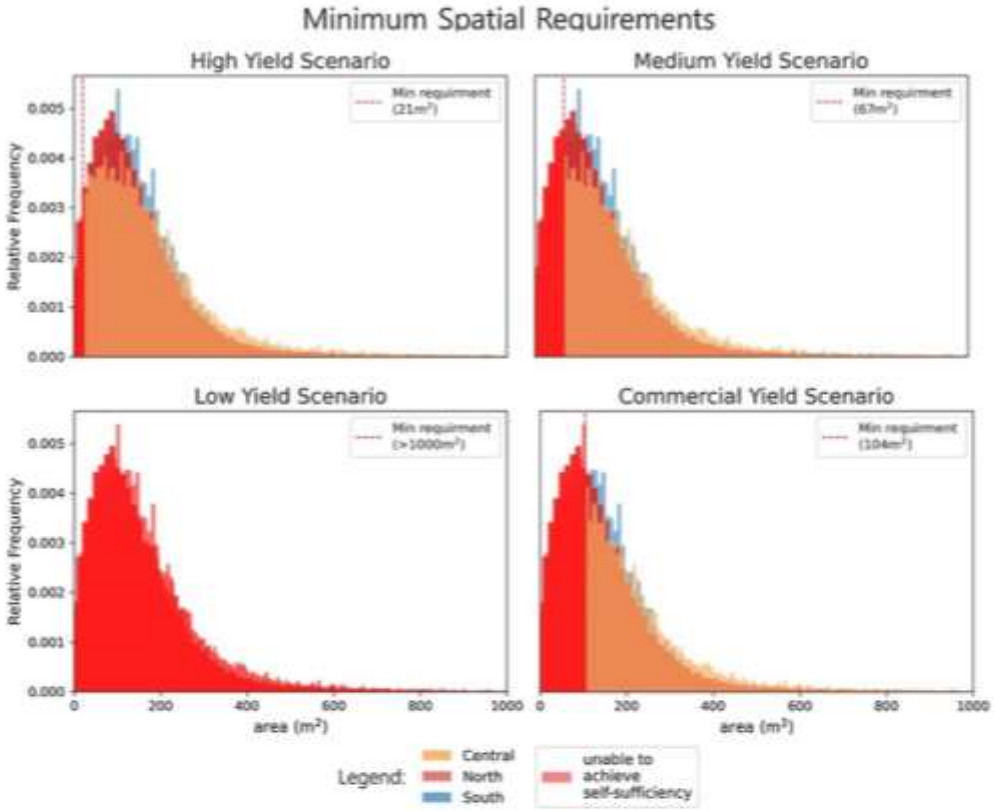

Fig. 7. The amount of lawn in each residential block (m^2), adjusted for land use change predicted in 2020, and the minimum cut off value for self-sufficiency under each of the three yield scenarios. These are: 1407 m^2, 67 m^2, 21 m^2, 104 m^2 for Low_E, $Medium_E$, $High_E$ and commercial yields respectively.

self-sufficiency through residential food gardens will require relatively high yields.

Self-sufficiency becomes increasingly important during times of stress. In the case that urban agriculture is employed to improve urban self-sufficiency in response to economic, social, or environmental shock, it is reasonable to assume all gardeners would be interested in maximising their productivity. However, the empirical yield data used in the model were taken from an observational study, where maximising self-sufficiency potential was not a motivating factor for gardeners. Instead, the top five motivations were: producing fresh tasty produce, enjoyment, health reasons, natural connection, and to save money (Csortan et al., 2020a). Furthermore, the data set was negatively skewed, meaning that a small number of gardens were significantly more productive than most (Csortan et al., 2020b). This trend is common among similar studies (CoDyre et al., 2015; McDougall et al., 2019; Zainuddin & Mercer, 2014). Through interviews with urban gardeners, CoDyre et al. (2015) found that high yields were achieved when gardeners were actively engaged in maximising their garden's efficiency in terms of space (i.e. by participating in biointensive practices). Importantly, gardeners were able to achieve high yields even with little gardening history (CoDyre et al., 2015). Therefore, high yields from observational studies are likely to represent an attainable, best practice scenario for most growers.

The high yield scenario in this research was 16.07 kg m^{-2} $year^{-1}$. Assuming these yields can be replicated throughout Adelaide, 93 % of houses across all study sites met the minimum spatial requirement for self-sufficiency through vegetable production; and just 23 % of lawn in a typical block would be required to achieve self-sufficiency. However, access to training and education would need to be considered if high yields are to be expected.

Even under the $High_E$, a proportion of properties do not meet the minimum spatial requirements to achieve self-sufficiency (Fig. 7). The model assumes each property block houses the same number of people, which is a simplified model of reality. This is a symptom of the spatial scale at which the statistical data was captured. Without further analysis at a finer spatial scale, it is unclear whether properties with smaller yards house fewer people, thus requiring less vegetable growing space. The model also doesn't capture those living in apartments. Less than 2% of dwellings in the Northern and Southern extent and 8% of dwellings in the Central extent are apartments and unit blocks (Australian Bureau of Statistics, 2020a). Therefore, their presence won't meaningfully impact the measure of a *typical block* (Fig. 4) but may impact the proportion of properties able to achieve self-sufficiency (Fig. 7). Future work that examines spatial resources at a finer scale would build on the results of this study. Nonetheless, those unable to achieve self-sufficiency through residential gardens can exploit street verges and community gardens, or explore raised garden beds and rooftop farming - methods which have been the subject of other work (Hara, McPhearson, Sampei, & McGrath, 2018; Lee, Lee, & Lee, 2015; McDougall et al., 2020). This study focuses specifically on residential agriculture because it is often overlooked in the broader context of urban agricultural research. The results highlight residential land as a valuable resource for urban agriculture and encourage further research into this practice.

The spatial resource appears to be shared relatively consistently across the three study sites despite the unique urbanisation history of each (see section 2.2.1). Martellozzo et al. (2014) identified population

density as the most significant factor in determining cities' self-sufficiency potential. The three study sites investigated have average population densities between 1,800−1,900 people km^{-2}, with a maximum of 2900 people km^{-2} in some neighbourhoods (Australian Bureau of Statistics, 2020b). Ambiguities in defining urban and peri-urban boundaries, and heterogeneity within cities, make it hard to categorise urban areas according to their relative density. One estimate is that around 10 % of the world's cities house less than 2000 people km^{-2} (Demographia, 2020). So, while this is a case study of one city, the results are likely to be relevant in other parts of the world including suburban areas associated with large cities.

In this case study, the spatial constraints on residential farming have increased over time (Fig. 5, Fig. 6). Given the Government of South Australia (2017a) plan for metro infill, further loss of arable urban land can be expected. However, policymakers should be aware of the spatial limits being placed on urban agriculture and balance them against other interests. Besides food provisioning, in-ground vegetable production can deliver a suite of ecosystem services to urban landscapes (Cleveland et al., 2017; Wielemaker et al., 2018). Work by Cleveland et al. (2017) demonstrated that vegetables grown in residential gardens could reduce greenhouse gas emissions by 2.10 kg $CO2_e$ for every kg of vegetable consumed. Given that Adelaide has a population of over 1.2 million, and a healthy adult should consume 137 kg of vegetables annually (Dietitians Association of Australia, 2019), there is significant potential for residential gardens to contribute to climate change mitigation. To preserve residential space that might be used for urban agriculture, policy intervention should be considered. Further modelling of eco-system services delivered by residential food gardens would better inform urban planning decisions.

The primary goal of this research paper was to examine the spatial constraints on food production. Food processing or cold storage required to supply food evenly throughout the year has not been considered in this paper, neither has crop losses or waste in the production phase. The model also assumes that 100 % of the crop harvested will contribute to dietary self-sufficiency without considering the % inedible (e.g., skin, seeds, stem). For most crops the % inedible will be a negligible amount, however almost 50 % of sweet corn ears are inedible. Sweet corn made up 2.6 % of total yield harvested by urban gardeners in the $High_E$, $Medium_E$ and Low_E scenarios (Csortan et al., 2020b), and occupies 6% of the land over a one-year cycle in the commercial yield scenario (Table 2). Additionally, this paper has not examined the micronutrient content of the crops grown to see if a nutritionally balanced diet can be achieved. This will be an important next step in self-sufficiency research.

Self-sufficiency through urban agriculture would result in a suite of economic and social interactions. Without overcoming the initial obstacle of space, social or economic barriers to urban agriculture uptake are trivial. However, given that space is not likely to be a limiting factor (Figs. 6 and 7), the next step is to investigate the social and economic factors. The second half of this discussion explores the potential barriers to validate the conclusions drawn at the end of the paper.

The self-sufficiency model relies on two classes of assumptions. The first is that all available land will be uniformly productive. The second is that the model correctly identifies all available land.

To address the first assumption, not all of the lawn space may be suitable for vegetable production due to heavy metal contamination in the urban environment (Rouillon, Harvey, Kristensen, George, & Taylor, 2017). Salomon, Watts-Williams, McLaughlin, and Cavagnaro (2020) tested heavy metal pollutants in various locations around Adelaide, including those with industrial histories, and were unable to detect any concentrations above the safety guideline limits. However, Rouillon et al. (2017) previously found that Sydney's pollutant concentrations vary greatly, even within a single residency. Using Rouillon et al. (2017)'s finding as a highly conservative estimate, up to 28 % of available land might be unsuitable due to contamination (McDougall et al., 2020). This would still allow enough uncontaminated land to meet the daily requirements of residents under $Medium_E$ and $High_E$ conditions (Fig. 5).

It has already been discussed that households will need relatively high yields in order to achieve self-sufficiency. However, the labour required to achieve high yields in low-tech urban gardens may limit the feasibility of self-sufficiency. Csortan et al. (2020a), who provided empirical yield data for this study, recorded labour inputs to be 9 min. m^{-2} 30 $days^{-1}$ (1.8 h m^{-2} $year^{-1}$) for mixed horticultural production with an average yield ($Medium_E$). Therefore, the $Medium_E$ scenario (Figs. 5–7) would require each person to invest 20 min. of labour day^{-1} (Csortan et al., 2020a). However, yields are positively correlated with labour inputs (Csortan et al., 2020a), meaning higher labour inputs would be required that to achieve $High_E$ (Figs. 5–7). Furthermore, labour inputs vary significantly between studies. CoDyre et al. (2015) reported inputs of 41 min m^{-2} 30 $days^{-1}$ over a 4 – 5-month growing season, while McDougall et al. (2020) found labour inputs to be 6 h m^{-2} $year^{-1}$ in urban food gardens. The cause of these discrepancies is unclear. Csortan et al. (2020a) found gardeners significantly over-estimated time spend working their garden when they were surveyed (3.5 h $week^{-1}$), compared to when actual time spent was documented (1.3 h $week^{-1}$). Therefore, methods of recording, and definitions of what constitutes *labour* could alter the result.

All of the labour estimates cited come from observational studies in which gardeners are not necessarily engaged in maximising productivity. As was discussed in relation to yield efficiency, if urban agriculture was employed to improve self-sufficiency in response to stress, gardeners may be more engaged in maximising efficiency. Taken together, evidence suggests that labour will not limit self-sufficiency potential; however, the evidence is not consistent across studies. To better understand human factors, future work should investigate maximum labour efficiency as well as attitudes towards gardening.

In addition to labour, production requires water and nutrients. A major advantage of producing food within the urban boundary is the opportunity to utilize residual water and nutrients found in cities - primarily household waste and stormwater. Previous work illustrates that these resources are abundant, and can sustain wide-spread urban food production (Cleveland et al., 2017; McDougall et al., 2020; Wielemaker et al., 2018). Importantly, the technologies that will facilitate decentralised waste treatment are being developed (Saini, Lohchab, Anil, & Kulbir, 2018).

The second class of assumptions is that the model correctly identified all available land. Four-band spectral imagery proved to be a valuable tool, able to classify urban space with a high degree of accuracy (Table 3). However, all maps are models of reality and will therefore include some error and inherent limitations. Widespread misclassification, such as the errors observed in Fig. 3, could threaten the integrity of the self-sufficiency model. The similarity between spectra of lawn and trees is a common limitation in spectral classifications of vegetation (Kremer & DeLiberty, 2011; Wetherley et al., 2017). However, the spectral classification performed unexpectedly well on its own (Tables 1 *and* 3). The error associated with each classification has been deemed too small to have a meaningful impact on the final result (Table 3).

The imagery was captured in January (Austral summer) when Adelaide experiences a typically dry summer resulting in desiccation of (non-irrigated) lawn. Until the end of 2010 it was illegal for residents to irrigate their lawn due to water scarcities associated with the millennium drought (Government of South Australia, 2017b) (imagery for this study was acquired from January 2011). These exceptional conditions work favourably for a spectral classification as the dry lawn has a markedly different spectral reflectance to leafy trees (Table 3) and explains why the spectral classification performed so well (Table 1). It is assumed that all lawn in residential areas is treated similarly. Should a selection of residents irrigate their lawn during an otherwise dry period, it may lead to the systemic misclassification of lawn. This is an important point, particularly if the methods of the study are to be extrapolated. In the context of this research, the conditions may result in conservative

estimates of the total amount of lawn if some residents used bore water for irrigation. In wetter years, the inclusion of LiDAR data to differentiate between irrigated lawn and trees may be more effective than it was in the present study. The same may be true for regions with less a distinct seasonality.

Finally, the use areal of imagery makes it impossible to discern land use under the canopy of a tree. Therefore, any residential lawn that resides under a tree would not have been included in this analysis – making estimates of lawn in each block conservative.

5. Conclusion

This research demonstrates the plausible reality of self-sufficiency through vegetable production using Adelaide, Australia, as a case study. A remote sensing classification was used to quantify residential lawn space available for use in urban agriculture. The model revealed that household self-sufficiency of vegetables could occur via open-air, soil-based production. Under a medium yield scenario, 72 % of lawn space in a typical block would be required to meet the recommended vegetable intake of residents. Under a high yield scenario, just 23 % of lawn would be required. Other prerequisites that might limit production, such as labour, nutrients, and water, were explored in the discussion. Self-sufficiency, as described in this paper, would be contingent on high-yields, proper nutrient management, and maximising the efficiency of gardeners.

The results are encouraging for those seeking to increase local food production or harness ecosystem services provided by urban agriculture. However, in this case study, the self-sufficiency potential is not currently being realised, and residential growing space has decreased over time. While this research explores a production model that can be carried out by individuals without government interventions, policymakers should consider actions that preserve potentially productive residential land. Further modelling of eco-system services delivered by residential food gardens would better inform such decisions.

Declaration of Competing Interest

The authors reported no declarations of interest.

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Chapter 3
Lawn with a side salad: Rainwater harvesting for self-sufficiency through urban agriculture

Chapter 3 contains the PDF of the published research article. The supplement to the work is found in Appendix F.

Statement of Authorship

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Lawn with a side salad: Rainwater harvesting for self-sufficiency through urban agriculture

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ABSTRACT

In a rapidly urbanising world, urban agriculture has garnered much attention for increasing resilience to a range of interrelated stressors, including climate change, food insecurity, economic instability and most recently, public health crises. Critical to understanding the viability of urban agriculture and ensuring its environmental sustainability, is the intersection of available land and the supply of agricultural inputs. Here, we address existing knowledge gaps related to urban agriculture and rainwater harvesting, by quantifying the self-sufficiency potential of half a million homes in Adelaide, South Australia. We developed a model that combines high resolution spectral and LiDAR imagery with productivity and irrigation data that reflect the actual behaviors of urban growers. Results indicate that 65% of residential properties contain enough available land to provide dietary self-sufficiency of vegetables, while capturing and storing adequate rainwater for irrigation, even in the modelled Dry year scenario. The modelled edible garden and associated storage tank would occupy around half of the lawn space in a typical residential block. These results highlight the substantial contribution urban agriculture can make to a more sustainable food systems in a low-density city.

1. Introduction

The world's urban population has grown exponentially in recent decades. By 2050, its projected that 68% of people will live in urban areas (United Nations, 2019).The world's rapidly expanding urban areas are having a profound impact on global food production. As ore than 60% of the world's irrigated croplands are situated near urban boundaries, such urban expansion will displace croplands that are more productive than the global average (d'Amour et al., 2017). These issues are confounded by growing competition for water and energy, as well as the threat of climate change (Godfray et al., 2010). All of this is to say that the human population will need to produce more food than ever before, with less resources . One of the largest challenges for the coming decades will be engineering sustainable food systems to feed this ever-growing, and increasingly concentrated, urban population (Bren d'Amour et al., 2017, Godfray et al., 2010). Urban agriculture, the practice of growing food within the urban boundary, maybe be one such solution.

The production of fruit and vegetables through urban agriculture, which is also referred to as urban horticulture, can simultaneously address a range of urban challenges. These include, but are not limited to, biodiversity loss, public health, social cohesion, and economic growth (Artmann and Sartison, 2018). Furthermore, urban agriculture utilizes land that would not otherwise be put to productive use, and allows food to be grown in close proximity to where it will be consumed. Small scale urban agriculture has also been shown to produce high yields (Csortan et al., 2020a, McDougall et al., 2019), and is often posed as a solution to satisfy the need for fresh and safe food in cities and urban settlements (Khan et al., 2020). In recent years, the COVID-19 pandemic has accentuated existing issues of food insecurity, placing a spotlight on the value of local food production, and triggering increased interest in urban agriculture in recent year (Lal, 2020, Pulighe and Lupia, 2020, Khan et al., 2020).

The benefits of urban agriculture are well documented through-out the literature (Weidner et al., 2019). However, there is concern that the academic discourse around urban agriculture has tended towards advocacy, rather than critically analyzing the resource consumption and production potential (Neilson & Rickards, 2017; Weidner et al., 2019). For urban agriculture to realise its full potential there needs to be

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adequate access to the resources required. Additionally, to ensure the environmental sustainability of urban agriculture, judicious management of resources is required (McDougall et al., 2019). Despite this, one of the most critical agricultural inputs, water, is understudied in urban agricultural research (Pollard et al., 2018b).

Urban agriculture is uniquely placed to use harvested rainwater for irrigation due to its proximity to urban infrastructure. Rainwater harvesting (RWH) is commonly discussed in relation to the sustainable cities paradigm, for its extensive contributions towards environmental sustainability (de Sá Silva et al., 2022). RWH can help adapt food production to issues of climate change and increased water scarcities by reducing the reliance on freshwater abstractions (de Sá Silva et al., 2022, UNESCO, 2020). RWH also helps reduce the energy consumption associated with urban water supply (de Sá Silva et al., 2022, Vargas-Parra et al., 2013). Additionally, harvesting rainwater before it enters the stormwater stream reduces the volume of polluted stormwater discharged from urban areas (Zhang et al., 2020). However, research relating specifically to RWH for use in urban agriculture is limited (Amos et al., 2018).

1.1. Ground-based vegetable production systems

Urban agriculture is a diverse term relating to any, and all, production methods occurring on private land. However, in Australia, 90% of urban agricultural practitioners utilize residential back yards for ground-based food production, with 31% producing food in their front yards (not mutually exclusive) (Donati and Rose, 2020). Ground-based production refers to growing crops at ground level in native, amended, or introduced soil. In an urban setting, home gardening can improve dietary diversity, micronutrient deficiencies, and increase the frequency of fruit and vegetable consumption in children (Cabalda et al., 2011). Ghosh (2021) argues that growing food in home gardens connects people back to land, nature and memories and is also a potential method for improving health and resilience. Despite this, residential production remains understudied in the context of urban agriculture research (Kirkpatrick and Davison, 2018, Taylor and Lovell, 2014). Low-tech farming, such as ground-based production of vegetables, can be undertaken with little capital investment or complex infrastructure. This makes it immediately available to households or individuals with access to available, suitable land. Furthermore, nutrient-dense, perishable crops, such as vegetables, are efficient to grow in urban areas given their high economic value (Martellozzo et al., 2014, Safi et al., 2011, Weidner et al., 2019). Hence, focusing selectively on horticultural crops optimizes the benefits of urban agriculture (Weidner et al., 2019).

For these reasons, this work will focus on open air, ground-based production of vegetables, occurring on residential land. High-tech solutions, such as hydroponic production, may lead to increased water efficiency (Al-Kodmany, 2018). However, they present social and economic barriers for urban growers (Pollard et al., 2018a, Pollard et al., 2017), and are not the focus of this research.

1.2. Urban agriculture and self-sufficiency

Both the self-sufficiency and the RWH potential of home gardens have been measured to various degrees in the literature. Ghosh (2021) calculated that home gardens provide the recommended vegetable intake to 84% of city's population if 90% of potentially productive residential land were converted to food production. However, the self-sufficiency potential of individual houses was not calculated (i.e. does an individual property have the capacity to provided dietary self-sufficiency to residents). Recently, Hume et al. (2021) used remote sensing techniques to map and model food production potential in Adelaide, Australia. Over 90% of residential properties sampled could achieve dietary self-sufficiency of vegetables at the household scale, simply by dedicating a portion of the land that is currently residential lawn. However, the analysis of self-sufficiency did not extend to agricultural inputs such as water. Also using remote sensing techniques, McDougall et al. (2020) found that Sydney, Australia, could grow 15% of its entire vegetable supply on verges, vacant land, and a portion of yard space. Furthermore, captured rainwater from rooftops met the annual irrigation requirements of all production scenarios modelled (McDougall et al., 2020). However, results were not analyzed at the household scale, which could create a practical barrier for individual households in accessing both land and irrigation. Additionally, temporal lags between when water can be captured, and when it may be used for irrigation were not considered. Their model also assumed that the water requirements of urban food gardens would match that of commercial vegetable production in the surrounding basin (McDougall et al., 2020). Small scale urban agriculture plots produce both higher yields and are typically more diverse than their commercial counter parts (Pollard et al., 2018a, Salomon et al., 2020, Hume et al., 2021, Csortan et al., 2020a). These key differences in production methods between small-scale urban and traditional commercial farming would result in alternate irrigation requirements. A metric specific to polyculture food gardens, and which reflects the behaviour of urban gardeners, may further assist in planning.

1.3. Economic viability

Sustainable urban agriculture must also be economically viable (Van Veenhuizen and Danso, 2007). Commercial urban farms are generally operated for profit, while home and community gardens are generally considered to provide more social benefits (O'Sullivan et al., 2019). Perhaps for this reason, urban agriculture research in developed countries tends to focus more on health, emotional, and educational benefits, rather than economic outcomes (Wadumestrige Dona et al., 2021). However, in developed countries, those that rely most heavily on urban agriculture for food security are low-income earners (Donati and Rose, 2020).

Home vegetable gardens can be profitable (Langellotto, 2014). Csortan et al. (2020a) found that of 35 surveyed edible urban gardens in Adelaide, South Australia, 65% should break even in five or less years. However, this analysis did not include any costs associated with RWH. RWH systems are expensive and can have low long-term benefit-cost-ratios (Khan et al., 2021, Preeti and Rahman, 2021). If rainwater harvesting is employed to help adapt to, and mitigate against increasing issues of water stress, the economic cost of rainwater harvesting should be considered.

1.4. Novelty and aims

The potential benefits of back yard urban agriculture are well understood. However, there remains the need to critically analyze resource consumption and production potential. Water, which is one of the most critical agricultural inputs, is particularly understudied, and research relating specifically to RWH for use in urban agriculture is limited. Self-sufficiency through urban agriculture has been measured to various degrees in the literature, yet knowledge gaps remain. These include: a metric of self-sufficiency, which includes RWH, measured at the household scale; modelling self-sufficiency using a monthly time step; and the inclusion of productivity and irrigation data that accurately reflect the reality of urban growers. Addressing these gaps will add much needed sophistication to self-sufficiency modelling, which is lacking in the current literature. The inclusion of multiple scenarios and spatial forecasting will illuminate the opportunities and limitations for household self-sufficiency through urban agriculture.

Therefore, this study aims to critically analyze the self-sufficiency potential of households.

More precisely, the primary objective is to determine the potential for roof harvested rainwater to support dietary self-sufficiency through urban agriculture, using remote sensing and spatial modelling. For the purpose of this study, *dietary self-sufficiency* will refer to residents' ability

to produce enough food to meet their recommended vegetable intake (RVI). The term *harvested rainwater* refers to water that runs off the roof of a building and is captured in rainwater tanks from the building's gutters or downpipes prior to contact with the ground.

To achieve the main objective, the specific aims are:

1 to classify urban land use and parametrize growing space and roof area;
2 to calculate the water demand of crops to meet the RVI of households;
3 to determine the RWH potential of homes;
4 to ascertain the sensitivity of self-sufficiency to two potential stressors: low rainfall; and physical changes to urban landscape over time; and
5 to examine the economic viability of household RWH by considering the financial cost of water storage.

2. Methods

2.1. Adelaide as a case study

To investigate capabilities of urban agriculture and integrated RWH, the city of Adelaide, South Australia, is used as a case study. Adelaide (34.9° S, 138.6° E) is the capital city of the Australian state South Australia. Adelaide (Australia) was chosen as a case study based on the following criteria:

1 Adelaide is positioned in a productive agricultural landscape and has a Mediterranean Climate (Köppen climate classification Csa) of hot dry summers and cool wet winters. January, the hottest month on average, has an average high of 29.6°C and an average low of 17.3°C, with 20.1 mm of rain. July, the coolest month on average has an average high of 15.4°C and an average low of 7.6°C, with 76.3 mm of rain. The average maximum temperature in Adelaide is 22.5°C, and the average minimum is 12.3°C. The annual average rainfall is 547.1 mm (Bureau of Meteorology, 2020).
2 Adelaide is a relatively low density, sprawling city, which indicates good self-sufficiency potential (Martellozzo et al., 2014). Adelaide houses 1.3 million people over an 837 km $^{-2}$ footprint - approximately 1,620 people km^{-2} (Fig. 1). However, the population density is heterogenous, and reaches up to 3,000 people km^{-2} in areas (Australian Bureau of Statistics, 2020a). Free standing houses make up 75% of all residential dwellings in Adelaide, and a further 17% are semi-detached. Only 8% of residential dwellings are apartments (Australian Bureau of Statistics, 2020b). The high proportion of free standing, and semi-detached dwellings provides opportunities for food production to occur on residential land. Generally speaking, residential yards are an important land reserve as cities continue to densify (Ghosh, 2021).
3 There is also a body of in-depth literature characterising urban agriculture in the region (Pollard et al., 2018a, Csortan et al., 2020a, Hume et al., 2021), which provides a solid foundation for subsequent work.

Taken together, the relatively low density and mild climate make Adelaide an attractive place to practice residential urban agriculture. Food producing areas on Adelaide's urban fringe are protected from development to preserve peri-urban agriculture (SA Planning Portal, 2019). While there is no equivalent legislation or policy surrounding food production within Adelaide's metropolitan areas, both private and public urban agriculture does occur. Only the land that meets the objective definition of *urban character*, according to the Australian Bureau of Statistics (2017) is included in this research (Fig. 1).

Fig. 1. Location map of study site.

2.2. Model design and inputs

The model includes a land use classification based on spatial data, which is used to parametrize individual properties in terms of their roof area and growing (lawn) space. The water requirements are a function of growing space, irrigation requirements, and potential yields. Water capture potential is a function of both roof area and rainfall data (Fig. 2). The model has six primary data sources which have been analyzed to varying degrees prior to the commencement of this research. They are

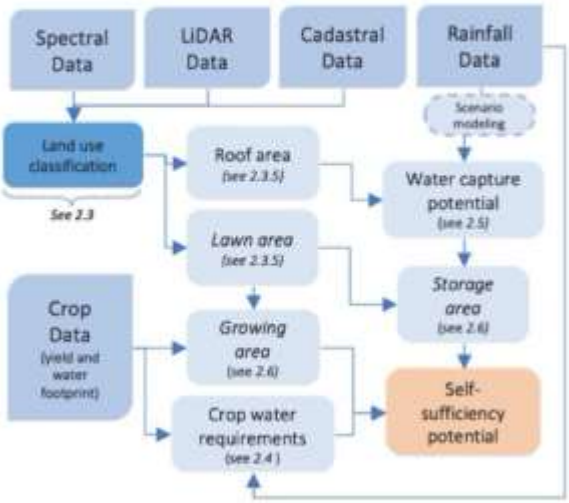

Fig. 2. Flow chart of method design.

described in Sections 2.2.1-2.2.5.

2.2.1. Spectral data

Spectral imagery is used in the land use classification to delimitate pixels with markedly different spectral reflectance (e.g., photosynthetic/no photosynthetic cover types). Spectral imagery was provided by Aerometrex (2022). Imagery was captured on the 24th and 25th of September 2018 on UltraCam Eagle Prime 210 (a VIS - NIR airborne sensor) at 11085 ft with a spatial resolution of 50 cm. Imagery was ortho-rectification and mosaiced by Aerometrex using high end commercial photogrammetric software and Aerometrex proprietary software.

2.2.2. LiDAR data

Light detection and ranging (LiDAR) provides 3D information to delineate spectrally similar, but structurally distinct cover types (e.g., lawn/trees) in the land use classification. LiDAR data was provided in LAS1.2 format by Mapland the South Australian Government spatial data library and retailer. The LiDAR data was collected with a nominal Average Point Spacings of 8 pts per sqm by a RIEGL VQ-780i LiDAR sensor. Vertical accuracy on average was +/- 0.1m (1 Sigma). Imagery was captured on the 23rd of April 2018 – additional meta data available online (Location SA, 2022a).

2.2.3. Cadastral data

Cadastral data provides the spatial boundary of each residential property, along with building attributes and ownership as of January 2021. Full valuation cadastre dataset was provided by Department for Infrastructure and Transport and accessed via the University of Adelaide. Meta data available online (Location SA, 2022b).

2.2.4. Rainfall data

Climate data was accessed from an Australian climate database, SILO (Queensland Government, 2021). As rainfall is not uniform across the study site, gridded monthly rainfall data was collected from 1900-2021 at a 25m spatial resolution. From this dataset, three rainfall scenarios are used in the model, the 0.25 decile, 0.5 decile, and 0.75 decile, hereby referred to as Dry, Typical and Wet respectively (Fig. 3).

2.2.5. Crop data: land and water requirements

Crop yields and crop irrigation data used in this research were collected by citizen scientists involved in the South Australian Edible Gardens Project, which has been published in the literature, and made publicly available for reuse (Csortan et al., 2020b, Csortan et al., 2020a). Hume et al. (2021) analyzed raw data from the Edible Gardens Project to calculate that a typical home in Adelaide, which is comprised of 2.5 people, would require 21.30 m^{-2} to achieve dietary self-sufficiency of vegetables. The model reflects a polyculture food garden with a range of vegetable crops that are currently produced by urban growers in Adelaide (Hume et al., 2021). Further description of Csortan et al. (2020b) data collection and analysis performed by Hume et al. (2021) is included in the supplement (S1).

The garden modelled by Hume et al. (2021) is used in the present study as the growing space required for self-sufficiency of vegetables via urban agriculture, and will simply be referred to as 'the modelled edible garden' hereafter. In addition to achieving complete dietary

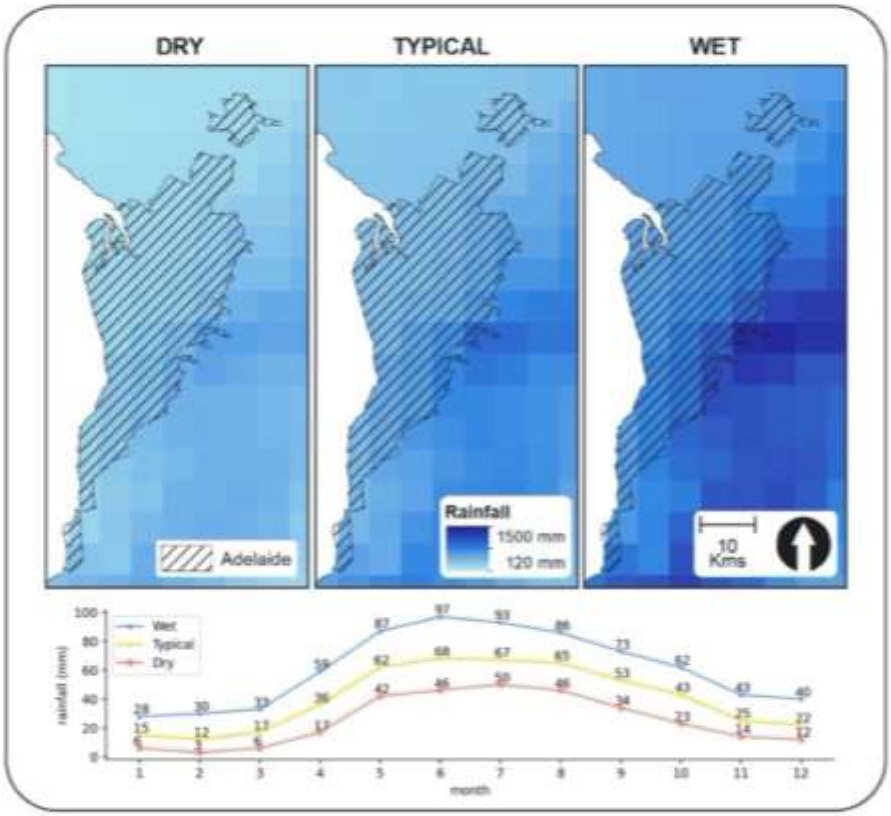

Fig. 3. Gridded annual rainfall of each of scenario (25m resolution), and average rainfall across Adelaide in each month for each scenario (line graph).

self-sufficiency of vegetables, two partial self-sufficiency scenarios are modelled. These are: 1, producing 75% of the RVI and 2, producing 50% of the RVI. It is assumed that the two scenarios would use 75% and 50% of the land required for complete self-sufficiency, respectively.

To accurately measure irrigation applied by growers from the Edible Gardens Project, Csortan et al. (2020a) provided participants with individual water meters. The irrigation applied (L m^{-2}) by urban growers is used in conjunction with the results of Hume et al. (2021) to calculate total water requirements for the modelled edible garden. For this study, only gardens with crops labelled as "Vegetables" or "Vegetables & Herbs mixed" from the Edible Garden Project were included.

2.3. Land use classification

Land use classification was undertaken using LiDAR and Spectral imagery and a number of remote sensing techniques (Fig. 4). The workflow of the land use classification is outlined in Fig. 4. Sections 2.3.1–2.3.5 outline each stage of the land use classification in detail. All stages of the land use classification were carried out using ArcGIS (ESRI, 2011).

2.3.1. LiDAR classification

The LiDAR point cloud was used to identify rooftops (building footprint) and delineate ground cover vegetation (lawn) from other vegetation - as identified by spectral reflectance.

Points were classified using tools from the ArcGIS (ESRI, 2011) 3D Analyst toolset. Ground points were classified using the conservative method within the *Classify LAS Ground* tool. The *Classify LAS Building* tool was applied to remaining unclassified points. In classifying buildings, minimum thresholds were applied to reduce misclassification. The building minimum height was set to 1m, and area set to 2m^{-1}. All remaining points were classified by height into 3 classes: 0 - 0.5m, >0.5 - 1.5m, >1.5 - 50m and >50m, to aid with the classification of vegetation when combined with spectral data *(see Section 2.3.3)*.

The final point cloud dataset was resampled as a raster with 0.5m resolution. Post classification processing involved a majority filter with eight neighbors, and a boundary clean using a *half majority* threshold. The resulting raster is the LiDAR Output (Fig. 4).

2.3.2. Spectral classification

The four-band spectral imagery was subject to a supervised

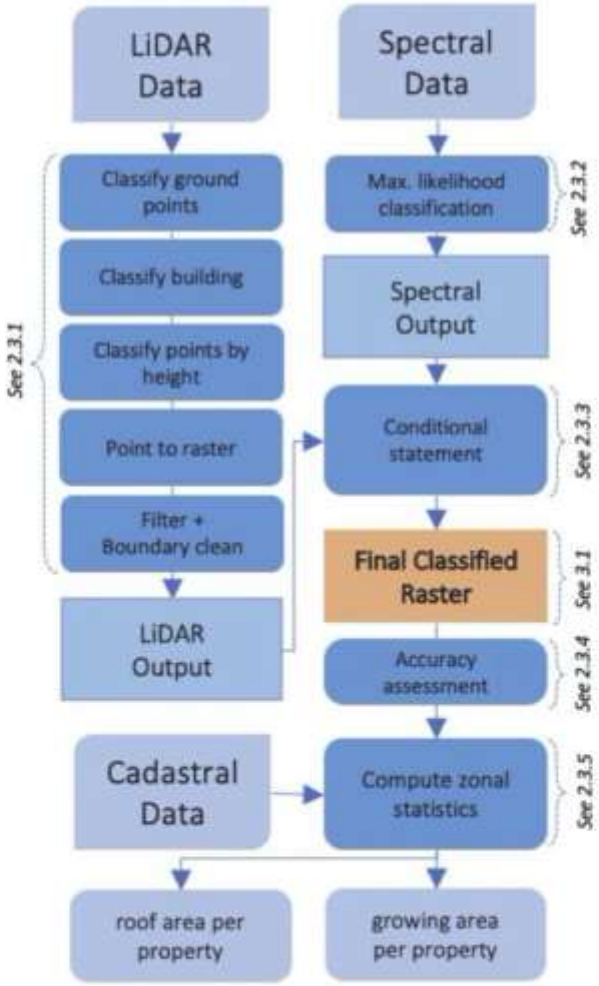

Fig. 4. Flow chart of remote sensing land use classification methods.

maximum likelihood classification. Pixels were classified into the following classes: *Vegetation*, *Sealed Surface*, *Shadow*, or *Water* (e.g., Swimming Pool). The maximum likelihood classification relies on training classes determined by the practitioner and uses Bayes' theorem of decision making to assign each pixel to the class with which it has the highest probability of being a member (ESRI, 2019). The resulting raster, with a 0.5 m pixel resolution is the Spectral Output (Fig. 4).

2.3.3. Conditional statement

The LiDAR Output and Spectral Output were analyzed concurrently to delineate ground cover vegetation (lawn) from other vegetation, using an *if-else* principle. Pixels identified as *Vegetation* in the Spectral Output and *less than 0.5 m high* in the LiDAR Output were classified as *Lawn*. *Buildings* were identified from the LiDAR Output and were given that classification unless pixels were also identified as *Vegetation* in the Spectral Output, in which case they were classified as *Tree* (to account for vegetation overhang and provide a more conservative estimate of rainwater capture). All remaining pixels were classified as per the Spectral Output: Non-lawn vegetation = *Tree*, *Sealed* = *Sealed*, *Shadow* = *Shadow*, and *Water* = *Water (Supplement S2)*.

2.3.4. Accuracy assessment

An accuracy assessment was carried out on the final model. A total of 500 random points were created to satisfy the commonly cited recommendation of creating 50 points per class (Congalton, 1991). Points were randomly stratified between classes to reflect the large disparity between class sizes. Each point was individually ground-truthed via high-resolution Google Earth imagery.

The error matrix resulting from the accuracy assessment details the overall accuracy (proportion of pixels correctly identified), Kappa coefficient, as well as producer's accuracy and user's accuracy of each class. The Kappa coefficient is regarded as the most robust measure of accuracy as it considers the probability of a pixel being allocated to the correct class purely by chance (Foody, 2002). The producer's accuracy quantifies pixels that have been ground-truthed as *Class X* but not identified as such in the model, i.e., false negative. Conversely, user's accuracy quantifies the proportion of pixels that have been classified as *Class X* by the model but contain other cover types in the reference data i.e., false positives.

2.3.5. Computation of zonal statistics

The *Compute Zonal Statistics* tool in ArcGIS was used to calculate the area of different land-use classes (*Lawn* and *Building*) within residential blocks. Only residential blocks that were completely contained within the study site and between 50 m^2 and 1500 m^2 in size were included. The size restrictions were implemented to exclude very small or large blocks. These may reflect errors in the data record due to poorly recorded boundary divisions, alternatively, they may contain many residencies but haven't been classified as such (e.g., residential care facilities), or, in some cases, represent individual privately owned carparking spaces.

The input layers for the *Compute Zonal Statistics* tool were pixels classified as Lawn, and pixels classified as Building. However, buildings smaller than 16 m^{-2} (approximately the size of a single car garage) were not included in the computation. This decision was based on the impracticalities of capturing water from small, free standing sheds and garages.

2.4. Calculation of crop water requirements

Crop water requirements in this study are based on the actual water applied by urban growers in the Edible Garden Survey (Csortan et al., 2020a). The irrigation applied during the Edible Garden Survey was summed with rainfall during the same time period to calculate total water footprint of crops. Therefore, water footprint (L m^{-2}) was calculated according to *Eq. 1*.

$$\text{WaterFootprint}_m = irr_m + Rainfall_m \quad (1)$$

where:

WaterFootprint (*L* m^{-2}) is the total water received by crops during month *m*,

irr (*L* m^{-2}) is the irrigation applied to crops in month *m*, and

Rainfall (mm) is the average rainfall in month *m*, within the corresponding period.

The total irrigation required for the modelled edible garden was calculated from the crops' water footprint, minus the amount of water the crops receive from rainfall. Therefore, the total irrigation (L), (and target volume of harvested rainwater) in each month was calculated according to *Eq. 2*.

$$Irrigation\ Tot_{m,s} = \left(WaterFootprint_m - Rainfall_{m,s}\right) * Gowing\ Area \quad (2)$$

where:

Irrigation Tot (L) is the irrigation applied to the modelled edible garden in month *m*, under rainfall scenario *s*,

WaterFootprint (*L* m^{-2}) is as described *Eq. 1* for month m,

Rainfall (mm) is the average total rainfall for month *m*, under rainfall scenario *s*, and

Growing area (m^2) is the area of edible garden big enough to sustain self-sufficiency, taken from Hume et al. (2021).

2.5. Rainwater capture potential at each property

As rainfall was not uniform across the entire study site, the gridded rainfall data (25m resolution) was related to the spatial data via a spatial join for each property in the study site.

The water capture potential of each property was calculated according to *Eq. 3*.

$$Capture_{m,s} = rainfall_{m,s} * RoofArea * e \quad (3)$$

where:

Capture (L) is the maximum potential rainwater capture for month *m*, and rainfall scenario *s*,

Rainfall (mm) is the rainfall for month *m*, and rainfall scenario *s* in the property's location (Fig. 3).

RoofArea (m^2) is the roof area of each individual property as calculated according to the land use classification outlined in Section 2.3, and

e is the efficiency of the rainwater harvesting system, accounting for losses due to leakage and spilling, which is assumed to be 0.95 (Imteaz et al., 2011).

The analysis also considers storage of water in residential water tanks for extended periods of time. The parameters of the storage model are determined by the results of earlier modelling. i.e., deficiencies in water capture compared to irrigation requirements.

2.6. Spatial analysis for water storage and food production

Area of lawn in each block (resulting from the land use classification (*Section 2.3*)) is considered the maximum space available for food production and water storage tanks. Lawn is preferred as it is immediately available for food production without significant costs (Hume et al., 2021). However, other residential land cover may also be suitable, e.g., paved courtyards. As the land-use classification does not allow for delineation of courtyards from incompatible sealed surfaces (i.e., driveways), lawn area is used as a conservative estimate of total land available for conversion to edible garden.

2.7. Sensitivity to urban densification

While the spatial data was captured at a single date, the cadastral dataset provides the date each house was built (original structure, not

subsequent additions and extensions). This has been used to identify trends in self-sufficiency potential. Additionally, published trends in residential developments have been analyzed to project house and block size into the future. While house sizes in Adelaide have remained relatively constant, there has been a trend towards smaller blocks with less yard (Hume et al., 2021). Between the years 2011 and 2020 the average yard space shrunk by 20 m^2 (Hume et al., 2021). This data is used to model self-sufficiency potential of homes if lawn area shrunk by a further 20 m^2 in the coming decade.

2.8. Economic viability

To assess the economic viability of irrigation with rainwater tanks, we compare the cost of harvested rainwater, to the cost of irrigating with the equivalent volume of mains water. We calculated the net present value (NPV) of the rainwater tanks, taking into consideration the upfront cost of the tanks, the annual water savings (i.e., cost of mains water), and the discount rate of the investment. The *discount rate* describes the interest that the investment may yield over its lifetime. In other words, money is worth more in the future than it is worth today, and the discount rate factors that into the calculations. NVP is calculated according to Eq. 4

$$\mathrm{NPV} = \left(\sum_{t=1}^{n} \frac{savings_t}{(1+r)} \right) - \text{initial investment} \quad (4)$$

where:

NPV is the net present value,
n is the number of years,
t is the time in years,
r is the discount rate, set at 0.035
savings is the sum of the annual water savings at time t, and
initial investment is the upfront cost of the water tank

3. Results

3.1. Growing space and roof area

The land use classification was successfully used to determine growing space and roof area in each property. The final classification had a Kappa coefficient of 0.89 (Table 1). Values between 0.81–1 indicate almost perfect model agreement (Landis and Koch, 1977). Fig. 5 shows the classified data, in addition to a subset of the study area to highlight the distribution of classes across a non-representative cluster of properties. The final data set contained 491,712 residential properties. The median lawn area across the individual blocks was 71 m^{-2} and the median roof area was 200 m^{-2}. Across all properties, the total residential lawn area was 4588 ha.

3.2. Crop water requirements

Over one year, an edible garden of 21.30 m^{-2}, which is large enough to satisfy the vegetable requirements of a typical household, would require a total of 50.42 kL of water from both irrigation and rainfall (2.37 kL m^{-2}, and approximately 0.15 kL kg $produce^{-1}$ y^{-1}). Therefore, in a Typical year, 40.91 kL of irrigation are required, whereas in a Dry year, 44.41 kL of irrigation are required. The typical (median) rainwater capture from a single house in a Typical rainfall year is 90.67kL. In a Dry year, the typical capture volume is 54.43 kL.

Taken together, the median annual rainwater capture at a single property more than satisfies the water requirements of crops for self-sufficiency, even in a Dry year. However, given the variation in the population of houses, there are some properties that fall below the line (Fig. 6). In a Typical Rainfall year, 88% of properties can capture enough rainwater to satisfy annual irrigation demand. In a Dry year, 66% of properties can capture enough rainwater to satisfy annual irrigation demand.

While the annual volume of rainwater runoff more than satisfies the modelled volume required for irrigation at a typical (median) property, there is a temporal lag between when water can be captured, and when it is required. Fig. 7 compares the rainwater capture potential of properties in Wet, Typical, and Dry years, and contrasts this to irrigation requirements in a monthly timestep. The lines of the graph indicate the volume of water required to irrigate crops for self-sufficiency. In the Wet scenario, less irrigation is required as crops receive more water from rainfall.

The boxes in Fig. 7 highlight the variation in potential rainwater capture across properties in the study site. In wetter years, the potential for rainwater capture is greater than in drier years.

Properties that sit above the line in Fig. 7 will be able to irrigate using only rainwater captured in that month. Properties that sit below the line will require water stored from wetter months where there is a surplus. The first step in exploring the opportunities and limitations of long-term water storage is to better understand the cumulative deficit of rainwater capture compared to irrigation requirements.

In the modelled Dry year, there are five consecutive months (November – March, late Austral Spring to early Autumn) where irrigation outside of what can be captured that month will be required. In the Typical rainfall year there are four consecutive months (December – March, Austral Summer- early Autumn), and in the Wet year there are three (December – February, Austral Summer)(Supplement Fig. S3). The total water deficit across these time periods are 29 kL, 20 kL, and 9 kL for dry, typical, and wet months respectively. As such, three storage scenarios are explored: 25 kL of storage, 30 kL of storage, and 40 kL of storage. These storage scenarios only consider the need for long term water storage from one month to another. They do not include rainwater that is harvested and used immediately on the garden.

3.3. Self-sufficiency analysis

The three different storage scenarios, 25 kL, 30 kL and 40 kL, are compared in terms of their physical attributes and the resulting feasibility of self-sufficiency (Table 2). To analyze the feasibility of self-sufficiency, two key metrics are used. The first is the percent of the

Table 1
Error matrix of land-use classification.

Classified data	Reference data Lawn	Tree	Sealed	Shadow	Water	Building	Total	UA
Lawn	108	10	2	0	0	0	120	0.90
Tree	0	55	0	0	0	1	56	0.98
Sealed	7	7	141	3	0	2	160	0.88
Shadow	0	0	0	4	0	0	4	1.00
Water	0	0	0	0	3	0	3	1.00
Building	0	0	1	1	0	77	79	0.97
Total	115	72	144	8	3	80	422	
PA	0.94	0.76	0.98	0.50	1.00	0.96		

Overall accuracy = 0.92, Kappa coefficient = 0.89.

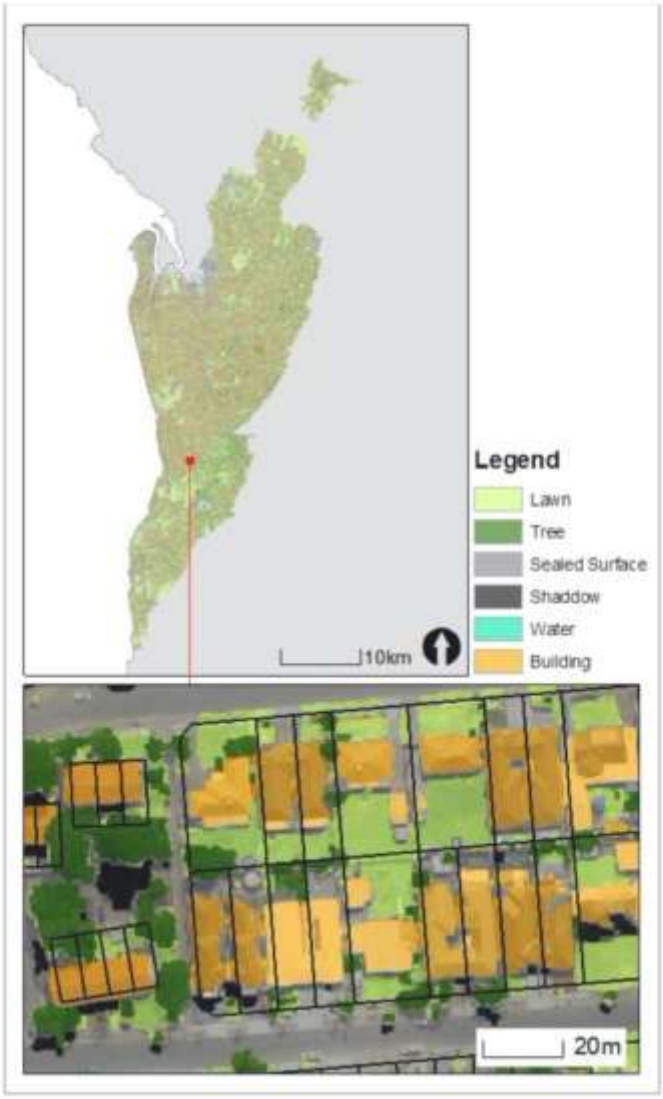

Fig. 5. All classified land area, and a subset of study area displaying results of land use classification.

population that could achieve total self-sufficiency (i.e., enough lawn area to produce the RVI and house the RWH tank, and enough roof area to harvest sufficient rainwater). The second metric is the percent of lawn area that a typical block would need to dedicate to RWH and food gardening in order to achieve self-sufficiency. In all scenarios, the lawn area in a typical block is 71m^2.

Of all properties sampled, 65% are able to achieve self-sufficiency in at least one of the scenarios. In a Dry year, most properties would require the largest storage scenario of 40 kL to achieve self-sufficiency (meeting the household RVI, and irrigation with only harvested rainwater). Most properties could achieve self-sufficiency in a Typical rainfall year with 25 kL of long term storage, only 11% of the population could achieve self-sufficiency in a Dry year using 25 kL of storage. Increasing from the 25 kL to the 40 kL tank size would require an additional 4 m^2 water storage space. With a 40 kL storage tank, a Typical property would need to dedicate 52% of the land that is currently lawn to growing food and storing water for self-sufficiency.

Free standing houses made up 73% of residency types across the population, but accounted for around 90% of the properties that could achieve self-sufficiency in each scenario (Supplement Table S4). None of the scenarios account for overflow, or potential water-use elsewhere in the house. A visualisation of the monthly water volume of a 30 kL tank attached to a typical property is displayed in Fig. 8. There are one or two months where water is most likely to become limiting. This occurs during late Summer or early Autumn (Fig. 8). Self-sufficiency is most attainable in Winter and Spring.

3.3.1. *Partial self-sufficiency*

Two other production scenarios are explored – growing 50% of the RVI, and growing 75% of the RVI (Table 3). In a Dry year, 71% of properties could produce 50% of the RVI, using 15 kL of water storage. Alternatively, 71% of properties could produce 75% of the RVI in a Dry year using 40 kL of water storage. The inclusion of the extremely large storage scenario illustrates the diminishing effect that increased storage

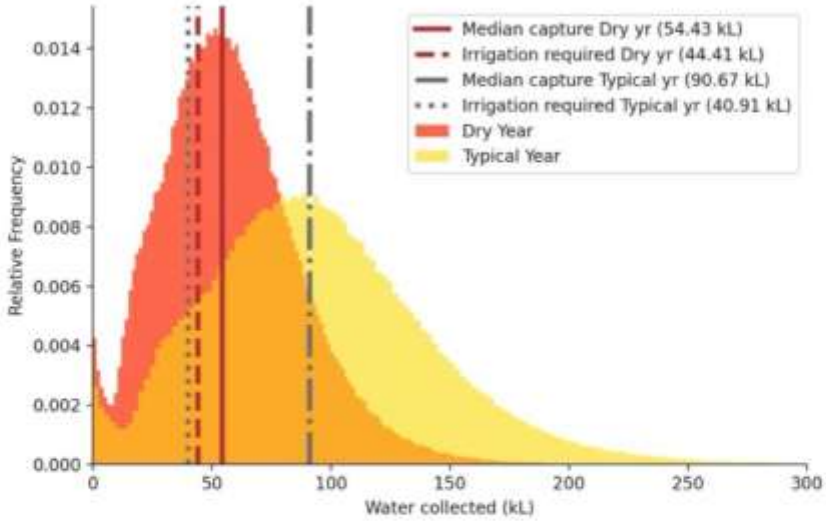

Fig. 6. Annual rainwater capture (kL) from each property in a modelled Dry year and modelled typical year, median values displayed, and the annual irrigation required to sustain dietary self-sufficiency of vegetables. X axis maximum is set to 300 kL.

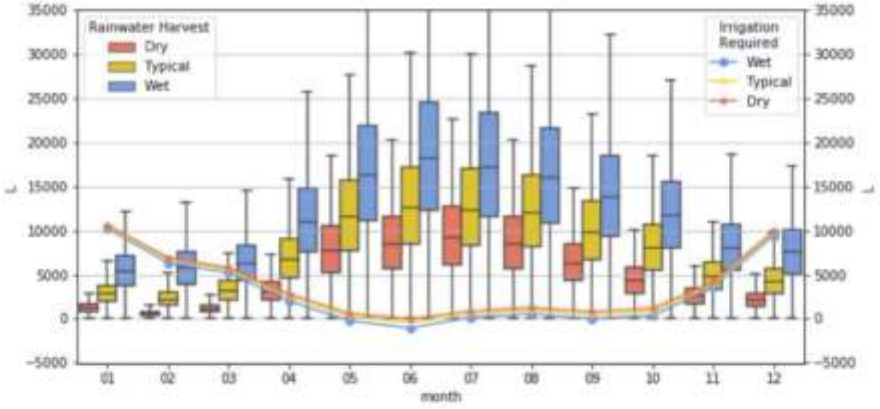

Fig. 7. Monthly rainwater capture potential of properties in Wet, Typical, and Dry years (boxes), compared to the volume of water required for irrigation in in Wet, Typical, and Dry years (points).

Table 2
Comparison of three different water storage scenarios: 25 kL, 30kL, and 40 kL.

Tank size	Footprint tank (m)*	Footprint for SS (tank + garden)	% Lawn area used**	% pop. able to achieve SS***			Reference
				Dry	Typical	Wet	
25 kL	11.22	32.52	45%	11%	60%	65%	Os Tanks (2021)
30 kL	12.07	33.37	47%	44%	65%	66%	ASC Tanks (2021)
40 kL	15.21	36.51	52%	65%	65%	65%	(Southern Tanks, 2021)

* all tanks are 2.8m in height.
** "% Lawn area used" refers to the % of lawn area at a typical property (71m2) that the modelled edible garden and associated storage tank would occupy.
*** "% pop. able to achieve SS" refers to the refers to the percentage of the population able to achieve dietary self-sufficiency of vegetables for 2.5 people (a function of a property's lawn area), as well as capture and store enough rainwater for irrigation of the modelled edible garden (a function of roof area, rainfall and lawn area)

has on self-sufficiency potential.

3.4. Sensitivity to urban densification

Houses built more recently are likely to have less lawn space and a higher proportion of the block dedicated to building (Fig. 9a). Houses built more recently are also less likely to be able to achieve self-sufficiency in a Typical rainfall year, using 30 kL of storage (Fig. 9b). Properties that can achieve self-sufficiency under this scenario have a mean year of construction in 1977. Those than cannot achieve self-sufficiency have a mean year of construction in 1997. However, there is no statistical difference in the year built between the two groups.

Fig. 8. The monthly water volume of a 30 kL storage tank, as it is recharged from a Typical Property.

Table 3
Outcomes for partial self-sufficiency in two difference cases: growing 50% of the RVI and growing 100% of the RVI, with three different water storage scenarios.

	tank size	footprint tank (m)[a]	footprint for SS (tank + garden)	% Lawn area used[b]	% pop. able to achieve SS[c]			reference
					Dry	Typical	Wet	
50%	5 kL	3.25	13.9	20%	2%	52%	73%	(Oz Tanks, 2022)
	10 k	5.19	15.84	22%	27%	71%	75%	(Oz Tanks, 2022)
	15 k	7.31	17.87	25%	71%	74%	74%	(Oz Tanks, 2022)
75%	15 kL	7.31	23.29	33%	4%	59%	70%	(Oz Tanks, 2022)
	25 kL	11.22	27.2	38%	62%	69%	69%	(Oz Tanks, 2022)
	40 kL	15.21	31.19	44%	71%	71%	71%	(Southern Tanks, 2021)

[a] tank heights vary.
[b] '% Lawn area used' refers to the % of lawn area at a typical property (71m2) that the modelled edible garden and associated storage tank would occupy.
[c] '% pop. able to achieve SS' refers to the percentage of the population able to achieve dietary self-sufficiency of vegetables for 2.5 people (a function of a property's lawn area), as well as capture and store enough rainwater for irrigation of the modelled edible garden (a function of roof area, rainfall, and lawn area).

Houses built before 1950 are excluded as the dataset does not account for building extensions, which are most likely to occur on holder houses.

If the average yard space were to shrink by 20 m^2 (as has occurred in the previous decade), 58% of properties could achieve self-sufficiency using a 40 kL of water storage in a Dry year - a reduction of 6 percentage points from current day figures. The tank and edible garden combined would take up 72% of the lawn space at a Typical property (Supplement Table S5).

3.5. Economic viability

The economic viability of the scenarios outlined in Section 3.3, and *Table 2* (producing 100% of the RVI) are examined. All currency values are in Australian Dollars (AUD). The local purchase price of a 40 kL, 30 kL, 25 kL is $5,500, $3,900, and $3,650, respectively (Oz Tanks, 2022, ASC Tanks, 2021, Southern Tanks, 2021). This does not include installation, delivery, or maintenance costs. Alternatively, South Australian households pay between $1.966 kL^{-1} and $3.040 kL^{-1} for potable mains water (SA Water, 2021). Therefore, the NPV of each scenario is a range between a lower and upper value, depending on the cost of water. The discount rate in all scenarios is set at 3.5% (0.035). Since 1990, the average interest rate in set by Reserve Bank of Australia (2022) has been 3.9%. However, it has declined more in recent years (Reserve Bank of Australia, 2022).

To calculate the economic rate of return on the initial investment, the NPV (Section 2.8) of each scenario after 25 years is calculated (*Table 4*). For example, after purchasing a 30 kL storage tank, and irrigating the modelled edible garden for 25 years under Typical rainfall conditions, the return on the initial investment is between -$1845.74 AUD and -$2,571.49AUD. In other words, the gardener is between $2,928.51 and $3,654.26 AUD out of pocket. None of the scenarios returned a NPV of less that 0 in the first 25 years.

4. Discussion

While urban agriculture may address a range of urban challenges (Artmann and Sartison, 2018), judicious management of inputs is required to ensure its environmental sustainability (McDougall et al., 2019). This research investigated the potential for residential urban agriculture to provide dietary self-sufficiency while addressing issues of water scarcity by irrigating crops using captured rainwater. The work directly addresses existing knowledge gaps surrounding roof harvested rainwater for use in urban agriculture and food production (Amos et al., 2018). In this case study, a typical property was found to contain enough available land (i.e. lawn) to provide dietary self-sufficiency of vegetables, as well as capture and store adequate rainwater for irrigation, even in a modelled Dry year.

4.1. Self-sufficiency of crops and water

Using remote sensing, we were able to take into account the heterogeneity of urban spaces, and provide a city-wide overview of self-sufficiency potential. In a Dry year, approximately two thirds of properties (65%) could grow the RVI, irrigating with only harvested rainwater. An additional 6% of the population can achieve *partial* self-sufficiency – growing 75% of the RVI (Table 3). Additionally, in a Dry year, 71% of properties could produce half of the RVI, using just 15 kL of storage. In this final scenario, the combined land area dedicated to both

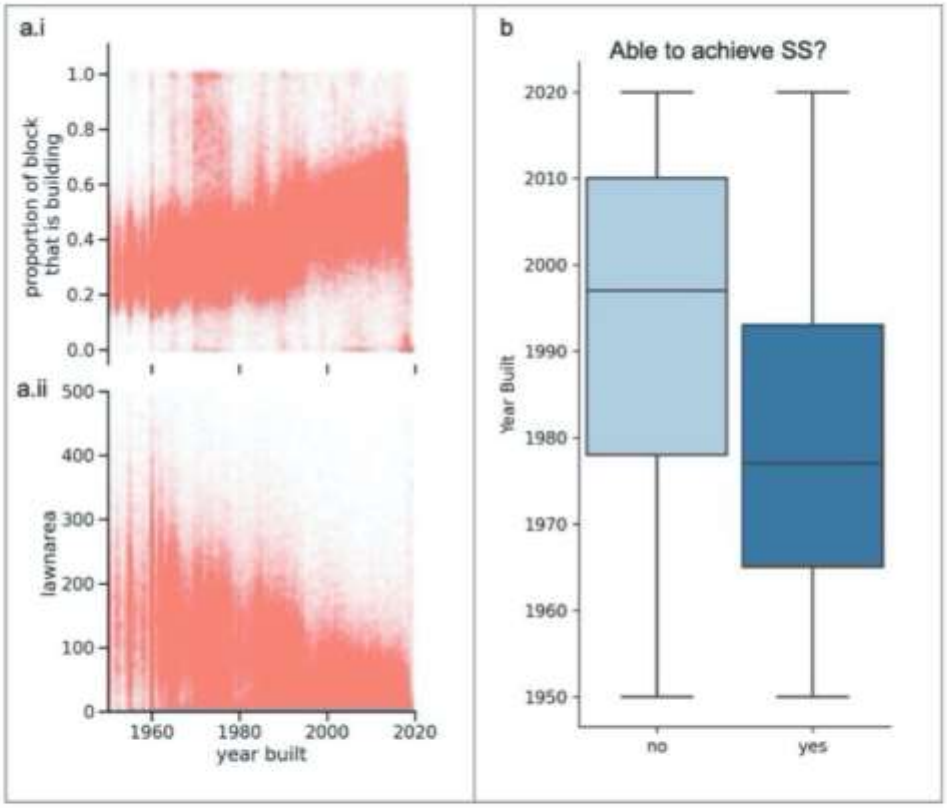

Fig. 9. Panel a: lawn area in each property (a. i) and proportion of residential block that is building (a.ii) versus year that the home was built. Both scatter plots' X-min is set to 1950 and displayed with an alpha of 0.002. Y-max in a.ii set to 500 m. Panel b: distribution of year built across properties that are able to achieve Self-sufficiency in a Typical rainfall year, with 30 kL of storage (yes = 65%, no = 35%).

Table 4
The NVP (net present value) (AUD) of rainwater harvesting after 25 years. 'Annual savings' is the alternate cost - irrigating with mains water. 'Initial investment' is the purchase price of the tank.

tank size	*investment*	*Dry*[*]		*Typical*[**]		*Wet*[***]	
		lower	*upper*	*lower*	*upper*	*lower*	*upper*
25 kL	$3,650.00	-$2,224.28	-$1,445.43	-$2,321.49	-$1,595.74	-$2,483.50	-$1,846.26
30 kL	$3,900.00	-$2,474.28	-$1,695.43	-$2,571.49	-$1,845.74	-$2,733.50	-$2,096.26
40 kL	$5,500.00	-$4,074.28	-$3,295.43	-$4,171.49	-$3,445.74	-$4,333.50	-$3,696.26

[*] In a Dry year 44 kL of irrigation is required, resulting in annual savings between $ 86.504 -$133.76.
[**] In a Typical year 41 kL of irrigation is required, resulting in annual savings between $80.60 - $124.64.
[***] In a Wet year 36 kL of irrigation is required, resulting in annual savings between $70.78 - $109.44

the RWH system and edible food garden would be 17.87 m^2 (Table 3). This could be appealing to many residents for space saving and efficiency, especially considering most Australians eat no more than 2 serves of vegetables per day (Australian bureau of Statistics, 2018) (RVI is five serves per day). Furthermore, partial self-sufficiency might be a more achievable goal for densely populated cities where space is limiting.

As outlined in the introduction, until now, research relating specifically to RWH for use in urban agriculture has been limited (Amos et al., 2018), however, studies do exist. McDougall et al. (2020) calculated the annual water resources available to urban agriculture in Sydney, Australia, a city with a population density of 9,301 km^{-2} (City of Sydney, 2022). Roof-top capture exceeded water demand for 100% self-sufficiency of vegetables (measured in 20 community scale blocks, housing approx. 400 people in each) (McDougall et al., 2020). The current work provides further insights by considering temporal lags between rainwater capture and use of water for irrigation, as well as the allocation of land for water storage. Furthermore, McDougall et al. (2020) considered all street verges, some public miscellaneous spaces and private yard space, as available land for food production. The present study only considers residential lawn area, and investigates self-sufficiency potential of households rather than cohorts of the population. At this scale, resources would be immediately available to residents with less capital investment and no legislative interventions, however, as the resources are not shared, more people might miss out. Compared to the work of McDougall et al. (2020), this investigation focused on relatively water stressed conditions. Sydney has an annual rainfall of 1,083 mm. By comparison, the modelled Dry year equated to

298 mm of annual rainfall, and the Typical year equated to 486 mm of annual rainfall. For context, arid zones are defined as having less than 250 mm of annual rainfall. Taken together, the current work has added necessary complexity to previous modelling of water capture for urban agriculture and considered relatively water stressed conditions to better understand the limitations of RWH for urban food production.

This research considered partial self-sufficiency (achieving 50% or 75% of the RVI), rather than seasonal self-sufficiency (achieving self-sufficiency at certain times of the year). During the months of late Summer or early Autumn, water is most likely to become limiting (Fig. 8). An additional cohort could produce the RVI relying on harvested rainwater where possible, and supplementing with mains water as necessary. Given that self-sufficiency without RWH has been the focus of other work (Hume et al., 2021), it has not been investigated here. However, seasonal self-sufficiency could still deliver positive outcomes for food production and water management.

At the city scale, the volume of water that could be harvested and directed towards urban agriculture is significant. South Australia's combined mushroom and vegetable growing industry extracted 26.2 GL of water and used an additional 9.6 GL of distributed water in the last financial year (Australian Bureau of Statistics, 2021a). In a Dry year, the volume of water harvested by properties that are able to achieve self-sufficiency is 14.19 GL, which is 54% of the water currently extracted by the vegetable growing industry in South Australia. This number does not include properties that could achieve partial self-sufficiency. Taken together, RWH integrated with urban agriculture not only has significant food producing capabilities, but could also provide adaptive food production methods that reduce water abstractions.

4.2. Water use efficiency

Until recently, a limitation in advancing this area of research has been a lack of reliable data around water use in low tech urban agriculture. In lieu of reliable irrigation data specific to urban gardens, previous researchers have relied on commercial irrigation rates (McDougall et al., 2020), or modified crop growth models such as CROPWAT (Ward et al., 2014). In the present study, actual urban agriculture irrigation rates are sourced from a citizen science project (Csortan et al., 2020a). The average irrigation rate in a Typical year, calculated from Csortan et al. (2020a), is 2.04 kL m^{-2}. By comparison, the annual water-use of vegetable crops grown commercially in the Adelaide Greater Region in 2019-2020 was 1.21 kL m^{-2} (Australian Bureau of Statistics, 2021b). This indicates that using observational data provides a great advantage for modelling a more realistic scenario for urban growers.

Urban food gardens typically produce higher yields than their commerical counterparts (McDougall et al., 2019, Csortan et al., 2020a). It may be assumed that higher yields require increased irrigation. The modelled edible garden used in this study consumed 147 L kg produce $^{-1}$ y^{-1}, which is within the range of values modelled for vegetable crops grown in Adelaide using CROPWAT (Ward et al., 2014). However, urban gardens differ from commercial farms in ways that affect their water consumption. These include: the area of land under production, production methods, crops grown, and planting arrangements (Pollard et al., 2018b, Ward et al., 2014). Small scale urban gardens are also managed by community groups, households, or individuals. These groups have been observed to over-apply other key agricultural inputs (Salomon et al., 2020). Given that there was no effort in the current research to maximize the water use efficiency of gardens, RWH capabilities for urban agriculture may be even greater than reported in this research if crop water use was optimized. Other measures could also be taken to improve water use efficiency. These could include water sensitive crop selection or more water efficient production methods. Hydroponics, or aquaponics would both improve water use efficiency of crops (Sambo et al., 2019). However, there are educational, economic, and social barriers hindering their uptake (Pollard et al., 2017). Taken together, there remains some uncertainty for optimal water use in urban gardens which limits the accuracy of RWH modelling. Nevertheless, the chief benefit of the observational data used in this study is that is it reflects the actual behaviors of urban growers. As such, the model represents the most *realistic* scenario.

It is important to note that the modelled edible garden in this research replaces residential lawn. While the current study does not attempt to calculate water use on lawns, given the climate of the study site, it is assumed lawns are irrigated to various degrees. As such, the discussion on the environmental sustainability and water use of edible gardens cannot be had without acknowledging the diversion of water from residential lawn.

4.3. Economic viability

As outlined in Section 3.5, the value of purchasing a rainwater tank, rather than potable mains water for irrigation does not result in an economic return on investment (Table 4). Most houses in this case study require between 30 kL and 40 kL of storage to achieve self-sufficiency in Dry years. The purchase price of a 40 kL tank in South Australia is around $5,500 AUD (Table 4). Alternatively, the economic cost of irrigating the modelled edible garden with mains water in a Typical rainfall year would be between $81 – $125, depending on other household water use. If all of the water captured at a Typical home could be used to displace mains water elsewhere in the home, the cost of water tank would still not be paid off in 25 years (Supplement Table S6). This is the approximate lifespan of a rainwater tank, and does not consider any other costs associated with miniatous or repairs.

While there are a suite of reasons why people engage in urban farming, those that grow a significant amount of food at home (at least 30% of their total diet), are low income earners (Donati and Rose, 2020). The model of self-sufficiency assumes that households are able to finance the materials for the edible garden and RWH system. As such, the cost of engaging a RWH system may exclude the very populations that are most likely to take up urban farming. Of course, those who can afford the upfront cost of rainwater harvesting may do so for altruistic reasons. In this case economic *return on investment* may be irrelevant.

There are additional complexities to the economic modelling that were not considered in this research. More thorough investment calculations on rainwater tanks have been performed elsewhere. Such investigations highlight the importance of tank size selection to optimize the return on the initial investment (Khastagir and Jayasuriya, 2011). In this study, none of the tanks resulted in a return on investment, however, larger tanks were the least profitable (Table 4). There is a body of literature that supports this finding. In general, larger tanks are less profitable than smaller tanks due to the large capital investment (Hajani and Rahman, 2014, Khan et al., 2021, Preeti and Rahman, 2021).

As mentioned, the modelled NPV does not consider increasing price of water. A recent report by Infrastructure Australia (2017) warned that the cost of water to Australian households could more than double in the next 20 years, which would improve the benefit to cost ratio. However, Khan et al. (2021) investigated the Cost Benefit Ratio (CBR) of RWH systems and found that water prices would need to increase three times before the modelled RWH system would become financially feasible in the Adelaide region.

The financial viability of RWH systems depend on the water requirements, climate conditions, rainfall variability and price of water, and are therefore variable across different regions (Hajani and Rahman, 2014, Khan et al., 2021, Preeti and Rahman, 2021). Rainwater harvested specifically for urban agriculture may also require water quality monitoring which would have associated ongoing costs (Deng, 2021). The model assumes vegetable production replaces (in part or full) urban lawn, the economic costs of irrigation for self-sufficiency must also be compared to the economic cost of lawn irrigation. Finally, in this case, RWH was considered alone and not in conjunction with the other

elements of urban agriculture and food production.

4.4. Urban densification

This study quantified a decrease in capacity for sustainable urban agriculture over time, due to patterns of urban densification. Properties with newer builds contained less lawn area on average, and a had a larger proportion of the block occupied by building, compared to properties containing older houses. Additionally, newer builds were less likely to meet the requirements for self-sufficiency compared properties containing older houses. A densification model was built to simulate further losses of potentially productive land (see Section 2.7). Under the densification model, most houses were still able to achieve self-sufficiency in the largest water storage scenario (40 kL). However, the proportion of households able to achieve self-sufficiency dropped 6 percentage points, from 65% to 58%. Under the densification model, 73% of the lawn area would be occupied by the edible garden and storage tank.

If we are to keep the door open to greater urban agriculture and urban resilience, retaining open space in residential developments is an important issue. Martellozzo et al. (2014) identified population density as the most significant factor in determining cities' self-sufficiency potential. As outlined in Section 2.1, Adelaide's average population density is 1,620 people km^{-2} and reaches up to 3,000 people km^{-2} in areas (Australian Bureau of Statistics, 2020a). By comparison, Richardson and Moskal (2016) analyzed the self-sufficiency potential of homes in Seattle, USA. Seattle has a city-wide population of 3429 people km^{-2}, and is one of the most densely populated cities in the United States, with some neighborhoods reaching 14,644 people km^{-2} (Statistical Atlas, 2018). Converting residential lawn into urban food gardens would feed less than 1% of Seattle's population on an entirely vegetarian diet, which includes nuts and legumes (Richardson and Moskal, 2016). However, Richardson and Moskal (2016) modelled their results using national agricultural yields, rather than yields achieved by urban growers.

The comparison between Adelaide and Seattle highlights the unique opportunity for urban agriculture to provide meaningful amounts of food in low to medium density cities. The heterogeneity within cities, along with ambiguous urban/peri-urban boundaries make it hard to categorize cities according to their relative density. However, by one estimate around 10% of the world's cities house less than 2,000 people km^{-2} (Demographia, 2020). Therefore, this case study relates to many other parts of the world including suburban areas associated with large cities. Our modelling revealed that 36.51 m^{-2} of space is required for both self-sufficiency of vegetables and 40 kL of water storage. This could be used as a guide for policy makers and community members alike seeking to retain open space for food provisioning - yet similar yields and RWH must also be achieved.

4.4. Assumptions, limitations, and future directions

One foundational assumption to this work is that RWH and urban agriculture contribute to environmental sustainability. RWH can provide many benefits to the environment including decreased fresh water abstractions, and a reduction of polluted stormwater (Richards et al., 2013). However, RWH systems, particularly those with pumps, also consume significant amounts of power (Amos et al., 2018). In France, a domestic RWH system was found to be worse than the local municipal drinking water in terms of their comparative environmental impact (Vialle et al., 2015). In contrast, in Washington, USA, a commercial scale RWH system outperformed the municipal mains water in areas of energy demand and global warming potential, albeit, only slightly (Ghimire et al., 2017). As such, larger scale RWH systems may be more energy efficient than the domestic RWHs systems modelled here. Unsurprisingly, RWH has been shown to substantially out-perform mains water in areas of eutrophication and freshwater withdrawals (Ghimire et al., 2017) Taken together, the environmental impact of RWH likely depends on both the system set up, and the value placed on different services - which could differ across location and time. A thorough life cycle assessment (LCA) of RWH for urban agriculture will be an important next step in this area of research. The LCA should also consider the energy saved by eliminating food miles and food waste along the production line (Dorr et al., 2021).

As with most models that attempt to predict and understand complex systems, this study makes use of averaged and generalized data. The analysis only considered the total amount of land available, not the distribution of that land on the property. We assumed that residential lawn is the only land use available for both farming and water storage, however, sealed surfaces such as courtyards could also house storage tanks. This study also does not consider competing uses of lawn area and does not consider overflow management of tanks during high rainfall periods. This study does not consider houses in Adelaide that already have rainwater tanks installed, making the results of the study conservative. Potential limitations relating to labor shortages and the need for education and training are not discussed in this paper as they have been explored in previous research (Hume et al., 2021, McDougall et al., 2020). Issues of contamination and pollutants in harvested rainwater have been discussed elsewhere in the literature (Deng, 2021). While such issues were not addressed in this model, doing so will be an important next step in this area of research. We also limited our study to private land, however, there remains the opportunity to expand both food production and RWH into public areas. Finally, while the research tested many variables relating to rainfall, storage capacities, level of self-sufficiency, and even land use changes over time, it did not test sensitivities to yield. Yield and productivity in relation to self-sufficiency have been examined else-where (Hume et al., 2021). As such, only one yield was modelled, with the assumption that this is an achievable best-case scenario. Further justification can be found in Section 2.2.5.

5. Conclusions

In this study we investigated the potential for residential urban agriculture to provide dietary self-sufficiency, while mitigating against and adapting to water scarcity, by irrigating crops using captured rainwater. We found that 65% of residential properties contain enough available land to provide dietary self-sufficiency of vegetables, as well as capture and store adequate rainwater for irrigation, even in a Dry year. This scenario requires 40 kL of water storage volume for year-round irrigation. Properties that achieve self-sufficiency could divert 14.19 GL of harvested rainwater water from entering stormwater waste streams, and displace a significant portion of fresh water extracted by the vegetable growing industry.

We examined two potential threats to self-sufficiency through urban agriculture: the economic viability of RWH, and loss of yard space from urban densification. None of the RWH systems examined yielded an economic return on investment within 25 years. The potentially prohibitive cost of RWH systems may exclude members of the population most likely to rely on urban agriculture for food security. Additionally, projected urban densification will result in a 20 m^2 reduction of yard space in the coming decade. The proportion of properties able to achieve self-sufficiency in the densification sub-model was 6 percentage points lower than today's estimate. However, partial self-sufficiency may be a viable, space saving, alterative. In a Dry year, 71% of households could grow half the recommended vegetable intake using a 15 kL storage tank. The footprint of both the garden and the tank would be less than 18 m^2.

Taken together, RWH integrated with urban agriculture not only has significant food producing capabilities, but could also help to adapt food production methods to issues of increasing water scarcity. Retaining open space in residential developments will be important if we are to keep the door open to urban agriculture in the future.

Declaration of Competing Interest

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

Data Availability

The authors do not have permission to share data.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.scs.2022.104249.

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Chapter 4
Backyard farming: Who benefits and who misses out? A Geospatial Analysis.

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Name of Principal Author (Candidate)	Isobel Hume		
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Overall percentage (%)	90%		
Certification:	This paper reports on original research I conducted during the period of my Higher Degree by Research candidature and is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in this thesis. I am the primary author of this paper.		
Signature		Date	22/05/2023

Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

i. the candidate's stated contribution to the publication is accurate (as detailed above);

ii. permission is granted for the candidate in include the publication in the thesis; and

iii. the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

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Abstract

Urban populations in the Global North face a rising problem of food security, which can be addressed through urban agriculture. Backyard farming can improve access to nutritious foods and reduce household expenditure on food. Previous research has shown that backyard farming can contribute significantly to dietary self-sufficiency, however, targeting urban agriculture initiatives effectively requires mapping and auditing available land to understand the distribution across the population. Here, geospatial analysis is used to investigate the relationship between household dietary self-sufficiency potential and economic risk factors for food insecurity, using a case study of Adelaide, Australia. Areas with at-risk populations for food insecurity were found to have high household self-sufficiency potential, illustrating that at-home urban agriculture could be an effective strategy to alleviate food insecurity. However, the bespoke spatial analysis of backyard farming potential also exposes specific areas where vulnerable communities lack access to privately owned land for food production. Such analyses identify opportunities and constraints within each suburb that can inform targeted and policy interventions to support the most vulnerable members of the community. While population density was found to be an indicator of self-sufficiency potential, policymakers should perform additional analysis to avoid overlooking opportunities and limitations for specific populations.

Highlights

1. Backyard farming and self-sufficiency can alleviate food insecurity in urban areas
2. Geospatial analysis reveals high household self-sufficiency potential in at-risk areas
3. Yet, areas where vulnerable communities lack access to land are also identified
4. Targeted policy interventions are required to support vulnerable communities
5. Geospatial analysis is a powerful tool in urban agriculture research and planning

1. Introduction

Despite living in areas of abundance, urban communities in the Global North are facing food insecurity due to a range of socio-economic and environmental factors (Gulyas & Edmondson, 2021). The recent COVID-19 pandemic has highlighted the fragility of global supply chains and the need for urban areas to become more self-sufficient in their food supply (Gulyas & Edmondson, 2021). With the world's population becoming increasingly urbanised, and climate change threatening traditional agricultural practices, urban agriculture has emerged as a promising way to produce fresh, healthy food within cities. While there is growing evidence that urban agriculture can provide multifunctional benefits to urban communities (Evans et al., 2022; Weidner et al., 2019), further research is required to understand how these benefits are distributed within cities. Additionally, there is a need to understand how urban agriculture can be integrated into urban planning and development policies to best support vulnerable communities (Sarker et al., 2019).

1.1 Food security definitions

Ensuring food security is a crucial issue in the 21st centaury. Food security occurs when all people have access to enough safe and nutritious food to maintain an active and healthy lifestyle and are able to access that food in socially acceptable ways (Bowden, 2020). In short, food insecurity is the opposite. Food insecurity can manifest in different ways and exists on a spectrum of severity. This could include an inability to afford or access enough food, skipping meals, reducing food portions or quality, relying on low-

cost, low-nutrient dense foods, or experiencing prolonged periods of hunger and malnutrition (Miller & Li, 2022). Similarly, food justice goes beyond availability and affordability of an individual's dietary needs to address the underlying cause of food insecurity. Food justice considers the systemic inequalities that effect food security, and emphasises the importance of healthy, environmentally sustainable, and culturally appropriate food as a fundamental right (Murray, et al., 2023).

1.2 The state of food security in Australia

While there is no consistent measure of food security at the population level in Australia (i.e., census reporting), reporting indicates that it is on the rise. Previous estimates showed that 4% to 13% of the Australian population experience food insecurity (Bowden, 2020), while the most recent Foodbank Hunger Report indicates that one third of Australian households are food insecure (Miller & Li, 2022). Food insecurity is caused by confounding factors working in combination, yet the most common drivers in Australia are increased cost of living and reduced income (Miller & Li, 2022). When a household's financial resources are reallocated due to changes in economic circumstances, nutritious and diverse diets are often sacrificed (Seivwright et al., 2020). While low income is the most reliable predictor of household food insecurity, higher income does not grant immunity (Kleve et al., 2018). In addition to low-income earners, Indigenous peoples, geographically isolated individuals, culturally diverse groups, the young, single-parent households, older individuals, and people experiencing homelessness are particularly at risk of food insecurity in Australia (Bowden, 2020).

1.3 The role of urban agriculture in food security

Cultivating edible gardens enhances dietary diversity and improves the food security of low-income households (Diekmann et al., 2020; Donati & Rose, 2020). Strategically applied, edible gardening can reduce the household expenditure on food, particularly as supermarket prices rise. Edible gardening also provides peace of mind and dignity to those struggling with food insecurity (Donati & Rose, 2020), and can fulfil needs for culturally appropriate foods (Diekmann et al., 2020). By producing food within the urban boundary, urban agriculture also increases food justice. In particular, community participation helps to democratise food systems while empowering individuals to grow healthy and sustainable food (Murray et al., 2023). Diversifying food systems to include urban agriculture can help to provide vulnerable communities more reliable access to diverse food groups. Urban agriculture is not a panacea for food insecurity, particularly in areas of extreme food insecurity associated with poverty and extreme poverty (Du Toit et al., 2022). It is, however, recognised as an effective strategy towards achieving targets pertaining to dietary diversity, nutrition, food justice, and (FAO et al., 2020).

Backyard farming is by far the most common form of food production practiced by urban growers (Donati & Rose, 2020). Ghosh (2021) contends that home gardening can connect individuals to land and nature, while also improving health and resilience. Soil-based vegetable production requires little investment or infrastructure, making it accessible to households with enough suitable land (Salomon & Cavagnaro, 2022). Vegetables are an efficient crop to grow in urban areas as they are nutrient dense and economically valuable (Weidner et al., 2019). However, residential vegetable production

receives far less attention than community gardens or more high-tech solutions within urban agriculture (Kirkpatrick & Davison, 2018; Taylor & Lovell, 2014). Despite being underrepresented in the literature, recent work demonstrates that residential land in low-to-medium density urban areas can provide a significant resource for food production and contribute to dietary self-sufficiency (Hume et al., 2021, 2022).

To gain a better understanding of the importance of urban agriculture across demographics, following the COVID-19 lockdowns, a survey was conducted among Australian gardeners (Donati & Rose, 2020). The survey, which had over 9,000 respondents and represented 62% of Australian postcodes, provided valuable insights into the experiences of individuals and a cross-section of the Australian population. The results showed that households within the lowest income bracket (< \$50K year $^{-1}$) were most likely to grow a significant portion of their food at home (30% or more of their diet). This further validates the significance of urban agriculture for food security in low-income populations.

One of the key self-reported barriers to urban gardening is access to land (Donati & Rose, 2020). With 43% of survey respondents citing access to land as a barrier to expanding their gardening activities a key recommendation of the Pandemic Gardening survey was to 'make much more land available for edible gardening' (Donati & Rose, 2020). However, in contrast to this Hume et al. (2021) conducted a spatial analysis of residential properties in Adelaide, Australia, and found that 93% of residential properties sampled had enough lawn area to grow the recommended vegetable intake (five serves of vegetables per person per day). Additionally, further research found that

65% of homes could grow the recommenced vegetable intake and irrigate while also capturing and storing adequate rainwater for irrigation (Hume et al., 2022). However, the situation is not as promising in densely populated cities. For example, a case study of Seattle, USA, by Richardson and Moskal (2016) found that converting residential lawn into urban food gardens would feed less than 1% of the population following a complete vegetarian diet that includes nuts and legumes. Seattle has a city-wide population of 3,429 people km^{-2}, compared to Adelaide's 1,620 people km^{-2} (Hume et al., 2022; Richardson & Moskal, 2016). While other factors such as climate, crop selection, and achievable yields also influence self-sufficiency potential, research supports the notion that density is a primary driver of a location's ability to achieve self-sufficiency (Martellozzo et al., 2014).

1.4 Aims

Access to resources for urban agriculture is a critical area that needs investigation. Who has access to the land that is required for successful urban farming, and how is it distributed across the population? This information is important for policy makers and urban planners who seek to support vulnerable communities. To target urban agriculture initiatives effectively, available land resources need to be mapped and audited (Donati & Rose, 2020). Therefore, this research aims to:

1. analyse the spatial distribution of self-sufficiency potential within the city and identify areas with high and low potential for dietary self-sufficiency through urban agriculture,

2. investigate the relationship between self-sufficiency potential and key indicators of vulnerability in relation to food security such as income and housing stress, and
3. provide insights into targeted policy interventions that can support and improve food security and food justice for vulnerable populations.

Adelaide (Australia) is used as a case study as in our earlier work on this topic (Hume et al., 2021, 2022).

2. Methods

This analysis takes key data from secondary sources to compute the self-sufficiency potential, and vulnerability of Statistical Areas across Adelaide's urban extent (Figure 6). Two primary datasets are analysed, 1): The classification of the self-sufficiency potential of homes in Adelaide, performed by Hume et al. (2022), and 2): 2021 Census of Population and Housing General Community Profiles (Australian Bureau of Statistics, 2022a), which provides demographic data such as income levels. Geospatial analysis is used to map the areas at risk of food insecurity, and areas of high self-sufficiency potential (Figure 7).

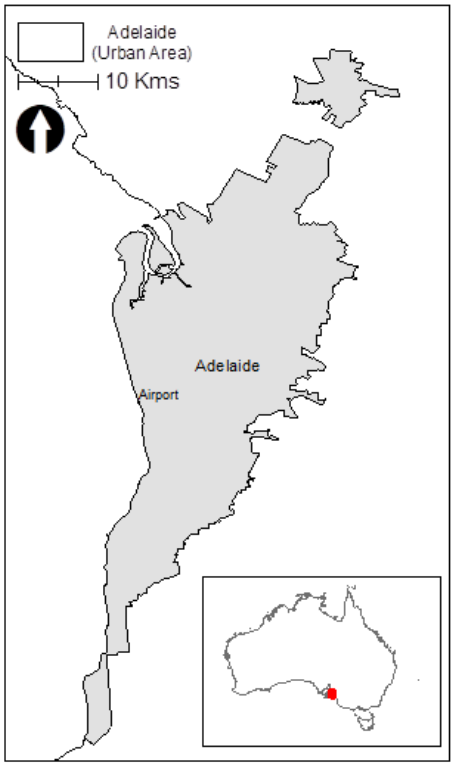

Figure 6 location and boundary of the Adelaide Urban Area.

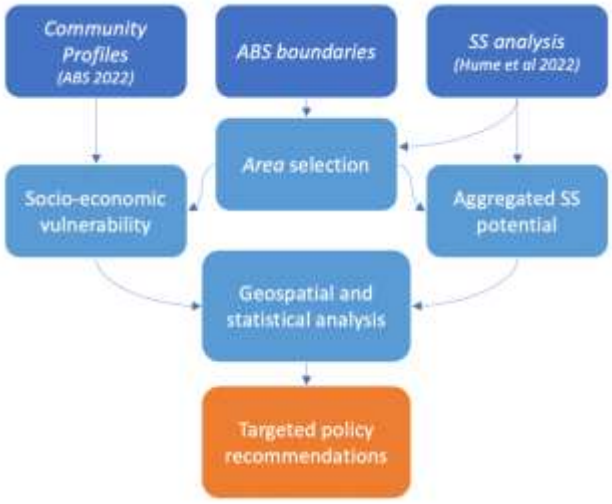

Figure 7 Overview of research workflow

2.2 Statistical boundaries

Data is analysed at the resolution of Statistical Areas Level 2 (SA2s). SA2s are geographic areas defined by the Australian Bureau of Statistics (ABS). They are designed to represent a community that interacts together socially and economically. Within cities, they are often based on gazetted suburbs. However, as suburb size is variable, smaller suburbs may be clustered, or extremely large suburbs broken up. SA2s generally have a population range of 3,000 to 25,000 persons, with an average population of approximately 10,000 persons. Whole SA2s can be combined to form larger statistical areas, such as Statistical Areas Level 3 (SA3). In cities, SA3s approximate Local Government Areas (LGAs) (Australian Bureau of Statistics, 2021). This paper will hereby refer to SA2s as *Areas.*

2.3 Area Selection

Geospatial analysis was conducted using GeoPandas (version 0.11.1) in Python (version 2.7) (Python Software Foundation). The properties analysed by Hume et al. (2022) are located across 97 Areas. However, many of the boundaries of these Areas extend beyond the boundary of Adelaide's defined Urban Area (Australian Bureau of Statistics, 2017). As such, only 83 Areas located wholly within a 2.2km buffer of Adelaide's Urban Centre ware included in this analysis (Figure 8).

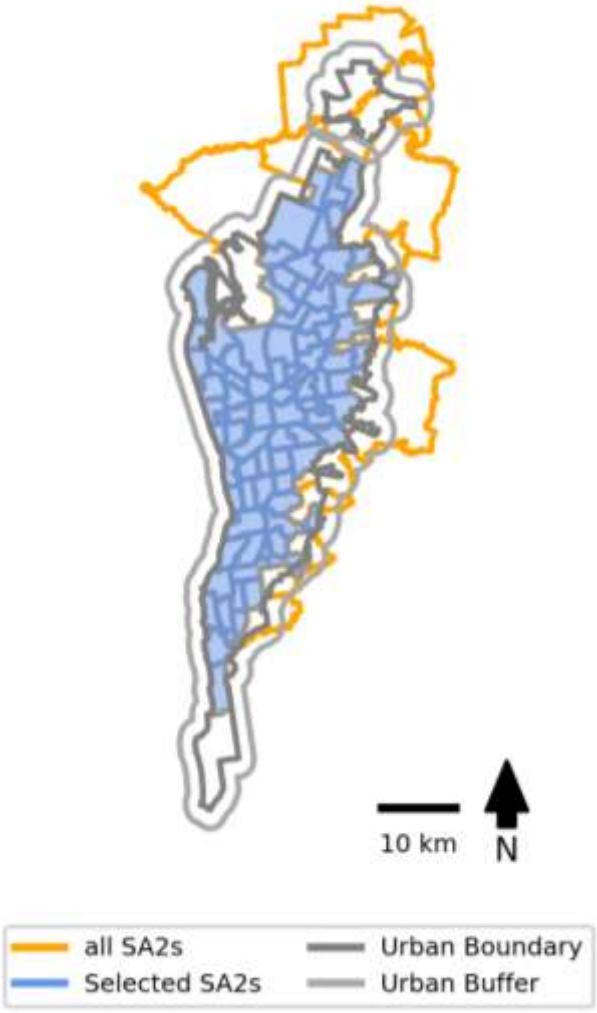

Figure 8 elected SA2s in Adelaide's urban extent. Only SA2s which are fully, or mostly contained within the urban extent are included in the study. SA2s that extend beyond 2.2km buffer of the urban boundary are excluded.

2.1 Measure of self-sufficiency potential

Previous studies have analysed the self-sufficiency potential of individual homes in Adelaide (see (Hume et al., 2022) and (Hume et al., 2021). Using a binary classification of a home's self-sufficiency (SS) potential, this study aggregates the results of Hume et al. (2022) at the Area level by calculating the ratio individual properties that are able to achieve dietary self-sufficiency of vegetables and those that are not.

A brief summary of Hume et al. (2022) is as follows: High-resolution imagery was combined with productivity and irrigation data to quantify the self-sufficiency potential of 491,712 individual homes in Adelaide. To meet the criteria for self-sufficiency, a single property must have been found to have enough available land to grow the households' recommended vegetable intake, while capturing and storing adequate rainwater for irrigation using a 40 kL storage tank (typical rainfall year). The recommended vegetable intake was five serves per person, per day (1 serve of vegetables is 75 g), and the average house size was 2.5 people. Urban agriculture was assumed to yield 5.08 kg of vegetables m^{-2} (Csortan et al., 2020; Hume et al., 2022).

The SS potential of each Area was calculated as the portion of properties that are able to achieve SS. Areas were then assigned a ranking from 0-6 depending on their relative rate SS potential. Values of 0, 2, 4 or 6 were assigned to Areas in the fourth, third, second, and first quartile (Q4, Q3, Q2, Q1), respectively, with higher scores being assigned to Areas with the lowest SS potential.

2.4 Socio-economic vulnerability

Data for this study was obtained from the Australian Bureau of Statistics Census Community Profiles (Australian Bureau of Statistics, 2022a), which provided information on the socio-economic status of households in Australia. An Income Stress Score and a Housing Stress Score were assigned to each Statistical Area based on the relative level of household weekly income and household mortgage stress, respectively, using variables

available in the Census Community Profiles. There variables are reliable predictors of household food insecurity (Bowden, 2020; Miller & Li, 2022).

2.4.1 Income stress

The most recent Household Income and Wealth survey of Australia reveals that the 40th percentile of weekly household income is $1470, and the 20th percentile is $583 per week (Australian Bureau of Statistics, 2022b). This approximates to the $1500 and $650 per week income brackets in the Census data, which are used to identify low income households and very low income households, respectively.

Areas were then assigned a ranking from 0-3 depending on their relative rate of low income families and very low income families. Values of 0, 1, 2 or 3 were assigned Q1, Q2, Q3 and Q4, respectively (Table 1). The ranking values for Low and Very Low were then summed to create an *Income Stress* score. For example, if an Area sits in 50%-75% quantile (Q3) for relative rates of low income households, and Q4 for relative rates of very low income households, its Income Stress score would be 5. By design, the Income Stress score is weighted towards rates of very low income families.

$$Income\ Stress\ Score = low\ income\ score + very\ low\ income\ score$$

Table 1 Summary statistics (Q1, Q2, Q3, Q4) of the proportion low income and very low income households in each Area.

	Low income (% Total)	very Low income (% total)
Q1	39%	14%
Q2	46%	17%
Q3	51%	20%
Q4	72%	35%

2.4.2 Household financial vulnerability

To provide a more comprehensive picture of financial stress, income is also compared to outgoing costs. A common example of this is housing affordability stress (Australian Housing and Urban Research Institute, 2019). If housing costs become too high relative to household income, it leads to housing stress, which can impact spending on other essential items such as food. Households are defined as financially vulnerable if they are both low income, and spend a significant portion of their income on housings costs (Australian Housing and Urban Research Institute, 2019). The underlying assumption is that households with high incomes who choose to spend a larger portion of their income on housing do not face the same negative consequences and are still able to afford other essentials.

Financial stress is measured as the proportion of household income spent on housing costs. Data were collected from the ABS. Therefore, the two relevant variables are: Housing Cost (proportion of weekly income spent on Housing cost), and Average Household Income. Average Household Income is reported by the ABS, and Housing Stress is calculated as follows:

$$housing\ cost = \frac{(rent\ stress + mortgage\ stress)}{2}$$

Where:

$$rent\ stress = \frac{weekly\ rent}{weekly\ household\ income}$$

And

$$mortgage\ stress = \frac{weekly\ mortgage\ repayments}{weekly\ household\ income}$$

Each Area was then assigned a ranking from 0-3 depending on their relative Housing Cost and Average Household Income. (0, 1, 2 or 3 for Q1, Q2, Q3 and Q4 of Housing Cost, respectively, and 0, 1, 2 or 3 for Q4, Q3, Q2 and Q1 of Average Household Income, respectively) (Table 2).

Table 2 Summary statistics (Q1, Q2, Q3, Q4) of household weekly income and housing cost (% of household income spend on rent/mortgage repayments) for the 83 Areas in the study site.

	Household income (weekly)	Housing cost (% of income)
Q1	1689.50	21%
Q2	2017.00	22%
Q3	2273.50	24%
Q4	2854.00	47%

Housing Stress is the sum of Housing Cost score and Average Household Income scores, resulting in a value from 0-6.

$$Housing\ Stress = Housing\ Cost\ score + Household\ Income\ rank$$

4. Results and Discussion

Part 1: The spatial distribution of SS potential and physical characteristics

The first aim of this research was to analyse the spatial distribution of self-sufficiency potential within the city and identify areas with high and low potential for dietary self-sufficiency through residential urban agriculture. Areas closer to the city centre appear to have lower SS than Areas further away from the city centre. This was confirmed by analysing the correlation between *Distance to Adelaide's city centre (CBD)* and SS potential (Spearman correlation coefficient of 0.54 (p=<0.001) (Figure 9). However, properties close to the coast (Southern border) also appear to have lower SS potential (See supplement Figure S1, Appendix F). Across all Areas, the Q1, median, Q3 and maximum rates of SS were 0.57, 0.66, 0.79 and 0.97, respectively. Twenty-two Areas were in the lowest quantile of SS potential. Of those Areas, the most heavily represented SA3s (approximate LGAs) were Charles Sturt (4), Norwood - Payneham - St

Peters (3), and Port Adelaide – West (3), which are all in close proximity to the city centre (Figure 9).

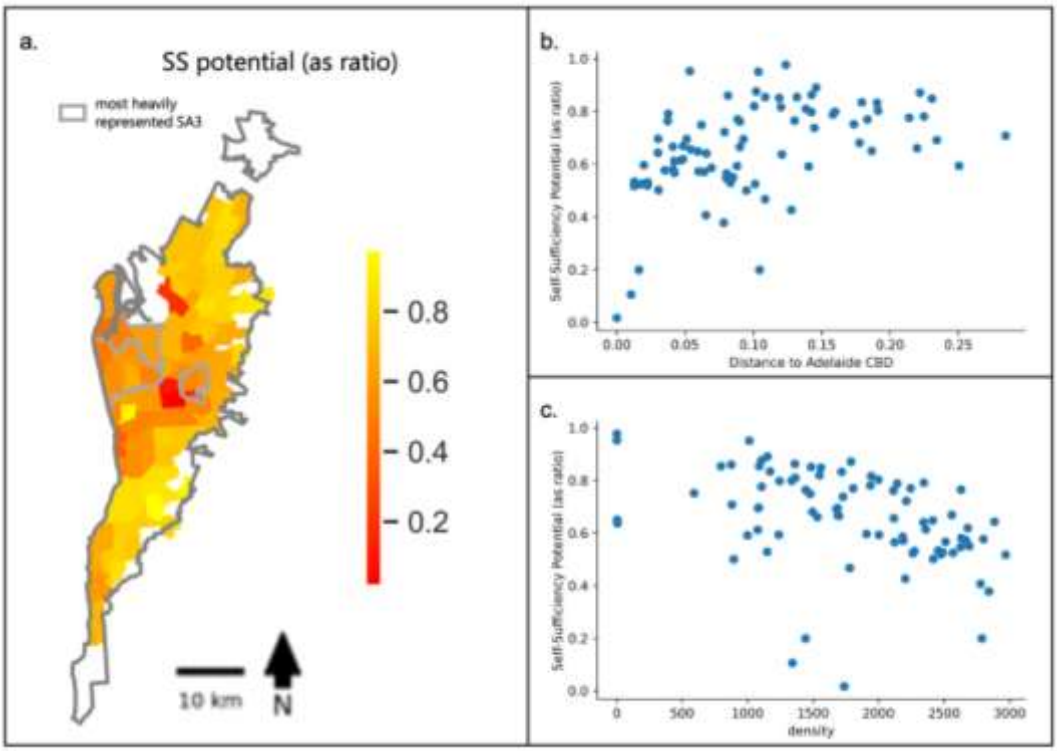

Figure 9 – Distribution of Self-sufficiency (SS) potential across Adelaide: Panel a. illustrates the geographical spread of SS potential across Areas within Adelaide, while panels b., and c. examine the correlation between SS potential and Distance to CBD, and population density, respectively.

Population density is negatively correlated with the SS potential (Spearman correlation coefficient of -0.495 (p=<0.001). The five most densely populated Areas are: Unley – Parkside, Prospect, Glenelg, Nailsworth – Broadview and Norwood. By comparison, Adelaide is the 43rd most densely populated Area, and North Adelaide the 60th. Areas with low population density and low SS potential may have opportunities for urban agriculture to occur on public land. For example, Adelaide and North Adelaide are considered low density due to the expansive public parklands (to further illustrate this

point, and provide further context, arial images have been included in supplementary material Figure S2, Appendix F).

To understand the drivers of SS potential it is useful to consider the physical characteristics of each area. For example, Adelaide city centre had the lowest self-sufficiency potential, with just 2% of properties meeting SS criteria. The average number of people per household is less than the citywide average – at 1.8 people per household. Flats and apartments make up 58.9% of the total dwellings, with free standing, semi-detached and row houses making a combined 41.4% (Australian Bureau of Statistics, 2022a).

Three other Areas had less than 20% of properties meeting the criteria for SS. These are North Adelaide, Norwood, and Mawson Lakes (Globe Derby Park). Flats and apartments make up 28%, 22%, and 14% in North Adelaide, Norwood and Mawson Lakes (Globe Derby Park), respectively (Australian Bureau of Statistics, 2022a).

Previously, Martellozzo et al. (2014) found that population density is the most critical factor in determining the potential for urban agriculture to contribute to a country's self-sufficiency in food production. Here, we show that while population density is an important factor in determining the SS potential of smaller Areas within a city, using it as the only metric may lead policy makers to mis-understand the limitations and opportunities for specific populations.

Part 2: Distribution of housing and financial stress

4.2 Distribution of income stress

In order to investigate the relationship between self-sufficiency potential and key indicators of vulnerability, we will first examine the spatial distribution of housing stress and financial stress. The portion of low income households ranges from 30% to 72% within each Area, while the portion of very low income households ranges from 8% to 35% in each Area. Low income households and very low income households are concentrated in the North and South of Adelaide's Urban extent. The CBD also has relatively high rates of low income households and very low income households compared to the surrounding inner city Areas. There is no statistical difference in the SS potential of Areas with different income stress scores. However, Areas with the lowest and highest rates of income stress appear to have greater SS potential than Areas in the middle range of income stress (Figure 10).

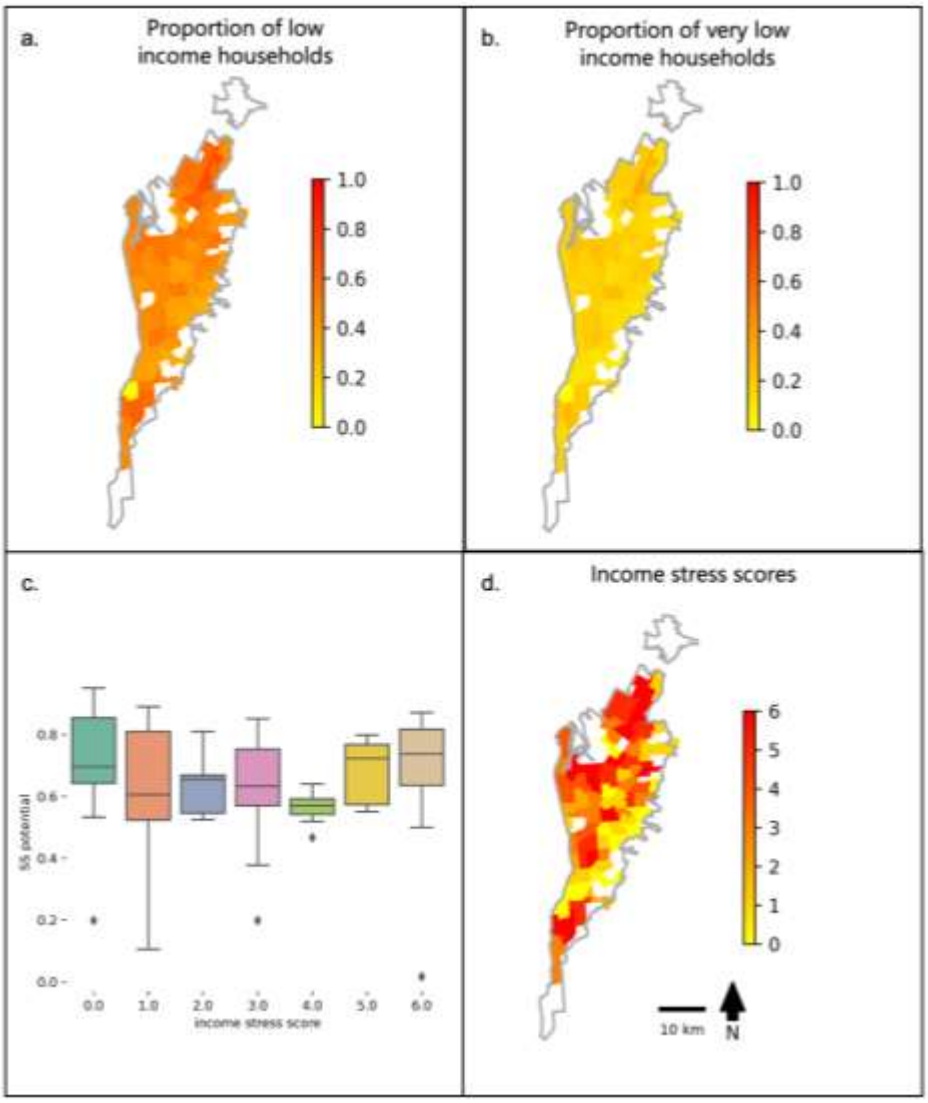

Figure 10 – Income stress and self-sufficiency potential of Areas within Adelaide: Geographical distribution of low income households (panel a.), very low income households (panel b.), and income stress scores (Areas with a score of 6 have the highest rates of low income households and very low income households(panel d.); and the distribution of SS potential across Areas of varying income stress scores (panel c.).

4.3 Distribution of Housing stress

Housing stress considers the proportion of outgoing household costs in addition to income. Housing stress is spatially related to income stress and follows a similar distribution when compared to SS potential (Figure 10, Figure 11). However, housing

stress appears to be more randomly distributed than income stress (Figure 11). While Areas located in the North and South of Adelaide's urban extent are still highlighted, now hot spots of housing stress also arise in inner city Areas.

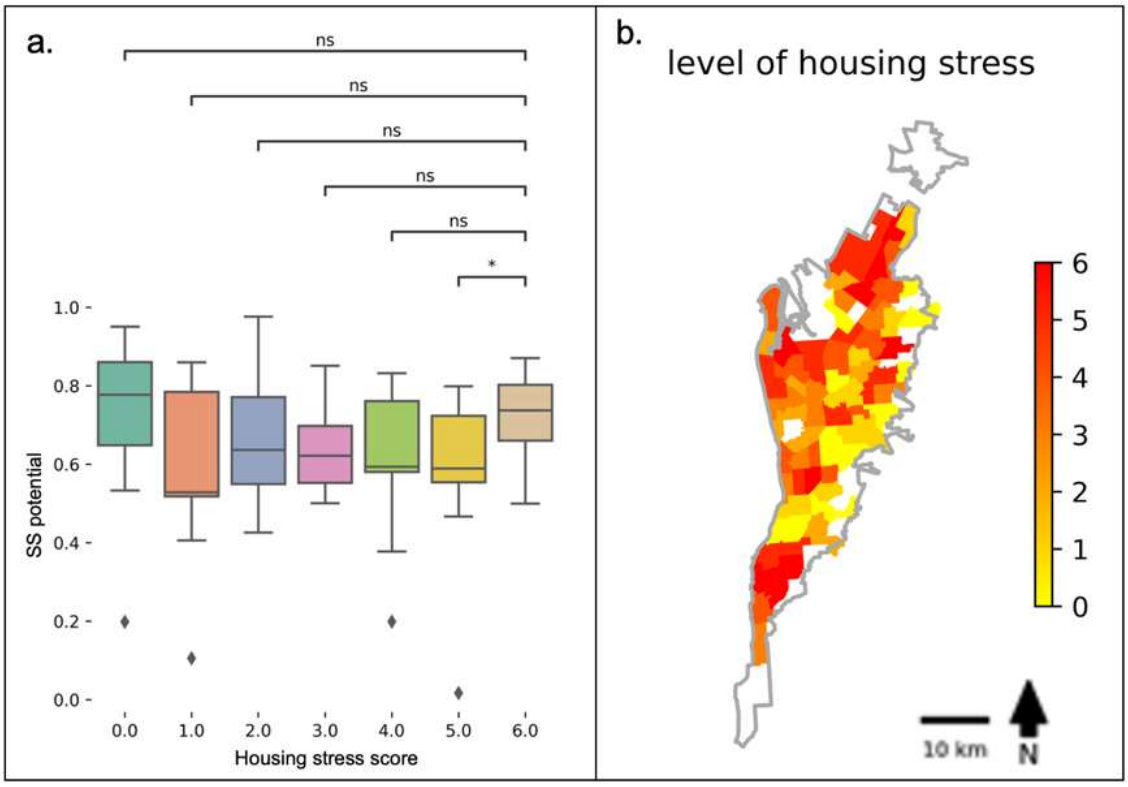

Figure 11 SS potential (as ratio of properties able to achieve household SS) of areas with varying levels of Housing Stress (panel a.). Spatial distribution of housing stress across the Adelaide Urban Area (6 being the highest level of housing stress, and 0 being the least) (panel b.).

Eleven Areas were ranked 6 – the highest level risk of housing stress. The SA3 most represented was of Onkaparinga, which contained five Areas with a housing stress score of 6, and Playford, which contained two. Playford is located in the Northern end of the study site, and Onkaparinga in the South. Areas with highest levels of housing stress have some of the highest rates of household SS potential (Figure 11).

Pairing the levels of Housing Stress with Self Sufficiency potential in each area provides insights into targeted policy interventions that can support and improve food security and food justice for vulnerable populations. For instance, In Areas where vulnerability (measured as income stress) is high, and properties have high SS potential, councils have an opportunity to take proactive steps to empower community members to engage in backyard farming successfully. This research identified four Areas that have extremely high levels of housing stress and significant household SS potential. The Areas of Elizabeth, Smithfield - Elizabeth North, Hackham West - Huntfield Heights and Morphett Vale have housing stress scores of 6, yet 80% - 87% of properties meet the criteria for SS. These Areas are marked in green in Figure 12. These Areas have a unique opportunity, as they already have many of the physical resources required for in-ground vegetable production. Nevertheless, access to land is not the only challenge. Australian survey respondents have identified inputs and materials, knowledge and training, and financial support as resources that would help them expand their edible gardening (Donati & Rose, 2020). Support could come in the form of workshops, mentoring programs or factsheets. Financial and material aids supplied to vulnerable populations would also provide essential gardening infrastructure. However, there is a need to deepen understanding of the barriers faced by backyard farmers specifically to ensure that they receive the support they need.

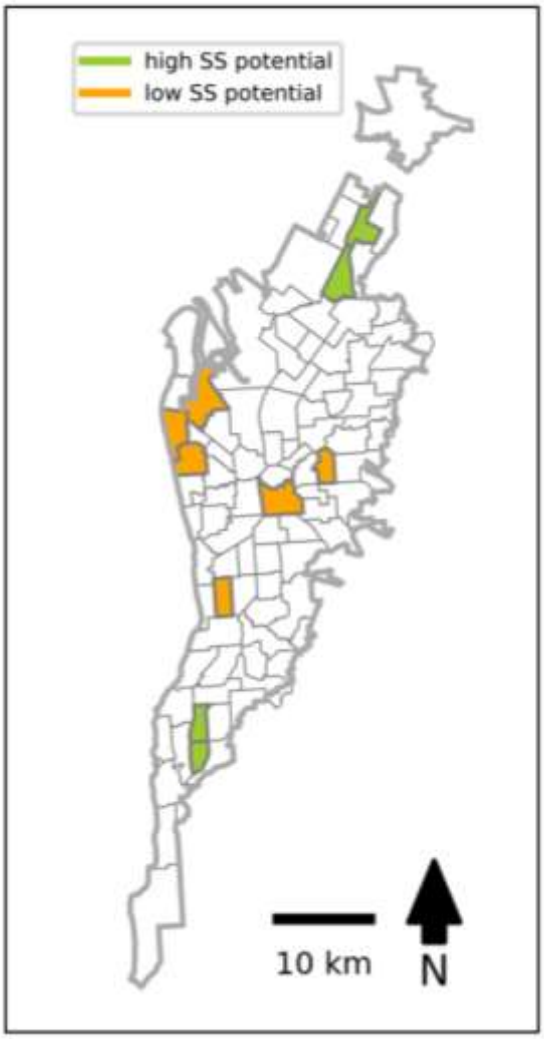

Figure 12 Areas that have high Housing Stress, and are also in the highest (green), or lowest (orange) quantile of Self-sufficiency potential. More specifically, Orange Areas are in the lowest quantile of SS potential, with a Housing Stress Score >= 5, while green Areas are in the highest quantile of SS with Housing Stress scores of 6.

While the analysis demonstrates a general trend of higher self-sufficiency potential in areas with higher housing stress scores, it is important to note that there are pockets of vulnerability that do not follow this trend. Notably, Port Adelaide ranked in the lowest quantile for SS potential, with 50% SS potential, and had a housing stress score of 6. Curiously, Port Adelaide is also one of the least densely populated Areas with 893 people km^{-2}. However, unlike Adelaide and North Adelaide, Port Adelaide has a large industrial footprint, containing non-residential land that is not plantable. As already

discussed, by contrast, much of the non-residential land in Adelaide and North Adelaide is plantable parklands (to further illustrate this point, and provide further context, arial images have been included in supplementary material Figure S2, Appendix F). Additionally, several other Areas in the lowest quantile for self-sufficiency potential had housing stress scores of 5. As has already been discussed, *Adelaide* is an outlier, with a SS potential of 2%. The SS potential in the other Areas range from 47% to 57%.

4.4 Urban Agriculture outside of private land

For Areas with a high population at risk of food insecurity, but low potential for residential urban agriculture, policy makers should prioritise integrating food production into the planning of public urban spaces. As outlined in *Section 1,* Adelaide and North Adelaide have low residential SS potential, but possess extensive open public space that could be volunteered for urban food production in community gardens. However, these initiatives efforts must be highlighted by planners and decision-makers. As such, it is crucial to provide planners with appropriate education and training to ensure that they understand the benefits and risks of urban agriculture (Pires, 2011).

Promoting community gardening assumes plantable public space is available. As a part of a city-wide investigation of tree canopy cover, Holt (2021) investigated the spatial distribution of plantable space in Adelaide. While the investigation was not aimed at mapping available land for urban agriculture, Holt (2021)'s work provides insight into the opportunities for SS as it investigates plantable space in private and public land. Indeed, there is an opportunity to increase both canopy cover and food supply (i.e., fruit trees).

Plantable space is least available in suburbs closest to the city, within a 5km to 6km radius of the CBD (Holt, 2021). Hence, Areas such as Norwood, which have low household SS potential, high levels of housing stress and low rates of plantable areas, may have limited resources to support urban agriculture projects.

The Planning profession aims to improve the health and well-being of communities. It has generally been considered the role of urban planners to lead the effort to integrate urban agriculture by allocating and regulating land use (Sarker et al., 2019). However, these results show that residential properties in areas with the most vulnerable communities have significant space for back-yard farming (Figure 11), which shifts the focus away from land use planning as the primary point of call. While this analysis provides valuable insights into the distribution of household self-sufficiency potential, one limitation is that its results were analysed at the aggregate level. At this resolution, vulnerable individuals may be overlooked. Future research could consider examining resource distribution at a finer scale. Relatedly, considering other stressors that impact food insecurity, such as age, family makeup, and employment status (Miller & Li, 2022), would further aid decision making. To realise the full potential of urban agriculture, work should be undertaken to understand the barriers to home gardening.

Extreme food insecurity is often associated with poverty. It is not our intention to suggest that urban agriculture could or should replace food relief as a critical crisis response for food insecure peoples. While it may not be a panacea, improving dietary diversity and food access with urban agriculture can improve how poverty is experienced. Urban agriculture is a valuable tool available to help address the complex

issue of food insecurity, however, the economic inequalities that lead to poverty and extreme food insecurity should be a priority of all government levels and addressed independently.

5. Conclusions:

This research explores distribution of resources for urban agriculture and investigates the correlation between self-sufficiency potential and financial stress indicators across one city. Examining the relationship between self-sufficiency potential and income stress variables illustrated that the most vulnerable communities have significant potential to produce food in their own back yards. Conversely, the bespoke spatial analysis of backyard farming potential also identified specific Areas where vulnerable communities do not have access to privately owned land for food production. These are the locations where policy makers should aim interventions to expand urban agriculture on open public space. Taken together, the results show that there is no single trend that can be generalised across the population. Bespoke spatial analyses identify specific opportunities and constraints within each Area that can inform targets and policies that promote urban agriculture and support the most vulnerable members of the community. While population density is important in determining the self-sufficiency potential of urban areas, policymakers should consider additional analysis to avoid overlooking opportunities and limitations for specific populations.

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Chapter 5
Is the grass always greener? Comparing the global warming potential of backyard farming and lawn.

Statement of Authorship

Title of Paper	Is the grass always greener? Comparing the global warming potential of backyard farming and lawn
Publication Status	☐ Published ☐ Accepted for Publication ☑ Submitted for Publication ☐ Unpublished and Unsubmitted work written in manuscript style
Publication Details	This manuscript has been submitted to the Journal of Cleaner Production

Principal Author

Name of Principal Author (Candidate)	Isobel Hume
Contribution to the Paper	Literature review Formal analysis Interpretation Writing - original draft
Overall percentage (%)	90%
Certification:	This paper reports on original research I conducted during the period of my Higher Degree by Research candidature and is not subject to any obligations or contractual agreements with a third party that would constrain its inclusion in this thesis. I am the primary author of this paper.
Signature	Date: 22/05/2023

Co-Author Contributions

By signing the Statement of Authorship, each author certifies that:

i. the candidate's stated contribution to the publication is accurate (as detailed above);

ii. permission is granted for the candidate in include the publication in the thesis; and

iii. the sum of all co-author contributions is equal to 100% less the candidate's stated contribution.

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Please cut and paste additional co-author panels here as required.

Abstract

As the global urban population continues to grow, immediate action is required to address food insecurity, climate change, and sustainable urban development. Urban agriculture is widely acknowledged as a multifunctional strategy that addresses these interconnected issues. The COVID-19 pandemic highlighted vulnerabilities within food systems, and sparked citizen-led engagement in urban agriculture. Despite the growing movement, the environmental impacts of residential urban agriculture remain under studied. This paper analyses urban agriculture as an alternate food production system, using a life cycle approach to establish the global warming potential (GWP) of back yard vegetable farming and residential lawn. Maintenance (irrigation, nutrients, and mowing) and soil bio-geochemical processes were considered. Urban agriculture was found to have a GWP of 22.55 t CO_2e ha $^{-1}$ year $^{-1}$ to 28.37 t CO_2e ha $^{-1}$ year $^{-1}$, whereas lawn had a GWP of 1.04 t CO_2e ha $^{-1}$ yr $^{-1}$ to 4.66 t CO_2e ha $^{-1}$ yr $^{-1}$. Compared to business as usual, back yard farming could result in emission reduction of -0.14 kg CO_2e kg vegetables $^{-1}$ yr^{-1} to -0.32 kg CO_2e kg vegetables $^{-1}$ yr^{-1}. Dietary patters, household make up, and Adelaide's urban form were considered. In the best-case scenario, the uptake of urban agriculture could achieve 1.10% to 0.86% of the emissions reduction targets outlined for households in The Paris Agreement.

Keywords: urban agriculture; vegetable production; global warming potential; life cycle; soil GHG emissions; food system modelling.

Highlights

1. GWP of home-grown vegetables and lawn were modelled using life cycle approach

2. DeNitrification DeComposition (DNDC) software modelled soil greenhouse gas emission

3. Maintenance (irrigation, nutrients, and mowing) reflect actual behaviours of growers

4. Dietary patterns, household make up, and urban form were also analysed

5. Home production reduced GWP of vegetables by up to -0.32 kg CO_2e kg veg^{-1} yr^{-1}

1. Introduction

Global food production will need to double by 2050 in order to meet the needs of our rapidly growing population (van Dijk et al., 2021). For countries in the economically developed Global North, where the majority of the population lives in urban areas (United Nations, 2018), food security is an increasingly concerning issue (Miller & Li, 2022; Sharifi, 2021). At the same time, climate change is directly threatening the integrity of arable land, and the stability of food supply chains (Dasgupta & Robinson, 2022; Mbow et al., 2017). Urbanisation has increased energy consumption and greenhouse gas emissions, and contributed to land degradation, soil erosion and water shortages, all of which can affect food production (Mbow et al., 2017; Revi et al., 2014). Despite these challenges, cities have significant potential for climate change adaptation and mitigation (Sharifi, 2021). Urban agriculture, the cultivation and distribution of food in urban areas, is widely acknowledged as a multifunctional strategy that addresses issues of food security, environmental sustainability, economic development, and social cohesion (Evans et al., 2022; Langemeyer, Madrid-Lopez et al., 2021). However, many questions about the plausibility of truly sustainable urban agriculture remain (Hume, et al., 2022).

The United Nations recognizes the vital role that urban agriculture plays in advancing several Sustainable Development Goals (SDGs), including Zero Hunger (SDG 2), Sustainable Cities and Communities (SDG 11), and Climate Action (SDG 13) (FAO, 2020). Recently, the COVID-19 pandemic resulted in worldwide upheaval of food supply chains and revealed vulnerabilities in cities and urban centres with traditionally sound food

security (World Bank, 2021). The disruptions drove a surge of bottom up, citizen led engagement in urban agriculture to improve food access (Kirkpatrick & Davison, 2018; Music et al., 2021). These efforts successfully achieved their intended social benefits (Donati & Rose, 2020). However, it has been suggested that many claims on the environmental sustainability of urban agriculture have been made prematurely given the lack of quantitative assessments (Goldstein et al., 2016; Salomon & Cavagnaro, 2022). Weidner et al. (2019) argue that studies on urban agriculture have tended to take an advocacy viewpoint, rather than critically analyse the potential consequences. Additionally, the benefits of a particular production method are often attributed to the whole spectrum of urban agriculture systems, leaving an incomplete understanding of the strengths and weaknesses (Goldstein et al., 2016; Weidner et al., 2019).

A recent literature review and meta-analysis confirmed that our understanding of the environmental impacts and resource use of urban agriculture is still in its early stages (Dorr et al., 2021). Models that do strive to quantify the environmental impact of urban agriculture focus on commercial or highly controlled (e.g., indoor) systems that are not representative of typical home gardens (Dorr et al., 2021). For example, Lucertini and Di Giustino (2021) found roof top urban agriculture to be an effective climate change mitigation and adaption strategy, sequestering carbon, mitigating urban heat island effects, and reducing the electricity required to cool buildings. However, such results cannot be extrapolated to in ground vegetable production. Compared to high-tech solutions or community gardens, research into back-yard farming receives relativity little attention (Kirkpatrick & Davison, 2018; Mullins et al., 2021; Taylor & Lovell, 2014). Despite being under-represented in the literature, as much as 90% of urban agricultural

practitioners utilise residential yards for ground-based food production (Donati & Rose, 2020). In low to medium density urban areas, residential land can be a significant resource for food production and provide a substantial portion of dietary self-sufficiency (Hume et al., 2021; Hume et al., 2022).

As the literature on Urban Agriculture matures, there is a need to shift from advocacy to critical and quantitative assessment. Upstream processes, including the energy and material profiles of inputs must be carefully considered (Goldstein et al., 2016). In the case of community-driven urban agriculture, this is particularly challenging as the outcomes are dependent on the specificities of the system and management practices used. Consequently, only few researchers have documented resource use in low tech, community led urban agriculture (Csortan et al., 2020a; McDougall et al., 2019). With in-ground farming in residential yards dominating urban farming systems, there are large gaps in knowledge that require attention. Moreover, previous studies have relied on data from conventional agricultural inputs and yields (Cleveland et al., 2017; Kulak et al., 2013), rather than data from residential yards (Csortan et al., 2020a; Hume et al., 2021, 2022; McDougall et al., 2019; McDougall et al., 2020).

When assessing the footprint of urban agriculture, it is necessary to compare it to other urban land uses. Residential lawn is well suited to in ground vegetable production as it is immediately available for conversion, with little capital investment and no legislative interventions (Hume et al., 2022; Richardson & Moskal, 2016). Moreover, the conversion of lawn to food production does not come at the expense of tree canopy or infrastructure. Across the globe, lawns are the most ubiquitous form of urban vegetation

and are a symbol of modern urban landscapes (Ignatieva et al., 2015). Residential lawns hold social, cultural, and aesthetic value, and are often viewed as 'natural' in the western world (Ignatieva et al., 2015). Individuals may not consider the environmental impact of managed lawns, however, managed urban lawns can be net emitters of green-house-gasses (Gu et al., 2015; Tidåker et al., 2017; Townsend-Small & Czimczik, 2010a, 2010b; van Delden et al., 2016). Emissions come from upstream processes or inputs, such as mowing, irrigation, and fertilizers, and direct emission from soil.

Direct emissions from soils result from bio-chemical processes influenced by land use, climate, soil type, and nutrient and substrate availability (Rowlings et al.,2015; van Delden et al., 2016). The climate relevant trace gases in this context are carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Nitrification and denitrification in soil (van Delden et al., 2016)., as well as the metabolism of O_2 and CH_4 by methanogens (Rowlings et al., 2015), release greenhouse gasses into the atmosphere. Carbon can also be stored as Soil Organic Carbon (SOC), acting as a sink for atmospheric greenhouse gasses (Ontl & Schulte, 2012).

While in-ground food production in residential yards is the dominant form of food production in urban areas, there is a paucity of research quantifying the environmental outcomes. Therefore, here, we present results of a study that examines the Global Warming Potential (GWP) of back-yard farming, and residential lawn. Global Warming Potential is a measure of the carbon dioxide equivalents (CO_2 e) associated with a process and its materials. To calculate the GWP, we consider direct greenhouse gas (GHG) emissions from soil, soil organic carbon (SOC) and GHG associated with

management inputs (i.e., fertilizers, compost, and irrigation). As urban agriculture intersects challenges of climate change, sustainable urban development, and resilient food systems, this analysis also examines consumer behaviour, dietary choices, and land and resource availability in urban environments, using a low-density city in the global North (Adelaide, Australia) as a case study. Specifically, this paper asks:

1. What are the direct GHG and GWP of residential lawn and residential food gardens? This part of the investigation will employ DeNitrification DeComposition (DNDC) software to model trace gas emissions, as well as fluxes in the SOC pool,
2. What are the GHG emissions, or CO_2 equivalents (CO_2 e), of the upstream processes and materials required in lawn management and back-yard vegetable gardening? and,
3. How does the uptake of urban agriculture affect a household's global warming budget? This part of the analysis will use functional scenarios that mimic consumer behaviours and reflect the urban form of Adelaide, Australia.

2. Methods

2.1 Study site

To investigate the Global Warming Potential (GWP) of back yard urban farming (urban agriculture), the city of Adelaide, Australia has been selected as case study. Home to 1.5 million people, Adelaide (34.9° S, 138.6° E) (Figure 13) is the fifth most populous city in Australia and the capital of South Australia. Adelaide has been used as a case study in

our previous work on land, and resource, availability for urban agriculture (see Hume et al., 2021, 2022)

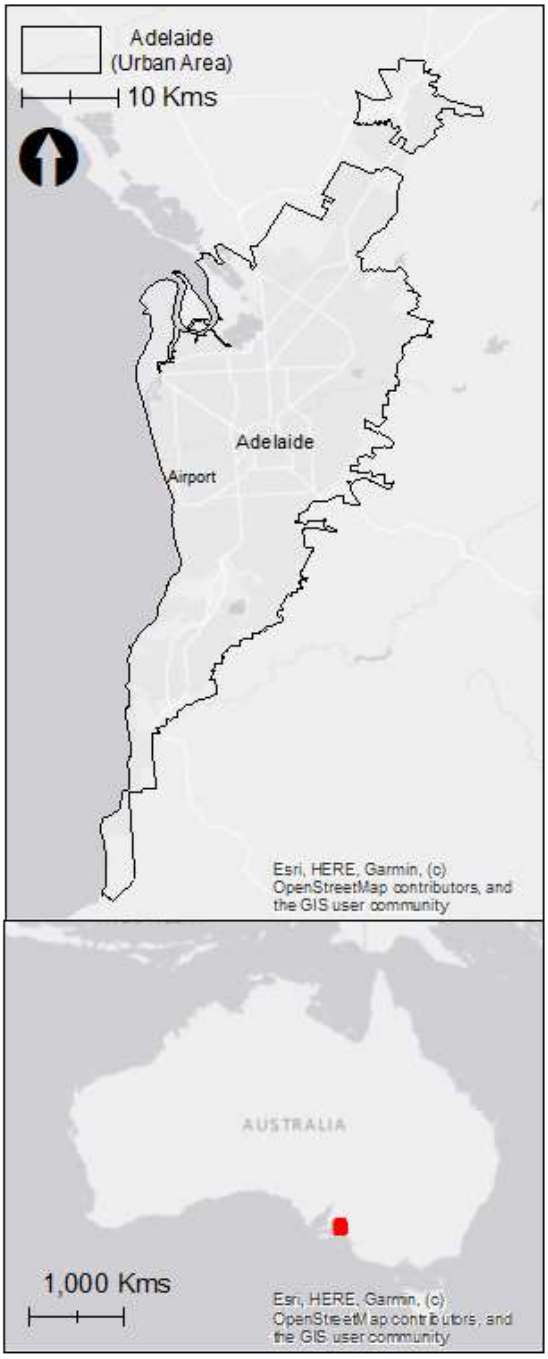

Figure 13 Location map of study site (Adelaide, Australia)

Adelaide is situated on fertile agricultural land. Adelaide's Mediterranean climate is characterised by hot, dry summers and cool, wet winters (classified as Csa according to

the Köppen climate classification). Adelaide has an average maximum temperature of 22.5°C and an average minimum of 12.3°C, and the annual average rainfall is 547.1 mm The hottest month is January, with an average maximum of 29.6°C while the coolest month is July, with an average maximum temperature of 15.4°C (Bureau of Meteorology, 2020). This climate is conducive to year-round open air vegetable production.

Adelaide's low-density urban landscape is characterised by free-standing houses and private yards, presenting significant opportunities for food production to occur on residential land (Australian Bureau of Statistics, 2020; Hume et al., 2022). Hume et al. (2021) estimated that 93% - 69% of residential properties have enough yard space to achieve household self-sufficiency of vegetables, while 65% of properties have enough yard space to grow the recommended vegetable intake, while also capturing and storing enough rainwater to irrigate crops (Hume et al., 2022). These results highlight the significant food producing capabilities of back yard farming in Adelaide.

Additionally, Adelaide has been subject of extensive research in low-tech, community led urban growing. Csortan et al. (2020a) conducted a detailed investigation into capabilities and potential of Adelaide home food gardens, including productivity and resource efficiency. Raw data from the investigation has since been made publicly available for reuse (Csortan et al., 2020b). This dataset details climate-appropriate management, including crop type, irrigation schedules, and reflects the actual behaviours of back-yard growers, providing a significant advantage for modelling the environmental impact of urban gardeners.

Taken together, given Adelaide's climate, urban landscape, and previous research on low-tech, community-led urban growing, it is an attractive case study for investigating the potential of back yard farming. This work will expand the current body of literature, providing insights into the sustainability outcomes of gardener practices and building on previous work in logical succession.

2.2 Model set up

This work assesses the GWP of two distinct land use types: lawn; and urban agriculture in the form of a polyculture food garden. The model considers two primary processes, namely: GWP of soil emissions, and GWP of inputs (Figure 14). The basic parameters and management practices of each scenario are illustrated in Figure 15.

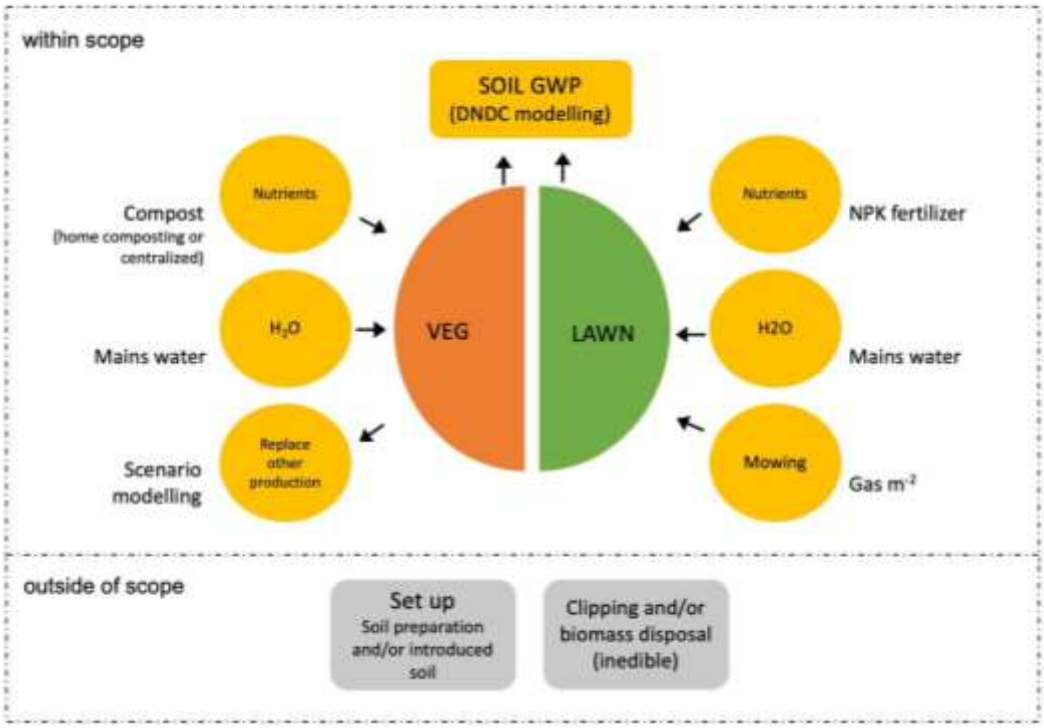

Figure 14 system boundary for GWP comparison of Lawn and Vegetable garden.

The urban agriculture scenario consists of a polyculture vegetable garden managed by individual households, as described by (Hume et al., 2022). Where possible, management of home gardens is modelled from average practices reported by home gardeners involved in the South Australian Edible Gardens Project (Csortan et al., 2020a). The original raw, and partially analysed data set is publicly available online (Csortan et al., 2020b). The edible gardens project surveyed 34 South Australian home gardeners from September 2016 to June 2018, who grew a range of crops including vegetables, fruit, eggs, and honey.

Two different management scenarios are considered for the computation of Lawn GWP: a High Maintenance (HM) and Low Maintenance (LM) scenario (Figure 15). These are based on household lawn management practices found in the literature (Gu et al., 2015), and common lawn care recommendations tailored to the Adelaide region (Great Aussie Lawns, 2021; SA Water, 2015).

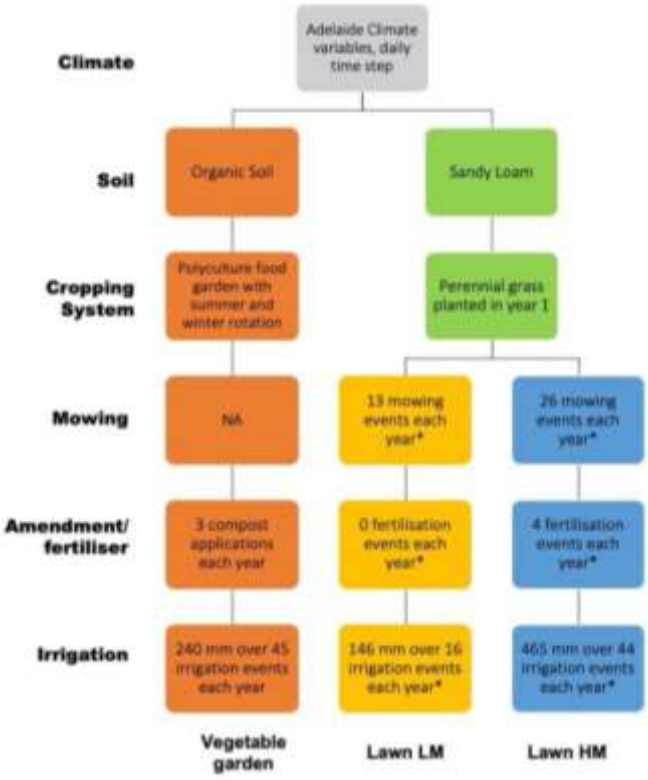

*Figure 15 overview of systems modelled (Vegetable garden, Low Maintenance (LM) Lawn and High Maintenance (HM) Lawn). The key variables are climate, soil, vegetation (cropping system), and management practices. The vegetable garden is harvested and planted on an annual cycle, whereas lawn is perennial. * values for all years after year 1.*

2.3 Soil GWP (DNDC):

This study uses the DNDC (DeNitrification DeComposition) (version 9.5) model to simulate trace gas emissions. The DNDC model is a process-oriented computer simulation model of carbon and nitrogen biogeochemistry in agroecosystems (Abdalla et al., 2022; Gilhespy et al., 2014). It has previously been used to model trace gas emission from lawn (Gu et al., 2015) and urban vegetable production (Budiman et al., 2020). The entire model is driven by four primary ecological drivers: climate, soil, vegetation (cropping system), and management practices. A comprehensive list of all DNDC input values can be found in the supplementary table (S1, Appendix G). A summary of the input values is described below.

2.3.1 Climate

Climate data was retrieved from the Bureau of Meteorology (2021), for Adelaide Airport weather station (situated centrally in the Adelaide metro/study area extent). The variables included are: max temp, min temp, precipitation, and radiation. Climate data from 2017 is used for each year of the simulation, which was determined by the availability of observational irrigation data for vegetable gardens (see S1, Appendix G). Rainfall in 2017 was close to the average, with mean temperatures typically between half and one degree warmer than average (Bureau of Meteorology, 2018).

2.3.2 Vegetable Garden

2.3.2.1 Soil

The vegetable garden soil was modelled from a recent survey of chemical and biological properties of existing soil found in urban agricultural in Adelaide (Salomon et al., 2020) Soil texture was set as *Organic Soil* in in the DNDC software, with the following soil parameters adjusted: pH, 7.1; SOC at surface: 7.9%; nitrate (mg kg $^{-1}$): 9.1; and ammonium (mg kg $^{-1}$): 6.2. The remaining soil parameters remained as default settings for *Organic Soil* (S1, Appendix G).

2.3.2.2 Cropping system

Urban agriculture sites are typically diverse in terms of planting, and highly individualised (Salomon et al., 2020). Nevertheless, there are some commonalities in what is grown (largely due to seasonality and dietary preferences) (Csortan et al., 2020a, 2020b). This information was used to design a representative planting sequence for a polyculture urban food garden. The cropping system consists of a summer and winter rotation of vegetables, which repeats on an annual cycle for every year of the simulation. The crops selected mimic, as closely as possible, those which were reported to be grown by urban growers in the Adelaide region (Csortan et al., 2020b). All crop growth parameters remain as default settings in the DNDC software.

The winter crops are: broccoli, green beans, onions, radish, cabbage. The summer crops are: beans, beetroot, tomatoes, lettuce, spring onion, capsicum, and carrot. The same amount of space is dedicated to each crop. However, as there are more summer crops than winter crops, broccoli and green beans are planted twice in the winter rotation (i.e.

they are assumed to take up twice as much space as the other crops). Planting and harvest schedules are included in the supplement (S2, Appendix G)

2.3.2.3 Management

Amendments

It is well established that UA producers in Adelaide typically adhere to the principles of organic farming (Salomon et al., 2022; Santo et al., 2021); in particular, nutrients are typically delivered to plants using compost. Compost application occurs three times each year as recommended by Salomon (2021). Two primary applications of 20 mm depth occur between the summer and winter planting (Feb 29, Aug 31), and one secondary application of 10 mm occurs on Nov 30 (Salomon, pers. comm., 2021). This equates to a typical application of 2,619 kg of organic N ha^{-1} yr^{-1}. No synthetic fertilizers are applied.

The nutrient content of the compost (Organic N, NH_4^+, and NO_3^-) is taken from Salomon et al. (2022) (S1, Appendix G) and the compost has a bulk density of 582 kg m^{-3} (Khater, 2015).

Irrigation

The frequency and depth of irrigation was based on the survey of urban gardeners in Adelaide collected by Csortan et al. (2020a). As gardeners adjust their irrigation according to rainfall events (Hume et al., 2022; Ward et al., 2014), only records from one year (2017) were included in the model. Doing so allows these records to be paired with climate data for the same year in the DNDC simulation. Gardens are irrigated with 240

mm across 57 irrigation events each year. The irrigation schedule is outlined in the supplement (S1, Appendix G)

2.3.2 Lawn

2.3.2.1 Soil

Both lawn management scenarios occur on Sandy Loam soil. Soil bulk density is 1.55, which is standard for sandy loam texture and supported by analysis of turf grass in the literature (Qian & Follett, 2012). The soil pH is 6.5. Surface soil organic carbon (SOC) is 0.9%, based on Qian and Follett (2012) finding that turfgrass systems contain 1.5% soil organic matter, and the finding that SOC comprises 58% of SOM mass (Salomon et al., 2022). All other soil parameters are set to default for Sandy Loam soil (S1, Appendix G).

2.3.2.2 Cropping system

Lawn is planted in March (early Austral Autumn), year one of the simulation, and grows for the duration of the model. Crop Type is Perennial grass, however leaf stem C:N ratio changed from default to 20, and root C:N ratio changed from default to 40 (Gu et al., 2015). All other crop parameters remain default for perennial grass (S1, Appendix G).

2.3.2.3 Management

Both LM and HM scenarios are treated identically for the first three months. Seeds are watered with 10 mm every day for three weeks, and then regular watering as per their scenario's schedule occurs (Yates, 2022). A germination specific fertilizer (Great Aussie Lawns, 2021) is applied at planting, and again after three weeks, then regular fertilization occurs as per the scenario's schedule. Fertilizer is applied at a rate of 30 g m^{-}

^{2} and contains 4.8% N and 5.7% P (Great Aussie Lawns, 2021). No cutting events occur for three months then regular mowing as per their scenario's schedule occurs.

Fertilizer

The LM scenario has no fertiliser applications after establishment (Gu et al., 2015). The HM scenario is fertilized with All Season Reno Fertiliser, as per the as per the manufacturer's recommendations (Great Aussie Lawns, 2021). There are four fertilization events each year: early Spring, mid-Summer, and twice in Autumn. The recommendation falls within the upper range of fertilizer applications reported by lawn managers in the literature (Gu et al., 2015). Fertilizer is applied at a rate of 30 g m^{-2}, and contains 22% N and 5% P (Great Aussie Lawns, 2021).

Irrigation

Irrigation schedules were calculated using SA Water's Irrigation Management Toolkit, which is based on the Code of Practice for Irrigated Public Open Space (SA Water, 2015). The toolkit relies on climatic variables and the desired vigour of the lawn to optimise irrigation. Evapotranspiration and rainfall data were collected from Adelaide Airport weather station for the year 2017 (Bureau of Meteorology, 2021). For the HM scenario, the functional purpose, which implies desired vigour, was set to *Local Sports Turf* (TQVS3). In the LM scenario, half of the recommended watering events for *Passive Recreational Turf* (TQVS4) were included. Further information on the Code of Practice for Irrigated Public Open Space, justification of the irrigation schedules selected, and a description of the frequency and depth of irrigation events can be found in the Supplement (S1, Appendix G).

Mowing

In the HM scenario, lawn is cut every two weeks, and clippings removed. In the LM scenario lawn is cut every four weeks, and clippings are removed. At each mowing event, 30% of the above ground biomass (stem, leaves, and grain) is removed.

2.2.3 Calculation of GWP from DNDC results

DNDC model computes DeNitrification DeComposition processes which are then used to calculate the GWP using the following equation (Gu et al., 2015; Li et al., 2005):

$$GWP = (25 * 16 * CH_4 / 12) + (298 * 44 * N_2O / 28) - (44 * dSOC / 12)$$

Where:

GWP = 100-year global warming potential (kg CO_2 equivalent ha $^{-1}$ y $^{-1}$),

CH_4 = CH_4 flux (kg C ha $^{-1}$ y $^{-1}$), as calculated by the DNDC model,

N_2O = N_2O flux (kg N ha $^{-1}$ y $^{-1}$), as calculated by the DNDC model, and

dSOC = change in soil organic carbon (SOC) (kg C ha $^{-1}$ y $^{-1}$), i.e., carbon sequestration rate, as calculated by the DNDC model.

2.4 GWP of inputs

In addition to modelling the GWP from soil biogeochemical process, the analysis seeks to quantify the GWP from maintenance and inputs. This includes, irrigation, nutrients (fertilizer and compost) and lawn mowing (gasoline consumption). Soil preparation and physical tools such as garden hoses, trowels, lawn mowers or garden bed boundaries are

outside the scope of the analysis. The GWP of inputs are summarised in Table 3 and expanded upon in the following text. Further calculations are displayed in the Supplement (S3, Appendix G)).

Table 3 Global warming potential of upstream processed, management practices, and inputs.

Input	Value	Units	Justification
N	4.76	kg CO_2 e kg N -1	The conversion factor for Nitrogen fertilizer as reported by Lal (2004) via Gu et al. (2015).
P	0.73	kg CO_2 e kg P -1	The Conversion factor for P fertilizer as reported by Lal (2004) via Gu et al. (2015).
K	0.55	kg CO_2 e kg P -1	The conversion factor for K fertilizer as reported by Lal (2004) via Gu et al. (2015).
Compost	0.149.	kg co_2 e / kg compost	For Home Compost (HC) (Lu, Qu, & El Hanandeh, 2020).
	0.129	kg co2e / kg compost	For Centralised Composting (CC) (Lu et al., 2020).
Irrigation	0.346	kg co2e m3 of irrigation	From the OzLCI, analysed using openLCA under the IPPC 100yr GWP impact method.
Mowing	18.167	kg co2e ha $^{-1}$ mowing event $^{-2}$	7.764 L gasoline ha ^{1} per mowing event (Saidani & Kim, 2021). The CE conversion of gasoline is 8.87 kg CO2 per gallon of gasoline (Lal, 2004). This = 2.34 kg CO2e per L. To determine the total kg of CO2 emitted each year by gasoline, we multiply 7.764 L ha−1 by 2.34 kg CO2 gal−1 and the mowing events each year.
Replacing conventional farming	-27430	kg CO2 e Ha	0.54 kg co2e kg $^{-1}$ produce grown (Clune, Crossin, & Verghese, 2017) . Av. yield from the EG survey was 5.08 kg m^2. Therefore, 0.54 * 5.08 = 2.743 kg m^{-2} = 27430 kg.

2.4.1 Fertilizer

The fertiliser used to derive NPK values and application schedule used in the DNDC model is the *Reno fertilizer* and *Kickstart fertilizer* produced by Great Aussie Lawns (2021). The CO2 e conversion factors for the N, P, and K fertilizers are taken from Lal (2004) via Gu et al. (2015). The conversion factors are as follows: 4.76 kg CO_2 kg N $^{-1}$, 0.73 kg CO_2 kg P $^{-1}$, and 0.55 kg CO_2 kg K $^{-1}$.

2.4.2 Compost

In Adelaide, kerbside municipal green waste collection is commonplace. Green waste is transported to one of two commercial windrow composting facilities, before being sold

as a commercial product (Green Industries SA, 2022). Therefore, home gardeners have the option of either home composting (HC scenario) or purchasing the centralised composting waste end-product (CC scenario).

Lu et al. (2020) performed a life cycle assessment on both home composting and centralised composting systems in Brisbane (Australia), which is used as a proxy for values in Adelaide (Australia). The HC system included a green waste collection bin and a food waste bin; garden tools such as garden chipper, and two compost bins for waste composting (Lu et al., 2020). The CC system assumes organic waste is collected from residential areas, sorted, shredded, and placed in compost windrows. The CC scenario includes 50 km of transportation to deliver finished compost to the user (assumed to be farmland in the analysis), with a 50 km return trip (Lu et al., 2020). Further description of the system boundary of each scenario is outlined in (Lu et al., 2020).

The GWP of compost is 0.129 kg CO_2 e kg compost $^{-1}$, and 0.129 kg CO_2 e kg compost $^{-1}$ for the CC, and HC scenarios, respectively (Lu et al., 2020). It was assumed that household green waste production can meet the compost requirements of crops (McDougall et al., 2020).

2.4.3 Irrigation

The GWP of irrigation was take form the Evah OzLCI2019 Free Database (The Evah Institute, 2019) for the item Pump & Use Town Water in South Australia. Data was analysed using openLCA with the IPPC GWP 100-year impact method (GreenDelta,

2022). Water was found to have a 100-year GWP of 0.346 kg CO_2 e m^{-3} (0.000346 kg L^{-1}).

2.4.4 Lawn Mowing

A conventional walk-behind lawn mower consumes 7.764 L of petrol (gasoline) ha^{-1} during each mowing event (Saidani & Kim, 2021). Gasoline contains 2.34 kg CO2e L^{-1} (calculated from Lal (2004)). Therefore, to determine the kg of CO_2 e emitted each year from lawn mowing, we multiply 7.764 L ha^{-1} by 2.34 kg CO_2 L^{-1} and the number of mowing events each year.

2.4.5 Replacement of conventional agriculture

The emissions associated with conventional farming are used as a baseline to which urban agriculture can be compared. Additionally, the replacement of conventional farming with urban agriculture results in avoided emissions. We analysed data collected by Clune et al. (2017) in their systematic review of greenhouse gas emissions for different fresh food categories. The average GWP of vegetable crops grown in Australia is 0.54 kg of CO_2e kg produce $^{-1}$ (Clune et al., 2017) (see supplement S4, Appendix G). This value includes post-harvest on-farm activities but does not consider emissions associated with transporting produce from the farm gate to the consumer. Marquez et al. (2010) calculated an additional 134.5 kg of emissions for every tonne of vegetable transported to consumers in Melbourne, Australia. Values vary greatly depending on the produce's origin location (< 50 kg t^{-1} for local produce to > 200 kg t^{-1} for Australia wide produce). The average is within the range expected for transport associated emissions in

the food industry (Clune et al., 2017). The value includes supply-chain refrigeration and consumer transport (Marquez et al., 2010).

2.5 Functional scenario analysis

Functional scenarios consider dietary intake of vegetables and Adelaide's urban form to quantify the net impacts of urban agriculture. Hume et al. (2021) analysed the Edible Garden Survey raw data (Csortan et al., 2020b) to determine expected yields from urban vegetable gardens. Yields ranged from 0.24 kg m^{-2} year-^{1}, to 16.07 kg m^{-2} $year^{-1}$, with a mean value of 5.08 kg m^{-2} $year^{-1}$. The mean will be used in this study as a conservative estimate of productivity. We also consider the amount of lawn currently being managed. Hume et al. (2022) parametrized 491,712 residential properties in Adelaide, identifying 4,588 ha of residential lawn in total. The median lawn area at a single property was 71 m^{-2}.

3. Results

3.1 Soil emissions

Results from the DNDC analysis revealed that both of lawn scenarios are net emitters of greenhouse gasses, while the veggie scenario resulted in a negative GWP value. At 100 years, the cumulative GWP were 123.86 t CO_2 e ha $^{-1}$, 19.22 t CO_2 e ha $^{-1}$ and -1581.76 t CO_2 e ha $^{-1}$ for Lawn HM, Lawn LM and Veggies respectively (Figure 16). Therefore, the annual GWP of each of the three land uses over the time of the duration of the simulation were 1.24 t CO_2 e ha $^{-1}$ yr $^{-1}$, 0.19 t CO_2 e ha $^{-1}$ yr $^{-1}$, and -1.581t CO_2 e ha $^{-1}$ yr $^{-1}$ for Lawn HM, Lawn LM and Veggies respectively. Taking the average over the 100 years

of the simulation results in a conservative estimate of soil GWP for vegetables (Figure 16).

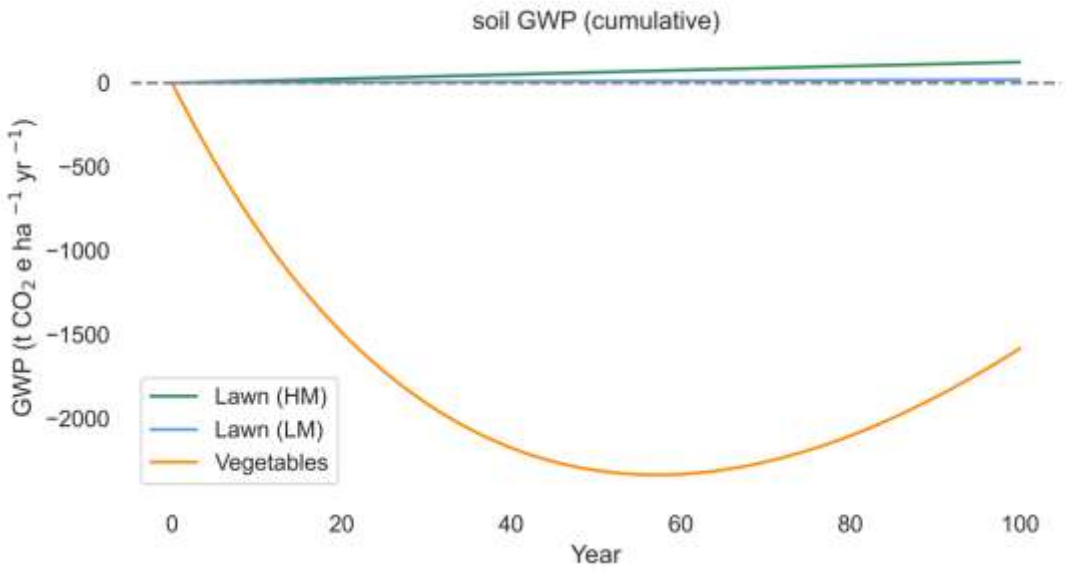

Figure 16 Cumulative soil emissions over 100 years under three different systems: High Maintenance (HM) Lawn, Low Maintenance (LM) Lawn, and Polyculture Vegetable Garden.

3.2 GWP upstream emissions

The four scenarios with upstream emissions modelled are: polyculture vegetable garden with home compost applied (Veggie HC), polyculture vegetable garden with centralised compost applied (Veggies CC), low maintenance lawn (Lawn LM), and high maintenance lawn (Lawn HM). All four scenarios modelled resulted in positive GWP values. The land use with the greatest GWP was the Veggie HC scenario, with compost contributing the most to the total GWP. The Veggie CC scenario resulted in the lowest GWP, followed by Lawn LM, and then Lawn HM (Figure 17). Figure 17 also displays the comparable emissions for conventional agriculture, labelled as Farm Veg. This standardised value has been created as a baseline by which to compare urban agriculture. It is a function of the

GWP of conventional produce (kg CO_2 e kg produce $^{-1}$), and the yields of urban agriculture.

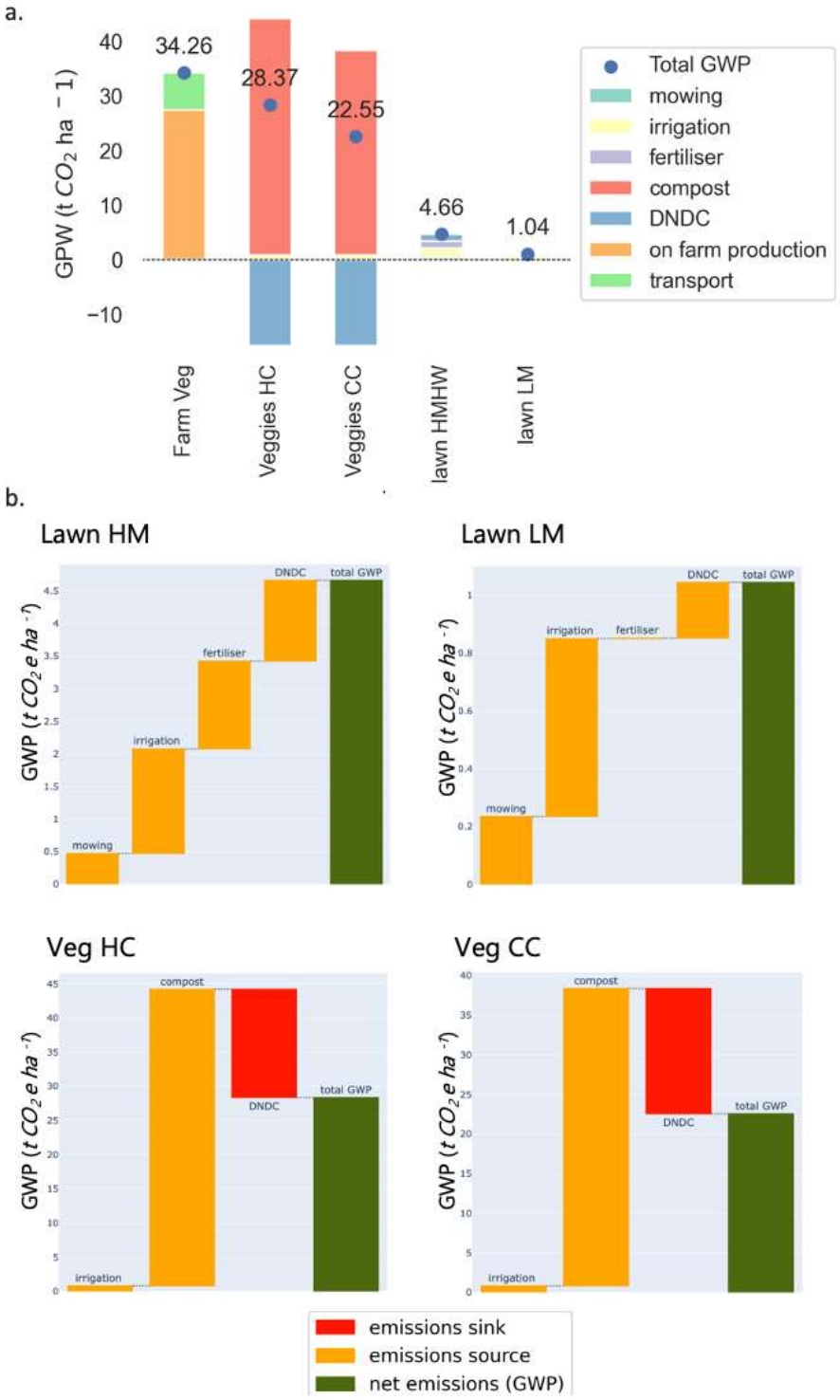

Figure 17 Annual GWP emissions. Panel A illustrates differences in net GWP between four modelled scenarios (Veggies with Home Composting (HC), Veggies with Centralised Composting (CC), High Maintenance (HM) Lawn, and Low Maintenance (LM) Lawn, as well as on farm emissions from conventional vegetable production (value taken from the literature). Panel B illustrates the relative contribution of system processes for each of the modelled scenarios. Y-axes are scaled individually for each plot.

3.3 Functional scenario: GWP of recommended dietary intake

The modelled vegetable garden yields 5.08 kg of produce m^{-2}. Therefore, each kg of produce grown using centralised compost (CC) will emit 0.444kg CO_2 e kg^{-1}. Each kg of produce grown using home composting (HC) will emit 0.559 kg CO_2 e kg^{-1}. One serve of vegetables is 75 g and the Recommended Vegetable Intake (RVI) is five serves per person per day (Dietitians Association of Australia, 2019).

In this scenario, we compare the GWP of purchasing the RVI (base line scenario) or growing the RVI at home. Purchasing the RVI involves importing five serves of vegetables from conventional agriculture and maintaining 27 m^2 of residential lawn (Figure 18). Growing five serves involves no conventional agriculture, and replacing 27 m^2 of lawn with vegetable production. Therefore, the four modelled outputs in this functional scenario are: Veggies HC replacing LM lawn (HC LM), Veggies HC replacing HM lawn (HC HM), Veggies CC replacing LM lawn (CC LM), and Veggies CC replacing HM lawn (CC HM).

Figure 18 also illustrates *avoided emissions*. I.e., in the CC HM scenario, growing zero serves of vegetables at home emits 105.41 kg CO_2 e and growing 5 serves day^{-1} $year^{-1}$ emits 76.43 kg CO_2 e. Therefore, growing five serves each day avoids 28.98 kg CO_2 e per year. These models do not consider other lawn maintained at the property. The net avoided emissions, i.e., net emissions compared to the baseline scenario are outlined in

Table 4.

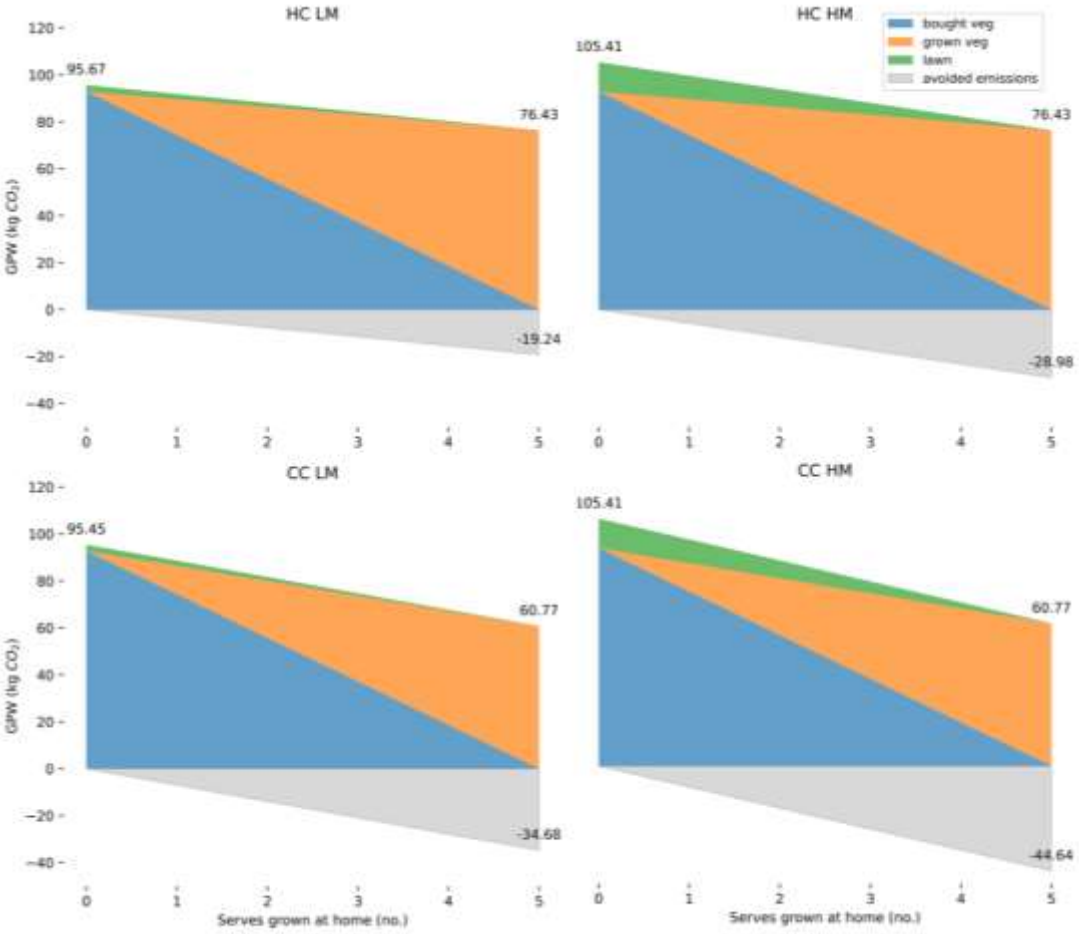

Figure 18 Emissions associated with purchasing an average of 5 serves of vegetables per day (baseline scenario) vs growing an average of 5 serves per day at home for 1 year.

Table 4 Yield based emissions of urban agriculture, compared to the baseline scenario. Baseline = 1kg of bought + 0.20 m2 of lawn. Outcome = 1kg of home grown produce - baseline.

	Outcome (compared to baseline)
FROM HM	kg / kg
veggies HC	-0.21
veggies CC	-0.32
FROM LM	
veggies HC	-0.14
veggies CC	-0.25

3.4 Functional scenario: GWP of a typical household

A typical property in Adelaide contains 71 m^2 of lawn space. Therefore, lawn at a typical property would emit 33.12 kg CO_2 e $year^1$, and 7.41 kg CO_2 e $year^{-1}$, for the HM and LM scenarios, respectively (Figure 19).

Despite the dietary guideline, most people in Australia eat half the RVI (2.5 serves) (ABS, 2022). As such, we consider a household with 71 m^2 of lawn space in two different functional scenarios. In the first scenario, the household grows 2.5 serves per person each day, which replaces 2.5 serves purchased. In the second scenario, the household grows 2.5 serves per person each day, which is consumed in conjunction with the 2.5 purchased to reach 100% of RVI. The change in GWP potential from lawn to vegetable garden across all scenarios are outlined in Table 5.

Table 5 Net change in GWP for every ha of lawn converted to vegetable production – in t/ha. (to convert to kg / m2, divide by 10)

scenario	*FROM HM*	*FROM LM*
veggies HC	-10.56	-6.93
veggies HC no replace	23.71	27.33
veggies CC	-16.38	-12.75
veggies CC no replace	17.89	21.51

Growing 2.5 serves of vegetables per person per day would require 33.5 m^2 of lawn space, or 47% of the land that is currently lawn at a typical property (Hume et al., 2021) (See *section 2.5 Functional scenario analysis* for explanation of yields). Both functional scenarios assume 33.5 m^2 of vegetable garden and 37.5 m^2 of lawn.

A typical household with a low maintenance lawn would emit 7.41 kg CO_2 e $year^{-1}$ in the baseline scenario. If that household grew 2.5 serves of vegetables at home using home composting, then the emissions from the lawn and associated vegetable garden would be 98.96 kg CO_2 e yr^{-1}. If the products of urban agriculture replace vegetables purchased, and avoided emissions are accounted for, then the lawn and associated vegetable garden would become net a GWP sink (Figure 19).

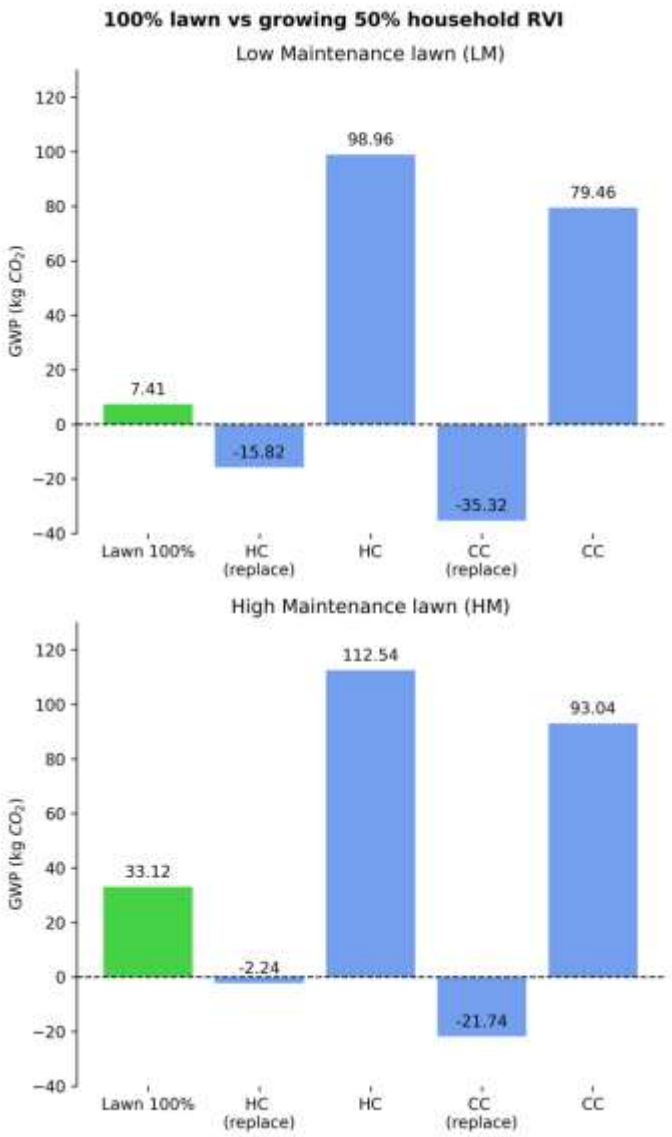

Figure 19 GWP of a typical property which is either subject to high maintenance (HM) or low maintenance (LM) management (baseline). The baseline is compared to the GWP of growing half of the recommended vegetable intake (RVI) through urban agriculture (scenario). In each scenario, 47% of the lawn space is converted to vegetable garden. Scenarios are Home Composing (HC) or Centralised Composting (CC) treatment, and replacing conventional agriculture (replace) or not.

4. Discussion

This paper analyses the GWP of urban agriculture in the form of back yard vegetable production by quantifying the direct soil GHG emission and upstream emissions, as well as emissions from residential lawn. Maintenance (irrigation, fertilizers, composting and lawn mowing) and soil bio-geochemical processes were accounted for. Functional scenarios, which reflect dietary patterns, household make up, and Adelaide's urban form, contextualise the results and investigate the relevance of urban agriculture in achieving sustainability outcomes. Urban agriculture was found to have GWP of 22.55 t CO_2 e ha $^{-1}$ year $^{-1}$ to 28.37 t CO_2 e ha $^{-1}$ year $^{-1}$ (Figure 17). By comparison, conventional agriculture result in emissions of approximately 34.26 t CO_2 e ha $^{-1}$. In order to adhere to the 1.5°C goal of the Paris Agreement, lifestyle emissions in wealthy countries will need to reduce by around 2–2.5 t CO_2 e per capital by 2030 (UNEP, 2020). In the best-case scenarios outlined in Figure 19, the uptake of urban agriculture can achieve 1.10 % to 0.86 % of this emissions reduction target.

Estimating the environmental impact of community-driven urban agriculture is a challenging task. Perhaps this is why previous LCAs analysing urban agriculture systems skew towards highly controlled systems (Dorr et al., 2021). However, it is important to ensure that the model accurately reflects reality. While the model used in this study relies on secondary sources to estimate the GWP of upstream inputs, the large advantage of this work is that it uses observational data on weather, irrigation, yields, and crop type. As a result, the output simulates the reality of a typical home gardener reflecting the outcomes of backyard farming. Despite some uncertainty due to the use

of secondary sources, the reliance on observational data strengthens the validity of the findings.

4.1 Global Warming Potential of Vegetables

The yield based GWP of vegetable gardens ranged from 0.444 kg CO_2 e kg^{-1} produce in the Centralised Compost (CC) scenario and 0.559 kg CO_2 e kg^{-1} in HC scenario (*section Functional scenario: GWP of recommended dietary intake)*. Stone et al., (2021) estimated the GWP of home gardens in Iowa, USA to be approximately 0.25 kg CO_2 e kg^{-1} produce. Conversely, Cleveland et al. (2017) found vegetables to have a GWP of −2.10 kg CO_2e kg^{-1} vegetables yr^{-1}. This single figure factored in replacement of urban lawn, replacement of purchased vegetables, and the diversion of organic waste and wastewater from landfill and treatment facilities. Many of these processes are external consequences, also known as avoided emissions. External consequences are processes (or avoided processes) which are a direct consequence of the urban agriculture system and can be credited within the system boundary (Dorr et al., 2021). Accounting for all avoided emissions in the model, the present research found urban agriculture to have a GWP of -0.14 kg CO_2 e kg^{-1} yr^{-1} to -0.32 kg CO_2 e kg^{-1} yr^{-1} (Figure 18). Given that Australians consume on average 2.5 serves of vegetables per day (ABS, 2022), a typical household could save 23.23 kg CO_2 e to 54.74 kg CO_2 e each year by replacing purchased vegetables with home grown produce (Figure 19). This model did not aim to optimise the sustainability outcomes of urban growing. On the contrary, this work looks at the current practices of urban growers to quantify their environmental sustainability. Given that compost was the largest contributor of GHG emissions in home gardens (Figure 17), there is an opportunity to improve the overall GWP by optimizing compost application.

4.2 External Consequences

External consequences can have enormous effects on the final model result, yet they are inconsistently applied in life-cycle research throughout the literature (Dorr et al., 2021). A 2020 guideline recommend that negative CO_2 emissions obtained from substitution must be clearly stated as being compared to a benchmark, and not as a CO_2 sink over the life cycle of an individual product (Müller et al., 2020). To distinguish the difference, the negative emissions reported in Figure 16 relate to carbon flux in the SOC pool, whereas the negative emissions reported in Figure 18 and Figure 19 relate to avoided emissions compared to a baseline. An important feature of the present study is that it considers how individuals and households consume. Rather than creating a single value to denote the GWP of urban agriculture, here, we create an emissions budget with GWP values for purchased vegetables, lawn, and urban agriculture.

4.3 External Consequences: System Boundaries

The functional scenarios in this analysis incorporate the consumer into the model. Despite the dietary guideline, Australians consume only 2.5 serves of vegetables per day, half of the RVI (ABS, 2022). In light of this, we investigate the outcomes of urban agriculture for a household with 71m ^{2} of lawn space (Hume et al., 2022), considered in two different functional scenarios. In the first scenario, the household replaces vegetables purchased with produce grown at home. In the second scenario, the household consumed home grown produce in addition to vegetables purchased (Figure 19). The first scenario results in a net negative GWP compared to the baseline, whereas

the second scenario, results in a significant increase in GWP. However, it is reasonable to assume that urban agriculture must replace some element of the diet. In a longitudinal study analysing behaviours of urban agriculture participants, Puigdueta et al., (2021) observed that community gardeners reduced their meat consumption over time. Given that vegetables embody little energy compared to other fresh food categories (Clune et al., 2017), a small shift towards a more vegetable based diet can have a large impact. Puigdueta et al. (2021) found that the food related carbon footprint was 12% lower in urban gardeners than in the control, owing primarily to a decrease in meat consumption. Such literature highlights the fluidity of system boundaries, particularly in a system so closely related to consumer behaviour.

4.4 External Consequences: a note on composting and establishing baselines

Composting had the largest effect on the GWP of home gardens. In the HC scenario, composting contributed to 72% of the emissions flux (Figure 17). Recycled materials, such as compost, present complications for emissions modelling as the same material or process is used in different products. Even in Life Cycle Analyses (LCAs) that follow universal guidelines, there are various methods for dealing with recycled inputs with no definitive consensus (Ekvall et al., 2020). The suitability of different methods depends on the requirements of the LCA and its application (Ekvall et al., 2020). In Adelaide, centralised composting is the business-as-usual scenario (BAU). Therefore, the results of this analysis are conservative. An alternate approach would be for emissions associated with collecting, and processing compost to lie within the system boundary of municipal waste management, rather than the urban garden.

In this analysis, a comprehensive approach was taken by including all emissions associated with home composting within the system boundary. This approach differs from other research, which compared the emissions from home composting to the business-as-usual scenario (Cleveland et al., 2017). In the case study conducted by Cleveland et al. (2017), which was conducted in Santa Barbara County (USA), municipal green waste is typically sent to landfill. Therefore, home composting resulted in avoided emissions. However, in the Australian context, a slightly increased GWP was observed for home composting when compared to the BAU, (Lu et al., 2020). Interestingly, if the method employed by Cleveland et al. (2017) had been used in the present study, a contribution of 5.82 t CO_2 ha^{-1} yr^{-1} would have been observed for home composting, rather than the value of 43.36 t CO_2 ha^{-1} yr^{-1} used in this analysis (Figure 17). This would have the result of reducing GWP of the HC scenario by 37.54 t CO_2 ha^{-1} yr^{-1} – making the vegetable garden a net reducer of GHGs.

The model, which is based on observations of gardeners, included only compost as a nutrient source. In a recent survey of Adelaide community gardens, Salomon et al. (2020) found that, *"All sites were committed to principles of organic farming",* with compost being the nutrient source for crops. Similarly, in Baltimore, USA, Santo et al. (2021) found 80% of city farms and gardeners used no chemical fertilisers, 3% used them everywhere, while 17% them some places (n=95).

4.5 Global Warming Potential of Lawn

Establishing the baseline emissions from urban lawn is another important factor in determining the relative GWP of urban agriculture. The GWP of urban lawns were found

to be 1.04 t CO_2 e ha $^{-1}$ yr $^{-1}$ and 4.66 t CO_2 e ha $^{-1}$ yr $^{-1}$ for LM and HM scenarios respectively (Figure 17), which is consistent with findings in the literature. In California (USA), total GWP of athletic field lawns ranged from 4.05 t CO_2 ha^{-1} yr^{-1} in a low fertilization scenario to 7.98 t CO_2 ha^{-1} yr^{-1} in a high fertilizer scenario, while ornamental lawns were found to have lower GWP values ranging from −108 g CO_2 m^{-2} yr^{-1} (-1.08 t ha $^{-1}$) to 285 g CO_2 m^{-2} yr^{-1} (2.85 t ha $^{-1}$) for low and high fertilizer scenarios, respectively (Townsend-Small & Czimczik, 2010a, 2010b). In the present study, neither of the management scenarios resulted in a negative GWP (Figure 16, Figure 17). However, the present study did not retain grass clippings in the system. Retaining the clippings would improve the SOC stocks (Gu et al., 2015). Gu et al. (2015) also simulated GWP in residential lawns managed at varying intensity levels. Results ranged from 0.697 t CO_2 e ha $^{-1}$ yr $^{-1}$ to 2.443 t CO_2 e ha $^{-1}$ yr $^{-1}$. In Sweden, GHG emissions from turf management were found to be 6.2 t CO_2e h^{-1} and 6.8 t $CO2_e$ ha $^{-1}$ at two different golf courses (Tidåker et al., 2017). However, the study included the transport of sand to be applied to the green, which accounted for 2-7 % of the total GHG emissions.

Given that is 4588 ha of lawn has been identified across 491,712 residential properties in Adelaide (Hume et al., 2022), we estimate that residential lawn could be responsible for between 4772 t and 21380 t of CO_2 e each year. To better understand these estimates, it's important to consider how people maintain their lawn. After interviewing 248 households in Nashville (USA), about their lawn management practices, Gu et al. (2015) concluded that households can be categorized into architypes based on intensity of irrigation and fertilization. Half of the survey respondents applied no irrigation to their lawn, while around 45% applied no fertilizer. This approximately corresponds to the LM

scenario in the present study (adjusting for climatic differences between Adelaide and Nashville). Lawn care can be influenced by functional use (i.e., sports turf, golf green, or ornamental), or the presence of children or pets (Carrico et al., 2012). However, lawn care is also associated with individual concerns over reputation, appearance, and social pressures (Carrico et al., 2012). Understanding the prevalence of different architypes in Adelaide would give more accurate estimations of the city-wide greenhouse gas emissions from residential lawns. This work does not consider social and cultural motivations for food gardening, of which there are many.

4.6 Soil emissions (DNDC Modelling)

Soil GHG emissions of both lawn and vegetable gardens were calculated using the Denitrification Decomposition (DNDC) computer simulation, which models carbon and nitrogen biogeochemistry in agroecosystems (Gilhespy et al., 2014). The modelled polyculture vegetable garden was a net global warming sink for the 100-year duration of the simulation, while both lawn simulations were net global warming sources. Comparison of measured and simulated values indicate that the DNDC model generally shows a good performance in simulating SOC stocks (Lembaid et al., 2021). Calibration and validation studied on the DNDC modelling software have found that DNDC effectively estimates cumulative N_2O emissions, however, N_2O estimates are more accurate for scenarios with N fertilization than for those without (Abdalla et al., 2020; Xue et al., 2013). DNDC has not commonly be used to estimate GHG from soils under horticulture. One study found that with out calibration, DNDC proved poor at predicting GHG emission in horticultural peat soils (~35% and ~70% SOM content)(Taft et al., 2019). The SOM in the present work was ~14% so the results are not directly

comparable. However, it does highlight the need for field measurements to properly validate soil GHG emissions.

4.7 Limitations

Taken together, this paper provides necessary insight into the role of urban agriculture in the sustainable cities paradigm. However, there are many practical limitations to urban agriculture which may impede the feasibility of its wide-spread uptake. Previous authors have examined the limitations of labour and space, and have investigated the economic viability of establishing urban food gardens in the Australian setting (Csortan et al., 2020a; Hume et al., 2021, 2022; McDougall et al., 2020). Resources such as land, money and time are not evenly distributed throughout the population, making household self-sufficiency an unattainable goal for some. The model also assumed that the compost demand will be met by urban waste streams. To grow 50% of the RVI, a household would require 902.1 kgs of compost (application rate outlined in section *Amendments)*. In 2019-2018, Australians generated 249 kgs of household organic waste per capita. Assuming a loss of 15.2% biomass, a household could only produce 529 kgs of compost annually. Centralised composting facilities can process and redistribute organic matter form the agricultural and service industries. Centralised composting also provides the benefits of quality control and pathogen screening, reducing the risk of contamination (Buscaroli et al., 2021).

A limitation to the modelling applied in this work is a relance on secondary data sources. Parameters used in the model and justification for values chosen are outlined in supplement Table S1 (Appendix G). The work explores sensitivity to farming practice and

lawn management though the scenario analysis conducted in *Section 3.1* and *Section 3.2.* A thorough sensitivity analysis of parameters used in the model would be an important next step, although outside the scope of the current study.

5. Conclusion

Urban agriculture is a sustainable alternative to conventional agriculture and presents significant benefits in terms of climate change and global warming potential of food production. Urban agriculture could reduce the GWP of produce by -0.14 kg CO_2 e kg $^{-1}$ yr^{-1} to -0.32 kg CO_2 e kg $^{-1}$ yr^{-1}. Based on current dietary behaviours, this equates to household GHG reductions of 23.23 kg CO_2 e to 54.74 kg CO_2 e each year. The findings of this study suggest that the most significant advantage of urban agriculture in mitigating climate change is its ability to replace conventional vegetable production. Without replacing conventional agriculture, land-use change from lawn to urban agriculture does not improve household GWP.

As low-density urban sprawl continues to capture rural ecosystems, the vegetated portion of urban areas is expected to increase (Richards & Belcher, 2020). Policy makers looking to address climate change (SDG 13), food security (SDG2), and sustainable urban development (SDG11), should consider urban agriculture a viable solution. Residential lawn space could provide a significant resource for climate change mitigation through urban agriculture, but opportunities also exist on public land. Governments should continue to support services that facilitate sustainable urban agriculture and encourage knowledge sharing through community gardens. Importantly, the expansion of urban

agriculture initiatives should focus on increasing the share of urban agriculture produce consumed.

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General discussion

Significance

Feeding a rapidly growing population while maintaining ecological integrity is one of the biggest challenges of our time. Urban areas house more than half of the world's population but represent less than 1% of the earth's surface (Clinton et al., 2018). Consequently, urban processes have a disproportionate impact on society and the environment. As the urban population continues to grow, issues of food insecurity, climate change impacts, and sustainable urban development all required immediate attention. Urban Agriculture, the practice of growing food within the urban boundary, is commonly put forth as a solution to these interconnected urban issues. The overarching goal of this study was to explore the plausibility of these claims using the greater City of Adelaide region as a case study.

The opening chapter (Chapter 1) of this thesis demonstrates that urban agriculture has the potential to contribute substantially to sustainable development across the SDGs. Yet, there are concerns that the academic discourse on urban agriculture favours an advocacy viewpoint over critical analysis of the feasibility and practicability at large scale (Neilson & Rickards, 2017; Weidner et al., 2019). Before such an analysis of urban agriculture can take place, the production methods must be clearly defined, as the benefits of one practice may not apply to the full spectrum of activities. In the context of this research, 'urban agriculture' refers to soil-based, open-air production of vegetables occurring on residential land.

A global analysis conducted by Martellozzo et al. (2014) suggested that dedicating one third of the total global urban area to food production could meet the vegetable requirements of urban dwellers. This estimate does not consider the suitability of urban land, or the distribution of potentially suitable land within the urban boundary. However, in suburban cities around the world, residential land represents a significant opportunity for food production. Front and back yard vegetable production also happen to be the most commonly practiced form of urban agriculture in Australia (Donati & Rose, 2020).

Despite this, residential farming tends to be overlooked and understudied in comparison to other production methods. Indeed, private vegetable patches are seen as a hobby rather than a serious farming system, especially within economically developed nations in the Global North. Certainly, it has been unclear whether this kind of production can result in meaningful amounts of food being grown across a city. Residential production does differ from traditional farming systems in several significant ways. As urban farms are (for the most part) tended by hand, and often planted as dense polycultures, there is evidence that they can produce more food on less land (McDougall, Kristiansen, & Rader, 2019). However, there is also evidence of urban growers using resources inefficiently, with perverse outcomes for environmental sustainability (Dorr et al., 2023; McDougall et al., 2019; Salomon et al., 2022).

In addition to land, water is one of the most critical agricultural inputs and an increasingly scarce natural resource, yet little is known about its use in home gardening (Pollard et al., 2018). Unlike traditional farming systems, proximity to urban

infrastructure presents opportunities for mitigating against and adapting to issues of water scarcity, by irrigating urban crops using captured rainwater. Home gardens differ from conventional food systems in other significant ways too: fertilizing nutrients may be delivered through urban waste products, like compost, and food travels significantly shortened distances before it is consumed. All these factors impact the overall sustainability of urban food production, but not always in predictable ways.

Taken together, the above information led to the aims of this PhD thesis, which is a multidimensional analysis of urban agricultural to explore dietary self-sufficiency potential and environmental integrity, using a focal city of Adelaide (Australia). In my efforts to assess the plausibility of urban agriculture producing meaningful amounts of food. I explored four key questions:

1. Is there enough land for household dietary self-sufficiency of vegetables? (Chapter 2)
2. Is there enough rainwater to sustain household dietary self-sufficiency of vegetables? (Chapter 3)
3. Who misses out on urban agriculture, and which community groups have the resources for back yard food production? (Chapter 4)
4. What is the GWP of back yard urban agriculture, and how does it compare to the current food system? (Chapter 5).

In addition to this, I also undertook a collaborative review of the potential for urban agriculture to address the SDGs (Chapter 1). In undertaking this work several broader themes became apparent, as will be discussed in this final chapter.

Contributions to knowledge

Chapter 2 provides new insights into the potential for food production to occur on residential properties. High-resolution multispectral imagery was used to classify residential lawn within the urban landscape, and to quantify land immediately available for vegetable production. Approximately 40,000 properties across three discrete extents within Adelaide were parametrized. Based on gardener surveys published in the literature, three yield scenarios were modelled. Under the high-yielding scenario, 23% of lawn area in a typical block would be required to meet the recommended vegetable intake of residents. Under the medium yield scenario, 72% would be required. Other resources that might limit production, such as labour, water and nutrients, were explored in the discussion. This work was published in the *Journal of Sustainable Cities and Society* (Hume et al., 2021).

Chapter 3 explored the intersection of available land and the supply of agricultural inputs, addressing existing knowledge gaps related to urban agriculture and rainwater harvesting. This work also addressed the need to adapt food production methods to issues of increasing water scarcity. Building on from the previous work, this research quantified the self-sufficiency potential of half a million homes in Adelaide. In the work, the study site was expanded from the three discrete extents investigated in Chapter 2, to cover the entire Adelaide Urban Area. LiDAR imagery was combined with high resolution multispectral imagery and climatic variables to quantify not only growing space, but rainwater harvesting and storage potential as well. Around two thirds (65%) of residential properties were found to contain enough available land to provide dietary

self-sufficiency of vegetables, while capturing and storing adequate rainwater for irrigation, even in the modelled Dry year scenario. The modelled edible garden and associated storage tank would occupy around half of the lawn space in a typical residential block. This has also been published in the *Journal of Sustainable Cities and Society* (Hume et al., 2022).

The first two research articles examine space as a potentially limiting factor for food production. Adelaide's urban extent contains 4,588 ha of residential lawn (Hume et al., 2022). Given that 3,312 ha of land would be required to meet the recommended vegetable intake of Adelaide's urban population (Table 6), the recommended vegetable intake could be met by converting 72% of the residential lawn in Adelaide's urban extent to in ground vegetable production (Table 6). However, the resource of available land is not distributed evenly across individual properties. Chapter 4 explores the distribution of self-sufficiency potential across the city and asks the question "who has the resources to be successful in urban agriculture, and who misses out?" This analysis compared self-sufficiency potential to financial stress indicators and identified specific areas where vulnerable communities do not have access to resources required for household self-sufficiency. Bespoke spatial analysis identified specific opportunities and constraints within individual suburbs that can inform targets and policies that promote urban agriculture and support the most vulnerable members of the community.

Table 6 Adelaide SS potential

yield (kg m2)	serve (kg)	serves/m2	people	serves/year	ha needed
5.08	0.075	67.73	1,229,280*	2 243 436 000	3312

**Population = no. of properties * number of people in a typical property (2.5).*

Community-led urban agriculture is often studied for its contributions to social dimensions of urban sustainability, such as food security and economic inclusion, while in parallel rigorous environmental assessments examine highly controlled high-tech farming systems. To address this gap, Chapter 5 took stock of the GWP of home gardening using a mixed method model that incorporated DeNitrification DeComposition (DNDC) modelling, with a Life Cycle Analysis (LCA) approach. Urban agriculture was found to be a sustainable alternative to conventional agriculture and presented significant benefits in terms of climate change and global warming potential of food production. In the best-case scenario, the uptake of urban agriculture could achieve 2.15% to 2.76% of the emissions reduction target for households, as is outlined in The Paris Agreement. The most significant advantage of urban agriculture in mitigating climate change is its ability to displace conventional vegetable production. Without replacing conventional agriculture, converting residential lawn (turfgrass) was not found to improve a household's global warming footprint (Chapter 5).

Emerging themes

The core question at the heart of my thesis is: Is Urban Agriculture as a meaningful source of food a plausible reality, or just a nice idea? In undertaking this work, several

intersecting themes emerge. The following sections discuss the overall significance of these themes, and future directions of the topics explored.

Optimisation (advocacy vs critically analyse)

Residential food gardens differ from highly controlled conventional farming systems. As they are managed by individuals, resource use and efficiency are not likely to be optimised. The variability makes it hard to model the inputs and outputs with a high degree of certainty. An on-going question that arose when conducting this research was "*am I trying to show the best case scenario, or simply record and expand on the behaviours of growers"*. Ultimately, I tried to account for both. At the project design scale, this research takes a highly conservative approach to measuring the self-sufficiency potential as it only considers residential land and open air, soil-based production - there exist a whole suite of production methods (e.g., roof tops, indoor cropping) and untapped physical resources not explored in this research. This was a conscious decision, which aimed to highlight that even with little capital investment, and dedicating only a small portion of urban land to food production, meaningful outcomes could be achieved. However, within these conservative bounds, it was soon apparent that optimised high yields would be a precursor for dietary self-sufficiency. In the best-case scenario, 23% of lawn in a typical block would be required to achieve self-sufficiency of vegetables, however, modelling using the low yield scenario required requires approximately 15 times more land than is available in a typical block. Both the "high yield" and "low yield" scenarios were taken from observational surveys of urban growers in Adelaide (See Chapter 2).

As outlined in the literature review in Chapter 1, sensitively applied, urban agriculture has great potential to help ameliorate pressing issues of our time. However, a literature review can go only so far. Therefore, in the latter chapters, a more pertinent (and in this author's opinion, interesting) question to ask is *do the current practices of urban growers realise this potential?* A key advantage to this work is that models are built using variables that reflect the actual behaviours of growers. This was made possible by a few key resources that observed how urban gardens were managed (Csortan et al., 2020; Salomon, pers. comm., 2021). In Chapter 3 and Chapter 5, the crop water footprint of urban agriculture reflects irrigation applied to urban farms in Adelaide. Similarly, in Chapter 5 compost application reflects the application rates seen in community gardens in the focal region (Adelaide). This is important because unlike in commercial farms, which are operated primarily for profit, inputs into home gardens are not necessarily applied based on a complete understanding of endogenous soil nutrient levels and crop requirements. Other researchers have documented the oversupply of nutrients by urban growers, and reported on the impacts to soil health, and long-term nutrient imbalances (Salomon et al., 2022). Here, I took a different approach - quantifying the embodied emissions in compost and water applied to gardens. Even without optimising resource use, urban agriculture showed significant benefits over traditional farming (Chapter 5).

This work provides much needed insights into environmental sustainability of low-tech urban farming through the lens of CO_2 e and GWP. While GWP is an important metric for understanding the environmental impact of a system or product, it is not the only factor to consider when evaluating the sustainability of a system like domestic rainwater

harvesting and composting. For example, of the inputs modelled in Chapter 5, home composting resulted in most CO_2 emissions. As such, the oversupply of compost also had significant impact on the associated GWP of urban produce. However, compost is commonly employed as a sustainable initiative in urban spaces as it conserves resources and reduces waste. Notably, in cities without centralised composting, home composting also dramatically decreases the greenhouse gas emissions of municipal waste management by diverting green waste from landfill. In a similar vein, rainwater harvesting provides many benefits to urban spaces, including decreased water abstraction, and adapting to issues on increased water scarcity. However, small scale domestic rainwater tanks can consume a lot of energy to build, transport and replace, particularly if they are reinforced with concrete. This affects their overall GWP, and in practice, researchers have found rainwater harvesting systems to perform worse than mains water in some metrics (Ghimire et al., 2017). All of this is to say that, while metrics such a GWP provide a useful snapshot, they can erode the nuances required to fully understand sustainability. An important advancement in this area would be to perform a thorough life cycle assessment of home gardens using software such as SimaPro or Open LCA. These software have the capacity to measure impacts beyond GWP. The importance of different impact categories, e.g., GWP and freshwater abstraction, may change over time or location.

System boundaries

Scientific research often aims to isolate and understand a single process or event. However, the system boundaries relevant to urban agriculture are obscure. Chapter 5 discussed system boundaries in a very practical sense. Life Cycle Assessments require

clearly defined boundaries to determine which emissions are attributed to one process, and which are omitted. In this work, it was evident that the boundaries drawn by the researcher can have a profound impact on the reported sustainability outcomes. To return to an earlier point, the goal of this research was not to unequivocally advocate for backyard urban gardening, but to examine the practice for what is it, and identify potential inefficiencies that arise. Therefore, in Chapter 5 all emissions associated with home composting and centralised composting were attributed to urban agriculture.

Another boundary was drawn around the household as a singular unit. Only land within the property boundary was available to the household. In reality, there are opportunities for produce-sharing between households, and for households to use street verges or other public greenspace for food production. The benefits of such activities go well beyond food production and sharing, including improved outcomes for the socially isolated (See Chapter 1). Similarly, in Chapter 3, a boundary was drawn between water use in the garden and water use inside the home. Yet integrating rainwater harvesting with multiple domestic uses could increase its economic efficiency and reduce buy-back time. Other boundaries are more fluid. For example, the very act of engaging in urban agriculture can change a consumer's diet, or increase pro-environmental activities (Kim, 2017). Exploring these interactions will be important next steps in advancing this area of research.

The value of GIS

Remote sensing and GIS techniques proved to be valuable tools. These modelling techniques allowed me to audit the city, find available resources, and correlate

outcomes to important social indicators. The land use classification used to identify lawn combined high resolution spectral and LiDAR imagery – techniques that have been used in the past to model urban tree canopy cover. While these techniques were relatively sophisticated, the true novelty of this work comes from the application.

A fascinating question explored in this work is *who has access to the resources to be successful in urban agriculture?* In Chapter 4, it was found that the most vulnerable communities (as determined by housing stress) have significant potential to produce food in their own back yards. Donati and Rose (2020) reported that lowest income earners are the demographic most likely to grow a significant portion of their diet themself. This suggests the idea of individuals who are 'asset rich' (i.e., have access to land), but are income poor as an important demographic in the context of urban agriculture. However, from Chapter 3, we learn of the potentially prohibitive cost of rainwater tanks, which would help to mitigate and adapt to issues of water scarcity. Therefore, vulnerable households seeking food security through urban farming may not be contributing to the other dimensions of sustainable development so often attributed to urban agriculture.

The role of planners

Having the capacity for self-sufficiency will retain the resilience of the urban space. In times of prosperity, food security does not seem like a pressing issue. This is true for most people in Australia. While the COVID-19 pandemic is not the only reason that urban agriculture is a worthwhile and relevant topic in today's world, it did highlight the fragility of our food systems. In response, many people turned to urban farming. The

economic effects of the pandemic are still being felt, with on going effects on food security.

Planning will have an important role in retaining the open space critical to urban food production. Currently, a trend towards smaller block sizes with less open space is threatening the long-term prospects of residential urban agriculture. Retaining yard space is not likely to serve the financial motivations of private developers, and so interventions may be required to legislate for potentially productive land to be retained in new builds. Some suburbs, such as the Adelaide CBD, have ample public green space that could service the nearby apartments. These opportunities were not explored in this Thesis directly, but the spatial analyses outlined in Chapter 4 illustrate the information and decision making that could inform such policy. Mixed use urban developments that combine high-density dwellings with shared open space would facilitate urban agriculture. However, unchecked, urban densification will continue to erode urban food production capacity.

Urban agriculture has been largely overlooked by planners in Adelaide. The 30-year greater plan for Adelaide, established in 2017 (Government of South Australia, 2017) aims to make Adelaide *A green liveable city.* The target is achieved through maintaining native habitat, supporting biodiversity, increasing tree canopy cover, provisioning spaces for recreation and connection, and increasing permeable surfaces for stormwater management. Meanwhile, the plans only reference food producing capabilities in the context of urban sprawl and encroachment into surrounding agricultural areas. This is a missed opportunity to promote urban agriculture, which would contribute to priorities

set out by the plan, including climate change resilience; social inclusion and fairness; environment protection, restoration and enhancement; natural resource management; and healthy, safe and connected communities. The findings of this Thesis can be used to guide policy and promote sustainable urban development that includes urban agriculture.

A note on lawn

While the focus of the work is urban agriculture and food production, the results also reveal many interesting insights into lawn as an urban land use. Importantly, this work audits the amount of residential lawn found in Adelaide's urban extent – 4,588 ha. Lawn holds cultural and social value and is ubiquitous throughout modern urban landscapes. The average residential property was found to have 71m^2 of lawn space, which, as reported in Chapter 5, could emit between 7.41 kg of CO_2 e and 33.12 kg of CO_2 e annually. Across the entire city of Adelaide, residential lawn could emit between 4,772 t and 21,380 t of CO_2 e each year and require between 6.7 billion L and 21.3 billion L of irrigation. We make the case that urban agriculture would be a more valuable use of land. However, as has been established, there are barriers to urban agriculture uptake. Perhaps another question could be – *what other urban vegetation could substitute for lawn while providing ecological services?* Native grasses, for example, would reduce water use and support native fauna, while large trees provide shade and long-term carbon storage.

Concluding remarks

Urban agriculture can create positive economic, social and environmental impacts. However, to make it work, we need to determine its feasibility on a large scale. This work takes a critical lens to the feasibility, practicability, and sustainability of low-tech urban farming. It is my central thesis that low tech urban agriculture should be viewed as a serious farming system. Operated by individuals, at-home gardens differ from commercial farms. These differences affect their resource use and productivity and are important to study.

This work provides new insights into residential urban agriculture. In this case study, there is more than enough residential lawn space to supply an entire city's population with the recommended daily vegetable intake. Most individual households have adequate land to achieve household self-sufficiency of vegetables. Importantly, self-sufficiency could be achieved irrigating crops with roof-fed rainwater tanks. Chapter 4 works as an excellent case study, demonstrating the power of land-use classification and GIS modelling to provide valuable insights for policy makers. These insights can help target bespoke interventions that promote urban agriculture where it is most needed.

The findings illustrate the significant potential for residential urban agriculture to contribute to sustainable development through food provisioning, water regulation, and climate change mitigation. Together, the results make a compelling case for urban agriculture and invite future work to address social and economic barriers. While the work employs one city as a case study, the results are relevant to low-density urban

spaces, or suburban areas associated with larger cities across the world. An important next step will be to perform similar analyses in other cities and climates.

We have enough land for urban agriculture to be a key player in Australia's future food security. A typical household could grow the recommended vegetable intake, while capturing and storing enough rainwater for irrigation. Even accounting for the oversupply of inputs that is common in community-led urban agriculture, back yard farming could lower the environmental impact of food production. By tapping into the vast potential of urban agriculture, we can pave the way for a more environmentally friendly and food-secure future.

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Appendix A

The following text is an unmodified literature review completed in July 2020. The literature review constitutes the formal research proposal submitted to the University of Adelaide as part of the Core Component of the Structured Program (CCSP) for the Doctor of Philosophy.

The unmodified literature review is included as an Appendix to the thesis in the interest of being exhaustive. It does not systematically cover all topics relevant to research articles (Chapters 2-5). A review of up-to-date literature and relevant literature can be found in the introduction of each research article, and in the review article included Chapter 1: *The Case for Urban Agriculture.* Finally, the project aims cited at the conclusion of the literature review evolved over time, and do not accurately reflect the aims of the thesis in its entirety.

Literature Review: Modelling the ecosystem services of urban agriculture

Urban areas house more than half (50%) of the world's population but represent less than 1% of the earth's surface (Clinton et al., 2018). Consequently, urban processes have a disproportionate impact on both society and the environment. In recent times, there has been intense interest in the adaptive capacity, habitability, and environmental burden of cities. Urban agriculture, the practice of growing food within the urban boundary, is a common theme in the literary discussion of urban sustainability. As will be the topic of this review, urban agriculture has the potential to deliver a range of

ecosystem services to urban environments that go far beyond the primary function of food production.

Urban Agriculture is a universal term which can refer to: animal husbandry and aquaculture, allotments or rooftop gardens, building-integrated greenhouses and complex indoor 'plant factories' (Mok et al., 2014; Weidner, Yang, & Hamm, 2019). Food plant cultivation, sometimes referred to as urban horticulture, is the most common form of urban agriculture (Mok et al., 2014), and the focus of this discussion. Food production that utilises climate control, or rooftop farming requires capital to establish suitable infrastructure (Goldstein, Hauschild, Fernandez, & Birkved, 2016; Mohareb et al., 2017). Therefore, this research is focused on relatively small-scale food gardens, that utilise soil-based, open-air production. Focusing on nutrient dense foods (i.e., vegetables), rather than calorie dense foods (e.g., grains) is considered a more efficient use of land available for urban agriculture as vegetables are high value, perishable crops (Martellozzo et al., 2014; Weidner et al., 2019). Fruit trees are not entirely excluded from this review but are not the primary focus of the research.

Using this definition, I will investigate the interaction between urban agriculture and the environment, using *ecosystem services* as a framework to guide the discussion. This will in turn lead to more specific aims using Adelaide (Australia) as a case study. It is beyond the scope of this review to discuss the economic and social aspects of urban agriculture (e.g., poverty alleviation and localised economies). However, it is important to acknowledge that communities engage in urban agriculture for reasons beyond environmental sustainability.

2. Provisioning

Provisioning services are the products obtained from ecosystems. In the case of urban agriculture, provisioning services relate to the capacity of food production (specifically vegetable production according to our definition, under other definitions it can extend to animal derived food products and inedible fibres). Analysing food provisioning services and spatial potential for urban agriculture is an important first step to understanding other ecosystem services. For example, the extent to which urban agriculture can displace rural farming determines how much of that land can be reclaimed as native vegetation. Additionally, establishing the capacity for urban agriculture in any given location will inform the water and nutrient requirements. Food provisioning potential is best modelled through self-sufficiency: commonly expressed as the capacity to satisfy some part of the human diet - but can also be modelled using economic parameters. Either way, self-sufficiency models generally rely on yield estimates, spatial modelling, and population metrics.

2.1 Yields

Yield estimates for in-ground urban vegetable production vary both within, and between observational studies (Algert, Baameur, & Renvall, 2014; Gittleman, Jordan, & Brelsfold, 2012; McDougall, Kristiansen, & Rader, 2019). Despite their variability, urban food gardens have been shown to produce higher yields than conventional practices (CoDyre, Fraser, & Landman, 2015; McDougall et al., 2019; Zainuddin & Mercer, 2014). This is because small scale urban food gardens are tended to manually, which allows for higher

cropping density and intercropping with different species (i.e., biointensive methods). However, given the multi-functionality of many urban food gardens, the high yield potential of urban gardens is often not fully realised. CoDyre et al. (2015) noted that high yields will only be achieved through careful management to ensure garden efficiency is maximised in terms of space. More high-tech urban agricultural practices such and vertical, or indoor farming will produce much higher yields in terms of space (Goldstein et al., 2016), however these are not included in this review.

Given the discrepancy between typical urban agriculture yields and those from commercial farms, empirical data from polyculture urban food gardens in Australia would be preferable for the self-sufficiency model being built. Salomon, Watts-Williams, McLaughlin, and Cavagnaro (2020) characterised urban agricultural sites in Adelaide in terms of plant diversity, soil contamination, soil fertility and mycorrhizal abundance. However, there remains a lack of published yield data for the area. Productivity estimates of urban food gardens instead must come from data collected in Sydney and published by McDougall et al. (2019). It is expected that Adelaide would experience similar annual yields to Sydney, as both cities have relatively temperate climates, allowing for year-round crop production. However, Sydney experiences significantly higher rainfall than Adelaide (Bureau of Meteorology, 2020a, 2020b). McDougall et al. (2019) surveyed thirteen urban food gardens over the course of one year. The mean yield from all gardens was 5.94 kg m^{-2}, with a minimum of 1. 99 kg m^{-2}, and maximum of 15.52 kg m^{-2}. While vegetables made up most of the produce, the gardens also produced fruits, nuts and herbs - yet the exact proportions of crop types were not disclosed (McDougall et al., 2019). Given these caveats, commercial yields from the area around

Adelaide can also be included. This will give a robust yet conservative estimate of productivity.

2.2 Spatial modelling (remote sensing techniques)

Remote sensing allows landcover to be classified accurately and efficiently and can be used to find potentially productive land in urban environments. Remote sensing of urban areas has applications in decision-making and planning, disaster response, and sustainable development, making it the subject of much research (Gong, Liu, & Huang, 2019). The land cover that will be most easily converted to soil-based vegetable production is urban lawn (sometimes called turf grass). The Normalised Difference Vegetation Index (NDVI) is a common index used to map vegetation extent. It can be derived from multi spectral data with reflectance values for Near Infrared (NIR) and Red (R) wavelengths (Curran & Steven, 1983). NDVI produces a value between -1 and 1 and can be used to produce thematic classifications. Alternatively, vegetation can be mapped via a supervised classification. This technique classifies each pixel in an image based on spectral similarities or differences to a set of training classes, which are determined by the practitioner (Xie, Sha, & Yu, 2008). Unlike the NDVI, supervised classifications take into account reflectance in all bands, and can identify a range of surfaces, not just photosynthetic vegetation. However, a challenge when classifying urban vegetation is the spectral similarity of functionally different vegetation types (such as lawn and trees) (Gong et al., 2019; Kremer & DeLiberty, 2011; Small, 2005; Wetherley, Roberts, & McFadden, 2017).

Digital Surface Models, derived from Light Detection and Ranging (LiDAR) data, provides textural information describing variance in elevation amongst adjacent pixels. Therefore, LiDAR is useful in distinguishing between spectrally similar, but structurally distinct surfaces, and is often combined with spectral information to classify vegetative structures (Abed, Mills, & Miller, 2014; Anderson et al., 2018; Bartesaghi-Koc, Osmond, & Peters, 2019; Iovan, Boldo, & Cord, 2008; J. Richardson & Moskal, 2013; Voltersen, Berger, Hese, & Schmullius, 2014; Walton, Nowak, & Greenfield, 2008). LiDAR data has been used to create 3D models of urban landscapes (Anderson et al., 2018) and indicate the boundary of vegetation types within a vegetated area (Iovan et al., 2008; Voltersen et al., 2014). Depending on the density of points, it can also estimate the size and extent of tree canopies (Bartesaghi-Koc et al., 2019).

2.3 Review of self-sufficiency publications

Remote sensing and GIS have been used to varying degrees to model self-sufficiency in the past (Table 7). The results are varied, due to a variation in the data used, and growing scenarios modelled (Table 7). Rather than develop a remote sensing classification, it is possible to use city planning data and GIS, to determine residential block size, as well as the number and size of vacant lots. For example, Grewal and Grewal (2012) modelled self-sufficiency potential via scenario analysis that assumed 10% of residential blocks, and 80% of vacant lots could be used for food production. This assumption does not account for different configurations of land-use within a single block, nor does it assess the suitability of garden beds and lawn for food production. Alternatively, (McDougall, Rader, & Kristiansen, 2020) employed remote sensing to exclude building footprints from residential blocks and infer yard space. Doing so

revealed that Sydney's entire vegetable supply could be produced through urban agriculture if all street verges, miscellaneous blocks (e.g., vacant blocks), and 25% of total yard space was employed. However, as different ground level surfaces within properties were not identified, impervious surfaces, such as driveways, were not excluded from the analysis.

Table 7 a synopsis of previous studies that modelled the potential urban agriculture output form in ground vegetable production within a city or region. Studies that modelled fruit tree production exclusively, or focused primarily on rooftop and indoor farming methods, are not included in the table, nor are studies that only model current urban agriculture production

Study	Maximum projected outcome*	Growing Scenario	Yield data	Land data
Grewal and Grewal (2012) Cleveland, USA	Between 31% and 68% of the required fruits and vegetables, 94% of both poultry and eggs, and 100% of honey could be supplied.	10% of total residential blocks, and 80% of vacant lots could be used for food production^	Secondary data from both US Department of Agriculture statistics for rural areas and data for urban environments derived primarily from the literature.	City planning data for land ownership and block size.
CoDyre et al. (2015) Guelph, Canada	49 % of the population's current fruit and vegetable consumption (11.6 % of total recommended calories) could be supplied.	10 % of residential 'yard' space (max scenario).	Primary data from 50 backyard gardeners	'yard' space = residential block minus the size of the buildings.
Colasanti and Hamm (2010) Detroit, USA	65% of vegetables and 39% of the fruits currently consumed could be produced (not the recommended intake – which far greater than what is being consumed).	Only fully vacant parcels were included.	Secondary data from published biointensive methods.	Vacant parcels were identified from the City of Detroit dataset.
Haberman et al. (2014) Montreal, Canada	128.4% of the total vegetable supply for residents.	All of yard space	Applied *yield ratios* to Statistics Canada data for conventional vegetable yields in Québec. (i.e. high yield = conventional yield*3.5 based on literature)	Sketched the yards from randomly sampled residential block in Google Earth for low-, medium- and high-density residential housing areas.
Hara, McPhearson, Sampei, and McGrath (2018) New York, USA	6.7 % of vegetable consumption (1% of total calories)	Current community gardens, vacant lots and land classified as grass/shrub and bare surface (e.g. public parks and verges).	Average of output across all community gardens divided by total community garden space.	NYC Planning data to identify vacant lots, Gov. land-cover data with grass/shrub classification/

Study	Maximum projected outcome*	Growing Scenario	Yield data	Land data
Johnson, Lathuillière, Tooke, and Coops (2015) Vancouver, Canada	5% of total calories could be met – using beans as a reference crop, or 37% of the vegetable requirements.	All open vegetated surfaces (within a 1km density populated area)	Combined literature values for species specific crop yields and modelled dry matter production values.	LiDAR remote sensing data used to classify vegetation based on reflectance (number or returns, and intensity of NIR reflectance).
Hara, Murakami, Tsuchiya, Palijon, and Yokohari (2013) Quezon City, Philippines	5.1 % of total calories could be met (per capita consumption based on household surveys rather than recommended intake)	All areas classified as grassland in the study site (one 32 km^2 suburban neighbourhood).	Primary data from a single urban farm in the area.	Object based image analysis (2m resolution) to classify landcover and identify grassed areas.
McClintock, Cooper, and Khandeshi (2013) Oakland, USAz	60 % of *actual* vegetable consumption (8.8 % of total caloric intake) or 12.7 % of recommended vegetable intake could be met.	75% of 'underutilised' public land and vacant private lots (including lots that appeared vacant or contained lawns, fields, and other open spaces, and parking lots that appeared to have been abandoned) -assuming 25% taken up by UA infrastructure.	Secondary data from published biointensive methods.	Alameda County Tax Assessor's parcel data obtained from the City of Oakland's GIS database, in conjunction with visual interpretation to select parcels.
McDougall et al. (2020) Sydney, Australia	entire vegetable supply, or 34 % total food supply, could be produced	75 % of yard space plus all street verge and miscellaneous space identified as 'suitable'	Primary data from 13 gardeners in the Sydney area. (Data was for vegetables, nuts and fruit gown together)	Government provided spectral imagery and land ownership data.
J. J. Richardson and Moskal (2016)	Less than 1% of the population could be ben on an entire vegetarian diet (including nuts and grains and fruit trees).	All existing lawn space occurring in single family residential homes used for food production.	Yields retrieved from National Agricultural Statistics Service	Object Based Image Analysis (1m resolution), using aerial imagery and aerial LiDAR to quantify grassed area.

*maximum projected outcome that sits within the scope of this review. For example, high land use and yield scenarios, excluding production on rooftops, indoor farms, or under climate control production (e.g., polytunnels and glasshouses).

Many researchers have assessed the suitability of vacant lots, existing community gardens, or other public green spaces within municipals. By contrast, residential food gardens have received little attention, and are very rarely investigated as a stand-alone resource (Table 2). Haberman et al. (2014) found that residential gardens in Montreal, Canada, could provide more than enough space to meet the daily vegetable requirements of residents. However, this conclusion was drawn from extrapolating from a small selection of randomly sampled homes. Both Kremer and DeLiberty (2011) and Richardson and Moskal (2016) successfully used spectral data to perform a remote sensing land-use classification and identify residential lawn to be reclaimed for urban agriculture. Of the two, only Richardson and Moskal (2016) went on to measure self-sufficiency potential, using Seattle, USA as a case study. However, the measure of self-sufficiency was a complete vegetarian diet, which included barley as a source of carbohydrate. Using all residential lawn in single family homes, less than 1% of the city's food needs could be met. This reiterates an earlier point that producing vegetables may be the most sensible use of urban agriculture, as other dietary elements (e.g., grains) can easily be imported (Martellozzo et al., 2014; Weidner et al., 2019).

Taken together, remote sensing can play an integral part in accurately identifying available land to be used in urban agriculture. Determining the extent of available land reveals the food provisioning potential of a given area. On its own, food provisioning is an important ecosystem service, especially in areas of poor food security (Mok et al., 2014). However, the extent and productivity of urban agriculture, i.e. the scale at which urban agriculture can occur, will also affect the operation of other ecosystem services. Therefore, quantifying these parameters is the first step in ecosystem service modelling.

3. Regulating service

3.1 Water regulation

Abstraction of water for irrigation or domestic use detracts from environmental flows and obstructs ecosystem functioning of rivers and wetlands (Grizzetti, Lanzanova, Liquete, Reynaud, & Cardoso, 2016). Water consumption in urban agriculture is not well studied (Pollard, Ward, & Roetman, 2018b). Both residential food gardens and community gardens share water meters with either the home or adjacent building (Ward, Ward, Saint, & Mantzioris, 2014). This makes it extremely difficult to distinguish between water used in the home (e.g., showering), on ornamental gardens (e.g., watering a rose bush), and on food crops. In response, Ward et al. (2014) created a '*theoretical irrigation footprint*' for domestic food gardens. Food gardens are diverse polycultures (McDougall et al., 2019), however, urban gardeners are likely to irrigate a polyculture uniformly to satisfy the crop with the highest water demand (Ward et al., 2014). Therefore, Ward et al. (2014) discerned that urban food gardens are likely to be less water efficient than commercial monocrops. Furthermore, mains water, commonly used in urban agriculture (Pollard, Ward, & Roetman, 2018a), has a greater embodied energy than the lower grade irrigation systems used in commercial agriculture (Mohareb et al., 2017).

At the same time, the urban environment is a source of contaminated stormwater runoff, which poses a significant ecological threat to waterways by altering the volume, pattern and quality of flow (Walsh, Fletcher, & Burns, 2012). Widespread impervious surfaces, which characterise urban landscapes, result in unnaturally large quantities of

run-off. The volume of stormwater entering waterways can be reduced by increasing urban infiltration, evapotranspiration, and capture of stormwater for urban use. Australia's Cooperative Research Centre for Water Sensitive Cities (2020) does extensive research on water sensitive design in cities, with case studies on specific design practices at various scales. Comparatively, urban water capture for use in urban agriculture receives little attention. However, many of the design principles are transferable. For example, *rain gardens* are designed bioretention and filtration systems for storm water runoff (Davis, Hunt, Traver, & Clar, 2009). While raingardens are not commonly comprised of edible plants, purpose-built edible rain gardens can produce similar yields to traditional urban food gardens (Richards, Farrell, Tom, Williams, & Fletcher, 2015). In this way, urban agriculture can help to reduce stormwater runoff, while simultaneously reducing the need for rural water abstractions.

Stormwater refers to water captured from all impervious services; however most household storm water catchment occurs from roofs. Rainwater tanks are associated with some chemical and biological hazards that pose human health risks. These include the use of lead paint on roof surfaces, and some reports *Salmonella* outbreaks from contaminated stormwater (Environmental Health Committee, 2010). Nonetheless, The Environmental Health Committee (2010) declare that under most circumstances Australian roofed rainwater tanks can be used for drinking, food preparation and garden watering.

Using LiDAR point clouds to create building footprints, rates of precipitation and evaporation, McDougall et al. (2020) calculated that amount of water that could be

collected from rooftop-fed rainwater tanks in Sydney (Australia). This captured rainwater met the irrigation requirements of urban agriculture in all of the scenarios modelled by McDougall et al. (2020), however, the model assumed that the water requirements of urban food gardens would match that of commercial vegetable production in the surrounding basin. A metric specific to polyculture food gardens, such as Ward et al. (2014) '*theoretical irrigation footprint'*, may be a more accurate. Finally, McDougall et al. (2020) found that roof-fed rainwater tanks didn't deliver water uniformly across the city, in which case water would need to be transported requiring energy and infrastructure.

Taken together, urban food gardens align with water sensitive design principles by intercepting and capturing storm stormwater. Harnessing and repurposing stormwater would also reduce the need to abstract water from rural river systems. However, reasonable steps should be taken to minimize the risk or urban pollution and other contaminants (Environmental Health Committee, 2010).

3.2. Waste management

By recycling urban nutrients on site, urban agriculture provides the opportunity to *close the loop* on urban metabolism. Circular metabolism involves the localised collection and treatment, of waste (solid waste and wastewater) to recover resources such as water, nutrients or organic matter. Currently, cities, and their associated agricultural regions, consume and dispose of nutrients in a linear fashion, leading to global resource depletion (Cordell & White, 2011). Recycling the nutrients found in urban waste would elevate the burden of waste management and provide a sustainable nutrient source for

food production - creating a symbiosis between urban agriculture and the city. The following paragraphs will discuss the feasibility, opportunities and challenges of recycling various forms of urban waste.

In Australia, households produce more organic waste than any other industry or sector. In fact, 51% of all waste generated by households is organic waste. Food scraps and garden organics are the two main type of household green waste; however, garden organics are much more likely to be recycled than food waste. The ABS reports that each year 2,623,494 kg of food waste is sent to landfill, compared to 285,911 kg which is recycled (an additional 1,038,172 kg is used in energy recovery). A kerbside bin audit conducted in one Adelaide council region found that less than 10% of food waste was correctly presented in the *organics* bin (the study was examining methods to increase green waste recycling, this was the pre-trial baseline) (Gallou, Rawson, & Heinrich, 2019). The ABS use the municipal solid waste stream to calculate its annual waste accounts. This is waste collected from households through kerbside waste and recycling collections, meaning the actual amount of food sent to landfill is likely to be more than the figures reported by the ABS (Gallou et al., 2019).

The decomposition of green waste in landfill causes anerobic decomposition of organic matter, resulting in methane production, a highly potent GHG (Hellebrand, 1998). However, food scraps, yard waste, and paper can all be composted at the household level, allowing nutrients and organic matter to be recovered (Vázquez & Soto, 2017; R. Wielemaker, Oenema, Zeeman, & Weijma, 2019). As well as diverting food waste from landfill, home composting avoids indirect GHG emissions by eliminating collection and

transport of green waste products. Using national waste data from the ABS (which is likely to be conservative), McDougall et al. (2020) calculated that the average Australian could produce 156.88 kg of compost per person each year. This would be enough compost to satisfy approximately 10 m^2 to 20 m^2 of land, depending on the quality of the soil. The US Composting Council recommends 8 kgm^{-2} yr^{-1} compost (Cleveland et al., 2017), however degraded urban soils may require up to 15 kg m^{-2} (Beniston, Lal, & Mercer, 2016).

Domestic wastewater is the other main waste stream with the potential for nutrient recovery. Domestic wastewater is categorised according to the level of contaminates; black water is wastewater contaminated with faecal matter, whereas greywater is that which is uncontaminated (i.e., all streams excluding the wastewater from toilets). Centralised treatment of domestic wastewater produces emissions from transport, electricity consumption and production of chemicals used in the treatment process. Potent GHGs, like methane and nitrous oxide, are also produced biologically during wastewater and sludge treatment (Daelman et al., 2012; Parravicini, Svardal, & Krampe, 2016).

New Sanitation is a new paradigm proposed to better manage household wastewater and recover resources that can be used in urban agriculture. New Sanitation relies on source separation technologies such as urine deviated vacuum toilets and allows for different types of waste (e.g., greywater and blackwater) to be treated separately in decentralised sanitation systems. Grey water has a high potential for reuse as it is relatively low in pollution, while black water contains large amounts of nitrogen and

phosphorus (de Graaff, Temmink, Zeeman, & Buisman, 2010). Therefore, source separation allows technologies such as *Upflow Anaerobic Sludge Blanket* (UASB) reactors (Kujawa-Roeleveld & Zeeman, 2006) to efficiently recover nutrients and heat energy from blackwater (de Graaff et al., 2010; R. C. Wielemaker, Weijma, & Zeeman, 2018).

R. C. Wielemaker et al. (2018) investigated how the New Sanitation paradigm can be used to satisfy the input requirements of urban agriculture, in a case study in of Rotterdam (Netherlands). Household waste (blackwater and kitchen waste) was shown to provide enough P, N and OM to fertilize all of the arable land within the city (including suitable rooftops) (R. C. Wielemaker et al., 2018). Saini, Lohchab, Anil, and Kulbir (2018) created a lab prototype UASB reactor measuring 66 cm by 15 cm with working volume of 10.2 L, large enough to serve a household of four persons. Not only could recycled fertilisers displace the need for synthetic fertilisers, but urine fertilisers have also been shown to replenish micronutrients in the soil and cause less soil acidification than synthetic fertilisers (Chrispim, Tarpeh, Salinas, & Nolasco, 2017). Despite the demonstrated advantages, widespread use of human waste as fertilizer may be impaired by social norms (Andersson, 2015).

Despite the potential benefits of nutrient recycling, meeting the nutrient requirements of urban food gardens is a significant challenge (Taylor & Lovell, 2014; R. Wielemaker et al., 2019; R. C. Wielemaker et al., 2018). Improper nutrient management, in particular the over-application of nutrients, can lead to leaching, acidification, and contamination of ground water (Taylor & Lovell, 2014; Tian & Niu, 2015; Wielemaker et al., 2019). Numerous studies in various locations have found a surplus of soil nutrients in urban

food gardens using organic fertilizers, often exceeding soil fertility recommendations (Dewaelheyns, et al., 2013; Metson & Bennett, 2015; Salomon et al., 2020; Witzling, Wander, & Phillips, 2010). This is likely to be due to over-fertilisation, a lack of practitioner knowledge, and the chemical composition of the compost used. For example, when compost is applied at a rate to meet the Nitrogen requirement of crops, the amounts of Phosphorous and Potassium applied often exceed crop requirements (R. Wielemaker et al., 2019). This can be counteracted by the addition of substances with high Nitrogen:Phosphorous ratios such as coffee grounds or wood chips (Salomon et al., 2020), or integration of N-fixing crops such as legumes (Peoples et al., 2009). However, such practices require gardeners to have a working knowledge of complex nutrient management principles.

3.3 Climate Regulation

ex situ: Carbon sequestration and reduced carbon emissions

Urban agriculture is thought to provide an alternative to traditional food systems with a lower green-house gas (GHG) footprint. This concept has inadvertently been introduced through the previous discussion on waste management, as the processing of urban waste has an associated global warming potential. However, to properly understand the impact urban agriculture can have on global warming, one must assess the environmental impact of inputs and raw materials, as well as waste management. The formal assessment of a product from cradle to grave is called Life Cycle Assessment (LCA) (Finnveden et al., 2009; Muralikrishna & Manickam, 2017), and can be used to compare urban agriculture to traditional farming. Comprehensive LCAs have proven to

be impractical due to data needs, time, expense, and uncertainty in the final results (Muralikrishna & Manickam, 2017). Consequently, a method of Streamlined LCA has been developed based on the relative significance of environmental impacts – i.e., climate change (GHG emissions), which is of crucial concern. This retains the fundamental concept of LCAs, while allowing actual implementation to be more straight forward (Muralikrishna & Manickam, 2017). In addition, LCAs succinctly summarise a range of services/disservices along the production line. This is particularly useful given that many regulating services can be quantified according to the resultant GHG emissions.

One LCA of residential food gardens in Santa Barbara County (USA), found that urban agriculture could reduce emissions by 2.10 kg of CO_2e per kg of vegetables each year when compared to commercial farms (Cleveland et al., 2017). The model assumed food gardens would replace residential lawn, and that water and organic matter would come from household greywater and green waste, diverting it from treatment plants. Removing purchased conventional vegetables contributed to just 37% of the total GHG reduction (associated with production and transport, and consumer pickup from the grocery store), with the largest contribution in the model coming from the diversion of organic household waste (47%). Other contributions were replacing lawn (13%), and diversion of greywater from treatment plants (3%) (Cleveland et al., 2017).

Cleveland et al. (2017) estimate of CO_2e reduction has been used by McDougall et al. (2020) to model potential net GHG reductions in Sydney (Australia). The authors state that "*Cleveland et al. (2017) is the only study we are aware of which quantifies the GHG*

emissions involved in urban vegetable gardening" (McDougall et al., 2020). However, in Santa Barbara County there is no separate food waste collection. Therefore, all the household green waste is usually sent to landfill (Cleveland et al., 2017), whereas in Sydney, food scraps are collected separately. This has a great impact on the baseline GHG emissions and is a major caveat in the extrapolation of Cleveland et al. (2017) data. Additional LCA of food gardens in a range of cities would facilitate more accurate modelling in Australia and Europe, where green waste is collected and composted separately from other household waste. Furthermore, Cleveland et al. (2017) did not investigate the potential carbon sequestration in reforesting agricultural land as traditional food systems become displaced.

Fisher (2014) compared LCAs of small-scale urban vegetable farming and commercial farming in Denver (USA). The net reduction in GHG emissions accredited to urban agriculture ranged between approximately 0.2 - 0.65 pounds of CO_2e per pound of vegetables, depending on the crop species (Fisher, 2014). These figures differ greatly from Cleveland et al. (2017) findings. Fisher (2014) assumes urban turf is converted to urban food gardens. and include displacement of commercial farmland to be left fallow. While compost was presumed to be used in lieu of other fertilizers in small scale urban farming, Fisher (2014) did not investigate the diversion of organic waste from treatment facilities.

In London, production of food in community gardens was found to reduce global warming potential by an average of 2.23 kg of CO_2e per kg of fresh produce, or 1.11 kg with two outliers eliminated (calculated by Cleveland et al. (2017) from Kulak, Graves,

and Chatterton (2013)). Unlike residential food gardens modelled by Cleveland et al. (2017) and Fisher (2014), the London community gardens produced emissions associated with refrigeration and distribution of produce (Kulak et al., 2013). In some instances, production methods included the use of polytunnels, deviating from the definition of urban agriculture set out by this review. Growing strawberries in polytunnels in London would produce 12% *more* GHG emissions than importing strawberries from Spain (Kulak et al., 2013). Soil-based, open-air vegetable production has lower energy inputs than climate regulated practices (Goldstein et al., 2016). However, this does raise the issue of climate appropriate crops, and consumer demand. Fisher (2014) looked at just four widely consumed, climate appropriate crops, while the Cleveland et al. (2017) assessment was not crop specific. Of the three LCAs reviewed, Kulak et al. (2013) was the only one to consider crops currently being grown through urban agriculture.

Taken together, urban agriculture can lower the GHG emissions of vegetables production through several avenues. It is true that producing food close to where it is consumed will reduce GHG associated with travel (Lee, Lee, & Lee, 2015), however LCAs reveal that food miles alone are not the best indicator for environmental sustainability, as outcomes are subject to crop and production design. Existing LCAs are limited in number, and each have their own limitations. Additional LCAs would extend the knowledge to a broader range of urban forms and systems, as well as provide more data from which trends could be established. LCAs are useful as they can reveal the trade-offs along various stages of production (e.g., climate regulation and food miles (Kulak et al., 2013)), and highlight practices with greatest potential for emissions reductions (e.g.,

recycling organic household waste (Cleveland et al., 2017)). The LCAs reviewed collated primary data (direct observations) with secondary data sources such as peer-reviewed literature, industry publications, government databases and previously published life-cycle inventories.

4. Conclusion and Project Aims

Under sensitive management practices, urban agriculture can deliver ecosystem services to urban environments. This is particularly true in the areas of food provisioning, waste management, water regulation, and climate regulation. However, residential food gardens are overlooked and understudied compared to other forms of urban agriculture. This research will model outcomes from residential food gardens in relation to self-sufficiency, decentralised circular metabolism, and climate regulation. Each chapter will build on the earlier work. The modular approach will result in a comprehensive analysis of ecosystem services of residential food gardens.

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Appendix B

Supplement to 'The case for Urban Agriculture: Opportunities for sustainable development'

Table 8 Representative examples of the contribution of urban agriculture to the SDGs through People ( ) and Planet ( ). See main text for more detailed exploration of SDGs 2, 11, and 12. For a full listing of each Target, see https://sdgs.un.org/.

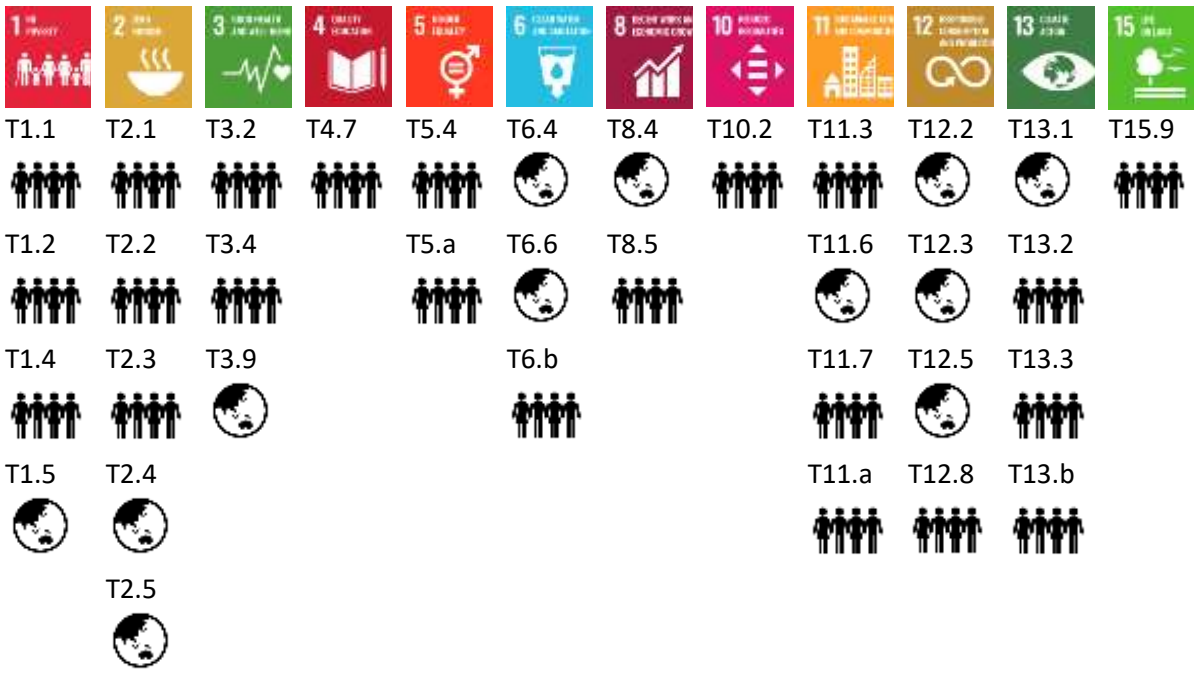

Table 9 Urban agriculture applied effectively has the potential to help achieve a multitude of Targets within SDG 2 (Zero Hunger).

Target 2.1 By 2030, end hunger and ensure access by all people, in particular the poor and people in vulnerable situations, including infants, to safe, nutritious and sufficient food all year round.

Target 2.2 By 2030, end all forms of malnutrition, including achieving, by 2025, the internationally agreed targets on stunting and wasting in children under 5 years of age, and address the nutritional needs of adolescent girls, pregnant and lactating women and older persons.

Target 2.3 By 2030, double the agricultural productivity and incomes of small-scale food producers, in particular women, indigenous peoples, family farmers, pastoralists and fishers, including through secure and equal access to land, other productive resources and inputs, knowledge, financial services, markets and opportunities for value addition and non-farm employment.

Target 2.4 By 2030, ensure sustainable food production systems and implement resilient agricultural practices that increase productivity and production, that help maintain ecosystems, that strengthen capacity for adaptation to climate change, extreme weather, drought, flooding and other disasters and that progressively improve land and soil quality.

Target 2.5 By 2020, maintain the genetic diversity of seeds, cultivated plants and farmed and domesticated animals and their related wild species, including through soundly managed and diversified seed and plant banks at the national, regional and international levels, and promote access to and fair and equitable sharing of benefits arising from the utilization of genetic resources and associated traditional knowledge, as internationally agreed.

Table 10 Urban agriculture applied effectively has the potential to help achieve a multitude of Targets within SDG 11 (Sustainable Cities and Communities).

Target 11.3 By 2030, enhance inclusive and sustainable urbanization and capacity for participatory, integrated and sustainable human settlement planning and management in all countries.

Target 11.7 By 2030, provide universal access to safe, inclusive and accessible, green and

Target 11.6 By 2030, reduce the adverse per capita environmental impact of cities, including by paying special attention to air quality and municipal and other waste management.

public spaces, in particular for women and children, older persons and persons with disabilities.

Target 11.a Support positive economic, social and environmental links between urban, peri-urban and rural areas by strengthening national and regional development planning.

Table 11 Urban agriculture applied effectively has the potential to help achieve a multitude of Targets within SDG 12 (Responsible Consumption and Production)

Target 12.8 By 2030 ensure that people everywhere have the relevant information and awareness for sustainable development and lifestyles in harmony with nature.

Target 12.2 By 2030, achieve the sustainable management and efficient use of natural resources.

Target 12.3 By 2030, halve per capita global food waste at the retail and consumer levels and reduce food losses along production and supply chains, including post-harvest losses.

Target 12.5 By 2030, substantially reduce waste generation through prevention, reduction, recycling and reuse.

Appendix C

THE CONVERSATION

Academic rigour, journalistic flair

Shutterstock

Fennel looking a bit feeble? Growing enough veggies to feed yourself depends on these 3 things

Published: June 9, 2022 1.46pm AEST

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University of Adelaide

Farming inside city boundaries is on the rise as countries become more urbanised and people seek to connect with the source of their food and improve their sustainability.

But despite the productivity potential of home food gardens and the like, they are rarely analysed as serious farming systems. There's little data, for example, on how much can be grown on an average suburban property.

As climate change threatens global food supplies, however, building sustainable urban food systems will be crucial.

Our research has examined how productive the average home vegetable garden really is, and how to get the most from your patch.

Home gardens are rarely analysed as serious farming systems. Shutterstock

Lawn with a side of salad?

Urban agriculture refers to growing produce and raising livestock inside a city's boundary. In Australian cities, it might involve a home vegetable patch, community garden, backyard beehives, an edible rooftop garden on an apartment block, indoor hydroponics, a communal orchard and more.

Sometimes, especially in developing countries, urban farming can help address issues such as poverty, unemployment and food insecurity.

More broadly, it can increase access to healthy, fresh produce and lead to more sustainable food production. It can also help us save money and improve our well-being.

Societies have traditionally lent on urban farming during times of stress. So it's no surprise the practice resurged during the COVID pandemic. In Australia, keeping edible gardens significantly helped people maintain mental health during lowdown, particularly those on low incomes.

But to what extent can we rely on our backyard gardens to meet all our fresh produce needs? Our research shows these three factors are key.

Read more: 3 ways community gardens often exclude migrants and refugees — and how to turn this around

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Gardening helped people get through COVID lockdowns, especially those on low-incomes. David Crosling/AAP

1. Give up some lawn

We looked at the potential for food production at about 40,000 residential properties in suburban Adelaide – mostly free-standing homes.

We calculated the amount of land required for a household of 2.5 people to grow the recommended five servings of vegetables per person each day. Then, using high-resolution aerial imagery to get a birds eye view of properties, we identified those with enough lawn area to make that happen.

Some 21m² of lawn is needed to produce the recommended vegetable intake. In a scenario where a garden is high-yielding, this would require converting 23% of lawn area on a typical block into a vegetable patch. Of the properties modelled, 93% had the room to a create 21m² garden from the total lawn space.

In a medium-yield garden, 72% of lawn on a typical block would need converting to produce enough vegetables to feed a household – equating to 67m².

We limited the research to in-ground veggie production and didn't include fruit trees. So a property's potential to grow food would be even higher if food gardens or fruit trees already exist, or other garden beds or paved areas could be converted.


house with front lawn and sold sign depicting blonde woman

Converting just 23% of lawn can provide enough room to grow your own vegetables. Dave Hunt/AAP

2. Up your gardening game

Research out of Adelaide, which surveyed about 30 home gardeners, found yields per square metre ranged from 0.24kg to 16.07kg per year. This suggests a high rate of variability in home garden productivity – notwithstanding the fact people grow different crops.

Not all of us have green thumbs and in some cases, your veggie patch might not yield as much as you hoped.

Perhaps you gave it too much or too little water. Maybe you didn't have time to pull out weeds or harvest produce. Pests and fungus might have struck down your crop. You may have planted the wrong seeds at the wrong time or just have poor soil.

Our research suggests low-yield gardens would need 1,407m² of converted lawn to meet the vegetable needs of a household. However, less than 0.5% of properties in the analysed Adelaide sites had so much land. So to reach self-sufficiency in urban agriculture environments, medium to high yields are preferred.

Skilled gardeners with high yields will need much less land. Given the space constraints in cities, upskilling gardeners is important to maximising production.


straggly plants in pots with bead bush

Your garden may not yield as much as you'd hoped. Shutterstock

3. Know what's in your soil

Good soil is a key factor in productive gardens. It needs a good structure (one that allows water and air to enter and drain easily, while retaining enough moisture) an ample supply of plant nutrients and a rich microbial community.

In city areas, heavy metal contamination and pollution of soils can be a concern. We examined soils at 12 urban agricultural sites in Adelaide, and found in all cases that metal concentrations did not exceed health guidelines for residential areas – even at sites with an industrial history.

But this might not always be the case. An analysis of residential and community gardens in Melbourne, for example, showed some soils were contaminated at levels which could pose a human health hazard. This highlights the importance of testing urban soils before planting.

Proper management of inputs – particularly fertiliser – is also key. Our research has found urban gardeners can choose from a variety of organic waste-based fertilisers such as spent coffee grounds, food scraps or lawn clippings. But this abundance can lead to imbalances.

In Adelaide, for example, the widespread use of freely available horse manure led to excessive phosphorous levels in almost all of the 12 tested sites. This imbalance can depress plant growth and damage the broader environment.

Read more: These 3 tips will help you create a thriving pollinator-friendly garden this winter

---


garden bed with rake and manure on top

Using too much manure on a garden can lead to excessive phosphorus levels. Shutterstock

Helping city gardens flourish

Urban agriculture has been identified as a A$4 billion economic growth opportunity for Australia. However, suburban blocks are trending towards smaller yards with less growing space.

Given the many benefits of urban farming, it's time to think more seriously about maximising efficiency and scale.

Community gardens are well placed for knowledge-sharing. Research on 13 community gardens in Sydney revealed they were very high-yield – around twice as productive than the typical Australian commercial vegetable farm.

Funding for more community gardens, and other education opportunities for urban gardeners, would be a valuable investment in improving public health and sustainability.

This should be coupled with policy and planning decisions designed to increase the amount of urban farming space in our cities.

Appendix D

Urban food production: backyard hobby or valuable farming system?

Isobel Hume[1], David Summers[2], Timothy Cavagnaro[1]
[1] University of Adelaide
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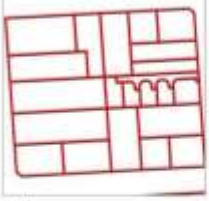
THE UNIVERSITY of ADELAIDE

Researcher Profile

Spatial data layers

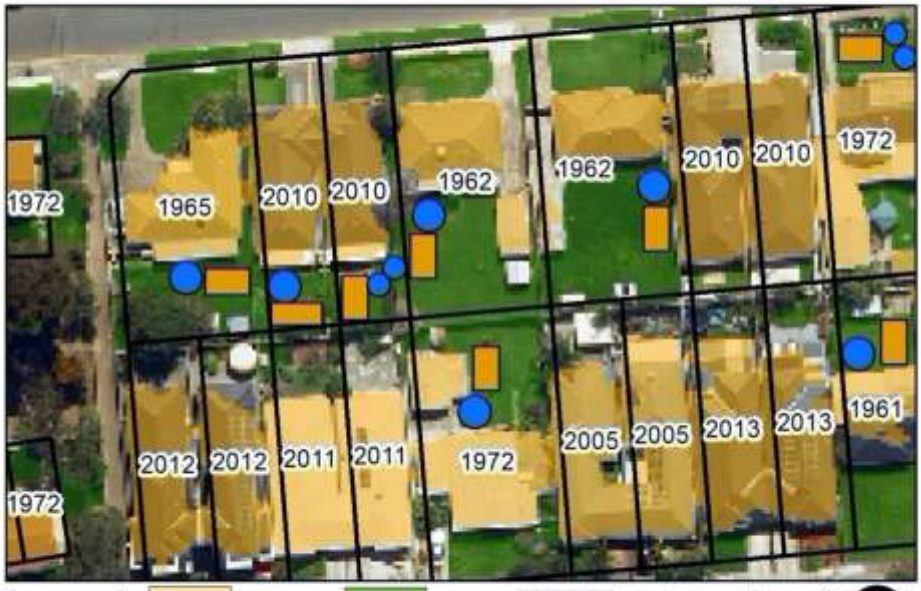
Land-use

Land ownership



crop data (yields + irrigation) **& climate variables**

Rainwater harvesting and self-sufficiency through urban agriculture



Highlights

- Almost half a million homes were modelled across one focal city (Adelaide, Australia).
- **Almost two thirds of households could produce the recommended vegetable intake, and harvest enough rainwater for irrigation,** even in a Dry year.
- This would require 40 kL storage, and a 21.3 m² garden (pictured).
- Self-sufficiency potential of homes is negatively correlated with the year the house was built.
- Replacing urban lawn with vegetable production reduces soil greenhouse gas emissions.

Background
Urban agriculture has garnered much attention for increasing resilience to a range of interrelated stressors. However, it is unclear if urban agriculture is just a nice idea, or a viable way of producing meaningful amounts of food for urban dwellers.

Conclusions
Backyard food gardens can produce meaningful amounts of food, address issues of increasing water stress, and lower the global warming potential of urban greenspace.

The scale at which urban agriculture can occur highlights the significant contribution these benefits could have toward sustainable food-systems.

Acknowledgements
Thankyou to my collaborators and supervisors, Timothy Cavagnaro and David Summers, everyone in the CaCuli for their ongoing input and support, and The University of Adelaide. This research was funded by an RTP scholarship.

Appendix E

Supplementary material for 'Lawn with a side salad: Rainwater harvesting for self-sufficiency through urban agriculture'

S1 Data were collected from 34 South Australian home gardens for varying lengths of time between September 2016 to June 2018. Amongst other things, participants recorded the size (m^2) of the growing area, crop type, yields (kg), and irrigation applied (L). The project's raw data has been made publicly available for reuse (Csortan, Ward, & Roetman, 2020).

Hume, Summers, and Cavagnaro (2021) analysed raw data from the Edible Gardens Project to calculate land use requirements to achieve dietary self-sufficiency of vegetables. Yields were found be to highly variable; however, the authors found that an achievable, best-case scenario for most growers would be 16.07 kg m^{-2} yr^{-1}. Importantly, this value is similar to other yields reported for soil-based urban agriculture sites in Australia (McDougall et al., 2019). With these yields, Hume et al. (2021) calculated that a typical home in Adelaide, which is comprised of 2.5 people, would require 21.30 m^{-2} to grow the daily RVI of residents. The model reflects a polyculture food garden with a range of vegetable crops that are currently produced by urban growers in Adelaide (the raw dataset contained 44 different vegetables and fruit vegetables as well as several herbs (Supplement Table S1)). The names of crops grown were recorded by participants of the survey and can be accessed in the publicly available raw data (Csortan et al., 2020).

Table S12 List of vegetables harvested by gardeners in the edible gardens project (as entered by participants) (Csortan et al., 2020)

1	Red cabbage	11	Fennel	21	broad beans	31	sweetcorn	41	Broccoli
2	Capsicum	12	Garlic	22	peas	32	beans	42	Boc Choy
3	Carrots	13	mixed greens	23	potatoes	33	Spinach	43	Brussel Sprouts
4	Cauliflower	14	Kale	24	pumpkin	34	corn	44	Cabbage
5	Celery	15	leeks	25	Radish	35	sweet potato		
6	Chillies	16	lettuce	26	Rainbow Chard	36	Tomatoes		
7	Jalapeno	17	Olives	27	Rocket	37	Turnips		
8	Cucumber	18	onions	28	Silverbeet	38	zucchini		
9	Eggplant	19	spring onions	29	snow peas	39	mixed (misc)		
10	Asparagus	20	Parsnip	30	Sprouts	40	Beetroot		

S2 Conditional statement use to classify ground cover according to processed spectral and LiDAR data, written with ArcPy (ESRI, 2011):

```
OutRas = Con((spec == 2) & (LiDAR < 4), 1, Con((LiDAR == 6) & (spec == 2), 2, Con((LiDAR == 6), 8, spec)))
```

$S3$ Deficit between water captured in month *m* and irrigation required in month *m* (Figure S3). Only months where water capture < irrigation required for at least one scenario are displayed on graph.

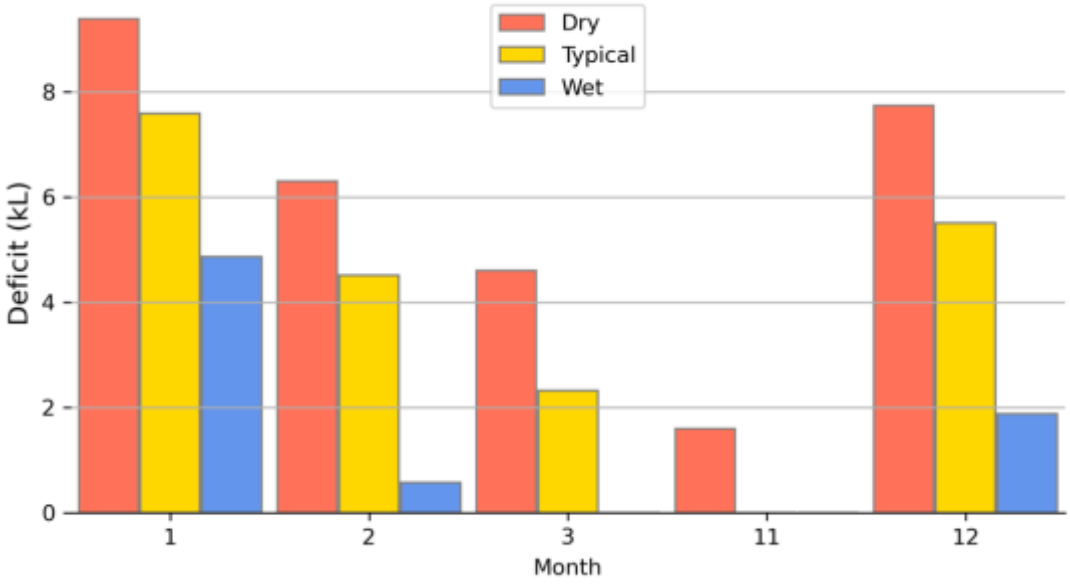


Figure S3 Deficit between water captured and irrigation required for self-sufficiency in a particular month. The deficit (Y axis) is the volume of water in kilolitres (kL). The months of January (1), February (2), March (3), November (11) and December (12) are displayed on the X axis. Three different rainfall scenarios are displayed: Dry, Typical, and Wet.

S4

Table S4: Self-sufficiency by residential property type. "Of total yes" is the break-down of property types within the cohort of properties that can achieve self-sufficiency under a given scenario. "Of type" is the proportion of that property type that is able to achieve self-sufficiency under a given scenario.

Type	count	proprtion of total	Able to achieve Self-sufficiency?																	
			25 kL						30 kL						40 kL					
			Dry		Typical		Wet		Dry		Typical		Wet		Dry		Typical		Wet	
			of total yes	of type	of total yes	of type	of total yes	of type	of total yes	of type	of total yes	of type	of total yes	of type	of total yes	of type	of total yes	of type	of total yes	of type
House	360427	73.30%	0.92	0.00	0.97	0.45	0.92	0.80	0.96	0.06	0.95	0.69	0.99	0.90	0.95	0.71	0.90	0.79	0.89	0.79
Ground Floor Home Unit Only	34875	7.09%	0.00	0.00	0.00	0.01	0.02	0.14	0.00	0.00	0.01	0.05	0.02	0.17	0.01	0.06	0.02	0.16	0.02	0.16
Maisonette	28851	5.87%	0.00	0.00	0.01	0.07	0.04	0.44	0.00	0.00	0.02	0.21	0.05	0.58	0.02	0.23	0.05	0.59	0.05	0.60
Townhouse - Defined As Home Unit With Both Ground And First Floor Areas	16792	3.42%	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.01	0.00	0.03	0.00	0.01	0.00	0.04	0.00	0.05
Independent Living Unit	8473	1.72%	0.00	0.00	0.00	0.01	0.00	0.10	0.00	0.00	0.00	0.06	0.00	0.10	0.00	0.07	0.00	0.09	0.00	0.10
Single Storey Flats - Purpose Built	7897	1.61%	0.02	0.00	0.00	0.10	0.01	0.21	0.01	0.03	0.00	0.13	0.01	0.30	0.00	0.13	0.01	0.32	0.01	0.35
Detached Single Storey Home Unit	7041	1.43%	0.00	0.00	0.00	0.03	0.01	0.25	0.00	0.00	0.00	0.11	0.01	0.26	0.00	0.12	0.01	0.24	0.01	0.24
Ground Floor Home Unit In A Multi-Storey Block	5573	1.13%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01	0.00	0.01
Row House	4764	0.97%	0.00	0.00	0.00	0.01	0.00	0.15	0.00	0.00	0.00	0.04	0.00	0.19	0.00	0.07	0.00	0.19	0.00	0.20
First Floor Home Unit	3794	0.77%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Unfinished House	3346	0.68%	0.00	0.00	0.00	0.05	0.00	0.12	0.00	0.01	0.00	0.08	0.00	0.15	0.00	0.08	0.00	0.16	0.00	0.20
House & Granny Flat	1173	0.24%	0.01	0.01	0.01	0.72	0.00	0.87	0.01	0.16	0.00	0.83	0.00	0.97	0.00	0.83	0.00	0.96	0.00	0.96
Second Floor Home Unit	1035	0.21%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Retired And Aged Accommodation	954	0.19%	0.01	0.01	0.00	0.04	0.00	0.05	0.00	0.03	0.00	0.05	0.00	0.06	0.00	0.05	0.00	0.07	0.00	0.08
Town House Style Flats	921	0.19%	0.00	0.00	0.00	0.01	0.00	0.03	0.00	0.00	0.00	0.01	0.00	0.06	0.00	0.02	0.00	0.09	0.00	0.13
Two Storey And Higher Flats - Purpose Built	920	0.19%	0.01	0.01	0.00	0.09	0.00	0.15	0.00	0.02	0.00	0.13	0.00	0.16	0.00	0.13	0.00	0.17	0.00	0.18
Flats	608	0.12%	0.00	0.01	0.00	0.34	0.00	0.49	0.00	0.08	0.00	0.41	0.00	0.50	0.00	0.42	0.00	0.50	0.00	0.50
Townhouse - Home Unit Over Two Levels In Which The Lower Level Is Above Ground Level	582	0.12%	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01	0.00	0.02
House With Unestablished Grounds/Gardens	512	0.10%	0.00	0.00	0.00	0.31	0.00	0.49	0.00	0.07	0.00	0.44	0.00	0.49	0.00	0.42	0.00	0.47	0.00	0.49
Third Floor Home Unit	471	0.10%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Shop And Dwelling	429	0.09%	0.01	0.01	0.00	0.25	0.00	0.32	0.00	0.07	0.00	0.30	0.00	0.31	0.00	0.29	0.00	0.30	0.00	0.30
Ninth Floor Or Above Home Unit	252	0.05%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Home Industry Where Owner Resides In Property	177	0.04%	0.00	0.03	0.00	0.46	0.00	0.61	0.00	0.12	0.00	0.53	0.00	0.59	0.00	0.53	0.00	0.56	0.00	0.56
Fourth Floor Home Unit	165	0.03%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Single Storey Flats - Pair Of Maisonettes	162	0.03%	0.00	0.00	0.00	0.17	0.00	0.33	0.00	0.03	0.00	0.23	0.00	0.41	0.00	0.23	0.00	0.45	0.00	0.49
Single Storey Flats - Built For Strata Titling	162	0.03%	0.00	0.01	0.00	0.15	0.00	0.36	0.00	0.03	0.00	0.22	0.00	0.45	0.00	0.23	0.00	0.45	0.00	0.48
Sixth Floor Home Unit	145	0.03%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fifth Floor Home Unit	128	0.03%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
House & Flat	123	0.03%	0.00	0.02	0.00	0.66	0.00	0.85	0.00	0.15	0.00	0.80	0.00	0.82	0.00	0.79	0.00	0.81	0.00	0.81
House & Office	108	0.02%	0.00	0.04	0.00	0.34	0.00	0.56	0.00	0.13	0.00	0.52	0.00	0.55	0.00	0.52	0.00	0.54	0.00	0.55
Single Storey Flats - House Converted To Flats	99	0.02%	0.00	0.00	0.00	0.57	0.00	0.74	0.00	0.17	0.00	0.67	0.00	0.74	0.00	0.70	0.00	0.74	0.00	0.74
Seventh Floor Home Unit	98	0.02%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
House With Manufacturing & Service Industry	94	0.02%	0.00	0.01	0.00	0.43	0.00	0.56	0.00	0.15	0.00	0.52	0.00	0.56	0.00	0.49	0.00	0.53	0.00	0.53
Home Unit	88	0.02%	0.00	0.00	0.00	0.00	0.00	0.17	0.00	0.00	0.00	0.03	0.00	0.18	0.00	0.05	0.00	0.19	0.00	0.19
Eighth Floor Home Unit	76	0.02%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Independent Living Unit B	47	0.01%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
House & Surgery	40	0.01%	0.00	0.00	0.00	0.53	0.00	0.63	0.00	0.25	0.00	0.58	0.00	0.65	0.00	0.58	0.00	0.65	0.00	0.65
Private Hotels And Boarding Houses And Boarding Houses	39	0.01%	0.00	0.00	0.00	0.26	0.00	0.33	0.00	0.08	0.00	0.26	0.00	0.33	0.00	0.26	0.00	0.33	0.00	0.33
Institutional Residential Accommodation N.E.C.	38	0.01%	0.00	0.03	0.00	0.32	0.00	0.39	0.00	0.21	0.00	0.37	0.00	0.39	0.00	0.29	0.00	0.29	0.00	0.29
Rural Residential House (House Without Primary Production)	36	0.01%	0.00	0.00	0.00	0.72	0.00	1.00	0.00	0.33	0.00	0.92	0.00	1.00	0.00	0.92	0.00	1.00	0.00	1.00
Town House Style Flats - Built For Strata Titling	36	0.01%	0.00	0.00	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.03	0.00	0.14	0.00	0.03	0.00	0.36	0.00	0.39
Town House Style Flats - Pair Of Two Storey Maisonettes	32	0.01%	0.00	0.00	0.00	0.03	0.00	0.13	0.00	0.03	0.00	0.06	0.00	0.13	0.00	0.03	0.00	0.06	0.00	0.13
House With Single Bed & Breakfast	31	0.01%	0.00	0.03	0.00	0.29	0.00	0.58	0.00	0.13	0.00	0.48	0.00	0.55	0.00	0.52	0.00	0.55	0.00	0.55
Multiple Unit	23	0.00%	0.00	0.00	0.00	0.17	0.00	0.43	0.00	0.00	0.00	0.35	0.00	0.48	0.00	0.35	0.00	0.43	0.00	0.43
Two Storey And Higher Flats - Two Storey House Converted To Flats	21	0.00%	0.00	0.00	0.00	0.33	0.00	0.43	0.00	0.00	0.00	0.43	0.00	0.43	0.00	0.43	0.00	0.43	0.00	0.43
Religious Quarters - Monasteries Etc.	13	0.00%	0.00	0.00	0.00	0.54	0.00	0.62	0.00	0.31	0.00	0.62	0.00	0.62	0.00	0.62	0.00	0.62	0.00	0.62
Two Storey And Higher Flats - Built For Strata Titling	11	0.00%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Other Residential Hall Or Dormitory	9	0.00%	0.00	0.00	0.00	0.11	0.00	0.22	0.00	0.00	0.00	0.11	0.00	0.22	0.00	0.11	0.00	0.22	0.00	0.33
Supported Residential Care	7	0.00%	0.00	0.14	0.00	0.43	0.00	0.43	0.00	0.29	0.00	0.29	0.00	0.29	0.00	0.29	0.00	0.29	0.00	0.29
Basement Home Unit	6	0.00%	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Short Term Accommodation - Single Unit	4	0.00%	0.00	0.00	0.00	0.00	0.00	0.75	0.00	0.00	0.00	0.25	0.00	0.75	0.00	0.25	0.00	0.75	0.00	0.75
Institutional Residential	2	0.00%	0.00	0.00	0.00	1.00	0.00	1.00	0.00	0.00	0.00	1.00	0.00	1.00	0.00	1.00	0.00	1.00	0.00	1.00
House & Holiday Cabin	2	0.00%	0.00	0.00	0.00	0.50	0.00	0.50	0.00	0.00	0.00	0.50	0.00	1.00	0.00	0.50	0.00	1.00	0.00	1.00

S5

Table S5 Comparison of three different water storage scenarios: 25 kL, 30kL, and 40 kL, assuming lawn area in a Typical property has shrunk by 20m².

tank size	*footprint tank (m)*	*footprint for SS (tank + garden)*	*% Lawn area used⁺*	*% pop. able to achieve SS^*			*reference*
				Dry	Typical	Wet	
25 kL	11.22	32.52	64%	10%	57%	52%	Oz Tanks (2021)
30 kL	12.07	33.37	66%	41%	59%	59%	ASC Tanks (2021)
40 kL	15.21	36.51	72%	58%	58%	58%	(Southern Tanks, 2021)

** all tanks are 2.8m in height*

⁺'% Lawn area used' refers to the % of lawn area at a typical property (51m2) that the modelled edible garden and associated storage tank would occupy.

^'% pop. able to achieve SS' refers to the percentage of the population able to achieve dietary self-sufficiency of vegetables for 2.5 people (a function of a properties lawn area), as well as capture and store enough rainwater for irrigation of the modelled edible garden (a function of roof area, rainfall, and lawn area)

S6

All water used

Excel version					
discount rate		0.035			
Annual water savings		90			
purchase price		5500			
water value		1.966	3.04		
Annual savings		176.94	273.6		
				Different water costs	
Year		Annual savings		196.6	304
	0	-5500	-5500		
	1	176.94	273.6	-$5,329.04	-$5,235.65
	2	176.94	273.6	-$5,163.87	-$4,980.24
	3	176.94	273.6	-$5,004.28	-$4,733.47
	4	176.94	273.6	-$4,850.09	-$4,495.05
	5	176.94	273.6	-$4,701.11	-$4,264.68
	6	176.94	273.6	-$4,557.17	-$4,042.11
	7	176.94	273.6	-$4,418.09	-$3,827.06
	8	176.94	273.6	-$4,283.72	-$3,619.29
	9	176.94	273.6	-$4,153.90	-$3,418.54
	10	176.94	273.6	-$4,028.46	-$3,224.58
	11	176.94	273.6	-$3,907.27	-$3,037.18
	12	176.94	273.6	-$3,790.17	-$2,856.11
	13	176.94	273.6	-$3,677.03	-$2,681.17

14	176.94	273.6	-$3,567.72	-$2,512.15
15	176.94	273.6	-$3,462.11	-$2,348.84
16	176.94	273.6	-$3,360.07	-$2,191.05
17	176.94	273.6	-$3,261.48	-$2,038.60
18	176.94	273.6	-$3,166.22	-$1,891.30
19	176.94	273.6	-$3,074.18	-$1,748.99
20	176.94	273.6	-$2,985.26	-$1,611.49
21	176.94	273.6	-$2,899.34	-$1,478.63
22	176.94	273.6	-$2,816.33	-$1,350.27
23	176.94	273.6	-$2,736.12	-$1,226.26
24	176.94	273.6	-$2,658.63	-$1,106.43
25	176.94	273.6	-$2,583.76	-$990.66
26	176.94	273.6	-$2,511.42	-$878.80
27	176.94	273.6	-$2,441.53	-$770.72
28	176.94	273.6	-$2,374.00	-$666.30
29	176.94	273.6	-$2,308.75	-$565.41
30	176.94	273.6	-$2,245.71	-$467.94
31	176.94	273.6	-$2,184.80	-$373.75
32	176.94	273.6	-$2,125.95	-$282.76
33	176.94	273.6	-$2,069.10	-$194.84
34	176.94	273.6	-$2,014.16	-$109.89
35	176.94	273.6	-$1,961.08	-$27.82
36	176.94	273.6	-$1,909.80	$51.48
37	176.94	273.6	-$1,860.25	$128.10
38	176.94	273.6	-$1,812.38	$202.12
39	176.94	273.6	-$1,766.12	$273.64
40	176.94	273.6	-$1,721.43	$342.75
41	176.94	273.6	-$1,678.25	$409.51
42	176.94	273.6	-$1,636.54	$474.02
43	176.94	273.6	-$1,596.23	$536.35
44	176.94	273.6	-$1,557.28	$596.57
45	176.94	273.6	-$1,519.66	$654.76
46	176.94	273.6	-$1,483.30	$710.97
47	176.94	273.6	-$1,448.17	$765.29
48	176.94	273.6	-$1,414.24	$817.76
49	176.94	273.6	-$1,381.44	$868.47
50	176.94	273.6	-$1,349.76	$917.46

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Appendix F

Supplementary Material for 'Backyard farming: Who benefits and who misses out? A Geospatial Analysis'

The following images are providing more context to the results in the paper main text. Figure S1 shows the entire study site, showing the built-up urban area with the Mount Lofty Rangers on to the east and ocean to the west. Agricultural regions can be seen to the North and South of the urban area.

Figure S1 Satellite (Google Earth) image of the Greater Adelaide Region.

Figure S2 Highlight specific areas which are discussed in the paper – illustrating the clear difference in plantable public land between Areas.

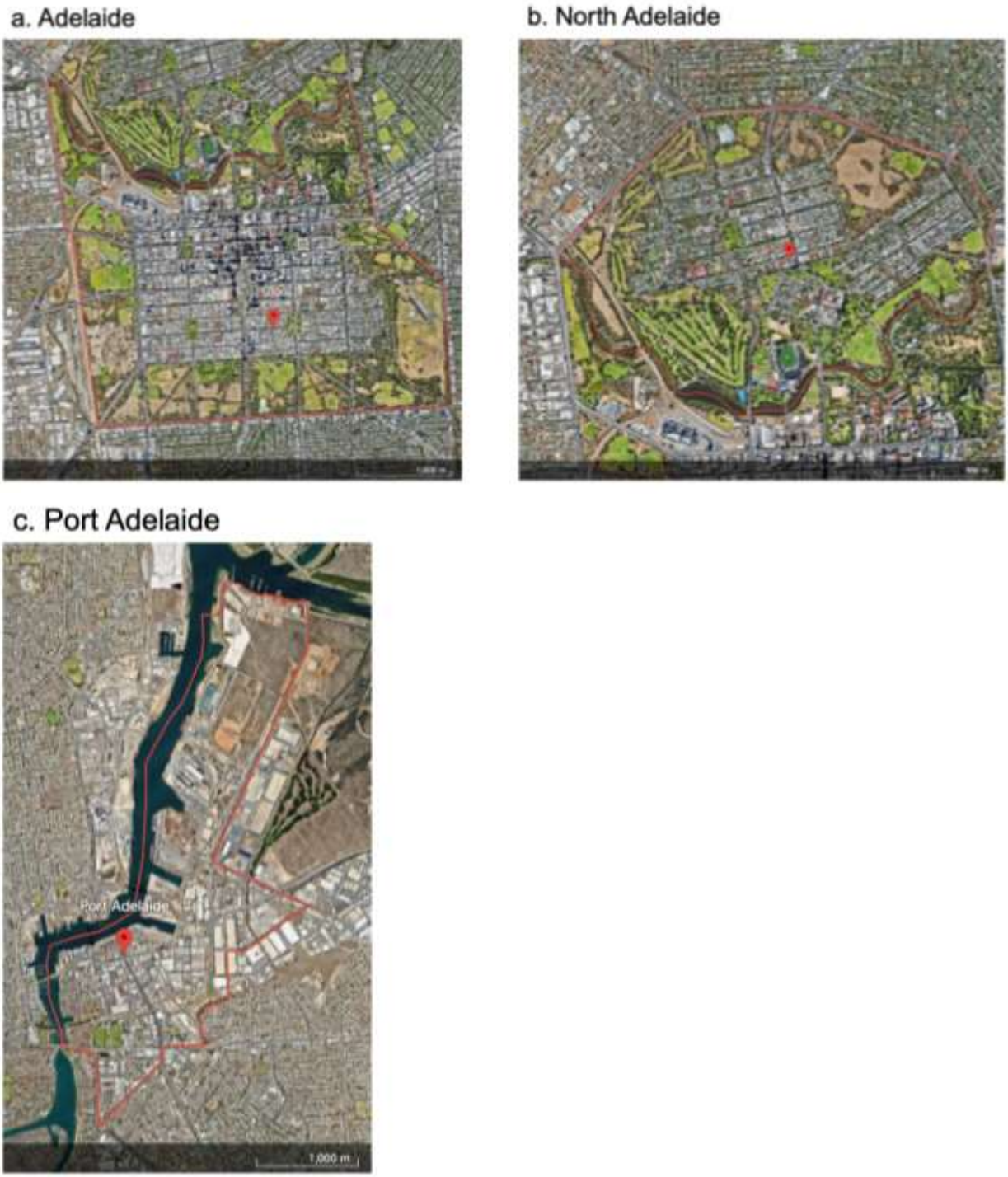


Figure S2 Satellite (Google Earth) image of three suburbs within Adelaide –a. Adelaide (CBD), b. North Adelaide, and c. Port Adelaide.

Appendix G

Supplementary material for 'Is the grass always greener? Comparing the global warming potential of backyard farming and lawn'

S1 DNDC input values for each scenario (Lawn HM, Lawn LM, Vegetable (Veggie) Garden).

variable	measurement	justification/source	variable	measurement	justification/source
	ALL				
time	100 years				
lat	-34.9				
N in rainfall	0.1				
soil	LAWN (ALL)		**soil**	VEGGIE GARDEN	
texture	sandy loam	- all other soil properties not listed remained at default.	texture	Organic Soil	- all other soil properties not listed remained at default.
bulk density	1.55	based on standard for Sandy loam, supported by (Qian and Follett 2002) who found soil bulk density to be 1.6 g cm−3 for putting greens and 1.5 g cm−3 for fairways on golf course.	bulk density	1.55	Default
ph	6.5	"Ideal" assuming that it will be managed by gardener to this level (sandy, east to manipulate)	ph	7.1	Salomon, Watts-Williams, McLaughlin, and Cavagnaro (2022)
field capacity	0.32	Default	field capacity	0.55	Default
wilting point	0.15	Default	wilting point	0.25	Default
clay fraction	0.09	Default	clay fraction	0.06	Default
conductivity	0.1248	Default	conductivity	0.015	Default
porosity	0.435	Default	porosity	0.701	Default
SOC at surface (kg/kg soil)	0.90%	Qian & Follett 2012 found OM to be appox. 1.5% in turfgrass planted with sand-based soil. This figure is multiplied by 0.58 as SOC comprises 58% of SOM mas.	SOC at surface (kg/kg soil)	7.9 %	From Salomon et al. (2022), assuming that all soil carbon is SOC.

nitrate mg/kg	0.5	Default	nitrate mg/kg	9.1	From Salomon et al. (2022)
ammonium mg/kg	0.05	Default	ammonium mg/kg	6.2	from Salomon et al. (2022)
crop	LAWN (ALL)		**crop**	VEGGIE GARDEN	
type	perennial grass	leaf stem C:N ratio changes to 20 and root changed to 40. other parameters default	type	various	see section explaining the polyculture set up.
planted	1-Mar	year 1 only - never harvested	planted	various	see section explaining the polyculture set up.
irrigation	LAWN HIGH MAINTENANCE		**irrigation**	VEGGIE GARDEN	
no. of events	44	Irrigation schedule calculated from "Basic Irrigation Management Toolkit" outlined in SA Waters' Code of Practice for Irrigated Public Open Space (SA Water, 2015). The details can be found in the "Watering Table". There are 44 watering events every year but the first year (n1). In n1, lawn is watered every day for three weeks after sowing, then follows regular watering as per the watering table. Therefore, n1 has 46 events (none for Jan/Feb, heavy watering in March).	no. of events	57	Taken from Edible Adelaide Gardener Survey (Csortan, Ward, & Roetman, 2020)
event dates	see watering table		events	see watering table	
type	sprinkler		type	sprinkler	
depth	see watering table		depth	see watering table	
irrigation	LAWN LOW MAINTENANCE		Compost		
how many times	16	Gu et al. (2015) conducted face to face interviews with household specifically to find out about their lawn management practice, they found that around half of the people did not irrigate at all. Given that Aus. is dryer that this site, we are assuming exactly half of the high maintenance TQVS4 (Passive Recreational Turf) every year but n1, n1 has 32	We assume the only fertilisation comes from organic compost. While some home gardeners do use syntenic fertiliser (Taylor & Lovell, 2015), in Baltimore, (USA), Santo et al. (2021) found 80% of city farms and gardeners used no chemical fertilisers, 3% used them everywhere, while 17% them some places (n=95). Additionally, Salomon, Watts-Williams, McLaughlin, and Cavagnaro (2020) found that in Adelaide, *"All*		

		(none for Jan/Feb, lots in Mar).		*sites were committed to principles of organic farming".*	
events	See watering tables	every second event from TQVS4	applications / year	3	Two main events between each planting season. One smaller event added during planting season (Salomon, pers. comm., 2021).
type	Sprinkler		dates	Feb 29, Aug 31 (main) Nov 30 (smaller)	(Salomon, pers. comm., 2021)
depth	see watering tables	every second event from TQVS4			
			solid C/N ratio	11	From Salomon et al. (2022)
fertilizer	LAWN LOW MAINTENANCE			All estimates based on a bulk density of 582 kg m^{-3}, a value within the rage of literature findings (Khater, 2015)	
applications year 1	2	"Based on the instructions from "Aussie lawns" - https://www.aussielawns.com.au/page/8/feeding-your-lawn-how-to-fertilise	Organic C (kg C/ha/event)	11058	From Salomon et al. (2022) – half for smaller application event. 27645 kg total
applications year n	0	Gu et al. (2015) conducted face to face interviews with household specifically to find out about their lawn management practice, they found that around half of the people did not fertilise at all.	Organic N (kg N/ha)	1047.6	From Salomon et al. (2022) – half for smaller application event. 2619 kg total
dates	Mar 1, Mar22		NH4+ (kg N/ha)	0.00163	From Salomon et al. (2022) – half for smaller application event.
Applied N (kg/ha)	14.4	'Kickstart' fertilizer	NO3- (kg N/ha)	0.0277	From Salomon et al. (2022) – half for smaller application event.
Applied P (kg/ha)	17.7	'Kickstart' fertilizer			
			method	Surface spread	(Salomon, pers. comm., 2021)
fertilizer	LAWN HIGH MAINTENANCE		depth cm	2cm (main) 1cm (secondary)	(Salomon, pers. comm., 2021)
applications year 1	4	Based on the instructions from "Aussie lawns" - (Great Aussie Lawns, 2021a)			
applications year n	4				

dates year 1	Mar 1, Mar 22, Apr 12, Sept 1	First two are "Kickstart", second two are "Reno"			
dates year n	Jan 1, Mar 1, Apr 12, Sept 1				
Applied N (kg/ha)	14.4	For 'Kickstart' fertilizer applications (Great Aussie Lawns, 2021c)			
Applied P (kg/ha)	17.7	For 'Kickstart' fertilizer applications (Great Aussie Lawns, 2021c)			
Applied N (kg/ha) -	66	For "Reno" fertilizer applications (Great Aussie Lawns, 2021b)			
Applied P (kg/ha)	15	For "Reno" fertilizer applications (Great Aussie Lawns, 2021b)			
mowing	**LAWN HIGH MAINTENANCE**				
events	26	Once every 2 weeks, but not for the first three months after sowing. Therefore, there are 16 events in the first year (plant in March).			
cut part	leaf, grain, stem				
cut fraction	0.3				
mowing	**LAWN LOW MAINTENANCE**				
events	13	Once every 4 weeks (half of the HM scenario), but not for the first three months after sowing. Therefore, there are eight events in the first year (plant in March).			
put part	leaf, grain, stem				
cut fraction	0.3				

Watering Tables:

Below is the irrigation schedule for the High Maintenance lawn management scenario, as calculated using the "Basic Irrigation Management Toolkit" outlined in SA Water's Code of Practice for Irrigated Public Open Space (SA Water, 2015), and rounded to the nearest whole number.

Month	1	2	3	4	5	6	7	8	9	10	11	12	Sum
no. water events	9	7	5	2	0	0	0	0	2	5	7	8	45
mm\event	10	10	11	13	0	0	0	0	12	10	10	10	
mm total	90	70	55	26	0	0	0	0	24	50	70	80	465
extra parameters:													
functional purpose	TQVS3 (Local Sports Turf)												
weather station	Adelaide Airport												
dist. uniformity	80%												
turf type	Warm												
Root Zone (mm)	150												

Outlined in SA Water's Code of Practice for Irrigated Public Open Space (SA Water, 2015). For the Low Maintenance management scenario *HALF of the recommended irrigation was applied.*

Month	1	2	3	4	5	6	7	8	9	10	11	12	Sum
no. water events	7	5	4	2	0	0	0	0	1	3	5	6	33
mm\event	10	12	12	8	0	0	0	0	15	12	12	10	
mm total	70	60	48	16	0	0	0	0	15	36	60	60	365
extra params:													
functional purpose	TQVS4 (Passive Recreational Turf)												
weather station	Adelaide Airport												
dist. uniformity	80%												
turf type	Warm												
Root Zone (mm)	150												

For vegetable garden – taken from the edible gardens survey.

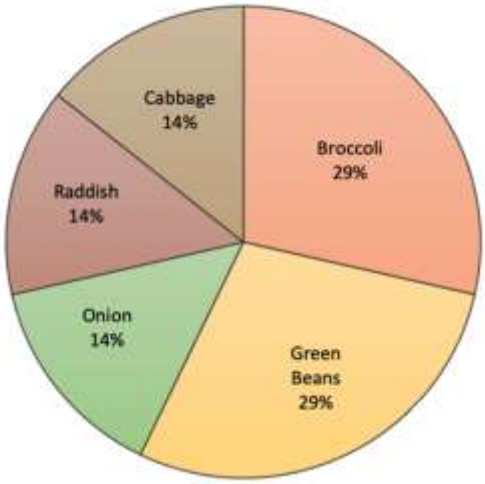
Month	1	2	3	4	5	6	7	8	9	10	11	12	Sum
no. water events	7	5	6	5	2	4	1	1	2	4	5	3	45
mm/event	3	8	5	5	8	2	10	5	5	5	5	10	
mm total	21	40	30	25	16	8	10	5	10	20	25	30	240

S2: Planting and harvesting
All crops are planted on the 1st day of the month and harvested on the 28th day of the month. Within each rotation some crops have multiple plantings based on the growth time until maturity, these are as follows (**Plant month/**harvest month):

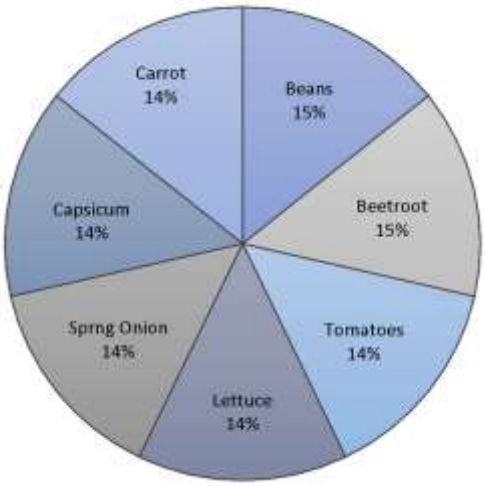
broccoli	***Mar***		*May*	***June***		*Aug*	***AW****
beans	***Sept***		*Nov*	***Dec***		*Feb*	***SS****
green beans	***Mar***		*May*	***June***		*Aug*	***AW****
beetroot	***Sept***		*Nov*	***Dec***		*Feb*	***SS****
onions	***Mar***					*Aug*	***AW****
tomatoes	***Sept***		*Nov*	***Dec***		*Feb*	***SS****
radish	***Mar***	*Apr*	***May***	*June*	***Jul***	*Aug*	***AW****
lettuce	***Sept***		*Nov*	***Dec***		*Feb*	***SS****
cabbage	***Mar***					*Aug*	***AW****
spring onion	***Sept***	*oct*	***Nov***	*Dec*	***Jan***	*Feb*	***SS****
broccoli	***Mar***		*May*	***June***		*Aug*	***AW****
capsicum	***Sept***		*Nov*	***Dec***		*Feb*	***SS****
green beans	***Mar***		*May*	***June***		*Aug*	***AW****
carrot	***Sept***		*Nov*	***Dec***		*Feb*	***SS****

* Season, either Autumn/Winter (AW) or Spring Summer (SS)

Autumn Winter



Summer Spring



S3

Input	**Value**	**units**	**Justification**
mowing	18.1672	kg co2e ha $^{-1}$ mowing event $^{-2}$	7.763784 L/Ha per mowing event (Saidani & Kim, 2021). The CE conversion of gasoline is 8.87 kg CO2 per gallon of gasoline (Lal, 2004). This = 2.34 kg CO2e per L. To determine the total kg of CO2 emitted each year by gasoline, we multiply 7.764 L ha−1 by 2.34 kg CO2 gal−1 and the mowing events each year.
	472.347	kg co2e / ha /yr / HMHW	26 events
	290.675	kg co2e / ha /yr / HMHW (1st)	16 events

	236.1736	kg co2e / ha / yr / LM	13 events
	145.3376	kg co2e / ha / yr / LM (1st)	8 events
irrigation	0.346	kg co2e m3 of irrigation	This is the value taken from the ozLCA dataset, analysed using openLCA under the IPPC 100yr GWP impact method.
	1608.900	kg co2e / ha /yr / HMHW	465 mm of irrigation = 4650 m^3 ha^{-1}
	1453.200	kg co2e / ha /yr / HMHW (1st)	420 mm of irrigation = 4200 m^3 ha^{-1}
	608.960	kg co2e / ha / yr / LM	176 mm irrigation = 1760 m^3 ha^{-1}
	1141.800	kg co2e / ha / yr / LM (1st)	330 mm of irrigation = 3300 m^3 ha^{-1}
	830.400	kg co2e /ha / yr / veg	240 mm of irrigation = 2400 m^3 ha^{-1}
N	4.76	kg CO2 e kg N -1	The conversion factor for Nitrogen fertilizer as reported by Lal (2004) via Gu, Crane, Hornberger, and Carrico (2015).
	1256.64	kg co2e / ha /yr / HMHW	4 lots of reno fertilizer - 66 kg / ha of N each fertilizing event
	765.408	kg co2e / ha /yr / HMHW (1st)	2 lots of Quickstart (14.4 kg / ha of N per fertilizing event) + 2 lots of Reno
	0	kg co2e / ha / yr / LM	No fertilization
	137.088	kg co2e / ha / yr / LM (1st)	2 lots of Quickstart (14.4 kg / ha of N per fertilizing event)
P	0.73	kg CO2 e kg P -1	The Conversion factor for P fertilizer as reported by Lal (2004) via Gu et al. (2015).
	43.8	kg co2e / ha /yr / HMHW	4 lots of Reno fertilizer - 15 kg / ha of P each fertilizing event
	47.742	kg co2e / ha /yr / HMHW (1st)	2 lots of Quickstart (17.7 kg / ha of P per fertilizing event) + 2 lots of Reno
	0	kg co2e / ha / yr / LM	no fertilization
	25.842	kg co2e / ha / yr / LM (1st)	2 lots of Quickstart (17.7 kg / ha of P per fertilizing event)
K	0.55	kg CO2 e kg P -1	The conversion factor for K fertilizer as reported by Lal (2004) via Gu et al. (2015).
	52.8	kg co2e / ha /yr / HMHW	4 lots of reno fertilizer - 24 kg / ha of P each fertilizing event
	46.2	kg co2e / ha /yr / HMHW (1st)	2 lots of quick start (18 kg / ha of P per fertilizing event) + 2 lots of reno
	0	kg co2e / ha / yr / LM	no fertilization
	19.8	kg co2e / ha / yr / LM (1st)	2 lots of Quickstart (18 kg / ha of P per fertilizing event)
Compost	0.020	kg co2e / kg compost	For Home Compost (HC) only (Lu, Qu, & El Hanandeh, 2020).
	5820	kg co2e / ha / yr	5cm a year (2 cm x 2 events + 1 cm X 1 event), with PB of 582 kg/ m3 (Khater, 2015) 5cm over 1 ha = 500m3 of compost. 500m3 at 582kg/m3 = 291 000 kg/ha

Replacing conventional farming	-27430	kg CO2 e Ha	0.54 kg of co2e / kg of produce (Clune, Crossin, & Verghese, 2017). Av. Yield from the EG survey was 5.08 kg m3. 0.54 * 5.08 = 2.743 kg/m3 = 27430 kg.

S4

GWP of various crops grown commercially in Australia, as reported by Clune et al. (2017) in their systematic review of greenhouse gas emissions for different fresh food categories. Only vegetable crops grown in Australia were selected from Clune, Crossin and Verghese's (2017) meta-analysis and included in the calculation for this work.

Crop	**Original Data Source**	**kg CO2 e/kg produce**
Asparagus	(Maraseni, Cockfield, Maroulis, & Chen, 2010)	2.54
Beetroot	(Maraseni et al., 2010)	0.24
Broccoli	(Maraseni et al., 2010)	1.73
Cabbage	(Maraseni et al., 2010)	0.23
Capsicums	(Maraseni et al., 2010)	0.23
Carrots	(Maraseni et al., 2010)	0.20
Cauliflowers	(Maraseni et al., 2010)	0.38
Celery	(Maraseni et al., 2010)	0.18
Chillies	(Maraseni et al., 2010)	0.66
Cucumbers	(Maraseni et al., 2010)	0.13
Lettuce	(Gunady, Biswas, Solah, & James, 2012)	0.22
Lettuce	(Maraseni et al., 2010)	0.32
Onion	(Maraseni et al., 2010)	0.21
Peppers	(Maraseni et al., 2010)	0.23
Potatoes	(Maraseni et al., 2010)	0.21
Pumpkins	(Maraseni et al., 2010)	0.25
Tomatoes	(Maraseni et al., 2010)	0.22
Zucchini/Button Squash	(Maraseni et al., 2010)	1.17
AVERAGE		**0.54**

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