Quantifying and Mapping Urban Ecosystem Services in Johannesburg, South Africa

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A Dissertation submitted to the Faculty of Science, University of the Witwatersrand, Johannesburg, in fulfilment of the requirements for the degree of Master of Science.

Johannesburg

2050

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Signed 10 June 2024

Candidate's Declaration

I declare that this Dissertation is my own, unaided work. It is being submitted for the Degree Master of Science at the University of the Witwatersrand, Johannesburg. It has not been submitted before for any degree or examination at any other University.

Candidate's Signature: ____________________ **Date:** _______10 June 2024_____

Acknowledgements

I would like to thank my supervisor, Dr Ute Schwaibold, for her guidance, mentorship and support throughout the research and writing of this thesis. I would also like to thank my parents and my partner, Georgia, for their support throughout this process. Thanks also to the management at the Johannesburg country club for allowing me to sample on their premises. I also acknowledge the kindness of all the individuals who permitted me to conduct field sampling in their private gardens. Finally, I would like to thank Carl Belger and Thulani Hadebe for their assistance in conducting some field samples.

Abstract

Modern cities face a wide range of challenges such as flooding and heat stress, which are driven by urbanisation and exacerbated by the impacts of climatic change. The ecosystem services provided by green spaces in cities have become a crucial element in addressing these challenges by supporting climate change mitigation and adaptation. The first step in maintaining and improving the supply of these services is their quantification and mapping. However, large knowledge gaps exist in South Africa and Johannesburg relating to the provision of urban ecosystem services.

This study aimed to quantify the supply of three important urban ecosystem services (carbon storage, runoff retention and cooling) and map their distribution across the wards of Johannesburg. Carbon storage was quantified through field sampling of four urban forest types (roadside trees, parks, gardens and nature reserves) and the use of biomass equations. InVEST's urban flood risk mitigation model was used to quantify runoff retention, while cooling was quantified by deriving land surface temperatures from Landsat satellite imagery, which were then used as inputs for a cooling indicator. All three services were mapped across the wards of Johannesburg and then normalised for comparison.

The results revealed that Johannesburg's urban forest stores 2.4 million tonnes of carbon, with significant differences in carbon storage between forest types. Johannesburg's ecosystem services provide great value in mitigating urban challenges, retaining 20.9 million m^3 of runoff during a 50 mm storm, and providing cooling services across most of the city. However, the supply of these services is unequal, with large spatial disparities between the northern and southern regions in the city. Numerous wards receive critically low supply of these services, making them vulnerable to the impacts of climatic change. The northern-central wards have optimal supply of all three services, highlighting synergies between services.

Ultimately, these three services have immense value in the Johannesburg context and play key roles in supporting the city's climate change mitigation and adaptation, through the multi-functional delivery of ecosystem services from urban green infrastructure. By mapping these services at the ward scale, our findings can be used to accurately inform authorities and decision makers of priority areas for intervention, as well as key areas for conservation.

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List of Abbreviations

Abbreviation	Definition
AGC	Aboveground Carbon
AGB	Aboveground biomass
BA	Basal Area
CoJ	City of Johannesburg
CN	Curve number
DBH	Diameter at breast height
ES	Ecosystem services
ESA	European Space Agency
GI	Green infrastructure
GEE	Google earth engine
HSG	Hydrologic Soil Groups
inVEST	Integrated valuation of Ecosystem Services and Trade-offs
IPCC	Intergovernmental Panel on Climate Change
LST	Land Surface Temperature
NDC	Nationally Determined Contribution
OSM	OpenStreetMap
RS	Roadside
SANLC	South African National Land Cover
SCS	Soil Conservation Services
UES	Urban Ecosystem Services
UHI	Urban Heat Island
UGS	Urban greenspace(s)
UF	Urban Forest
UFRM	Urban Flood Risk Mitigation
UGBS	Urban Green and Blue Spaces
UNCCC	United Nationals National Climate Change Conference

Chapter 1: Introduction

Modern society is undergoing a dramatic, widespread, and rapid transition toward urban living (Grimm *et al*., 2008). The urban population increased by 3.4 billion between 1950 and 2018, with more than half the global human population currently living in urban systems (United Nations, 2019). This demographic transition shows no signs of slowing down in the near future, with projections indicating that nearly 70% of the global population will reside in urban areas by 2050 (United Nations, 2019). This increase in the urban population is expected to occur almost entirely (90-95%) on the Asian and African continents, which are already urbanising at a greater rate than anywhere else in the world (United Nations, 2019).

The global urban transition over the last century has taken place on less than three percent of the earth's terrestrial surface (Schneider *et al*., 2010; Liu *et al*., 2018; Huang *et al*., 2021). Consequently, despite the magnitude and intensity of population growth and associated land cover change, these effects have been concentrated on a relatively small proportion of the earth's surface, placing immense pressure on the humans and ecosystems within these systems (Grimm et al., 2008; De Carvalho & Szlafsztein, 2019). While urbanisation has spurred economic growth and cultivated technological innovation, it has also brought about profound changes in the earth's biosphere and atmosphere, driving environmental change at local and global scales.

At the global scale, urban systems are key drivers of climatic change (Grimm *et al*., 2008; Seto *et al*., 2014). Cities are responsible for nearly 80% of global carbon emissions driven by their role act as hotspots for global material production and resource consumption (Grimm *et al*., 2008; Seto *et al*., 2014; Garrick *et al*., 2019). These anthropogenic CO_2 emissions underly dangerous climatic change, which has and will continue to increase global surface temperatures and drive more frequent and severe extreme weather events (IPCC, 2021). Furthermore, the impacts of high resource demands within cities can have large-scale effects on ecosystems far beyond the boundaries of the cities themselves (Grimm *et al*., 2008).

While being heavily implicated in global climatic change as principal greenhouse gas (GHG) emitters, urban systems are also affected by local-scale climatic changes associated with urbanisation (Seto *et al*., 2014). The most well-known and widely reported instance of local climate alteration due to anthropogenic activity is the urban heat island (UHI) effect (Voogt & Oke, 2003; Grimm *et al*., 2008; Santamouris, 2015; Santamouris, 2018; Nastran *et al*., 2019), an urban specific climate in which urban, built-up areas have higher temperatures than their surrounding rural or natural environments (Oke, 1997; Voogt & Oke, 2003; Busato *et al*., 2014) as they retain more heat. The increase in temperatures caused by the UHI effect have major consequences for both human and ecosystem health, particularly in the context of climate change (Kleerekoper *et al*., 2012; Heaviside *et al*., 2016; Santamouris, 2018; Li *et al*., 2019; Yang *et al*., 2020; Santamouris, 2020; Yin *et al*., 2023).

In addition, this increased impervious land cover which is synonymous with urban systems has also altered hydrological functioning in cities, creating numerous local challenges including water pollution and enhanced flooding (Grimm *et al*., 2008). The replacement of natural surfaces with artificial ones diminishes the capacity for water infiltration, and other means of hydrological losses (e.g. percolation in soils), thereby lowering the threshold at which stormwater runoff is initiated (Villarreal & Bengtsson, 2005; Berland *et al*., 2017), making cities vulnerable to flooding events (Grimm *et al*., 2008; Villarreal & Bengtsson, 2005; Huong & Pathirana, 2013). This vulnerability is enhanced by the absence of vegetation and tree canopies, which intercept rainfall and slows surface flow, enabling greater percolation and reducing pressure on stormwater drainage systems (Bolund & Hunhammar, 1999; Villarreal & Bengtsson, 2005; Pataki *et al*., 2011; Berland *et al*., 2017).

Air pollution has also arisen as another challenge in megacities, due to these systems acting as point sources for a wide range of greenhouse gasses and pollutants due to the aggregation of industrial and transportation activities (Molina & Molina, 2004; Pataki *et al*., 2006; Xie *et al*., 2017). A further key impact of urbanisation is the loss of biodiversity, as species richness and evenness for the majority of biotic communities are reduced (Paul and Meyer, 2001; McKinney, 2002; McKinney, 2008; Fenoglio *et al*., 2020) due to ecological degradation, the fragmentation of landscapes and disturbed urban ecosystems caused by urbanisation (Zhang & Muñoz Ramírez, 2019). Across the world, urbanisation and the growth of cities act as an immense homogenising force, in which urban-adapted species are becoming widespread and abundant, whilst indigenous species are diminished (McKinney, 2006).

In addition to these already existing urban problems, global climatic change is set to exacerbate urban challenges and pressures. It is indisputable that anthropogenic influence has caused the warming of the earth's land, oceans, and atmosphere alongside driving swift and intensive changes to the biosphere, oceans, and atmosphere (IPCC, 2021). The magnitude and scale of changes to the earth's climate system are unprecedented (IPCC, 2021) and presents a major threat to the sustainability of human society. Both hot extremes and severe rainfall events have and will continue to increase in both frequency and intensity, particularly in Africa and Asia (IPCC, 2019IPCC, 2021). In addition, global surface temperatures will continue to rise unless drastic reductions in GHG emissions take place (IPCC, 2021). Consequently, already existing urban pressures such as urban heat islands, biodiversity loss and flooding will be greatly exacerbated (Seto *et al*., 2014; Dodman *et al*., 2021).

Urban ecosystem services (UES), provided by biodiversity and green infrastructure within cities, have emerged as an important solution to existing and future urban challenges, which can act as a tool for both climate mitigation and adaptation. In particular, the regulation of local temperatures and mitigation of the UHI (Bowler *et al*., 2010; Schwarz *et al*., 2011; Lafortezza *et al*., 2009; Rocha *et al*., 2022), flood regulation (Pataki *et al*., 2011; Berland *et al*., 2017; Li *et al*., 2019), mitigation of climate change through carbon sequestration and storage (Nowak *et al*., 2013; Lindén *et al*., 2020; Dhyani *et al*., 2021) and air purification (Janhäll, 2015) have been highlighted as important ecosystem services in the urban context.

The quantification of these services is an imperative first step In their preservation and improvement (Radford & James, 2013). The quantification of regulating services such as local climate regulation (cooling) and flood regulation is crucial for informing decision making and addressing climate change impacts in growing urban systems (Farrugia *et al*., 2013). Meanwhile, the quantification of carbon stocks is the crucial first step in creating and informing effective climate change mitigation strategies in urban systems (Nowak *et al*., 2013; Davies *et al*., 2013; Liu and Li, 2012). Determining the spatial distribution of these services also aids in identifying key areas with low supply as well as priority areas for conservation (Farrugia *et al*., 2013). Maintaining services which build resilience to climatic change are particularly important in the African context, due to the continent's vulnerability to climate change impacts.

1.1 Knowledge gap

For a prolonged period, issues relating to biodiversity were not adequately considered in the context of urban development (Sieber & Pons, 2015). As a result there are still significant knowledge gaps regarding cities and their ecosystems (Gómez-Baggethun & Barton, 2013; Hubacek & Kronenberg, 2013; Haase *et al*., 2014; Shackleton *et al*., 2021). Both ecosystem services (ES) and UES remain markedly under-studied in Africa (Wangai *et al*., 2016; du Toit *et al*., 2018; Veerkamp *et al*., 2021). The vast majority of UES research has been undertaken in Europe, North America and China (Haase *et al*., 2014; Crossman *et al*., 2013; Veerkamp *et al*., 2021). Thus there is clear need for the enhancement and broadening of the knowledge base surrounding the application and implementation of the ES framework in Africa, supported by the conjunct necessity to meet human needs, particularly in rapidly expanding urban areas (Wangai *et al*., 2016).

Very few studies have explored UES in South Africa and where they have, the focus has been on a single service (e.g. Stoffberg *et al*., 2010; O'Donoghue & Shackleton, 2013) or they are limited to small-scale case-studies (Schäffler & Swilling, 2013). In fact, very few studies have mapped urban ecosystem services, at the city-scale in South Africa outside of an assessment of ES in the City of Cape town which utilised a land cover scoring method (O'Farrell *et al*., 2012) and an unpublished study in Durban, which mapped ES using inVEST models (Glenday, 2012). Studies which have mapped ES in South Africa have generally been conducted at national (See Egoh *et al*., 2008; Turpie *et al*., 2017) or regional scales (See SANBI, 2021).

To my knowledge, no biophysical quantification of UES supply has been done for the City of Johannesburg, or any other South African cities. In fact, despite being South Africa's most populous and heavily treed city, only limited UES research has been conducted in Johannesburg (see Schäffler & Swilling, 2013), with no mapping or quantification of ES supply having been conducted.

1.2 Aim and Objectives

This study aimed to assess ecosystem service supply spatially and quantitatively in the City of Johannesburg, South Africa. The contribution of Johannesburg's green infrastructure (GI) toward climate resilience and human wellbeing was investigated through three key urban ecosystem services (UES).

Objectives:

(1) To quantify the supply of three urban ecosystem services (carbon storage, runoff retention, and cooling) from green infrastructure in Johannesburg, using bio-physical or non-economic metrics;

(2) To map the spatial distribution of urban ecosystem service supply across all the administrative wards in Johannesburg and make recommendations for planning

1.3 Layout of this dissertation

In quantifying three different ecosystem services in Johannesburg, three distinct methodologies were used, including field sampling, remote sensing and modelling approaches. These methods produced different results, specific to each service. In order to aid the readability and comprehensibility of this thesis, the methods and results sections are partitioned between the different services. After the introduction and literature review (Chapters 1 and 2), the methods for quantifying carbon storage, followed by the carbon storage results are presented in Chapter 3. Then, the methods and results for cooling and runoff quantification are presented in Chapter 4. Following this, the methods and results for the comparison between services are presented in Chapter 5. These chapters are followed by a discussion and conclusion in Chapter 6. The dissertation closes with a comprehensive reference list and eight appendices

Chapter 2: Literature Review

Modern society is rapidly shifting towards urban living, with over half the global population residing in urban areas (United Nations, 2019). This proportion is projected to increase to 70% by the year 2050, with an increase in the urban population of 2.5 billion. This growth in the urban population is expected to occur almost entirely on the African and Asian continents, as developing countries approach the level of urbanisation evident in current industrialised countries (United Nations, 2006; United Nations, 2019).

Cities serve as primary hubs for humans and infrastructure and as centres of economic functioning; in turn, their resource requirements are vast, applying immense pressure on both the surrounding and global environment (Grimm *et al*., 2008; Garrick *et al*., 2019). Furthermore, the process of urbanisation serves as a key driver of global land use change as urban land is expanding at a greater rate than any other land use (Pickett *et al*., 2001; McKinney, 2002; Meyerson *et al*., 2007; Davies *et al*., 2009; Holt *et al*., 2015). Thus, urban systems prompt environmental changes at multiple scales. In this context, the urban transition has been identified as the largest human-environment experiment in the history of humankind (Meyerson *et al*., 2007).

At the global scale, cities are principal emitters of greenhouse gasses, playing a major part in driving global climatic change (Grimm *et al*., 2008; Seto *et al*., 2014). However, urban systems also face numerous local challenges associated with the built up and transformed nature of urban landscapes. One important challenge is the urban heat island (UHI) effect, a local climate phenomenon in which built-up areas have higher temperatures than their surrounding rural environments (Oke, 1997; Voogt & Oke, 2003; Busato *et al*., 2014). The UHI represents a distinctly urban pressure and its formation is influenced by a multitude of urban characteristics (Grimm *et al*., 2008; Santamouris, 2015; Mohajerani *et al*., 2017; Li *et al*., 2020). These include large areas covered by impervious surfaces which have a high heat retention capacity alongside limited albedo, as well as a reduction in vegetation and waterbodies, which limits the loss of heat through evaporative cooling (Stone *et al*., 2010; Santamouris, 2015; Mohajerani *et al*., 2017). Anthropogenic heat sources such as industrial machinery, cars and generators also contribute to the formation of the UHI (Stone *et al*., 2010; Li *et al*., 2020). The increase in temperatures caused by the UHI effect poses major risks to both human and environmental health, particularly in the context of climate change

(Kleerekoper *et al*., 2012; Heaviside *et al*., 2016; Santamouris, 2018; Li *et al*., 2019; Yang *et al*., 2020; Santamouris, 2020; Yin *et al*., 2023).

The UHI effect is not the only adverse impact of the transformed urban landscape, as cities also face heightened flood risks. By substituting natural, pervious surfaces with artificial, impervious ones, urban surfaces have reduced infiltration capacity and greater surface runoff (Villarreal & Bengtsson, 2005; Berland *et al*., 2017). The increased volume of runoff, alongside the limited capacity of impervious surfaces to slow down the flow of water makes cities vulnerable to flood events (Villarreal & Bengtsson, 2005; Grimm *et al*., 2008; Huong & Pathirana, 2013). Additionally, the loss of green spaces in urban systems reduces the interception of rainfall by tree canopies and vegetation, as well as the capacity for vegetation to slow down the flow of water (Villarreal & Bengtsson, 2005; Pataki *et al*., 2011; Berland *et al*., 2017). Consequently, urban flooding has become a key challenge facing cities and have already driven economic losses, fatalities and illnesses worldwide (IPCC, 2022).

Numerous other challenges face urban systems, including but not limited to air pollution , water pollution and the widescale loss of biodiversity (Grimm *et al*., 2008; McKinney, 2008; Fenoglio *et al*., 2020; Pataki *et al*., 2006; Xie *et al*., 2017). Furthermore, the increase in temperature and rainfall extremes associated with climatic change will exacerbate many of these urban pressures, including the UHI effect and urban flooding (IPCC, 2021; Dodman *et al*., 2022). As a result, many of the negative and damaging feedback of climatic change are felt most acutely in cities. As a consequence of the ever-increasing proportion of the global population living in urban systems, more and more people are directly faced with challenges associated with global climatic change, including a consistent increase in the occurrence of extreme weather events and the loss of biodiversity and terrestrial ecosystems, alongside the ecosystem services which they provide (IPCC, 2019). This challenge is exacerbated by the severe vulnerability of urban systems to extreme weather events (De Sherbinin *et al*., 2007; Dodman *et al*., 2022). In this context, it is clear that we need solutions that address typical urban challenges, reduce anthropogenic emissions, enhance urban resilience to climatic change and reduce the loss of biodiversity so as to preserve human well-being in cities and ensure a sustainable urban future (Radford & James, 2013).

2.1 Urban Ecosystem Services (UES)

The extreme challenges facing human-environmental systems around the world have created a need for integrative and adaptive management to simultaneously deal with social and environmental pressures (Burkhard *et al*., 2010; Niemelä, 2014; Babí Almenar *et al*., 2021). In this context, the concept of ecosystem services arose as a means to integrate social, environmental and economic goals (Seppelt *et al*., 2011; Granek *et al*., 2010). The concept was popularised by the release of the Millennium Ecosystem Assessment (MEA, 2005) which served as the earliest, comprehensive, globe-wide assessment of the ecosystem services (Haines-Young & Potschin, 2010; Hubacek & Kronenberg, 2013). Since then, ES considerations have become mainstream, particularly over the last decade (Zhang & Muñoz Ramírez, 2019), evident in its increasing political profile and adoption into EU policy (Bouwma *et al*., 2018). The increasing prominence of the concept is explained by its ability to reframe the value of the conservation of nature, rather than being in contrast to or additional to human and societal goals, but rather as a crucial element to achieving these goals (Hubacek & Kronenberg, 2013).

The ES concept serves as a framework for the characterisation, definition and analysis of the interdependencies and linkages between natural and anthropogenic systems (Burkhard *et al*., 2010). Thus it serves as a principal model for establishing connections between the functioning of ecosystems and human well-being (Fisher *et al*., 2009). The understanding of this link between humans and nature is crucial in a wide variety of decision making contexts, particularly in urban systems.

Various environmental pressures and changes are driven by urban systems, whilst simultaneously affecting these systems (Grimm *et al*., 2008), enhancing their complexity and elevating their importance to both global and local changes in the environment. However, urban systems were largely neglected by ecologists throughout the 20th century, and consequently environmental issues in urban systems received little aid from ecological study. It is now clear that we need solutions that address typical urban challenges, enhance urban resilience to climatic change and reduce the loss of biodiversity so as to preserve human well-being in cities and ensure a sustainable urban future (Radford & James, 2013). The concept of urban ecosystem services (UES) has arisen as a key means of studying and addressing these urban challenges and integrating social, economic and environmental goals. UES refer to the services provisioned by urban ecosystems and their components (Gómez-Baggethun & Barton, 2013) and is derived from the understanding of ecosystem services (ES), defined as the "contributions of ecosystem structure and function to human wellbeing" (Burkhard and Maes, 2017, p25) or the direct or indirect benefits which humans derive from ecosystems (TEEB, 2010). ES are generally classified as belonging to one of four categories, namely regulating (moderate ecosystem processes and include air quality regulation, local climate regulation), cultural (non-material benefits which humans receive from ecosystems such aesthetics, recreation), provisioning (material outputs from ecosystems such as wood, water or food) and supporting (underpin the production of other services, e.g. habitat provision) services (MEA, 2005; TEEB, 2010)..

For a prolonged period, issues relating to biodiversity were not adequately engaged with within the context of urban development (Sieber & Pons, 2015). As such, urban ecosystems are relatively less pronounced in ES research than systems such as wetlands or forests (Gómez-Baggethun & Barton, 2013). Regardless, since the publication of the seminal article by Bolund & Hunhammar, (1999), a growing body of literature has built knowledge towards the understanding of UES through biophysical (Pataki *et al*., 2011; Escobedo *et al*., 2011), economic (Sander *et al*., 2010) and socio-cultural perspectives (Chiesura, 2004; Erik Andersson *et al*., 2007). UES were also represented in large scale initiatives (MEA, 2005; TEEB, 2010) and are establishing a strong foothold in policy debates on green infrastructure (EEA, 2011). Publications relating to the study of urban ecosystem services have followed a trend of exponential growth since their onset (Fisher *et al*., 2009; Hubacek & Kronenberg, 2013). The rise in attention paid to UES is primarily driven by their capacity to reduce, moderate and regulate various pressures and impacts linked to urbanisation (Elmqvist *et al*., 2015; Escobedo *et al*., 2019).

Different habitats provide varying ES and in turn alter the importance of particular ES provided, whereby forests for instance are important for carbon sequestration, and agroecosystems for food production. The UES delivered by green and blue infrastructure in urban systems provides a wide range of benefits to human health and well-being, which is crucial for the management of existing and future urban problems (Gómez-Baggethun & Barton, 2013). Some of the UES which have been emphasised in the literature as particularly important for the maintenance of human well-being in urban systems include regulating the urban heat island effect and mitigating the negative health impacts of increasing urban temperatures (Bowler *et al*., 2010; Schwarz *et al*., 2011; Lafortezza *et al*., 2009; Rocha *et al*., 2022); reducing the risks and buffering the impacts of flooding (Bolund & Hunhammar, 1999; Villarreal & Bengtsson, 2005; Pataki *et al*., 2011; Berland *et al*., 2017; Li *et al*., 2019); reducing mortality rates and enhancing human mental and physical health (Tzoulas *et al*., 2007; Maas, 2006; Gascon *et al*., 2015; van den Berg *et al*., 2015; Nowak *et al*., 2018); mitigating climate change through carbon storage and sequestration (Liu & Li, 2012; Nowak *et al*., 2013; Sun *et al*., 2019; Lindén *et al*., 2020; Dhyani *et al*., 2021); air purification (Janhäll, 2015); and improving environmental awareness (Gómez-Baggethun & Barton, 2013; Haase *et al*., 2014).

In the context of the UHI effect (Busato *et al*., 2014) and increasing global temperatures (IPCC, 2021), the provision of local climate regulation or cooling service becomes particularly important. An effective means of mitigating the UHI effect and regulating urban temperatures is through the utilisation of urban green spaces and the enhancement of vegetation cover (Hardin & Jensen, 2007; Grimm *et al*., 2008; Bowler *et al*., 2010; Busato *et al*., 2014; Skelhorn *et al*., 2014; Nastran *et al*., 2019). Vegetation can reduce surface and air temperatures through evapotranspiration (Hardin & Jensen, 2007; Bowler *et al*., 2010; Skelhorn *et al*., 2014; Nastran *et al*., 2019). This cooling effect is enhanced where foliage density and canopy cover is greater, due to the larger surface area from which water can be transpired (Hardin & Jensen, 2007). Furthermore, tree canopies provide shading, effectively minimising the proportion of solar radiation which reaches the ground (Rosenzweig *et al*., 2006; Hsieh et al., 2018).

In the context of the immense air pollution challenges faced by cities around the world (Molina & Molina, 2004; Grimm *et al*., 2008) and their hazardous health effects (Leiva G *et al*., 2013), air purification is another key service provided by urban green spaces. Vegetation provides an inherent air purification capacity due to its ability to absorb pollutants (Salmond *et al*., 2016), and affect the dispersion of and deposition of pollutants (Tiwary *et al*., 2009; Baró *et al*., 2016; Xing & Brimblecombe, 2019). A large body of research provides empirical evidence that urban vegetation enhances air quality through the removal of air pollutants from the atmosphere (see McDonald *et al*., 2007; Escobedo *et al*., 2008; Yang *et al*., 2008; Escobedo & Nowak, 2009; Tallis *et al*., 2011; Baumgardner *et al*., 2012; Janhäll, 2015; Jim & Chen, 2008; Selmi *et al*., 2016; Nowak *et al*., 2018; Vieira *et al*., 2018; Lindén et al., 2023).

The regulation of flooding or runoff retention is another service that is particularly important in an urban context (Whitford, 2001; Farrugia *et al*., 2013). When precipitation occurs over the impervious surfaces in urban systems, it is converted into stormwater runoff, creating issues relating to both the quantity and quality of water that enters close-by natural water bodies (Berland *et al*., 2017). A key solution is the utilisation of urban green space (UGS) with dense vegetation which provides a runoff retention service. Heavily vegetated areas have a far higher infiltration capacity as they allow water to move through the vegetation slowly, thereby reducing the amount of runoff that occurs after precipitation events (Villarreal & Bengtsson, 2005). This service is driven by the interception of rainfall via plant leaves and tree canopies alongside the soil beneath them, which enables the storage of water in pore spaces until percolation occurs (Bolund & Hunhammar, 1999; Villarreal & Bengtsson, 2005; Pataki *et al*., 2011). This improved infiltration capacity aids in mitigating flood events downstream and reducing pressure on stormwater systems (Bolund & Hunhammar, 1999; Villarreal & Bengtsson, 2005; Pataki *et al*., 2011).

Both the UHI effect and urban flooding as well as other impacts of climate change are already affecting all inhabited regions across the globe and will continue to worsen unless drastic reductions in GHG emissions occur (IPCC, 2021). There is thus a great need to reduce GHG emissions and mitigate climatic change to avoid devastating consequences for both ecosystem and human health. One strategy, which has become central to climate mitigation efforts, is the carbon storage and sequestration services provided by natural vegetation (Muñoz-Vallés *et al*., 2013; Kuittinen *et al*., 2016; Chen, 2015; Strohbach & Haase, 2012; Nowak *et al*., 2013; Ariluoma *et al*., 2021). The preservation and enhancement of existing natural carbon sinks as well as the addition of new ones are currently considered among the principal strategies by which global climate change can be addressed, and are pursued in tandem to execution of carbon emission reduction policies (Muñoz-Vallés *et al*., 2013).

Evidence is mounting to suggest that carbon sinks in urban areas are far more substantial than previously assumed (Pataki *et al*., 2011; Davies *et al*., 2013). In turn, urban greening strategies are increasingly being incorporated into urban climate change mitigation policies, due to their capacity to offset urban emissions and act as a local sink (Weissert *et al*., 2014). This service is rooted in the ability of vegetation to regulate atmospheric CO_2. Through the process of photosynthesis they can sequester carbon and store it in their biomass during growth, thereby removing it from the atmosphere (Nowak & Crane, 2002; Liu & Li, 2012; Dorendorf *et al*., 2015; Kuittinen *et al*., 2016). Ultimately, by providing carbon storage and sequestration services in addition to mitigating other urban challenges, urban green spaces can be instruments of climate change mitigation and adaptation (Muñoz-Vallés *et al*., 2013).

Beyond regulating services which address biophysical urban challenges, cultural ES contribute significantly to the wellbeing of social-ecological systems and cities (Martín-López *et al*., 2009; Niemelä, 2014). Cultural services are particularly important in urban systems as compared to rural settings and build beneficial conditions for urban citizens (Haines-Young and Potschin, 2008; Niemelä, 2014). This relates largely to the decoupling of humans and nature in urban systems, where human-nature connections are diminishing despite their importance in mental and physical well-being., making cultural services a priority with regard to preservation and delivery in urban systems (Radford & James, 2013).

The value of urban ecosystem services in addressing urban challenges and enhancing the well-being of urban dwellers is further enhanced by the capacity for green infrastructure to deliver multiple services simultaneously. The multi-functionality of green infrastructure draws clear distinctions from traditional grey infrastructure, which is generally designed and implemented for a single use or purpose. Meanwhile, ecological systems have the capacity to provide an array of functions with great flexibility and at multiple scales, simultaneously providing a range of ES (GCRO, 2013; Derkzen *et al*., 2015; Liu and Russo, 2021). This is valuable for planning, due to its unique ability to meet multiple municipal goals, simultaneously (GCRO, 2013).

2.1.1 Urban ecosystem services and Resilience

In the context of rapidly unfolding, global climatic change, urban systems are increasingly facing threats and challenges associated with environmental extremes which will occur with greater frequency and intensity (Seto *et al.*, 2014; IPCC 2021; Dodman *et al*., 2022). In this context, the need for building urban resilience becomes evident and accordingly has risen to prominence in recent years, in the domains of science and practice, particularly in the field of urban planning (Quinlan *et al*., 2016; Fu *et al*., 2021; Cobbinah, 2021; Elmqvist *et al*., 2019; Moser *et al*., 2019) and in ES literature (De Luca *et al*., 2021; McPhearson *et al*., 2015; Fedele *et al*., 2017). It is pivotal that modern cities build adaptive capacity and resilience to enable them to mitigate, absorb and adapt to various disturbances whilst still preserving their social, economic and environmental functions and the general organisation of the system (Fu *et al*., 2021).

In this context, UES can make an important contribution to the enhancement of urban resilience and adaptive capacity (Gómez-Baggethun & Barton, 2013). Urban pressures and challenges can be effectively diminished by UGS, which simultaneously provide various other services (Derkzen *et al*., 2015). UGS provide a supply of crucial services that meet several key urban challenges needing to be tackled *in situ* within the time and place in which the challenge arises, such as urban heat stress or extreme stormwater runoff (Derkzen *et al*., 2015). Thus, the preservation of ES serves as a primary adaptation measure which can be used to bolster the adaptive capacity of society (IPCC, 2018).

Ultimately, UES enhance the quality of life, maintain human health, mitigate climate change and build resilience towards its impacts (Gómez-Baggethun & Barton, 2013; Berglihn & Gómez-Baggethun, 2021). Yet, as a consequence of urban development, many urban systems have greatly reduced ES supply and a simultaneously ever-increasing demand for those ES (Eigenbrod *et al*., 2011; Liyun *et al*., 2018; Liu *et al*., 2019; De Carvalho & Szlafsztein, 2019). In turn, it has been suggested that such services need to be more explicitly addressed in the generation of strategies and plans geared toward sustainable development, resilience and habitability in urban contexts (Gómez-Baggethun & Barton, 2013). Long-term strategies which maintain and enhance ES are pivotal to creating resilient, sustainable and liveable cities (Ahern *et al*., 2014). Furthermore, the supply and preservation of ES are largely reliant on suitable urban planning (Colding, 2011). A prominent and fairly novel approach toward the integration of ES into urban planning and policy is the green infrastructure concept (Niemelä, 2014).

The concept of green infrastructure (GI) can be broadly understood as a deliberately designed network of natural or semi-natural spaces alongside other environmental elements, which is managed and organised to supply ES (European Environment Agency, 2011; de Manuel *et al*., 2021). The implementation and establishment of green infrastructure has been proposed as an effective means of maintaining the supply of ES in urban areas alongside conserving the biodiversity from which these services stem (European Environment Agency, 2011; Pauleit *et al*., 2011). A wide variety of urban components can make up the green infrastructure, including urban forests, parks, street trees, green rooves, nature reserves, remnant vegetation and various others.

Attempts to make the ES approach functional and operational for effective GI planning have been gaining increasing attention across scientific, policy making, governance and decision making contexts (EEA, 2014; Kabisch, 2015; Bouwma *et al*., 2018). One prominent approach to improving the planning of green infrastructure for ecosystem service delivery is the mapping of ecosystem service supply. ES assessments and mapping provides spatial analyses on a particular scale, which in turn enables the maintenance of multi-functional landscapes and aids in urban green infrastructure planning (Zhang & Muñoz Ramírez, 2019). Thus, this study will quantify and map ecosystem service supply in Johannesburg, to support green infrastructure planning.

2.2 Quantifying and mapping the supply of ecosystem services

The quantification of ES supply (at multiple scales) is imperative for the preservation and improvement of ES (Radford & James, 2013; Derkzen *et al*., 2015). The quantification of regulating services such as local climate regulation (cooling) and flood regulation is important for informing decision making and tackling climate change impacts in growing urban systems (Farrugia *et al*., 2013). In the case of carbon storage services, the quantification of carbon stocks is the crucial first step in creating and informing local management policies and strategies to maintain the provision of this important ecosystem service (Liu & Li, 2012; Davies *et al*., 2013; Nowak *et al*., 2013). Furthermore, UES quantification also enables the contribution of UGS or green infrastructure toward human well-being to be measured (de Manuel *et al*., 2021). Measuring these benefits allows for integration into decision making contexts, more effective urban planning, highlights areas for conservation and areas which need enhanced green infrastructure (Lindén *et al*., 2020; de Manuel *et al*., 2021). Moreover, the quantification and mapping of UES as provided by urban green space allows for an understanding of the spatial distribution and effectiveness of green space in supplying different services, which can be useful information for consideration by urban planners (de Manuel *et al*., 2021). Policy makers and developers could benefit from ES quantification, by enabling the identification of key target areas within their policies and projects which would greatly benefit from improved green infrastructure (Farrugia *et al*., 2013). Consequently, understanding the variation in spatial distribution of ES in urban systems is crucial for addressing issues of unequal access to these services (Ernstson, 2013; Kremer *et al*., 2016). It is crucial that local decision making is supported by accurate and in-depth information pertaining to the supply of ES (Wangai *et al*., 2016). High resolution, local scale UES valuations are crucial for integration into urban policy, urban planning and to enable effective decision making (Haase *et al*., 2014).

In order to quantify ecosystem services, it is pivotal that quantification is presented in spatially explicit units, given that ecosystem service provision and demand are spatially explicit phenomena (Troy & Wilson, 2006). Furthermore, the supply and demand of ES can vary geographically (Fisher *et al*., 2009; Bastian *et al*., 2012). ES mapping is an approach which refers to the presentation of visualised spatial information about ES (Drakou *et al*., 2015). Mapping of ES has been undertaken at varying spatial scales, ranging from global (Naidoo *et al*., 2008; Wolff *et al*., 2017; Jiang *et al*., 2021) to local or city scale (Burkhard *et al*., 2012; Derkzen *et al*., 2015; Liu & Russo, 2021), and the body of literature which evaluates and maps UES has been growing steadily (Crossman *et al*., 2013; Malinga *et al*., 2015). The growth and scientific traction behind UES quantification and mapping in recent years has amounted to a large body of literature (see Liu & Russo, 2021; Derkzen *et al*., 2015; Holt *et al*., 2015; de Manuel *et al*., 2021; Ramyar *et al*., 2020; Zhang & Muñoz Ramírez, 2019; Basnou *et al*., 2020; Kremer, Hamstead and McPhearson, 2016; Haase *et al*., 2012). Regulating ES are the focus of the majority of the literature relating to mapping of ES, followed by provisioning services, then cultural services, while supporting services are the least prominent in the literature (Crossman *et al*., 2013; Malinga *et al*., 2015; Englund *et al*., 2017). Certain studies evaluate UES provided by differing types of UGS (Liu & Russo, 2021; Derkzen *et al*., 2015; de Manuel *et al*., 2021), whilst others assess UES more broadly as provided by the sum of all green space or green infrastructure (Ramyar *et al*., 2020).

The research, methods and approaches used to assess UES have grown and diversified since Bolund & Hunhammar's (1999) seminal article, addressing various elements of the complex, socio-ecological systems in which these UES are produced (Hubacek & Kronenberg, 2013). The approach of quantifying and mapping UES is heavily based on modelling and is rooted in an environmental science perspective, relating to various components of ecosystems and ecological assessments of ES (Hubacek & Kronenberg, 2013). Throughout the literature, authors have engaged in ES mapping to determine the correlation between ES and conservation goals (Egoh *et al*., 2009; Davids *et al*., 2016) as well as how the supply of these services vary spatially in response to land use changes (Lautenbach *et al*., 2011; Kertész *et al*., 2019; Kusi *et al*., 2020; Tolessa *et al*., 2017). Other research has addressed 'bundles' of multiple ecosystem services and how they interact through trade-offs and synergies (Haase *et al*., 2012; Derkzen *et al*., 2015; Holt *et al*., 2015; Liu & Russo, 2021). Further research has assessed the supply and demand of ecosystem services (Baró *et al*., 2015; González-García *et al*., 2020; de Manuel *et al*., 2021; Jiang *et al*., 2022), while others estimated the economic value of these services (e.g. Stoffberg *et al*., 2010). The methods used to quantify particular services are equally broad, with methods including the use of indicators, proxies, modelling, land-cover matrices, remote sensing, socio-cultural valuations, economic valuations, and bio-physical quantification (Burkhard and Maes, 2017).

A wide range of services have been quantified and mapped in recent years. In particular, services which are crucial to the urban context form a large proportion of the literature, including cooling services (Scholz *et al*., 2018; Wang *et al*., 2019; Rocha *et al*., 2022; Chen *et al*., 2022), runoff retention or flood mitigation (Barth & Döll, 2016; Li *et al*., 2019; Quagliolo *et al*., 2021; Mori *et al*., 2021) and carbon storage (Liu & Li, 2012; Agbelade & Onyekwelu, 2020; Lindén *et al*., 2020; Amoatey & Sulaiman, 2020). Although methods vary even within the same service, quantification of cooling services usually uses remote sensing or index methodologies (e.g. Schwarz *et al*., 2011; Kremer *et al*., 2016; Murtinová *et al*., 2022), while runoff retention is generally determined by hydrological modelling in conjunction with remote sensing (Jahan *et al*., 2021; Quagliolo *et al*., 2021; Wübbelmann *et al*., 2022). The quantification of carbon storage sometimes relies on secondary data from prior studies (Derkzen *et al*., 2015; Liu & Russo, 2021), but is generally quantified by determining biomass through field sampling using direct (measurements) and indirect (estimation) methods (Huynh *et al*., 2021).

Direct (destructive) sampling determines biomass by harvesting and directly weighing the trees or determining dry mass and density after exposing the trees to various temperatures (Jo *et al*., 2002; Pearson *et al*., 2005; Henry *et al*., 2010; Mangwale *et al*., 2017; Huynh *et al*., 2021), Although this method is most accurate it is expensive, time-consuming, labour intensive and ultimately releases CO_2 back into the atmosphere (Parresol, 1999; Henry *et al*., 2010; Nowak *et al*., 2013; Yao *et al*., 2015). Alternatively, indirect methods revolve around the estimation of a parameter which is difficult to measure (such as stem volume or tree weight) from a parameter which is relatively easy to capture in the field, such as DBH or tree height (Henry *et al*., 2010) and use statistically developed allometric equations (Brown 1997; Yoon *et al*., 2013; Chave *et al*., 2014; Mangwale *et al*., 2017). This approach is better suited to large scale sampling of diverse tree species and has been applied in a wide range of studies globally (Strohbach & Haase, 2012; Lindén *et al*., 2020; Pregitzer *et al*., 2022; Amoatey & Sulaiman, 2020; Stoffberg *et al*., 2010). However, this approach is associated with its own uncertainties, primarily stemming from the lack of equations derived specifically for urban vegetation, resulting in the use of models derived from natural forests (Yoon *et al*., 2013; Davies *et al*., 2013).

While research on urban ecosystem services have been growing in recent years, with a broad range of methodologies being developed and applied, there remain significant knowledge gaps in Africa (Wangai *et al*., 2016; du Toit *et al*., 2018; Veerkamp *et al*., 2021). This presents a major challenge, given that Africa is most vulnerable to the impacts of climatic change and has great need for ES supply.

2.3 African Vulnerabilities and Knowledge Gaps

The African continent is of particular importance in relation to ES, given the wide range of socio-economic and climatic vulnerabilities present on the continent, in conjunction with rapidly occurring urbanisation. Ecosystem degradation is a pertinent issue in Africa (AEO, 2013). Alongside this, water scarcity, poverty, food insecurity, housing insecurity and the prevalence of slums enhance Africa's socio-economic vulnerability (Bates *et al*., 2008; Frayne & McCordic, 2015; Ravallion *et al*., 2007; UN-Habitat, 2008; UNPF, 2007). It has also been acknowledged that the consequences of biodiversity loss and ecosystem degradation will be acutely and disproportionately experienced by the poor (Díaz *et al*., 2006). Africa stands as the least equipped region to deal with the impacts of climatic change (Velarde *et al*., 2005), exemplified by the history of extreme weather events having catastrophic consequences on the continent, both socially and economically (Galvin *et al*., 2001; Wangai *et al*., 2013). South Africa is no exception to this trend, as the South African people face many socio-economic challenges, driven by high poverty rates, lack of adequate housing, medical facilities and food security, alongside the greatest occurrence of HIV AIDS and tuberculosis in the world (Wright *et al*., 2021). Consequently, the South African people are generally vulnerable to the impacts of climatic change and have limited capacity to adapt to these changes.

As urbanisation continues to unfold on the global stage, the achievement of sustainable development is becoming increasingly dependent on the sustainable and successful management of urban growth in lower income regions, which will accommodate the most rapid urbanisation (United Nations, 2019). It is in this context of extreme and rapid urbanisation and population growth, in conjunction with already existing structural, socio-economic and climatic vulnerabilities, that the preservation of ES, particularly in urban systems, is an urgent priority in Africa. The demand for ES will increase dramatically alongside these population increases and increasing population density in urban settings; it is therefore evident that an adequate supply of ES needs to be created and sustained (Wangai *et al*., 2016). Despite the importance of ES in the African context, large knowledge gap exist around ecosystem service supply in cities, particularly in Africa (Wangai *et al*., 2016; du Toit *et al*., 2018; Veerkamp *et al*., 2021).

Although ecosystem services are understudied in Africa relative to the rest of the world, existing research on the continent is largely centred in South Africa (Wangai *et al*., 2016). Several studies have assessed ES in South Africa through economic (de Wit *et al*., 2012; Anderson *et al*., 2017; Turpie *et al*., 2017; Abd Elbasit *et al*., 2021) and non-economic perspectives (Egoh *et al*., 2008; Reyers *et al*., 2009; Egoh *et al*., 2009; Egoh *et al*., 2010). Others assessed the social perceptions of ES and how their use varied across socio-economic factors (Shackleton *et al*., 2015; Mensah *et al*., 2017) as well as the cultural values associated with them amongst traditional communities (Mowat & Rhodes, 2020). Some research has not directly quantified ecosystem services, but rather assessed their threats (Davids *et al*., 2018), identified hotspots (Davids *et al*., 2016) or overlap with biodiversity priority areas (Egoh *et al*., 2009). Several studies have assessed the ES produced by natural systems (e.g. Ntshane & Gambiza, 2016; Rebelo *et al*., 2019). While research on ES is growing in South Africa, urban ecosystem services remain understudied. Very few studies have explored UES in South Africa and the limited research that has, focuses on a single service (Stoffberg *et al*., 2010; O'Donoghue & Shackleton, 2013). Furthermore studies which have mapped urban ecosystem services are also limited. Studies which have mapped ES have generally been conducted at national or regional scales (e.g. Egoh *et al*., 2008; Turpie *et al*., 2017; SANBI, 2021). Furthermore, to my knowledge, little to no urban carbon storage research has been conducted in any South African city, although carbon sequestration has been quantified in the roadside and parking lot trees in the city of Tshwane (see Stoffberg *et al*., 2010) and several towns in the Eastern cape (see O'Donoghue & Shackleton, 2013). Ultimately, while research is growing in South Africa, urban ecosystem service remain understudied, their supply is rarely quantified and services in cities are rarely mapped. Furthermore, the City of Johannesburg has been largely neglected from ecosystem service research. These areas represent major knowledge gaps which this study aims to address by quantifying and mapping three contextually important ecosystem services in Johannesburg.

2.4 The Johannesburg case

The relevance and importance of different UES differs between cities, in accordance with the specific socio-economic and environmental characteristics of the city (Gómez-Baggethun & Barton, 2013). This relates to the context-specificity of ES and the social-ecological systems in which they are embedded (Costanza, 2008; Burkhard *et al*., 2012). Given the contextual nature of UES, the services investigated in this study were selected in accordance with their relevance to the socio-economic and environmental characteristics of Johannesburg and in alignment with the city's environmental priorities according to policy documents, such as the Johannesburg climate action plan (City of Johannesburg, 2021).

Carbon storage: It has been suggested that in the context of climate change mitigation, urban forests can play a role as carbon sinks (Stoffberg *et al*., 2010; Nowak *et al*., 2013). There is great potential for urban forests and trees to act as carbon sinks which store and sequester carbon (Nowak & Crane, 2002; Pataki *et al*., 2006; Nowak *et al*., 2013; Davies *et al*., 2011; McPherson, 2005). South Africa has a key role to play in curbing global CO_2 emissions, given their role as the 13th highest emitter worldwide and the largest emitter in Africa (Crippa *et al*., 2020). The estimation of carbon storage within the urban forest of Johannesburg is a crucial first step in assessing the potential urban forests can play in reducing atmospheric CO_2, thereby mitigating climatic change (Nowak *et al*., 2013; Liu & Li, 2012; Nowak & Crane, 2002). From a policy perspective, the climate action plan adopted by the city of Johannesburg aims to align with the targets of the Paris Agreement and has set an ambitious target of achieving net-zero emissions by the year 2050 (City of Johannesburg, 2021), highlighting the importance of carbon related services such as carbon storage.

Cooling and flood regulation: The City of Johannesburg has already experienced increases in temperature alongside severe rainfall variability, and average monthly temperatures are expected to increase under all climate change scenarios (City of Johannesburg, 2021).The city's climate action plan highlights the danger of rising temperatures in relation to human health and emphasises the need for additional cooling within the city, which may increase energy demands (City of Johannesburg, 2021). Projections of temperature increases, in combination with the UHI effect, are expected to create average temperature increases in the range of 2.2-2.7 ^{o}C by the year 2050 and consequently heat risk to human health as a result of increased temperature and the UHI effect has been highlighted as a key risk to the city (City of Johannesburg, 2021). This emphasises the importance of cooling ES. It is also expected that increases in rainfall intensity alongside the frequency of flood events during the summer months will become more prevalent by 2030 (City of Johannesburg, 2021). Thus flood reduction services are critical, particularly given the occurrence of extreme Highveld thunderstorms in Johannesburg, alongside the vast impervious urban surfaces, resulting in increasingly dangerous flash flood events (Schäffler & Swilling, 2013). The building of climate resilience and implementation of adaptation strategies have been highlighted as key goals in the City of Johannesburg (City of Johannesburg, 2021), with a central focus on flood and drought regulation strategies, creating resilient infrastructure and improving health and well-being across communities (City of Johannesburg, 2021).

The potential value of ecosystem services in Johannesburg is high, due to the city's large urban forest, connected greenspaces and remnant grassland vegetation. However, the city's vegetation cover is highly spatially variable, with large disparities in the distribution of tree cover between the northern and southern regions of the city (Schäffler & Swilling, 2013). The high potential value of ecosystem services in Johannesburg, alongside these disparities, heighten the necessity for the supply of ES to be quantified and mapped. This will identify UES inequalities and highlight vulnerable regions which need to be targeted by policy and decision makers. By identifying areas with low ES supply at the ward scale, decision makers will be better equipped to identify areas vulnerable to heat stress and flooding and can implement targeted solutions. This can aid planners in more effectively addressing Johannesburg's greenspace inequality and ensure equitable ES supply.

In addition to the city's climate action plan, the importance of preserving ES supply for building resilience to the impacts of climate change is further emphasized in Johannesburg's spatial development framework (City of Johannesburg, 2021b). This research can contribute to developing spatial planning instruments which are supported by spatially explicit ES supply information. South Africa's national development plan, further highlights the need for maintain ecosystem services as a climate change mitigation strategy (National planning commission, 2017), supporting the need for research on this topic.

3. Carbon Storage methods and results

For this study, a combination of remote sensing, modelling and field sampling methodologies was used to quantify and map three UES that were selected on the basis of their importance in the local context as well as their importance for climate change resilience and human well-being in the CoJ (see section 2.1). Temperature regulation (microclimatic cooling), runoff retention and carbon storage services were quantified and mapped according to the administrative wards of Johannesburg to assess the current supply of key ecosystem services by green infrastructure, identify supply deficiencies and create new knowledge for the design and planning of a climate resilient and sustainable city.

Deriving methodologies from literature to quantify and map UES is an approach which has previously been used in European and American studies (see Aevermann & Schmude, 2015; Derkzen *et al*., 2015; Kremer *et al*., 2016; Liu and Russo, 2021). However, these studies relied on existing primary data and prior studies in the region to derive values of UES provision. In South Africa, and Africa in general, there is a severe knowledge gap relating to UES (Haase *et al.*, 2014; Tavares *et al.*, 2019; Veerkamp *et al*., 2021). As a result, primary data and studies on UES in Johannesburg are largely unavailable. An extensive literature review was thus undertaken, and methods were assessed across multiple different research contexts to derive a suitable methodology to quantify and map UES in Johannesburg and overcome the paucity of data and prior studies. Methods relating to ES modelling, remote sensing techniques, field sampling as well as ES indicator methodologies were reviewed (See Whitford, 2001; Egoh *et al*., 2008; Ayanu *et al*., 2012; Crossman *et al*., 2013; Kanniah *et al*., 2014; Derkzen *et al*., 2015; Baró *et al*., 2015; Kremer *et al*., 2016; Basnou *et al*., 2020; de Manuel *et al*., 2021; Liu & Russo, 2021). Various methods were then adapted or combined to develop applicable approaches for Johannesburg, relative to the data available.

The overarching methodology used in this study, where applicable, involved quantifying the supply rate of selected UES per pixel (20x20 m) in Johannesburg in a GIS environment. This provides the overall supply of each service across the city and enables their supply to be mapped. Finally, the service supply values are normalised on the same scale, to enable comparison between services. This approach has been applied in the US and Malaysia to quantify and map UES (Kremer *et al*., 2016; Lourdes *et al*., 2022).

To do this, the study area was divided into several sub-units, in this case, administrative wards of Johannesburg , to make the results spatially explicit (as per Derkzen *et al*., 2015; Liu and Russo, 2021; de Manuel *et al.*, 2021; Hou *et al*., 2023)

3.1 Study Area

This study was conducted in the City of Johannesburg Metropolitan Municipality, commonly known to as the City of Johannesburg (CoJ) (Figure 1). The city is located on the eastern plateau of South Africa, known as the Highveld region, at an average altitude of 1700 m asl (Goldreich, 1992; City of Johannesburg, 2021). It is situated deep inland in the Gauteng Province of South Africa, the smallest but wealthiest province (City of Johannesburg, 2009). The CoJ is the country's largest and fastest growing city, expanding across an area of 1645 km^2 (City of Johannesburg, 2021). Despite the city's relatively small land surface area, Johannesburg remains South Africa's most densely urbanised and populated city. It hosts a rapidly growing population of 5.05 million inhabitants and a population density 40 times larger than the national average, with 2363.6 individuals per km^2 (City of Johannesburg, 2009; City of Johannesburg, 2009b; Hardy and Nel, 2015; City of Johannesburg, 2021).

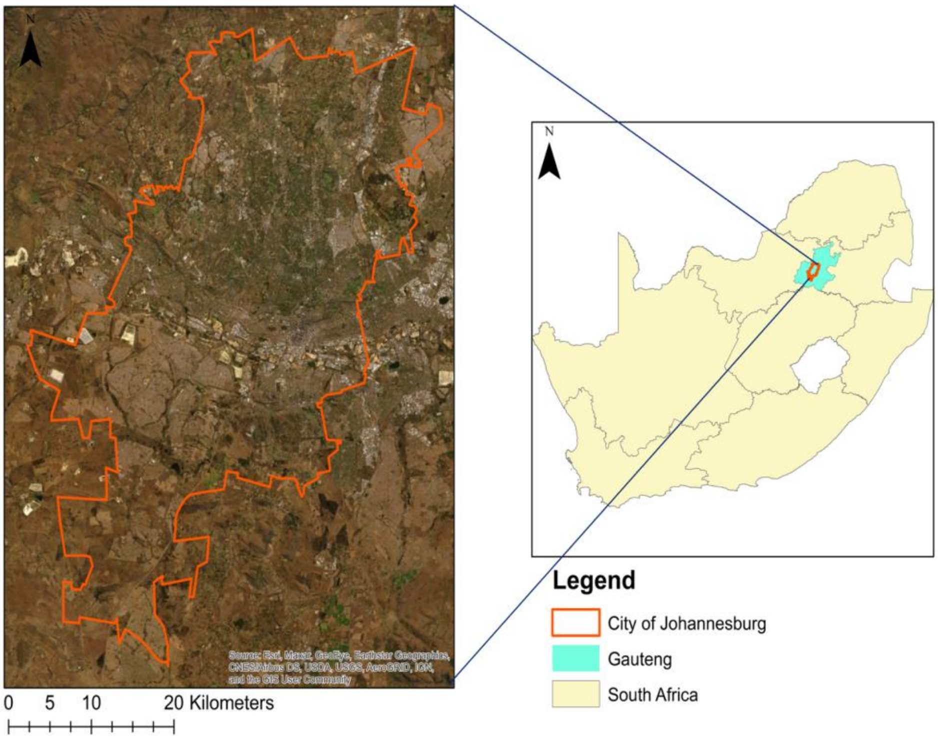

Figure 1- Study Area: The Greater metropolitan Johannesburg Municipality (City of Johannesburg), Gauteng Province, South Africa.

Johannesburg's climate is characterised as sub-tropical, having warm summers with rainfall and mild, sunny and dry winters (City of Johannesburg, 2021). Summer is characterised by intense tropical, convective rainfall events, which contribute the majority of the 800 mm annual precipitation (Goldreich, 1992; SAWS, 2021). The average temperatures during summer and winter are 25 ^{o}C and 18 ^{o}C, respectively (City of Johannesburg, 2021). The landscape of Johannesburg is characterised by dense urban areas in the central regions of the city, which are highly transformed and extensively covered by artificial, impervious surfaces, in which commercial, industrial, mining, and residential land uses are prominent (City of Johannesburg, 2009b). Moving north and south of the central regions, residential land uses, and associated activities become more prominent, with the city's suburbs being heavily vegetated by trees and bushes (City of Johannesburg, 2009b). Towards the northern and southern boundaries of the city the residential areas decline, with irrigated and natural grasslands being the dominant vegetation in these areas (City of Johannesburg, 2009b).

The city faces a range of environmental challenges, including flooding, heat stress and ecological degradation, which are expected to worsen due to climate change (City of Johannesburg, 2009b; City of Johannesburg, 2021). These issues are exacerbated by high rates of poverty (45.2%) and unemployment (31.5%), which increase pressure for further development and service delivery (City of Johannesburg, 2021). Simultaneously, Johannesburg contains a valuable and unique ecological asset, one of the largest urban forests in the world, estimated to include 10 million trees covering 16.1% of the city's area (Schäffler & Swilling, 2013; GCRO, 2013; Hardy and Nel, 2015) (Figure 2).

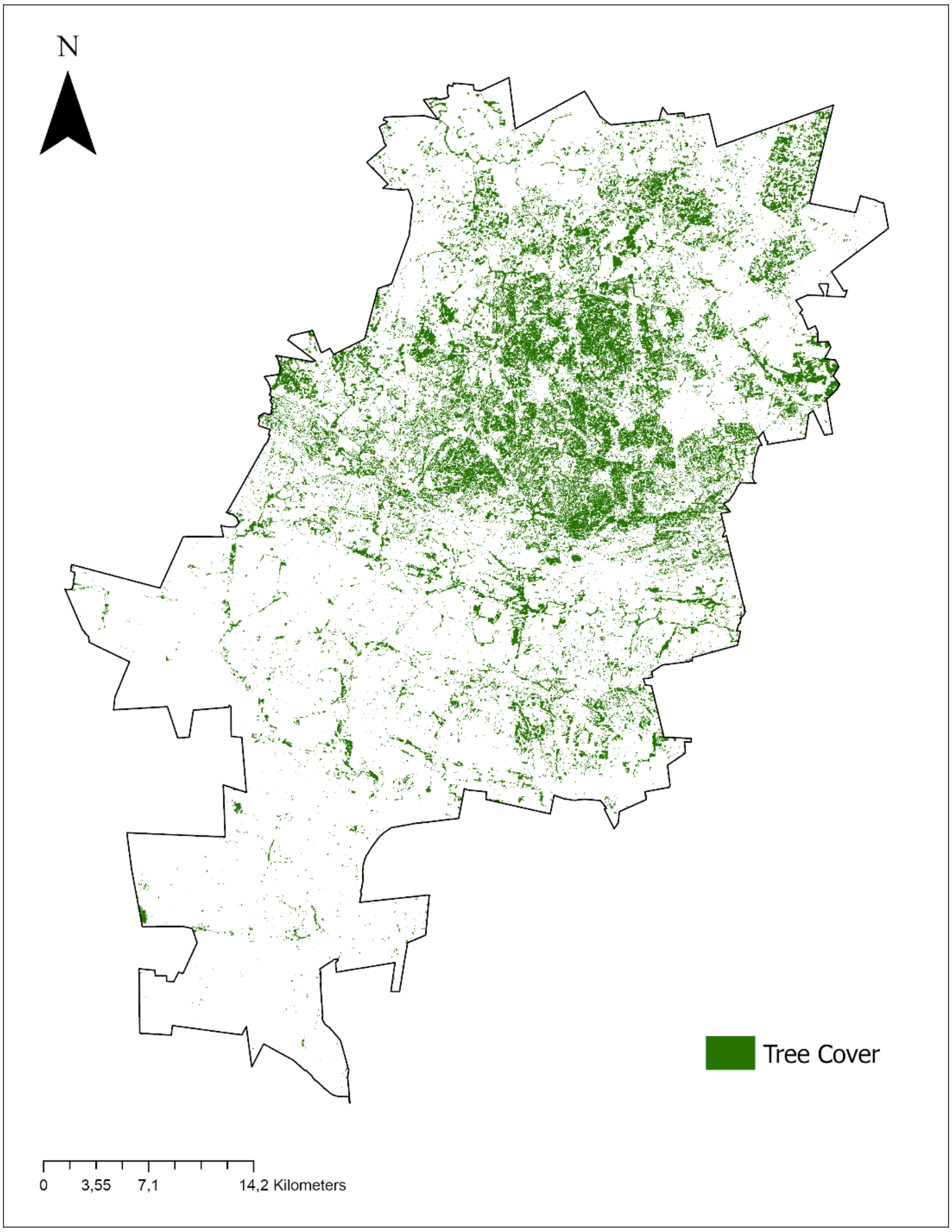

Figure 2: Tree cover distribution in the city of Johannesburg, South Africa. Source- ESA WorldCover 10m v200 (2021)

The CoJ lies in South Africa's the grassland biome, however its colonial and mining history has resulted in the introduction of a significant number of non-indigenous tree species, including Jacaranda, Eucalyptus, Oaks, and London planes (Turton *et al*., 2006). The city is largely characterised by artificial green areas, including the massive non-indigenous forest alongside sizable public parks and a multitude of private gardens (GCRO, 2013). Despite being highly transformed, the city retains remnant grassland and bushveld areas. Its natural vegetation reflects temperate inland grasslands alongside highveld grasslands (GCRO, 2013).

Due to the large urban forest and extensive, diverse green spaces within the city, the potential value of UES in Johannesburg is high. However, the city's vegetation cover is highly spatially variable, with large disparities in the distribution of tree cover between the northern and southern regions of the city (Figure 2; Schäffler & Swilling, 2013).

The CoJ has two administrative divisions, namely wards and regions. The city has 135 administrative wards and seven, broader administrative regions, named region A through F (Figure 3). The administrative wards were used for mapping ecosystem service supply in this study, while the region scale was used for broader spatial analysis.

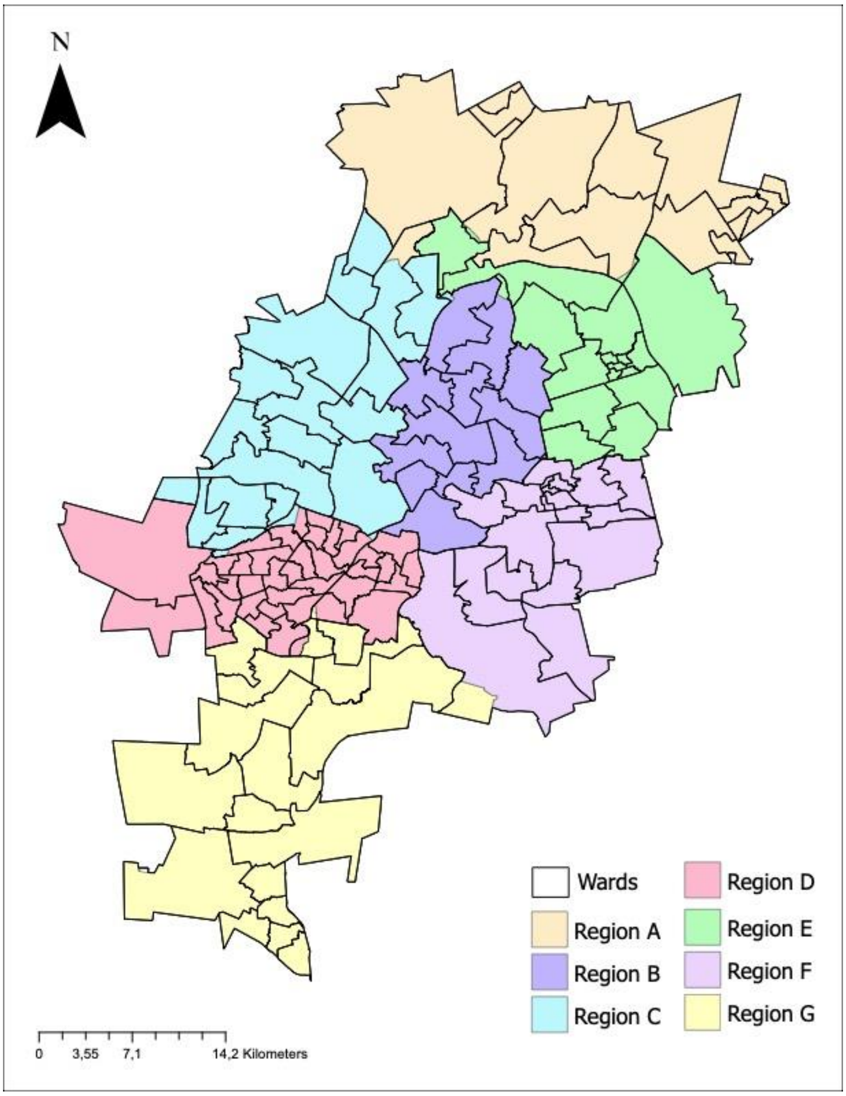

Figure 3: Boundaries of the administrative wards and regions of the City of Johannesburg, South Africa

3.2 Carbon storage methodology

For the purpose of this study, carbon storage is defined as the metric tonnes of aboveground carbon per hectare of urban green space (t/ha). The aboveground carbon (AGC) stocks are understood as the total amount of carbon contained within the aboveground biomass of vegetation which upon the death and decomposition of vegetation would be emitted back into the atmosphere (Reyers *et al.*, 2009; Nowak *et al*., 2013).

As in other South African ecosystem services studies (see Egoh *et al.*, 2008; Reyers *et al*., 2009; Egoh *et al*., 2010), carbon (C) storage rather than carbon sequestration is quantified and mapped due to data deficiency (Stoffberg *et al.*, 2010), and a lack of certainty regarding sequestration rates in the study area. Furthermore, the quantification of C sequestration necessitates large amounts of primary data on growth rates and population structure of trees in Johannesburg. These data are not currently available and would require long term, extensive field sampling beyond the scope of this study.

Prior empirical studies in South Africa found the aboveground carbon storage to be very low in indigenous grassland vegetation and to be highly related to the woodiness of that vegetation (See Mills *et al*., 2005). This is supported by international studies which found herbaceous and shrub vegetation to store a minute proportion of total aboveground carbon stocks, while the majority is stored in trees (See Davies *et al*., 2011; Nowak *et al*., 2013; Derkzen *et al*., 2015). In turn, indigenous grasses, non-woody and herbaceous vegetation are not considered in our carbon assessment due to their insignificant role in aboveground carbon storage. In line with international, urban carbon storage research (e.g. Raciti *et al*., 2014; Agbelade & Onyekwelu, 2020; Dhyani *et al*., 2021; Pregitzer *et al*., 2022), this study will thus assess aboveground carbon storage in trees and woody vegetation only.

3.2.1 Field Sampling

This study aimed to determine the AGC storage in trees in the urban forest of Johannesburg and subsequently map these carbon stocks. To achieve this, field sampling was conducted alongside a standard carbon stock methodology based on biomass equations (see Schäffler and Swilling, 2013). The study area consisted of the city's urban forest, defined as the sum total tree cover within the CoJ's boundaries, as depicted in Figure 2.

A simple, random, stratified sampling technique was employed to assess carbon stocks across the City of Johannesburg (CoJ). The CoJ's urban forest was stratified into four types of urban green space based on their function (management), structure and relative frequency. To select meaningful strata, the European Space Agency (ESA) tree cover layer was integrated with the South African National Land Cover (SANLC) map, and the land cover classes were sorted by their percentage tree cover and area (ha) of tree cover (Appendix A, Table A1). The land cover classes which contribute most to tree cover (by area and proportion) in the city were then chosen for sampling. The SANLC dataset contains 72 land cover classes, many of which contribute to tree cover in the CoJ, thus these classes were grouped at a higher level into four 'urban forest types'. Several SANLC classes were combined into single strata based on similar functions, structures, or compositions, for example 'Urban Recreational Fields' and 'Open and Sparse Planted Forest' SANLC classes were combined to form the 'park' stratum (see Appendix A Table A1). In addition, ground truthing was used to inform the categorisation of SANLC classes into broader strata, which represented the four dominant green space types in the city and were used for sampling.

Strata were differentiated by their composition and environmental conditions, thus were delineated as four dominant biotopes. The four strata included nature reserves, roadsides, private gardens, and parks, and are defined in Table 1, below. These four strata are referred to as urban forest types hereafter. While it is acknowledged that in isolation, they may not classify as 'forests' in of themselves (e.g. trees planted in intervals along a road), these four types contribute to Johannesburg's urban forest as a whole and serve as the primary constituents of the city's tree cover.

Table 1: Urban forest types used as stratification criterion for conducting carbon storage sampling in Johannesburg, South Africa

Urban forest type	Definition
Parks	Public and semi-private recreational, urban green spaces. Inclusive of public parks, golf courses and country clubs.
Nature Reserves	Municipal and private nature reserves or protected areas. Primarily consisting of remnant, natural grassland, and associated woodland.
Roadsides	Municipally planted roadside or pavement trees, occurring within 4 m of the roadside.
Gardens	Private, residential gardens within the boundary of the City of Johannesburg.

These four strata contribute more than 80% of the tree cover area in the city. Of the 10 million trees estimated to inhabit Johannesburg, approximately 4 million occur in private gardens, 1.5 million along roads and pavements and 2 million in parks and nature reserves (CoJ JCP, 2013; Schäffler & Swilling, 2013). The key urban green spaces which contribute to tree cover in the city which were not sampled include wetlands, smallholdings, and fallow lands (~11.25% of tree cover) (Appendix A, Table A1). All other land cover classes contribute only 2% of the city's tree covered area, based on ESA tree cover data.

Stratification was necessary to accurately quantify carbon storage in Johannesburg, due to the heterogeneity of the city's urban forest. Due to the immense size of CoJ's urban forest (Schäffler & Swilling, 2013; GCRO, 2013), tree cover occurs across a diverse range of land uses and biotopes. As in traditional forest sampling, where sampling is often stratified into forest zones of heterogenous communities or structure (e.g. Berliner, 2015), the urban forest was stratified into different forest types which have distinct communities, structure and/or physical environments. This allowed for different vegetation types, tree densities and land use types to be accounted for, ensuring the sample population accurately represents the entire population. Stratification of urban forest types is common practice in urban carbon stock assessments (Liu and Li, 2012; Amoatey & Sulaiman, 2020; Gülçin & van den Bosch, 2021).

After delineating the four dominant urban forest types in Johannesburg as strata, sample points were generated in each of them. The SANLC dataset did not contain a land cover which demarcates roadside trees, thus one was created manually and reclassified from existing SANLC land classes (Appendix A, Figure A1). To delineate roadside tree plantations across the city, two data sources were used: ESA Worldcover 10 m v200 and OpenStreetMap (OSM) roads layer. The ESA dataset was created by the European Space Agency in 2021, based on Sentinel 1 and Sentinel-2 imagery, with a resolution of 10 m. It consists of 11 LC classes, of which only tree cover was utilised in this study. The dataset was downloaded in Google Earth Engine and tree cover was extracted in Johannesburg from this raster. The OSM dataset consists of shapefiles representing roads, from which the Johannesburg area was extracted.

Both ESA and OSM Datasets were clipped to the study area (CoJ) and converted to polygons in ArcGIS Pro. The OSM dataset was used to create a 5 m buffer on either side of the roads, using the buffer tool in ArcGIS. The ESA tree cover layer was then clipped to the roadside buffer, extracting all areas in Johannesburg where tree cover occurs within 5 m of roads. These points were dissolved into a single layer and RS samples drawn from this dataset, using the create random points tool in ArcGIS Pro. This layer was later used for extrapolating roadside plantation carbon storage. The full GIS workflow, alongside zoomed in imagery of the process (datasets) are presented in Appendix A**.**

Nature reserves were also not demarcated by the SANLC Dataset, as such, polygons were manually drawn around the nature reserves in the city, from which samples were randomly drawn. Garden samples were drawn from the SANLC2020 class 'Formal residential (Tree)'. Park samples were drawn from NLC classes 'Dense Forest & Woodland', 'Contiguous & Dense Planted Forest', 'Open and Sparse Planted Forest', 'Temporary unplanted forest' as well as 'Urban recreational fields (Tree)'. All the constituent land cover classes were joined into a single layer from which samples were randomly drawn. See Appendix A for all constituent SANLC classes used for sampling each forest type.

Urban forest areas were extracted from the study area data using ArcGIS, after which stratified sampling was applied, using the urban forest type as the stratum. Sample plots were randomly selected in each of the four strata across the city. Each plot consisted of a 200 m^2 circle (radius = 8 m), with the centre point determined via the generation of random points in

ArcGIS. During field surveys, the centre point was located using GPS co-ordinates using Google Maps after which the plot was delineated with the aid of a laser range finder (SNDWAY 1000A Laser Rangefinder, 104x76.5x41mm). In cases where the randomly generated point was inaccessible due to physical terrain or other obstacles (e.g. dense thicket), the plot was shifted north by 50m or to the nearest accessible point thereafter. After delineating the plot boundaries, all trees with more than 50% of the stem falling within the plot were sampled. Trees with branches falling inside the plot, but stem falling outside the plot were excluded. All woody vegetation with a diameter of less than 2.5cm were excluded from sampling.

Due to the irregularity of garden sizes and shapes, a consistent plot size could not always be determined (Table 1). Instead, sampling occurred in the centre point of the largest, continuous garden area. An effort was made to create a plot as close to $200m^2$ as possible. For roadside sampling, due to the shape of the roads, a 50m transect was determined along both sides of the road. All stems falling within 4m of the roadside were sampled, creating a $200m^2$ plot, consistent with the other strata.

In total, 151 plots (with 1184 individual trees) were assessed. The number of plots for each stratum was determined by the area of the stratum, the variability within each forest type and the accessibility to sampling locations (Nowak, 2003). As in prior stratified carbon sampling studies, a minimum number of plots per stratum was implemented, frequently 10 plots per stratum (as per Nowak, 2003; Liu and Li, 2012). This study used a minimum of 20 plots per stratum. In the case of gardens, accessibility to sampling locations was limited by the necessity to obtain permission (from property owners) to enter private property to conduct the samples. Particularly in Johannesburg, where crime rates are significant, home-owners were hesitant to allow access to their property. As a result, fewer garden plots (21) were assessed relative to the large area of private gardens across the city. Parks and roadsides occur across a large area in the city, hence more plots were assessed (49 and 52 respectively). Nature reserves occur across a much smaller area in the CoJ, hence fewer plots were assessed (28). The distribution of plots and individuals across the strata is presented in Table 2 below.

Table 2: Number of plots assessed, number of individual trees, and mean area of sampled plots across four urban forest types in Johannesburg, South Africa

Urban forest type[a]	Number of plots	Number of individuals	Average area of plots (m^2)
Park	49	294	200
Nature reserve	28	456	200
Roadside	52	189	200
Garden	21	245	152
Total	**151**	**1184**	**188**

Field sampling was conducted between March-July 2023. For each of the 151 sample plots, tree species, diameter at breast height (DBH) in cm, stem length (m), total height (m) and branch percentage were recorded for each tree. The coordinates of each plots centre point were also recorded. Tree circumference was measured at 1.37m above the ground and divided by π to determine diameter at breast height (DBH). For slanted, forked or leaning trees and other exceptions, measurements were taken as per standard rules (see UNFCCC, 2015). For multi-stemmed trees, the DBH of the five largest stems were recorded and total DBH was calculated as the square root of the sum of the squares of each individual DBH. Dead trees were excluded from measurement. Stem length represented the height of the stem in meters (from ground to main branches) and was measured using measuring tape or a rangefinder where the trunk was too tall to measure by hand. The total height of the tree, from the ground to the highest branches, was recorded using a rangefinder (SNDWAY 1000A Laser Rangefinder, Dimensions: 104x76.5x41mm). Branch percentage was estimated as a proportion of the total volume of the tree. The collected field measurements were then used to determine the carbon storage in each tree measured.

Each tree was identified to species level using van Wyk & van Wyk (2013) and Glen & van Wyk (2016) as field guides. Where species level identification was not possible, the tree was identified to genus level. In cases of high uncertainty, physical samples were collected and analysed at the University of the Witwatersrand's H.M. Moss Herbarium, South Africa.

3.2.2 Carbon storage calculations

To quantify carbon storage, biomass was estimated by non-destructive (indirect) sampling methods (see Schäffler & Swilling, 2013). Although direct (destructive) methods most accurately determine biomass by harvesting and weighting trees (e.g. Jo, 2002; Pearson *et al*., 2005; Henry *et al.*, 2010; Mangwale *et al*., 2017; Huynh *et al*., 2021), they present several practical limitations. The trees sampled in this study were located on public roads, parks, protected areas and in private gardens. Hence, it was not feasible to conduct destructive sampling and a non-destructive method based on biomass equations and field data was used. Indirect methods are statistically developed allometric equations, in which physical parameters (independent variable) and biomass (dependent variable) are used to form a regression model (Brown 1997; Yoon *et al*., 2013; Chave *et al*., 2014; Mangwale *et al*., 2017) or physical parameters are used to determine tree volume using volumetric equations, which is converted to biomass using wood-specific densities (see Henry *et al*., 2011; Schäffler & Swilling, 2013; Chave *et al*., 2014; Dhyani *et al*., 2021). A general biomass equation was used due to the impracticality of species-specific equations in species rich forests (Smithwick, 2019), as well as the unavailability of species-specific equations for Johannesburg due distinct knowledge gap in the urban, South African context (Henry *et al*., 2010; Stoffberg *et al*., 2010).

On account of the lack of applicable biomass equations and the paucity of data relating to carbon storage in the urban context of Johannesburg, the methodology of Schäffler & Swilling (2013) was employed to estimate biomass. This general equation was chosen as it was derived specifically for the urban forest of Johannesburg, originally adapted from studies on the wild coast of South Africa (Geldenhuys & Berliner, 2010).

The relevant steps to estimating AGB (kg) used in this study, adapted from Schäffler & Swilling (2013), are presented below.

Basal area (BA) is calculated using measured DBH (cm):

$$BA = \frac{\pi * DBH^2}{40000} \qquad \textit{(Equation 1)}$$

Subsequently, the following equation is applied:

$$AGB = 0.7 * (BA * SL * 0.7)/(1 - B) \qquad \textit{(Equation 2)}$$

where *BA*= basal area, *SL*= stem length (m) and *B*= branch percentage (%)

Initially, the standard equation for basal area was used with DBH as the input (Eq. 1). Then, stem volume was calculated by multiplying basal area with stem length and the form factor of the average approximate stem shape (Eq. 2). Volume was then corrected (divided by 1- B%) to account for the branch and crown contribution to tree volume, and converted to biomass (kg) by multiplying by an average wood-density coefficient (0.7 g/m^3) for urban trees in Johannesburg (Eq. 2), as per (Schäffler & Swilling, 2013) . Multiplying the stem volume by specific wood-density is a common practice to convert tree volume to biomass (see Mchale, 2009; Henry *et al*., 2011; Dhyani *et al*., 2021). Average wood-density was used in this methodology due to the large uncertainties and variability associated with wood densities. This is particularly prominent in both urban and South African settings, where specific wood densities are largely untested. Furthermore, even within individual trees, wood density varies between different parts of the tree and can even vary within the trunk itself (see Henry *et al*., 2010).

Field measurements were entered into the equations and aboveground biomass was calculated for each tree sampled. Following prior South African carbon quantification studies (Mangwale, 2017; Smithwick, 2019), the mean carbon storage determined was compared with the means produced by internationally recognised equations, as well as two local equations. This enabled the comparison and validation of my results relative to local and international methods. The equations used for comparison are presented in Table 3.

Table 3: Biomass equations used for calculating biomass estimates from field measurements

Biomass equations	References
$AGB = exp\{-1.996 + 2.32 * In(DBH)\}$	Brown (1997)*
$AGB = 0.1464(DBH)^{2.3679}$	Ziannis (2008)
$AGB = 0.035(\mathrm{DBH})^{2.5}$	Netshiluvi and Scholes (2001)
$ABG = 0.127(DBH)^{2.335}$	Mangwale (2017)

AGB= above-ground biomass (kg), ln= Natural logarithm, *DBH*= Diameter at breast height (cm), *p*= specific wood-density (g/cm^3), *H*= tree height (m)
*From UNCFF (2009)

Netshiluvi & Scholes (2001) and Mangwale (2017) present equations derived for South African woodlands. Meanwhile, equations from Brown (1997) and Ziannis (2008) are widely used or standardised international, general biomass equations (see Amoatey & Sulaiman, 2020 ; O'Donoghue & Shackleton, 2013; Mangwale *et al*., 2017; Smithwick, 2019). These equations were selected in-line with prior carbon storage literature in South Africa (Mangwale, 2017; Smithwick, 2019) and due to their widespread use, which allows for effective sense-checking and comparison of our results to other standard methodologies. Due to the large uncertainties and knowledge gaps associated with wood-density in South Africa and urban settings (see Henry *et al*., 2010), standard allometries which require species-specific wood-density inputs such as (Chave *et al*., 2014) could not be implemented.

Urban trees are often heavily pruned and consequently have less ABG than is predicted by biomass equations, particularly those derived from natural forests (Nowak, 1994). As such, it is common practice to apply an urban biomass reduction factor of 0.8 (Liu & Li, 2012; Strohbach & Haase, 2012; Nowak *et al*., 2013; Dhyani *et al*., 2021), to reduce the biomass estimate by 20%, making it more accurate in the urban context. In this study, trees sampled in the urban environment where pruning was common (gardens, parks and roadsides), a 20% biomass reduction was applied. Trees in nature reserves are kept mostly in their natural state, thus no conversion was applied. After conversions, the AGB was summed for each plot. AGB was then converted to carbon by multiplying by 0.5 (as per Lindén *et al*., 2020 and Dhyani *et al*., 2021). Finally, the carbon density per plot (200m^2) was calculated and upscaled to produce metric tonnes per hectare (t/ha).

To determine the CO_2 equivalent of the carbon stock, the following formula was applied (as per Aguaron & McPherson, 2012; Chen, 2015):

$$CO_{2eq} = C * 3.67 \qquad \textit{(Equation 3)}$$

The mean carbon storage was then calculated for each urban forest type, across all sampled plots. The values produced by these calculations were then used for extrapolation and mapping of carbon stocks across Johannesburg.

3.2.3 Extrapolation and Mapping

Carbon storage values derived from field sampling (t/ha) were assigned to each LU class in the SANLC2020 dataset (20m resolution). This enabled carbon storage values to be extrapolated to the entire CoJ using ArcGIS. Each SANLC pixel is 400m^2 in size, thus values from the field were adjusted accordingly. The full list of carbon storage values assigned to each land cover as well as the detailed methodology, can be found in Appendix D.

Prior to extrapolation, several adjustments were made to the SANLC dataset as does not contain a layer for tree cover, and the amount of tree cover varies between LU classes. In order to spatially extrapolate to tree cover across the city, additional datasets were used. The ESA WorldCover LU dataset (10m resolution) was used to extract the tree covered area within each SANLC LU. This enhances the accuracy of the extrapolation, by ensuring that carbon storage is only assigned to pixels covered by trees. Three additional LU classes were then added to the SANLC dataset through reclassification, to enhance the accuracy of the extrapolation. One SANLC layer was reclassified into three classes and a new class was added. A roadside tree LU was created by reclassifying the SANLC data where tree cover overlaps with roads or pavement (See Appendix A: Figure A1).

Another key issue which prevented accurate extrapolation was the overlap of planted forest with natural forest SANLC LU classes in Johannesburg. As the SANLC is a national land cover map, covering the extensive natural landscapes of South Africa, its accuracy in urban settings is reduced. Parks and other artificial green systems are frequently detected as natural or indigenous vegetation land covers. As such, adjustments were made to enable better distinction of constructed and natural green spaces in the city. Full details and GIS methodology behind the reclassification of the SANLC dataset, as well as the data sources used are also available in Appendix D.

Outside of the four key strata which were sampled in the field (which compromise the majority of the tree cover and urban green spaces in Johannesburg), the remaining land covers which contribute to tree cover (agricultural, fallow fields, small holdings) were assigned a generic value. This value was determined by averaging the carbon storage of the four sampled strata. Waterbodies were assumed to have 0 C/ha ABG, thus aquatic vegetation was excluded. However, tree cover in and around rivers and wetlands was accounted for. Carbon storage in grass and turf grass was not accounted for. Field values were assigned to each SANLC LU class in an attribute table and these values were extrapolated to the city scale using the look-up tool in ArcGIS Pro. The extrapolation produced a 30m resolution carbon storage map of the entire CoJ. This map was sub-divided using the administrative wards of the city. The carbon storage map was integrated with the CoJ ward boundaries and the total carbon storage (per ward) and mean carbon storage (per pixel) were calculated. The results were then aggregated to the ward level and mapped accordingly. Additionally, the same procedure was followed to calculate the carbon storage for each administrative region in Johannesburg.

Due to the large variability in the sizes of wards in Johannesburg, total carbon storage per ward was normalised by area (ha) of that ward. This enabled the wards to be accurately compared in terms of their carbon storage performance, per unit area, and minimised differences in values based on size.

3.2.4 Statistical Analysis

To test for differences in biomass, carbon storage, height and DBH between the four urban forest types, a non-parametric Kruskal-Wallis test was used, as the data were not normally distributed, even after transformations. Dunn's post-hoc test was used to determine where, if any, significant differences occurred.The median was used as the measure of central tendency, due to the data being non-parametric in distribution. Differences in tree density between forest types were assessed using a one-way ANOVA, and Tukey's HSD post-hoc test. Relative frequency distributions were constructed for each urban forest type to assess the distribution of DBH and height within and across types. Finally, a chi-square test was used to test for differences in the proportions of exotic vs indigenous species across four urban forest types. All statistical analyses were undertaken using the Statistica V14.0 (StatSoft, Inc, 2020) alongside Prism 9 for macOS V9.5.1 (GraphPad Software, Inc) and were run at a significance level of 0.05.

3.3. Carbon Storage Results

This study aimed to quantify and map the aboveground carbon stocks in Johannesburg. Field sampling was done to assess the structure and composition of Johannesburg's urban forest across four sampled forest types, and a biomass equation was used to evaluate their capacity to store carbon in AGB. Finally, a GIS assessment was done to extrapolate findings from the field to the entire city, to determine the spatial distribution of carbon storage across the CoJ and determine the city's total carbon stock. The results show that Johannesburg's green spaces store 39.92-97.50 t/ha of carbon and that there are large spatial disparities in the distribution of carbon storage across the city.

3.3.1 Urban Forest Structure

A total of 151 plots were sampled and 1186 individual trees were measured across the four urban forest (UF) types. The distribution of sampled plots and individual trees across strata is presented in Table 4. The mean (SD) number of stems per plot was 7.85 (7.08) across all urban forest types. Density differed significantly between urban forest types ($F_3 = 41.34$, $p < 0.001$). Tukey's post-hoc revealed that tree density in nature reserves (810 trees/ha) was significantly higher than other forest types ($p<0.05$), as individuals were located in dense, natural woodlands, with many small individuals growing in proximity. Meanwhile, roadsides and parks were found to have significantly lower density (177 and 298 trees/ha, respectively) than both parks and gardens ($p<0.001$) but did not differ from each other ($p=0.096$). In seven plots, no trees were found, six of which were along roadsides.

Table 4: Number of plots (200 m^2) assessed, number of individual trees, mean tree density per plot and per ha, across four urban tree types sampled in Johannesburg, South Africa. RS- roadside, NR- nature reserve

	No. Plots	No. individuals	Tree Density (/Plot)	Tree Density (/Ha)
Park	49	295	6	298.50
RS	53	189	3.566	177.41
Garden	21	246	11.71	582.80
NR	28	456	16.28	810.23

Note: Plot = 200m^2 (8m radius circle)

Across the 1184 sampled trees in Johannesburg, the mean height was 8.85 m ± 0.17. The minimum recorded height was 1.2 m, while the maximum was 42 m. The median tree height across all samples was 6.91 m. Parks contained the tallest trees, with a mean (±SE) height of 13.16 m ± 0.411, followed by roadsides at 11.34 m ± 0.323. Nature reserves had the shortest trees, with a mean height of 6.21 m ±0.18, while gardens represented the second shortest class, with a mean of 6.64 m ±0.25.

However, the mean height of parks is slightly skewed by extremely large values, as most extremely tall trees occurred in parks. As a result, median tree height (95% CI) was highest in roadsides at 11.51 m (10.81-12.01), followed by Parks at 10.96 m (10.01-12.00) (Figure 4).

Tree height differed significantly between UF types (H=382.4, df=4, p<0.001), with both park and roadside individuals being significantly taller than nature reserve and Garden individuals (p<0.001).

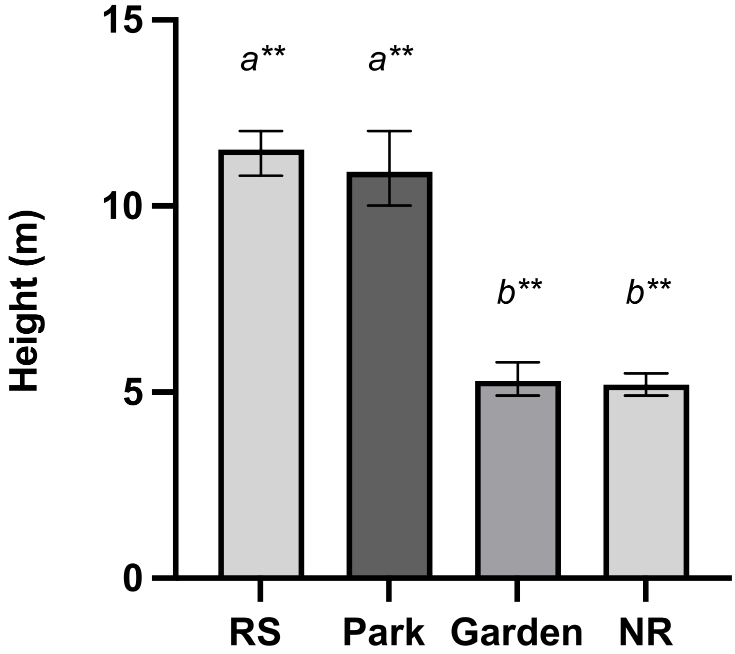

Figure 4: Median tree height ± 95% CI in four urban greenspace types across sampled locations in Johannesburg, South Africa. Medians denoted by different letters indicate significant differences between groups. RS- Roadside plantation, NR- Nature reserve, **- P value <0.001 with the Kruskal-Wallis test.

Across all sampled trees, the majority were between 4-10 m in height (58.36% of sampled individuals). Parks were dominated by individuals between 6-12 m (44.56%), but about half the population was >12 m in height, with 22.79% being taller than 20 m (Figure 5). Roadsides were dominated by taller trees, with 56.61% between 10-16 m in height. However, roadsides comprised of much fewer very tall trees (>20 m, 5.29%) when compared to parks (Figure 5). Gardens were dominated by shorter trees, with most sampled individuals between 3-7 m in height (57.96%). Gardens had fewer tall trees and even fewer very tall trees compared to parks and roadsides (Figure 5). Nature reserves were also dominated by short trees, with 75% of the sampled individuals between 0-6m in height and only 12.9 % >10 m and 1.97%>20 m.

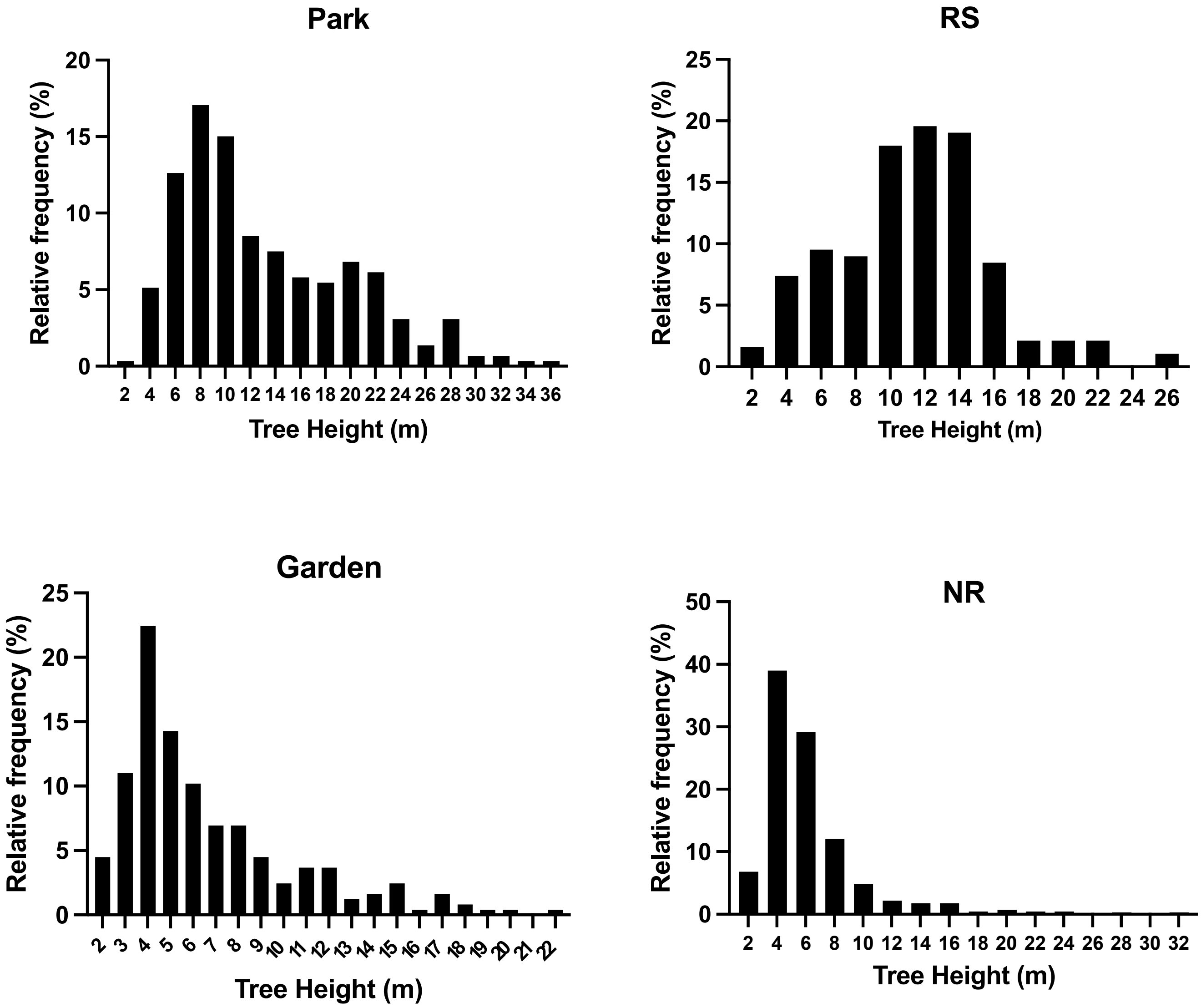

Figure 5: Tree height relative frequency distribution across sampled individuals in four urban forest types in Johannesburg, South Africa. NR= Nature Reserve, RS= Roadside. One value removed from park dataset to reduce axis (43.01 m).

The median DBH differed significantly between UF types ($H=515.2$, $df=3$, $p<0.001$). The thickest trees were found along roadsides with an average (±SE) DBH of 46.94cm ±1.58, while park individuals were the second largest (40.73 cm ±1.21). Nature reserve and garden trees were much smaller, with an average DBH of 13.64 cm ±0.55 and 18.78 cm ±1.12, respectively. Recorded DBH ranged from 2.7cm-123.5 cm, with an average of 26.75 ±0.64 across all sampled individuals. The median DBH was 19.15 cm across all samples. The largest stem sampled belonged to a *Platanus x acerifolia* standing in a roadside plot, with diameter at breast height of 123.5 cm. The post-hoc test revealed that both trees in roadside and park plots were significantly larger than garden and nature reserve plots (p<0.001), while garden trees were larger than nature reserve trees (p<0.005).

Nature reserve and garden trees were generally smaller, with the majority between 0-30 cm (Figure 6). These two UF types made up most samples between 0-20 cm size classes. Meanwhile, RS and Parks contained much larger trees, with the majority between 40-70cm. RS and Park trees accounted for almost all very large trees with DBH >60 cm (Figure 6). RS and Park individuals were more evenly distributed between 20-80 cm size classes, while Garden and Park trees dominated by 0-30 cm trees (Figure 6).

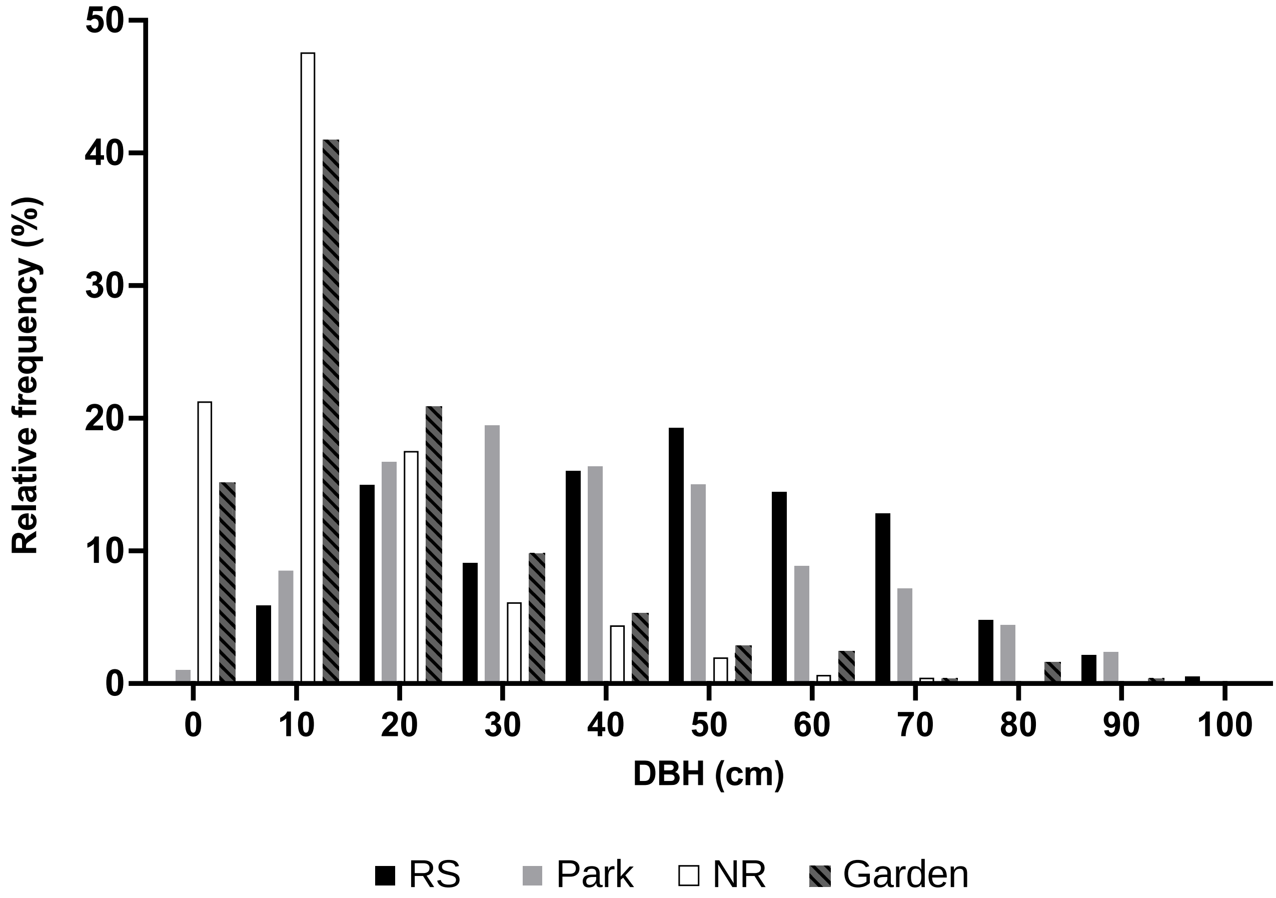

Figure 6: Tree diameter at breast height distribution across different urban forest types in Johannesburg, South Africa. NR= Nature Reserve, RS= Roadside

Roadside trees were evenly distributed between 20-80 cm DBH, with the majority occurring between 40-70cm (49.2%). Very large trees (DBH>50 cm) were very common on roadsides, making up 54.49% of the population. Park samples were relatively evenly distributed amongst 20-60 cm size classes, with 30-40 cm size classes being most common (19.39%). Large trees were also prevalent in parks, with 38.09% of the sampled population larger than 50 cm. Nature reserved were dominated by very small trees, with 39.7% of sampled individuals less than 10 cm in size and 74.4% below 20cm. Nature reserve samples were centred in the 5-10 cm size class, with the samples in each subsequent size class reducing evenly. Garden trees were slightly larger than NR on average and were dominated by the 10-20 cm size class (40.8%). Both NR and Garden had very few large trees, with only 2.41% and 8.16% of the sample population >50 cm DBH, respectively.

3.3.2 Species composition

Forest composition was assessed for all four UF types. Of the 1186 individuals sampled, 97.8% were identified to species level and 99.86% to genus level. 18 individuals (1.43%) could not be identified. In total, 164 species and 116 genera of tree and shrub were identified across all four strata. Garden samples contributed the most to this high diversity, with 62 genera and 80 species sampled amongst only 246 individuals (Table 5). Meanwhile, Parks had the lowest number of genera (31), but more species than roadsides and nature reserves. Nature reserves represented the second highest number of taxa (37) but had by far the most individuals sampled (Table 5). Relative to the fewer individuals sampled (189), Roadside trees still maintained high diversity, with 34 taxa represented. Overall, all four strata showed high diversity with (>31) taxa represented (Table 5). The full genus and species composition for each forest type is presented in Appendix **G**.

Table 5: Number of individuals and species diversity of sampled trees across four urban forest types in Johannesburg, South Africa.

	No. individuals	No. Taxa	No. Species	Unidentified/ Genus level*
Park	295	31	57	0/10
RS	189	34	37	1/1
NR	456	37	51	4/4
Garden	246	62	80	4/3

The diversity of each urban forest type was fairly unique, with only 45 species occurring in more than one forest type, and only three species (*Celtis africana, Celtis sinensis, Searsia lancea*) occurring in all four urban forest types. The dominant species also varied according to forest type (Table 6). *Pinus spp* (15.99%), *Celtis spp*. (12.93%), *Quercus* spp. (9.86%) and *Eucalyptus* spp. (7.48%) were the most common taxa in Parks (Table 6). In roadside plots, *Jacaranda mimosifolia* (32.28%), *Platanus x acerifolia* (12.17%) and *Brachychiton* spp. (7.94%) were dominant.

Gardens were more diverse, resulting in less dominance and a more equal distribution of species (Table 6). However, *Viburnum* spp. (7.35%), *Prunus* spp. (6.94%) and *Celtis* spp. (6.53%) were the most represented taxa in gardens in Johannesburg. Nature reserves were dominated by *Celtis* spp. (13.22%), *Senegalia* spp. (11.23%), *Eucalyptus* spp. (10.13%) and *Vachellia* spp. (9.91%). Most of the Eucalyptus trees were found in private reserves, while municipal reserves were dominated by exclusively indigenous vegetation.

Overall, roadside plots had the highest dominance of individual genera, with the three most common genera making up 52.9% of the sampled population. Meanwhile, gardens had the lowest dominance, with the three most common taxa making up only 20.8% of the samples.

Table 6: Top five most commonly occurring tree genera across four sampled urban forest types, in Johannesburg, South Africa.

Strata	Genus	Relative frequency (%)	Observations
Park	*Pinus*	15.99	47
	Celtis	12.93	38
	Quercus	9.86	29
	Eucalyptus	7.483	22
	Casuarina	6.46	19
Roadsides	*Jacaranda*	32.28	61
	Platanus	12.17	23
	Brachychiton	8.47	16
	Searsia	5.82	11
	Celtis	5.82	11
Nature Reserves	*Celtis*	13.22	60
	Senegalia	11.23	51
	Eucalyptus	10.13	46
	Vachellia	9.91	45
	Searsia	8.37	38
Garden	*Viburnum*	7.35	18
	Prunus	6.94	17
	Celtis	6.53	16
	Vachellia	6.12	15
	Syzygium	5.31	13

The proportion of exotic and alien species differed significantly between forest types (X^2=739, df= 3, p<0.001). Exotic or introduced species formed the majority of sampled individuals in Parks (84.4%), RS (82.5%) and Gardens (76.3%). Nature reserves were dominated by indigenous species (85.7%) with a much lower proportion of exotic species (13.2%) relative to the other strata (Figure 7).

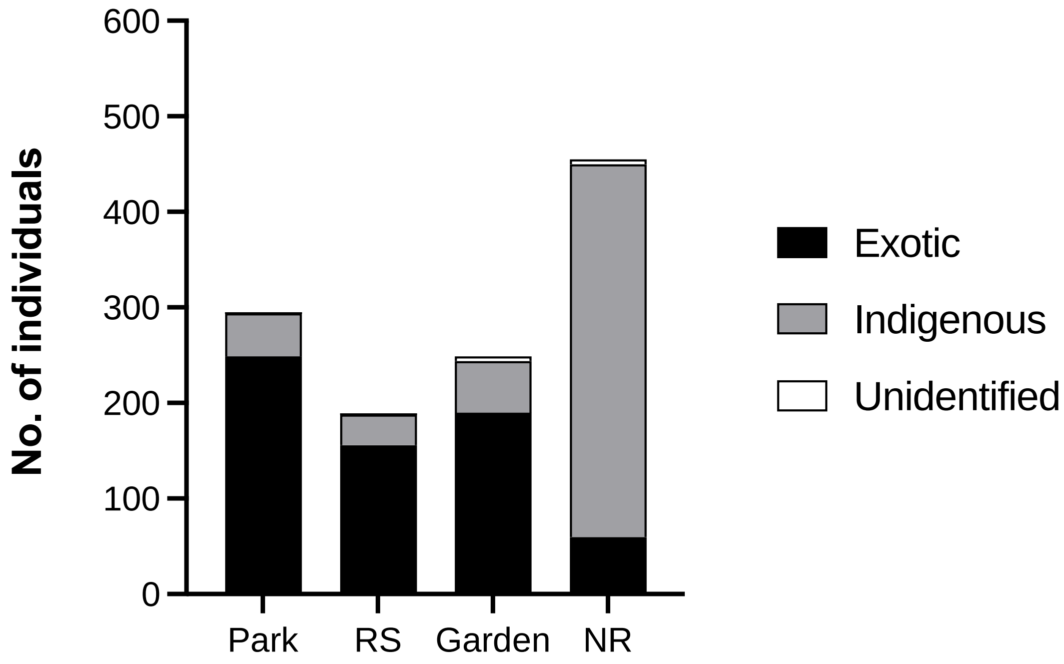

Figure 7: Exotic vs indigenous species composition across four urban forest types in Johannesburg, South Africa. NR- Nature reserve, RS- Roadside plantation.

3.3.3 Urban Forest Function

The function of Johannesburg's urban forest was then assessed, through its capacity to store above-ground biomass (AGB) and carbon.

In order to quantify carbon storage, the AGB was assessed for each sampled tree. The mean (±SE) biomass per tree across all samples was 437.2 kg ±24.12. This value was brought up by a small proportion of the population with extremely large biomass, hence the median biomass is much lower, at 70.38 kg. Minimum biomass recorded was 0.29 kg, while the maximum was 7493 kg. Most sampled individuals were between 0-500 kg (67.85%), while 13.62% were between 500-1000 kg and 7.53% between 1000-1500 kg. Only 2.79% of samples were found to have a biomass greater than 3000 kg.

As the data were not normally distributed, a Kruskal-Wallis test was run which revealed that biomass of trees differed significantly between urban forest types H= 474, df= 3, $p<0.001$), with RS and Park being significantly larger than the other UF types ($p<0.001$) and gardens being larger than NR ($p<0.05$).Amongst the four urban forest types, the largest mean biomass was found in RS trees (947.1 kg ±72.54) , followed closely by parks (816 kg ±60.44), while the lowest was found in NR trees (99.28 kg ±12.89) (Table 7). Thus, the heaviest trees with the most aboveground biomass were in RS and Park UF types, while garden and NR trees were much smaller on average. The median biomass in each UF type was much smaller than the mean, due to the presence of extremely high values (>5000 kg) in each UF type except NR (table 7).

Table 7: Descriptive statistics of Aboveground Biomass (Kg) per tree, across four sampled UGS types in Johannesburg, South Africa.

	Individuals (n)	Mean	Std Error	Min	Max	Median
Park	187	816	60.44	0.85	5699	383.9
RS	294	947.10	72.54	3.56	5851	646.1
Garden	245	222.60	45.92	0.29	7493	26.71
NR	456	99.28	12.89	0.34	3320	15.09

For garden and NR samples, the majority of individuals had a biomass between 0-500 kg (87.35% and 91.45% respectively). Meanwhile, RS and Park trees were much heavier, with a greater proportion of individuals >1000kg (47.06% and 34.01%, respectively) compared to Gardens and NR (7.34% and 2.85% respectively).

Carbon storage was quantitatively compared between UF types and spatially assessed across the wards and regions of Johannesburg. The urban forest of Johannesburg stores an estimated 2 406 377 tonnes of carbon (tC), equivalent to 8 832 84.59 tCO_2 equivalent. On average 14.65 t/ha are stored across the entire city. Carbon storage per ha of tree cover varies between UF types.

Carbon storage differed significantly between UF types (H= 14.34, df= 3, p<0.005), with parks and RS storing more carbon than NR (p<0.05). The data show that Parks in Johannesburg store 97.50 t/ha (SE: 13.68), Roadside plantations store 67.84 t/ha (SE: 8.59), Gardens store 65.36 t/ha (SE: 16.06), while nature reserves store 39.92 t/ha (SE: 8.01). Thus, of the four UF types, Parks stored the most Carbon, RS store the second most followed closely by RS while NR stores the least amount of carbon in AGB (Table 8).

Table 8: Mean Aboveground Biomass, Carbon storage and CO_2 equivalent (tonnes/ha) across four sampled UGS types in Johannesburg, South Africa

UGS Type	Plots (n)	ABG (t/ha)	ABC (t/ha)	CO_2 eq. (t/ha)
Park	49	194.99	97.50	357.83
RS	53	135.68	67.84	248.97
Garden	21	130.72	65.36	239.87
NR	28	79.84	39.92	146.51

In comparing the biomass equation used in this study to alternative local and general equations, it was found that Garden carbon storage was 65.36 t/ha in this study, 60.5-91.63 t/ha using local equations and 92.46-119.83 t/ha using international equations (Table 9). Meanwhile, alternative equations produced values ranging from 26.01-68.28 t/ha for NR, compared to 39.92 t/ha in this study. The equation used in this study, developed specifically for the Johannesburg urban forest, produces a more conservative estimate of C storage than the alternative equations, with the exception of Netshiluvi & Scholes (2001) (Table 9). The estimates produced by this study are within a similar range of other local equations (Netshiluvi & Scholes, 2001; Mangwale, 2017) suggesting a reasonable accuracy. Of the general and international equations, Brown (1997), shows the greatest similarity to the estimates produced by this study's methodology (Table 9).

Table 9: Comparison of aboveground carbon storage in this study to values derived from four alternative allometric models.

	Mean Carbon storage (tons/ha)			
	Garden	**Park**	**RS**	**NR**
This study	65.36	97.50	67.84	39.92
Netshiluvi & Scholes (2001)	60.50	101.79	53.23	26.01
Mangwale (2017)	91.63	124.42	97.59	52.78
Brown (1997)	92.46	125.23	97.61	53.58
Ziannis (2008)	119.83	167.59	128.35	68.28

Carbon storage amongst sampled trees was largely driven by structural traits, in which taller and larger trees stored disproportionately more carbon than shorter and smaller trees. Trees taller than 10m (n=374) contributed 86.36% of the total C and biomass amongst all the sampled trees, while making up 31.64% of the sampled population. Meanwhile, trees >15m (n=162) represented 13.7% of the sample population but contributed 51.4% of the total C storage.

A similar yet more pronounced trend occurred in the relationship between DBH and C storage. Most of the carbon stored within sampled trees in Johannesburg occurs in large trees with high DBH. Trees with DBH>50cm (n=199) stored 72.2% of the carbon stored across all sampled trees, despite making up only 16.8% of the samples. Similarly, trees with a DBH of greater than 60 (n=116) stored 53.67% of the carbon, while constituting less than 10% of the total trees sampled. Furthermore, the 28 largest individuals, 2.37% of the sampled trees, contributed 17.38% of the sampled C stock.

Beyond structural traits, the C stock of sampled trees in Johannesburg varied greatly between alien and indigenous species (Figure 8). Alien or introduced species contributed most of the biomass in sampled trees across Parks (94.8%), Roadside plantations (93.4%) and gardens (84.5%). However, the biomass was more evenly distributed between native (50.2%) and alien (49.8%) species in Nature reserves. Ultimately, across all four strata, alien species contributed 705.1kg per tree on average, while indigenous species were 105.74 kg on average.

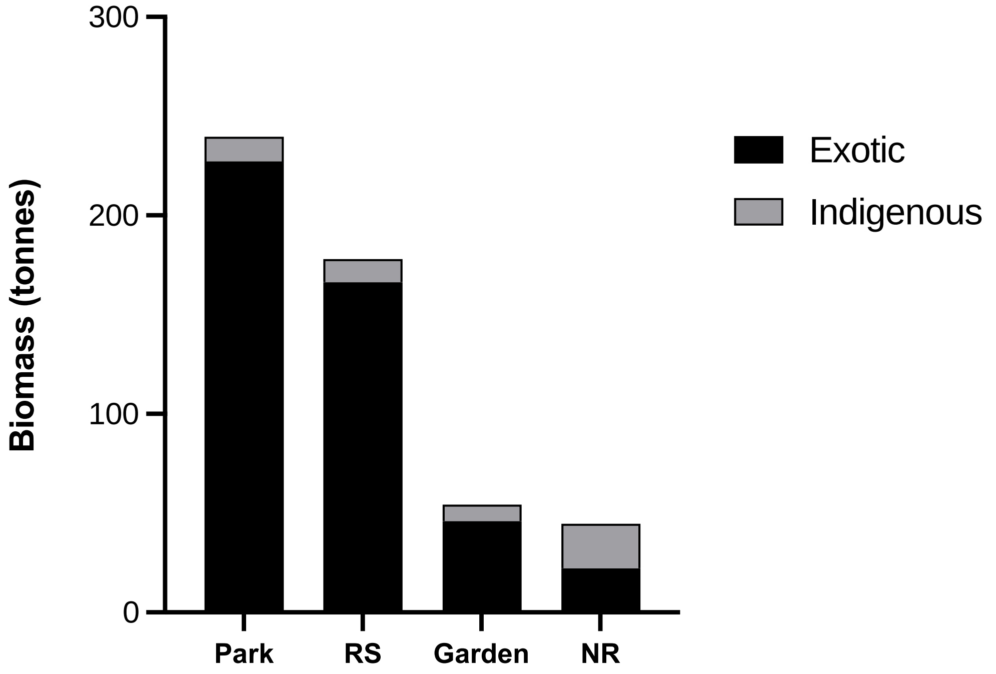

Figure 8: Biomass contribution of indigenous vs exotic species, sampled across four urban forest types in Johannesburg, South Africa.

Prior to assessing carbon storage at the fine ward scale, the carbon stock was aggregated to the broader, administrative region level (Table 10). The results revealed great spatial variation in C storage between the 7 administrative regions of Johannesburg. Regions E and A contribute the largest portion of C storage in the city, while regions B, C and F contribute considerably (Table 10). Regions D and G contribute far less C storage relative to their area while region B overperforms its area in terms of C storage. Regions D and G have a large discrepancy between their relative area and relative C storage. Ultimately regions B and E store the most carbon per ha, 26.92 and 25.19 t/ha respectively. Meanwhile, regions D and G store by far the least carbon per ha (4.21 and 5.20 t/ha respectively). Thus, the Northern central and North-eastern regions perform best in terms of C storage, while the Southern and South-western regions perform worst.

Table 10: Carbon storage total, proportion, and density across the seven administrative regions of Johannesburg, South Africa.

Region	Area (ha)	C Storage (tonnes)	Area (%)	C storage (%)	C Density (t/ha)
A	31349.88	508212.02	19.09	21.12	16.21
B	14910.84	401395.98	9.08	16.68	26.92
C	25376.88	392357.48	15.45	16.31	15.46
D	18664.64	78615.57	11.36	3.27	4.21
E	19859.36	500216.43	12.09	20.79	25.19
F	21928.16	358216.69	13.35	14.89	16.34
G	32151.72	167304.45	19.58	6.95	5.20

Finally, carbon storage is assessed spatially, across the wards of Johannesburg (Figure 9). Most of the Northern half of Johannesburg was found to have maximal or high carbon densities, while carbon storage in the South was lower and more variable (Figure 9). The southwestern region has the lowest carbon density, with many wards having the minimum supply (<3.71 t/ha), while the southernmost and westernmost wards also show low supply. In the Northern regions the small group of wards in the northeast (Alexandra) stand out as having minimal carbon storage, despite being surrounding by high carbon storage wards on all sides (Figure 9). The above-ground carbon storage distribution across Johannesburg largely mirrors the tree cover distribution in the city (see Figure 2). The highest carbon density is found in the northern-central region of the city, where most of the urban forest occurs. Outside of this area, the north-eastern ward is the only other ward with maximal carbon storage density (Figure 9). Meanwhile, the Southern half of the city is largely void of tree cover, except for the south-eastern wards, which have significant tree cover and in turn higher carbon storage than the surrounding wards (>15.21 t/ha).

Ultimately, of the 2.4 million tonnes of carbon stored by Johannesburg's urban forest, most of it is stored in Northern Johannesburg. The distribution of carbon storage across the city is highly variable and unequal, with clear discrepancies between and within the Northern and Southern regions (Figure 9). Forty three wards in the city have minimal carbon density (<3.71 t/ha). Of these, the majority are situated in the south of Johannesburg, with wards in the south-west, southernmost and northeast (Alexandra) standing out as critically low C storage supply areas.

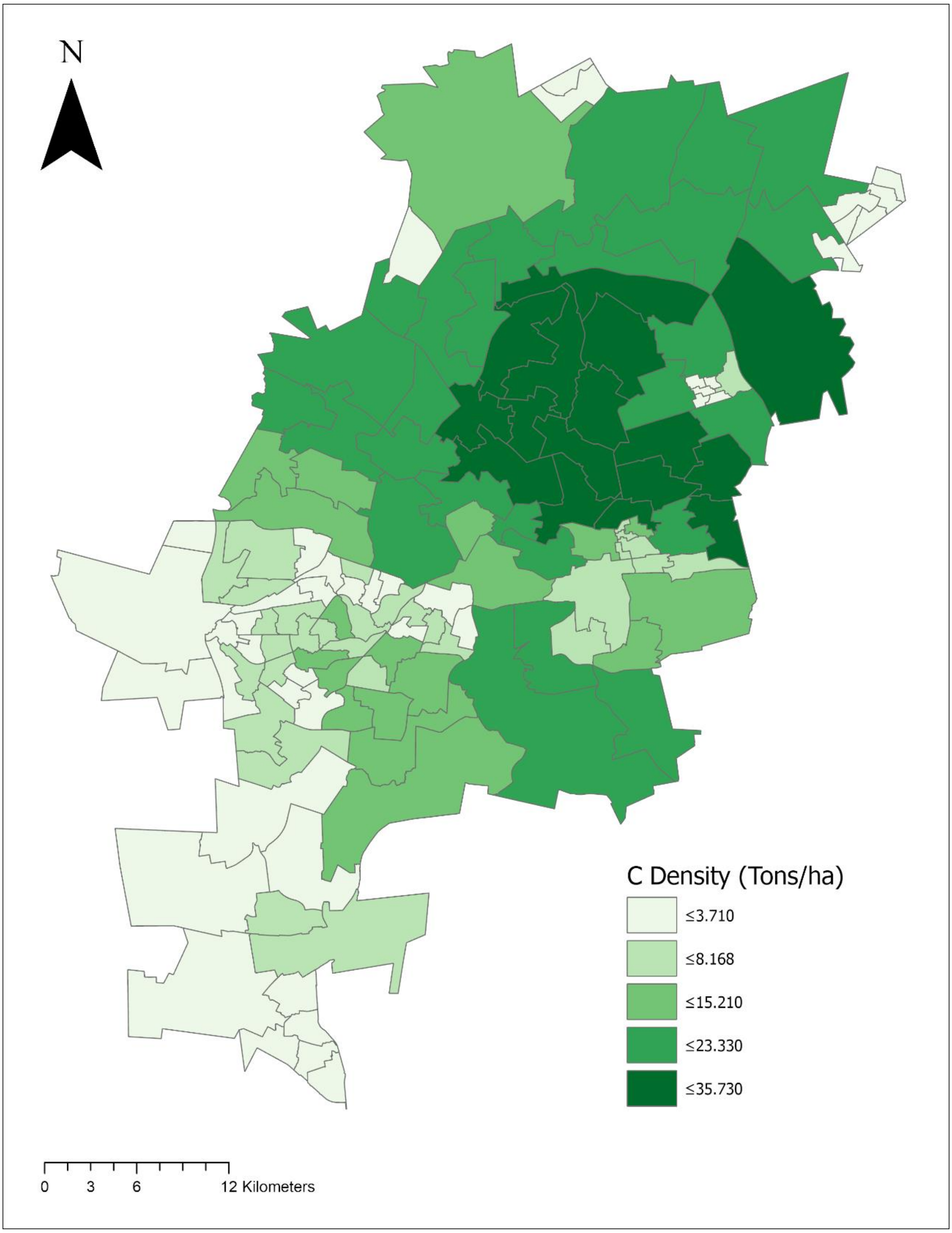

Figure 9: Carbon storage density across the administrative wards of the city of Johannesburg, South Africa.

4. Runoff Retention and Cooling Methods and Results

4.1 Runoff Retention Methods

Runoff retention is defined as the litre of runoff retention per m^2 of urban green space (l/m^2), as per Derkzen *et al*. (2015) and Liu and Russo (2021). It is also measured as the runoff ratio (runoff value relative to precipitation volume), ranging from 0-1.

To quantify the runoff retention provided by City of Johannesburg's green spaces, runoff infiltration as well as interception by vegetation was determined using inVEST's (v3.12.1) urban flood risk mitigation model. This modelling technique is adapted from the Soil Conservation Services (SCS) Curve number approach (Cronshey, 1986; Chow 1988). The SCS method is the most commonly used empirical method of hydrological modelling, originally created by the Soil Conservation Services (SCS, 1972) to estimate the volume of runoff in catchments for a given rainfall event (Muthu & Santhi, 2015). This method has been widely used in the estimation of runoff depth after rainfall events and for calculating runoff retention in urban settings (see Whitford *et al*., 2001; Weng, 2001; Tratalos *et al*., 2007; Derkzen *et al*., 2015; Kremer *et al*., 2016; Radford and James, 2013; Ramyar *et al.*, 2020; Basnou *et al*., 2020; Liu and Russo, 2021; Kadaverugu *et al*., 2021).

4.1.1 Soil Conservation Services Curve Number model theory

The SCS approach is derived from the equation used for determining surface runoff:

$$Pe = (P - 0.2S)^2 / P + 0.8S \qquad \textit{(Equation 4)}$$

where *Pe* represents surface runoff, *P* the precipitation and *S* represents the maximum potential runoff retention.

For the first variable, precipitation data were acquired from TAHMO (Trans-African Hydro-Meteorological Observatory) weather stations and the South African Weather Service (SAWS, 2021). Three design storms with varying return periods for Johannesburg were selected (12, 30 and 50 mm). For a one-hour rainfall event, these design storms represent 18-day (12 mm/hr), 2-year (30 mm/hr) and 20 year (50 mm/hr) return periods in the City of Johannesburg (SAWS, 2021).

An average, frequent heavy rainfall event in Johannesburg was input into the model (12 mm), alongside two high-intensity design storms with longer return periods (30 mm, 50 mm). This allowed the estimation of runoff retention under conditions where this service is most critical, extreme rainfall conditions, where flooding may occur and can be regulated by this service. In Johannesburg, floods are usually driven by heavy rainfall events over short time intervals (SAWS, 2021), hence 1-hour events were modelled. However, runoff retention is also important during moderate rainfall events, where it reduces drainage and water treatment costs (Derkzen *et al*., 2015), hence a more common rainfall event was also included.

The second input is the maximum potential runoff retention (S), derived from the following equation:

$$S = 2540/CN - 25.4 \qquad \textit{(Equation 5)}$$

where *CN* denotes the curve number, a value calculated by the Soil Conservation Service for various combinations of land cover classes and hydrological soil types. The CN serves as a quantitative characterisation of the land use and soil characteristics which influence runoff processes, under normal antecedent moisture conditions (Whitford *et al*., 2001).

To derive curve numbers for Johannesburg, land use and hydrological soil groups (HSG) datasets were required. Land use data were sourced from the South African National Land Cover dataset (DFFE, 2020), with a resolution of 20m. The second dataset required was the hydrological soil groups, a classification of four soil groups (A, B, C, D) according to their minimum infiltration rate (Table 11). Soil data relating to the hydrological soil groups (HSG) in Johannesburg were acquired from Schulze (2010), who mapped the HSGs across South Africa (Figure 10).

Table 11- Hydrologic soil groups (HSG) and their properties (Zeng *et al*., 2017).

HSG	Soil textures	Properties
A	Sand, loamy sand or sandy loam	Low runoff potential and high infiltration even when thoroughly wetted.
B	Loam, silt loam or silt	Moderate infiltration rate and moderately low runoff potential.
C	Sandy clay loam	Moderately high runoff potential and low infiltration rates when thoroughly wetted.
D	Clay loam, silty-clay loam, sandy clay, silty clay, or clay	Highest runoff potential, very slow infiltration rates.

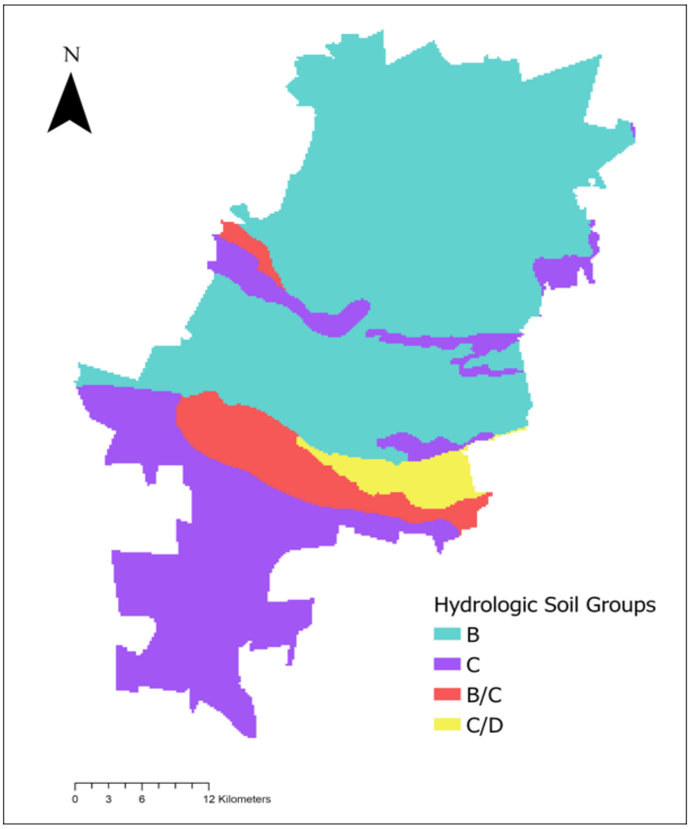

Figure 10: Distribution of hydrologic soil groups across the city of Johannesburg, South Africa (Schulze, 2010)

Curve numbers (CN) for different land uses in Johannesburg were determined using the SCS CN look-up tables (see Appendix B, Table B1), sourced from the TR-55 report on urban hydrology in small watersheds (Cronshey, 1986). For each South African National Land Cover (SANLC) land use, a CN value was assigned to each combination of that land use and the four HSG groups. CN values were assigned based on the closest respective land use in the SCS-CN tables (See Cronshey, 1986), or if not possible, the values used in prior literature in urban settings (see Vojtek & Voteková, 2016; Shrestha *et al*., 2021; Jahan *et al*., 2021; Liu and Russo, 2021). The complete list of CNs assigned to each land use in Johannesburg is presented in (Appendix B, Table B1).

Where a direct SCS-CN equivalent for a land use was absent, a composite CN was calculated according to Equation 6.

To calculate composite CN where multiple land uses exist in an area:

$$CN\ Composite = \frac{\sum A_i * CN_i}{\sum A_i} \qquad \text{(Equation 6)}$$

Where A= Area and CN= Curve number

Composite CNs were used to incorporate the effect of tree cover in land uses with significant tree cover and to adjust CN values to reflect the Johannesburg context accurately. For example, only lawn is accounted for in residential areas in the original Cronshey (1986) tables, based on American residential areas, but for Johannesburg the significant tree cover in residential areas needed to be accounted for.

Tree cover data from the ESA Worldcover V200 (10m) dataset was used to determine the area of tree cover used in calculating composite CN's. The weighting of the composite varied depending on the respective land use's tree cover and dominant vegetation. Where land uses are sub-divided by their dominant vegetation cover in the SANLC2020 dataset (e.g., smallholdings (tree), smallholdings (bush), smallholdings (grass), smallholdings (bare)), composite CNs were calculated with a weighting of 75% for their primary land use (e.g. smallholding) and 25% of their respective vegetation cover (e.g. tree or bush). No composites were calculated for bare land uses.

Other general assumptions included that impervious areas had a CN= 98, waterbodies and wetlands were assumed to be fully inundated and thus assigned CN= 100, open green space covered by trees within the city was in good hydrologic condition (Cronshey, 1986; Vojtek & Voteková, 2016). The SCS-CN look-up tables classify some LU according to their conditions. In Johannesburg, good conditions were assumed for all land uses with >25% tree cover, while fair condition was assumed for the remaining land uses, except for bare land uses, which were assumed to have poor condition (see Cronshey, 1986). The original SCS tables assign CN values to residential areas based on their proportion of impervious cover. The ESA Worldcover V200 (10m) land use dataset was determine proportion impervious cover in residential areas in Johannesburg, and assign CN values accordingly. The ESA Worldcover V200 (10m) land use dataset was determine proportion impervious cover in residential areas in Johannesburg, and assign CN values accordingly.

The next step in the model relates to a runoff coefficient, denoted by **Q**, determined by

$Q = Pe/P$ (*Equation 7)*

and ultimately, the runoff retention is determined by multiplying **P x Q** (Whitford, 2001; Tratalos, 2007). The proportion of precipitation retained during the rainfall event is determined by **1- Q/P (**Whitford, 2001; Kremer *et al*., 2016).

This model enables the quantification of runoff retention across urban areas (see Kremer *et al*., 2016; Liu & Russo, 2021). Furthermore, this method has been incorporated into open-source, hydrological modelling software such as inVEST (Integrated valuation of Ecosystem Services and Trade-offs) (Natural Capital Project, 2023). This study uses the inVEST (v3.14.1) urban flood risk mitigation (UFRM) model, which implements the SCS model in a spatially explicit manner. This enabled the mapping and spatial analysis of runoff retention in the CoJ in ArcGIS. For further detail how the model executes in each pixel, as well as the input data requirements see Appendix E.

The URFM implements the SCS model in each pixel of the study area, in a spatially explicit manner. The model was run for three design storms (12, 30 and 50 mm) across the City of Johannesburg. The three outputs of the model, runoff retention (m^3), runoff (mm) and runoff retention ratio (0-1), were available as rasters (20 m resolution) for each design storm. Each of the raster outputs was imported to ESRI's ArcGIS (v2.6.0) for analysis and mapping.

These rasters were integrated with the CoJ's administrative wards using the zonal statistics and join field tools in ArcGIS. Subsequently, runoff maps were created by aggregating the runoff retention to the ward level and calculating the sum and mean for each ward. The values per pixel (400m^2) were converted to per ha and per m^2 values respectively. Finally, the total (sum) values for each parameter were calculated at the city-wide scale, as well as at the region scale, for each design storm.

4.2 Local climate regulation (cooling) methods

The methodology used to quantify, and map cooling services was based on the generation of land surface temperature (LST) maps using Google Earth Engine (GEE) according to the methodology of Ermida *et al.* (2020). alongside the adaptation of the index methodology used by Kremer *et al*. (2016) and Schwarz *et al.* (2011).

LST data were collected for the City of Johannesburg, using a remote sensing approach. This method is indirect but spatially precise and has become the key method for quantifying temperature distributions at the city scale (Marando *et al*., 2019) due to its extent of spatial coverage, higher resolution and spatially continuous data as compared to *in situ* or weather station air temperature data (Mendelsohn *et al*., 2007; Tomlinson *et al*., 2011; Marando *et al.*, 2019; Zhou *et al*., 2019). The city-scale of this study called for a high resolution and spatially continuous dataset, therefore remotely sensed LST data were used.

Landsat 8 satellite scenes of Johannesburg during the summer months (Dec-Feb) for 2016-2022, were used to derive surface temperatures in Google Earth Engine (GEE) according to the methodology of Ermida *et al.* (2020). Landsat 8 satellite data were used in this study due to its high spatial resolution (30m) relative to other satellites (Tomlinson *et al.*, 2011; Zhou *et al.*, 2019). This enables discrimination of spatial features (see Loche *et al.*, 2022) and the assessment of temperature patterns at the city and urban green space scales.

Furthermore, there is an existing body of methodological frameworks for the extraction of LST data from Landsat satellite imagery which have been translated into code (Wang *et al*., 2015; Meng *et al.*, 2019; Ermida *et al.*, 2020) for Google Earth Engine (see Ravanelli *et al.*, 2018; Loche *et al*., 2022).

Ermida *et al*. (2020) provide a code repository which contains the GEE scripts, coded in JavaScript, needed to derive LST from Landsat data. The methodology is based on an algorithm which determines the empirical relationship between top of atmosphere brightness and LST through the application of a simple linear regression (see Sun *et al*., 2004; Li *et al*., 2013; Duguay-Tezlaff *et al*., 2015; Ermida *et al.*, 2020). The algorithm, as implemented by Ermida *et al.* (2020), alongside the source-code for GEE, is presented in Appendix C.

Ermida *et al.'s* (2020) methods were used to generate LST raster layers of Johannesburg for the years 2016-2022. Summer data were used as signs of cooling were expected to be more prominent when daytime temperatures were highest. Furthermore, heat stress and UHI intensity are highest during summer (Yao *et al*., 2020), making it the period of interest for the provision of cooling UES. All LST maps for each Landsat image acquisition (every 16 days) during the study period were downloaded in GEE. From these data (48 rasters), 13 rasters were selected for use. The selection of Landsat scenes was based primarily on the availability of summer scenes with less than 5% cloud cover to eliminate data gaps (as per Kremer *et al*., 2016). Upon generating LST maps, the data was extracted and transferred into ArcGIS. These data were then cleaned and re-projected for use in Kremer *et al*., (2016)'s local climate regulation index (*Equation 5 below*).

The index methodology of Kremer *et al*. (2016), see below, was adapted to enhance accuracy. The original methodology used a single Landsat scene (temperatures on a single day), whereas this study used multiple L8 scenes as inputs into the index. The mean LST for each pixel, across 13 selected LST rasters, was calculated in ArcGIS Pro (v2.6.0). Thus, an average raster was calculated, producing a spatially continuous LST raster, representative of the average temperatures across multiple days over 8 years, with high spatial resolution (30m). This LST map was then used as the input for the cooling capacity index in ArcGIS (see *Equation 8*). By using several days across multiple years, a spatially continuous average raster was created. This enabled the LST across Johannesburg to be measured under different climatic conditions, enhancing accuracy and reducing anomalies. Furthermore, this allowed the provision of cooling UES to be quantified under a range of temperature conditions, thereby ensuring the service is more accurately and reliably quantified. These data were then used to assess temperature differences between urban green and blue spaces (UGBS) as compared to impervious surfaces across the city (Kremer *et al.*, 2016). The mean LST data were integrated with land cover data, namely the South African National Land Cover (SANLC) 2020 dataset using the zonal statistics and join field functions in ArcGIS. The temperature of each land use in the SANLC dataset was extracted from the mean LST raster. The mean LST of all pixels covered by green spaces, waterbodies and bare ground was calculated, to serve as an input into the index calculation (*Equation 8*).

4.2.1 Cooling Capacity Index

The cooling service provisioning was quantified for each pixel in the LST raster of Johannesburg, using the following index as per Kremer *et al*. (2016):

$$\text{Local climate regulation indicator (i)} = Temperature\ (i)/Temperature\ (g)\ x\ 100 \qquad \textit{(Equation 8)}$$

where (i) denotes an individual pixel in the average LST raster within ArcGIS and (g) represents the mean LST of pixels which are covered by urban green space, water, or bare ground. This index enables the quantification of the cooling service provided by Johannesburg's UGBS based on LST data by providing values for each pixel, allowing for this service to be mapped.

The cooling capacity index dataset, representative of local climate regulation UES provision, was imported in ArcGIS and the output mapped at 30m resolution. This provided a spatially continuous, high resolution cooling capacity index across the COJ, allowing for the cooling capacity to be quantified by comparing temperatures in each pixel to the mean temperature of urban green space. The resultant map was then normalised to fit a scale of 0-100 and inverted, so that 0 represents the lowest cooling capacity and 100 represents the maximum cooling capacity. This cooling capacity map was integrated with a shapefile of Johannesburg's administrative wards, using the zonal statistics and join field functions in ArcGIS Pro . Finally, the mean cooling capacity was calculated for each ward of the CoJ and mapped accordingly. A schematic overview of the complete GIS workflow, alongside the ArcGIS pro tools used at each step, is presented in Figure 11.

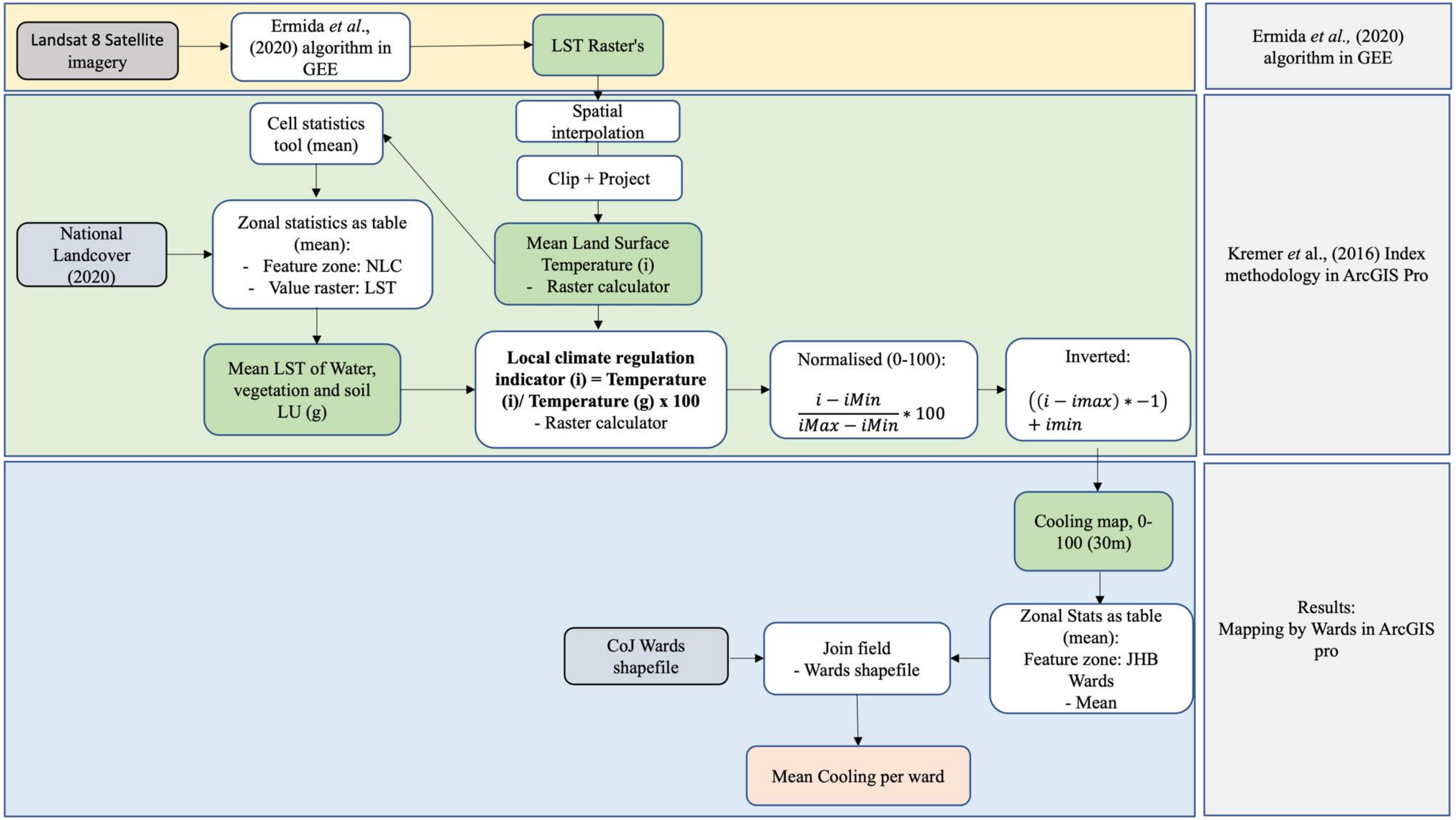

Figure 11: Workflow for Cooling UES quantification in Google Earth Engine and ArcGIS Pro (version) *Note:* GEE= Google Earth Engine, LST= Land surface temperature, i = individual pixel in map, in ArcGIS Pro. Projections were made to *Albers equal area conic* projection system. Grey boxes represent inputs, Green boxes represent key outputs. White boxes indicate methodological steps, GIS tools or temporary products.

4.3 Runoff retention and Cooling Results

This study aimed to quantify the supply of runoff retention and cooling (local micro-climate regulation) ecosystem services in the city of Johannesburg and map them according to the city's administrative wards. The inVEST UFRM model was used to implement the SCS CN model in a GIS environment to estimate runoff and runoff retention across Johannesburg, under three design storms. A remote sensing approach in conjunction with an indicator were used in GIS to determine the supply of cooling services in each pixel across the city. The results revealed distinct spatial variation in the supply of these services across Johannesburg, with high supply in the north and lower, more variable supply in the south.

4.3.1 Runoff Retention

The results of the inVEST model revealed that the total runoff in Johannesburg is 11.16 million m^3 under a 12 mm h^{-1} rainfall event, while a more extreme, longer return period event of 50 mm h^{-1} generates 21.89 million m^3 of runoff across the city (Table 12). Flood mitigation through Johannesburg's UGS is quantified through runoff retention, with the average volume of runoff retained ranging from 113.05 to 366.75 m^3/ha under the modelled storms (Table 12). The total runoff retained across the city for a 50mm h^{-1} rainfall event is 60.24 million m^3, compared to 18.57 million m^3 under a 12mm h^{-1} event.

The runoff retention ratio is a proportional measure of the volume of runoff retained relative to the precipitation volume. The average runoff retention ratio across the city decreases from 0.943 in a 12 mm event to 0.733 in a 50 mm event (Table 12).

Table 12: Sum and mean (ha^{-1}) runoff, runoff retention and runoff ratio for the entire city of Johannesburg, under 12 mm h^{-1}, 30 mm h^{-1}, and 50 mm h^{-1} design storms.

	Mean per ha			Sum		
Design storm (mm)	**12**	**30**	**50**	**12**	**30**	**50**
Runoff (m^3)	6.79	46.38	133.29	1115627	7619181	21894573
Runoff Retention (m^3)	113.05	253.64	366.75	18570103	41663949	60243978
Runoff Ratio	0.943	0.8454	0.7334	-	-	-

The average and total values for each variable (retention, runoff, retention ratio), for each ward in the CoJ, under each design storm, are available in Appendix L.

The highest runoff retention ratio in a ward decreased from 0.994 in a 12 mm event, to 0.945 in a 30 mm event, to 0.859 in a 50 mm event (Appendix L). Meanwhile, the lowest runoff ratio in the CoJ's wards was 0.734 (12mm event), 0.636 (30mm event), and 0.529 (50 mm event). Thus, the poorer-performing wards, with low ES supply, are affected much more than the best-performing wards as rainfall intensity increases from 12 mm h^{-1} to 50 mm h^{-1}.

Runoff retention, runoff, and retention ratio were modelled across the wards of Johannesburg, under three design storms (12 mm, 30 mm and 50 mm), allowing for the spatial distribution of these services to be assessed (Figure 12). The maps for 12- and 30 mm events are presented in Appendix M. The results show that as storm intensity increases, the supply of runoff retention becomes more variable across the city, evident in the larger differences in runoff ratio and retention between wards in a 50 mm event (Figure 12) compared to a 12 mm event (Appendix G: Figure 1, Figure 3).

Both runoff and runoff retention are distributed unevenly across Johannesburg, during a 50 mm hr-1 rainfall event, with large spatial variation in the amount of runoff generated and retained (Figure 12). The runoff in the northern regions of Johannesburg is considerably less than in the southern regions, while runoff retention is higher in the north as compared to the south (Figure 12).

Maximum runoff retention is provided to the majority of the northern wards, particularly those in the central northern suburbs, in turn the northern wards receive the lowest amount of runoff (Figure 12). The north-east corner and wards in and around Alexandra (north-east) stand out as exceptions in the north, receiving high runoff and having low runoff retention (Figure 12). Meanwhile, the southern region of the city is characterised by low runoff retention and high levels of runoff, particularly in the southwestern and southernmost wards which receive minimum runoff retention (Figure 12). The highest supply of runoff retention in the south, is provided in the south-eastern wards (Figure 12). In addition to the southern wards, the central-eastern region (Johannesburg CBD) also stands out, receiving maximum runoff and minimum retention (Figure 12). Overall, high retention values were observed over areas with high proportions of green space as well as open spaces. Meanwhile, high levels of runoff are associated with high proportions of impervious surfaces, limited green space and/or proximity to water bodies.

The runoff retention ratio displays the same spatial trends as runoff retention, with high rations in the north (0.78-0.86) but low ratios in the southern regions (Figure 12). The mean retention ratio per ward ranges from 0.859 to a minimum of 0.529. Thus, in the wards with the highest supply of runoff retention, nearly 86% of precipitation is retained, while in the wards with the lowest supply, only 53% of precipitation is retained, resulting in more than double the amount of runoff during a 50 mmhr-1 rainfall event. The variation in runoff ratio across the city has large consequences for the capacity of different regions to mitigate flooding under extreme rainfall events, reflected in the differences in runoff across different regions of the city (Figure 12). The low retention ratios in many of the city's Southern wards indicates that high runoff will occur under extreme events and these regions are vulnerable to flooding (Figure 12). Alternatively, the high ratios in the north reflect the role of green spaces in effectively mitigating runoff, even under extreme rainfall events.

The 5 wards with the highest provision of runoff retention, under a 50 mm event, all occur in Northern Johannesburg, with a maximum of 429.50 m^3/ha retained. Meanwhile, the 5 wards with the lowest provision of runoff retention under a 50 mm event, all occur in Southern Johannesburg, with a minimum of 264.25 m^3/ha. Wards in the south-west (Soweto), central-east (CBD) and southernmost regions of the city stand out as being critically undersupplied (Figure 12).

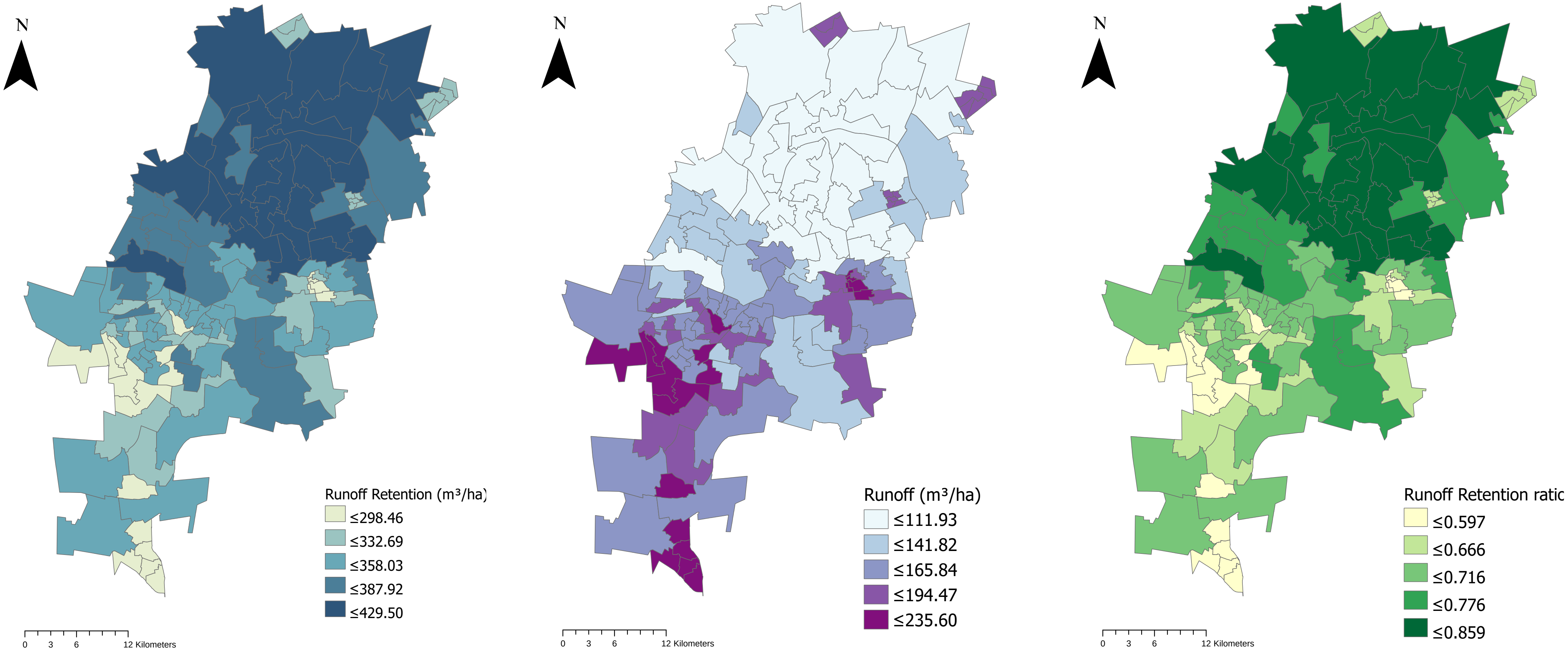

Figure 12: Average runoff retention, runoff, and retention ratio across the administrative wards of Johannesburg, during a 50 mm h^{-1} rainfall event.

4.3.2 Cooling Results

Cooling service supply in Johannesburg was quantified at both the broader administrative region scale, as well as the finer, ward scale (see Figure 3). Within Johannesburg's administrative regions, the highest cooling capacity was provided by regions B and E (north central and north-eastern regions), while the lowest was provided in regions D and G (Table 13). The large differences in cooling capacity correspond with large temperature differences between regions. Regions with the lowest cooling capacity (D and G) correspond with the highest average LST (41.21 and 40.80 ^{o}C, respectively), while the highest cooling capacity (regions B and E), have the lowest average LST (37.49 and 37.89 ^{o}C respectively). The largest difference in average temperature between regions is 3.72 ^{o}C, corresponding to a cooling capacity difference of 11.64 (Table 13). Thus, the regions with the lowest cooling ES provisioning, are exposed to the highest temperatures. Regions D and G (Southern and South-western regions) are particularly vulnerable to high urban temperatures, with limited mitigation through supply of cooling services.

Table 13: Mean cooling service supply and land surface temperature (LST) in the administrative regions of Johannesburg, South Africa.

Region	Mean Cooling Capacity	Mean LST (^{o}C)
A	58.99	38.96
B	63.59	37.49
C	59.76	38.71
D	51.95	41.21
E	62.33	37.89
F	59.59	38.77
G	53.23	40.80

Cooling UES supply was quantified and mapped across the administrative wards of Johannesburg, revealing distinct spatial variation across the city (Figure 13). There are clear disparities in cooling capacity in the city, with the northern half having considerably higher supply of cooling services compared to the south. The maximum cooling UES supply occurs in the central northern wards of Johannesburg. A small cluster of wards in the north-east (Alexandra) stand out as an exception, having the lowest cooling supply in the city, despite being surrounded by regions of generally high supply (Figure 13). Within the southern regions, the highest supply is provisioned in the south-eastern region, (Figure 13). The rest of the southern regions are characterised by low supply. In particular, the south-western, westernmost and south-westernmost wards display a critically low supply of cooling services (Figure 13).

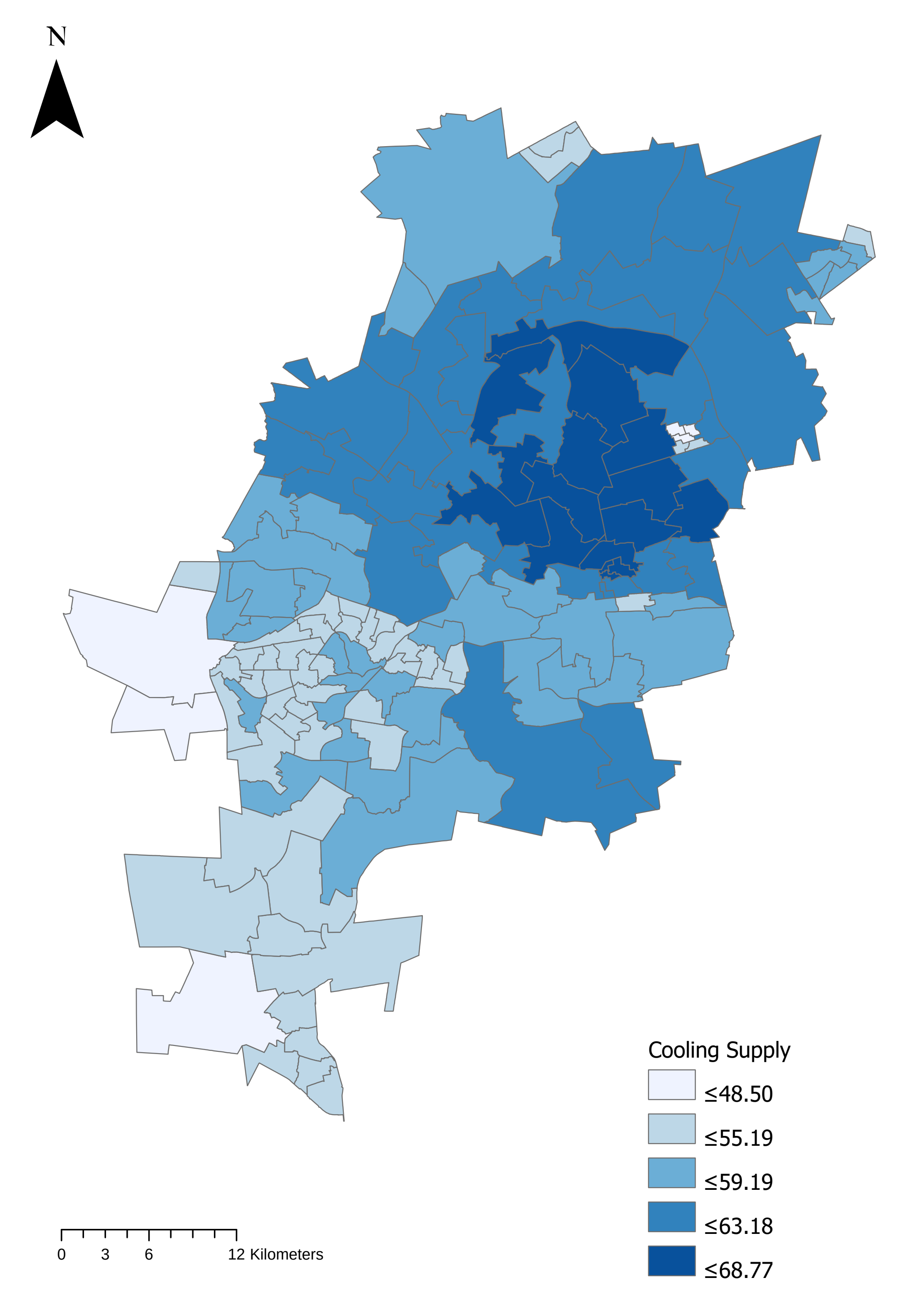

Figure 13: Supply of cooling ecosystem service across the administrative wards of Johannesburg, South Africa.

Overall, at both the region and ward scale, the cooling supply across the CoJ is extremely spatially variable, with a generally higher and more consistent supply of cooling in the North, but more variable and often critically low cooling capacity in the South.

The cooling capacity index is available in a finer resolution (30 m), before aggregation to ward level (Figure 14). The supply of cooling services at both the ward and 30 m scale, shows distinctive spatial overlap with the distribution of tree cover across the city (See Figure 2). The region with the highest cooling supply (north-central), corresponds to the highest density of tree cover and urban greening in the city. Meanwhile, areas of minimal or low supply are characterised by an absence of tree cover and a high proportion of impervious surfaces or bare ground. However, despite low tree cover in the south-western regions of the city, various wards display moderate cooling supply due to their proximity to rivers and waterbodies. The cooling supply of these rivers, and the absence of other cooling sources in the south-west can be seen in Figure 14. Aside from the rivers and greenspace in the south-east, the rest of the South has almost no cooling supply (Figure 14).

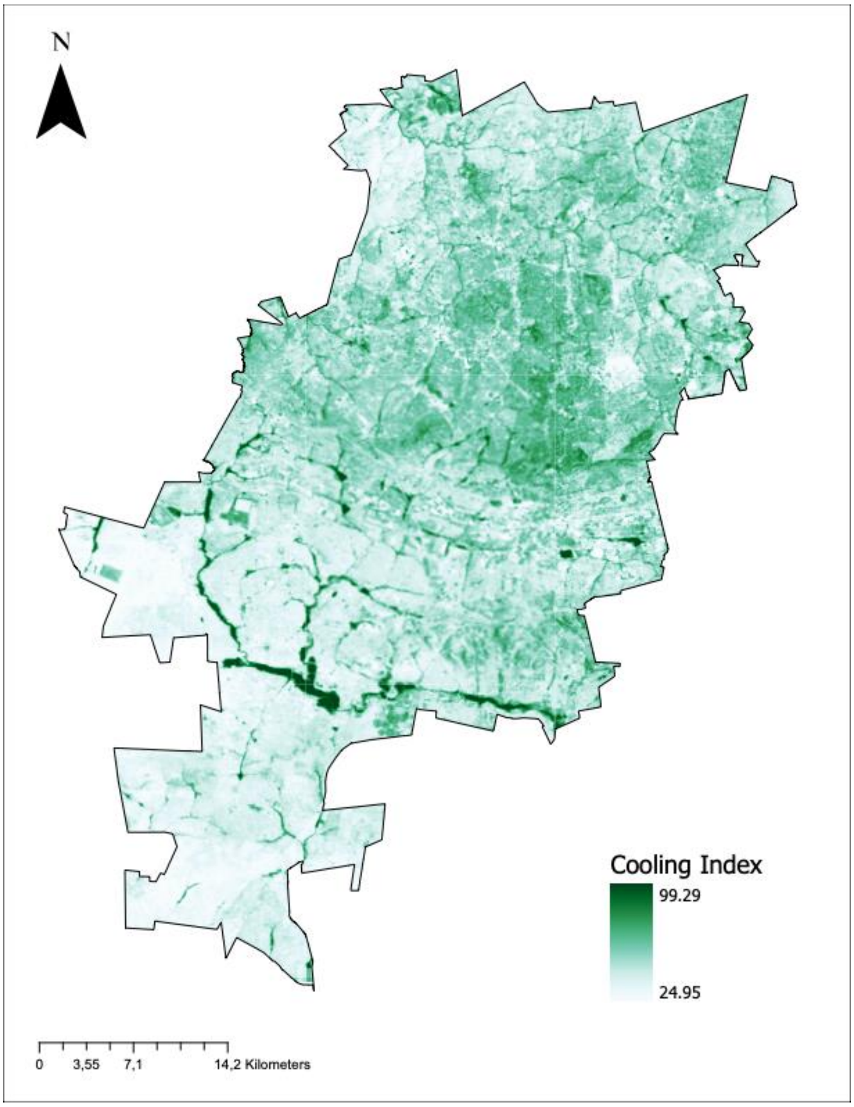

Figure 14: Cooling ecosystem service supply index (30 m resolution) across the city of Johannesburg, South Africa, based on land surface temperature data (2015-2022).

The average land surface temperature map of Johannesburg, used as an input into the cooling index, is presented in Figure 15. The areas of low cooling supply coincide with temperature hotspots, while areas of high supply generally represent cool spots (Figure 14, 15). The delineation of hotspots and cool spots is informed by visual interpretation of areas with distinctly higher and lower temperatures, as compared to their surroundings. Temperatures are distinctly cooler in the northern regions, in accordance with their high cooling service supply (Figure 15). The south is characterised by high temperatures, coinciding with low cooling provision. Key hotspots include Alexandra, the north-western corner, the south-western corner, the majority of Soweto as well as orange farm and the southern townships, including Orange Farm (Figure 15).

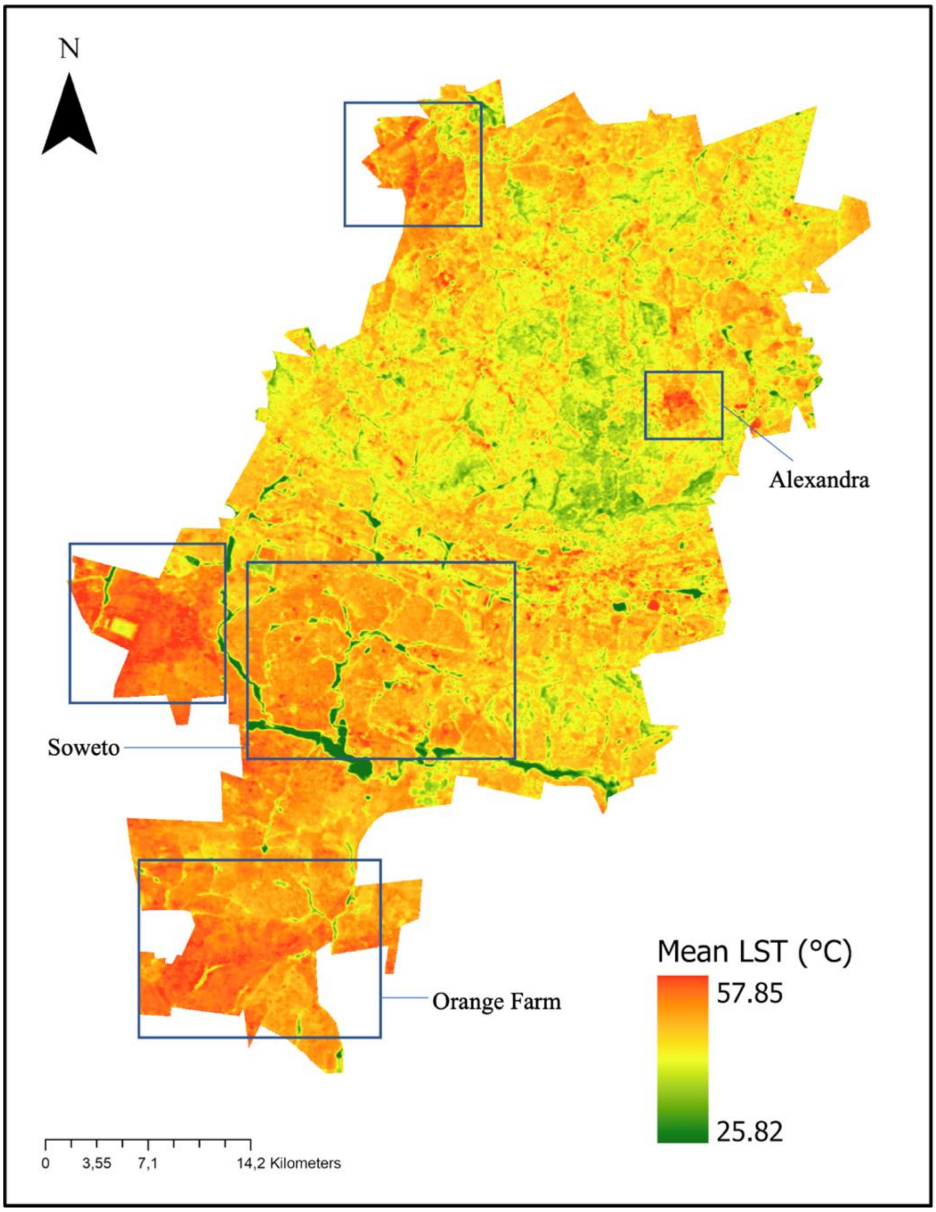

Figure 15: Mean land surface temperature across the city of Johannesburg, with hotspots highlighted, based on selected Landsat 8 satellite images during summer months of 2015-2022.

5. Ecosystem service comparison

5.1 Methods

After spatially quantifying carbon, cooling and runoff supply in each ward in Johannesburg, we then normalised the ES supply values according to equation 9 below. This allowed us to derive relative values of ES provision, on a scale from 0-100, to enable the comparison of services between wards.

$$I_{norm} = \frac{i - i_{min}}{i_{max} - i_{min}} \qquad (\textit{Equation 9})$$

Where I_{norm} is the value of ecosystem service supply at a given pixel (i), normalised on a scale of 0-100. i_{min} and i_{max} are the maximum and minimum value of ES supply.

The normalised service values were then aggregated to the ward scale and mapped for all three services in ArcGIS Pro (v2.6.0).

5.2 Results

5.2.1 Spatial trends across UES

The three UES in Johannesburg were normalised on the same scale (0-100) to enable the spatial and quantitative comparison between services and the identification of synergies and trade-offs. The spatial comparison of carbon storage, cooling, and runoff retention services is presented in Figure 16.

The key spatial trends across all services include a higher and more consistent supply of UES in northern Johannesburg, but a lower and more highly variable supply in southern Johannesburg (Figure 16). The central Northern suburbs of the city have maximal supply across all three services. However, northern Johannesburg does not receive a uniformly high supply, with the northwest corner of the city having reduced supply across all three services. A small group of wards in the north-east (Alexandra) also stands out as a key exception in the north, with all its wards having low to minimum provisioning across all three services, despite being surrounded by regions of high supply (Figure 16).

Southern Johannesburg is characterised by high spatial variability in UES supply and generally low to minimum provisioning of services (Figure 16). In particular, carbon storage is the most undersupplied service in the south, followed by cooling while runoff has the greatest variation. The southwestern region and the southernmost region are consistently undersupplied across all services, while the south-eastern wards have a relatively higher supply (Figure 16). Central Johannesburg is characterised by moderate, low, and minimal provisioning with some variation across services. The central-east (CBD) wards have low supply across all services, with the greatest supply of cooling and the least supply of runoff retention (Figure 16).

Overall, runoff retention has the most consistent distribution across the city, while carbon storage has the most variable distribution in its supply (Figure 16). The most critically undersupplied regions are the southwestern, westernmost, southernmost and north-eastern wards (Figure 16).

5.2.2 Synergies and Trade-offs

The relationship between services in Johannesburg is largely characterised by synergies rather than trade-offs, albeit trade-offs do occur. There is distinct synergy and spatial overlap between the provisioning of cooling and carbon storage services, likely due to the influence of tree cover in the delivery of both services (Figure 16). The extensive and diverse greenspaces in the central northern wards of Johannesburg provide maximum supply of all three services simultaneously (Figure 16).

While the city's greenspace provides synergistic supply of multiple services, this is not universal and in several instances trade-offs occur. Despite the overlap between cooling and carbon storage, the southwest region has a moderately higher supply of cooling than carbon storage in several wards (Figure 16). In particular, this occurs in wards with close spatial proximity to rivers and waterbodies, which provide cooling services, but do not contribute towards above-ground carbon storage. Thus, rivers represent a trade-off between cooling and carbon storage services, evident in the northeast corner of the city. Rivers represent further trade-offs between cooling and runoff retention. This relates to the cooling effect of rivers running through the region but the simultaneous assumption in the runoff model that under full inundation, the rivers will not retain any runoff. This is evident in the southeasternmost ward, which receives high supply of cooling services but a low supply of runoff retention (Figure 16). Wards in the southernmost and westernmost regions show high values for runoff retention, but low supply of carbon storage and cooling, representing a trade-off (Figure 16).

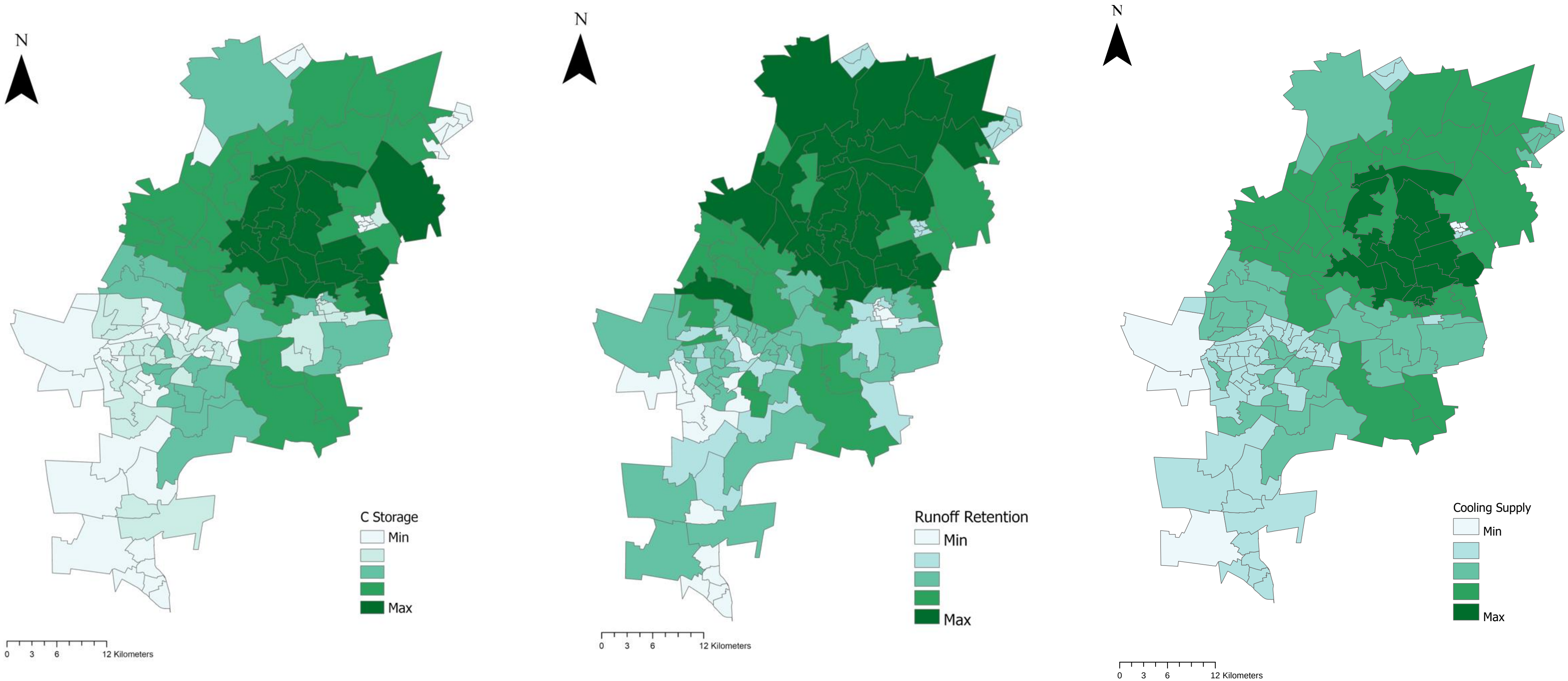

Figure 16: Normalised UES supply score for carbon storage, cooling, and runoff retention services mapped across the administrative wards of Johannesburg, South Africa.

The total UES supply score (0-300) for groups of 4-6 wards in the city is presented in Figure 17. The wards are ordered along the x-axis as they occur spatially, from south to north and east to west. This highlights the spatial disparity in the supply of all three services across Johannesburg, with low values across the left side (southern Johannesburg) but extremely high values on the right side (northern Johannesburg).

Several wards in Johannesburg receive a high supply (>200) of all three crucial services, while numerous wards receive a low supply (<100) of all three services (Figure 17). The majority of wards in the city face an undersupply across all services. The right side of the graph portrays two distinct peaks, the first in the central northern and north-eastern suburbs and the second in the further northern suburbs (Figure 17). These northern suburbs show nearly maximum supply in all three services.. Additionally, there are two distinct troughs in the northern half of the graph (Figure 17). The first and most extreme is in Alexandra, which represents a stark contrast to its surrounding wards. The second, although not as extreme, is still a large reduction from its surroundings and represents the northeast corner of the city (Figure 17). Meanwhile, the southern half of the city is largely undersupplied (Figure 17).

The trends in ES supply further support synergies rather than trade-offs between services, although there are exceptions. Generally, where a ward receives a high supply (>80) of one service, the other two services are also well supplied (Figure 17). However, trade-offs do occur between the services in several regions of the city. In cases where both carbon storage and runoff retention values are very low, but cooling is not, the ward generally occurs over rivers or water bodies. This is a fairly common trend in the southernmost and southwestern wards, on the leftmost side of the x-axis (Figure 17). Generally, a low supply of carbon storage (>10), with a moderate supply of cooling and or runoff retention (<40) is associated with green and blue spaces not inclusive of trees. In these cases, rivers and or grasslands can provide a moderate supply of cooling and runoff retention, despite being limited in carbon storage. There also exist cases where runoff retention is far higher than the other two serviced (Figure 17), suggesting a trade-off. This usually occurs in regions with a high proportion of bare ground or bare soil.

Generally, where one service is critically undersupplied (>15), the other services are also in low supply (<50).

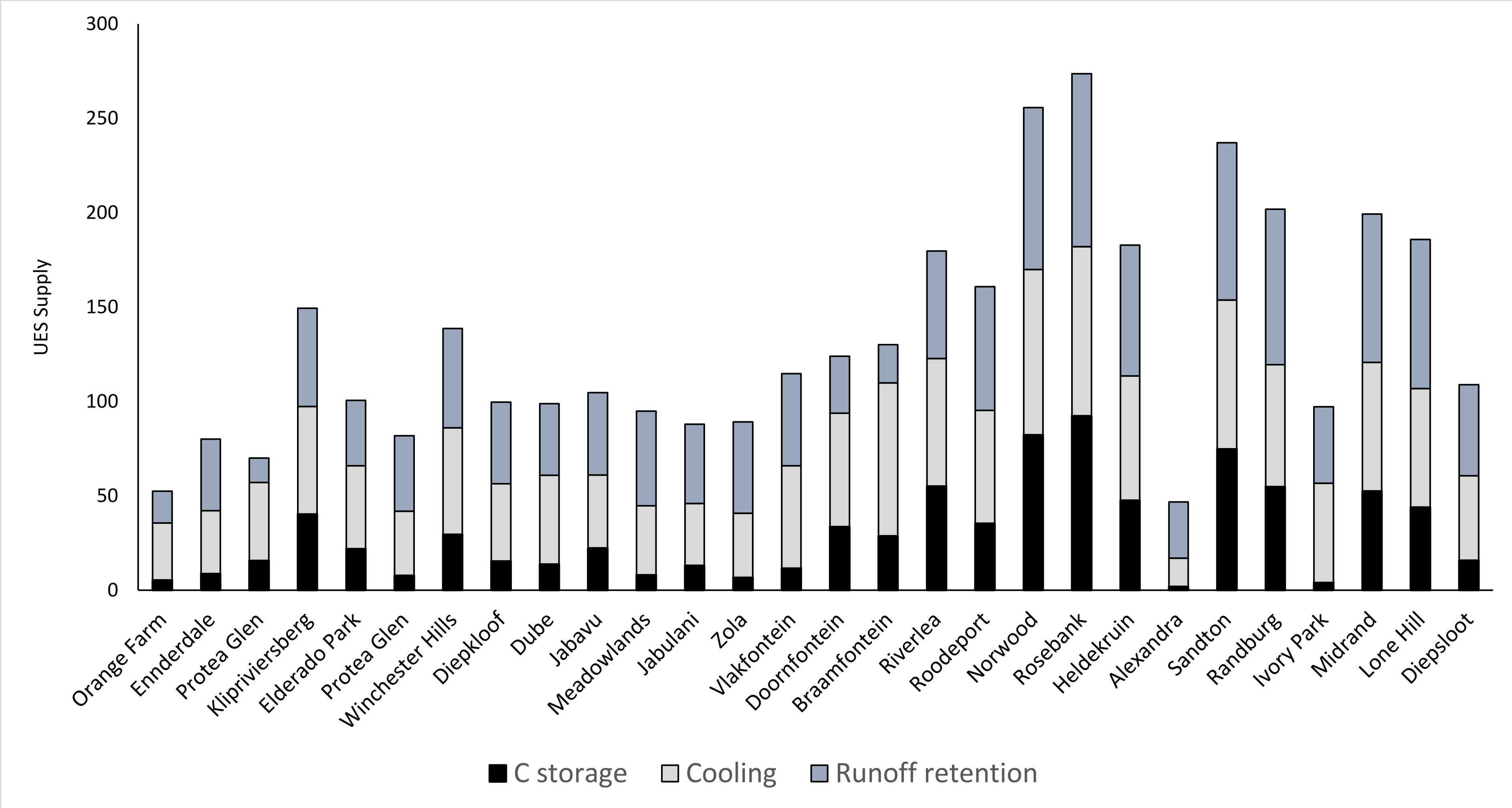

Figure 17: Normalised (0-100) UES supply score for carbon storage, cooling, and runoff retention services across groups of administrative wards in Johannesburg, South Africa. Neighbouring wards were grouped, in groups of 4-6 wards. Key named locations within those groups are presented. The groups are presented along the x-axis as they occur spatially, from South to North.

Chapter 6 . Discussion and Conclusion

Urban areas face a range of significant challenges on account of their morphology, including the urban heat island effect and urban flooding, which are exacerbated by the impacts of climate change and threaten the sustainability of cities into the future. Urban green spaces and the ecosystem services they provide have emerged as a valuable solution which can address these challenges, mitigate climatic change, and bolster urban resilience to climate change impacts. To enhance and maintain these services, and ensure their equitable distribution, it is critical that their supply is quantified and made spatially explicit. In this study, I set out to assess the supply of three crucial ecosystem services (carbon storage, runoff retention and cooling) spatially and quantitatively in the city of Johannesburg. Specifically, I quantified the supply of these three services in Johannesburg using bio-physical and non-economic metrics, then I mapped their spatial distribution across the administrative wards of Johannesburg.

6.1 Above-ground carbon storage in Johannesburg

This study estimates the C storage of Johannesburg's urban forest to be 2.4 million tC, equivalent to 8.83 million tCO_2. Across the entire city, an average of 14.65 t/ha of carbon are stored in the aboveground biomass, with the carbon storage per unit tree cover ranging from 39.92-97.50 t/ha between different UF types (Chapter 3, Table 8). The CoJ's above ground carbon stock is an impressive and significant source of C storage, surpassing the C stock of several other cities (e.g. Nowak *et al*., 2011; Liu & Li, 2012; Strohbach & Haase, 2012; Dorendorf *et al*., 2015; Sun *et al*., 2019).

Prior research conducted in Johannesburg estimated 5.3 million metric tonnes of carbon storage across the city (Schäffler & Swilling, 2013). However, it should be noted that Schäffler and Swilling (2013) conducted only a single plot of field sampling, producing a limited scope case study, with greater focus on other dimensions. In my study I increased the plot number substantially, sampled a diverse range of UF types and took steps to improve the accuracy of the extrapolation. This is particularly important in Johannesburg, which is highly heterogenous in structure, with great diversity in green spaces. Schäffler and Swilling (2013) extrapolated carbon storage from a park plot, assuming a uniform tree cover for the entire city. In reality, only 16.1% of the city's area is covered by trees which this study accounted for during extrapolation of carbon storage values to the whole city. Therefore, my results which present a more conservative carbon stock of 2.4 million tC may be a stronger estimation for Johannesburg.

Beyond this case study there is, to my knowledge, a distinct lack of comparable carbon storage studies in South African cities, highlighting the knowledge gap in the region (Haase *et al*., 2014; Crossman *et al*., 2013; O'Donoghue & Shackleton, 2013; Weissert *et al*., 2014). While some research has been conducted on the capacity of street and parking lot trees to sequester carbon in Tshwane and the Eastern Cape of South Africa (Stoffberg *et al*., 2010; O'Donoghue & Shackleton, 2013), these studies do not address carbon storage.

Research on the carbon storage of South Africa's natural forests generally reports higher values of carbon density than those found in Johannesburg, but within range of the values found in parks (98.5 t/ha), the highest of the four UF types assessed. Indigenous South African forests in the coastal lowlands of the Eastern Cape were found to store 99.8 t/ha on average (Smithwick, 2019), while Mangwale *et al*. (2017) presented a broader range of carbon densities for the same region, from 12.4-113.7 t/ha. More generally, carbon stocks in sub-Saharan African forests typically range from 30-150 t/ha, with great variation between different climates and forest types (Houghton & Hackler, 2006; Bombelli *et al*., 2009). Meanwhile, extremely high carbon storage values (179 t/ha) were found in the high elevation, Afromontane mist belt forests in Limpopo, South Africa (Mensah *et al*., 2016). This is representative of the higher storage associated with African tropical montane forests, which average 149.4 t/ha in above-ground carbon stock across numerous African countries (Cuni-Sanchez *et al*., 2021). Thus, the average carbon stock of Johannesburg's four UF types (71.96 t/ha) is within range of South African forest carbon density, albeit somewhat lower. However, the carbon stock of Johannesburg remains notably lower than the high densities observed in tropical, Afromontane forests. Beyond natural forests, similarly high carbon storage was found in three plantation forest species in KwaZulu Natal, South Africa, with 103 t/ha in *Eucalyptus* and 143 t/ha in *Pinus* plantations (Dube & Mutanga, 2015). These values are similar to our findings in parks in Johannesburg (98.5 t/ha), which is reasonable given the predominance of plantation species such as *Eucalyptus* and *Pinus* in the city's parks (Chapter 3, Table 6). In contrast, studies on grassland and savanna systems generally presents aboveground carbon stocks lower than that of the Johannesburg urban forest. For example, the central lowveld savanna of South Africa was found to store 18 tC/ha (Shackleton & Scholes, 2011), while the Miombo woodlands of northern Mozambique store 29.8 t/ha in trees (Ribeiro *et al*., 2013). Meanwhile, Mills *et al*. (2005) presented extremely low values of 2 t/ha for aboveground carbon storage for mesic grasslands in South Africa.

The average carbon storage per unit tree cover of Johannesburg's urban forest is in the same order of magnitude as results from other cities in Europe, USA, Asia, and Africa (e.g. Jo, 2002; Nowak *et al*., 2013; Strohbach & Haase, 2012; Moussa *et al*., 2019). However, caution is advised when making comparisons of the findings of different studies in different cities. There is considerable variation in estimates of carbon storage across different cities, influenced by diverse spatial-temporal factors such as climate, land management practices, history of urbanisation, ecology, and population densities (Davies *et al*., 2013). A crucial factor contributing to these differences and hindering reliable inter-city comparisons is the variability in methodological approaches used to estimate urban carbon storage (Davies *et al*., 2013). This includes variation in sampling methods, land-use classifications and differences in the utilisation of allometric equations (McHale *et al*., 2009; Lindén *et al*., 2020). The absence of standardised methods, uniform definitions of 'urban,' and consistent approaches represents prominent challenges in urban ecology, impeding the comparability of research findings (Davies *et al*., 2013). With these differences in mind, it is possible to cautiously compare the mean carbon storage per ha of urban forest in Johannesburg to other studies in cities around the world.

The mean carbon storage across four urban forest types in Johannesburg (71.96 t/ha) is markedly similar to the average across 28 US cities with 76.9 t/ha per unit tree cover (Nowak *et al*., 2013), and to the urban forest of Liepzig, Germany with 68.2 t/ha (Strohbach & Haase, 2012). Meanwhile, Los Angeles and Sacramento in the US had values of 59 and 84.5 t/ha (McPherson *et al*., 2013), showing further similarities to Johannesburg. Several studies in Asia presented values far lower than those found in Johannesburg. The average carbon density of UGS in the 35 major Chinese cities is 21.34 t/ha (Chen, 2015), with other Chinese cities showing similarly low values of 30.25 t/ha in Hangzhou (Zhao *et al*., 2010) and 33.2 t/ha in Shenyang (Liu and Li, 2012). This is primarily attributed to the predominance of young vegetation in Chinese green spaces (Zhao *et al*., 2010; Chen, 2015). Amongst other Asian cities, Nagpur, India also showed a lower carbon storage than Johannesburg, with 31.51 t/ha, while Seoul, South Korea showed greater similarity, storing 58.7- 60.1 t/ha (Jo, 2002). Conversely, some studies presented values vastly higher than those observed in Johannesburg. Addis Ababa, Ethiopia and Kumasi, Ghana both showed substantially higher C storage than Johannesburg, with 172 and 228 t/ha, respectively (Woldegerima *et al*., 2017; Nero *et al*., 2018). Johannesburg's urban forest also stored far less carbon than the urban green spaces investigated in in Leicester, UK (281- 289 t/ha) (Davies *et al*., 2011). These differences in carbon storage per unit tree cover may be due to differences in tree density, size distributions and species composition (Nowak *et al*., 2013).

Moving beyond carbon storage per unit tree cover, C storage at the city-wide scale shows further variation between different cities. For instance, Leicester, UK, exhibits a notably high carbon storage level at 31.6 t/ha (Davies *et al*., 2011), while Meran in Northern Italy demonstrates a comparatively lower value of 13.5 t/ha (Speak *et al*., 2020). Boston, US, showcases a carbon storage level of 28.8 t/ha (Raciti *et al*., 2014), contrasting with Beijing's lower estimate of 7.8 t/ha (Sun *et al*., 2019). Similarly, Leipzig, Germany, presents a modest level of 11.8 t/ha (Strohbach & Haase, 2012), whereas Hamburg, Germany, illustrates a higher carbon storage level across land use at 27.4 t/ha (Dorendorf *et al*., 2015). Johannesburg's mean carbon storage (14.65 t/ha) aligns closely with certain cities such as Meran, Italy, and Leipzig, Germany, yet falls short of the notably higher estimates found in Leicester, UK; Boston, US and Hamburg, Germany. Interestingly, American cities such as Los Angeles and Sacramento present city-wide average estimates of carbon storage at 8.15 t/ha and 15.4 t/ha, respectively (McPherson *et al*., 2013), which closely resemble Johannesburg's mean carbon storage.

This variation reflects the diverse ecological and urban characteristics which drive carbon storage at the city-scale. One particularly important driver of carbon storage at this scale, is the proportion of tree cover within the city. Cities with higher carbon density at the city-scale than Johannesburg, generally have much higher tree cover. For example, the city of Boston stores nearly twice the amount of carbon as Johannesburg per ha of urban area, due to having 25.5% tree cover (Raciti *et al*., 2014), compared to Johannesburg's 16.1%. Similarly, Johannesburg has less tree cover than 21 of the 28 US cities assessed by Nowak *et al*. (2013). Other factors contributing to these differences may include each city's unique history, climate and management of green spaces (Zhao *et al*., 2010; Davies *et al*., 2013; Nowak *et al*., 2013; Lindén *et al*., 2020).

Although methodological differences may drive some of the variation in carbon storage estimates between cities, Johannesburg's carbon stock is significant in the context of comparable urban forests. Our results for C density across four forest types are in line with findings across a wide range of cities in Europe, Asia, America, and Africa. However, at the city-wide scale, Johannesburg stores less carbon than several other cities due to the disproportionate distribution of tree cover in the northern region of the city. Thus, Johannesburg's urban forest simultaneously exemplifies the key role that can be played by green infrastructure in supporting climate change mitigation, while leaving great room for improvement and a need to extend tree cover in the city to ensure a more equal distribution of C storage.

6.2 Variability of Carbon storage within the city and between UF types

Urban carbon storage is subject to a wide range of local drivers, which are highly site and scale dependant. These may include age structure of trees, tree density, management practices, species composition as well as land use (Zhao *et al*., 2010). It is clear across a wide range of studies that urban carbon storage is characterised by large variation between different Land uses (see (Hutyra *et al*., 2011; Strohbach & Haase, 2012; Nowak *et al*., 2013), due to the heterogeneity of urban systems. This is reflected in Johannesburg, in the large differences observed in aboveground carbon storage between nature reserves and parks, as well as the high spatial variation in the city. This variation across land use is evidenced in numerous cities around the world, including Barcalona, Spain (Chaparro & Terradas, 2010) and Leipzig, Germany where carbon storage ranged from 15.3-96.7 and 6.8-98.5 t/ha, respectively (Chaparro & Terradas, 2010; Strohbach & Haase, 2012).

While the delineation of land uses varies across different studies, some research used similar greenspace delineations as this study, allowing for comparison. In comparing the carbon storage of parks and gardens in Johannesburg to similar land uses in published literature various similarities and discrepancies are revealed. For instance, parks in extreme northern latitudes exhibit far less carbon than those in Johannesburg. Trees in constructed parks in Helsinki, Finland stored 22-28.1 t/ha (Lindén *et al*., 2020), compared to 98.5 t/ha in Johannesburg. The large differences may be explained by the lower tree density in Helsinki (150 trees/ha) as compared to Johannesburg (299 trees/ha), as well as the large proportion of small trees (<15 cm DBH) found in constructed parks in Helsinki (Lindén *et al*., 2020). Meanwhile, a broader range of carbon storage was observed in public parks in Muscat, Oman, ranging from 103.43-313.74 t/ha (Amoatey & Sulaiman, 2020), owing to a high proportion of very large trees. 'Green urban areas' in Leipzig, Germany stored less than Johannesburg parks with 46.4 t/ha, while urban forest and woodland areas stored similar amounts, ranging from 73.2-98.5 t/ha (Strohbach & Haase, 2012). This parallels findings from American cities such as Los Angeles and Sacramento, where open green space was found to store 38 and 75.6 t/ha, respectively (McPherson *et al*., 2013).

Residential areas in Leipzig, including single or semi-detached houses, show carbon storage levels akin to Johannesburg's gardens, with 63.8 t/ha of carbon stored (Strohbach & Haase, 2012). This resembles findings from American cities like Los Angeles and Sacramento, where low-density residential spaces demonstrate carbon storage densities of 40.1 and 71.3 t/ha, respectively (McPherson *et al*., 2013). In contrast, gardens of Boston stored far less than Johannesburg, with 32.8 t/ha (Raciti *et al*., 2014). Investigations into the carbon storage of roadside trees displayed similar variation across different cities, with values ranging from as low as 13.9 t/ha in Beijing, China (Zhao *et al*., 2016) to as high as 485.15 t/ha in Muscat, Oman (Amoatey & Sulaiman, 2020). Johannesburg's roadside carbon density (67.84 t/ha) corresponds to findings in Korea, where the largest street tree species stored 69.7 t/ha, while the average of all species assessed was lower, at 24.9 t/ha (Yoon *et al*., 2013).

Tree density and size are key drivers of carbon storage (Nowak *et al*., 2013). These structural attributes may explain the significant differences between carbon storage and biomass between forest types in Johannesburg, at both the plot and tree level. Carbon storage in Johannesburg ranged from 39-98 t/ha between urban forest types. Nature reserves (NR) in Johannesburg held the lowest carbon stocks of the four UF types despite having the highest tree density per plot, due to the predominance of shorter and smaller (DBH) trees. NR had a significantly smaller average DBH than all other UF types, with nearly 40% of the sample having a DBH below 10 cm. Meanwhile, roadside (RS) plantations had the lowest density per plot, but were frequented by large, high DBH and tall trees, resulting in a higher carbon storage. Despite having marginally smaller trees than RS, parks held large trees and had greater density than RS, resulting in the highest carbon storage of 98.5 t/ha (Table 8). Similar to nature reserves, gardens had a relatively high density, but held small trees, resulting in the second lowest carbon storage. The carbon storage in nature reserves was expected to be low, being dominated by smaller indigenous woodland and grassland tree species, which have generally found to store lower amounts of carbon (Mills *et al*., 2005; Shackleton & Scholes, 2011; Ribeiro *et al*., 2013). However, it should be considered that the soil organic carbon pool, not assessed in this study, is likely very significant in these areas (Mills *et al*., 2005; Bai & Cotrufo, 2022).

The results suggest that tree size and density are the two primary factors which govern the amount of carbon stored in aboveground biomass in urban green spaces (Nowak & Crane, 2002; Davies *et al*., 2013; Lindén *et al*., 2020). Additionally, tree girth is proportional to age (Schmitt-Harsh *et al*., 2013), making the age of trees another strong determinant of above-ground carbon storage. Several South African studies have evidenced this relationship, showing that older, more mature trees have greater biomass (Dube & Mutanga, 2015; Hlatshwayo *et al*., 2019; Abutaleb *et al*., 2021). Internationally, Linden *et al*. (2020) showed that time has a positive effect on the carbon stocks in urban parks. This relationship explains some of the key differences between UF types found in Johannesburg. Parks and roadside trees were dominated by older stands with larger trees (Figure 6), resulting in higher carbon storage capacity than nature reserves and gardens (Table 8). Furthermore, the prevalence of very large, mature trees throughout Johannesburg drives the high overall carbon storage of 2.4 Mt C.

My findings indicate the disproportionate role played by large trees in contributing towards the carbon stock, which has been reflected in other urban areas (Nowak, 1994; Nowak & Crane, 2002; Stoffberg *et al*., 2010; Davies *et al*., 2011; Nowak *et al*., 2013; Schmitt-Harsh *et al*., 2013; Lindén *et al*., 2020). The top 10% of trees in terms of size (girth) in Johannesburg contributed 53.67% of the carbon stored amongst sampled individuals, and carbon storage increased as DBH increased. Similarly, taller trees contributed disproportionately more biomass relative to shorter trees in Johannesburg. While small trees store much less carbon than larger individuals, they frequently made up a large proportion of the trees in urban areas (Davies *et al*., 2011; Hutyra *et al*., 2011; Liu & Li, 2012; Lindén *et al*., 2020). This is partially true for Johannesburg's urban forest, where small individuals make up a large proportion of both NR and garden trees (Figure 6).

Anthropogenic factors also play a role in driving urban carbon storage, as the structural attributes of urban forests are moulded by land development practices, historic species preference, conservation efforts and tree planting practices (Berland, 2012). Johannesburg's long history of greening and gardening contributes to the current, significant carbon stock and explains the abundance of large, old trees standing in the city, particularly in Northern Johannesburg, where numerous green spaces were built and preserved as the city continued to expand and urbanise (Turton *et al*., 2006; GCRO 2013; Schäffler & Swilling, 2013). For example, parks like Zoo Lake and Emmerentia Park were built in the early 1900s and have been maintained to this day. Additionally, both parks and roadsides are characterised by mature tree stands composed of mainly exotic species such as Jacarandas, Pines and Eucalyptus, which have been planted in the city for more than a century (Turton *et al*., 2006; Schäffler & Swilling, 2013). This history contextualises Johannesburg’s high tree cover and abundance of large, old trees which drive the total carbon storage of 2.4 Mt C.

A high proportion of large trees in urban areas is a common phenomenon, accounting for some findings where urban forests store more carbon than local, natural forests (Nowak & Crane, 2002; Nowak *et al*., 2013). This aligns with my findings that Johannesburg's urban forest stores similar amounts of carbon to natural South African forests and more than grassland and savanna woodlands, despite having generally lower tree density. The large carbon storage and sequestration potential of urban trees can be attributed to several factors. Firstly, urban vegetation generally has lower density than in natural ecosystems, resulting in fewer space constrictions, allowing for open crown structures which enhance the photosynthetic capacity of the plant (Nowak & Crane, 2002). Secondly, fast-growing species are frequently planted in urban areas to achieve the intended aesthetic or landscape effect in minimal time (Chen, 2015). Finally, trees in private gardens are frequently supplied with optimal resources for growth, in the form of watering or nutrients through fertiliser (McHale *et al*., 2009).

The distribution of urban trees is largely dependent on the history and intensity of management of these green spaces (Dhyani *et al*., 2021). The four UF types assessed in Johannesburg had significantly different densities, tree sizes and species compositions (see Chapter 3). It is likely that the history and management of these trees played a key role in determining their current structure and composition. In the case of roadside plantations and parks, the management of local authorities and urban planners would have influenced their structure through formal planning processes. In the case of gardens, individuals are key actors, whose decisions affect the diversity and structure of garden trees. Nature reserves, however, remain as remnant vegetation and therefore follow the characteristic structure and diversity of indigenous savanna grassland and woodland ecosystems (GCRO, 2013). The role of human management is reflected in the high proportion of exotic species in all forest types except for nature reserves (Figure 7), mirroring a common trend in cities around the world (Nowak *et al*., 2011; Nagendra & Gopal, 2011; Dhyani *et al*., 2021) and explaining the high diversity found in Johannesburg. In general, urban forests represent a mixture of remnant, native tree species which existed pre-development, alongside exotic species introduced by the humans in the city (Nowak *et al*., 2011). Johannesburg is dominated by exotic species (Figure 7), with a long history of their introduction. The exception lies in nature reserves, where indigenous species prevail, emphasising the importance of their conservation. Johannesburg, similar to Los Angeles (Nowak *et al*., 2011), is conducive to the growth of exotic trees from a wide range of origins, resulting in extremely high diversity (Table 5), which may exceed the diversity of surrounding natural landscapes. The highest diversity in Johannesburg was found in gardens, due to the human influence and introduction of exotic and ornamental species, similarly to gardens in the US (Schmitt-Harsh *et al*., 2013).

The results revealed great spatial variation in carbon storage across the city of Johannesburg, with much higher carbon densities in the north as compared to the south (Figure 9). Given that urban carbon storage is generally a function of tree cover (Nowak *et al*., 2013), it is unsurprising that this variation was driven by the unequal distribution of tree cover across the city. Northern Johannesburg maintains high tree cover, coinciding with large carbon stocks, while much of the south has minimal tree cover, and consequently low carbon stocks. In particular, the north-central and north-eastern suburbs are characterised by dense tree cover, with heavily treed private gardens and numerous, connected green spaces. Additionally, the trees in the northern regions were generally found to be larger and older, consistent with the findings of previous studies in the city (Abutaleb *et al*., 2021).

6.3 The role of exotic species in delivering ecosystem services

Similar to prior research undertaken in urban areas in South Africa (O'Donoghue & Shackleton, 2013; Newete *et al*., 2022) and internationally (Dhyani *et al*., 2021; Gülçin & van den Bosch, 2021), the majority of sampled species in Johannesburg were exotic (Figure

7). This is a common trend within urban forests around the world, given that the primary drivers of biodiversity in urban systems are anthropogenic.

In the case of Johannesburg, the prevalence of exotic species relates to the city’s colonial and mining history (Turton *et al*., 2006; Schäffler & Swilling, 2013; GCRO, 2013). During the gold rush of the 19th century, large scale tree planting was undertaken to mitigate the high levels of dust, as well as to provide timber to meet the high demand for mining poles (Turton *et al*., 2006). To meet the demand, fast growing species such as eucalyptus and black wattle were planted *en masse*. Massive timber plantations, consisting of non-indigenous species, were supported by investment from the mining industry as well as private landowners, after indigenous tree species could no longer met the demands for mining timber. Along with this tree planting boom came the introduction of street trees familiar to the colonials, such as oaks and London planes (Turton *et al*., 2006). In addition, wide-scale suburban development occurred alongside the rapid expansion of the city. These suburbs were also subject to the tree planting boom evidenced in the naming of suburbs such as 'Forest town' and 'Parkview' (Turton *et al*., 2006; GCRO, 2013).

Ultimately, these activities had a profound effect on the landscape of Johannesburg, transforming a natural grassland into the characteristic, urban forest which is seen today. Thus, the seemingly incongruent situation of an urban forest occurring in a natural grassland is explained by the wide-scale planting of exotic trees during the colonial beginnings of the city (Schäffler & Swilling, 2013). Many of these introduced species continue to dominate the urban forest of Johannesburg, reflected in my own findings and in prior research in the city (Abutaleb *et al*., 2021b; Newete *et al*., 2022). My sampling indicated that exotic species make up more than 75% of individuals in all forest types except nature reserves (Figure 7).

Interestingly, my results show that these exotic trees provide exceptionally high ecosystem services. Most of the carbon storage in Johannesburg's urban forest is provided by large, exotic trees (Chapter 3, Figure 8). Additionally, the regions characterised by the highest supply of all three UES, the northern suburbs, are dominated by exotic species (Figure 16). My results mirrors the findings of prior studies, which found high carbon storage and sequestration amongst exotic species in South Africa (O’Donoghue & Shackleton, 2013; Dube & Mutanga, 2015) and other African countries (Woldegerima *et al*., 2017; Moussa *et al*., 2019). This relates primarily to the large size of exotic tree species in the city, given that large trees are crucial providers of ecosystem services (Schmitt-Harsh *et al*., 2013; Nowak *et al*., 2013), particularly in the urban context (Schlaepfer *et al*., 2020). Given that exotic species present numerous threats to ecological systems, their management is complicated by the positive socio-economic or ecological benefits they provide (Roy *et al*., 2012; Zengeya *et al*., 2017; Abutaleb *et al*., 2021b). In turn, the high carbon storage amongst exotic tree species in Johannesburg brings about several management implications.

Exotic tree species are known to present a range of negative impacts and disservices, in conjunction to their beneficial ecosystem services (Almas & Conway, 2016; Potgieter *et al*., 2017; (Gaertner *et al*., 2016; Le Maitre *et al*., 2020). In the Johannesburg context, the city's largely exotic urban forest has attracted criticism on account Johannesburg's lack of a natural watershed and associated water challenges in relation to the water-intensive exotic species which make up its urban forest (Schäffler & Swilling, 2013). The forest has therefore been criticised for inducing additional hydrological pressures due to its erection in a natural grassland.

Thus, the value of exotic tree species in delivering ES in Johannesburg, contrasted with the potential negative impacts of these same species, presents a predicament for the management of these species into the future. Currently, the management of alien species in South Africa is informed by the National Environmental Management: Biodiversity Act (Act 10 of 2004) (RSA, 2004; Zengeya *et al*., 2017). This act prevents the future planting of a wide range of alien tree species which exist in Johannesburg. However, the management of species which provide both benefits and negative impacts are complex, with possible disagreement between stakeholders regarding the extent of their positive or negative impacts (Zengeya *et al*., 2017). Some species have no negative impact, but several have positive impacts and are increasingly regulated on a per area basis. This research further supports the positive impacts of for example, *Jacaranda mimosifolia*, *Eucalyptus grandis* and other large urban trees with minimal destructive effects (see Zengeya *et al*., 2017). The more complex issues relate to species which have both positive and negative impacts.

Research on the trade-offs between benefits and costs related to exotic trees has already begun in South Africa (See Zengeya *et al*., 2017), with urban specific case studies having been conducted in the city of Cape town (Gaertner *et al*., 2016). It has been suggested that existing frameworks for the management of invasive species in South Africa are often developed for rural or natural areas and therefore are inadequate for application in urban areas (Gaertner *et al*., 2016). Another key challenge for implementing invasive species management in South Africa are the conflicting views amongst stakeholders or general conflicts of interest, whereby for example, goals of biodiversity conservation and ecosystem service delivery may be contradictory. Research in the city of Cape Town has suggested that management frameworks need to incorporate divergent stakeholder perceptions and that some alien species may need to be accepted in the urban context (Gaertner *et al*., 2016). A similar sentiment has been expressed in the European context, that the general exclusion of non-native species is not suitable, as native tree species may be too limited to provide the ecosystem services and resilience required in urban settings (Sjöman *et al.*, 2016). Thus, the dangers of a "native only" approach in the European urban context have been elucidated (Sjöman *et al*., 2016).

My research supports this viewpoint, by demonstrating the immense carbon storage capacity of alien tree species in Johannesburg. This capacity is extremely important in the context of South Africa's climate change mitigation efforts and should not be dismissed due to broad alien species policies. Rather, the benefits provided by these species should be considered, within the context of altered urban ecosystems, and should be contextually assessed in relation to the disservices they provide at the species level. Therefore, future research is required to develop a more comprehensive understanding of the trade-offs between costs and benefits within urban species in Johannesburg, accounting for a wide range of stakeholder perceptions. The city should be cautious in applying overly broad and prescriptive restrictions on alien species planting, which may result in the loss of contextually important ecosystem services. It is therefore plausible that some species which do not have major negative effects, but provide high levels of ecosystem services, should be tolerated in urban systems, even if they are exotic. The decision of whether to tolerate a particular species is context-dependent, contingent upon its current and future potential to inflict significant impacts, as well as its perceived value (Gaertner *et al*., 2016). This acknowledgment aligns with the understanding that certain ecosystems have undergone such extensive transformations that restoring them to their original historic condition is not feasible (Hobbs *et al*., 2014; Corlett, 2015; Martin *et al*., 2014).

The aim of this research was not to address invasive species management, and further research is required to develop a framework more comprehensively for managing invasive species in the context of Johannesburg. However, we raise the question- given the clear evidence of high ES delivery by alien species in Johannesburg, which are critical in the context of climate change adaptation and mitigation, should these species be broadly excluded and removed from the city? We suggest that novel ecosystems may need to be managed and designed for the optimisation of functionality as opposed to restoration of historic conditions. In the context of the urgent need to mitigate and adapt to climatic change, caution is required in ensuring that the intrinsic value of conservation does not interfere with crucial climate change interventions, particularly in the already transformed and novel urban systems. To achieve optimal UES delivery and green space planning, both ecosystem services and disservice's need to be considered- the trade-offs and synergies need to be assessed fully and further research is required to determine which species and at what levels the trade-offs are too large. It should however be considered that despite indigenous species being less effective at storing carbon, they provide important benefits and services which were not considered in this UES assessment, for instance promoting native biodiversity. In the context of the findings of this study, it is advised that the COJ proceed with caution on this issue and that the benefits of the alien species are fully considered when determining future responses.

6.4 Spatial variation in Cooling and Runoff services in Johannesburg

In addition to carbon storage, green spaces in Johannesburg provide cooling and runoff retention services (Figure 16). While these two services were not evaluated in the field at the same micro-scale as C storage was, the broader trends can still be contextualised. Broadly, these services follow similar trends to C storage, with high supply in the densely vegetated north and low supply in the south of Johannesburg.

Green spaces, including parks, trees, and waterbodies, are acknowledged as an effective strategy for reducing surface and air temperatures by minimising solar radiation through shading, and by cooling surfaces through evapotranspiration (Santamouris, 2015; Hardin & Jensen, 2007; Bowler *et al*., 2010; Nastran *et al*., 2019; Skelhorn *et al*., 2014). This is reflected in Johannesburg, where cooling services coincide heavily with the distribution of green spaces across the city (Figure 14). The widespread provision of cooling in the north is driven by the numerous, connected green spaces and dense tree cover in the region. This relates to the key role played by trees in delivering cooling services (Gillner *et al*., 2015; Lee *et al*., 2016; Lindén *et al*., 2016). Thus, cool spots are observed in central northern Johannesburg, where there is the greatest tree cover. Meanwhile, regions of low supply generally occur in southern Johannesburg, which has minimal tree cover (Figure 13). Hotspots, with elevated land surface temperatures, were identified in areas in and around Alexandra and Soweto (Figure 15), supporting the findings of previous research in the city which identified similar hotspots (Souverijns *et al*., 2022; Freimond and Schwaibold, submitted). The southernmost and westernmost regions also displayed heightened surface temperatures, in conjunction with a low supply of cooling. These areas with minimal supply of cooling services generally have low tree cover and high proportions of bare ground or impervious surfaces. Alongside the absence of evaporative cooling and shading from vegetation, the highly impervious surfaces increase surface heat retention. These surfaces have far greater thermal storage capacity than natural surfaces, as well as reduced albedo (Golden and Kaloush, 2006; Grimm *et al*., 2008; Stone *et al*., 2010). In turn, large amounts of solar radiation are absorbed, stored, and re-radiated into the urban environment by artificial surfaces and materials (Rizwan *et al*., 2008).

The value of cooling services in urban systems is high, and is expected to increase into the future, as the UHI effect is exacerbated by climate change impacts, particularly increasing global temperatures and more frequent heatwaves (IPCC, 2021). In this context, and in conjunction with the major effects of the UHI on human health and the environment (Kleerekoper *et al*., 2012; Heaviside *et al*., 2016; Li *et al*., 2019; Grimm *et al*., 2008), the provision of cooling services is critically important. In the local context, climate change projections indicate likely increases in temperatures and the frequency of heatwaves in Johannesburg (City of Johannesburg, 2021). The city's climate action plan emphasises the danger of rising temperatures to human health and identifies the need for additional cooling (City of Johannesburg, 2021). Projections of temperature increases in conjunction with the UHI effect predict increases in temperature up to 2.7 ^{o}C by the year 2050, posing a serious risk to human health. This makes the cooling supply identified in Johannesburg immensely valuable. The high supply of cooling in the highly vegetated northern regions of Johannesburg (Figure 13) aligns with the frequent findings that urban greenspaces mitigate the UHI and provide cooling services (Busato *et al*., 2014; Bartesaghi Koc *et al*., 2018; Chen *et al*., 2022; Emmanuel & Loconsole, 2015; Lee *et al*., 2016; Lin *et al*., 2016; Marando *et al*., 2019). Our findings also follow the trend of UHI formation in cities, with several heat islands observed in south-western and central Johannesburg, in addition to the region of Alexandra (Figure 15).

The distribution of runoff retention services showed some similarities to carbon storage and cooling with high supply in the north but greater variability in southern Johannesburg (Figure 16). This relates to the greater dependency of runoff retention services on surface and soil characteristics relative to the other two services (Quagliolo *et al*., 2021). Runoff retention has high supply in northern Johannesburg where extensive greenspace exists. Vegetated areas have a higher infiltration capacity, which reduces the amount of runoff that occurs following a precipitation event (Villarreal & Bengtsson, 2005; Grey *et al*., 2018). The high supply of this service is driven by interception of rainfall by canopy cover, as well as the soil beneath vegetation which enables the storage of water in pore spaces (Villarreal & Bengtsson, 2005; Berland *et al*., 2017). Meanwhile areas of low supply in southern Johannesburg are generally characterised by high proportions of impervious surfaces. Impervious surfaces diminish the capacity of the ground to infiltrate water, resulting in greater volume and velocity of runoff (Berland *et al*., 2017; Hemmati *et al*., 2022). This contributes to the low runoff retention supply in the highly impervious CBD and south-western regions of Johannesburg.

Another factor which influences the runoff retention in Johannesburg is soil type. The hydrological soil group is a key input into the SCS model used to determine runoff retention in Johannesburg (see Chapter 4). Northern Johannesburg is dominated by soil group B, characterised by a moderate infiltration rate, while southern Johannesburg is covered by soil type C which has lower infiltration rates. An additional contributor to the pattern of runoff retention is the presence of waterbodies. Despite enhancing cooling, waterbodies diminish runoff retention due to the modelling assumption that they are fully inundated and thus do not retain any rainfall, explaining low runoff retention in the south-western region surrounding rivers. Ultimately, greenspace distribution, urban surface characteristics and soil type all play a role in dictating the delivery of runoff retention services in Johannesburg.

Contemporary urban systems face urban stormwater management issues, due to the high proportion of impervious surfaces which diminish infiltration capacity (Berland *et al*., 2017; Quagliolo *et al*., 2021), in conjunction with climate change impacts which drive more frequent and intense rainfall event which leading to pluvial flooding (Rosenzweig *et al*., 2019; IPCC, 2021). In this context, the value of runoff retention services in Johannesburg is extremely high. Our results show that a total 60.2 million m^3 of runoff is retained by the greenspace and soil, under a high intensity 50 mm rainfall event, with an average of 366 m^3/ha retained across the city (Table 12). Numerous wards in northern Johannesburg benefit greatly from runoff retention services, retaining 86% of rainfall, even under a 50 mm event (Figure 12). In the context of urban vulnerability to flooding, and increasing flood risks under climatic change, this service is crucial in ensuring the safety of urban dwellers and building resilience against climate impacts. The contextual importance of this service related to the high levels of imperviousness across Johannesburg, in conjunction with the occurrence of extreme highveld thunderstorms, which result in dangerous flash flood events (Schäffler & Swilling, 2013). As a result, the city's climate action plan identified the building of climate resilience and implementation of adaptation strategies as key goals, with a particular focus on flood regulation strategies (City of Johannesburg, 2021). Our results show that the GI in Johannesburg plays a key role in regulating runoff and retaining runoff retention during heavy rainfall events.

Various synergies and trade-offs between services emerged from our results. Due to the importance of tree cover in delivering both cooling and carbon storage services, the spatial distribution of both is markedly similar (Chapter 5, Figure 16). Some minor differences occur in southern Johannesburg, where there is limited tree cover and the only major sources of cooling in the south of Johannesburg are the two rivers and the greenspace associated with Klipriviersberg nature reserve (Figure 14). Blue spaces such as rivers generate latent heat flux through evaporative cooling, reducing surrounding temperatures (Zuo *et al*., 2016; Wu & Zhang, 2019).

Beyond those already addressed, other trade-offs between services occur in the wards of Johannesburg, although they are not prominent, as the general pattern of supply is similar across all three services. For example, rivers represent a trade-off between carbon storage and cooling, with high supply of cooling services but no storage of aboveground carbon. This was evident in the more consistent distribution of cooling in the southern wards, compared to C storage (Figure 16). Bare ground and grasslands represent a trade-off between runoff retention and the other two services. For example, the north-west corner of the city showed high runoff retention supply, but low supply of the other two services due to the capacity of grasslands and bare ground to retain runoff, but their lack of aboveground C storage and high heat retention. While trade-offs do occur, the three services assessed in Johannesburg are largely characterised by synergies, with the multi-functional delivery of ecosystem services from urban green spaces, particularly in northern Johannesburg (Figure 1). However, our results show that these services are not supplied equally across the city.

6.5 Green spaces and UES inequality

While the value of the UES in Johannesburg is high, a key trend which emerged from the results was that all three services are unevenly distributed across the city (Figure 16). All three services show distinct spatial variation, with higher and more consistent supply in the north, but lower and more variable supply in the south. The low tree cover in southern Johannesburg results in extremely low C storage at the ward level (<3.71 t/ha), compared to high supply in the north (23-35 t/ha) (Figure 9). Runoff retention is also unequally distributed, with numerous southern wards retaining 110 m^3/ha less than the best performing wards in the north (Figure 12). Furthermore, runoff increases disproportionately in wards with low supply as storm intensity increases. This has major implications in the context of climatic change, which is expected to increase the severity and frequency of extreme precipitation events (ICPP, 2021; City of Johannesburg, 2021).

There are also clear disparities in cooling capacity between the northern and southern regions of the city (Figure 13), resulting in the formation of hostpots in regions of low supply (Figure 15). This too poses a major health risk to the inhabitants of these areas, especially when considering rising global temperatures and increased heatwaves associated with climatic change (ICPP, 2021). Although the south maintains most wards with low supply, areas in the north such as Alexandra and the north-western corner of the city show reduced supply across all services, despite being surrounded by regions of high supply (Figure 16). Overall, Alexandra, the south-western and southern-most regions of the city are consistently undersupplied across all three services.

My findings evidence that the 'luxury effect', "a pattern of higher biodiversity in affluent neighbourhoods" (Leong et al., 2018), extends to UES supply in Johannesburg, South Africa. The wealthier northern-central regions of Johannesburg show a markedly higher supply of ecosystem services, compared to the poorer southern regions. These findings align with prior research in the city of Johannesburg (Schäffler & Swilling, 2013; GCRO, 2013; Abutaleb *et al*., 2021), which highlighted the inequality in the distribution of tree cover and greenspace across Johannesburg. Broader, national-scale research in South Africa found distinct discrepancies in green space accessibility and abundance between high and low-income areas in South African cities, with an abundance of green space in areas where previously advantaged racial groups reside (Venter *et al*., 2020).

This relates to the disparities in the development and provision of green infrastructure in South Africa due to the legacy of apartheid (Venter *et al*., 2020). The geographical areas in which communities were previously disadvantaged are referred to as 'Townships' (Stoffberg *et al*., 2010). These areas tend to be poorly vegetated because of the legacy of apartheid and its associated history of politically driven urban development strategies (Stoffberg *et al*., 2010; Abutaleb *et al*., 2021). Consequently, Johannesburg's urban forest which provides many crucial services, is immensely unequal in its distribution, being centred in the city's historically wealthy northern suburbs (Schäffler & Swilling, 2013). This research adds to prior findings by demonstrating that these spatial inequalities in tree cover, lead to inequalities in the supply of important ecosystem services.

African cities are already recognised as being particularly vulnerable on account of climatic severity and structural and socio-economic vulnerabilities (Du Toit *et al*., 2021; Shackleton *et al*., 2021). This will be exacerbated by the impacts of climatic change, to which African and global south countries are disproportionately vulnerable (Jalayer *et al*., 2015; Scoville-Simonds *et al*., 2020). In the context of Johannesburg, high rates of poverty, 45.22% and unemployment, 31.5% (City of Johannesburg, 2021), alongside other adverse socio-economic conditions, render much of the population vulnerable to climate change, with limited capacity for adaptation (Wright *et al*., 2021). Health risks posed by the UHI, and flooding will be aggravated by socio-economic conditions. Moreover, structural vulnerabilities will add to this suite of challenges, due to the prevalence of housing poverty in South Africa (Wright *et al*., 2021; Naicker *et al*., 2017). For example, a significant portion of South African residents reside in dwellings constructed from corrugated iron or tin, characterised by inadequate ventilation and heightened indoor temperatures of up to 5 ^{o}C (Naicker *et al*., 2017), rendering them particularly susceptible to the UHI effect, heatwaves, and escalating temperatures (Naicker *et al*., 2017). These factors, in conjunction with the established vulnerability of urban systems to flooding and heat stress (Grimm *et al*., 2008; De Sherbinin *et al*., 2007; Herslund *et al*., 2016; Revi *et al*., 2014), create a serious challenge for the city of Johannesburg in the context of the unequal distribution of ecosystem services.

My findings show that the regions of lowest supply of ES coincide with the poorest populations in Johannesburg, characterised by townships and informal settlements such as Alexandra, Soweto, Ivory Park and Orange farm. Furthermore, these areas have the highest population density of any regions in the city. In particular, the southernmost region, the south-western region of Soweto, the CBD, Alexandra, and the north-west corner of the city stand out as regions of critical undersupply and are simultaneously the most densely populated areas in the city (Todes *et al*., 2018; Abrahams & Everatt, 2019). The areas in and around Soweto and Diepkloof in the south-west, for example, house over a third of the city's total population (Abrahams & Everatt, 2019). The culmination of low ES supply with high population densities, structural, and socio-economic vulnerabilities make these regions extremely vulnerable to both flooding and heat stress. These urban challenges pose a major risk to the safety of the inhabitants of these areas, especially under climate change. These areas must be targeted by authorities to improve ES supply and build resilience against climate change impacts. My study provides evidence that the socio-economic inequalities, related to the political history of South Africa, extend to ecosystem service and ecological inequalities, through unequal supply of runoff retention, C storage and cooling ecosystem services.

Understanding the variation in spatial distribution of ES in urban systems is crucial for addressing issues of unequal access to these services (Ernstson, 2013; Kremer *et al*., 2016). By mapping ES supply at the ward scale, our findings can be used to accurately inform authorities and decision makers of which areas need to be targeted with greening projects or improvements. Our quantification and mapping of ES supply can aid planners to identify priority areas for intervention, where large populations are inequitably undersupplied.

6.6 Implications for policy and decision-making

In the context of increasing extreme weather events, in conjunction with the vulnerability of cities to flooding and heat events (De Sherbinin *et al*., 2007; Seto *et al*., 2014; Dodman *et al*., 2022), the sustainability and safety of cities are largely dependent on addressing these challenges. Cooling and runoff retention services make critical contributions to building urban resilience and adaptive capacity (Gómez-Baggethun & Barton, 2013). The cooling and runoff services in Johannesburg effectively meet the challenges of flooding and heat stress, which need to be tackled *in situ* at the time and place they arise (Derkzen *et al.*, 2015). The preservation of these ES serves as a primary adaptation strategy which can be used to support the adaptive capacity of the city (IPCC, 2018). The maintenance of these services in the urban context is a crucial pathway to building a sustainable city and ensuring the health and well-being of urban dwellers in the face of climatic change (Haase *et al.*, 2014; Liu & Russo, 2021). Johannesburg's climate action plan highlights the need to build climate resilience and implement adaptation strategies, with a focus on flood and heat regulation (City of Johannesburg, 2021). The UES assessment results can inform local authorities of the value and role of these services in mitigating flooding and heat stress. In turn, their value can be incorporated into urban planning, policy, and decision-making contexts (Dhyani *et al.*, 2021; Farrugia *et al.*, 2013; de Manuel *et al.*, 2021). By mapping UES supply across administrative wards, their value is made spatially explicit and priority areas can be identified to support green infrastructure planning (Derkzen *et al.*, 2015; de Manuel *et al.*, 2021). On the other hand, while our results show the value of UES in Johannesburg, they also indicate the distinct inequalities in the supply of these services across the city (see section 6.5). Several wards are critically undersupplied and must be targeted with interventions and greening programmes to bolster resilience. By making our results spatially explicit, priority areas for targeting with greening interventions are clearly defined at a scale conducive to decision-making by authorities.

The results of our UES assessment, and the comparison of supply between services, highlighted the various synergies which exist between services in Johannesburg, although some trade-offs do occur. Generally, wards with high supply of one service, had a high supply in all three. Carbon storage and cooling services have distinctly similar spatial distributions, suggesting a strong synergy (Figure 16). All three services are simultaneously optimally supplied in the northern-central region of the city by the extensive and diverse green spaces that characterises the region. This region elucidates the multi-functionality of UES, a key element of the green infrastructure framework (Lovell & Taylor, 2013; Hansen & Pauleit, 2014; Monteiro *et al.*, 2020), which has great value in an urban planning context, given its unique capacity to meet multiple municipal goals simultaneously (GCRO, 2013). This is indispensable in the African context, where countries are economically constrained, with limited capacity for investment into urban governance and a range of socio-economic issues competing with environmental issues for priority (Shackleton *et al.*, 2021). The low tax base and per capita budget for implementing urban planning and ecosystem management in Africa (Shackleton *et al*., 2021), suggests that the cost-effective delivery of multiple services from singular green infrastructure investments which address more than one problem, may be of critical importance. Moreover, this study did not exhaustively analyse the services provided by green infrastructure in Johannesburg, suggesting that the multi-functionality may extend beyond these three services. The trade-offs and synergies identified in this study can be used to maintain and enhance the multi-functionality of green infrastructure in Johannesburg and optimise ecosystem service delivery.

The results showed that Johannesburg's urban forest stores 2.4 Mt C which may have implications for Johannesburg's climate change mitigation strategy. South Africa has a key role to play in curbing global CO_2 emissions, given their role as the 13th highest emitter worldwide and the largest emitter in Africa (Crippa *et al*., 2020). As the country seeks to reduce emissions, it is apparent that no singular strategy will be sufficient (O'Donoghue & Shackleton, 2013). Instead, a diversity of strategies will need to be employed, spanning environmental, social, and economic spheres. While there are ongoing efforts to reduce emissions through renewable energy and economic restructuring (TPRSA, 2023) which are necessary, this study indicates that tree planting in urban areas can contribute significantly to the strategies for reducing national emissions and meeting the required contribution of the country to mitigating global climate change. Tree planting has been practiced extensively in South African cities and towns, but the value and costs of these practices remain understudied and poorly quantified (Shackleton, 2006; GCRO, 2013). In particular, the role of urban forestry in sequestering and storing carbon has been understudied (Stoffberg *et al*., 2010). This study provides the first comprehensive baseline assessment of C storage in Johannesburg's urban forest, which can serve decision makers and stakeholders in guiding tree planting strategies and land development policy to mitigate climatic change.

Given the ongoing, rapid urbanisation and industrialisation occurring in South Africa and expected future increases in Africa as a whole (Pauleit *et al*., 2021), it is expected that associated CO_2 emissions will continue to increase in coming decades. This will further increase the importance of maintaining and enhancing carbon sinks. The quantification of C storage within the urban forest of Johannesburg is a crucial first step in assessing the potential and actual role which urban forests play in reducing atmospheric CO2, thereby mitigating climatic change (Nowak *et al*., 2013; Liu & Li, 2012; Nowak & Crane, 2002). The findings of this study serve as an initial foundation for the integration of urban green infrastructure into strategies for climate change mitigation in South African cities.

South Africa is a signatory to both the United Nations convention on climate change (UNCCC) and its Kyoto protocol, which was ratified in 2002, indicating a national commitment to GHG reductions. As a result, South Africa is legally obligated to report GHG sources and sinks through the Kyoto protocols. The GHG inventory is crucial in informing and measuring South Africa's nationally determined contribution (NDC) to emissions reductions under the UNCCC. However, despite having published their 8th national GHG Inventory report, South Africa does not fully account for Urban C sinks based on collected data (see DFFE, 2022). A similar undervaluation of urban C sinks has been observed in other cities around the world (Dorendorf *et al*., 2015; Davies *et al*., 2011). Our findings indicate that C storage in urban systems is underestimated in the national budgets. Our findings may improve estimation of C storage in South African cities. For most of Johannesburg 10 t/ha is assumed in the national budget. Meanwhile this study shows that AGB in Johannesburg can store up to 98 t/ha, prior to the inclusion of the soil organic carbon pool. Urban areas are sources of uncertainty in determining AGB in South Africa’s national inventories (DEFF, 2020), making our results valuable for national and local carbon budgets and enabling a more accurate inclusion of urban system C sinks.

From a policy perspective, the climate action plan adopted by the city of Johannesburg aims to align with the targets of the Paris agreement and has set an ambitious target of achieving net-zero emissions by the year 2050 (City of Johannesburg, 2021). This study highlights the important role that greening can play in achieving these goals and in Johannesburg's climate mitigation policy. My results have shown that Johannesburg's urban forest stores 2.4 Mt C, equivalent to 8.8 Mt CO_{2e}. This is a significant amount, equivalent to 42.26% of Johannesburg’s annual emissions (City of Johannesburg, 202). It is clear that urban greening can play an important role in urban climate change mitigation policies (Weissert *et al*., 2014) but the potential carbon storage of urban areas is yet to be appropriately integrated into policy in South Africa or into greening institutions. At the municipal scale, a range of operational and administrative measures can be implemented to support climate change mitigation (Chen, 2015). These may include, but are not limited to, the creation of new green spaces, the maintenance of existing green spaces, preserving natural land and planting fast growing tree species which have extended life spans and mature into large individuals. South Africa faces various challenges relating to the lack of finances available to support the preservation or construction of urban green spaces. Consequently, financial policies which provide incentives for public or private investment into green infrastructure could be pivotal in ensuring sustainable development in South African cities.

My results may hold further important policy implications for municipal spatial planning instruments, such as municipal spatial development frameworks (SDF). Spatial planning and development in Johannesburg is guided by the City of Johannesburg's metropolitan municipalities SDF, which provides guidance to the general spatial distribution of existing and desired land uses (See City of Johannesburg, 2021b). The municipal SDF aims to realise the goals, objectives and overall vision of the municipal integrated development plan (IDP) (City of Johannesburg, 2022). The findings presented in this study can inform more effective spatial and land use planning in Johannesburg for the delivery of critical ecosystem services which support climate change adaptation and mitigation. The SDF emphasises the importance of natural structures as providers of valuable ecosystem services, viewing these areas as critical assets rather than unused land for future development (City of Johannesburg, 2021b). My research can inform the SDF's evaluation of critical biodiversity areas, to include urban spaces that supply ecosystem services. To develop a resilient city, the SDF focuses on the natural environment's role in buffering environmental impacts and protecting green infrastructure. Open space systems are identified as crucial for mitigating the urban heat island effect, managing stormwater, improving air quality, and providing recreational spaces. The framework aims to integrate climate change adaptation, protect biodiversity, and maximise ecosystem services (City of Johannesburg, 2021b). My research provides a foundation for achieving these goals by offering spatially explicit information on the supply of key ecosystem services, which contribute to climate change mitigation.

Quantifying and mapping urban ecosystem services in Johannesburg is vital for informing the city's spatial development frameworks and land use planning, as it provides a detailed understanding of the distribution, value, and function of natural resources and land-uses within urban areas. By incorporating ecosystem service assessments into the planning process, city planners can identify and prioritise areas that are critical for maintaining ES supply and enhancing residents' quality of life. Moreover, spatially explicit data on ecosystem services enable the development of targeted policies and interventions that support sustainable urban growth, resilience to environmental changes, and equitable access to natural benefits across different urban communities (Dhyani *et al*., 2021; Farrugia *et al*., 2013; de Manuel *et al*., 2021). By mapping ecosystem services the specific land uses and areas which provide high values of ES supply can be adequately protected and conserved and can be incorporated into the SDF. Simultaneously, areas of critically low supply of ES can be targeted by the SDF as key intervention areas, where ES supply can be enhanced and the implementation of GI and greenspace construction can be effectively targeted (Derkzen *et al*., 2015; de Manuel *et al*., 2021). The results of my and future research, should be used to inform the creation of spatial planning instruments which more explicitly address the protection and enhancement of ecosystem service supply. The role of ecosystem services should also be incorporated more effectively into the city's integrated development plan, where they receive insufficient attention (see City of Johannesburg, 2022). While the SDF aims to maximise the value of ecosystem services, implementation of the concept or integration into spatial planning instruments remain under-utilised. The SDF identifies key spatial opportunities in protecting biodiversity and the ES it provides and my research provides an initial foundation for these objectives and opportunities to be actioned and realised, through the use of spatially explicit information on the supply of three ecosystem services.

As with the other two ES, there is large variability in C storage across the city, with the majority being stored in the northern suburbs. This highlights the potential to improve the C storage capacity of the city by planting more trees and expanding the urban forest to the rest of the city. While initiatives have been undertaken (see van Staden & Stoffberg, 2021), further action is needed to ensure that the maximum potential carbon storage within the city is achieved. Our maps highlight the areas which can take advantage of this, as well as evidence the 'best-case' potential of effective ecosystem service delivery through green infrastructure in an African city. Given the available space in Johannesburg, with only 16.1% of the city covered by trees (Schäffler & Swilling, 2013), there is great potential for increasing the carbon stock in the city. The inequality in tree cover distribution has important management implications for the potential to increase tree coverage and enhance the city’s climate change mitigation strategies, while also addressing ES inequalities and building resilience in vulnerable communities.

Differing urban forest types in Johannesburg have significantly different carbon storage capacities due to variation in structure, density, and species composition, which has implications for urban planning and greenspace planning. Our findings can inform local authorities and decision makers in creating green spaces with the highest contribution to climate change mitigation through carbon storage. In conjunction with the findings relating to runoff and cooling, UGS planning, and design can be optimised for delivery of crucial ecosystem services. Our results further indicated that many of Johannesburg's trees were large and old. Although this drives their high C storage in AGB, it also brings out management implications for replacing these trees at the end of their life cycle. Johannesburg maintains a large C sink of 2.4 million tonnes of carbon; however, this C sink is not permanent. The maximum carbon storage capacity is reached when the urban forest reaches maturation (Chen, 2015). With the onset of senescence of the city’s vegetation, their contribution to the carbon cycle will diminish and carbon will be released back into the atmosphere upon the death of the trees. The maturity of Johannesburg's urban forest entails that many of the large trees in the city are nearing the end of their life cycle, thus it is pivotal that the city plans and strategies adequately to replace these C sinks which will soon diminish. This process is complicated by the fact that most of the largest trees are alien species, which are no longer planted. The question must be raised as to how these large trees will be adequately replaced when indigenous species are generally smaller. Thus, active management of the city’s UGS is required to ensure tree cover is maintained and that trees at the end of their life cycle are replaced. By re-planting trees, new C sinks will replace the old and additionally, young, and growing trees contribute significantly to carbon sequestration. The CoJ must consider strategies to ensure that C sinks are maintained and enhanced, to preserve the C storage ecosystem service which contributes to the city’s climate change mitigation. Long-living, low-maintenance tree species should be prioritised to maximise climate change mitigation (Chen, 2015; Davies *et al*., 2011; Jo, 2002; Nowak *et al*., 2002). It is also crucial to ensure that young and growing trees replace the older trees reaching the end of their life cycle (Lindén *et al*., 2020).

My results showed that exotic tree species contribute most of the carbon storage in Johannesburg. It is therefore advised that the COJ ensure that the benefits of exotic tree species are fully considered when determining future responses to exotic tree species. In the context of the already transformed urban ecosystem of Johannesburg, which has long since lost its grassland characteristics and entered novel ecosystem status, caution is advised when prioritising the conservation of past status, rather than functionality and UES. In the context of rapid climate change, and the increasing impacts on cities- building urban resilience and mitigation of climate change is pivotal. Functionality must be prioritised to address climate change and ensure safe, habitable cities under current and future climatic change. This will have key management implications in Johannesburg, particularly when the old, large exotic species currently standing in the city die and need to be replaced.

6.7 Conclusion

This study aimed to quantify and map three contextually important ecosystem services in the city of Johannesburg, South Africa. A combination of field sampling, modelling and remote sensing methodologies were used to quantify the supply of carbon storage, runoff retention and cooling services in Johannesburg, which were then mapped across the administrative wards of the city. The results revealed an impressive carbon stock in Johannesburg's urban forest of 2.4 million tonnes, with carbon storage differing significantly between urban forest types. The highest carbon storage was observed in parks, with 97.5 t/ha, while the lowest was observed in nature reserves, with 39.92 t/ha, with differences in structure and composition driving this variation. A high supply of cooling and runoff retention services were observed in Johannesburg, with an average of 73% of precipitation being retained under a 50 mm design storm, and a wide distribution of cooling in the city.

These services have immense value in the context of the urban challenges facing Johannesburg, including flooding and heat stress, both of which will be exacerbated by the impacts of climate change. The three services assessed in this study make important contributions to both climate change mitigation and adaptation strategies. However, the results revealed distinct spatial disparities in the supply of these services, with higher supply in the north and lower supply in the south. Several areas in the city face a critical undersupply of ecosystem services, heightening their vulnerability to the impacts of climatic change. By mapping the supply of ecosystem services at the ward scale, my findings can inform decision makers in addressing areas of low supply and preserving biodiversity in areas of high supply. My results have numerous implications for the integration of ecosystem services into climate change adaptation and mitigation strategies and can aid in greenspace planning to ensure an equitable distribution of ecosystem services and ensuring the sustainability and habitability of Johannesburg into the future.

6.8 Limitations

The primary limitations of this study relate to the spatial datasets used and their resolutions. The SANLC dataset is used as a landcover/land-use dataset in the assessment of all three ES. The resolution of this dataset is 30m, meaning intricacies of ES supply which occur at a scale below 30m are not accounted for. For example, the occurrence of singular or small patches of trees or greenspace might not be detected by this dataset and therefore not included in carbon extrapolation or runoff modelling. This research could be improved in the future, when higher resolution data becomes available (e.g., 1m LiDAR data). This would enable more accurate runoff modelling, temperature assessments and would enhance the accuracy of C storage extrapolation. The SANLC dataset is considered 85% accurate and therefore may introduce uncertainties, although several steps were taken to limit their effect. The accuracy of C storage estimations is largely dependent on the spatial resolution of the data used (Davies *et al*., 2013). Higher resolution data fails to capture the fine scale land use mosaic which characterises urban areas (Gill *et al*., 2008). Davies *et al*. (2013) showed that the use of 10 and 25m spatial resolutions has a relatively small effect on the variability of C stock estimations, relative to using higher spatial resolutions. Thus, the resolutions used in this study can be deemed reliable, although there is room for future improvement.

It should also be considered that the indicators used to represent cooling and runoff retention ES are simplified representations of processes which are far more complex, despite their common use in urban ecosystem service assessments (Haase *et al*., 2014; Kremer *et al*., 2016). The InVEST UFRM model to determine runoff retention, is a simplification of more complex processes, as all models are. In turn, this model does not consider the spatial variability of rainfall and is not temporally explicit (Kadaverugu *et al*., 2021).

Regarding the cooling quantification methodology, the primary limitation of using a remote-sensing approach is effect of cloud cover, which results in missing data, thereby introducing uncertainties. We attempted to mitigate the influence of cloud cover by using multiple satellite scenes and determining the average across several days in different years, with varying climatic conditions, to reduce the effect of anomalous conditions. Furthermore, no imagery with cloud cover of greater than 10% was included in the analysis.

Numerous uncertainties and challenges were faced in assessing the urban C stock of Johannesburg. This study stratified the sampling a the four dominant UGS types within Johannesburg. This enabled the key differences in C storage capacity amongst different UGS types with differing structures and compositions to be assessed and used to improve the accuracy of extrapolation. However, not all land use types in Johannesburg were sampled. Consequently, some land uses, such as wetlands and agricultural LU, were assigned an average value based on the sampled forest types, likely reducing accuracy. Another limitation was the low number of garden plots conducted, due to the inaccessibility of private property.

The accuracy of results stemming from allometric, or biomass equations may be limited by the lack of equations derived specifically for urban trees (Aguaron & McPherson, 2012; McHale *et al*., 2009; Davies *et al*., 2013). However, the development of these equations in the urban context is often challenging, given that it is usually not possible to cut down trees and determine their biomass through destructive sampling. Currently, very few allometric equations which were derived specifically for urban trees exist, resulting in a reliance on equations derived from traditional forests (McHale *et al*., 2009; Davies *et al*., 2013). No allometric equations exist for South African urban tree species. Hence, this study used a generalised biomass equation which had been derived from natural forests (Schäffler & Swilling, 2013). To minimise uncertainty, our results were compared to several commonly used local and international biomass equations (Table 9), enabling cross-comparison and validation of our findings in the context of other methods. However, there remain numerous uncertainties associated with using general biomass equations in the urban context (McHale *et al*., 2009; Yoon, 2013). Future research should focus on developing urban-specific allometric equations in the Johannesburg or South African context, to enable the estimation of C stocks with greater accuracy.

6.9 Future Research

More research is necessary to achieve a more complete understanding and accounting of the carbon cycle of urban green infrastructure. Firstly, there is a distinct lack of comparable C storage studies in South Africa. Further research is required to quantify C storage in other South African cities, with different climatic, geographic, and historical conditions. This will support the development of a more complete understanding of the role of urban areas in mitigating climatic change in South Africa. Secondly, other biotic carbon pools must be considered, especially organic soil carbon, which was not assessed in this study but has been shown to have high C storage potential in urban areas (Pataki *et al*., 2006; Dorendorf *et al*., 2015; Lindén *et al*., 2020; Seboko *et al*., 2021). Research has been undertaken in some regions of Johannesburg, which indicates significant C stocks in the region (Seboko *et al*., 2021), future research should assess the city in its entirety. Soil C stock in greenspaces needs to be assessed in conjunction with AGB (Dorendorf *et al*., 2015) to fully characterise this ecosystem service. Finally, the maintenance processes associated with urban trees need to be assessed, to ensure low-carbon maintenance can be achieved. Other important considerations are the economic feasibility of constructing green infrastructure in South African cities, as well as operational requirements from a policy perspective (relating to funding or incentives), to implement green infrastructure as a climate change mitigation strategy need to be explored (Chen, 2015; Pataki *et al*., 2011).

Another key area for future research is the phenological characteristics of trees in Johannesburg, including lifespans and growth rates. This information could be used to determine C sequestration and ensure that tree planting efforts are timed such that continuous tree cover is maintained and old, dying trees are replaced by young, growing trees (Schmitt-Harsh *et al*., 2013). Carbon sequestration was not assessed in this study due to the absence of data on growth rates, but serves as an important process in dictating the overall carbon flux in urban systems and has great potential to mitigate CO_2 emissions in urban systems (Stoffberg *et al*., 2010; Liu & Li, 2012; Nowak *et al*., 2013; O'Donoghue & Shackleton, 2013; Chen, 2015).

Our UES assessment addressed the three most contextually important services in Johannesburg, cooling, runoff retention and carbon storage. However, urban greenspaces provide a wide range of benefits and services beyond these, including air purification, recreational potential, enhancing physical and mental health, increase biodiversity, noise cancellation and numerous others (Bolund & Hunhammar, 1999; Gómez-Baggethun & Barton, 2013; Haase *et al*., 2014; Tzoulas *et al*., 2007; Maas, 2006; Gascon *et al*., 2015; van den Berg *et al*., 2015; Goddard *et al*, 2010; Hostetler *et al*., 2011). Consequently, to develop a wholistic understanding of the value of UES in Johannesburg and fully assess their trade-offs and synergies, other services need to be assessed in future research. Air pollution, cultural services and biodiversity need to be assessed to understand trade-offs between services. This is relevant for example, in our findings for C storage which show that alien trees provide high C storage but their potentially damaging impact on native biodiversity is not assessed. Furthermore, the low C storage found in nature reserves, relative to other LU, may be balanced out by their high contribution to biodiversity. Without a full ecosystem service assessment, the value of different UGS cannot be comprehensively assessed. Other key areas for future research in Johannesburg include the quantification of ES demand (Jiang *et al*., 2022; Li *et al*., 2021; Baró *et al*., 2015) and the economic quantification of the economic value of ES (de Wit *et al*., 2012; Gómez-Baggethun & Barton, 2013; Schäffler & Swilling, 2013). The assessment of a suite of ecosystem services can further enable a more comprehensive integration of key ES into municipal planning instruments such as the spatial development framework (See Johannesburg 2021b). Future research should assess other services and explore the potential of spatially explicit ES supply information for the formation of spatial planning instruments to mitigate climate change impacts in the city of Johannesburg.

Another dimension which was not addressed in this study, but is important and needs mention, is the role of culture, values and beliefs in regulating ES supply and demand. This is particularly important in the African context, as the cultural-symbolic dimension is generally considered to be more pronounced in the global south than the global north (Shackleton *et al*., 2021). These considerations affect decision-making and value creation amongst individual actors, who play a key role in moderating ES flows in urban systems (e.g. (Schmitt-Harsh *et al*., 2013). Furthermore, the role of individuals is reflected in Johannesburg by the large C storage capacity of privately owned gardens in Johannesburg, storing 65.36 t/ha. Additionally, the areas of highest ES supply in Johannesburg, across all three services, lie in the Northern regions where private gardens are extensive and heavily greened, estimated to contain 4 million trees. Human agency is a key element of ES delivery and demand, as ES themselves are co-produced by humans and nature (Haines-Young & Potschin-Young, 2018). Consequently, the factors that influence ES supply and demand at the individual level need to be assessed.

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8. Appendices

Appendix A: Field sample points and strata selection

Strata selection

To select meaningful strata, the ESA tree cover layer was integrated with the SANLC LU map, and the land use classes were sorted by their percentage tree cover and area (ha) of tree cover (Table 1). This data, alongside ground truthing, was used to select the four dominant green space types in the city as strata for sampling (Table 2).

Table A1: Percentage and area tree cover of each South African National Landcover (2020) land use in Johannesburgt.

NLC Land Use	Tree Cover (%)	Area Tree Cover (ha)	Proportion of total Tree cover (% area) *
Residential Formal (Tree)	75.722	63779200	24.529
Residential Formal (Bush)	56103	9006400	3.464
Contiguous Low Forest & Thicket (combined classes)	56.028	3534000	1.359
Temporary Unplanted Forest	53.673	6149600	2.365
Smallholdings (Tree)	53.302	17263600	6.639
Contiguous & Dense Planted Forest (combined classes)	51.871	12688800	4.880
Smallholdings (Bush)	47.211	904000	0.348
Dense Forest & Woodland (35 - 75% cc)	44.991	28515200	10.967
Natural Rivers	42.301	208800	0.080
Open & Sparse Planted Forest	41.688	1642000	0.632
Residential Informal (Tree)	29.352	195600	0.075
Urban Recreational Fields (Bush)	27.841	78400	0.030
Residential Informal (Bush)	27.143	15200	0.006
Open Woodland (10 - 35% cc)	24.637	3073200	1.182
Urban Recreational Fields (Tree)	22.420	2784800	1.071
Residential Formal (low veg / grass)	22.150	51527600	19.817
Herbaceous Wetlands (currently mapped)	21.890	8414000	3.236
Village Scattered (bare only)	18.704	3255600	1.252
Herbaceous Wetlands (previous mapped extent)	18.309	3433600	1.321

NLC Land Use	Tree Cover (%)	Area Tree Cover (ha)	Proportion of total Tree cover (% area) *
Smallholdings (low veg / grass)	14.528	9591200	3.689
Fallow Land & Old Fields (Bush)	12.139	50400	0.019
Cultivated Commercial Permanent Orchards	10.469	11600	0.004
Fallow Land & Old Fields (Trees)	10.359	440800	0.170
Urban Recreational Fields (Grass)	10.060	1542000	0.593
Artificial Dams (incl. canals)	8.785	366000	0.141
Sparsely Wooded Grassland (5 - 10% cc)	6.911	6800	0.003
Fallow Land & Old Fields (wetlands)	6.825	274400	0.106
Natural Grassland	6.162	18308400	7.041
Natural Pans (flooded at obsv time)	5.622	63600	0.024
Dry Pans	5.585	16800	0.006
Bare Riverbed Material	5.386	27600	0.011
Village Dense (bare only)	5.000	203200	0.078
Smallholdings (Bare)	4.853	898800	0.346
Commercial Annuals Non-Pivot Irrigated	4.131	29200	0.011
Artificial Sewage Ponds	3.706	22800	0.009
Residential Informal (low veg / grass)	3.697	467200	0.180
Artificial Flooded Mine Pits	3.070	25600	0.010
Urban Recreational Fields (Bare)	2.856	78000	0.030
Landfills	2.856	31200	0.012
Natural Rock Surfaces	2.854	68400	0.026
Mines: Extraction Sites: Open Cast & Quarries combined	2.786	270800	0.104
Commercial	2.338	2433200	0.936
Residential Formal (Bare)	2.301	6072800	2.336
Subsistence / Small-Scale Annual Crops	2.201	62400	0.024
Other Bare	1.893	338800	0.130
Roads & Rail (Major Linear)	1.393	85200	0.033

NLC Land Use	Tree Cover (%)	Area Tree Cover (ha)	Proportion of total Tree cover (% area) *
Mines: Waste (Tailings) & Resource Dumps	1.357	273600	0.105
Industrial	1.066	648000	0.249
Fallow Land & Old Fields (Grass)	0.740	484400	0.186
Low Shrubland (other regions)	0.626	6400	0.002
Fallow Land & Old Fields (Bare)	0.586	28800	0.011
Commercial Annuals Crops Rain-Fed / Dryland / Non-Irrigated	0.520	233600	0.090
Residential Informal (Bare)	0.457	77600	0.030

*Represents the proportion each land use contributes to the total tree cover (area) in the city of Johannesburg.

Table A2: SANLC classes used for sampling, proportion of total tree cover and estimated number of trees for each of the four selected urban forest types in Johannesburg.

Urban Forest Type	Constituent SANLC Classes	Tree Cover Proportion (%)
Park	Contiguous & Dense Planted Forest (combined classes), Temporary Unplanted Forest, Urban Recreational Fields (Bush), Urban Recreational Fields (Tree), Urban Recreational Fields (Grass), Dense Forest & Woodland, Open & Sparse Planted Forest	20.57
Nature Reserve	Open Woodland, Dense Forest & Woodland, Natural Grassland, Sparsely Wooded Grassland and *manual reclassification	8.23

Garden	Residential Formal (Tree), Residential Formal (Bush), Residential Formal (Bare)	47.18 %
Roadside	*Manually re-classified from existing classes	N/A

**Note* Roadside trees are not categorised in the SANLC Dataset and were therefore manually re-classified from existing classes (see below). Tree cover proportion is based on the ESA tree cover dataset with reference to the constituent SANLC classes and is therefore unavailable for Roadside trees.

Sample point generation

The NLC dataset did not contain a LU which demarcates roadside trees, thus one was created manually and reclassified from existing NLC land classes according to the methods presented in **Figure A1**.

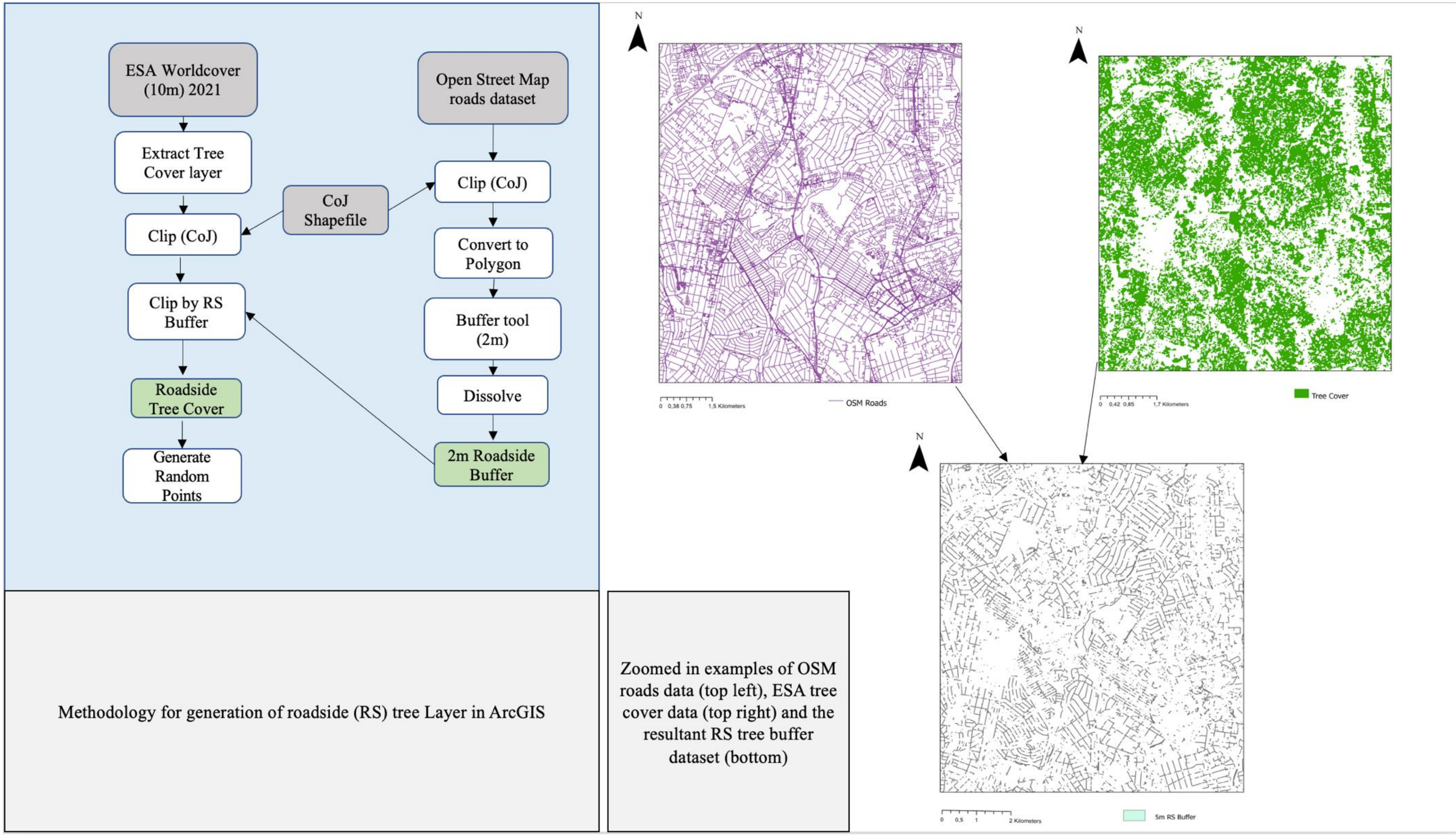

Figure A1: GIS Workflow and example imagery of the creation of a roadside (RS) tree cover layer. Green boxes represent key outputs, grey boxes inputs and white boxes represent temporary products, methodological steps or GIS tool.

Appendix B: Curve number value assignment

Table B1: Curve numbers assigned to each land use in the South African National Land Cover (SANLC) 2020 dataset, for use in the inVEST urban flood risk mitigation model.

Land Use Description	**LU code**	**CN Value**			
		Hydrologic Soil group			
		A	**B**	**C**	**D**
Contiguous Low Forest & Thicket (combined classes)	2	30	55	70	77
Dense Forest & Woodland (35 - 75% cc)	3	30	55	70	77
Open Woodland (10 - 35% cc)	4	35	60	73	79
Contiguous & Dense Planted Forest (combined classes)	5	30	55	70	77
Open & Sparse Planted Forest	6	40	65	76	82
Temporary Unplanted Forest	7	32	58	72	79
Low Shrubland (other regions)	8	30	48	65	73
Sparsely Wooded Grassland (5 - 10% cc)	12	39	60	73	79
Natural Grassland	13	39	60	73	79
Natural Rivers	14	100	100	100	100
Natural Pans (flooded at observed time)	18	100	100	100	100

Land Use Description	**LU code**	**CN Value**			
		Hydrologic Soil group			
		A	**B**	**C**	**D**
Artificial Dams (incl. canals)	19	100	100	100	100
Artificial Sewage Ponds	20	100	100	100	100
Artificial Flooded Mine Pits	21	100	100	100	100
Herbaceous Wetlands (currently mapped)	22	100	100	100	100
Herbaceous Wetlands (previous mapped extent)	23	100	100	100	100
Fallow Land & Old Fields (wetlands)	73	100	100	100	100
Cultivated Commercial	32	32	58	72	79
Permanent Orchards					
Commercial Annuals Pivot Irrigated	38	63	73	81	84
Commercial Annuals Non-Pivot Irrigated	39	63	73	81	84
Commercial Annuals Crops Rain-Fed / Dryland / Non-Irrigated	40	63	73	81	84
Subsistence / Small-Scale Annual Crops	41	63	73	81	84
Fallow Land & Old Fields (Trees)	42	70	80	86	89

Land Use Description	**LU code**	**CN Value**			
		Hydrologic Soil group			
		A	**B**	**C**	**D**
Fallow Land & Old Fields (Bush)	43	69	80	86	88
Fallow Land & Old Fields (Grass)	44	74	83	88	90
Fallow Land & Old Fields (Bare)	45	77	86	91	94
Smallholdings (Tree)	57	59	69	79	82
Smallholdings (Bush)	58	59	69	78	82
Smallholdings (low veg / grass)	59	62	72	81	84
Smallholdings (Bare)	60	65	75	83	86
Natural Rock Surfaces	25	75	85	89	94
Dry Pans	26	72	82	88	90
Bare Riverbed Material	30	72	82	88	90
Other Bare	31	74	84	90	92
Landfills	72	82	82	82	82
Residential Formal (Tree)	47	41	62	75	81
Residential Formal (Bush)	48	49	68	79	83
Residential Formal (low veg / grass)	49	57	72	81	86

Land Use Description	**LU code**	**CN Value**			
		Hydrologic Soil group			
		A	**B**	**C**	**D**
Residential Formal (Bare)	50	70	81	87	90
Residential Informal (Tree)	51	56	72	81	85
Residential Informal (Bush)	52	57	72	81	86
Residential Informal (low veg / grass)	53	76	84	89	91
Residential Informal (Bare)	54	77	85	90	92
Village Scattered (bare only)	55	61	70	78	82
Village Dense (bare only)	56	62	71	79	83
Urban Recreational Fields (Tree)	61	37	60	73	79
Urban Recreational Fields (Bush)	62	46	66	77	83
Urban Recreational Fields (Grass)	63	39	61	74	80
Urban Recreational Fields (Bare)	64	68	79	86	89
Commercial	65	83	89	92	94
Industrial	66	81	88	91	93

Land Use Description	**LU code**	**CN Value**			
		Hydrologic Soil group			
		A	**B**	**C**	**D**
Roads & Rail (Major Linear)	67	98	98	98	98
Mines: Surface Infrastructure	68	76	85	89	91
Mines: Extraction Sites: Open Cast & Quarries combined	69	76	85	89	91
Mines: Extraction Sites: Salt Mines	70	76	85	89	91
Mines: Waste (Tailings) & Resource Dumps	71	76	85	89	91

Appendix C: Ermida Land surface temperature method

The methodology proposed by Ermida *et al.* (2020) is based on an algorithm which assesses the empirical relationship between top of atmosphere brightness and LST by applying simple linear regression (Sun *et al.*, 2004; Li *et al.*, 2013; Duguay-Tezlaff *et al.*, 2015). The algorithm, as implemented by Ermida *et al* (2020), is presented in equation (C1).

$$LST = A_i \frac{Tb}{\varepsilon} + B_i \frac{1}{\varepsilon} + C_i \qquad (equation\ C1)$$

Where Tb and ε denotes the top of atmosphere (TOA) brightness temperature and surface emissivity, respectively, within the same thermal infrared (TIR) channel. A_i, B_i and C_i represent the algorithm coefficients, derived from linear regressions of radiative transfer simulations, calibrated for different classes of total column water vapor. For further details on the input datasets, calibration, and validation of the algorithm, see Ermida *et al* (2020).

The algorithm is implemented in Google Earth Engine, an Application Programming Interface (API), which runs on Javascript and is able to access satellite imagery databases, including Landsat. The code, written in Javascript, used to call upon the Ermida algorithm, using Landsat imagery on GEE, to determine land surface temperatures across Johannesburg, is presented below (see 'Source Code’). Note that the date inputs were subject to change, as temperature maps were generated across numerous time frames.

Source Code

```
// connects to code that calculates LST
var LandsatLST = require('users/sofiaermida/landsat_smw_lst:modules/Landsat_LST.js')

// select relevant date, study area and Landat satellite
var geometry = ee.Geometry.Rectangle([27.704722, -26.509722, 28.226389, -25.910833]);
var satellite = 'L8';
var date_start = '2022-03-16';
var date_end = '2022-03-30';
var use_ndvi = true;

// Get the Landsat collection with additional variables
// output includes fraction of veg cover, lst, surface emiisivity for thermal infrared band, total
water column vapor and landsat original bands which are clound matched
var LandsatColl = LandsatLST.collection(satellite, date_start, date_end, geometry, use_ndvi)
print(LandsatColl)

// Select the first feature
var exImage = LandsatColl.first();
```

```
var cmap1 = ['blue', 'cyan', 'green', 'yellow', 'red'];
var cmap2 = ['F2F2F2','EFC2B3','ECB176','E9BD3A','E6E600','63C600','00A600'];

Map.centerObject(geometry)
Map.addLayer(exImage.select('TPW'),{min:0.0, max:60.0, palette:cmap1},'TCWV')
Map.addLayer(exImage.select('TPWpos'),{min:0.0, max:9.0, palette:cmap1},'TCWVpos')
Map.addLayer(exImage.select('FVC'),{min:0.0, max:1.0, palette:cmap2}, 'FVC')
Map.addLayer(exImage.select('EM'),{min:0.9, max:1.0, palette:cmap1}, 'Emissivity')
Map.addLayer(exImage.select('B10'),{min:290, max:320, palette:cmap1}, 'TIR BT')
Map.addLayer(exImage.select('LST'),{min:290, max:320, palette:cmap1}, 'LST')
Map.addLayer(exImage.multiply(0.0001),{bands: ['B4', 'B3', 'B2'], min:0, max:0.3}, 'RGB')

Export.image.toDrive({
  image: exImage.select('LST'),
  description: 'LST',
  scale: 30,
  region: geometry,
  fileFormat: 'GeoTIFF',
```

Appendix D

Due to their relatively small area, nature reserves, protected areas, and remnant indigenous vegetation with tree cover (woodland) were manually delineated within ArcGIS. LU maps were compared to satellite imagery and ground truthing was used to verify the occurrences of planted forest or woodland and natural woodland in SANLC classes 1-7. Existing LU classes within these areas (SANLC Values: 3,5,7,61) were reclassified to a new class representing natural (indigenous) woodland (new value=75). Where sparse woodland occurred in these areas (6) they were reclassified to sparse natural woodland (new value =76). These LU classes were then merged into a single layer. The remaining areas with tree cover in the forest/woodland classes (3,5,6,7,61) were considered to represent planted forest and human-made urban green spaces. These three classes were then assigned carbon storage values accordingly based on NR and Park field values. The reclassified land uses were used as masks for the original SANLC raster, to ensure there is no overlap. Then, the mosaic to new raster tool was used to combine all LU's into a single raster for use in extrapolation.

Outside of the four key strata which were sampled in the remaining land uses were assigned a generic value. This value was determined by averaging the carbon storage of the four sampled Strata (64.4 t/ha). Natural (indigenous woodlands) were assigned a carbon storage value equivalent to municipal nature reserved from the field.

Other key assumptions included that 70% of the formal residential (tree) LU consisted of garden or tree cover. This was based on the % tree cover extracted from the ESA dataset and ground-truthing. Finally, the look-up table tool in ArcGIS was used to assign C storage values to the SANLC LU raster. The full value assigned to each SANLC land use is presented in Table D1 below.

Table D1: Carbon storage per pixel values assigned to each South African National Land Cover (NLC) land use for extrapolation, alongside the LU value and number of occurrences.

Land Use	C storage (tons/pixel 400m^2)	C storage (t/ha)	LU Value (NLC)	Count (Pixels)
Contiguous Low Forest & Thicket (combined classes)	3.900	97.500	2	15767
Dense Forest & Woodland (35 - 75% cc)	3.900	97.500	3	144769
Open Woodland (10 - 35% cc)	0.975	24.375	4	31178
Contiguous & Dense Planted Forest (combined classes)	3.900	97.500	5	60727
Open & Sparse Planted Forest	1.950	48.750	6	9704

Land Use	C storage (tons/pixel 400m^2)	C storage (t/ha)	LU Value (NLC)	Count (Pixels)
Temporary Unplanted Forest	3.900	97.500	7	28638
Low Shrubland (other regions)	0.000	0	8	17
Sparsely Wooded Grassland (5 - 10% cc)	1.116	27.900	12	8
Natural Grassland	1.116	27.900	13	50027
Natural Rivers	2.576	64.400	14	538
Natural Pans (flooded @ obsv time)	2.576	64.400	18	169
Artificial Dams (incl. canals)	0.000	0.00	19	867
Artificial Sewage Ponds	0.000	0.00	20	67
Artificial Flooded Mine Pits	0.000	0.000	21	53
Herbaceous Wetlands (currently mapped)	2.576	64.400	22	20626
Herbaceous Wetlands (previous mapped extent)	2.576	64.400	23	8339
Natural Rock Surfaces	0.000	0.000	25	237
Dry Pans	0.000	0.000	26	31
Bare Riverbed Material	0.000	0.000	30	74
Other Bare	0.000	0.000	31	1065
Cultivated Commercial Permanent Orchards	2.576	64.40-	32	29
Commercial Annuals Pivot Irrigated	2.576	64.400	38	9
Commercial Annuals Non-Pivot Irrigated	2.576	64.400	39	52
Commercial Annuals Crops Rain-Fed / Dryland / Non-Irrigated	2.576	64.400	40	635
Subsistence / Small-Scale Annual Crops	2.576	64.400	41	144
Fallow Land & Old Fields (Trees)	2.576	64.400	42	10623
Fallow Land & Old Fields (Bush)	2.576	64.400	43	166
Fallow Land & Old Fields (Grass)	2.576	64.400	44	1590

Land Use	C storage (tons/pixel 400m²)	C storage (t/ha)	LU Value (NLC)	Count (Pixels)
Fallow Land & Old Fields (Bare)	0.000	0.00	45	82
Fallow Land & Old Fields (Low Shrub)	2.576	64.400	46	2
Residential Formal (Tree)	1.802	45.050	47	209402
Residential Formal (Bush)	2.576	64.400	48	16725
Residential Formal (low veg / grass)	2.576	64.400	49	112729
Residential Formal (Bare)	2.576	64.400	50	19135
Residential Informal (Tree)	2.576	64.400	51	1639
Residential Informal (Bush)	2.576	64.400	52	28
Residential Informal (low veg / grass)	2.576	64.400	53	994
Residential Informal (Bare)	0.000	0.000	54	177
Village Scattered (bare only)	0.000	0.000	55	7156
Village Dense (bare only)	0.000	0.000	56	492
Smallholdings (Tree)	2.576	64.400	57	80111
Smallholdings (Bush)	2.576	64.400	58	1630
Smallholdings (low veg / grass)	2.576	64.400	59	20667
Smallholdings (Bare)	0.000	0.000	60	2658
Urban Recreational Fields (Tree)	3.900	97.500	61	30902
Urban Recreational Fields (Bush)	2.576	64.400	62	118
Urban Recreational Fields (Grass)	2.576	64,400	63	3505
Urban Recreational Fields (Bare)	0.000	0.000	64	235
Commercial	2.576	64.400	65	6992
Industrial	2.576	64.400	66	1680
Roads & Rail (Major Linear)	0.000	0.000	67	291
Mines: Extraction Sites: Open Cast & Quarries combined	0.000	0.000	69	770
Mines: Waste (Tailings) & Resource Dumps	0.000	0.000	71	765

Land Use	C storage (tons/pixel 400m^2)	C storage (t/ha)	LU Value (NLC)	Count (Pixels)
Land-fills	0.000	0.000	72	80
Fallow Land & Old Fields (wetlands)	2.576	64.400	73	682
NR Forest/Woodland	1.116	27.900	75	14121
NR sparse woodland	0.558	13.950	76	136
Roadside Plantation*	2.713	67.825	77	38485

* RS manually created LU, not included in original NLC Dataset.
Note: LU Value refers to the identifying value of each land use in the original NLC Dataset. Number of occurrences (pixels) refers to only the tree covered pixels within each land use.

Table D2: Landcover datasets used during extrapolation.

Dataset	Resolution	Classes	Description
ESA world cover, 2021	10m	11	The European Space Agency (ESA) world cover 10m, 2021 is a global landcover dataset, with a resolution of 10m. It is based on Sentinel-1 and -2 imagery. It consists of 11 LC classes, of which only tree cover (value =10) was utilised in this study.
SANLC 2020	20m	73	Raster format, South African National Land-Cover 2020 (SANLC 2020) dataset was generated by mapping models, based on Sentinel 2 satellite imagery (20m resolution) between January-December 2020.

Appendix E

UFRM Model implementation and mapping

The UFRM model estimates runoff (Q) in mm by executing the SCS-CN method for each pixel *i* , defined by a LU type and Soil characteristics:

$$Q_{p,i} = \left\{ \frac{(P-0.2S)^2}{P+0.8S} \; if \; P > 0.2S \right\} \qquad \textit{(equation E1)}$$

Where *P* is the design storm depth (mm), *S* is the maximum potential runoff and 0.2S represents the initial abstraction (the minimum rainfall depth required to initiate runoff). The initial abstraction (λ) factor of 0.2 times the S_{max} was considered for the entire study area, although it should be noted that this is a simplification, albeit frequently used (Derkzen et al., 2015; Kremer et al., 2016; Kadaverugu et al., 2021).

S (mm) is a function of CN where:

$$S_i = \frac{2540}{CN_{\mathrm{i}}} - 25.4 \qquad \textit{(equation E2)}$$

Runoff retention per pixel (R_i) is then calculated:

$$R_i = 1 - \frac{Q_{p,i}}{P} \qquad \textit{(equation E3)}$$

Runoff retention volume ($R(m^3)$) per pixel is then calculated as:

$$R(m^3) = R_i * P * pixel\ area * 10^{-3} \qquad \textit{(equation E4)}$$

With pixel area in m^2.

The inputs required to run the UFRM model in inVEST, alongside their formatting, are presented in Table E2.

Table E2: Inputs into the inVEST UFRM (Urban Flood Risk Mitigation model)

Data	Format	Description
Study Area*	Vector/Polygon	Area of interest, map of areas for results to be aggregated over, watershed or sewer-shed boundaries.

Rainfall depth	Unit (mm)	Depth of rainfall for design storm
Land use	Raster	Map of land use/land cover. Each value in the raster must have a corresponding value in the biophysical table. All results from the model are produced at the resolution of this raster.
Hydrologic soil group	Raster	Map of the HSGs across the study area. Pixel values may range from 1-4, corresponding to the HSG groups A, B, C and D respectively.
Biophysical table	CSV	Table of CN Data for each LU class across each of the four soil groups. Each LU code in the LU raster should have corresponding values in this table for each HSG.

*Water sheds were delineated using ArcGIS hydrology tools, with a DEM (Digital Elevation Map) as the input (Figure 2)

Appendix F: Diversity across four sampled forest types

Table F1: Diversity of sampled individuals in Roadside plantations across Johannesburg, South Africa.

Genus	No. of observations	Relative proportion (%)	Species*
Pinus	47	15.99	*P. canariensis (30), P. palustrus (7) P. pinaster (5), P. engelmannii (1), Unidentified (4)*
Celtis	38	12.93	*C. sinensis (25), C. australis (7), C. Africana (5)*
Quercus	29	9.86	*Q. suber (8), Q. acutissima (6), Q. robur (6), Q. rugosa (5), Q. cerris (2), Q. ilex (1), Q. palustris (1)*
Eucalyptus	22	7.48	*E. saligna (6), E. sideroxylon (6), E. camaldulensis (3), E. globulus (1),* Unidentified (6)
Casuarina	19	6.46	*C. cunninghamiana (13), C. equisetifolia (6)*
Robinia	12	4.08	*R. pseudoacacia*
Ligustrum	11	3.74	*L. lucidum*
Olea	11	3.74	*O. europaea subsp. Africana (10), O. europaea (1)*

Genus	No. of observations	Relative proportion (%)	Species*
Searsia	11	3.74	*S. lancea (9),* S. leptodictya (1), *S. pyroides (1)*
Ulmus	10	3.40	*U. carpinifolia (4),* *U. parvifolia (3),* *U. procera (3)*
Taxodium	9	3.06	*T.distichum*
Combretum	9	3.06	*C. kraussii (7),* *C. erythrophyllum (2)*
Vachellia	8	2.72	*V. sieberiana subsp. Woodii (6),* *V. gerrardi (2)*
Cedrus	7	2.38	*C. deodara*
Cupressus	7	2.38	*C. sempervirens (5),* C. arizonica (1), C. funebris (1)
Acer	5	1.70	*A. buergerianum*
Jacaranda	5	1.70	*J. mimosifolia*
Platycladus	5	1.70	P. orientalis
Sequoia	5	1.70	*Sequoia sempervirens*
Arbutus	4	1.36	*A. unedo*
Viburnum	4	1.36	*V. odoratissimum*

Genus	No. of observations	Relative proportion (%)	Species*
Tilia	3	1.02	*Tilia x europaea*
Callistemon	2	0.68	*C. viminalis*
Cinnamomum	2	0.68	*C. camphora*
Sengalia	2	0.68	*S. caffra*
Melia	2	0.68	*M. azedarach*
Araujia	1	0.34	
Morus	1	0.34	*M. nigra*
Platanus	1	0.34	*Platanus x acerifolia*
Styphnolobium	1	0.34	*S. japonicum*
Zelkova	1	0.34	*Z. carpinifolia*

*Number of observations per species in brackets where more than one species observed

Table 2F: Diversity of sampled individuals in Nature Reserves across Johannesburg, South Africa

Genus	No. of observations	Relative proportion (%)	Species*
Celtis	60	13.216	*C. Africana (55), C. sinensis (5)*
Senegalia	51	11.233	*S. caffra*
Eucalyptus	46	10.132	*E. sideroxylon (23), E. camaldulensis (14), E. globulus (5), E. saligna (2), Unidentified (2)*

Genus	No. of observations	Relative proportion (%)	Species*
Vachellia	45	9.912	V. robusta (25), *V. karroo (17), V. sieberiana subsp. Woodii (2), V. erioloba (1)*
Searsia	38	8.370	*S. leptodictya (17), S. pyroides (15), S. lancea (6)*
Afrocanthium	28	6.167	*A. gilfillanii (22), A. mundianum (6)*
Olea	24	5.286	*O. europaea subsp. africana*
Ziziphus	18	3.965	*Z. mucronate*
Combretum	17	3.744	*C. erythrophyllum (14), C. apiculatum (1), C. hereroense (1), Unidentified (1)*

Genus	**No. of observations**	**Relative proportion (%)**	**Species***
Dais	15	3.304	*D. cotinifolia*
Podocarpus	15	3.304	*P. falcatus (8),* *P. latifolius (4),* *P. henkelii (3)*
Diospyros	14	3.084	*D. lycioides*
Brachylaena	13	2.863	*B. rotundata*
Acacia	8	1.762	*A. decurrens (8)*
Mundulea	8	1.762	*M. sericea*
Ehretia	6	1.322	*E. rigida*
Protea	5	1.101	*P. caffra*
Euclea	4	0.881	*E. crispa*
Grewia	4	0.881	*G. occidentalis*
Unidentified	4	0.881	-
Uclea	4	0.881	*U. crispa*
Vangueria	4	0.881	*V. infausta*
Dombeya	3	0.661	*D. rotundifolia*
Leucosidea	3	0.661	*L. sericea*
Vepris	3	0.661	*V. reflexa*
Dovyalis	2	0.441	*D. zeyheri*
Aloe	1	0.220	*A. barberae*
Commiphora	1	0.220	*C. glandulosa*

Genus	No. of observations	Relative proportion (%)	Species*
Cussonia	1	0.220	*C. paniculata*
Ficus	1	0.220	Unidentified
Gymnosporia	1	0.220	G. buxifolia
Heteromorpha	1	0.220	*H. arborescens*
Kiggelaria	1	0.220	*K. africana*
Peltophorum	1	0.220	*P. africana*
Pittosporum	1	0.220	*P. viridiflorum*
Prunus	1	0.220	*P. serotina*
Trema	1	0.220	*T. orientalis*
Ximenia	1	0.220	*X. caffra*

*Number of observations per species in brackets where more than one species observed

Table 3F: Diversity of sampled individuals in Gardens across Johannesburg, South Africa

Genus	No. of observations	Relative proportion (%)	Species*
Viburnum	18	7.347	*V. odoratissimum (14), V. tinus (4)*
Prunus	17	6.939	*P. laurocerasus (7), P. domestica (5), P. cerasifera (1), P. persica (1), P. serotina*

Genus	No. of observations	Relative proportion (%)	Species*
			(1), P. serrulata (1), Unidentified (1)
Celtis	16	6.531	*C. africana (8), C. sinensis (8)*
Vachellia	15	6.122	*V. xanthophloea (6), V. robusta (5), V. sieberiana var. woodii (3), V. karroo (1)*
Syzygium	13	5.306	*S. paniculatum*
Laurus	12	4.898	*L. nobilis*
Ligustrum	9	3.673	*L. lucidum (6), L. ovalifolium (3)*
Chamaecyparis	8	3.265	*C. lawsoniana (7), C. obtusa (1)*
Ficus	7	2.857	*F. carica*
Pittosporum	7	2.857	*P. tenuifolium*
Duranta	6	2.449	*D. erecta*
Indigofera	6	2.449	*I. jucunda*
Lycianthes	6	2.449	*L. rantonnetii*
Nuxia	6	2.449	*N. floribunda*
Citrus	5	2.041	*C. limon*
Jacaranda	5	2.041	*J. mimosifolia*

Genus	**No. of observations**	**Relative proportion (%)**	**Species***
Callistemon	5	2.041	*C. viminalis (4), C. citrinus (1)*
Podocarpus	5	2.041	*P. henkelii (4), P. falcatus (1)*
Searsia	5	2.041	*S.pyroides (3), S.lancea (1), S. penduline (1)*
N/A	4	1.633	-
Populus	4	1.633	*P. deltoides*
Brachychiton	4	1.633	*B. acerifolius (3), B. populneus (1)*
Magnolia	4	1.633	*M. denudate (1), M. figo (1), M. grandiflora (1) Magnolia x soulangeana (1)*
Brunfelsia	3	1.224	*B. pauciflora*
Camellia	3	1.224	*C. japonica*
Ceratonia	3	1.224	*C.siliqua*
Olea	3	1.224	*O. europaea subsp africana*
Acer	2	0.816	*A. buergerianum*
Buddleja	2	0.816	*B. davidii*
Cedrus	2	0.816	*C. deodara*

Genus	No. of observations	Relative proportion (%)	Species*
Cupressus	2	0.816	*C. funebris*
Euonymus	2	0.816	*E. japonicus*
Harpephyllum	2	0.816	*H. caffrum*
Lagerstroemia	2	0.816	*L. indica*
Liquidambar	2	0.816	*L. styraciflua*
Platycladus	2	0.816	*P.orientalis*
Yucca	2	0.816	*Y. aloifolia*
Acca	1	0.408	*A. sellowiana*
Beaucarnea	1	0.408	*B. recurvata*
Bougainvillea	1	0.408	*B. spectabilis*
Caesalpinia	1	0.408	*C. ferrea*
Calpurnia	1	0.408	*C.aurea*
Carya	1	0.408	*C. illinoinensis*
Castanospermum	1	0.408	*C. australe*
Cinnamomum	1	0.408	*C. camphora*
Combretum	1	0.408	Unidentified
Coprosma	1	0.408	*C. repens*
Cordyline	1	0.408	*C. australis*
Cyathea	1	0.408	*C. australis*
Dais	1	0.408	*D. cotinifolia*
Fraxinus	1	0.408	Unidentified

Genus	No. of observations	Relative proportion (%)	Species*
Ginkgo	1	0.408	*G. biloba*
Hibiscus	1	0.408	*H. rosa-sinensis*
Mackaya	1	0.408	*M. bella*
Morus	1	0.408	*M. alba*
Nerium	1	0.408	*N.oleander*
Phoenix	1	0.408	*P.roebelenii*
Phytolacca	1	0.408	*P. dioica*
Pouteria	1	0.408	*P. campechiana*
Rhaphiolepsis	1	0.408	*R. indica*
Roystonea	1	0.408	*R. regia*
Ulmus	1	0.408	*U. carpinifolia*
Washingtonia	1	0.408	*W. filifera*

Table 4F: Diversity of sampled individuals in Roadside plantations across Johannesburg, South Africa

Genus	No. of observations	Relative proportion (%)	Species*
Jacaranda	61	32.275	*J. mimosifolia*
Platanus	23	12.169	*Platanus x acerifolia*
Brachychiton	15	7.937	*B. discolor (11),* *B. acerifolius (5)*

Genus	**No. of observations**	**Relative proportion (%)**	**Species***
Searsia	11	5.820	*S. lancea*
Celtis	11	5.820	*C. sinensis (6),* *C. africana (5)*
Quercus	9	4.762	*Q. palustris (6),* *Q. robur (1),* *Unidentified (1)*
Fraxinus	7	3.704	*F.angustifolia*
Pinus	5	2.646	*P. palustris*
Sengalia	5	2.646	*S. caffra*
Trachycarpus	4	2.116	*T. fortunei*
Acer	3	1.587	*A.beuergerianum*
Cussonia	3	1.587	*C. spicata*
Lagerstroemia	3	1.587	*L. indica*
Liquidambar	3	1.587	*L. styraciflua*
Roystonea	3	1.587	*R. regia*
Combretum	2	1.058	*C. erythrophyllum*
Corymbia	2	1.058	*C. ficifolia*
Grevillea	2	1.058	*G. robusta*
Ailanthus	1	0.529	*A. altissima*
Aloe	1	0.529	*A. barbera*
Calodendrum	1	0.529	*C. capense*

Genus	No. of observations	Relative proportion (%)	Species*
Cedrus	1	0.529	*C. deodara*
Ceiba	1	0.529	*C. speciosa*
Chamaecyparis	1	0.529	*C. lawsoniana*
Cupressus	1	0.529	*C. arizonica*
Erythrina	1	0.529	*E. caffra*
Melaleuca	1	0.529	*M. bracteata*
Morus	1	0.529	*M. alba*
N/A	1	0.529	-
Persia	1	0.529	*P. americana*
Pittosporum	1	0.529	*P.tenuifolium*
Schinus	1	0.529	*S. terebinthifolius*
Vachellia	1	0.529	*V.sieberiana*
Washingtonia	1	0.529	*W. filifera*
Yucca	1	0.529	*Y. aloifolia*

Appendix G

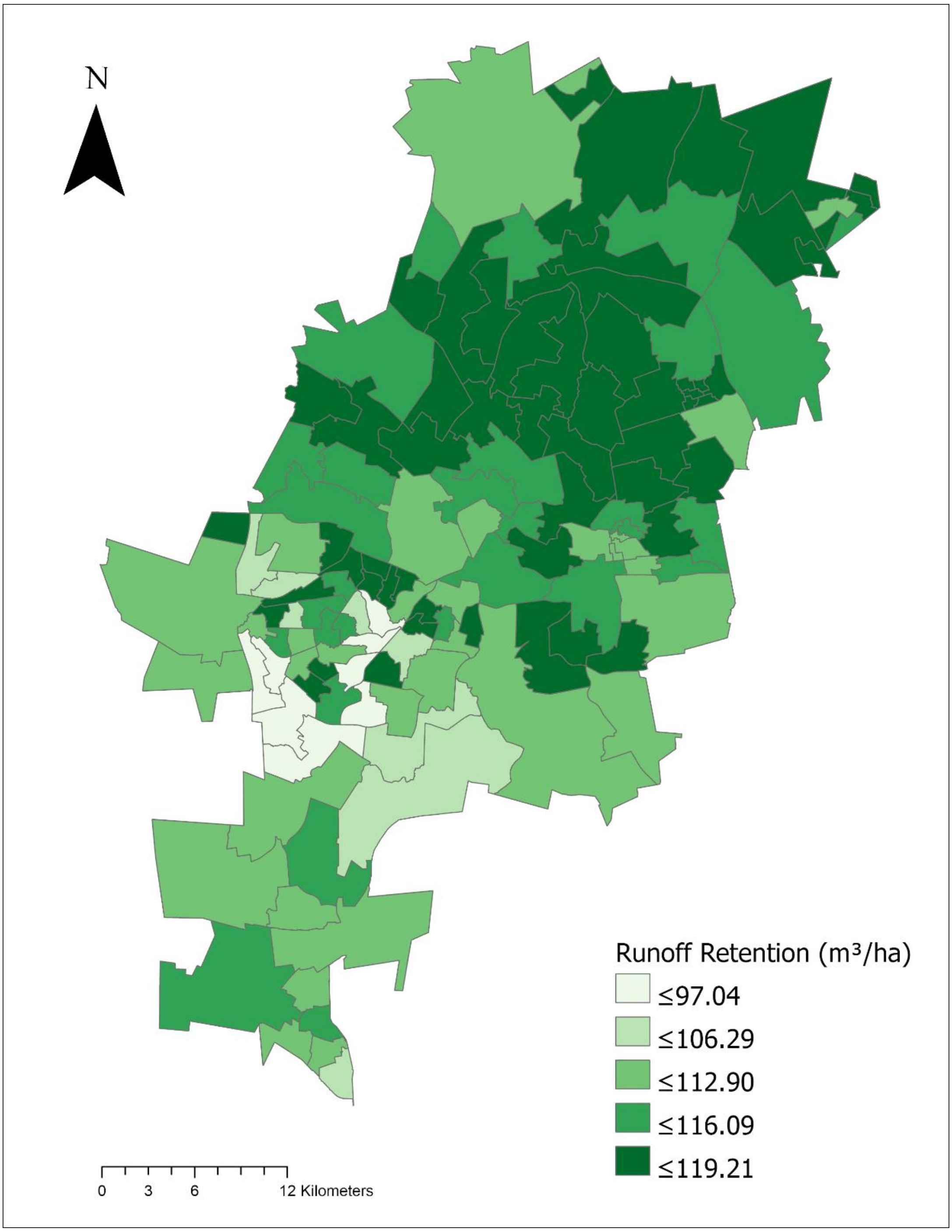

Figure G1: Average runoff retention across the administrative wards of Johannesburg, during a 12mm h^{-1} rainfall event.

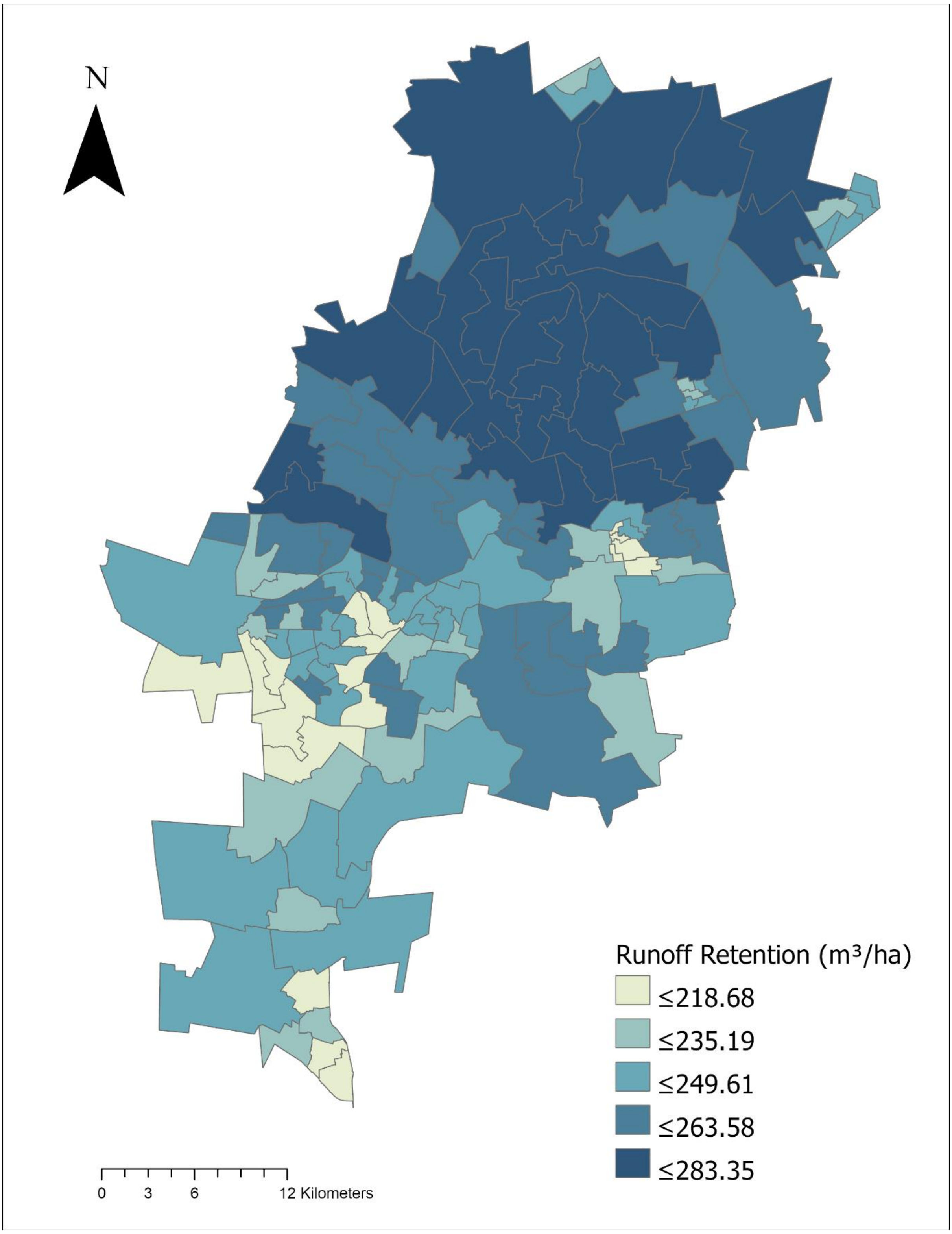

Figure G2: Average runoff retention across the administrative wards of Johannesburg, during a 30mm h^{-1} rainfall event.

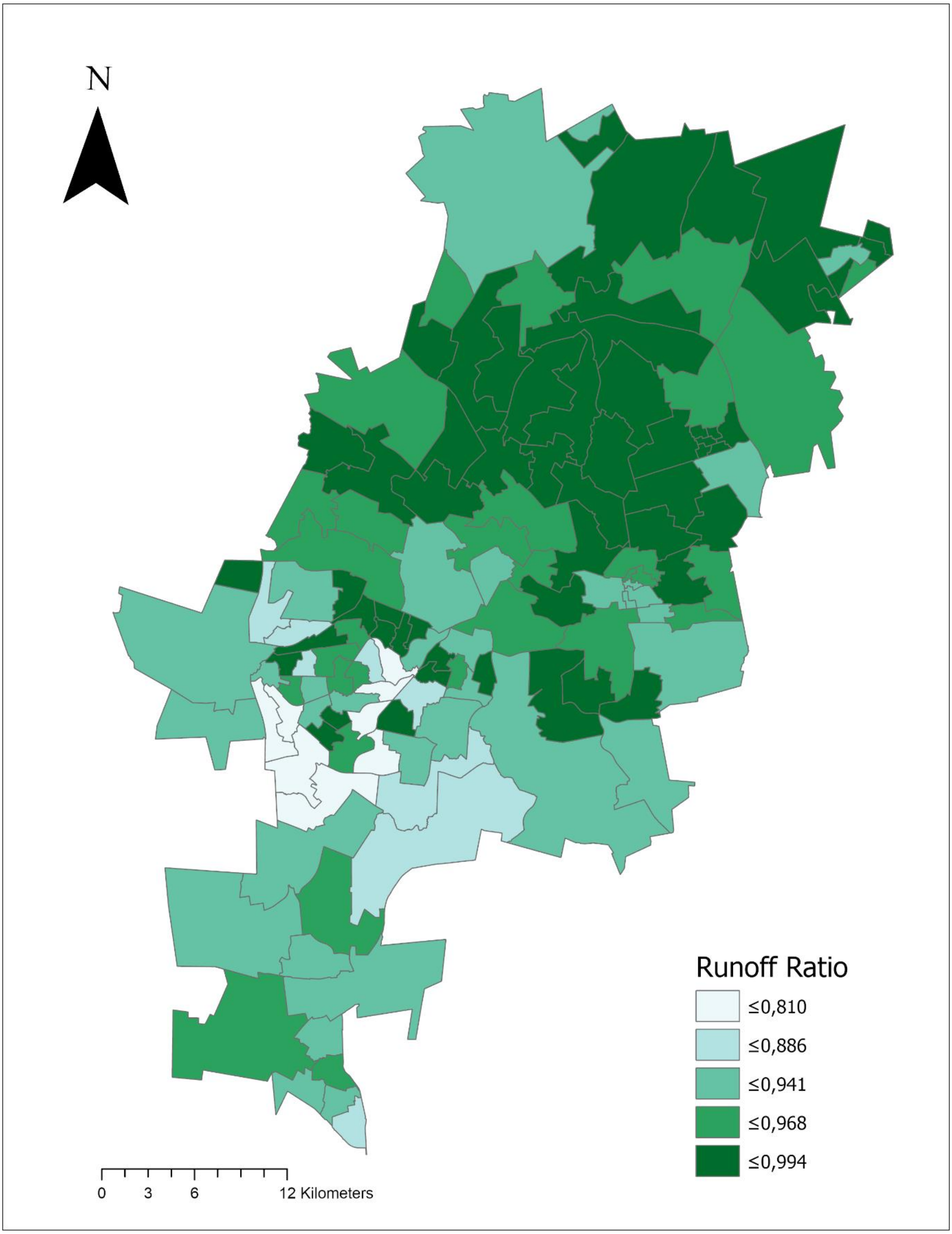

Figure G3: Average Runoff retention ratio across the administrative wards of Johannesburg, under a 12 mm h^{-1} rainfall event.

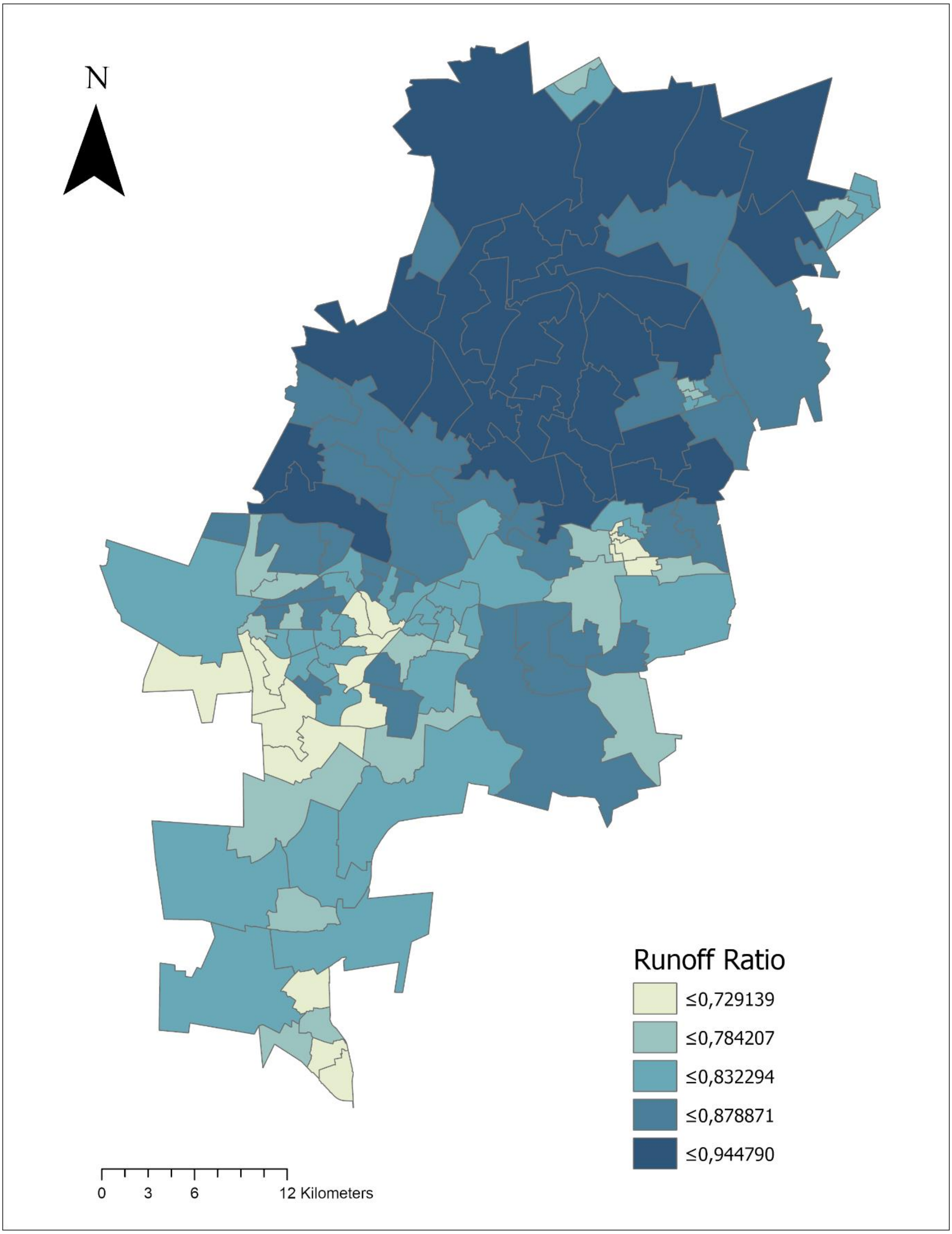

Figure G4: Average Runoff retention ratio across the administrative wards of Johannesburg, under a 30mm h^{-1} rainfall event.

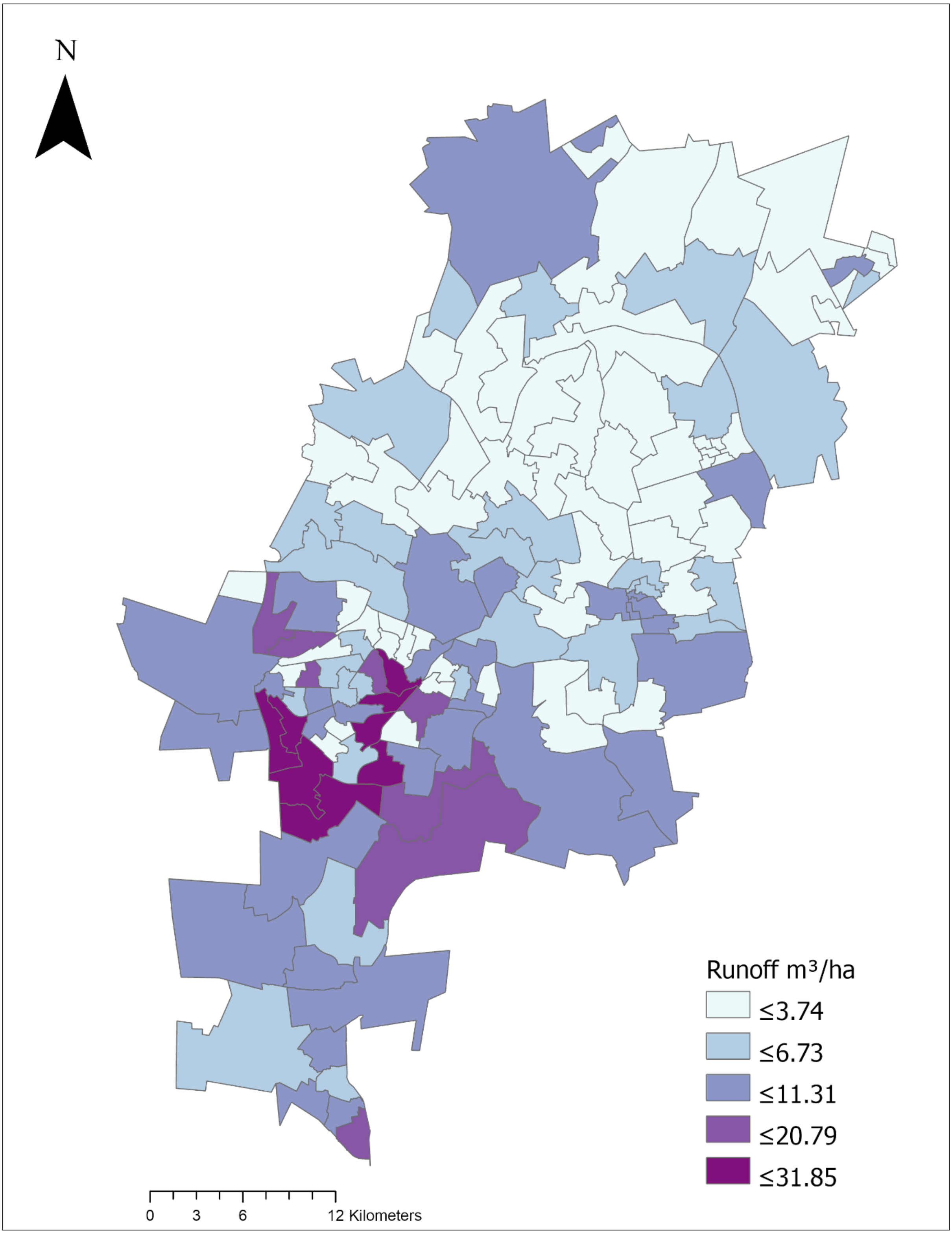

Figure G5: Average Runoff across the administrative wards of Johannesburg, under a 12 mm h^{-1} rainfall event.

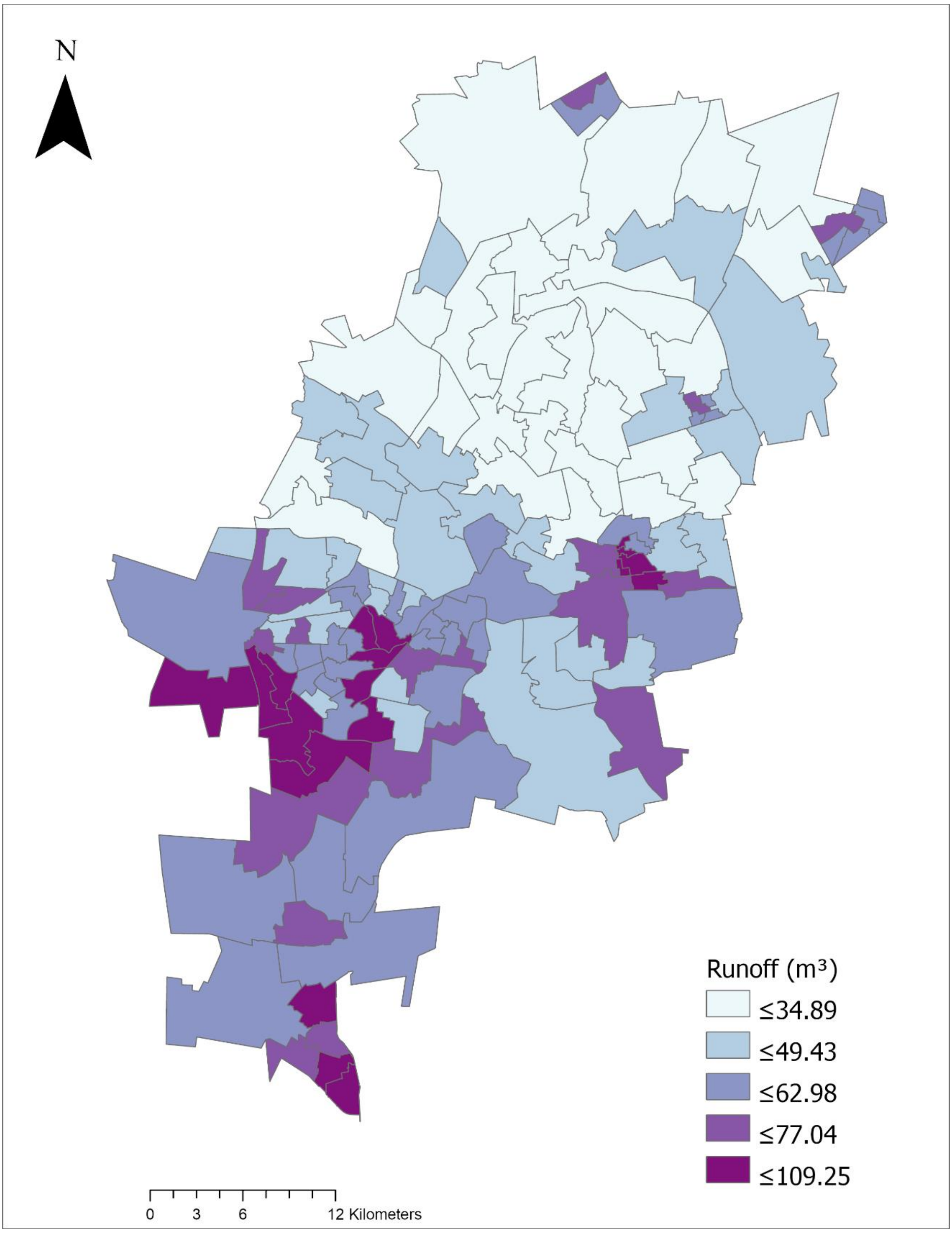

Figure G6: Average Runoff across the administrative wards of Johannesburg, under a 12 mm h^{-1} rainfall event.