

Impacts of Urban Intensification on the Production of Urban Ecosystem Services - A Case Study of the
Regional Municipality of Waterloo

by

Snehal Oswal

A thesis

presented to the University of Waterloo

in fulfillment of the

thesis requirement for the degree of

Master of Arts

in

Planning

Waterloo, Ontario, Canada, 2025

© Snehal Oswal 2025

Author's Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Abstract

As cities grow and intensify, the pressure on urban ecosystems to deliver essential services—such as clean air, water regulation, food production, and green space—continues to rise. In increasingly dense urban environments, the continuity and quality of these services are challenged not only by the reduction in green space but also by the form and pattern of urban growth. While compact development is often promoted as a sustainable alternative to urban sprawl, its ecological impacts remain contested, particularly in mid-sized urban regions like the Regional Municipality of Waterloo, where built forms and land use patterns are highly diverse and spatially uneven.

This study examines how different urban intensification patterns between 2013 and 2023 have influenced the production and spatial distribution of urban ecosystem services (UESs) across the Region. Using a 1 km² grid-based framework, urban areas are classified into typologies based on impervious surface ratios, and the degree of intensification is quantified using changes in building footprint and height. The analysis focuses on seven key UESs—food production, freshwater quantity and quality, air temperature regulation, air quality, stormwater runoff, and green space availability. By identifying trade-offs and synergies in their spatial behaviour through ecosystem service bundles, the study highlights how different urban landscapes exhibit distinct ecological patterns. Regression and correlation analyses further reveal how land cover changes and intensification levels drive these outcomes.

Findings underscore that the impacts of intensification are neither uniform nor linear; rather, they vary significantly across urban, peri-urban, and rural areas. While compact urban cores may show resilience in some services, they also face challenges like increased runoff and temperature rise. In contrast, sprawling developments often lead to fragmented ecosystems and more acute service degradation. These insights emphasise the need for spatially informed and context-sensitive planning strategies that recognise ecosystem interdependencies and aim to protect and enhance urban ecological functions amidst continued growth.

Acknowledgements

First and foremost, I would like to express my heartfelt gratitude to my supervisor, Prof. Marta Berbés-Blázquez, for your unwavering support, constant encouragement, and guidance throughout this research. Your belief in my work and your motivating presence, especially during the moments I needed it most, kept me moving forward with confidence and clarity.

I would also like to sincerely thank my committee member, Prof. Derek Robinson, for his generous support, technical expertise, and patient insights. It helped me navigate complex challenges and unknown technical territories.

To my partner, Anurag, thank you for being my rock, day and night. Your endless support, understanding, and quiet cheerleading behind the scenes meant more than words can say. Thank you for always being there and for lending an extra pair of eyes to my thesis when I needed them most.

Lastly, to my parents, thank you for making me who I am today. This journey would not have been possible without your lifelong support and belief in me.

Table of Contents

Author’s Declaration	ii
Abstract	iii
Acknowledgements	iv
List of Figures	ix
List of Tables.....	xi
Chapter 1 Introduction.....	1
1.1 Objectives.....	3
1.2 Research Questions	3
Chapter 2 Ecosystem Services in the Urban Context.....	5
2.1 Introduction	5
2.2 Urban Ecosystem Services	8
2.2.1 Provisioning Services	9
2.2.2 Regulating Services.....	12
2.2.3 Cultural services	17
2.2.4 Summarising Trends of Changing Urban Ecosystem Services in Rapidly Urbanising Cities	17
2.3 Urban Ecosystem Services Bundles	20
Chapter 3 Relevance of Urban Ecosystem Services in Urban Development	23
3.1 Urbanisation, Urban Typologies and Intensification as a Driver of Change.....	23
3.2 Understanding urban morphology and Varying Urban Typologies	24
3.3 Urban Intensification patterns and influence on Urban Ecosystem Services	25
3.4 Conclusion.....	28
Chapter 4 Case Study - The Regional Municipality of Waterloo.....	30

4.1 Introduction	30
4.2 Regional Overview	30
4.3 Rationale for Selection	33
4.4 Evolving Urban Typologies and Intensification Trends (2013–2023)	34
4.4.1 City of Kitchener: From Industrial Core to High-Density Centre	34
4.4.2 City of Waterloo: Institutional District and Mid-Rise Intensification:	35
4.4.3 City of Cambridge: Node-Based Growth and Heritage Preservation:	36
4.4.4 Rural Townships: Managed Growth and Agricultural Protection:	36
4.5 Summary and Analytical Significance	37
Chapter 5 Methods	38
5.1 Research Framework	38
5.2 Mapping Urban Ecosystem Services in the Regional Municipality of Waterloo	43
5.2.1 Food production (FP)	44
5.2.2 Freshwater Quantity	46
5.2.3 Freshwater Quality	48
5.2.4 Air Temperature Regulation	51
5.2.5 Air Quality	52
5.2.6 Stormwater Runoff Management	54
5.2.7 Green Spaces	56
5.2.8 Visualising Individual Urban Ecosystem Services on a 1km X 1km Grid	57
5.3 Spatial Assessment of Urban Intensification in the Regional Municipality of Waterloo	58
5.3.1 Mapping Urban Intensification	58
Chapter 6 Findings	60
6.1 Introduction	60

6.2 Urban Ecosystem Services	60
6.2.1 Crop Yield	60
6.2.2 Fresh Water Quantity - Ground Water Level	63
6.2.3 Fresh Water Quality	65
6.2.4 Air Temperature	68
6.2.5 Air Quality.....	71
6.2.6 Stormwater Runoff.....	72
6.2.7 Recreational Spaces.....	75
6.3 Urban Ecosystem Services Bundles	77
6.3.1 Bundle 1 – Suburban Bundle.....	77
6.3.2 Bundle 2 – Urban Core.....	77
6.3.3 Bundle 3 – Peri-Urban.....	78
6.3.4 Bundle 4 - Rural	79
6.4 Mapping the Urban Landscape and Intensification Patterns in the Region of Waterloo	80
6.5 Classifying and Visualising Urban Growth Clusters.....	85
6.6 Observing the Impacts of Varying Land Cover and Degree of Intensification in Urban Ecosystem Services Bundles	87
6.6.1 Multinomial Regression	87
Chapter 7 Discussion and Implications	95
7.1 Urban Growth and Ecological Trade-Offs at the Regional-Scale (2013 - 2023)	95
7.2 Trade-off and Synergies through UES Bundles	96
7.2.1 Rural versus Urban (Bundle 4 vs. Bundles 1, 2 and 3).....	97
7.2.2 Compact High-Rise versus Suburban Sprawl (Bundle 2 vs. Bundle 1)	99
7.2.3 Green Spaces in Sub-urban vs Peri-urban (Bundle 1 vs Bundle 3).....	100

7.3 Implications	101
Chapter 8 Conclusion	104
8.1 Limitations and Future Research.....	105
References	108
Appendix A	130
Deriving CN numbers: Overlaying MTO and SOLRIS CODES land use codes.....	130

List of Figures

Figure 1: Region Municipality of Waterloo - Cities and Townships	30
Figure 2: An aerial view of the City of Cambridge (Historic Galt Region)	31
Figure 3: An Aerial Photo of downtown Kitchener, looking northeast.....	31
Figure 4: Region of Waterloo Urban and Growth Centres.....	32
Figure 5: City of Kitchener Urban Structure	34
Figure 6: City of Waterloo Height and Density Policies.....	35
Figure 7: City of Cambridge Urban Land Use	36
Figure 8: Research Framework	42
Figure 9: Steps to Quantify Food Production.....	46
Figure 10: Steps to quantify Fresh Water Quantity	48
Figure 11: Steps to quantify Freshwater Quality.....	50
Figure 12: Steps to quantify Air Temperature.....	52
Figure 13: Steps to Quantify Stormwater Runoff.....	56
Figure 14: Maps for crop yield: (a) Map for crop yield in 2013; (b) Map for crop yield in 2023; (c) Map showing the change in crop yield.....	61
Figure 15: Distribution of change in crop yield	62
Figure 16: Distribution in farmland area change.....	63
Figure 17: Distribution of change in groundwater levels	63
Figure 18: Maps for groundwater level: (a) Map for groundwater levels in 2013; (b) Map for groundwater levels in 2023; (c) Map showing the change in groundwater level.....	64
Figure 19: Distribution of change in freshwater quality index.....	65
Figure 20: Change in average values of individual contaminants.....	66
Figure 21: Maps for freshwater quality: (a) Map for freshwater quality in 2013; (b) Map for freshwater quality in 2023; (c) Map showing the change in freshwater quality	67
Figure 22: Distribution of change in air temperature rise	68
Figure 23: Variation in air temperature among the municipalities.....	68
Figure 24: Maps for air temperature rise: (a) Map for air temperature in 2013; (b) Map for air temperature in 2023; (c) Map showing the change in air temperature	70
Figure 25: Graph showing change in the number of days in each AQHI.....	71
Figure 26: Graph showing the change in daily AQHI for 2013 and 2023	71

Figure 27: Change in average annual rainfall and surface runoff in 2013 and 2023.....	72
Figure 28: Maps for stormwater runoff: (a) Map for stormwater runoff in 2013; (b) Map for stormwater runoff in 2023; (c) Map showing the change in stormwater runoff.....	73
Figure 29: Distribution of change in stormwater runoff.....	74
Figure 30: Distribution of change in annual average rainfall	74
Figure 31: Distribution of change in green spaces	75
Figure 32: Maps for green spaces: (a) Map for green spaces in 2013; (b) Map for green spaces in 2023; (c) Map showing the change in green spaces	76
Figure 33: Urban Ecosystem Services Bundles.....	78
Figure 34: Graph showing change in urban parameters	80
Figure 35: Maps for impervious surfaces, built floor area and building volume in 2013 (a1, b1, c1) and 2023 (a2, b2, c2).....	82
Figure 36: Maps for change in impervious surfaces (a), built floor area (b) and built volume (c)	83
Figure 37: Distribution of change in impervious surfaces, built floor area and built volume.....	84
Figure 38: Map showing classification of urban clusters	86
Figure 39: Graph showing distribution of grid.....	86
Figure 40: Change in green spaces in each bundle between 2013 and 2023.....	99

List of Tables

Table 1: Indicators of Urban Ecosystem Services	19
Table 2: Urban Typologies and Densification Patterns.....	26
Table 3: Primary Data List	43
Table 4: Unit of Measurement for Quantified Urban Ecosystem Services	57
Table 5: Correlation among urban ecosystem services within each bundle	79
Table 6: Regression analysis for Bundle 1	89
Table 7: Regressions for Bundle 2	91
Table 8: Regressions for Bundle 3	92
Table 9: Regressions for Bundle 4	93
Table 10: Bundles and land-use	97

Chapter 1

Introduction

Urbanisation is one of the most transformative processes shaping the world today. Over the past 40 years, the global urban population has more than doubled, and by 2050, it's expected that nearly two-thirds of people worldwide will live in cities (OECD, 2020; UN, 2018). This rapid growth presents numerous opportunities but also poses significant challenges, particularly for the ecosystems within and around urban areas that provide vital services essential to human well-being. As cities expand and densify, they alter natural landscapes and disrupt essential ecological services, including water cycles, biodiversity, and climate regulation (Elderbrock et al., 2020). The resulting fragmentation and loss of green spaces put extra strain on these ecosystems, threatening their ability to deliver clean air, regulate temperature, manage stormwater, and more.

Urban ecosystems include parks, street trees, urban forests, wetlands, ponds, and other natural areas within and near cities. From an anthropocentric perspective, these spaces provide a range of ecosystem services for human well-being—from producing food and purifying air, to managing water, supporting biodiversity, and offering places for recreation and connection with nature (Costanza et al., 1998; Bolund & Hunhammar, 1999; Gómez-Baggethun & Barton, 2013). Understanding how these services are produced, interact and co-occur across the urban landscape is crucial, which is where the concept of urban ecosystem service (UES) bundles become important. UES bundles group together sets of services that tend to vary together, reflecting underlying patterns of land use, urban form, and ecological condition. Recognising these bundles helps reveal the complex trade-offs and synergies among provisioning, regulating, and cultural services in different urban contexts. As cities grow and change, these bundles shift in response, highlighting vulnerabilities and resilience in ecosystem service delivery. However, rapid and uneven urban growth increasingly threatens the stability and availability of these essential services, underscoring the need to consider the spatial dynamics of UES bundles when planning sustainable and resilient urban futures.

The Region of Waterloo in Ontario provides a clear example of this tension. As one of Canada's fastest-growing regions, it faces the challenge of accommodating population growth while trying to limit urban sprawl. Provincial policies like the Growth Plan for the Greater Golden Horseshoe (2005) and the Regional Official Plan (2012) promote compact, efficient urban development as a way to protect farmland and natural areas. But the real-world impacts of these intensification strategies on

urban ecosystem services are still not fully understood, especially at local and regional scales. While higher density development can theoretically reduce pressure on surrounding natural lands, it can also create localised environmental stress—such as worsening heat islands, water quality decline, and loss of green space—that undermine ecosystem functions and human well-being (Barthel et al., 2019).

Urban intensification means more people living in the same space through taller buildings, infill development, and mixed land use. On a broad scale, this can reduce sprawl and conserve open space. But the local effects depend heavily on how intensification unfolds—whether green infrastructure is preserved, how impervious surfaces increase, and how land cover changes. For instance, expanding concrete surfaces reduces groundwater recharge and increases stormwater runoff, while losing vegetation raises temperatures and reduces air quality. The net impact on ecosystem services is complex and varies by place, depending on existing ecological conditions and development patterns.

This study looks closely at how different urban intensification patterns in the Region of Waterloo between 2013 and 2023 have affected key ecosystem services. The region’s mix of urban cores, peri-urban, and rural areas offers a rich setting to explore how different development styles shape ecological outcomes. Using spatially explicit data on food production, water quantity, air temperature, water quality, green space, and stormwater management, the research investigates the trade-offs and synergies that come with sprawling suburbs versus compact high-rise developments.

What emerges is that urban intensification doesn’t have a one-size-fits-all impact. Suburban sprawl tends to fragment green spaces, raise temperatures, and degrade water quality, while compact, high-rise clusters show more resilience but still struggle with increased runoff and limited green cover. Beyond city limits, these changes ripple into surrounding rural and peri-urban areas, underscoring the interconnectedness of urban and regional ecosystems. These findings highlight the need for planning that goes beyond simple green space targets to consider landscape context, ecosystem health, and service interdependencies.

Adding to this complexity is climate change, which alters rainfall patterns, increases heat waves, and threatens biodiversity. Together with urban growth, these forces raise the stakes for managing urban ecosystem services wisely. Cities face growing risks from floods, droughts, and heat stress that can only be addressed by integrating ecological thinking into urban development strategies.

1.1 Objectives

The objectives of this study is to compare ecological impacts across different urban landscapes, such as urban, peri-urban, and rural regions.

- To assess the impact of urban intensification on the spatial distribution and production of key urban ecosystem services (UESs)—including food production, water quality and quantity, air temperature regulation, stormwater runoff, and green space—in the Region of Waterloo from 2013 to 2023.
- To classify urban ecosystem service bundles and analyse how different combinations of provisioning, regulating, and cultural services co-occur, revealing underlying patterns of land use, ecological condition, and urban form.
- To investigate the trade-offs and synergies among urban ecosystem services within these bundles, and how they relate to different intensification strategies such as suburban sprawl and compact high-rise development.
- To evaluate the role of land cover and green space change—particularly the expansion of impervious surfaces and loss of vegetation—in shaping the preservation or degradation of other ecosystem services.

1.2 Research Questions

The major aim of this study is to uncover the impacts of rising urban intensification in the Regional Municipality of Waterloo, specifically at regional and local scales. Further, this research investigates:

- How does intensification in different urban regions impact the spatial distribution and performance of the urban ecosystem services in the study area?
- What trade-offs and synergies emerge among the provisioning, regulating and cultural ecosystem services across distinct urban ecosystem service bundles, and how do these relate to the varying land cover changes and intensification patterns?
- How do varying degrees of land cover and green cover change impact the other urban ecosystem services? Does this degree of change differ in varying urban landscapes?

By unpacking how urban intensification shapes ecosystem service delivery across different landscapes, this study aims to provide actionable insights for sustainable urban growth. Understanding where trade-offs happen, where synergies can be strengthened, and how urban form influences ecological outcomes will help planners and policymakers balance development with ecosystem health. Ultimately, this research supports building cities that not only accommodate growth but also protect and enhance the natural systems that sustain them.

Chapter 2

Ecosystem Services in the Urban Context

2.1 Introduction

Urban Ecosystem Services (UES) are not just shaped by nature, but they are also continuously reconfigured by the form, density, and expansion of the cities themselves. Rather than viewing ecosystem services as passive benefits derived from nature, recent work highlights how urban morphologies and growth trajectories determine which services are sustained, traded off, or lost. Among these, *urban form and intensification* emerge as critical but unevenly understood drivers, altering how services are produced, distributed, and degraded across cities. This chapter reviews the growing body of literature that links changes in land use, built form, and spatial density to the functioning of urban ecosystems.

As an initial step, I define urban ecosystem services (UES) and outline the framework for narrowing the scope of this inquiry. Following definitions from Bolund and Hunhammar (1999) to recent studies (Caprioli et al., 2020; Veibiakkim et al., 2025), UES are the services generated by ecosystems within urban boundaries. Examples include urban forests, water bodies, wetlands, parks, gardens, street trees, etc. The U.S. Environmental Protection Agency (EPA) defines an ecosystem as the interactions among living organisms and their physical environment. In this study, I focus on non-human biotic elements. These are living components of the urban environment that exclude humans, such as vegetation, aquatic fauna, and soil microorganisms. They are rooted in natural systems and directly or indirectly sustain human well-being in cities.

UES can be further classified into provisioning (food and freshwater production), regulating (regulation of microclimate and air quality), cultural (availability and access to recreational spaces, sense of place, and cultural identity), and other supporting services (habitat and biodiversity conservation). (Shao et al., 2023; Caprioli et al., 2020; Kang et al., 2020; Gómez-Baggethun et al., 2013; Bolund & Hunhammar, 1999). Even assessing a single UES (whether provisioning, regulating, cultural, or supporting) is analytically challenging, let alone understanding how these services interact, co-evolve, or conflict within the broader urban system. This complexity arises because each aspect of an ecosystem service, ranging from production and demand to public perception, is shaped by distinct ecological, institutional, and socio-cultural processes (Langemeyer et al., 2024). These

processes are further influenced by the social, political, infrastructural, and governance systems embedded within cities (McPhearson et al., 2022).

Yet despite their differences, all services share a common dependency on transformations in the natural environment and ecological networks, which are deeply entangled with the urban built form. The *first scope* condition of this study is to limit my focus to *drivers associated with urban intensification*, particularly changes in land use, built form, and spatial intensification, rather than attempting to account for the full range of global or regional ecological influences on UES. A *second scope* condition concerns the *spatial scale* of analysis. While cities depend on ecosystem services generated both within and beyond their boundaries (Folke et al., 1997; McPhearson et al., 2022), this thesis confines its focus to services produced within urban areas. It does not attempt to model cross-boundary ecological flows or global socio-ecological linkages, which remain difficult to quantify due to geopolitical, institutional, and data-related constraints. Instead, it examines how *local* ecological changes, especially those shaped by urban form and the built environment, affect the capacity of cities to generate ecosystem services.

I conceptualise urbanisation as a multi-dimensional process comprising of a) spatial expansion, the outward growth of the built-up urban footprint beyond its existing boundaries; b) physical densification, the intensification of the built environment through increased building coverage, height, or land-use efficiency; and c) demographic concentration, the growth of population residing within urban settlements. Together, these three dimensions capture the spatial, physical, and demographic transformations that define urbanisation as observed in contemporary cities. Urbanisation, overall, affects the production of UES through both proximate and systemic pathways. Among the proximate drivers, there are two pathways. First, urbanisation places growing pressure on the supply and demand for ecosystem services, undermining the quality as well as the productive capacity of urban ecosystems. Second, it drives the expansion of built infrastructure and accelerates changes in land use and land cover, reshaping the physical and ecological fabric of cities (Sang et al., 2021). While managing demographic concentration is essential, the marginalisation of ecological infrastructure in urban development has long-term consequences (Lyu et al., 2018). At a systemic level, urbanisation interacts with broader forces such as climate change and biodiversity loss, compounding ecological degradation. Climate change has intensified the frequency and severity of extreme weather events, which in turn disrupt urban ecosystems and further diminish their service capacity. These impacts, though profound, unfold gradually and are often overlooked or deprioritised in urban planning

processes that tend to prioritize short-term development imperatives (Kang et al., 2020). Importantly, these drivers operate at different scales: while climate change functions at a global level, land-use decisions and built form transformations are governed locally and regionally.

Changes in urban infrastructure and land use/land cover (LULC) have become among the most significant local drivers of socio-ecological transformation (Rees & Wackernagel, 1996; Van der Biest et al., 2020; Wu et al., 2020a, 2020b; Vialatte et al., 2019; Wu, 2013, 2014; Young et al., 2020; Zhou et al., 2019a). Understanding their effects is essential not only for ecological resilience but also for integrating sustainability into urban planning frameworks that too often overlook the ecological consequences of built form and spatial structure. These shifts fundamentally reshape the structure and function of urban ecosystems, making it essential to consider the ecological consequences of development density and spatial intensification. Integrating these concerns into urban planning is critical for building cities that are not only sustainable and climate-resilient, but also socially just (Woodruff & BenDor, 2016).

Although the relationship between urban form and ecosystem services is well developed in theoretical and conceptual work, its integration into planning practice remains limited (Valencia Torres et al., 2021; McDonald et al., 2020). Environmental interventions tend to prioritise landscape design and green infrastructure, while the ecological implications of built form, development density, and spatial configuration receive considerably less attention (Sokolova et al., 2024; Cortinovis & Geneletti, 2018). Yet growing empirical evidence points to the uneven and context-specific impacts of urban intensification on ecosystem services. Across different urban settings, the ecological effects of densification vary significantly. In Singapore, rapid densification has resulted in the loss of natural vegetation and biodiversity despite efforts to integrate green infrastructure, underscoring persistent trade-offs not only within ecosystem services but also between ecological outcomes and spatial compactness (Tan & Abdul Hamid, 2014; Tan et al., 2021; Husain, 2024). In contrast, temperate cities such as Zurich and regions of Britain have seen reduced pressure on provisioning services, but often at the expense of regulating services such as microclimate control and air quality (Grêt-Regamey et al., 2020; Eigenbrod et al., 2011). In the United States, denser urban cores have experienced heightened flood risks and poorer air quality due to increasing impervious surfaces, while low-density sprawl in the Midwest has led to the loss of agricultural land and carbon storage capacity (Miller, 2012; Mohammadyari et al., 2023).

These cases reflect a broader pattern: while urban intensification often generates trade-offs between provisioning and regulating services. Yet, the direction and magnitude of these effects are highly dependent on local ecological conditions, urban form, and governance scale. The next sections identify the urban ecosystem services relevant for this study and their behaviour in urbanising regions. Further, in this chapter, the inter-dependencies of these services are discussed, highlighting the importance of ecosystem services bundles. Finally, the drivers of change for these services are discussed, specifically for the urban context.

2.2 Urban Ecosystem Services

UES was first introduced by Bolund and Hunhammar (1999), and the field saw a significant growth in publications during the early 2000s. Most scholars classify these services using the typology established by the Millennium Ecosystem Assessment (MEA, 2005), which distinguishes between provisioning, regulating, cultural, and supporting services—a framework that remains the most widely used in urban ecosystem research (Caprioli et al., 2020). While alternative assessment frameworks, such as that proposed by IPBES (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services), offer more nuanced accounts of the linkages between ecosystem service flows and human well-being, the MEA typology is retained in this thesis for its conceptual clarity and parsimony, which align well with the study's empirical objectives.

This thesis focuses specifically on the *production aspect* of UES, rather than assessing demand, perception, or patterns of service-use. The services examined include food and freshwater provision (provisioning services); temperature regulation, freshwater purification, air quality improvement, and stormwater control (regulating services); and access to recreational green spaces (cultural services). Supporting services—such as soil formation and biodiversity maintenance—are not included due to the lack of openly available and spatially explicit data. Prioritizing production of UES offers a clear and measurable lens, but critical questions around who benefits from ecosystem services, under what conditions, and with what forms of exclusion or inequality are kept outside the scope of the study for tractability.

Even within the production of UES, the indicators used to measure UES varies greatly across studies, depending on their specific goals, scales, and data availability. For instance, some assessments rely on broad measures—such as total green space at the city or regional level—while

others focus on finer-grained indicators like tree canopy cover at the neighbourhood or block scale (Söderman et al., 2012). The availability and resolution of spatial data, whether coarse or fine, also play a significant role in shaping indicator selection. Most crucially, the choice of indicators is shaped by the study's overarching aim. In this thesis, I prioritise indicators that capture the influence of the urban built environment on ecosystem service provision. I use measures such as impervious surface area, land cover change, built form typologies, and building volume density to analyse how urban intensification affects the capacity of cities to produce ecosystem services.

In this thesis, I evaluate the impact of urbanisation on ecosystem services at a regional scale, using the Region of Waterloo as a case study. I draw on publicly accessible environmental and urbanisation data from municipal, provincial, and federal government sources. Although the available government data is relatively coarse, it remains appropriate for regional-scale analysis. In the sections that follow, I examine seven ecosystem services—food production, water quality, water quantity, air temperature regulation, stormwater runoff, and the availability of green space; and discuss their corresponding indicators in detail. I focus on their relationship with the urban built environment and highlight patterns of change identified in the existing literature.

2.2.1 Provisioning Services

2.2.1.1 Food Production

Urbanisation has direct and indirect impacts on food production systems. As urban regions are expanding, the agricultural lands, especially along the urban fringe, are rapidly being converted for urban uses. This conversion of highly productive farmlands leads to loss of fertile lands, impacting the crop yield in the region (Langemeyer et al., 2021; Barthel et al., 2019a; Artmann and Sartison, 2018) as well as the livelihood of the rural and peri-urban farmers (Dadi et al., 2023). Studies conducted globally consistently highlight the trends of urban encroachment, often on high-yielding agricultural lands owing to the desirable topography, existing access and favourable soil conditions. For instance, a recent study shows that the cropland lost to cities in the United States and Western Europe is 30-40% more fertile than any other land available for replacement in the region. (Food and Agriculture Organisation, United Nations, 2023; Andrade et al., 2022) Bren d'Amour et al. (2017) projected the highest losses of crop yield in North America along the peri-urban regions owing to the rapid urban expansion.

Along with the direct loss of farmland, urban development also influences food production through changes in soil quality, water resource availability and altering micro-climate (Lal, 2020). Infrastructure development, pollution from socio-economic activities and expansion of impervious surfaces degrade soil health and compromise water quality, leading to reduced crop yield in urban peripheries (Evans et al., 2022). Moreover, a rise in demand of water for growing urban population can lead to increasing competition for this resource, indirectly impacting irrigation and crop yield in the region (Cao & Wang, 2025).

The recent literature on urban growth and its impacts on food production emphasises the advantages of compact urban development in preserving farmlands over urban sprawl. Further, it also highlights the role of urban agricultural initiatives such as rooftop farms, community gardens or plots and technologically enhanced peri-urban farming to support local food systems in compact urban regions. While these initiatives can contribute to local food supply and offer several other social-ecological benefits, it remains an empirically unaddressed question whether their scale is sufficient to compensate for the broader loss of crop yield and farmlands caused by urbanisation, depleting resources and degrading micro-climate.

In summary, rising urbanisation, changing urban typologies and intensification exert significant, and more than often negative impacts on the crop yield and farmland area. These dynamics are mediated by direct land conversion, resource competition, environmental degradation, and evolving food demand, with the effects varying according to local context and urban form. While some opportunities exist for integrating food production into urban landscapes, the current trend is one of heightened pressure on agricultural systems and increased challenges for sustaining food security in an urbanising world.

2.2.1.2 Freshwater Quantity

Urbanisation exerts significant and direct pressure on natural freshwater resources, such as groundwater levels and river systems, which are essential ecological systems to sustain human well-being. As urban areas expand, permeable natural surfaces are increasingly replaced by impervious infrastructure such as roads, buildings, and parking lots. This further leads to declining groundwater recharge rates and eventually falling groundwater tables. Multiple studies estimates that urban areas often experience a 30-50% reduction in groundwater recharge compared to pre-urbanised conditions,

leading to lower water tables and, in some cases, complete depletion of shallow aquifers (Foster and Chilton 2003).

The consequences of falling groundwater levels are multifaceted. Beyond serving as a major source of potable water and irrigation, over-extraction combined with reduced recharge of groundwater can lead to land subsidence, increased pumping costs and intrusion of pollutants or saline water into freshwater aquifers. The phenomenon of land subsidence are more common in coastal cities such as Jakarta, which experienced significant land subsidence due to excessive groundwater withdrawal owing to expanding urban regions (Chaussard et al., 2014; Abidin et al., 2011). Further, many urban centres are now facing shortage of freshwater and degrading quality of groundwater tables owing to falling groundwater levels and reduced infiltration. For instance, cities such as Los Angeles and Houston have faced severe losses in groundwater levels and quality with increasing urbanisation (Scanlon et al., 2005). Urbanisation profoundly impacts river systems. Increased impervious surfaces lead to higher volumes of surface runoff into urban streams and rivers. This leads to severe flooding, channel erosion and degraded aquatic habitats. For example, another study highlights the hydrological changes in Eastern American cities experiencing increased peak flows and reduced base flows owing to urbanisation (Meyer 2001). The loss of base flows in rivers is directly impacted by the falling groundwater levels, due to declining recharge or over-extraction. The loss of base-flows reduces the reliability of river systems for potable water in dry seasons.

High-density and compact urban forms, with concentrated impervious surfaces, can impact both groundwater levels and river flows. Conversely, sprawling developments have reduced impacts but on a larger area, resulting in diffused yet significant impacts on groundwater recharge and the natural drainage patterns. Studies conducted in European and North American cities show severe impacts on the freshwater resources due to fragmented watersheds in sprawling urban developments (Walsh et al., 2005).

In summary, different forms of urban intensification have diverse impacts on freshwater provisioning. The extent and nature of these impacts also depend on urban form, population density, as well as local hydro-ecological conditions and management responses. Recognising this heterogeneity is crucial for designing urban strategies that minimise trade-offs between densification and the sustainability of freshwater resources.

2.2.2 Regulating Services

2.2.2.1 Freshwater Quality

Increasing urbanisation prominently alters the chemical composition of the underground aquifers and river streams. Growing cities and expanding impervious surfaces, not only disrupt hydrological cycles but also profoundly increase surface water runoff and reduced natural infiltration and purification of rainwater (Paul and Meyer 2001, Walsh et al 2005,). Furthermore, with increasing pollution and human activities, numerous water contaminants - including nutrients, heavy metals, hydrocarbons and pathogens - are washed along the surface runoff into natural freshwater resources (Marsalek, and Rochfort 2004). This degrades water quality as well as destabilises aquatic ecosystems (Goonetilleke et al 2005).

Studies of urban ecosystem services have identified increased concentrations of nutrients (nitrogen and phosphorus), increased salinity, acidic pH values and reduced dissolved oxygen in highly urbanised regions (Kaushal et al 2008). In Ottawa's Carp River watershed, modelling studies correlate an increase in surface runoff to rising concentrations of nitrogen and phosphorus in rapidly urbanising sub-watersheds (Zhang and Chen 2017). Similarly, in the Greater Vancouver Area, urban streams such as Still Creek have experienced declines in water quality - altered pH, reduced dissolved oxygen, and increased nitrogen concentrations - directly attributed to increasing urban surface runoff, pollution and replacement of natural streams with engineered drainage systems (Gippel and Blackham 2002). Moreover, across Canada, freshwater quality is degrading, with rural and Indigenous communities facing the brunt (Hanrahan et al 2014). Additionally, modern farming technologies, including stronger and efficient fertilisers, add to the infiltration of chemicals and nutrients into the groundwater aquifers, influencing the water quality.

Urban development patterns further shape these impacts. Dense compact developments exhibit higher concentrations of pollutants; however, these can be easily managed with centralised and efficient water treatment infrastructure (Roy et al 2008, Fletcher et al 2014). However, urban sprawls, characterised by low density and dispersed developments, exacerbate water quality problems across larger regions owing to widespread impervious surfaces and surface runoff (Booth & Jackson, 1997; Brabec et al., 2002). The highly spread developments also make stormwater management challenging due to a lack of centralised wastewater treatment and stormwater management facilities. Moreover, the relation between urban intensification and water quality is not uniform. It highly relies on the

correlation between the degree of urban intensity and the supporting environmental management and availability of infrastructure to reduce degradation of water quality. For instance, highly urbanised watersheds with robust water management facilities have shown an improvement in water quality, while relatively less urbanised but poorly managed infrastructure is found to be highly sensitive to changes in the urban built environment, negatively impacting water quality (Roy et al 2008).

In summary, urbanised regions tend to see a decline in water quality; however, dense and compact regions with efficient water management see a lower impact as compared to those in peripheral sprawls. Further, irrespective of the water management facilities, dense and compact regions impact smaller areas as compared to dispersed low density developments. Further, agricultural activities also substantially impact the quality of groundwater and river streams.

2.2.2.2 Air Temperature Regulation

Air temperature is one of the most visibly and immediately affected ecosystem services under urban intensification, primarily due to the urban heat island (UHI) effect. UHI refers to the consistent tendency of urban areas to be warmer than their surrounding rural counterparts. This phenomenon predominantly occurs due to high impervious surfaces, reduced vegetation and concentration of heat-absorbing (low albedo) materials such as asphalt and concrete in urban areas. As cities grow and intensify, land cover changes, where natural landscape is replaced by urban built surfaces, increase heat retention and reduce evapotranspiration, elevating day-time and night-time air temperatures. Temperature regulation is one of the highly studied and documented urban ecosystem services globally. Studies in cities such as Edmonton and Calgary show pronounced impacts of the UHI effect, with nighttime temperatures rising to 2.86°C and 2.89°C, respectively (Alam et al., 2024). Similarly, the City of Winnipeg has seen a dramatic increase in the number of hot days above 30°C (Environment and Climate Change Canada, 2025; Phillips, 2025). The impact of UHI is not limited by the size of cities, with similar impacts seen in intensifying urban regions irrespective of their size (Zhou et al., 2017; Stewart & Oke, 2012).

Different urban forms mediate UHI in distinct ways. Dense, compact developments often intensify UHI due to concentrated vertical (building volume) and horizontal (impervious surface) expansion. These typologies tend to have reduced vegetation cover, increased thermal mass, and fewer permeable surfaces, which elevate both peak and average annual temperatures (Li et al., 2020; Ma &

Peng, 2021). For instance, dense neighbourhoods in Montreal with limited green space consistently record higher temperatures than areas with more trees and vegetation (Wang & Akbari, 2016).

Yet low-density, sprawling developments are not exempt. Despite offering more green space per capita, their extensive built area, fragmented vegetation cover, and higher vehicular emissions contribute to diffuse but significant temperature increases (Ewing & Rong, 2008; Stone et al., 2010; Zhou et al., 2017). Thus, both typologies generate UHI effects, albeit through different mechanisms.

In summary, rising air temperatures are a near-universal feature of urbanisation, but their magnitude and spatial distribution vary with urban form, density, material use, and green space availability. Vegetation remains the most critical moderating factor. Evidence from Toronto, for instance, shows that green roofs and reflective materials can substantially reduce local UHI effects in dense urban areas (Bass et al., 2002). Ultimately, evaluating temperature regulation at regional scales hinges not only on the built environment but also on the extent, continuity, and management of urban green infrastructure.

2.2.2.3 Air Quality

Air quality in urban regions is significantly shaped by rising concentrations of key pollutants such as particulate matter (PM), nitrogen oxides (NO_x), sulphur dioxide (SO₂), and ozone (O₃). These changes are primarily driven by increased population density, industrial activity, and vehicular emissions. While urban areas consistently show higher pollutant levels than rural counterparts, the relationship between urban intensification and air quality is complex and context dependent. It is mediated by urban form, socio-economic activity, and the presence or absence of technologies like public transit and emission controls.

High-density, compact developments—when well-integrated with robust public transportation—can reduce per capita vehicle use and lower emissions of harmful pollutants such as PM, NO_x, and SO₂ (Zhang et al., 2022). However, in the absence of efficient transit systems, adequate green spaces, or effective ventilation corridors, densely built environments can trap emissions and reduce air circulation, exacerbating local pollution levels. This often results in spatially uneven exposure, with neighbourhoods near highways, industrial zones, and commercial centers facing the greatest pollution burdens.

Conversely, peri-urban sprawl may allow pollutants to disperse over larger areas, but typically results in poorer air quality overall due to greater reliance on personal vehicles. Urban design features such as green spaces, vegetative buffers, and permeable surfaces can help mitigate these effects. Ultimately, the impact of urban form on air quality is mediated by the quality of transit infrastructure, availability of green cover, and how well urban layouts support air flow and reduce transport-related emissions.

Proximity to highways, industrial zones, and limited availability of green spaces are consistently associated with reduced air quality in urban environments. Studies from cities such as London in Ontario, and Madison (United States) illustrate how neighbourhoods located near major traffic corridors or industrial areas experience higher pollutant concentrations (Brook et al, 2018; Kanaroglou et al, 2005; Berman and Fann, 2012). In dense urban zones, particularly during summer months, concentrated traffic further exacerbates pollution levels (Cyrus et al, 2003). However, improved public transit access and walkability can partially offset these effects over time (Hankey et al, 2012).

In summary, urban intensification influences air quality in complex ways. Denser areas often exhibit higher concentrations of pollutants due to concentrated traffic and limited air circulation. However, such areas may also enable mitigation through greater access to public transportation and walkability. In contrast, sprawling suburban regions tend to be more car-dependent, contributing to diffuse emissions and wider spatial distribution of air pollutants.

2.2.2.4 Stormwater Runoff Management

Stormwater runoff management is crucial to mitigate floods, recharge freshwater resources and maintain the freshwater quality. Growing cities and rising impervious surfaces obstruct the natural infiltration of rainwater, causing increased surface runoff, higher peak flows, and shorter runoff durations during storm events. Some studies show a dramatic tenfold increase in runoff volume and 19 times higher peak runoff rates in urbanised regions compared to undeveloped watershed areas (Hollis 1975). This not only increases the flood risk, but also disrupts the water ecosystems, causing erosion, sedimentation and loss of aquatic habitats (US EPA, 2015). Similarly, a study of peri-urban catchments demonstrates a direct correlation between increasing impervious surfaces and higher stormwater runoff, emphasising the critical role of land cover change in urban hydrology (Miller et al., 2014).

Further, the intensity of stormwater runoff and its impacts vary across different urban typologies and intensification patterns. High-density, compact regions with infill in existing urban boundaries further intensify the stormwater runoff. Concentrated impervious surfaces in high-density developments lead to more severe and frequent flooding if not mitigated by green infrastructure or a robust drainage system. In contrast, sprawling, low-density development spreads impervious surfaces over a large area, increasing the net volume of runoff, which often overwhelms the existing drainage systems.

These patterns are increasingly evident in Canadian cities, where stormwater management challenges have intensified with rising urbanisation and the spread of impervious surfaces. Cities such as Hamilton, and Halifax have faced recurrent issues of flooding, runoff surges, and drainage overloads, particularly in older urban cores with ageing infrastructure (Li and Zhang 2019, Hamilton Stormwater Management Master Plan, Halifax Stormwater Management Strategy and Action Plan). The expansion of impervious surfaces due to densification and suburban growth has exacerbated the disruption of natural hydrological cycles. However, several municipalities have begun integrating green infrastructure strategies to mitigate these pressures. For instance, bioswales—vegetated channels designed to capture and filter runoff—are being implemented to restore infiltration pathways and reduce the velocity of surface flows (Davis et al, 2009). Likewise, interventions such as green roofs, permeable pavements, and constructed urban wetlands have gained traction as adaptive tools in stormwater management planning (Fletcher et al., 2015; Liu et al., 2020). Empirical evaluations of these strategies across Canadian cities show promising results. Studies report that such green infrastructure interventions can reduce stormwater runoff volumes by 43–61%, depending on land cover characteristics, soil type, and design scale (Liu et al., 2020).

In summary, urban intensification alters stormwater runoff by increasing impervious surfaces and reshaping land cover. High-density areas concentrate runoff and heighten flood risks, while low-density sprawl disperses runoff over wider areas, often overwhelming drainage systems. Though both typologies pose distinct challenges, their impacts can be mitigated through well-implemented green infrastructure, such as bioswales, permeable surfaces, and green roofs. The effectiveness of stormwater management ultimately depends on how urban form and ecological design are integrated into planning.

2.2.3 Cultural services

2.2.3.1 Availability of Green Spaces

While green spaces play a functional role in the performance of most of the above-discussed services, they also represent vital cultural services. Urbanisation reshapes the landscape of recreational spaces, specifically the urban green spaces globally. As the urban population increases, the demand for land for housing, infrastructure and socio-economic activities leads to reduced and fragmented green spaces, significantly impacting the environmental quality, public health and human well-being. Urban green spaces, such as urban parks and gardens, community allotments, urban forests and recreational facilities, play a pivotal role in mitigating the negative effects of urbanisation on other urban ecosystem services (for example, air quality, urban heat island effect and stormwater runoff), while also providing essential recreational, cultural and psychological benefits.

Urban intensification is often linked with declining green cover and diminished access to recreational spaces. Yet, well-planned high-density developments that integrate green infrastructure can enhance both ecosystem service provision and human well-being. Compact urban forms can also reduce ecological pressure beyond city boundaries by limiting land conversion. In contrast, urban sprawl may offer larger private green areas but often results in fragmented public green spaces with lower accessibility.

Empirical evidence from cities such as London, Michigan, and Madison underscores the risks of green space loss amid urban growth and infill, prompting policy shifts toward green infrastructure and preservation strategies (Benedict & McMahon, 2012; Madison Sustainability Plan, 2023; London Urban Forest Strategy Monitoring Report, 2024). In sum, while green spaces are central to sustainable urban systems, urbanisation and intensification continue to threaten their availability, accessibility, and quality.

2.2.4 Summarising Trends of Changing Urban Ecosystem Services in Rapidly Urbanising Cities

Urbanisation constitutes a primary exogenous driver of variation in urban ecosystem service provision. Although a prevailing expectation from the broader literature is that increasing urban intensification degrades the functioning of urban ecosystems, this relationship is not always uniform. Rather, it is conditioned by a range of contextual variables, including local climatic and socio-

economic conditions, the spatial configuration of urban development (e.g., compact versus dispersed forms), and the presence or absence of mitigation strategies such as green infrastructure integration.

Compact, high-density urban developments can reduce the amount of land that cities take up and limit the spread of hard, impervious surfaces. This can help protect key provisioning services like food production and freshwater supply. However, if green infrastructure is not properly included in these dense areas, other ecosystem services tend to suffer. These include regulating services like cooling the air, cleaning water and air, and managing stormwater, as well as cultural services such as access to parks, places for recreation, and overall mental well-being. The ability of green infrastructure to reduce the negative effects of urbanization depends on several factors. These include the type of urban development (such as compact or sprawling), how intensification is planned (for example, through infill or high-rise buildings), and how well the green infrastructure is maintained over time. If these factors are not carefully managed, green infrastructure may not be effective in protecting urban ecosystem services.

In contrast, sprawling and low-density developments increase the amount of land and resources used by each person. These patterns often require more extensive road networks, energy infrastructure, and water delivery systems, which further strain natural resources. The physical spread of built-up areas also fragments green spaces, reduces connectivity between ecological habitats, and limits opportunities for efficient public service delivery. Over time, these dynamics contribute to long-term declines in the provision of urban ecosystem services such as stormwater regulation, air and water purification, and recreational access.

In conclusion, the impact of urbanisation on the production of urban ecosystem services is uneven and cannot be categorically defined as either positive or negative. Its effects depend heavily on local context, the patterns and forms of development, and the scale of analysis. To better understand these dynamics, it becomes important to distinguish between different types of urbanisation processes—particularly along a densification gradient—as each form (e.g., compact infill versus peripheral sprawl) may exert distinct pressures on local ecosystem services.

Given this variability, the selection of appropriate indicators becomes especially important for distinguishing between different types of urban densification, as well as for accurately measuring UES. The table (2-1) below outlines the UES indicators chosen for this study, reflecting considerations of contextual relevance, spatial scale, and the availability and resolution of data. To

avoid misinterpretation in cases where baseline values are small, I report both the absolute change (in units) and the percentage change for site, where useful. This enables cross-site comparisons that account for both relative and actual magnitude of change.

Table 1: Indicators of Urban Ecosystem Services

Urban Ecosystem Services	Indicator/s	Key Citations
<i>Provisioning Services</i>		
Food Production	- Change in Crop Yield (% change) - Change in Area (% change)	Haines-Young et al. 2012; De Groot et al. 2010; Grima et al. 2023
Freshwater Quantity	- Change in Groundwater level - Change in Impervious Surfaces	Nedkov & Burkhard 2012; Egoh et al. 2012; Grima et al. 2023
<i>Regulating Services</i>		
Freshwater Quality	- Changes in Contaminant Concentration - Salinity - Nitrates - PH Values - Fluorides - Arsenic - Manganese - Economic activities (urban or agricultural)	Paul & Meyer 2001; Egoh et al. 2012; Grima et al. 2023
Air Temperature Regulation	- UHI Intensity (°C change in air temperature)	Davies et al. 2011; Maes et al. 2011a; Egoh et al. 2012

	- Green and Blue space coverage (% change)	
Air Quality	- Change in Pollutant Concentration (change in Air Quality Health Index) PM 2.5 NO2 O3	Grima et al. 2023; Egoh et al. 2012; Deeksha et al. 2022; Ontario Ministry of Public health, 2022
Stormwater Runoff Management	- Change in Runoff Volume (m3/s) - Annual Maximum Temperature (m2/s) - Maximum Retention Capacity (Soil Hydrology)	Nedkov & Burkhard 2012; Egoh et al. 2012; Grima et al. 2023
<i>Cultural Services</i>		
Availability of Green Spaces	- Green Space (% change)	Kienast et al. 2012; Milcu et al. 2013; Egoh et al. 2012

2.3 Urban Ecosystem Services Bundles

Urban ecosystems often provide multiple interconnected benefits rather than isolated functions, making it important to understand how these services occur together. A single urban park, for instance, can simultaneously offer recreational opportunities, regulate air temperature and quality, manage stormwater runoff, and support biodiversity within an urban environment (Gómez-Baggethun & Barton, 2013; Haase et al., 2014). These benefits are rarely delivered in isolation; rather, they co-occur as part of *ecosystem service bundles*—sets of services that repeatedly appear together in space and time due to shared drivers, ecological processes, or management regimes (Bennett et al., 2009). Recognizing these bundles is essential, as it shifts the focus from individual services to the interactive and cumulative impacts of urban ecosystems on human well-being (Raudsepp-Hearne et al., 2010;

Valencia Torres et al., 2021). The concept of ecosystem service bundles also helps identify potential synergies and trade-offs among services. For example, a reduction in urban green space can simultaneously impair temperature regulation, air purification, and stormwater management demonstrating a synergistic decline in multiple regulating services. In contrast, an increase in agricultural output from intensified farming practices may lead to water pollution, illustrating a trade-off between provisioning and regulating services (MEA, 2005).

Mapping ecosystem service bundles allows us to see how different services change together across space. When multiple services increase or decrease in the same area, it often signals that they are being influenced by common external factors—such as land use decisions, infrastructure development, or environmental policies. Literature suggests urban cores and high-density neighbourhoods tend to show different UES relationships compared to peri-urban or rural regions. These spatial differences help identify how changing urban forms, land cover, and intensification are shaping the production and distribution of ecosystem services. (Hu et al., 2025; Saidi, 2021; Alvarado, 2020; Bennett et al., 2009)

Moving beyond individual synergies and trade-offs, the ecosystem services bundle approach has important implications for urban planning and governance. Planners and policymakers often address stormwater management, green space provision, and public health as separate issues. The bundle framework challenges this separation by showing how multiple services change together and interact with one another. It provides three major insights.

First, the bundle approach helps identify unintended consequences of narrowly focused policies. For example, a policy that promotes food production may unintentionally harm water quality or reduce natural habitat. These trade-offs can be overlooked when services are considered in isolation. Second, bundles reveal patterns of socio-environmental inequality. When certain neighbourhoods consistently experience declines in multiple services—such as poor air quality, limited green space, and inadequate drainage—it may point to deeper problems of environmental injustice. Recognizing these clusters can help target resources and interventions more fairly. Third, this approach helps explain why similar forms of urban development lead to different results in different contexts. For instance, densification might improve or harm services depending on local factors like existing green infrastructure, soil type, or neighbourhood income levels. By mapping service bundles, researchers

and decision-makers can better understand local conditions and respond with place-specific strategies that improve both ecosystems and human well-being.

Chapter 3

Relevance of Urban Ecosystem Services in Urban Development

3.1 Urbanisation, Urban Typologies and Intensification as a Driver of Change

Despite growing interest in sustainable urbanism, a significant knowledge gap remains in understanding how different forms of urban intensification affect the production of ecosystem services (UES). Global urban policy increasingly promotes compact growth (OECD, 2018; Ahlfeldt & Pietrostefani, 2017). However, it remains empirically unclear whether such growth leads to better ecological outcomes overall. Some services, such as food and freshwater provisioning, may benefit from a reduction in urban sprawl. In contrast, regulating services like air quality, stormwater control, and temperature moderation often decline. This decline is especially common when cities fail to invest in parallel green infrastructure. These uneven effects show the importance of moving beyond a focus on individual services. Researchers need to assess how different forms of intensification affect bundles of services that occur together and shape urban well-being.

The Millennium Ecosystem Assessment (MEA, 2005) identifies urbanization—particularly its dimensions of spatial expansion and physical densification—as major drivers of ecosystem change. Their effects are especially visible at city and regional levels. Much of the existing literature focuses on the negative impacts of horizontal growth. This includes outward expansion and land conversion. However, this approach often overlooks vertical growth and changes in urban form. Recent studies show that the type of urban form has a major impact on ecosystem service outcomes (Dey et al., 2024; Ghosh & Sk., 2024; Grêt-Regamey et al., 2020; Liu et al., 2020). High-rise, mid-rise, and low-rise forms each influence ecosystem services differently. Sprawling low-density areas tend to use more land and resources. But dense areas can also cause concentrated environmental stress if they lack adequate green space. In contrast, mid-rise open neighbourhoods with parks, trees, and green roofs have been shown to support multiple services. These forms can accommodate population growth while preserving ecological functions (Grêt-Regamey et al., 2020).

Further, the link between urban intensification and ecosystem services is not always unidirectional. In China's Su-Xi-Chang region, researchers found a U-shaped pattern: as the city became denser, ecosystem services first declined (Wang et al., 2024). But over time, these services improved when compact buildings were combined with green infrastructure like parks and green roofs. For example,

food and water services became more stable, and climate-related services also started to recover. This case shows that the effects of urbanisation on ecosystem services are shaped not just by how much cities grow, but by *how* they grow. The type of development matters. For example, a high-rise compact neighbourhood will have different ecological impacts than a mid-rise open layout or a low-density suburban sprawl. These types interact with the degree of growth in ways that can either support or harm ecosystem services.

3.2 Understanding urban morphology and Varying Urban Typologies

Urban morphology refers to the study of the physical form, structure, and spatial organisation of urban areas. It provides a foundational framework for understanding how cities grow, evolve, and function. A key part of this is the classification of urban typologies, which are recurring patterns of development and land use. These typologies are shaped by planning history, economic processes, and cultural preferences (Gauthier & Gilliland, 2006). Urban morphology dissects the city into its constituent elements—streets, plots, buildings, and open spaces. It also examines how their arrangement shapes urban experience. Foundational studies by Moudon (1997) and Whitehand (2012) emphasise that urban forms are not fixed. Instead, they evolve through interactions between physical design, functional needs, and cultural values.

In the North American context, this evolution is evident in the historical layering of urban forms. Older city cores often exhibit grid-based street networks, compact block sizes, and mixed-use zoning. Postwar suburbanisation introduced a shift toward low-density, auto-dependent developments with hierarchical road networks, segregated land uses, and cul-de-sacs. Contemporary urban policy increasingly promotes densification through infill development, vertical expansion, and redevelopment of existing areas (Liu et al., 2024). As a result, cities now contain a mix of different urban forms, that defy rigid classification.

Two dominant urban typologies, however, still dominate urban development patterns in North American cities. The first is the compact, mixed-use core. These areas tend to have high-rise buildings, good public transit, and walkable streets. They combine residential, commercial, and institutional uses in proximity. However, in many mid-sized cities, the benefits of this design are limited. Weak public transit and fragmented planning reduce walkability and accessibility. The second typology is suburban sprawl. These areas are usually found at the urban edge. They feature single-family homes, wide roads, and curving street networks with cul-de-sacs. Land uses are

separated, which increases dependence on cars. These neighbourhoods often have green buffers, but they also consume more land and energy per person.

While these typologies provide a useful heuristic, they are not static. Both compact and sprawling areas are undergoing densification, often in ways that reconfigure their environmental and social impacts. Understanding these transitions requires close attention to urban morphology, not just as a physical descriptor, but as a lens through which urban change is interpreted. The following section explores how intensification reshapes these urban forms and alters their capacity to sustain ecosystem services.

3.3 Urban Intensification patterns and influence on Urban Ecosystem Services

Urban intensification implies increasing population and building density in the existing urban fabric. This has evolved as the central strategy to tackle the growing urban population and mitigate the ever-increasing impacts of the declining natural environment and climate change. While intensification is being equally implemented in both compact urban cores and sprawling suburban developments, the intensification strategies, emerging intensification patterns, and their correlation with change in urban ecosystem services largely vary (See **Table 2**).

In compact, mixed-use cores such as urban downtowns and central cities, intensification typically occurs in the form of redevelopment of underutilised plots, infill construction and the vertical expansion of existing buildings. This adds higher densities to already compact regions, featured by mixed land use and pedestrian or transit-oriented designs. Further, to promote the walkable and transit-oriented character of these regions, newer developments are often promoted along transit corridors and stations. Redevelopment of vacant or underused plots is usually defined by mid- to high-rise mixed-use developments. Mixed-use developments are encouraged to blend housing, retail, commercial and institutional uses, making the region more accessible. Municipal policies in Canadian cities explicitly promote compact, high-density developments along transit corridors and downtowns, aiming to optimise land use, where existing infrastructure and amenities can support the growing population.

In contrast, intensification trends in suburban sprawls are gentle, low-density, and often follow the segregated land uses and activities, maintaining the original character of the region. This intensification is often seen in the form of additional townhouses, walk-up apartments and additional residential units on larger plots. However, relatively higher densities and mixed land use have been

introduced in recent times through redevelopment along the automobile-oriented commercial strips. Further high-rise structures with residential and commercial uses have been on the rise along the expanding public transit stations and corridors. These intensification patterns are evident in municipalities such as Barrie and Collingwood in Canada, where intensification in predominantly low-density regions is aimed towards both maintaining the suburban character but also creating holistic neighbourhoods through strategic growth plans.

Mid-rise, mid-density developments have emerged as a balanced intensification approach, bridging the gap between high-rise urban cores and low-density suburban areas. Typically ranging from three to six storeys, these developments offer a human-scaled built form while accommodating a substantial increase in housing units and mixed-use spaces. They are often integrated into existing neighbourhoods along key transit corridors, main streets, or near local centres, providing a blend of residential, retail, and community services within walkable distances. This form of development supports gradual densification without overwhelming local infrastructure or drastically altering neighbourhood character, while still promoting vibrant, transit-supportive communities. In many Canadian municipalities, mid-rise projects are being actively encouraged as part of “gentle density” strategies, offering an adaptable model for both infill sites in mature neighbourhoods and new growth areas.

Table 2: Urban Typologies and Densification Patterns

<i>Typology</i>	<i>Key Features</i>	<i>Densification Patterns</i>	<i>Outcomes of Densification</i>
Compact Mixed-Use Urban Core	High-rise, high-density buildings Mixed-use (residential, commercial, institutional) Grid-like Street networks	Vertical redevelopment and infill Adaptive reuse of older buildings High-intensity transit-oriented development (TOD)	Enhanced transit uses and walkability Loss of open space Higher risk of heat island effects if not mitigated

	Transit-oriented Limited green space		
Mid-Rise, Mid-Density Urban Form	3–6 storey buildings Mixed residential-commercial use Moderate block sizes Some green space and street trees Often in inner suburbs or transitioning zones	Gentle infill Addition of secondary suites Mid-rise redevelopment along arterial corridors Transit-adjacent densification	Balanced density with some green space retention Improved service access Lower ecological stress compared to high-rise cores or low-density sprawls
Suburban Sprawl	Detached single-family homes Low-rise, low-density Curvilinear streets and cul-de-sacs Segregated land uses Car-dependent infrastructure	Lot subdivision Low-rise infill Expansion of suburban fringes and highway corridors	Continued car dependency Increased infrastructure costs Loss of peri-urban ecosystems Fragmented and inaccessible green space

Additional density in each of the above typologies has accelerated their typical impacts on the urban ecosystem services. However, sensitive and context-specific approaches in compact high-rise neighbourhoods can have improved results, while intensification in sprawling regions increases impacts by manifolds.

3.4 Conclusion

A review of current literature and supporting empirical evidence shows that urbanisation, urban form, and intensification are central drivers influencing the production of urban ecosystem services. These factors shape not only the extent of change in ecosystem services but also their spatial patterns and functional performance, both at the city and regional levels. The commonly held view that compact, high-density urban development inherently preserves urban ecosystem services is increasingly being contested. While compact forms may support efficient land use, transport, and infrastructure, they can also lead to concentrated environmental pressures—such as increased urban heat, reduced access to green space per capita, and higher stress on ecological systems—particularly when green infrastructure is not systematically integrated into the urban fabric.

Suburban sprawl is widely criticised for its inefficient use of land, low-density development, and fragmentation of natural landscapes. While it may offer some limited benefits—such as access to larger private green spaces—these are outweighed by its broader negative impacts on ecosystem services. Sprawling development patterns lead to increased land consumption, higher greenhouse gas emissions due to car dependency, and greater infrastructure demands. They also contribute to the loss of agricultural land and natural habitats at the urban fringe, disrupting ecological networks and reducing the overall resilience of urban ecosystems.

Recent research suggests that mid-rise, open urban forms—defined by moderate density, integrated green spaces, and mixed land use—may provide a more balanced approach to managing the trade-offs between urban growth and ecosystem function. This perspective moves beyond the binary debate of compact versus sprawl, and instead supports a context-based, morphology-informed planning approach.

Overall, urban intensification should not be viewed as a uniform or universally positive strategy. Its impacts are shaped by local urban form, ecological conditions, and the degree to which ecosystem

services are considered in planning. To support more resilient and sustainable urban development, intensification strategies must be flexible, evidence-based, and responsive to both spatial patterns and environmental dynamics.

Chapter 4

Case Study - The Regional Municipality of Waterloo

4.1 Introduction

The Regional Municipality of Waterloo (RMOW) offers a unique lens through which to examine the ecological implications of policy-led urban intensification in mid-sized Canadian cities. Located in southwestern Ontario's Greenbelt, RMOW is one of the fastest-growing regions in the province, shaped by deliberate planning interventions such as the Places to Grow Act (2006) and the Regional Official Plan (2009). These legal frameworks have sought to contain urban sprawl, prioritise infill development, and integrate transit infrastructure, particularly through the launch of the ION light rail system in 2019. While much of the scholarly focus on urban ecosystem services (UESs) centres on large metropolitan areas, RMOW represents a distinct and *underexplored context* where suburban foundations, rural-urban gradients, and mixed development patterns converge. This chapter introduces the spatial, demographic, and policy landscape of RMOW, outlining how diverse urban typologies have evolved across its municipalities. In doing so, it sets the empirical foundation for analysing how different trajectories of intensification shape UES outcomes across the region.

4.2 Regional Overview

RMOW comprises three urban municipalities—Kitchener, Waterloo, and Cambridge—and four rural townships: North Dumfries, Wellesley, Wilmot, and Woolwich (Figure 1). Together, the region spans approximately 1,390 square kilometres and had a population of 673,910 at the end of 2023. Since 2016, it has experienced steady annual population growth (~1.6%), spiking to 4.4% in 2023, largely driven by international migration, the student population, and a local economy anchored in education, technology, and advanced manufacturing.

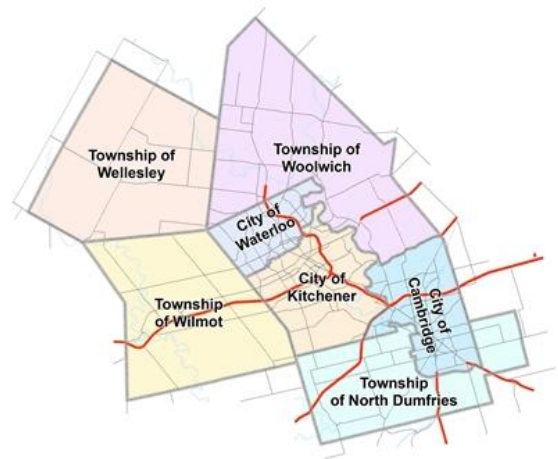


Figure 1: Region Municipality of Waterloo - Cities and Townships

The region's urban municipalities have seen distinct growth and development trajectories. Kitchener, the largest city and the administrative centre of the region, has transitioned from a manufacturing base to a growing centre for tech-based companies and entrepreneurship (Figure 2). Waterloo, which hosts two of the leading universities - the University of Waterloo and Wilfred Laurier University, is dominated by students, skilled professionals and academicians. Cambridge, historically known for its mills, is characterised by a historic downtown and riverfront setting, balancing heritage conservation with new suburban and infill developments (Figure 3). The surrounding townships maintain a rural agricultural identity, contributing to the region's mixed rural-urban character.

The region has experienced a substantial change in its population and urban density since the early 2010s. This period marks the implementation of intensification-focused planning policies such as the Places to Grow Act (2006) and the Regional Official Plan (2009), shaped by provincial and regional growth frameworks. These policies emphasised reducing urban sprawl, protecting farmland and natural features, while promoting intensification within existing urban boundaries. Further, the launch of the ION light rail transit system in 2019 played a pivotal role in determining the newer developments and urban typologies in the region. This development was marked by high-density mixed-use structures around the major stations, particularly in downtown Kitchener, Uptown Waterloo and some nodes in Cambridge. As shown in Figure 4, the agricultural systems legend in the Region of Waterloo's 2012 Official Plan distinguishes between



Figure 2: An Aerial Photo of downtown Kitchener, looking northeast

Source: Wikipedia, 2021
https://en.wikipedia.org/wiki/Kitchener,_Ontario



Figure 3: An aerial view of the City of Cambridge (Historic Galt Region)

Source: Sean Pavone, 2024 <https://www.shutterstock.com/image-photo/cambridge-ontario-canada-overlooking-grand-river-2539295387>

Protected Countryside, defined as rural or natural land preserved mainly for its ecological, landscape, or recreational value where development is highly restricted, and Prime Agricultural Area, defined as land with high-quality soils and climate conditions, protected primarily to sustain agricultural production..

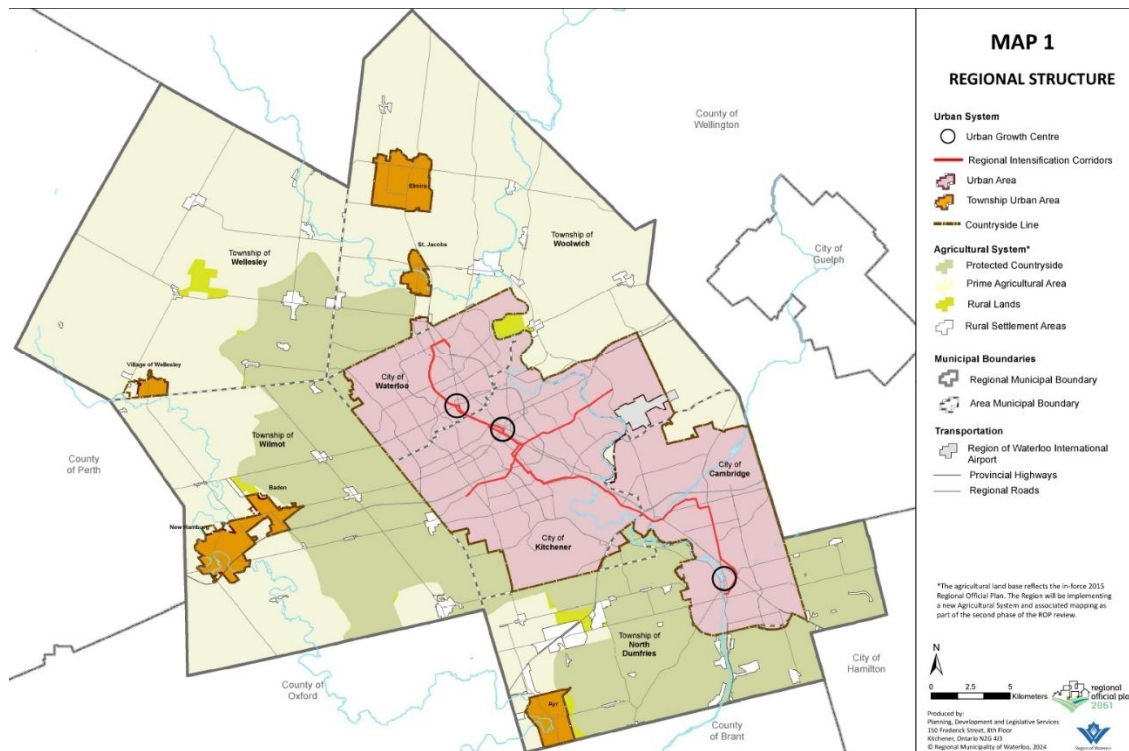


Figure 2: Region of Waterloo Urban and Growth Centres

Source: Region of Waterloo Official Plan, 2012

Increased development constraints within existing built-up areas have driven vertical growth, transforming mid-sized cities like the RMO from predominantly low-rise family homes to taller multi-unit apartments. This shift has been guided by transit-oriented planning, with over 70% of new residential units since 2010 built within existing urban boundaries—well above the 45% target in the Regional Official Plan. The transformation has been accompanied by investments in public transit, urban parks, and the protection of ecological features such as the Waterloo Moraine, the region's primary aquifer. While these measures aim to mitigate the environmental impacts of a growing

population, their effectiveness in sustaining urban ecosystem services remains to be evaluated in this thesis.

4.3 Rationale for Selection

While mid-sized cities often receive less attention than major metropolitan regions, they offer a valuable lens through which to re-examine established urban typologies. Rather than rejecting existing classifications such as core-periphery gradients or distinctions based on density and land-use intensity, this study uses the case of the Regional Municipality of Waterloo (RMOW) to explore how these typologies operate under different conditions. RMOW presents a particularly useful setting: it is a rapidly intensifying region where planning frameworks, such as the Places to Grow Act and the Regional Official Plan, are actively reshaping a predominantly suburban landscape. By applying standard urban classifications (e.g., levels of impervious surface or built form) to this mid-sized context, the study evaluates how typologies developed for large cities translate into settings where development is more dispersed, policy-driven, and intermixed with rural and agricultural land uses.

Studying mid-sized cities like RMOW allows us not only to observe different urban trajectories but also to test the limits and adaptability of conventional analytical tools. It highlights where existing typologies remain useful, where they require adjustment, and where new hybrid forms are emerging. In doing so, the study contributes to a more flexible and context-sensitive understanding of urbanisation and its ecological implications—one that better reflects the diversity of urban growth across scales and geographies.

This morphological diversity is evident in RMOW, with urban cores getting compact and higher (downtown Kitchener, uptown Waterloo and Hespeler/ Preston core regions in Cambridge), emerging nodes of high-density and high-rise developments in suburban sprawls around the major transit stations (around major transit stations in suburban Kitchener), and mid-rise developments along the transit corridor and major highways. Further, a rise in low-rise low-density developments along the existing urban boundaries also continues, especially in Cambridge and the townships.

In summary, the period from 2005 to the present has been pivotal in shaping the RMOW, marked by rapid urbanisation, increasing building density, and targeted planning interventions to promote intensification. However, due to data availability, this study focuses on the decade between 2013 and 2023—a critical phase for examining how urban intensification has influenced the production of urban ecosystem services across a diverse urban landscape

4.4 Evolving Urban Typologies and Intensification Trends (2013–2023)

Over the past decade, RMOW's urban fabric has transformed significantly under the combined influence of demographic growth, regional planning mandates, and strategic infrastructure investments. Each of the region's constituent municipalities has responded to these pressures with distinct urban development trajectories, producing varied typologies of intensification.

4.4.1 City of Kitchener: From Industrial Core to High-Density Centre

Kitchener, the largest city in the region and its administrative hub, has transitioned from a manufacturing-based economy to a vibrant tech and startup ecosystem. Between 2013 and 2023, its population grew by approximately 17.2%. The ION light rail system catalysed redevelopment along the Central Transit Corridor (CTC), particularly downtown and along King Street, with high-rise, mixed-use projects replacing older industrial and commercial properties. Developments such as a 45-storey residential tower with over 500 units exemplify the city's vertical intensification strategy.

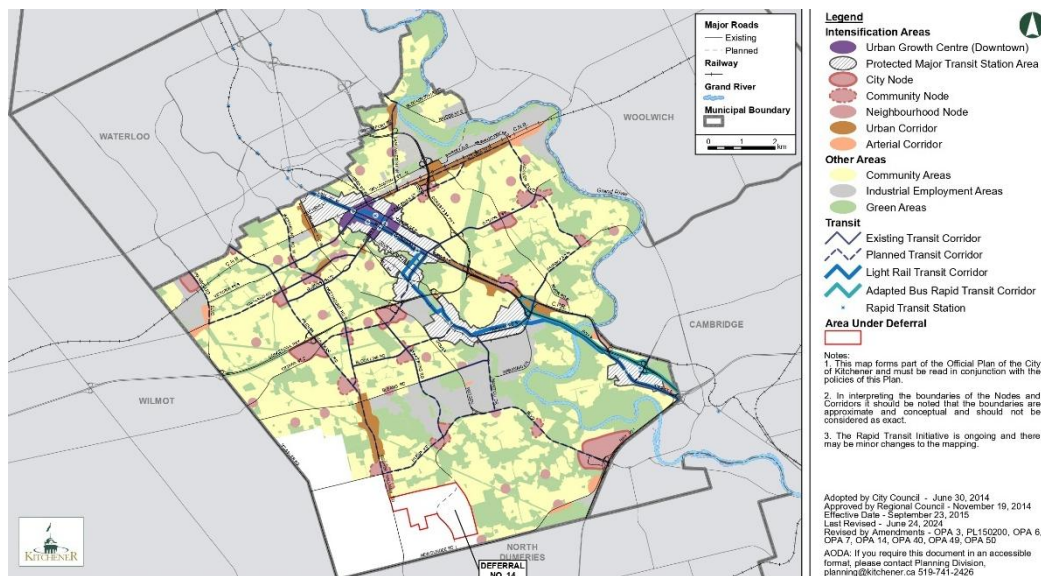


Figure 3: City of Kitchener Urban Structure

Source: City of Kitchener Official Plan, 2014

Kitchener now reflects a hybrid typology where historic low-rise blocks coexist with new high-density nodes, reinforcing the region's push for walkable, transit-connected communities.

4.4.2 City of Waterloo: Institutional District and Mid-Rise Intensification:

Waterloo's urban identity is shaped by its role as a university town. Hosting the University of Waterloo and Wilfrid Laurier University, the city has seen a 22.2% population increase since 2015, accompanied by substantial mid- and high-rise residential growth. Much of this development is concentrated in Uptown Waterloo and along the ION corridor, primarily serving students, researchers, and young professionals. The city's Urban Design Manual supports sustainable growth through green infrastructure and integrated public space planning. While intensification is most visible near the core, low-rise residential zones at the periphery remain largely intact.

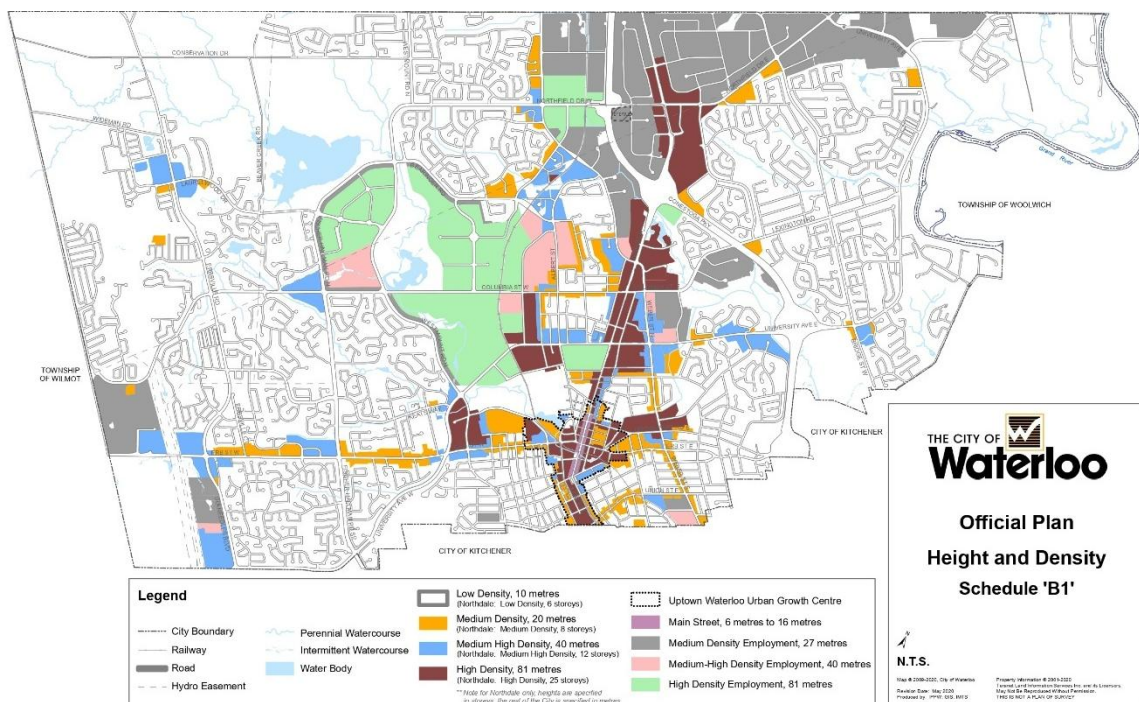
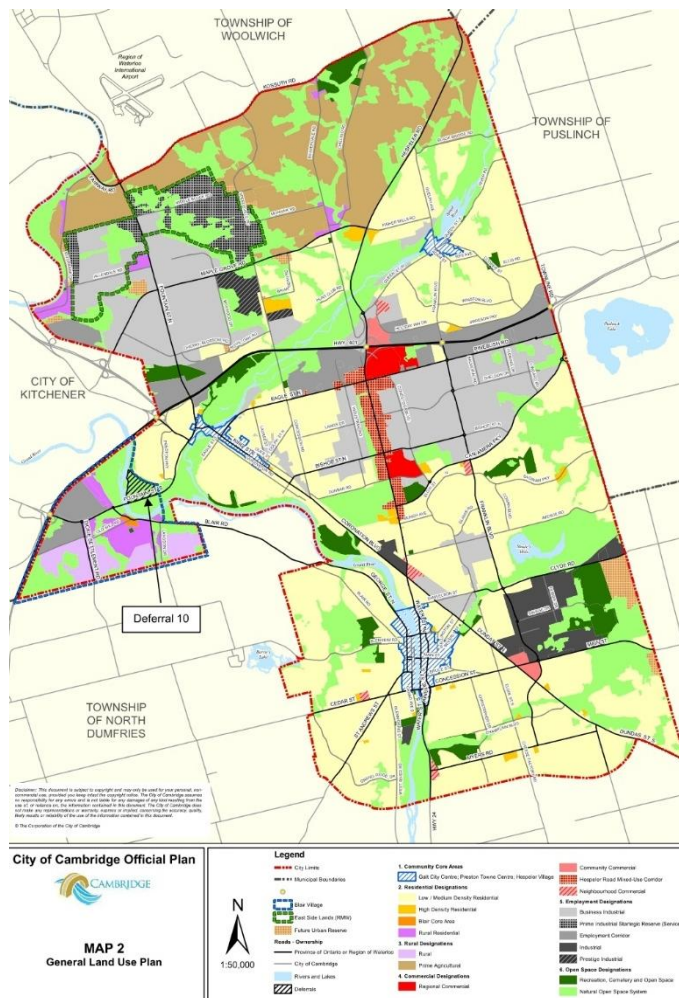


Figure 4: City of Waterloo Height and Density Policies

Source: City of Waterloo Official Plan, 2012

4.4.3 City of Cambridge: Node-Based Growth and Heritage Preservation:



Cambridge has experienced a more modest growth in its population (~9.3%) but presents a distinctive urban form centred around three historic cores: Galt, Preston, and Hespeler. Unlike the continuous intensification along transit corridors in Kitchener and Waterloo, Cambridge's growth is more node-based, characterised by mid-rise and townhouse developments in select areas near future transit nodes. The city balances infill development with strong heritage preservation goals, resulting in a predominantly low-rise suburban fabric with pockets of moderate density.

Figure 5: City of Cambridge Urban Land Use

Source: City of Cambridge Official Plan, 2018

4.4.4 Rural Townships: Managed Growth and Agricultural Protection:

The townships of North Dumfries, Wellesley, Wilmot, and Woolwich have grown collectively by ~9% over the last decade. These areas are defined by low-density agricultural landscapes and small settlement clusters. Growth is tightly managed through urban boundary restrictions and infill policies, aimed at preserving farmland and ecological features. While recent years have seen some increase in townhouse developments, the townships largely retain their rural character and low-density typologies.

4.5 Summary and Analytical Significance

Between 2013 and 2023, RMOW experienced a profound transformation in urban form, driven by intensification policies, transit investments, and demographic pressures. The shift from single-detached homes to townhouses and mid-to-high-rise developments, especially around transit nodes and urban cores, reflects a broader regional effort to promote compact, sustainable urbanism. Yet the variation across municipalities—with Kitchener and Waterloo embracing vertical growth, Cambridge opting for a node-based intensification with a suburban style of development, and the townships maintaining rural patterns—offers a unique opportunity to study how diverse urban typologies mediate the production and distribution of urban ecosystem services. This case study provides spatial and institutional grounding for the chapters that follow, where these typologies will be analysed in relation to observed patterns of ecological change.

Chapter 5

Methods

5.1 Research Framework

This thesis examines the correlation between varying urban patterns, increasing urban intensification and its impact on local urban ecosystem services (hereafter referred to as UESs) in the study area. More specifically, the research delves into the impact of urban intensification on the natural environment's capacity to produce UESs. This study adopts a temporal approach to analyse the effects over 10 years. As outlined in Chapter 4, population growth between 2013 and 2023 varied across the three cities, with Waterloo (~22%) and Kitchener (~17%) experiencing substantially higher growth than Cambridge (~9.3%). While population change is only one dimension of urban intensification, it provides important context for understanding the differing pressures on land use, built density, and urban ecosystem services (UESs) assessed in this chapter. The growth in urban intensification and the corresponding change in selected UESs were assessed for 2013 and 2023 to identify common trends in how UESs respond to varying urban typologies and degrees of urban intensification over time.

The research framework of this study comprises six components (see Table 5.1). **First**, variables are identified for both UESs and the urban landscape based on (i) relevance to the study area and (ii) availability of data. 7 UESs were identified—food production, freshwater quality and quantity, air temperature, air quality, stormwater runoff, and availability of green spaces—along with 3 urban parameters that quantify urban patterns and degree of intensification: built floor area, built volume, and impervious surfaces. The availability of data for the study region highly defines the selection of these variables.

The second and third components describe the process of data collection and operationalisation. This study was conducted with limited and coarse data available from government open data portals. For each of the seven UESs, measurable proxies were selected to quantify key aspects of service provisioning and to capture spatial patterns of production. Primary data sources and their operationalisation in measurable units are detailed in Section 5.2.

Data were obtained in two main formats: (1) point data, predominantly environmental measurements from regional and provincial weather stations or wells, and (2) polygon-based vector data, such as land cover, cropland, and urban parameters. Point data for continuous variables (e.g., temperature, rainfall) often require interpolation to estimate values for unsampled locations. In this study, the IDW (Inverse Distance Weighting) method in ArcGIS Pro was applied to all environmental point datasets, as it assumes nearby values have greater influence than distant ones—a reasonable fit for spatially correlated environmental data.

Kriging, while capable of producing accurate and visually smooth maps when the variogram is well-defined (Ding et al., 2018), is less suitable when observations are sparse or unevenly distributed. This is because its variogram, which describes how similarity between measurements changes with distance, can overfit random noise instead of capturing true spatial patterns. In the case of this study, weather station data for air temperature and precipitation, within the study area, were limited to three locations in close proximity to each other, making IDW a more appropriate choice. Nevertheless, the clustered distribution of stations still limits the accuracy of the interpolated surfaces. The resulting maps for each variable were generated as vector or raster layers for further analysis.

In the **fourth** component, the quantified data for each variable is visualised on a 1 km x 1 km grid. The 1 km² grid resolution was selected to match the coarsest input dataset in the study and to maintain consistency across all variables without resampling beyond the original data resolution. This choice ensured that each grid cell contained sufficient underlying data to produce reliable estimates and avoided creating empty or data-sparse cells in less monitored parts of the region. Other resolutions were not tested, as reducing the grid size would have required upscaling several datasets beyond their original precision, potentially introducing artificial accuracy. While finer resolution might reveal more local variation in heterogeneous areas such as urban–rural transition zones, the positional accuracy of key datasets (e.g., well locations, weather stations) would limit the reliability of estimates at smaller cell sizes. The absolute data of the individual variable for each grid cell is first standardised for more comparative accuracy and then normalised on a scale of 0-1 for visualisation purposes.

Grid analysis is suitable for this study for three key reasons: one, it offers spatially-explicit outcomes, second, it enables finer distinctions in urban form and UESs provisioning and lastly helps integrate diverse data streams into a consistent spatial framework. The impact on urban ecosystem services and their production both are rarely confined by administrative boundaries such as blocks or

neighbourhoods. Using a structured grid allows for a nuanced analysis of UESs patterns and influence across the boundaries (Shao et al., 2021). Further, this study aims to analyse the varying impacts of urban forms and intensification on the production of UESs. The grid framework enables the distinction of urban spatial variance within the smallest administrative boundaries (Guo et al., 2025). Lastly, a grid unifies disparate datasets (land cover, soil hydrology or air temperature) into a comparable framework (Hu et al., 2025). A 1km x 1km grid is selected based on the resolution of the data and the purpose of the study. A 1km² grid effectively captures the spatial heterogeneity of the urban environment, as well as being suitable for the coarse data available for this study (Assennato et al., 2022; Grêt-Regamey et al., 2020). In the upcoming sections, a 1km x 1km grid cell is also referred to as a grid cell.

The **fifth and sixth** components analyse the data in each grid cell. First, cluster analysis is performed to form UES bundles using the Self-Organising Map (SOM) tool in R (Kohonen package). In this study, SOMs were used as an unsupervised machine learning tool to identify patterns and group 1km x 1km grid cells based on changes observed in UESs indicators. SOM is a type of artificial neural network that projects high-dimensional data into a low-dimensional (typically two-dimensional) space while preserving the topological relationships in the data (Kohonen, 2001). This approach has been widely used in environmental and urban studies for its ability to handle complex, non-linear, and multidimensional datasets without requiring prior assumptions about data distribution (Vesanto & Alhoniemi, 2000; Liu et al., 2016). This method is particularly suited for this study because it allows simultaneous analysis of multiple UES variables across space, facilitates visual interpretation of UES bundles, and supports the detection of nuanced trade-offs and synergies among UESs.

The SOM was trained using 100 learning iterations ($r_{len} = 100$), with the resulting output grouped into four clusters for further analysis. Further, Pearson correlation analysis was conducted using the *base stats* package in R to assess the strength and direction of relationships between UESs in each formed bundle. Subsequently, using urban data for impervious surfaces, the grid cells were grouped into 4 urban clusters using thresholds defined by the percentage of impervious surfaces in each grid cell.

The final step of analysis involved performing regression analysis in R using the `lm` (linear regression model) function from the base stats package to quantify the impacts of urban cluster typologies and intensification change on UESs.

The next chapter discusses the data sources and operationalisation processes for each UES and urban intensification, respectively. They also describe the statistical methods and logic used to develop UES bundles and urban intensification clusters for final analysis.

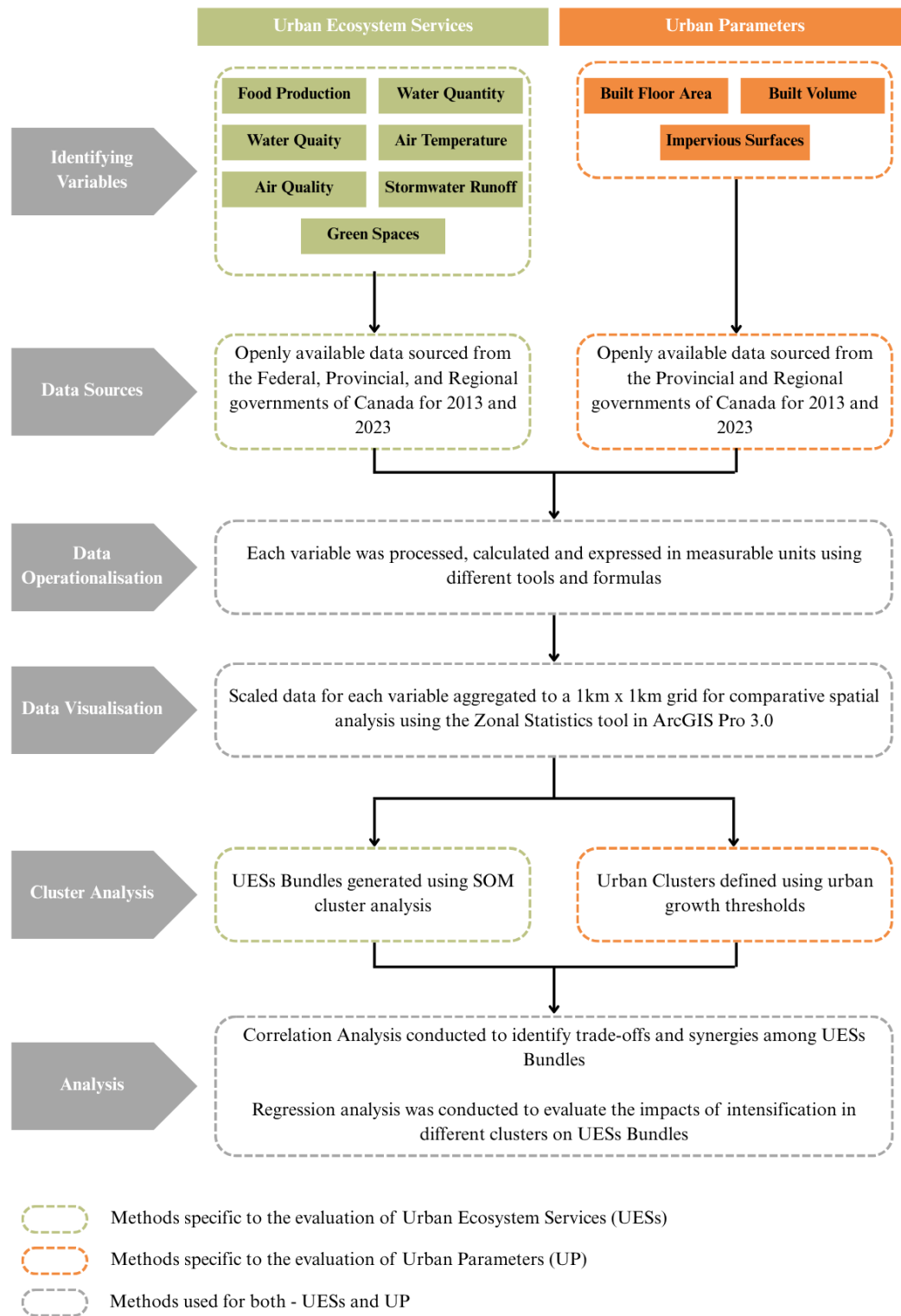


Figure 6: Research Framework

5.2 Mapping Urban Ecosystem Services in the Regional Municipality of Waterloo

This section describes the data sources for UESs, how they were cleaned, processed and finally operationalised for individual UES. It also defines how values for individual UESs are calculated for each grid cell, which are then used to formulate UES bundles used to evaluate the trade-offs and synergies among the services.

Table 3: Primary Data List

Data	Relevant UES	Data Source	Resolution	Reference
Region of Waterloo Administrative Boundary	All variables	Region of Waterloo Open Data Portal	-	Region of Waterloo, 2023.
Land Use Data	Air Temperature Regulation (ATR), Stormwater Runoff Management (SWR), Green Spaces (GS)	SOLRIS	15-20m	Ontario Ministry of Natural Resources and Forestry, 2008 Ontario Ministry of Natural Resources and Forestry, 2017
Crop Types and Agricultural Land Cover	Crop Yield (FP)	Ontario Annual Crop Inventory	30m	Agriculture and Agri-Food Canada. (2023).

Groundwater levels	Groundwater levels (FWQN)	Water Well Information System (WWIS)	-	Ontario Ministry of the Environment, Conservation and Parks, 2024
Contaminant concentration in freshwater	Chemical concentrations of groundwater (FWQL)	Provincial Groundwater Monitoring Network (PGMN)	-	Ontario Ministry of the Environment, Conservation and Parks, 2023
Air Temperature	ATR	Provincial Weather Stations	-	Environment and Climate Change Canada, 2025; University of Waterloo, 2025
Pollutant concentration in the atmosphere	Air Quality	Provincial Weather Station	-	Ontario Ministry of the Environment, Conservation and Parks, 2024
Soil Hydrology	SWR	Soil Survey Complex		Ontario Ministry of Agriculture, Food and Rural Affairs, 2019
Annual Rainfall	SWR	Provincial Weather Stations	-	Environment and Climate Change Canada, 2025

5.2.1 Food production (FP)

One method used to assess the urban ecosystem service of food production is by measuring changes in crop yield over time. This approach reflects both the availability of farmland and the quantity of agricultural output, providing a relevant indicator to measure the quantity of food produced in the study area.

5.2.1.1 Data Collection

To determine the spatial extent of agricultural land cover and identify crop types across the study area, the Ontario Annual Crop Inventory produced by Agriculture and Agri-Food Canada (AAFC) was used for the years 2013 and 2023. This step quantifies the availability of farmland and detects changes in land cover relevant to food production. The dataset was converted from raster to vector format, producing polygons representing homogeneous crop or land cover classes.

To evaluate the quantity of food produced, average crop yield values were required for each identified crop type. The Ontario Ministry of Agriculture, Food, and Agribusiness provides historic average yield (Bushels) per acre for major crops in Ontario. Crop yields for both years, 2013 and 2023, were available. This step ensured that total crop production could be estimated for the full set of land cover classes identified in the crop inventory.

5.2.1.2 Data Operationalisation

The first step involved clipping the provincial farmland shapefile to the administrative boundary of the Region of Waterloo using the clip tool in ArcGIS Pro. This was done to isolate and analyse only those agricultural polygons (referred to as farmland parcels henceforth) located within the study area. Each farmland parcel within the clipped shapefile was already coded in the original dataset to indicate the major crop type grown. Two layers corresponding to the years 2013 and 2023 were created.

In the second step, the estimated average crop yield (provided by OMAFRA) for each crop type was joined to the shapefile using the crop codes (Join and Relates tool in ArcGIS Pro). The codes for each crop yield were modified based on the crop names to match those in the shapefile. This quantified the crop produced by each farmland parcel. The total crop yield (expressed in bushels per acre) was calculated by multiplying the average yield value by the area of each farmland parcel.

The third step involved aggregating the total crop yield and total farmland area to the 1 km × 1 km grid cells. This summarises food produced at the grid cell level for comparison over time and with other variables. For each grid cell, the sum of all crop yields and the total farmland area were calculated using the Summarise Within tool in ArcGIS Pro.

While total farmland area indicates the extent of land available for cultivation, aggregated crop yields provide an estimate of the current state of agricultural production in the study area. These yields are not the same as maximum potential production—often referred to as food production capacity—because

they are influenced by yield gaps. A yield gap is the difference between the potential output achievable under ideal climatic, agronomic, and management conditions and the actual output observed in practice. Such gaps in the study area may arise from climatic variability, suboptimal farming practices, and land-use constraints (See Figure 9). For this reason, aggregated crop yield was used for trend analysis in this study, as it provides a more direct indicator of how each grid cell's observed output has changed over time.

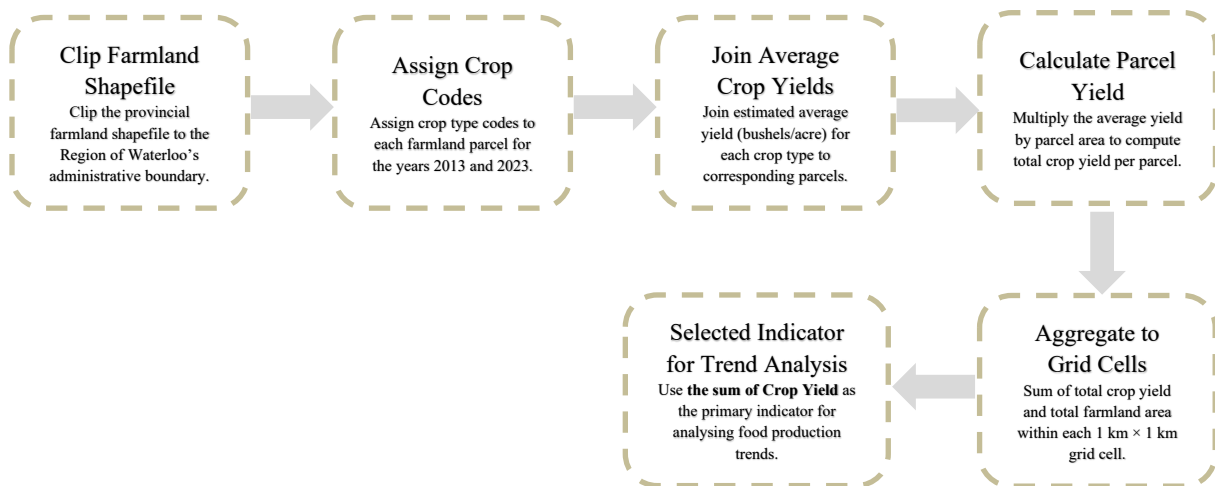


Figure 7: Steps to Quantify Food Production

5.2.2 Freshwater Quantity

The Regional Municipality of Waterloo sources its daily freshwater supply from two main sources: approximately 75–80% from underground aquifers and 20–25% from the Grand River (Region of Waterloo, 2024). In this research, the focus is on the primary source—underground water resources—to assess freshwater quantity and quality (see next subheading). The Region does not provide spatially disaggregated data on how much water is supplied by each source in different areas. The proportion of groundwater versus surface water use varies locally depending on demand and availability. As a result, this study does not attempt to estimate the individual contribution or spatial influence of each water source. However, given the Region's heavy reliance on groundwater, any fluctuation in its levels is likely to have a proportionately larger impact on the region's overall freshwater availability.

5.2.2.1 Data Collection

Groundwater level data were obtained from the Provincial Water Well Information System (WWIS), managed by the Ontario Ministry of the Environment, Conservation and Parks (2024). This step was necessary to assess the status of underground water resources in the Region of Waterloo for the years 2013 and 2023. The WWIS database compiles well records submitted by licensed contractors, as required under Ontario Regulation 903, and includes attributes such as well location, total depth, bedrock depth, and static water level. Groundwater levels were estimated using well record data extracted from the WWIS for each target year to quantify changes in underground water availability over time. For the year 2013, over 15,500 well records were analysed, while more than 26,000 well entries were used for 2023. These datasets provided sufficient spatial coverage to generate representative groundwater level estimates for the Region.

While Ontario's Permit to Take Water dataset records large-volume withdrawals (>50,000 L/day), it was not used in this study because it captures authorized capacity rather than actual withdrawals, does not account for smaller but potentially significant extractions, and lacks consistent temporal coverage for the full 2013–2023 period. However, such permit records represent a valuable complementary measure of freshwater extraction pressure and could be incorporated in future work.

5.2.2.2 Data Operationalisation

The first step involved extracting regional groundwater well data from the provincial Water Well Information System (WWIS). Only wells registered on or before the respective target years were included to ensure temporal consistency.

Groundwater levels (GWL) were calculated for each well using the following formula:

$$\text{GWL} = \text{Well depth}^* - \text{Static Water Level}$$

Where **GWL** stands for groundwater level, **well depth** is the total depth of the well from the ground surface, and **static water level** is the distance from the ground surface to the top of the water, when the water is not being pumped and the aquifer is at its normal level. *In exceptional cases, where the bedrock depth was lower than the well depth, bedrock depth was used to calculate the groundwater level.

To estimate continuous groundwater levels across the region, the IDW tool for interpolation was applied using ArcGIS Pro. This produced two raster surface layers representing spatially distributed groundwater levels for 2013 and 2023.

Finally, the interpolated groundwater surface was summarised to a $1 \text{ km} \times 1 \text{ km}$ grid using the Zonal Statistics tool in ArcGIS Pro. The mean groundwater level was calculated for each grid cell to enable comparison over time and spatial pattern analysis across the study area. (See Figure 10)

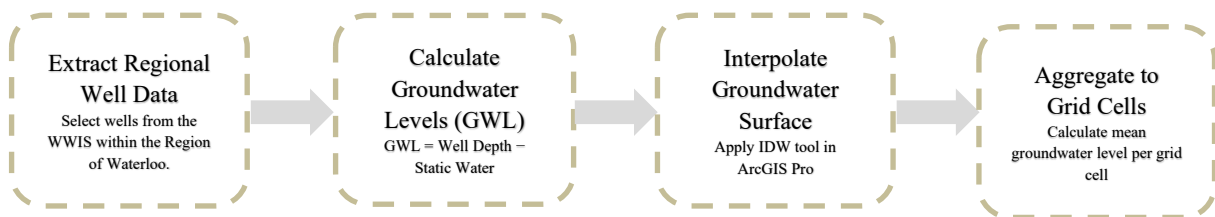


Figure 8: Steps to quantify Fresh Water Quantity

5.2.3 Freshwater Quality

As outlined in the previous section, this study focuses exclusively on underground water reservoirs to assess freshwater quality in the Regional Municipality of Waterloo. According to provincial and regional guidelines, both private and public well owners are required to monitor groundwater quality for key parameters, including salinity, acidity (pH), and the presence of specific minerals such as nitrates, fluoride, arsenic, and manganese (Region of Waterloo, 2024). While laboratory testing includes additional minerals, many of these do not pose direct risks to human health and primarily influence the water's colour or odour. In contrast, the selected minerals in this study are those with recognised health risks when concentrations exceed acceptable thresholds, as defined by both the Province of Ontario and the Region of Waterloo (Province of Ontario, 2024; Region of Waterloo, 2024).

Freshwater quality was quantified using a composite Freshwater Quality Index to provide a single measure that reflects multiple freshwater quality parameters. Each parameter—salinity, pH, nitrates, fluoride, arsenic, and manganese—is measured in different units and on different scales. These values are then standardised and normalised before they can be combined into a single index.

5.2.3.1 Data Collection

The first step involved obtaining groundwater quality data from the Provincial Groundwater Monitoring Network (PGMN), an initiative of the Government of Ontario established in 2000. This step was necessary to evaluate long-term, standardised measurements of chemical concentrations in groundwater.

The relevant monitoring wells were selected. To ensure sufficient spatial representation within the study area, 15 monitoring wells were selected—9 located within the administrative boundary of the study area and 6 situated within 50 km of the regional boundary. This step was undertaken to supplement the limited number of wells directly within the region for improved accuracy.

For each of the 15 wells, annual concentration data were extracted for the years 2013 and 2023. The parameters included were nitrates, fluoride, arsenic, manganese, salinity, and pH. This step ensured consistency in the analysis of water quality over time and alignment with the health-relevant indicators identified in provincial and regional guidelines.

5.2.3.2 Data Operationalisation

Each freshwater quality parameter was first interpolated across the study area to estimate spatial values beyond the sampled wells. This was done using the IDW interpolation tool in ArcGIS Pro, generating a continuous raster surface for each parameter.

Second, the mean interpolated value of each water quality parameter was extracted for approximately 1,500 grid cells using spatial analysis tools in ArcGIS Pro. This step ensured that all parameter values were spatially aligned at the 1 km × 1 km resolution used for this study.

To allow comparison across parameters with different units, the extracted values were standardised using the z-score formula:

$$z = (x - \mu) / \sigma$$
$$\sigma = \sqrt{ \Sigma(x_i - \bar{x})^2 / (n - 1) }$$

Where z is the standardised value, x is the original value, μ is the mean of the dataset for each quality parameter, and σ is the sample standard deviation of the dataset for each quality parameter. A sample standard deviation was calculated using the formula above, where σ is the sample standard deviation for the dataset, x_i is the original value, $\bar{x}$ is the mean of the dataset, and n is the total number of observations (grid cells) in the dataset.

A directional correction was applied to the standardised values of salinity, nitrates, fluorides, arsenic and manganese to reflect that, unlike pH, higher concentrations of these values generally indicate lower freshwater quality. Therefore, the standardised scores for all the values except pH were negatively corrected before inclusion in the index.

Next, to form the final freshwater quality index, a negative correction was applied to the standardised values of pH. While a lower presence of minerals and salinity (salts) indicates higher freshwater quality, a higher pH value (alkaline water) indicates higher freshwater quality. The freshwater quality index for each grid cell was obtained by adding all the standardised values.

Finally, before calculating the final index, all standardised values were normalised to a 0–1 scale using the min-max normalisation formula:

$$N = (z - z_{\min}) / (z_{\max} - z_{\min})$$

Where N is the normalised value, z is the standardised score, and $z_{\min}$ and $z_{\max}$ are the minimum and maximum z -scores for that parameter across all grid cells.

The Freshwater Quality Index for each grid cell was calculated by adding the normalised values of all six parameters. Lower index values represent poorer freshwater quality, whereas Higher values indicate better groundwater quality. (See Figure 11)

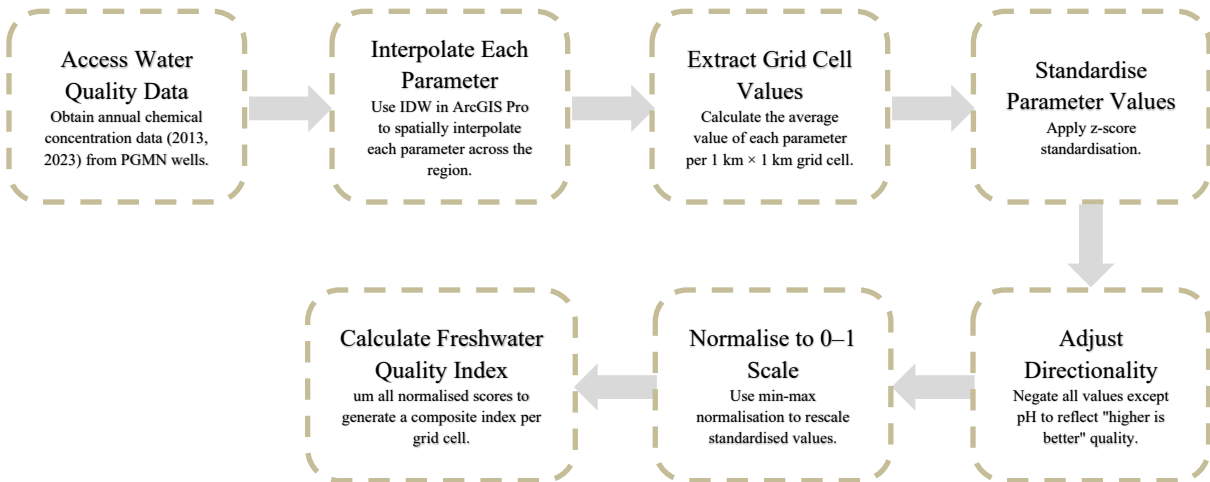


Figure 9: Steps to quantify Freshwater Quality

5.2.4 Air Temperature Regulation

Air temperature regulation is used to quantify changes in ambient air temperature across the Region. In this study, the service is defined such that higher regulation indicates cooler temperatures, reflecting improved temperature moderation by the urban landscape and green infrastructure. Conversely, reduced regulation reflects rising temperatures and thermal discomfort. The final normalised values obtained for air temperature are negatively corrected to directionally match other services, where higher values mean improving services and lower values mean degrading services. Hence, for clarity and simplicity in communication, the term air temperature regulation is used.

This service was assessed using average daily temperatures for the summer months—June, July, August, and September—which are most representative of urban heat impacts. The air temperature data were collected from three weather stations within the study area: two operated by the Province of Ontario and one by the University of Waterloo. To strengthen spatial interpolation, additional data were used from seven provincial weather stations in 2013 and six in 2023, located within a 50 km radius outside the regional boundary.

The weather stations within the Region remained constant across both years. However, the stations outside the boundary differed between 2013 and 2023, based on data availability for each year of study.

5.2.4.1 Data Collection

The daily mean temperatures were used from the weather station data for the summer months. Finally, an average value of daily mean temperatures was computed across the four months for each weather station in each study period. An average value was used to capture the long-term trends of air temperature changes, against maximum temperatures or the 95th percentile, which are more suitable to evaluate heat waves or the impacts of climate change.

The required land use and land cover data are obtained from SOLRIS (Southern Ontario Land Resource Information System). SOLRIS is a comprehensive landscape-level inventory developed by the Ontario Ministry of Natural Resources and Forestry to map natural, rural and urban land cover across southern Ontario. This land use is based on the Ecological Land Classification (ELC) framework, which integrates data from satellite imagery, aerial photography and field validation. The

ELC framework supports this study as it compiles data such as landforms, soil, vegetation, and urban forms, which provides rich ecological land cover attributes necessary for this study.

5.2.4.2 Data Operationalisation

To estimate air temperature values across unsampled locations, the average of daily mean temperatures was spatially interpolated using the Inverse Distance Weighting (IDW) tool in ArcGIS Pro. This step was necessary to generate a continuous temperature surface for the entire Region, enabling analysis at the 1km² grid scale.

Following interpolation, the mean air temperature for each 1 km × 1 km grid cell was extracted using the Zonal Statistics tool in ArcGIS Pro. This was done to standardise air temperature values across spatial units for comparable analysis. Other resolutions were not tested, as reducing the grid size would have required resampling several datasets beyond their original spatial precision, introducing false precision and potentially misleading spatial patterns.

(See Figure 12)

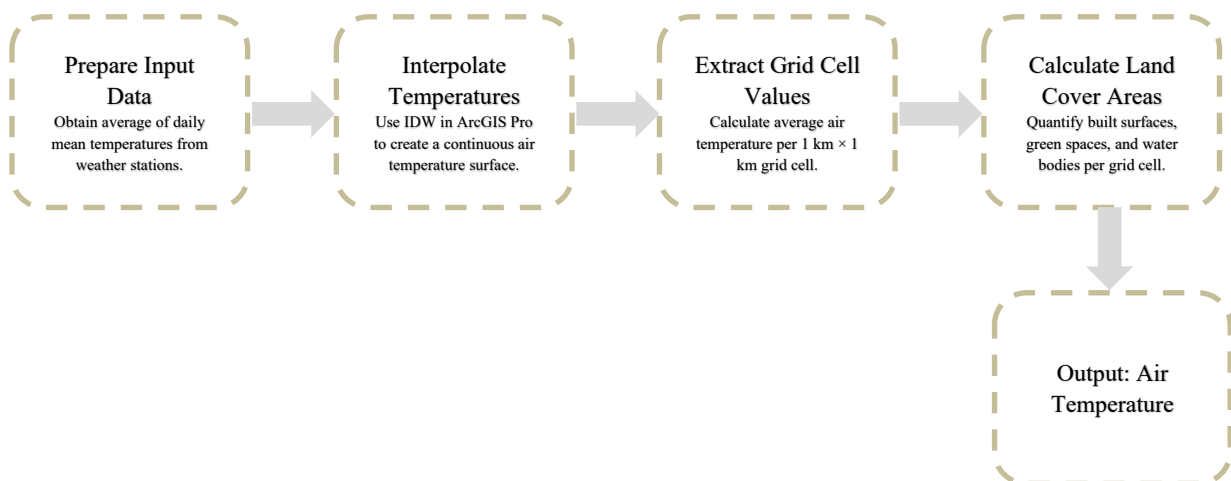


Figure 10: Steps to quantify Air Temperature

5.2.5 Air Quality

To quantify air quality as an ecosystem service, the Air Quality Health Index (AQHI) was used. This method was selected based on its scientific validity and adoption by Public Health Ontario (PHO), the provincial agency responsible for providing evidence-based health protection guidance (Public Health Ontario, 2013). Ontario and the Region of Waterloo have prescribed a list of harmful pollutants, their

threshold levels and the formulation of the AQHI. This study uses hourly data of these pollutants measured either as micrograms/ cubic meter or ppm (particles per million) from provincial weather stations. However, only 1 weather station is available in the entire region; hence, this study uses the highest recorded daily AQHI value for each year, 2013 and 2023 for this study. While this study does not map and analyse the fluctuations in air quality across the region, it is an essential regulatory service when analysing the impacts of urban intensification on local UESs.

5.2.5.1 Key Pollutants Observed

The study focuses on key atmospheric pollutants commonly monitored in urban environments due to their environmental and health impacts. The selected pollutants include:

- **Nitrogen Dioxide (NO₂):** Formed by the oxidation of NO, NO₂ is a major component of traffic-related air pollution and contributes to respiratory illnesses and smog formation.
- **Ozone (O₃):** Ground-level ozone is a secondary pollutant generated through photochemical reactions between nitrogen oxides (NO_x) and volatile organic compounds (VOCs) in the presence of sunlight. Unlike stratospheric ozone, ground-level O₃ is harmful to both human health and vegetation.
- **Particulate Matter (PM_{2.5}):** Refers to airborne particles with diameters of 2.5 micrometres or smaller. These fine particles are emitted from combustion sources and atmospheric chemical reactions and are associated with adverse cardiovascular and respiratory outcomes.

While this study only considers the parameters listed above, pollutants such as sulphur dioxide and carbon dioxide are crucial in evaluating air quality. However, due to a lack of data for the study years (2013 and 2023), these parameters are not considered.

5.2.5.2 Data Collection

A region-wide Air Quality Health Index (AQHI) was developed for the years 2013 and 2023. Hourly measurements of key air pollutants—NO₂, O₃, and PM_{2.5}—were obtained from Ontario’s open data portal. For each pollutant, the daily maximum concentration was calculated from the hourly data to represent the highest exposure level per day throughout the dataset.

5.2.5.3 Data Operationalisation

The AQHI was calculated based on the formula prescribed by PHO:

AQHI Formula:

$$\text{AQHI} = 0.084 * \text{NO}_2 + 0.052 * \text{O}_3 + 0.047 * \text{PM}_{2.5}$$

Where 0.084, 0.052 and 0.047 are constant coefficients calculated and provided by the PHO (simplifying the calculations of AQHI). These coefficients have been calculated using a time-series study over 12 Canadian cities adjusted to a 0-10 scale to assist spatial and temporal comparison of air quality (Public Health Ontario, 2013).

5.2.6 Stormwater Runoff Management

The stormwater retention capacity is evaluated using the Curve Number (CN) method, a widely applied hydrological approach to estimate stormwater runoff (Morgan et al., 2004; Wang et al., 2025). The CN method relates land use, soil type, hydrologic conditions and local rainfall readings to predict runoff potential.

5.2.6.1 Data Collection

5.2.6.1.1 Land use

Land use was derived from SOLRIS. Land use data for 2011 and 2020 were used as proxies for 2013 and 2023.

5.2.6.1.2 Hydrological soil type

Soil type across the Region was derived from the Soil Survey Complex. The Soil Survey Complex database is developed and maintained by the Ontario Ministry of Agriculture, Food and Rural Affairs (OMAFRA), which contains information on the detailed soil parcels in the province, their texture and drainage capacity. This dataset was available only for 2014, and hence same soil typologies and their

respective hydrological classification are used for both 2013 and 2023 calculations. The soil types change over very long periods, with no or very little change over 10 years.

5.2.6.1.3 Ontario Annual Crop Inventory

This dataset is used to further refine the land use dataset, assigning crop typologies to broadly classified agriculture parcels in the SOLRIS datasets. The type of crop grown and the method of farming have a crucial impact on the texture and drainage capacity of the soil.

5.2.6.1.4 CN numbers

These values were derived from the Ontario Ministry of Transportation (MTO). The CN values typically range between 30 - 100 and reflect the capacity of the land surface to absorb rainfall. The lower values indicate higher infiltration (e.g., forested or permeable areas), and higher values represent lower infiltration and greater runoff potential (e.g., urban or impervious surfaces).

5.2.6.1.5 Rainfall (mm)

Rainfall or precipitation data for the region were derived from weather stations (maintained by Environment and Climate Change Canada and the University of Waterloo) for the years 2013 and 2023. This data was collected for 10 weather stations (4 within the regional boundary and 6 within a 50 km buffer of the boundary). The annual maximum precipitation value was extracted for each station to assess stormwater runoff under peak rainfall conditions.

5.2.6.2 Data Operationalisation

The intersect tool in ArcGIS Pro was used to combine the land use, soil hydrology and crop data to generate a single layer where each polygon is assigned with land use type (urban, agriculture-crop type, forests, etc.) and soil hydrology classification. Soil hydrology ranges from A (very well-drained soil - sand or gravel) to D (very poorly drained soil - clay or loams).

Next, MTO's CN table was modified to correspond to SOLRIS's land use codes and CN values were assigned to each polygon in the layer created above (Refer to Table in Annexure 1). A weighted CN was calculated for each 1km² grid cell based on the proportional area of different land use and soil typology combinations.

The annual maximum precipitation values from 10 weather stations were interpolated to estimate rainfall across the region. An average precipitation value was calculated for each grid cell.

Using the weighted CN values and average rainfall, stormwater runoff was calculated for each grid cell using the SCS-CN equation:

$$Q = (P - 0.2 * S)^2 / (P + 0.8 * S)$$

$$S = (25400 / CN) - 254$$

Where Q is runoff (in mm), P is average rainfall (in mm), and S is the potential maximum retention capacity of the surface. (See Figure 13)

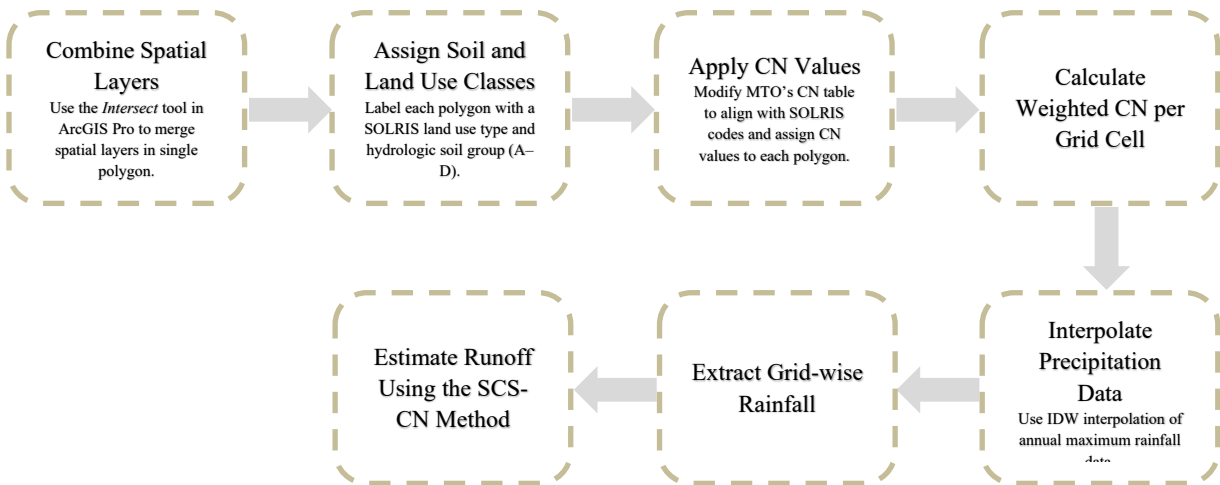


Figure 11: Steps to Quantify Stormwater Runoff

5.2.7 Green Spaces

The recreational spaces offer a wide variety of UESs (spaces for recreation, climate regulation, stormwater management, carbon sequestration, etc.), and multiple aspects of green spaces have been evaluated based on the approach of the study. In this study, which evaluates the impacts of urban intensification on the production of UESs, the quantity of available green spaces is evaluated. While the accessibility to and suitability of these spaces, along with their maintenance (such as availability of public facilities like washrooms), play a crucial role, these factors primarily reflect the social demands for recreational spaces. In contrast, this study focuses on the supply-side or production aspect, assessing the spatial distribution of the recreational spaces. This distinction helps evaluate whether the available green infrastructure supports the needs of a growing urban population.

5.2.7.1 Data Collection

To evaluate this service, SOLRIS land use data is used to identify green spaces and water bodies used as recreational spaces in the region for 2013 and 2023. Green spaces, such as urban forest, conservatories, parks, golf courses and sports fields, and water bodies such as lakes, ponds and rivers were extracted from the SOLRIS land use data (2011 and 2020 were used as proxies for 2013 and 2023, respectively).

5.2.7.2 Data Operationalisation

The selected polygons for green and blue land cover were verified manually with the satellite images for both the study years to reduce errors. A sum of all the recreational spaces was measured for each grid cell.

Finally, a percentage of available green spaces in each 1km x 1km grid cell was calculated by dividing the total area of recorded green spaces by the grid cell area.

Table 4: Unit of Measurement for Quantified Urban Ecosystem Services

Urban Ecosystem Service	Unit Measured	Unit of measurement
Food Production	Crop Yield	Bushels/ acre
Freshwater Quantity	Groundwater level	Meters
Freshwater Quality	Freshwater Quality Index	Cumulative Index (range: 0-6)
Air Temperature Regulation	Rise in air Temperature	°C
Air Quality	Air Quality Health Index	Index (range: 1-10)
Stormwater Runoff Management	Runoff Volume	Cubic Meter
Availability of Green Spaces	Green Space Area	Square Meter

5.2.8 Visualising Individual Urban Ecosystem Services on a 1km X 1km Grid

The 1km x 1km grid brings spatial uniformity to the highly heterogeneous datasets of diverse UESs. Further, these services are quantified in varying variables with different units of measurement. For a

comparable visualisation of these services, all the values for individual services are standardised using the z-score formula. Further, to limit the range of the values between 0-1, the min-max formula of normalisation is applied.

Finally, not all services are in the intuitive direction, where 0 means degradation of the service and 1 means improvement. The air temperature regulation and stormwater runoff management have an inverse interpretation of the normalised values, where lower runoff and a lower rise in temperatures are better. For a common understanding of the outcome, the normalised values of these two services are corrected for negation. In other words, the final normalised values were subtracted from 1 to reverse the direction and hence the interpretation.

The final normalised values for all the variables have the same direction, with lower values representing degradation and higher values referring to improvement in the performance of the services.

5.3 Spatial Assessment of Urban Intensification in the Regional Municipality of Waterloo

The quantification of urban growth is derived from the rise in built structures and the supporting infrastructure in the Region. This study specifically uses the built area ratio, which indicates the horizontal growth in the region, and building density (volume), which represents the vertical growth. While these indicators define the built forms, the change in impervious surfaces accounts for urban infrastructure such as roads, highways or outdoor recreational spaces such as tennis courts.

5.3.1 Mapping Urban Intensification

5.3.1.1 Data Collection

The primary data used to map urban intensification was the building footprint layer recorded by the Region of Waterloo. The Region's open data portal does not have access to archived data; however, the University of Waterloo's Geospatial Centre maintains the Region's previously published data. The building footprint data (shapefile) included information such as built floor area and number of storeys for each building polygon in the shapefile.

The road and highways, which contribute to the impacts of impervious surfaces, were extracted from the SOLRIS land use data.

5.3.1.2 Data Operationalisation

The number of storeys provided in the Region's building dataset was used to estimate the total height of each building. This step was necessary to derive vertical dimensions for built-up areas. An average storey height of 3 metres was assumed, and the total building height was calculated by multiplying the number of storeys by this average value.

To quantify the three-dimensional built form, the building volume was calculated for each building polygon. This was done by multiplying the built floor area with the previously derived building height, providing a consistent volume estimate for each structure.

To calculate the Impervious surfaces, the land cover coded as impervious surfaces and highways in SOLRIS's land use data was extracted for the study area boundaries. This land cover type in the data set included all the built surfaces, including built structures, roads and highways.

The total built floor area (m²), building volume (m³) and impervious area (m²) were aggregated for each 1 km × 1 km grid cell using the *Summarise Within* tool in ArcGIS Pro. This step enabled spatial analysis of built form at the grid level, allowing consistent comparison across the study area.

Finally, to make these values comparable with each other and the UESs, they are standardised using the z-score formula and then normalised for visualisation using the min-max formula.

Chapter 6

Findings

6.1 Introduction

This chapter provides an overview of the dependent and independent variables and examines the relationship between them. First, in *section 6.1*, the spatial distribution of each UES and urban intensification in the Regional Municipality of Waterloo for the years 2013 and 2023 is mapped and visualised using 1 km x 1 km grid cells. The changes over 10 years are also plotted on the same map. Second, in *section 6.2*, the outputs of cluster analysis are presented along with correlation tests between the services. Lastly, in the concluding *section 6.3*, I present multinomial regression analysis to examine the relation between UES and urban intensification across different clusters.

6.2 Urban Ecosystem Services

6.2.1 Crop Yield

Over the past decade, crop yield in the Regional Municipality of Waterloo has increased by 45.4%, while the total crop farmland area has grown by 22.47%, based on data from AAFC's Annual Crop Inventory. On average, this corresponds to an increase of nearly 4,500 bushels per acre in yield and an expansion of over 20 acres of farmland per grid cell (see Figures 15 and 16).

Figure 14 illustrates the spatial distribution of the change in crop yield across the region. In Figure 14c:

- 20% of grid cells, within the range of $[-22,700, 2 \text{ bu/acre}]$, show a decline in crop yield.
- Another 20%, in the range $[2, 2,003 \text{ bu/acre}]$, exhibit a moderate increase.
- The remaining 60%, ranging from 2,003 to 41,098 bu/acre, show a substantial rise in crop yield.

The lowest and highest ranges in Figure 14c represent outliers, marking areas of extreme decline or exceptional growth. These outliers are further visualised in Figure 15, which displays the distribution of changes in crop yield, including both negative and positive tails.

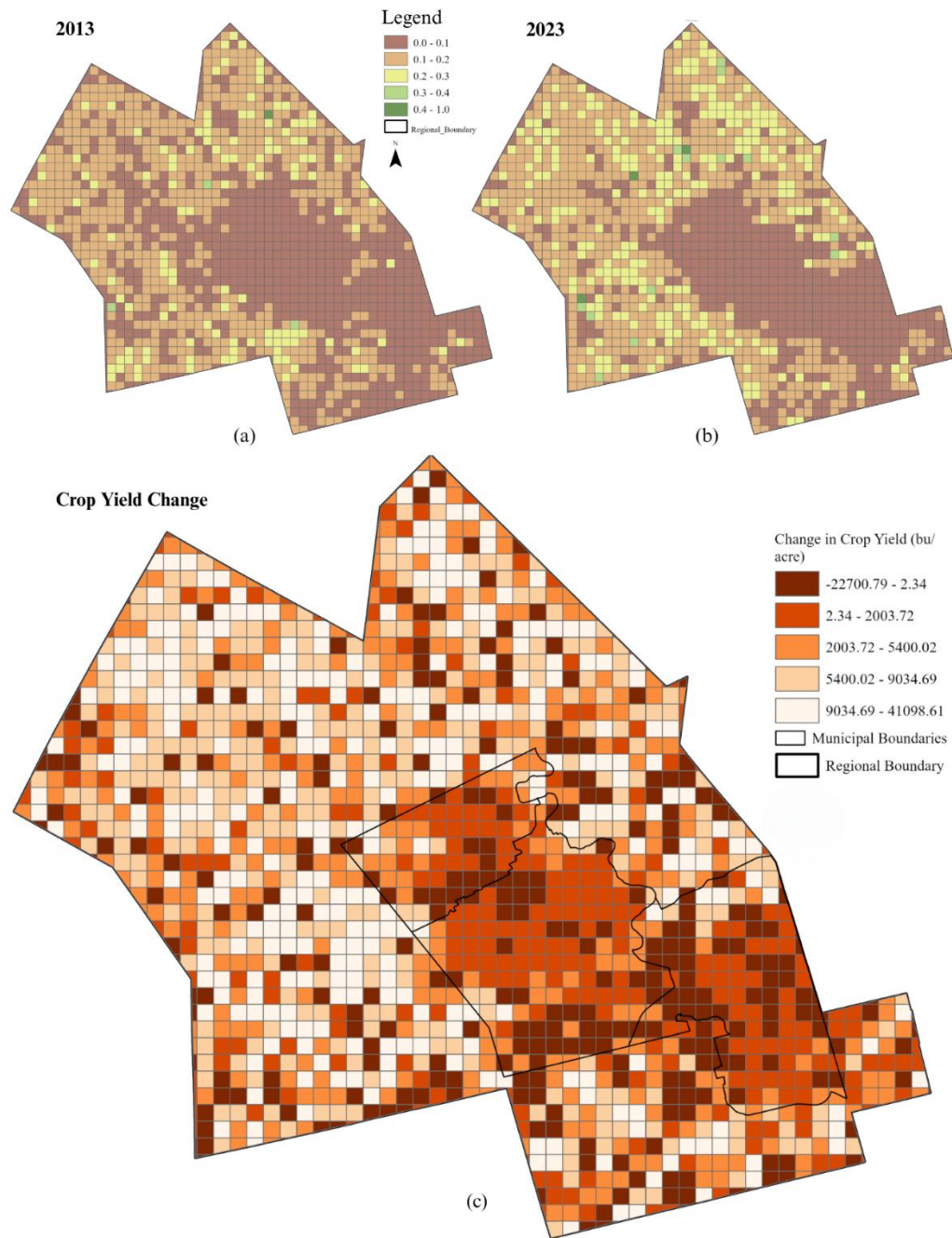


Figure 12: Maps for crop yield: (a) Map for crop yield in 2013; (b) Map for crop yield in 2023; (c) Map showing the change in crop yield

While the overall average crop yield has increased, the distribution is positively skewed, with several high positive outliers, as shown in the histogram (Figure 15). The median crop yield for the dataset is 3,218 bushels/acre, which is lower than the mean value of 3,566.48 bushels/acre, indicating that a small number of high-yield grid cells are pulling the average upward. The high standard deviation of 6,328.77 units reflects a wide variation in crop yield change across the Region. Notably, significant losses are observed along the edges of expanding urban areas (see Figure 14c, grid cells in the range -22,700.79 to 2.34), suggesting localised declines despite broader regional gains.

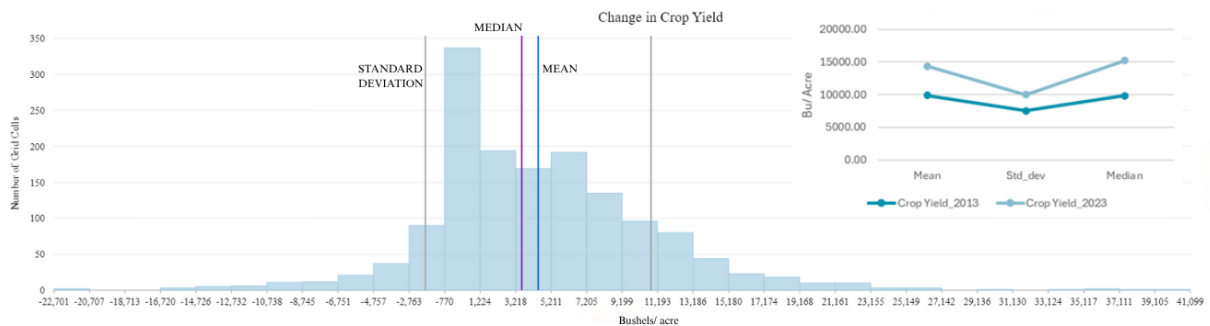


Figure 13: Distribution of change in crop yield

Compared to the ~45% increase in average crop yield, the growth in farmland area across the Region is relatively modest, at approximately 22%. However, the spatial distribution of farmland expansion mirrors the trends observed for crop yield (see Figure 16). Part of this increase may be attributed to revised federal definitions of agricultural cropland introduced in 2021 (Statistics Canada, 2021), which broaden the classification to include all land used for crop production.

Statistically, the median increase in farmland area per grid cell is 15 acres, which is lower than the average of 20 acres, suggesting that significant expansion has occurred only in a subset of grid cells, while most have experienced modest gains or even losses. This uneven pattern is further emphasised by a high standard deviation of 32 acres, highlighting the variability of farmland change across the region.

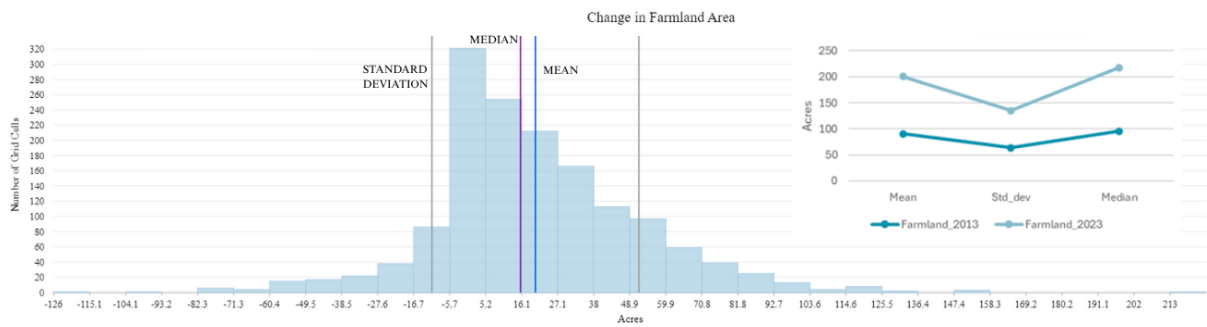


Figure 14: Distribution in the farmland area changes

6.2.2 Fresh Water Quantity - Ground Water Level

Groundwater levels have declined across the Regional Municipality of Waterloo, with an average drop of 2 meters per grid cell between 2013 and 2023 (see Figure 17). The spatial extent of low groundwater levels has expanded, particularly within and along the edges of urban boundaries (see grid cells in the [0, 0.2] range in Figures 18a and 18b).

Approximately 80% of the Region's population depends on groundwater for daily freshwater needs. As underground reservoirs shrink, greater pressure is likely to fall on the Grand River, which currently supplies the remaining 20% of the region's water demand (Region of Waterloo, 2024).

The most severe declines are observed along the urban periphery and in some agricultural areas, where groundwater levels have dropped by as much as 19.94 meters (see Figure 18c, grid cells in the range of -19.94m to -4.26m). Although urban areas also show a decline, the reductions are less extreme, typically ranging from -4.25m to -1.94m.

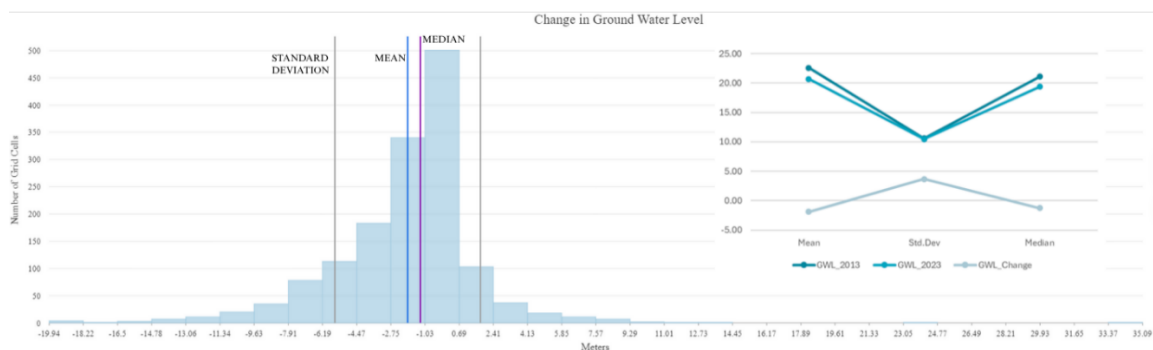


Figure 15: Distribution of change in groundwater levels

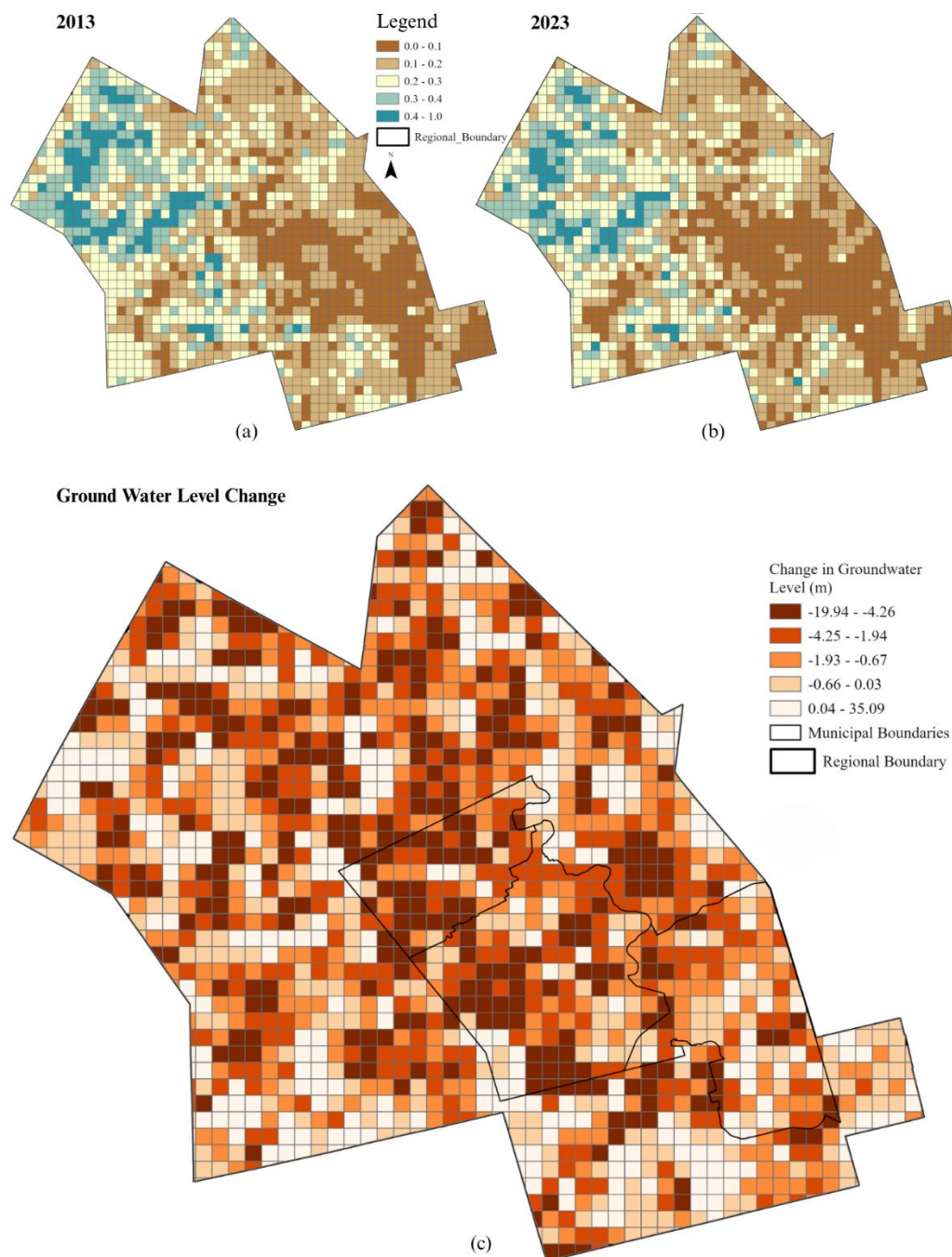


Figure 16: Maps for groundwater level: (a) Map for groundwater levels in 2013; (b) Map for groundwater levels in 2023; (c) Map showing the change in groundwater level

The relatively low standard deviation in groundwater level changes over the 10 years (see Figure 17) suggests a uniform regional decline. However, a few grid cells stand out as outliers, with groundwater levels showing either a sharp increase (up to 35 meters) or a steep decrease (up to 20 meters). These dramatic variations may be linked to local factors, such as protected lands, improved land use practices, or enhanced soil cover in areas with rising groundwater levels, and urban pressures, increasing impervious surfaces, or poor soil quality in areas experiencing decline.

6.2.3 Fresh Water Quality

Freshwater quality in the Regional Municipality of Waterloo, measured using the Water Quality Index (WQI), has experienced a slight average decline between 2013 and 2023. Although the average concentrations of individual contaminants used to calculate WQI have decreased slightly (see Figure 19), spatial analysis reveals notable degradation along urban areas (see Figures 21a and 21b).

The median WQI is lower than the mean, indicating that most grid cells have experienced a decline in water quality. In Figure 21c, which uses quantile classification to depict WQI change, approximately 60% of grid cells show a reduction, 20% show a modest increase, and the remaining 20% show a substantial improvement. However, these latter two categories include outliers, as reflected in the extreme ranges:

- $[-5, -0.48]$, indicating sharp declines, and
- $[0.69, 3.1]$, indicating sharp improvements (see Figure 21).

These outliers represent a small number of grid cells with dramatic changes and are also visible in the tails of the distribution shown in Figure 19.

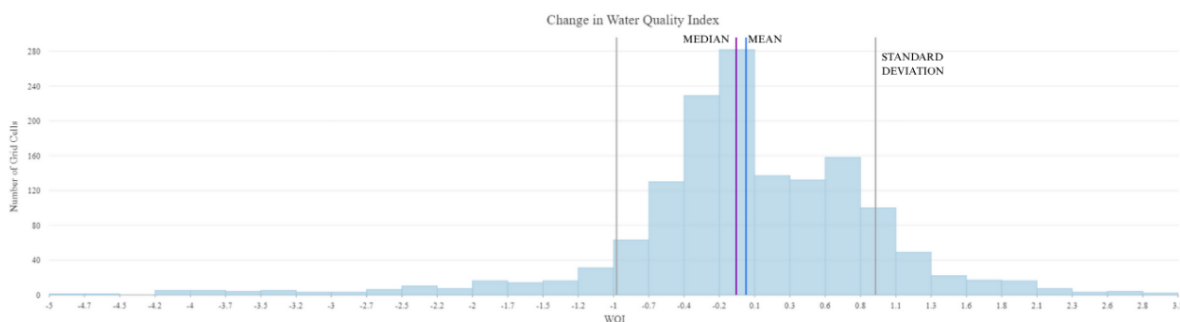


Figure 17: Distribution of change in freshwater quality index

The average rise of the contaminants in freshwater wells over 10 years, as seen in Figure 20, is approximately 0.7% for Salinity, 49% for Nitrates, 33% for Arsenic, 7% for Fluorides, 10% for Manganese and a fall in pH value by 2.7%. Despite a very concentrated spatial distribution of higher WQI change in urban regions (see Figure 21c), these values show a gradual but overall decline of water quality in the Region.

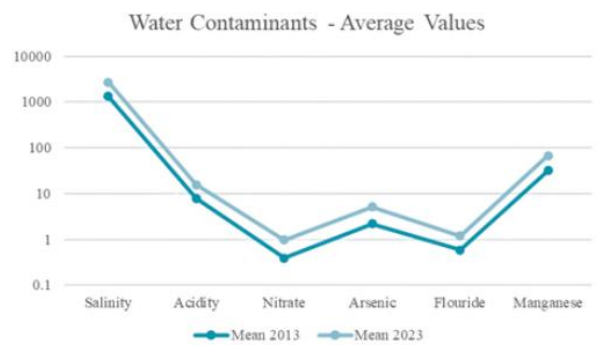


Figure 18: Change in average values of individual contaminants

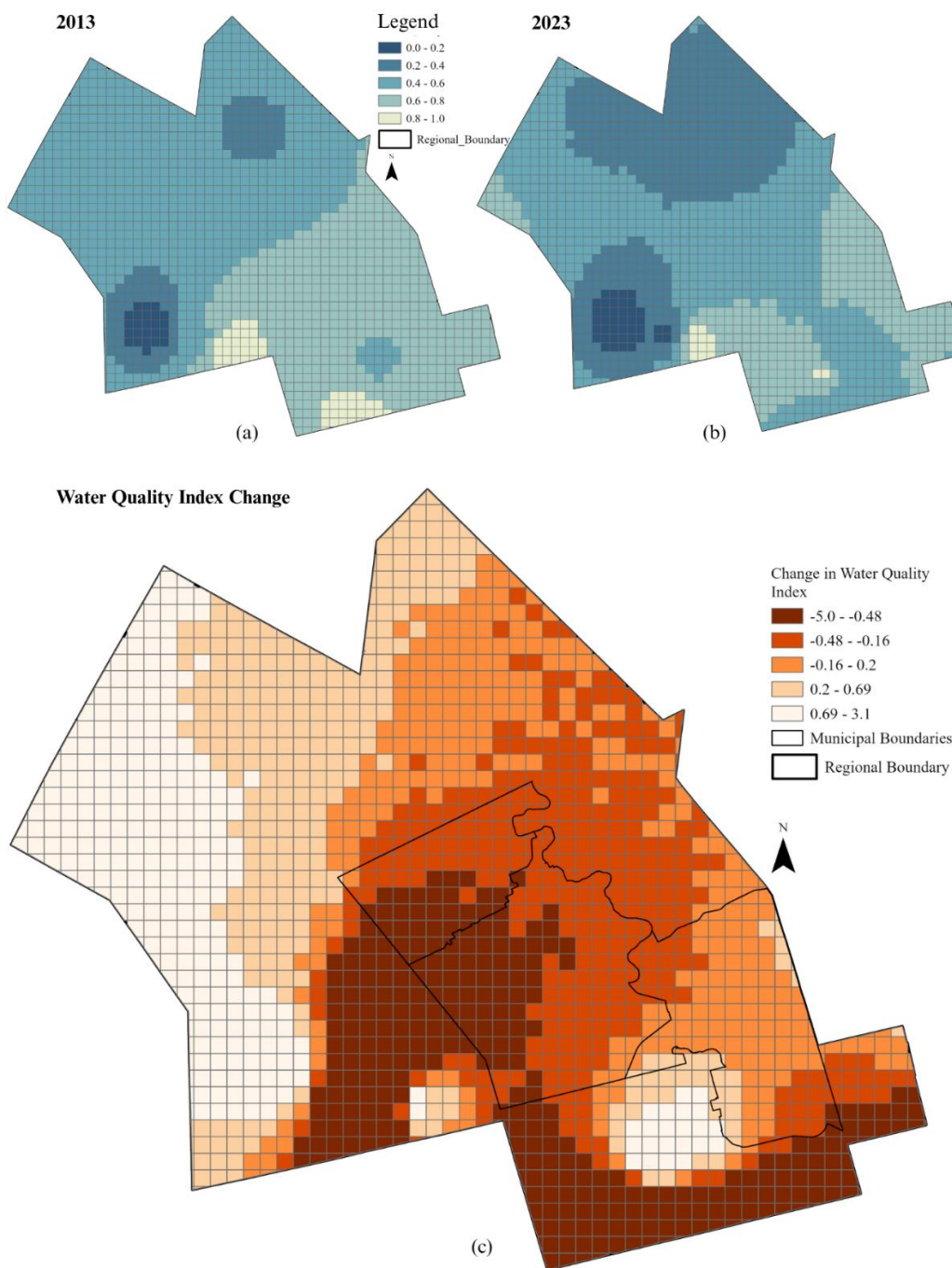


Figure 19: Maps for freshwater quality: (a) Map for freshwater quality in 2013; (b) Map for freshwater quality in 2023; (c) Map showing the change in freshwater quality

6.2.4 Air Temperature

The temperature-regulating ecosystem service for the Regional Municipality of Waterloo is assessed for the summer months (June to September). During this period, average air temperatures across the Region increased by 0.23°C. However, this rise is not spatially uniform. As shown in Figure 24c, some grid cells within the urban boundaries—particularly in the City of Waterloo—exhibit smaller increases, while steeper rises are observed in most grid cells within the Cities of Kitchener and Cambridge.

The distribution of temperature change is right-skewed, with over 60% of grid cells experiencing a below-average increase (see Figure 22). Along the regional boundary, grid cells tend to fall into either the highest or lowest categories of change (see Figure 24c), which may reflect minor inaccuracies due to the location of weather stations, many of which are positioned closer to urban cores. To enhance accuracy in these fringe areas, data from weather stations located just outside the administrative boundary have also been incorporated.

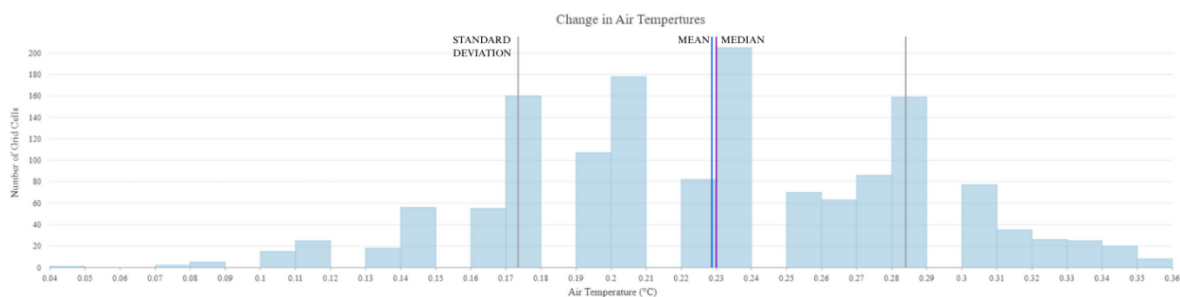


Figure 20: Distribution of change in air temperature rise

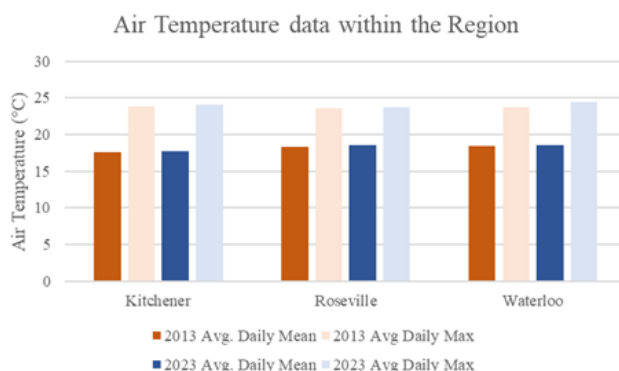


Figure 21: Variation in air temperature among the municipalities

The temperature data used in this analysis are derived from multiple weather stations located both **within and outside the regional boundary**. Based on records from **three stations within the Region**—two in the **City of Kitchener (Kitchener and Roseville)** and one in the **City of Waterloo**—the City of Kitchener experienced an **average increase of 0.2°C**

in both **daily maximum** and **daily mean temperatures** between 2013 and 2023 (See Figure 23).

In contrast, the City of Waterloo recorded a **larger rise in daily maximum temperatures**, averaging **0.7°C**, while the **daily mean temperature** increased by only **0.1°C** over the same period. Although the rise in Waterloo's daily mean temperature is relatively modest (slightly over 0.5%), the increase in daily maximum temperature is more substantial, amounting to an average rise of nearly **3%**.

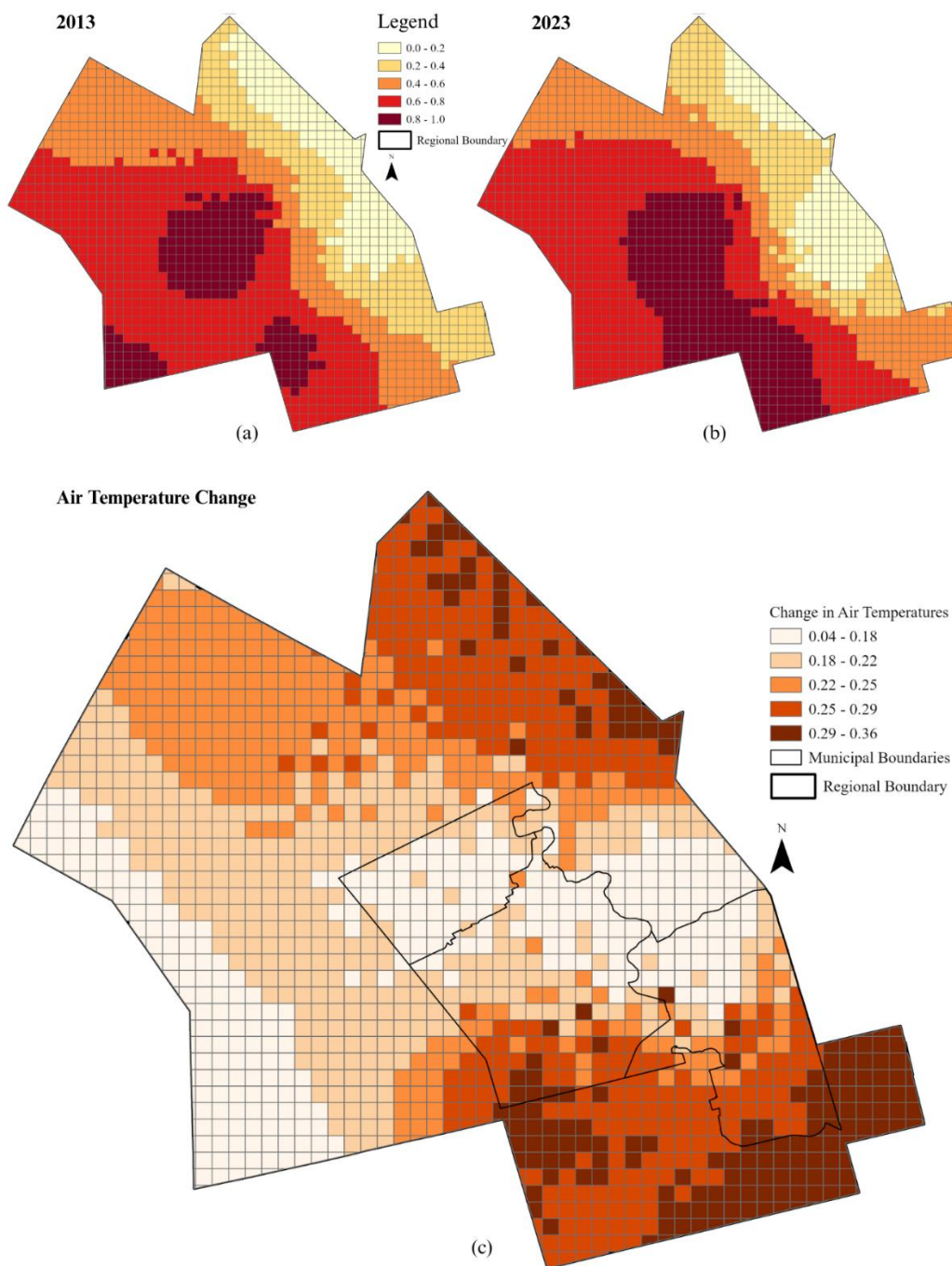


Figure 22: Maps for air temperature rise: (a) Map for air temperature in 2013; (b) Map for air temperature in 2023; (c) Map showing the change in air temperature

6.2.5 Air Quality

Air quality in the Regional Municipality of Waterloo has been assessed using the Air Quality Health Index (AQHI), which estimates the overall health and environmental impacts of key pollutants: NO₂, O₃, and PM_{2.5}. This analysis is based on data from a single monitoring station located in the City of Kitchener. Between 2013 and 2023, the average annual AQHI at this station improved slightly, decreasing from 4.02 to 3.79. According to Public Health Ontario's AQHI classification, both values fall within the moderate health risk category.

The data shows that the number of days with daily AQHI values in the 3–5 range (moderate risk) has increased, while the number of days in the 6–8 range (high risk) has decreased proportionally (see Figure 25). This indicates a potentially positive trend in overall air quality. However, it is important to note that extreme AQHI values have worsened: the highest daily AQHI recorded in 2023 was 11.86 (on three separate days), compared to a maximum of 9.57 (on two days) in 2013. This suggests that while average conditions have improved slightly, peak pollution events may be becoming more severe

The primary observations indicate an overall improvement in the average annual AQHI; however, the monthly trends reveal fluctuations in the rate of change and direction over time (see Figure 26). The months of April, May and June show a sharp rise in the AQHI (indicating higher health risks) while the months January and February show a substantial fall in the AQHI (indicating improvement of air quality).

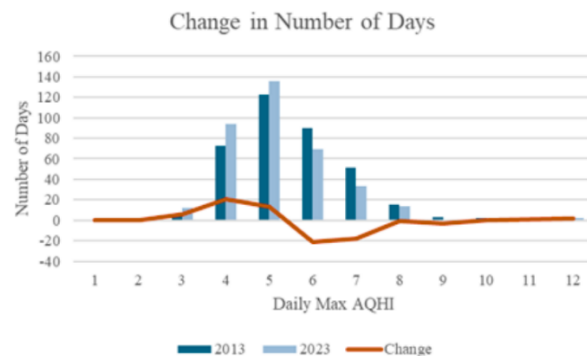


Figure 23: Graph showing change in the number of days in each AQHI

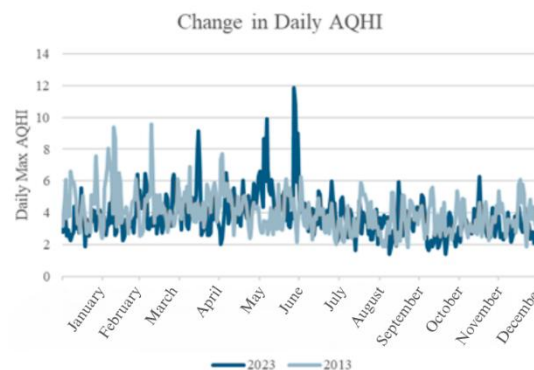


Figure 24: Graph showing the change in daily AQHI for 2013 and 2023

6.2.6 Stormwater Runoff

Stormwater runoff evaluates the capacity of land cover to retain or absorb incident rainfall in the Regional Municipality of Waterloo. Between 2013 and 2023, this capacity improved, meaning less water was running off into streets and drains. One reason for this change is that the region got about 16% less rain on average during this time. With less rain, the total amount of runoff water dropped by about 60%, showing there was simply less water flowing over the land. (See Figure 27)

While an overall decline in stormwater runoff is observed, the spatial distribution of this change is uneven (see Figure 28c). The last classification range ($-1,639.28 \text{ m}^3$ to $12,238.34 \text{ m}^3$), which accounts for 20% of all grid cells, reflects areas with either a minor decline or a significant increase in runoff. Within this group, approximately 30% of the grid cells show an increase in runoff, primarily located in the urban cores of Kitchener and Cambridge, with a few additional cells scattered along the edges of municipal boundaries.

Figures 28a and 28b further illustrate this pattern: decline in stormwater runoff are more prominent in non-urban areas, while grid cells with the highest runoff values (classified in the range 0.8-1.0) are increasingly concentrated in urban centres.

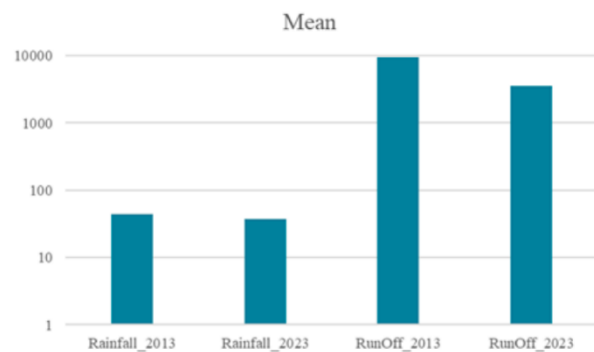


Figure 25: Change in average annual rainfall and surface runoff in 2013 and 2023

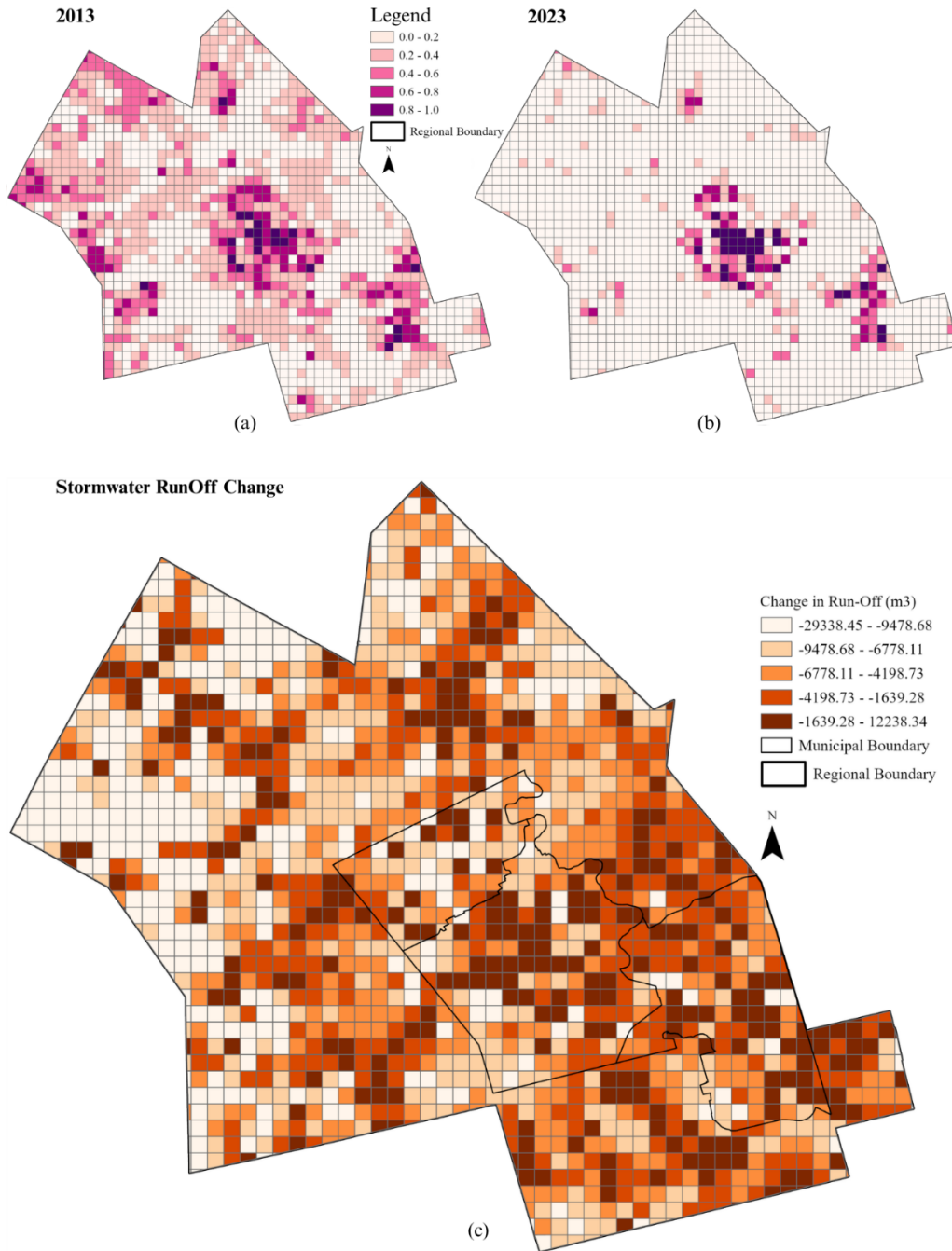


Figure 26: Maps for stormwater runoff: (a) Map for stormwater runoff in 2013; (b) Map for stormwater runoff in 2023; (c) Map showing the change in stormwater runoff

The average change in stormwater runoff per grid cell has decreased by approximately 5,700 cubic meters (see Figure 29). However, the large standard deviation highlights significant spatial variability in this change across the Region. Notably, only 6% of all grid cells exhibit an increase in stormwater runoff, including a small number of outliers that show especially sharp rises.

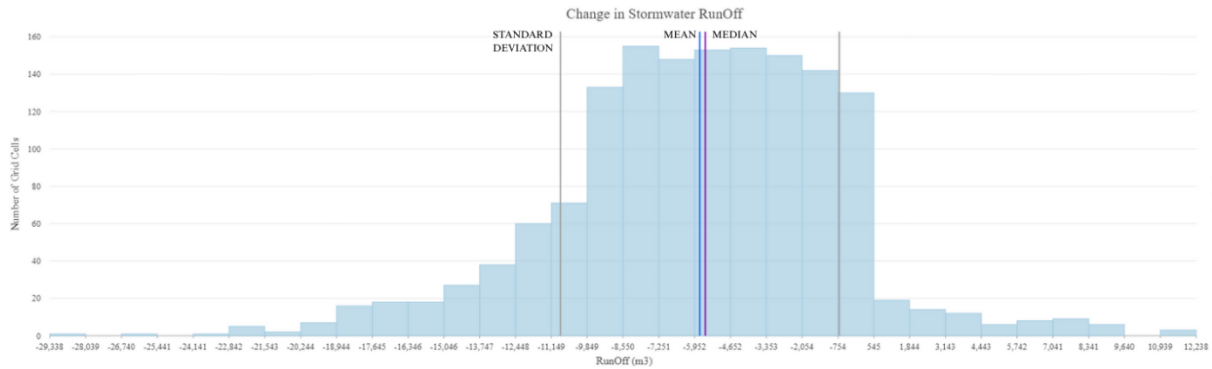


Figure 27: Distribution of change in stormwater runoff

The spatial distribution of grid cells with an increase in stormwater runoff can be further examined in relation to the Region’s rainfall data (measured in mm). Although the overall rainfall across the Region decreased between 2013 and 2023—by an average of 7.1 mm per grid cell—there is significant spatial variation in this trend (see Figure 30). The last two classification ranges in the map, primarily located within the municipal boundaries of Kitchener and Cambridge, show higher rainfall levels in 2023, ranging from 0 to 7.64 mm. These areas of localised increase in rainfall may help explain the corresponding rise in stormwater runoff observed in certain urban grid cells.

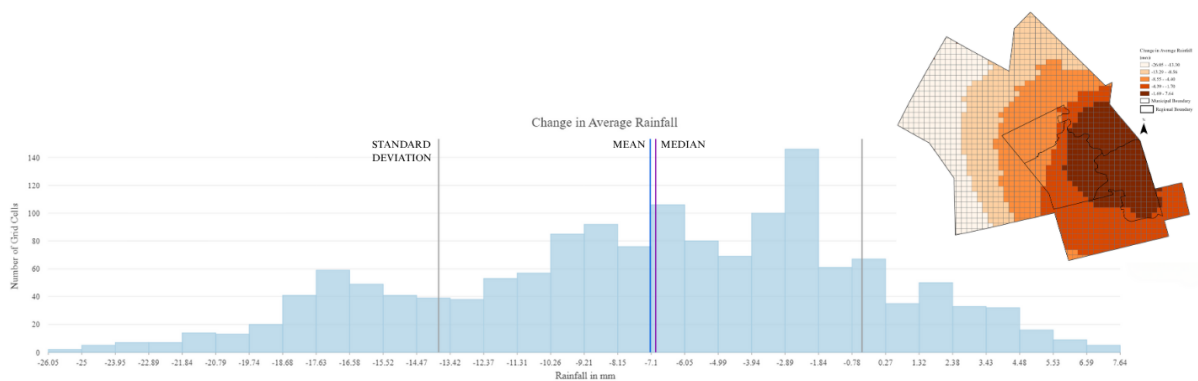


Figure 28: Distribution of change in annual average rainfall

6.2.7 Recreational Spaces

The urban ecosystem service (UES) of recreational spaces is assessed by measuring the change in green space area within each grid cell across the Regional Municipality of Waterloo. Between 2013 and 2023, the Region experienced an overall 40% decline in green space, with the most substantial losses occurring in urban areas, particularly along the fringes of municipal boundaries (see Figures 32a and 32b).

As shown in Figure 32c:

- 20% of grid cells show no change in green space (range [0, 0]).
- Another 20% experienced an increase in green space.
- The remaining 60% of grid cells show a decline, ranging from modest losses (0.04–0.12 km²) to significant reductions (0.13–0.83 km²).

The steepest losses, affecting about 20% of grid cells, are spatially clustered along municipal edges and in adjacent urbanised zones (see Figure 32c), highlighting the pressure of urban expansion on recreational land cover.

The median change in green space per grid cell is –0.02 units, which is higher than the average change of –0.07 units, indicating a negatively skewed distribution. This suggests that a larger number of grid cells experienced losses, including several extreme outliers (see Figure 31). Approximately 40% of grid cells show either a small decline, slight increase, or no change at all (range: –0.03 to 0.01), and these are primarily located outside urban areas or beyond municipal boundaries (see Figure 32).

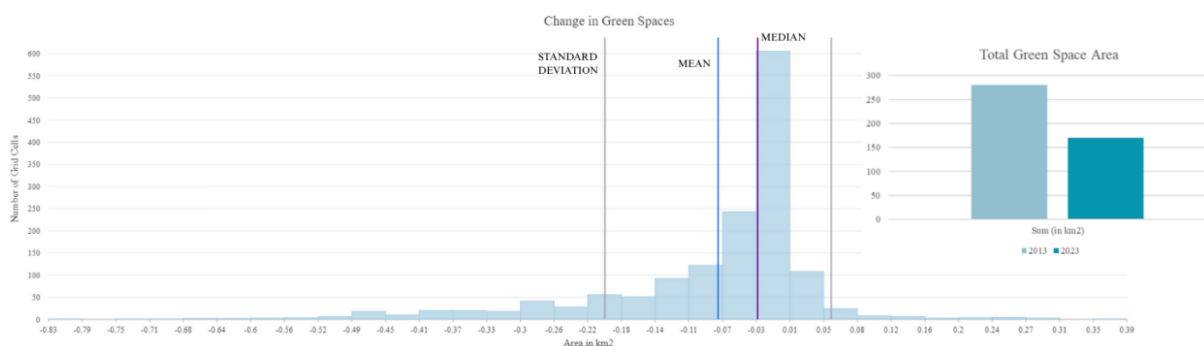


Figure 29: Distribution of change in green spaces

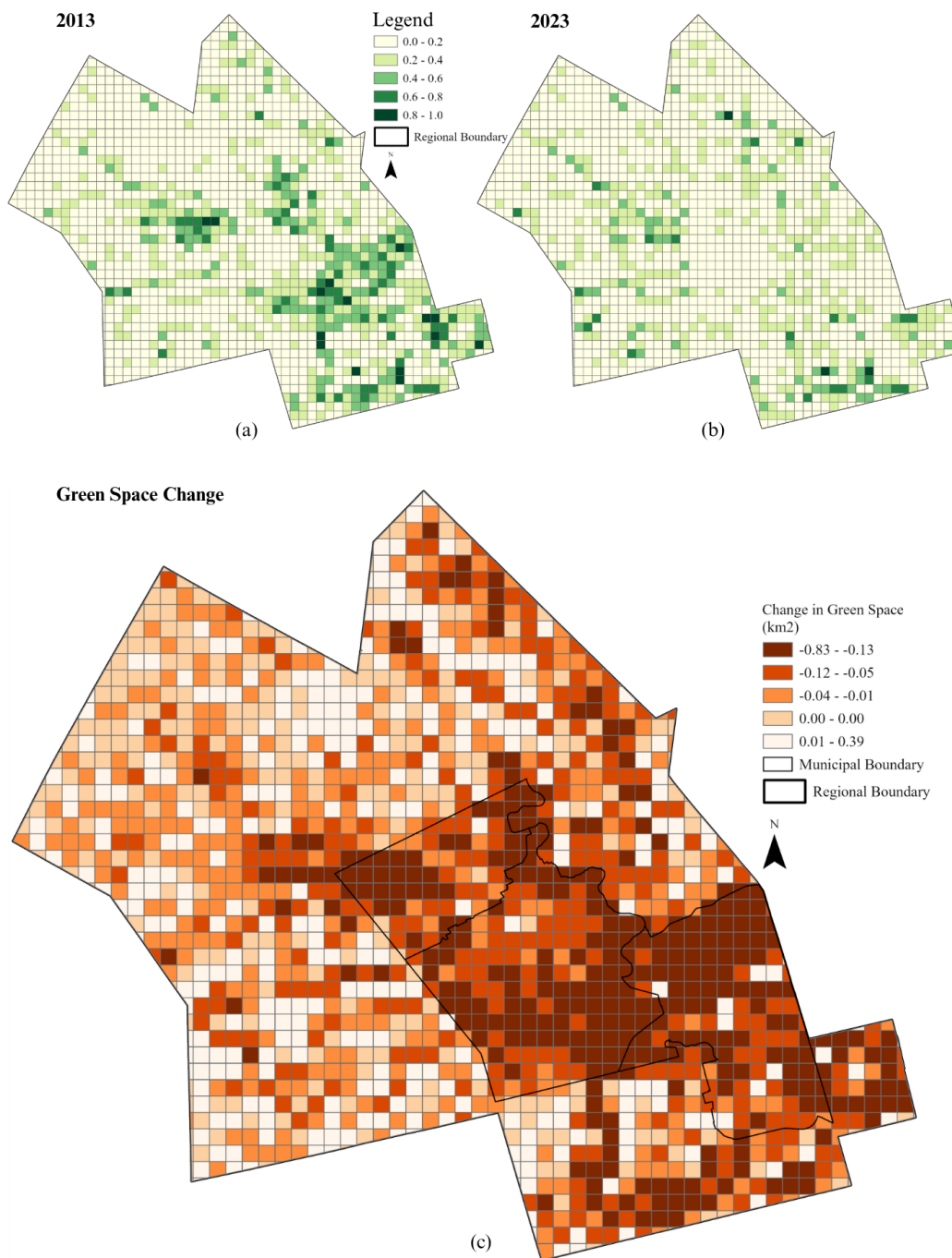


Figure 30: Maps for green spaces: (a) Map for green spaces in 2013; (b) Map for green spaces in 2023; (c) Map showing the change in green spaces

6.3 Urban Ecosystem Services Bundles

Initial observations from the spatial analysis of individual urban ecosystem services (UESs) suggest a degree of spatial coherence in the variation of their production capacities. For example, certain patterns—such as increased stormwater runoff or declining green spaces—are particularly evident along urban peripheries. To further investigate the spatial correlations among these changes, UESs were grouped into bundles across the entire region.

The bundling approach is used to identify clusters of grid cells that exhibit similar patterns of change across multiple services. This was achieved through Self-Organising Maps (SOM), a neural network-based tool for multidimensional clustering, pattern recognition, and dimensionality reduction. SOM preserves the topological structure of the data, making it well-suited for identifying spatially coherent clusters in high-dimensional datasets.

A SOM-based cluster analysis was conducted using the Kohonen library in R, running 100 iterations to ensure stability and consistency in the resulting clusters. Figure 33 below shows the results, clustering the grid cells into four bundles. While bundles 1 and 2 show a mix of urban and rural regions, bundles 3 and 4 are predominantly urban and rural, respectively.

6.3.1 Bundle 1 – Suburban Bundle

Bundle 1 consists of peri-urban regions and 2 townships. These cells are predominantly defined by the highest rise in air temperature (poor air temperature regulation, as seen in the rose diagram for bundle 1), moderate decline or maintenance of green spaces, and the highest decline in freshwater quality. A strong association is found in the air temperature regulation, freshwater quality and green spaces ($p < 0.01$); where if green spaces are reduced, the water quality drops and air temperature increases (see Table 5).

6.3.2 Bundle 2 – Urban Core

Bundle 2 consists of spatial variation including urban core regions, peripheries and rural townships. This bundle demonstrates a strong association between stormwater management and temperature regulation ($p < 0.01$; see Table 5), where both the services decline with an increased runoff and moderate rise in air temperature (see Table 5). However, the green spaces were maintained and

increased slightly, along with a relatively small decline in groundwater levels and improved underground water quality ($p < 0.01$).

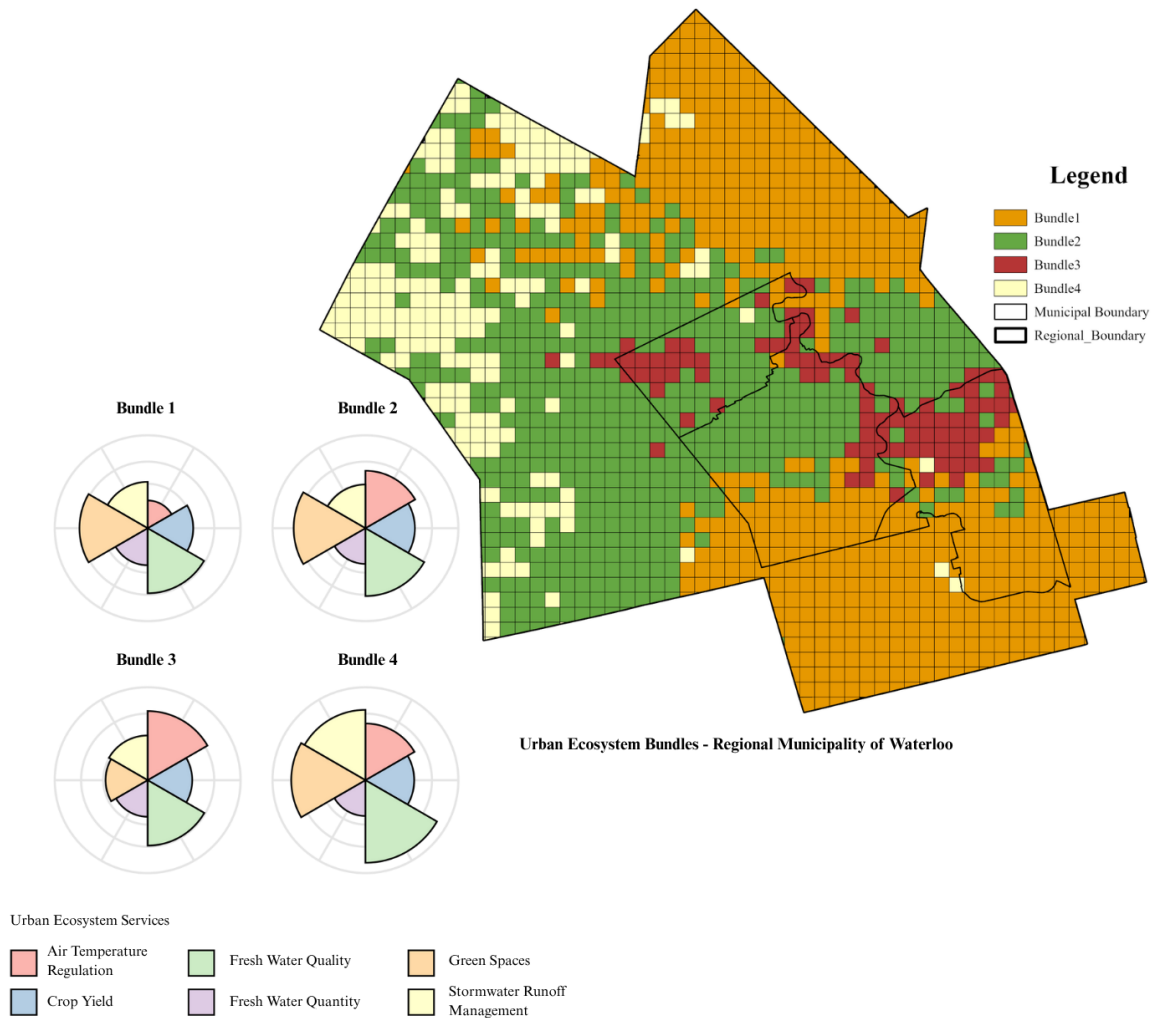


Figure 31: Urban Ecosystem Services Bundles

6.3.3 Bundle 3 – Peri-Urban

Bundle 3 mainly comprises urban cells, specifically those along the urban boundaries. While this region has seen the least rise in air temperature, other services have seen higher declines. The region experienced maximum loss of green spaces and forests. Additionally, the freshwater quantity has seen the highest decline in this region, with a modest decline in its quality. However, a strong negative correlation of statistical significance ($p < 0.01$) was observed only between temperature regulation and water quality (Table 5).

6.3.4 Bundle 4 - Rural

Finally, bundle 4, dominantly rural areas exhibit a steep rise in the green spaces, stormwater management (least runoff) and freshwater quality. The freshwater quantity is positively correlated ($p<0.01$) with these services, and it has seen a very small decline compared to other areas. The crop yield is inversely associated ($p<0.01$) with the services and has seen a below-average rise.

Table 5: Correlation among urban ecosystem services within each bundle

	<i>Urban Ecosystem Services</i>	<i>Crop Yield</i>	<i>Freshwater Quantity</i>	<i>Freshwater Quality</i>	<i>Air Temperature Regulation</i>	<i>Stormwater RunOff Management</i>	<i>Green Spaces</i>
Bundle1	<i>Crop Yield</i>	NA	0.09	0.028	-0.028	-0.016	0.17
	<i>Freshwater Quantity</i>	NA	NA	-0.038	-0.082	-0.107	-0.003
	<i>Freshwater Quality</i>	NA	NA	NA	0.323	-0.002	0.156
	<i>Air Temperature Regulation</i>	NA	NA	NA	NA	0.077	0.182
	<i>Stormwater RunOff Management</i>	NA	NA	NA	NA	NA	0.029
	<i>Green Spaces</i>	NA	NA	NA	NA	NA	NA
Bundle2	<i>Crop Yield</i>	NA	0.003	0.092	-0.093	0.102	0.166
	<i>Freshwater Quantity</i>	NA	NA	0.09	0.069	0.074	0.115
	<i>Freshwater Quality</i>	NA	NA	NA	0.037	0.1	0.146
	<i>Air Temperature Regulation</i>	NA	NA	NA	NA	0.133	0.009
	<i>Stormwater RunOff Management</i>	NA	NA	NA	NA	NA	0.128
	<i>Green Spaces</i>	NA	NA	NA	NA	NA	NA
Bundle3	<i>Crop Yield</i>	NA	0.115	-0.074	-0.073	0.149	0.089
	<i>Freshwater Quantity</i>	NA	NA	0.187	-0.121	0.012	-0.083
	<i>Freshwater Quality</i>	NA	NA	NA	-0.285	-0.062	-0.104
	<i>Air Temperature Regulation</i>	NA	NA	NA	NA	-0.162	-0.12
	<i>Stormwater RunOff Management</i>	NA	NA	NA	NA	NA	0.042
	<i>Green Spaces</i>	NA	NA	NA	NA	NA	NA
Bundle4	<i>Crop Yield</i>	NA	-0.253	-0.008	-0.26	-0.262	-0.05
	<i>Freshwater Quantity</i>	NA	NA	0.093	0.203	0.181	0.11
	<i>Freshwater Quality</i>	NA	NA	NA	0.26	0.21	-0.009
	<i>Air Temperature Regulation</i>	NA	NA	NA	NA	0.496	-0.038
	<i>Stormwater RunOff Management</i>	NA	NA	NA	NA	NA	0.023
	<i>Green Spaces</i>	NA	NA	NA	NA	NA	NA

Values Correlation with $p<0.01$

Values Correlation with $p>0.01$

While Bundles 3 and 4 are characterised by a dominant land use type, Bundles 1 and 2 display a heterogeneous mix of land uses. Variations in land cover composition—particularly the balance between impervious and pervious surfaces—as well as differences in the rate of urban intensification (i.e., the extent of new development), likely influence the functioning of ecosystem services within each bundle. To investigate these dynamics further, the following subsections present visualisations of land cover patterns across urban clusters, emphasising differences in intensification levels and their correlations with ecosystem service performance.

6.4 Mapping the Urban Landscape and Intensification Patterns in the Region of Waterloo

This section examines three key parameters of urban intensification: changes in impervious surfaces, built floor area (horizontal growth), and built volume (vertical growth). Between 2013 and 2023, the impervious surface area in the Regional Municipality of Waterloo increased by only 2.3%, while horizontal growth rose by 30%, and vertical growth by 47% (see Figure 34).

The relatively modest increase in impervious surfaces suggests limited outward urban expansion beyond existing urban boundaries. Instead, the spatial distribution of impervious surface growth—primarily located along the fringes of urban areas (see Figures 35a1 and 35a2)—points to the densification of semi-urban or previously developed zones, rather than the development of entirely new urban areas.

The significant increase in both horizontal development (30%) and vertical growth (47%) indicates that urban intensification is actively underway in the Regional Municipality of Waterloo. This trend makes the Region a compelling case for studying the impacts of intensification on urban ecosystem services (UESs).

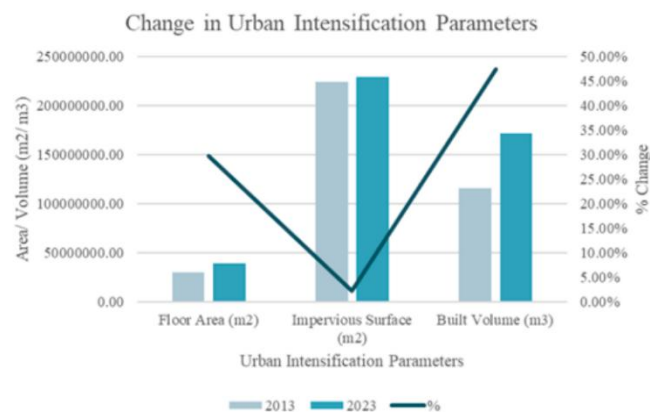


Figure 32: Graph showing change in urban parameters

Figure 35 presents normalised maps of urban development intensity, ranging from low to high, for each intensification parameter. To best capture spatial variability, classification ranges are tailored to each parameter, rather than using equal intervals. This approach accommodates the high degree of spatial variation and the presence of numerous outliers (see Figure 37). However, to support temporal comparisons, the same classification scheme is applied across both time periods within each parameter. The maps use the Jenks natural breaks method, which groups similar values while maximising differences between classes. This technique helps identify outliers and reveals spatial clustering in development patterns.

As shown in Figures 35b1, 35b2, 35c1, and 35c2, urban development is primarily concentrated within the municipal boundaries of the three cities. Figures 36a, 36b, and 36c, which illustrate changes from 2013 to 2023, further highlight spatial trends in growth. An increase in impervious surfaces is especially evident in peripheral grid cells, notably along the shared boundary between Kitchener and Cambridge, as well as on the outskirts of Waterloo.

Meanwhile, substantial horizontal and vertical development is most prominent in Waterloo and Cambridge, whereas central Kitchener shows moderate growth, with more intensified development along its fringes. All three parameters also reveal notable growth in select areas of the surrounding townships, indicating a broader regional pattern of intensification.

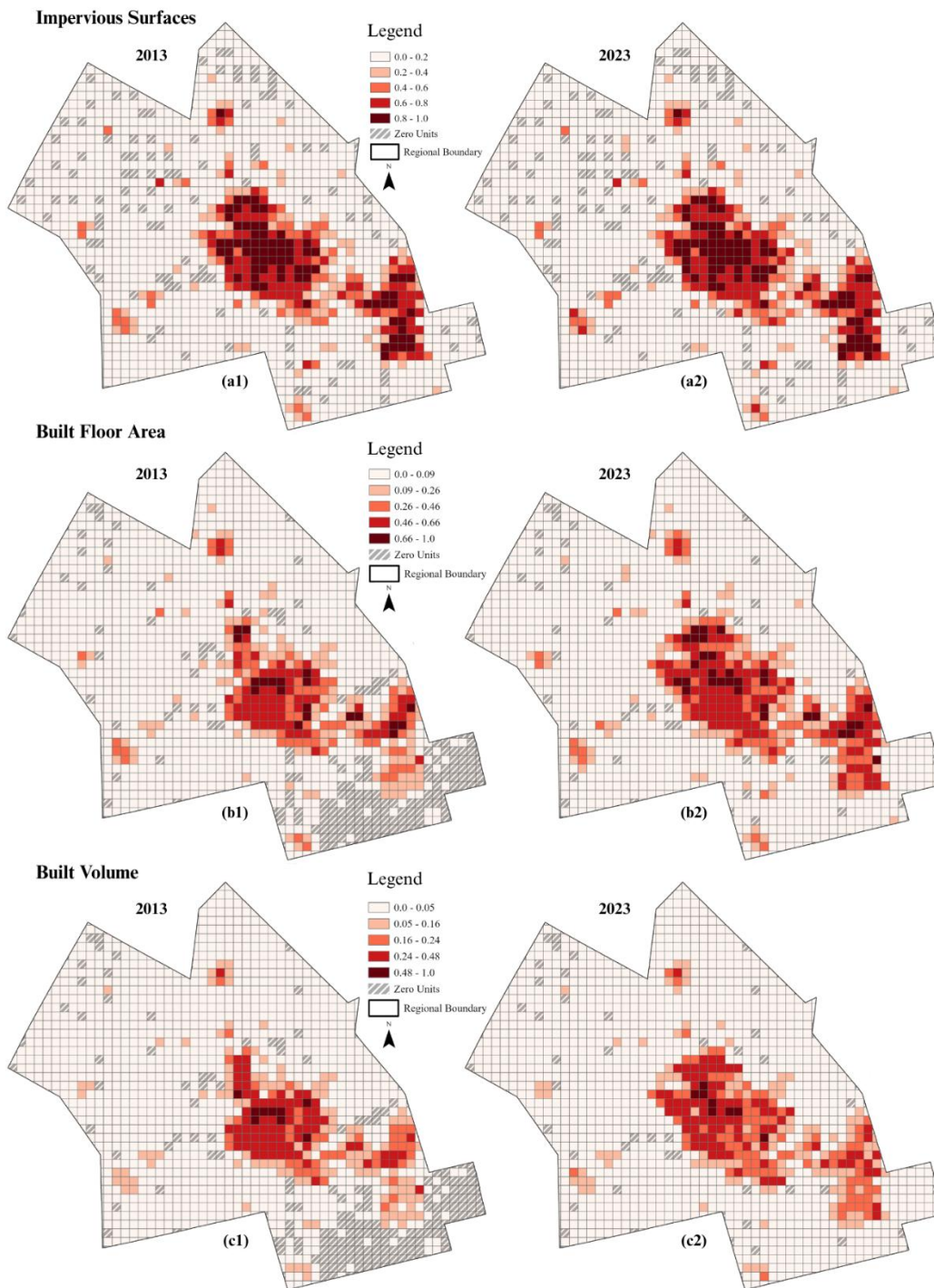


Figure 33: Maps for impervious surfaces, built floor area and building volume in 2013 (a1, b1, c1) and 2023 (a2, b2, c2)

Urban Spatial Changes Observed

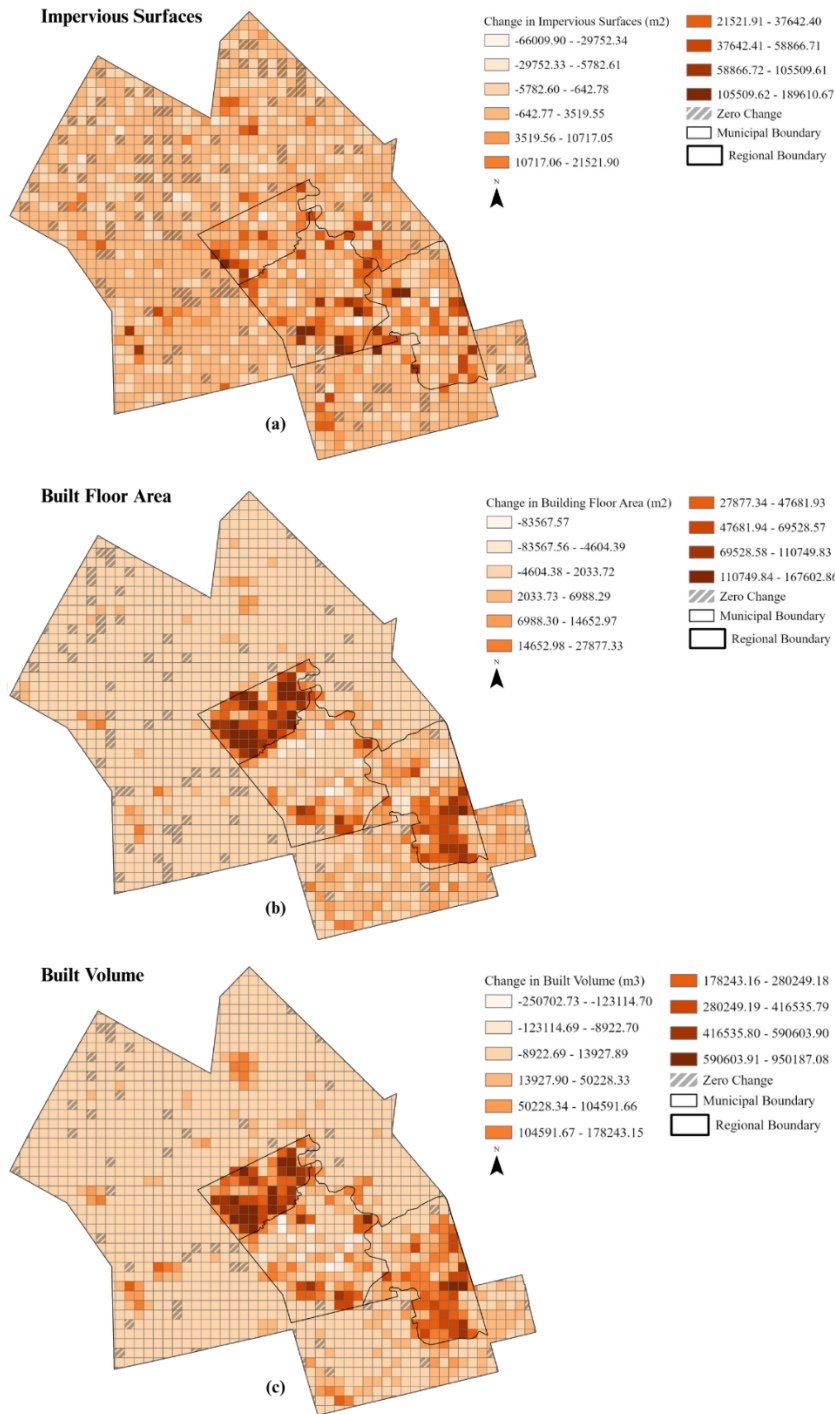


Figure 34: Maps for change in impervious surfaces (a), built floor area (b) and built volume (c)

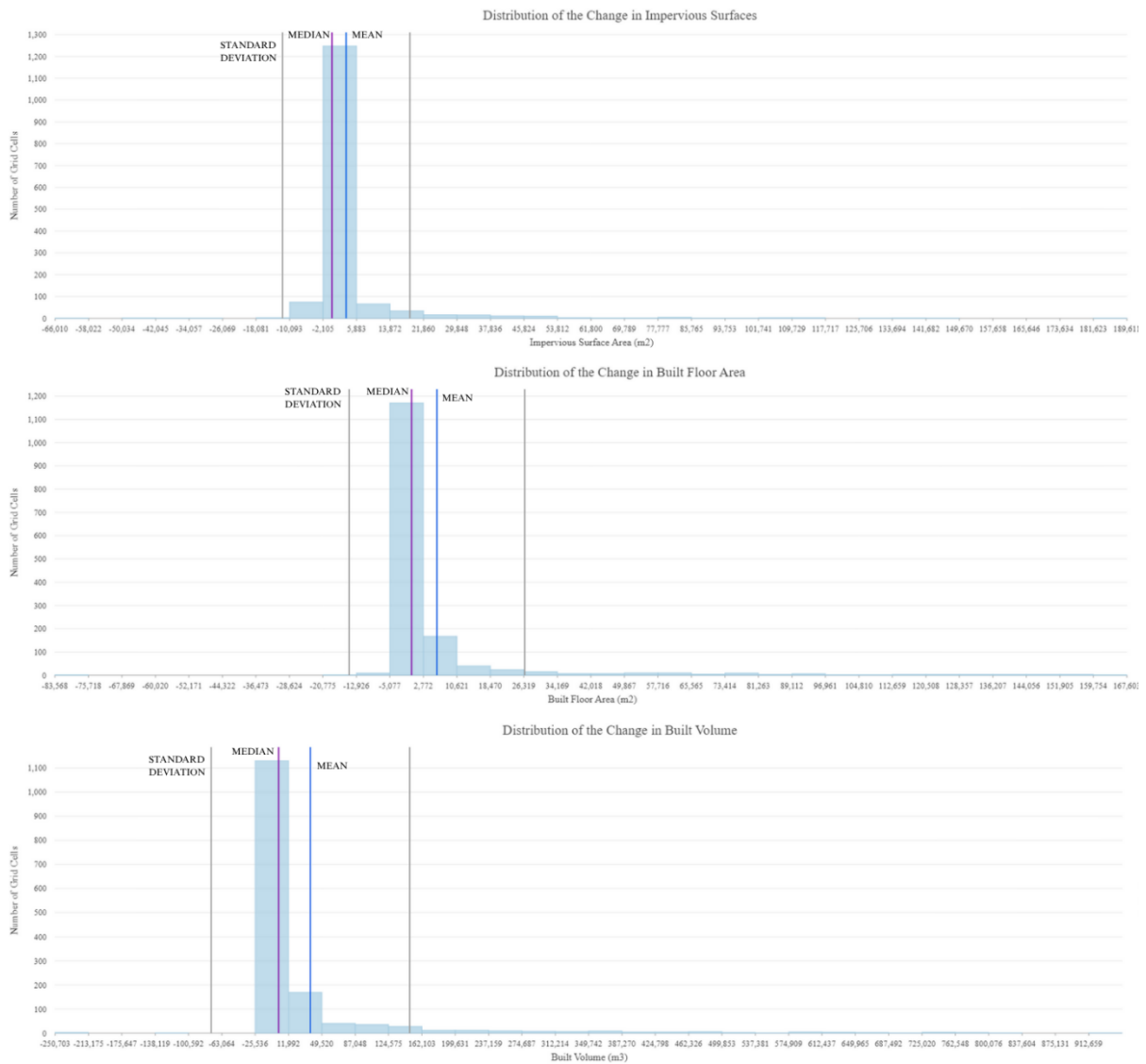


Figure 35: Distribution of change in impervious surfaces, built floor area and built volume

As shown in the histograms illustrating the distribution of changes in urban parameters across the Region, a large portion—approximately 1,100 grid cells—exhibits similar patterns of urban change between 2013 and 2023. This creates a limitation for analysis, as the uneven distribution of data points makes it difficult to apply uniform classification schemes. As a result, some grid cells are grouped into narrower data ranges, while others fall into broader categories. To avoid masking important variations, especially among grid cells with extreme values, naturally occurring data breaks

are used to define classification ranges. This approach allows for more accurate detection of spatial patterns and distribution across the Region.

6.5 Classifying and Visualising Urban Growth Clusters

Urban growth clusters in the Regional Municipality of Waterloo are classified using natural data breaks to reflect both land cover in 2013 and the relative intensification between 2013 and 2023, as described in the previous section. While much of the Region—particularly the non-urban townships—exhibits low levels of urban cover and minimal intensification, the urban municipalities display greater variation in development patterns.

Clusters are delineated based on the extent of impervious surfaces in 2013, which serves as the baseline for measuring urban intensification in this study. Grid cells with no or minimal built cover (<1%) are classified as Cluster 0 and excluded from further analysis. The remaining grid cells are categorized as follows:

- Non-urban clusters: <20% built cover (typically rural or agricultural land)
- Low urban clusters: 20–49% built cover
- Medium urban clusters: 50–79% built cover
- High urban clusters: ≥80% built cover

This classification captures the urban landscape gradient across the Region and supports further analysis of how urbanization levels affect ecosystem services (see map in Figure 38).

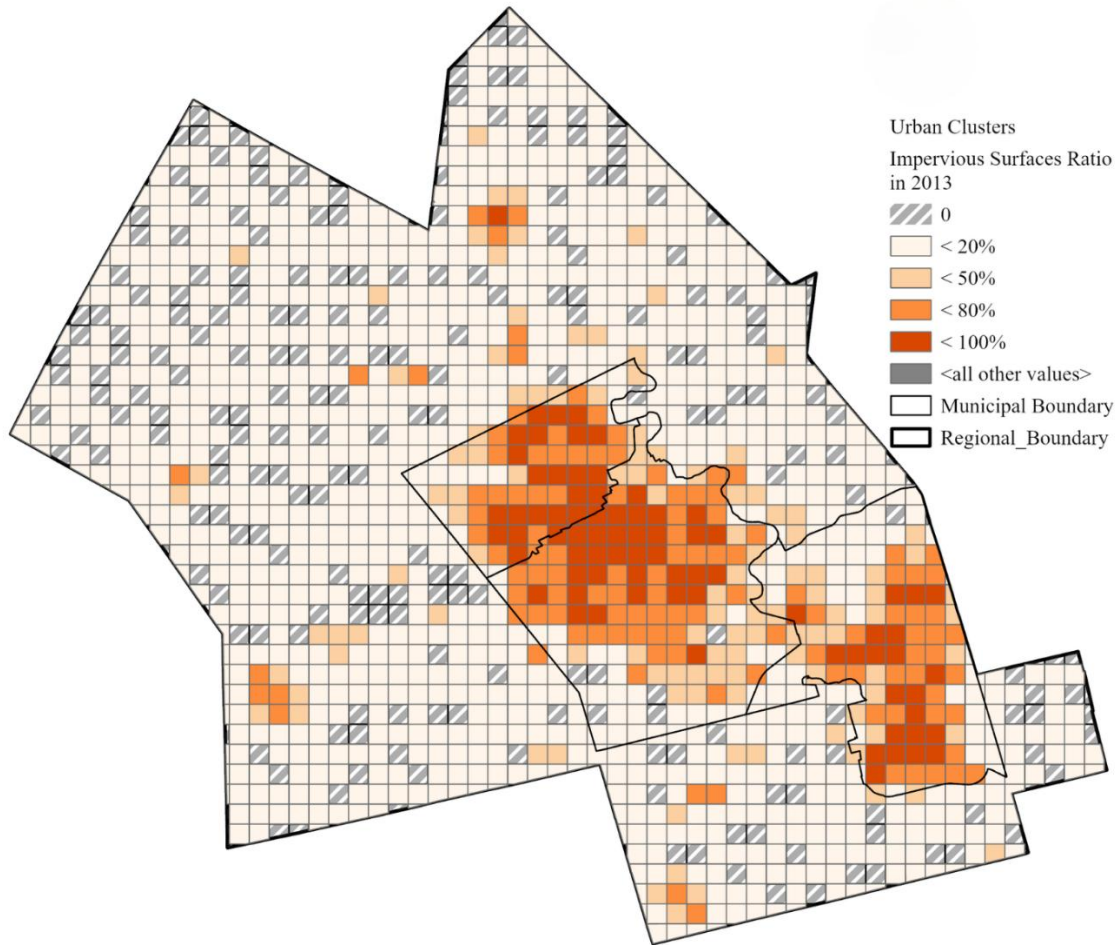


Figure 36: Map showing classification of urban clusters

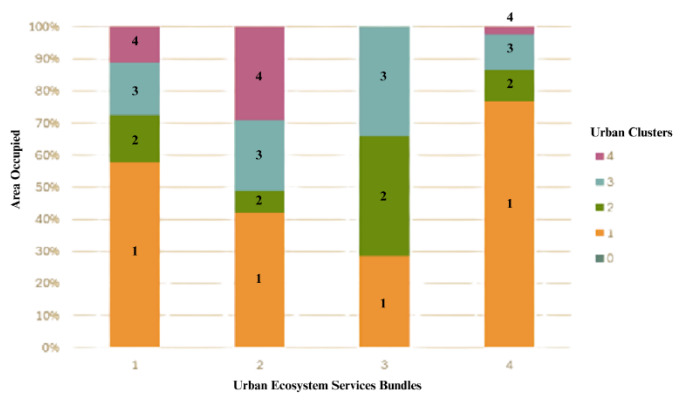


Figure 37: Graph showing distribution of grid cells in different clusters in each UES bundle

Further overlaying these urban clusters with the UESs bundles (Figure 39), a wide variation in urban land cover is observed, specifically in bundles 1 and 2, while very small variations are captured in bundles 3 and 4 owing to the single dominance of socio-economic activities. The next section reports the variations of impact in the UESs bundles using multinomial regression and correlation analysis.

6.6 Observing the Impacts of Varying Land Cover and Degree of Intensification in Urban Ecosystem Services Bundles

6.6.1 Multinomial Regression

The primary aim of this research is to examine the correlational impact of built land cover and the magnitude of its intensification on changes in urban ecosystem services (UESs). This is assessed using a multinomial regression model based on three key parameters:

- Average urban intensification within a UES bundle (i.e., built volume growth between 2013 and 2023).
- Baseline urban land cover in 2013, measured as the extent of impervious surfaces.
- Interaction effects between baseline land cover and subsequent intensification.

The first parameter isolates the independent impact of new urban growth. The second captures the effect of existing infrastructure (i.e., built cover in 2013), independent of later intensification. The third—and most critical—parameter captures how the effects of intensification vary depending on pre-existing land cover. For example, the same amount of new development may have very different impacts in areas with low vs. high initial urban cover. While the scale of intensification is accounted for, it is not classified into separate categories due to the resulting small sample sizes within each group.

Units of Measurement: *Urban intensification* is considered the change in built volume (m3), capturing both the horizontal and vertical dimensions of the total development in each grid cell. *Land Cover for TI* is measured using the sum of impervious lands in each grid cell from the SOLRIS data

for 2013. *UESs* have different units of assessment (refer to Table x in Chapter 5) and hence each UES is analysed in their respective units against 1 unit change in urban intensification (which is 1 m³).

The following regression model is applied across all four UES bundles for each individual ecosystem service:

Multinomial Regression Formula:

$$Y = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + \beta_3 (x_1 * x_2) + \varepsilon$$

$$\Delta UES_i = \beta_0 + \beta_1(\Delta UI) + \beta_2(LC_{t1}) + \beta_3 (\Delta UI * LC_{t1}) + \varepsilon$$

Where:

- ΔUES_i is the change in the *i*th ecosystem service.
- ΔUI denotes the change in urban intensification (built volume in m³).
- LC_{t1} is the baseline impervious surface area in 2013 (m²).
- ε represents the residual error.

And

- β_1 measures the effect of a 1 m³ increase in built volume, controlling for baseline land cover.
- β_2 captures the effect of existing land cover in 2013, relative to areas with no built cover, controlling for intensification.
- β_3 (interaction effect) reflects how the impact of intensification depends on the initial extent of land cover.

The tables below show the output results of regression analysis in regions occupied by different UESs bundles. The following tables read as follows: the values denote the impact on UESs (in specified units) for every 1 unit change in urban parameters (Intensification - 1 m³/ Urban Clusters - 1 m²), in reference to cluster zero - no built land cover or intensification as recorded in 2013.

6.6.1.1 Regression analysis for Bundle 1

Table 6: Regression analysis for Bundle 1

Impact Parameter (x)	Dependent Variables (Y)					
	Crop Yield (Bu/acre)	Freshwater Quantity (m)	Freshwater Quality (Index)	Air Temperature (°C)	Stormwater Runoff (m3)	Green Spaces (m2)
<i>Change in Urban Intensification</i>	-0.066	0.00001	-0.00001	0	0.039	-0.514
<i>Std.Error</i>	-0.042	-0.00002	-0.00001	0	-0.024	-0.855
<i>Urban Cluster 1</i>	851.106	-0.658*	0.021	-0.009***	-674.799*	11,026.92
<i>Std.Error</i>	-1,340.53	-0.989	-0.012	-0.022***	-1,498.353**	-45,735.420*
<i>Urban Cluster 2</i>	-1,251.26	-0.695	-0.166	-0.006	-711.896	-25,572.22
<i>Std.Error</i>	-693.97	-0.386	-0.092	-0.003	-394.83	-14,182.81
<i>Urban Cluster 3</i>	-2,121.31	-0.968	0.024	-0.026***	-1,599.500*	-75,787.600**
<i>Std.Error</i>	-1,582.12	-0.879	-0.21	-0.008	-900.136	-32,334.03
<i>Urban Cluster 4</i>	-3,435.51	-0.345	0.138	-0.029**	-2,213.53	63,999.32
<i>Std.Error</i>	-2,692.42	-1.496	-0.357	-0.013	-1,531.84	-55,025.49
<i>Urban Intensification in Cluster 1</i>	0.037	-0.00001	0.00001	0	-0.03	-0.208
<i>Std.Error</i>	-0.042	-0.00002	-0.00001	0	-0.024	-0.868
<i>Urban Intensification in Cluster 2</i>	0.054	-0.00001	0.00001	0	-0.032	-0.07
<i>Std.Error</i>	-0.043	-0.00002	-0.00001	0	-0.024	-0.873
<i>Urban Intensification in Cluster 3</i>	0.064	-0.00001	0.00001	0	=-0.04*	0.527
<i>Std.Error</i>	-0.042	-0.00002	-0.00001	0	-0.024	-0.862
<i>Urban Intensification in Cluster 4</i>	0.066	-0.00001	0.00001	0	-0.031	0.425
<i>Std.Error</i>	-0.043	-0.00002	-0.00001	0	-0.025	-0.888
<i>Constant</i>	3,684.648***	-0.955***	-0.260***	0.290***	-4,583.474***	-70,349.730***
<i>Std.Error</i>	-618.58	-0.344	-0.082	-0.003	-351.938	-12,642.06
<i>Observations</i>	624	624	624	624	624	624
<i>R2</i>	0.067	0.018	0.013	0.054	0.035	0.11
<i>Adjusted R2</i>	0.053	0.004	-0.002	0.04	0.021	0.097
<i>Residual Std. Error (df = 562)</i>	6,069.31	3.373	0.805	0.029	3,453.10	124,039.90
<i>F Statistic (df = 9; 562)</i>	4.898***	1.277	0.868	3.897***	2.473***	8.404***

Note *p<0.1; **p<0.05; ***p<0.01

The change in built volume, as part of UESs Bundle 1, does not show any statistically significant effect on any of the ecosystem services overall. The relatively small standard errors overlapping the

impact estimates suggest that the change is likely to have little to no real effect. While the individual urban clusters do display strong and statistically significant relationships with ecosystem services, the built volume itself, as an indicator of intensification, shows limited influence, with the exception of Cluster 3, which is predominantly urban. The rural cluster (>20% urban cover) has seen a drop in provisioning services, freshwater quantity and regulating services, such as temperature regulation and stormwater management, despite higher green spaces. Cluster 2 has no significant impacts; however, negative impacts on the provisioning services of food production and water quantity are seen. The 3rd cluster (predominantly urban) has the most significant impact. It shows a decline in regulating services such as temperature regulation and stormwater management, along with reduced green spaces. The fourth cluster sees an opposite impact with a drop in air temperature regulation despite increasing green spaces.

A substantial impact of change in intensification is seen only on the services of stormwater management in cluster 3, where for every 1000 m³ increase in built area, there is a rise in stormwater runoff by 40 m³.

6.6.1.2 Regression analysis for Bundle 2

Regression analysis for bundle 2 (dominantly peri-urban and rural), similar to bundle 1, shows no significant impact of intensification alone on any UESs. While the impacts of built land in clusters 1 and 2 are low, clusters 3 and 4 show heavy impacts. Rural cluster (1) has no substantial impact, with a slight increase in green spaces and lower runoff. Cluster 2 negatively impacts freshwater quantity and air temperature regulation with significant outcomes. Cluster 3, with medium urban cover, is the most degraded region in this bundle, showing highly significant negative impacts on all services except for a positive influence on stormwater. The most intense cluster demonstrates similar impacts to those of cluster 3, with a higher capacity to manage stormwater.

However, intensification in each cluster has no statistically significant impact on the services. Nevertheless, negative impacts on crop yield, water services, runoff and green spaces are recorded.

Table 7: Regressions for Bundle 2

Impact Parametere (x)	Dependent Variables (Y)					
	Crop Yield (Bu/acre)	Freshwater Quantity (m)	Freshwater Quality (Index)	Air Temperature (°C)	Stormwater Runoff (m3)	Green Spaces (m2)
<i>Change in Urban Intensification</i>	-0.051	-0.0002	0.0001	0	0.167	1.263
<i>Std.Error</i>	-0.548	-0.0003	-0.0001	0	-0.379	-5.516
<i>Urban Cluster 1</i>	389.57	-0.379	-0.184	-0.005	-391.716	12,213.38
<i>Std.Error</i>	-736.358	-0.433	-0.115	-0.004	-509.956	-7,416.73
<i>Urban Cluster2</i>	-2,010.29	-2.364**	-0.321	-0.025***	-1,140.01	-3,950.02
<i>Std.Error</i>	-1,666.96	-0.981	-0.259	-0.009	-1,154.44	-16,789.96
<i>Urban Cluster 3</i>	-6,313.376***	-2.452***	-0.572***	-0.010*	1,480.895*	100,293.800***
<i>Std.Error</i>	-1,158.17	-0.682	-0.18	-0.006	-802.077	-11,665.29
<i>Urban Cluster 4</i>	-6,970.090***	-1.315*	-0.673***	-0.009	5,043.426***	-42,188.550***
<i>Std.Error</i>	-1,187.94	-0.699	-0.185	-0.006	-822.691	-11,965.10
<i>Urban Intensification in Cluster 1</i>	-0.026	0.0002	-0.0001	0	-0.185	-1.651
<i>Std.Error</i>	-0.548	-0.0003	-0.0001	0	-0.38	-5.522
<i>Urban Intensification in Cluster 2</i>	0.036	0.0002	-0.0001	0	-0.172	-1.47
<i>Std.Error</i>	-0.548	-0.0003	-0.0001	0	-0.379	-5.518
<i>Urban Intensification in Cluster 3</i>	0.052	0.0002	-0.0001	0	-0.169	-1.254
<i>Std.Error</i>	-0.548	-0.0003	-0.0001	0	-0.379	-5.517
<i>Urban Intensification in Cluster 4</i>	0.051	0.0002	-0.0001	0	-0.174	-1.224
<i>Std.Error</i>	-0.548	-0.0003	-0.0001	0	-0.379	-5.516
<i>Constant</i>	7,161.425***	-1.319***	0.197*	0.201***	-4,295.059***	-33,363.770***
<i>Std.Error</i>	-648.14	-0.382	-0.101	-0.003	-448.862	-6,528.18
<i>Observations</i>	572	572	572	572	572	572
<i>R2</i>	0.178	0.087	0.05	0.031	0.107	0.238
<i>Adjusted R2</i>	0.165	0.072	0.034	0.016	0.092	0.226
<i>Residual Std. Error (df = 562)</i>	6,168.95	3.632	0.96	0.032	4,272.23	62,134.72
<i>F Statistic (df = 9; 562)</i>	13.503***	5.935***	3.262***	2.011**	7.462***	19.475***

Note *p<0.1; **p<0.05; ***p<0.01

6.6.1.3 Regression analysis for Bundle 3

Table 8: Regressions for Bundle 3

Impact Parameter (x)	Dependent Variables (Y)					
	Crop Yield (Bu/acre)	Freshwater Quantity (m)	Freshwater Quality (Index)	Air Temperature (°C)	Stormwater Runoff (m3)	Green Spaces (m2)
<i>Change in Urban Intensification</i>	173.031	-0.077	-0.002	-0.001	-222.221	27,144.070***
<i>Std.Error</i>	-304.15	-0.19	-0.002	-0.002	-197.756	-7,902.88
<i>Urban Cluster 1</i>	1,726.23	0.134	0.065***	-0.026	368.932	-138,305.70
<i>Std.Error</i>	-3,363.52	-2.106	-0.019	-0.021	-2,186.90	-87,394.83
<i>Urban Cluster 2</i>	-1,266.41	0.116	0.051**	-0.032	871.306	-214,275.400**
<i>Std.Error</i>	-3,461.46	-2.167	-0.02	-0.022	-2,250.58	-89,939.71
<i>Urban Cluster 3</i>	-2,788.48	-0.079	0.053**	-0.021	-783.62	-59,372.72
<i>Std.Error</i>	-3,675.80	-2.301	-0.021	-0.023	-2,389.95	-95,509.06
<i>Urban Intensification in Cluster 1</i>	-173.054	0.077	0.002	0.001	222.223	-27,144.520***
<i>Std.Error</i>	-304.154	-0.19	-0.002	-0.002	-197.756	-7,902.88
<i>Urban Intensification in Cluster 2</i>	-173.027	0.077	0.002	0.001	222.216	-27,143.750***
<i>Std.Error</i>	-304.154	-0.19	-0.002	-0.002	-197.756	-7,902.88
<i>Urban Intensification in Cluster 3</i>	-173.026	0.077	0.002	0.001	222.221	-27,144.070***
<i>Std.Error</i>	-304.154	-0.19	-0.002	-0.002	-197.756	-7,902.88
<i>Constant</i>	3,090.39	-1.444	0.540***	0.185***	-4,670.950**	-223,872.800***
<i>Std.Error</i>	-3,267.98	-2.046	-0.019	-0.021	-2,124.79	-84,912.48
<i>Observations</i>	91.00	91	91	91	91.00	91.00
<i>R2</i>	0.114	0.074	0.295	0.066	0.063	0.262
<i>Adjusted R2</i>	0.039	-0.004	0.236	-0.012	-0.016	0.199
<i>Residual Std. Error (df = 562)</i>	4,626.58	2.896	0.027	0.029	3,008.13	120,213.30
<i>F Statistic (df = 9; 562)</i>	1.52	0.944	4.962***	0.845	0.796	4.200***

Note *p<0.1; **p<0.05; ***p<0.01

This bundle is relatively smaller than the others and dominantly occupies urban regions. While an overall impact of intensification is seen across all the services, only the positive impact on green spaces has statistical significance. Further, individual clusters show a subtle but significant positive impact only on the water quality across the clusters. However, the impact varies within the clusters. Further, a heavy association with loss of green spaces, contrary to the impact of overall intensification, is significant in cluster 2. This contrasting effect on green spaces is significant in the impacts of intensification equally in all the clusters. Absence of grid cells in cluster 4 (highest artificial surfaces and intensification) indicates a relatively recent interaction with intensification.

While this cluster occupies the smallest region, each cluster has substantial instances of grid cells to consider these results for this study.

6.6.1.4 Regression analysis for Bundle 4

Table 9: Regressions for Bundle 4

Impact Parameter (x)	Dependent Variables					
	Crop Yield (Bu/acre)	Freshwater Quantity (m)	Freshwater Quality (Index)	Air Temperature (°C)	Stormwater Runoff (m3)	Green Spaces (m2)
<i>Change in Urban Intensification</i>	1.571**	-0.00002	0.0002***	0	-0.326	3.63
<i>Std.Error</i>	-0.64	-0.001	-0.0001	-0.00001	-0.431	-4.58
<i>Urban Cluster 1</i>	-186.28	-0.732	0.171*	0.002	819.78	13,434.400**
<i>Std.Error</i>	-897.55	-0.746	-0.102	-0.007	-607.85	-6,467.00
<i>Urban Cluster 2</i>	6,459.008***	-0.982	-0.01	-0.008	1,024.06	-3,610.95
<i>Std.Error</i>	-2,393.54	-1.989	-0.273	-0.019	-1,620.99	-17,245.91
<i>Urban Cluster 3</i>	-3,042.26	-1.96	-0.667**	-0.009	-340.083	34,260.130*
<i>Std.Error</i>	-2,807.49	-2.333	-0.32	-0.023	-1,901.33	-20,228.49
<i>Urban Cluster 4</i>	-1,181,177.000**	18.099	-154.653***	1.544	243,069.80	-2,637,864.00
<i>Std.Error</i>	-475,997.10	-395.556	-54.265	-3.843	-322,361.80	-3,429,654.00
<i>Urban Intensification in Cluster 1</i>	-1.566**	0.00003	-0.0002***	0	0.34	-3.48
<i>Std.Error</i>	-0.637	-0.001	-0.0001	-0.00001	-0.432	-4.591
<i>Urban Intensification in Cluster 2</i>	-1.605**	-0.00001	-0.0002***	0	0.308	-3.524
<i>Std.Error</i>	-0.637	-0.001	-0.0001	-0.00001	-0.431	-4.588
<i>Urban Intensification in Cluster 3</i>	-1.555**	0.00003	-0.0002***	0	0.338	-4.19
<i>Std.Error</i>	-0.637	-0.001	-0.0001	-0.00001	-0.431	-4.588
<i>Urban Intensification in Cluster 4</i>	Very low availability grid cell)					
<i>Std.Error</i>						
<i>Constant</i>	5,261.390***	-1.521**	0.930***	0.196***	-14,672.020***	-24,248.550***
<i>Std.Error</i>	-759.883	-0.631	-0.087	-0.006	-514.619	-5,475.11
<i>Observations</i>	191	191	191	191	191	191
<i>R2</i>	0.086	0.035	0.11	0.048	0.026	0.083
<i>Adjusted R2</i>	0.046	-0.008	0.07	0.006	-0.017	0.043
<i>Residual Std. Error (df = 562)</i>	5,296.22	4.401	0.604	0.043	3,586.78	38,160.30
<i>F Statistic (df = 9; 562)</i>	2.140**	0.816	2.798***	1.153	0.611	2.070**

Note *p<0.1; **p<0.05; ***p<0.01

The last bundle (4) shows contrasting built environment conditions to those of cluster 3, mainly demonstrating impacts in rural regions. Owing to its rural characteristics, the data points available to evaluate clusters 2, 3 and 4 are very low - 8, 6 and 1 grid cells of 1km X 1km each. Hence, results for these clusters are not discussed in this study. Despite its rural characteristics, the region has seen

some development (mostly <20%), which highlights the negative impact of overall intensification in the bundle ($p < 0.05$). It also has a very slight positive impact on the quantity of drinking water. Moreover, a positive impact on green spaces and water quality is seen in cluster 1. A statistically significant negative impact on crop yield further reiterates the loss of agricultural land to land use conversions. Finally, a small but contrasting result for the impact on freshwater quality is observed in this bundle, with a positive impact of overall intensification, but a negative impact of intensification in cluster 1.

Chapter 7

Discussion and Implications

7.1 Urban Growth and Ecological Trade-Offs at the Regional-Scale (2013 - 2023)

This chapter examines the ecological trade-offs among provisioning, regulating, and cultural services across the Regional Municipality of Waterloo between 2013 and 2023. While an aggregated perspective of UES shows improving provisional services and degrading regulating and cultural services, the local trends vary substantially. More specifically, the findings reveal that the regional changes in ecosystem service supply such as rising food production, declining water quality, increasing air temperatures, and loss of green space are spatially uneven and context-dependent.

Food production increased overall, but this rise was not spatially uniform. Most grid cells experienced either a decline or only marginal gains in crop yield and farmland area. This suggests that technological advancements and improved farming efficiency may have compensated for a limited spatial expansion of agricultural land. Decline in food production was more evident at the urban fringe, likely due to land fragmentation, and soil degradation associated with encroaching development.

Water-related services witnessed contrasting trends. Groundwater levels (measuring freshwater supply) declined across the region, particularly in the urban centres, reflecting growing demand and diminished recharge. This is likely influenced by the growing urban population and increasing impervious surfaces altering land cover. Water quality also showed a general decline, with sharp degradation in urban cores and isolated rural areas. However, this could be potentially skewed by outlier contamination in sparsely monitored areas. Notably, the well with the highest recorded decline of water-quality is located within a designated *protected* wellhead area under regional policy, raising concerns about the effectiveness of existing safeguards. The absence of data on underground-aquifer limits precise interpretation of the spatial variation in water quality across the region. Furthermore, water quality trends often reflect long-term cumulative pressures, suggesting current declines may also stem from historical land and water use decisions.

The rapidly increasing impervious surfaces also impact the stormwater runoff management. The runoff conditions deteriorated in urban and transitioning peripheral areas, corresponding with increased hardscaping, revealing a clear link between land cover change and hydrological regulation.

However, stormwater runoff shows improvement in rural zones, *potentially* due to reduced rainfall. A further examination of rainfall data, the role of engineered drainage systems, and topological dimensions such as land slopes can lead to a more nuanced understanding of the varying drivers of changes in stormwater runoff.

Air temperature showed an overall warming across the region, with pronounced increases in southern Kitchener and Cambridge, areas with increased built area and vegetation loss. Although the changes in air-temperatures were based on sparse meteorological data, this trend still aligns with the expected urban heat island effect. Rising temperatures also interacted with other services: for instance, summer 2023 witnessed a sharp seasonal drop in air quality health index in Kitchener, possibly due to elevated ozone levels, lower wind flow, and intensified anthropogenic activities.

The most substantial decline was observed in green space coverage, particularly in urban and peri-urban areas undergoing development. Beyond reducing access to recreation, the loss of vegetated areas undermines critical services such as stormwater regulation, temperature moderation, air purification, and groundwater recharge. This degradation underscores the foundational role of green infrastructure in maintaining urban ecological balance.

Overall, these findings suggests interdependencies among urban ecosystem services, where shifts in land cover and urban intensification produce cascading and often uneven ecological effects. Further, these spatially heterogeneous local impacts are not bound by the administrative urban or rural boundaries. The impacts of urbanisation often influence services beyond the city limits. Hence, for an integrated and spatially explicit evaluation of the impacts on urban ecosystem services, the next section analyses the patterns of UES across the regions' urban fabric using ecosystem service bundles.

7.2 Trade-off and Synergies through UES Bundles

The patterns of ecosystem service bundles in this study highlight two key aspects of urban ecological change. First, they show how different services—like food production, climate regulation, and recreation—interact with each other. Some services improve together (synergies), while others come at the cost of another (trade-offs). Looking at these interactions helps us understand the full picture, rather than focusing on one service at a time. Second, the bundles suggest that the physical

characteristics of urban areas—such as how densely built they are, how land is used, and how green and built spaces are arranged—strongly influence how well ecosystems function. These effects are not limited to the city itself. Changes in urban areas often affect nearby rural environments as well. Taken together, these findings offer a clear, spatial way to understand how urbanisation produces different ecological outcomes across the region. They also help guide local planning in ways that respond to each area’s unique conditions.

The study area demonstrates four urban ecosystem services bundles, which show distinct variations not only in the ecological trade-offs and synergies but also in the dominant land use and urban intensification patterns. As stated in Table 10, Bundles 1 and 2 include a combination of both urban and rural spaces. In contrast, bundles 3 and 4 are mostly made up of just one type of land use—peri-urban and rural areas, respectively. The subsections below highlight key trends in ecological transformations corresponding to the variations in land cover, dominant urban typologies and intensification strategies observed in the study area.

Table 10: Bundles and land-use

Bundles	Dominant land-use
1	Mix - predominantly suburban
2	Mix - predominantly high-rise compact
3	Peri-urban
4	Rural

7.2.1 Rural versus Urban (Bundle 4 vs. Bundles 1, 2 and 3)

The spatial distribution of ecosystem services across the Region reveals clear differences between urban and rural areas, supporting the idea that urbanisation reshapes ecological functions. While Bundles 1, 2, and 3 represent varying levels of urban development, Bundle 4—made up mostly of rural areas—exhibits distinct and more stable ecological dynamics. Overall, although the urban and peri-urban landscapes exhibit higher stress and vulnerability among the urban ecosystem service

bundles, the trends and severity of the impact vary with changing urban forms and spatial context. The rural bundle on the other hand demonstrates resilience.

Bundle 1 predominantly showcases the suburban style of development, and intensification shows strong inter-correlations between air temperature, water quality and green spaces, suggesting strong interdependencies among regulating services, where higher land and resource consumption coupled with fragmented green cover disrupts the existing balance of urban ecosystem services.

Bundle 2, dominated by high-rise compact developments of intensification in urban areas, shows a moderate decline in the services. However, *contrasting trends in rural and urban* regions of this bundle are observed. While stormwater runoff management and crop yield drastically decline in urban and peri-urban regions, rural areas demonstrate more resilience and improvement in these services.

Finally, bundle 3, dominated by peri-urban regions and suburban urban typologies and intensification, face the highest decline in green spaces, coupled with a moderate rise in temperature and a higher decline in regulating services. Although statistically significant correlations between services are limited, the observed decline in green space shows a strong association with patterns of urban intensification, indicating potential trajectories of continued ecosystem service degradation (see Table 6 in Chapter 6.6).

In contrast, Bundle 4, dominated by rural land uses, shows strong ecological performance. Positive correlations between water quality, stormwater management, and temperature regulation suggest that regulating services in these areas function well together, likely supported by intact rural landscapes. However, early warning signs are present. A moderate decline in green space and small drops in freshwater supply, despite rising crop yields, point to emerging trade-offs. These are supported by negative correlations between crop production, freshwater levels, and stormwater performance, possibly indicating rising agricultural intensity or water stress. Still, the overall trend in Bundle 4 remains positive, underscoring the importance of rural areas in sustaining multiple ecosystem services under low urban pressure.

7.2.2 Compact High-Rise versus Suburban Sprawl (Bundle 2 vs. Bundle 1)

A key difference between Bundle 1 and Bundle 2 lies not only in the degree of ecosystem service decline but also in the type and spatial pattern of urban development driving those changes. Bundle 1 is concentrated in peri-urban areas of Southern Kitchener, the historic core of Cambridge (Galt), and nearby rural zones. These areas have seen predominantly low-rise, low-density expansion between 2013 and 2023, consistent with suburban sprawl and horizontal land conversion. By contrast, Bundle 2 reflects compact, high-rise intensification along major transit corridors and within designated growth centres, primarily in the urban cores of Kitchener and Waterloo. Although intensification targets are consistent across the Region, implementation varies. Cambridge exhibits a slower rate of densification relative to Waterloo and Kitchener.

Even with similar levels of green space loss, suburban sprawl and fragmented growth in Bundle 1 lead to disproportionately greater ecological degradation—highlighting that the form of urban development can be as consequential as its extent. Both bundles experienced relatively similar green space loss (38% in Bundle 1 and 28% in Bundle 2, see Figure 40) but

Bundle 1 recorded significantly higher increases in air temperature, a sharper decline in groundwater quality, and reduced crop yield ($p < 0.01$; see Table 5 in Chapter 6.3). These impacts cannot be explained by green space loss alone. Furthermore, a portion of the green spaces impacted are designated as environmentally sensitive areas by the regional policies (ROP, 2012). A reduction in vulnerable green spaces, even if moderate, could trigger severe degradation of regulating services. Furthermore, rapid horizontal urban expansion in the vicinity of environmentally sensitive areas could accelerate the process of degradation, demonstrating the influence of urban pressures beyond urban boundaries.

Bundle 2 (Transitional) currently demonstrates more resilient ecological outcomes than Bundle 1, despite having a lower share of green cover. It shows smaller increases in air temperature and milder

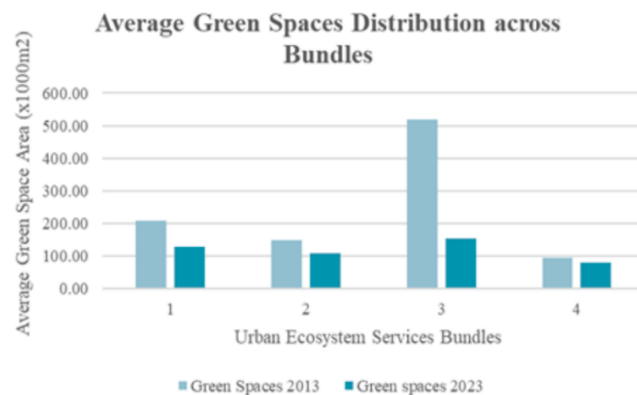


Figure 38: Change in green spaces in each bundle between 2013 and 2023

declines in water quality and crop yield. This suggests that compact, high-rise development—when well-managed—can mitigate some negative ecological impacts. However, this resilience is not guaranteed. As impervious surface area expands with continued densification, key ecosystem services come under pressure. Statistical analysis confirms strong correlations between increased impervious cover and rising stormwater runoff, reduced water availability, and elevated temperatures ($p < 0.01$; see Tables 5 and 6.3). These effects are most pronounced in dense urban zones where impermeable surfaces exceed 50% of land cover.

Notably, ecological trends within Bundle 2 vary by context. Rural segments maintain higher crop yields and experience lower runoff, underscoring the role of spatial layout, land-use design, and landscape buffers in shaping outcomes. While Bundle 2 still performs better than ecologically stressed areas, its growing exposure to land cover change signals future vulnerabilities—especially for regulating services like temperature control and water retention. As the region undergoes further population growth and vertical intensification, these risks are likely to intensify. This calls for proactive urban planning strategies that integrate green infrastructure, manage surface runoff, and maintain ecological buffers to sustain service provision.

7.2.3 Green Spaces in Sub-urban vs Peri-urban (Bundle 1 vs Bundle 3)

In peri-urban and suburban landscapes, the ecological consequences of green space loss are shaped less by the quantity lost and more by baseline conditions and spatial context—underscoring the disproportionate vulnerability of already fragmented ecosystems. This is evident in the comparison between Bundle 1 (suburban) and Bundle 3 (Peri-urban), both of which share low-rise suburban characteristics and similar patterns of intensification between 2013 and 2023. Yet, Green space loss in suburban bundles triggers disproportionately severe declines in ecosystem services—particularly regulating functions—where the baseline vegetation is already low and spatial fragmentation is high.

Bundle 3 has undergone the steepest decline in green spaces, losing approximately 70% of its vegetated cover, in comparison to a moderate loss of approximately 38% in bundle 1. Despite this loss, bundle 3 still maintains 20% more green spaces than bundle 1, largely due to its significantly higher baseline - over 150% more green cover than bundle 1 in 2013 (see figure 40). Yet, this apparent advantage does not translate into better ecosystem service outcomes. While air temperature increases were indeed moderate, water-related services—especially freshwater quantity, quality, and

stormwater regulations - saw a significant decline. While the stabilised rise in air temperature can be attributed to the declining impacts of urban heat island effect in regions farther away from the urban core, the decline in stormwater management, water quality and quantity reinforces the abrupt increase in impervious surfaces, reduced infiltration and fragmented green cover in the peri-urban regions. A strong negative impact of rising urban intensification on green spaces across the bundle 3 further reinstates a widespread conversion of green cover into built surfaces ($p < 0.01$, refer to Table 5 in Chapter 6.3).

In contrast, bundle 1, though it experiences a less dramatic fall of green cover, displays more acute ecosystem degradation, particularly in air temperature and water quality. This bundle has recorded the highest rise in air temperatures and the sharpest decline in groundwater quality, despite only a moderate reduction in green spaces. The strong correlations among green space availability, air temperature and water quality ($p < 0.01$, refer to Table 5 in Chapter 6.3) further highlight these interlinked service systems. However, the outcomes are not solely due to the change in quantity of green spaces, but the environmental vulnerability and low baseline stock of green spaces. A moderate additional loss in an already low green cover has tipped the balance towards ecological stress.

These contrasting trends challenge the assumption that maintaining high green cover alone is enough. Ecosystem vulnerability depends not only on how much green is lost, but also where it is lost, what baseline it started from, and what spatial context it occurs in. In already fragile peri-urban areas, even small losses can trigger larger ecological shifts. This underlines the need for region-specific planning strategies that consider baseline vegetation, landscape configuration, and functional linkages across services. While this study focuses on green space quantity, future research should also account for other ecological elements, such as urban tree cover and green infrastructure that influence service outcomes at finer spatial scales.

7.3 Implications

The findings of this study highlight the complex relationship between urban form, intensification patterns, and baseline ecological conditions in shaping the production of urban ecosystem services (UESs) within the Region of Waterloo. They indicate that while compact urban growth offers clear regional-scale benefits, its localised effects vary depending on the ecological capacity of the area in which intensification occurs. This underscores the need for a planning approach that is both scale-sensitive and responsive to site-specific conditions.

At the regional scale, compact urban form exerts a lower overall impact on UESs than low-density sprawl. By containing the urban footprint, this form of growth limits the loss of agricultural land, woodlands, wetlands, and other critical landscapes that contribute to regional biodiversity, climate regulation, and cultural value. This outcome directly supports the Region's Growth Management Strategy and the Regional Official Plan (Region of Waterloo, 2015), which aims to preserve the countryside line and protect rural lands from incremental encroachment.

However, when examined at the local scale (i.e., at a 1 km x 1 km grid scale), compact forms concentrate environmental pressures in already developed areas. Increased impervious surfaces, reduced green spaces, and rising urban heat island effect can erode local ecosystem functions, such as temperature moderation, runoff regulation, and provision of accessible green spaces. Without targeted mitigation, these changes risk diminishing everyday environmental quality in dense urban settings. This suggests that the benefits of compact growth at the regional level can only be fully realised if local environmental trade-offs are actively managed.

A further insight emerges from the analysis of where intensification takes place. High-density urban environments—often characterised by limited remaining permeable surfaces and fragmented vegetation—show a greater susceptibility to UES loss when accommodating additional growth. In such contexts, ecological offset measures are critical. These might include expanding tree canopy coverage, integrating green roofs, restoring riparian corridors, or increasing the quantity and quality of public green space. By contrast, medium- and low-density areas with greater existing green infrastructure and underutilised land can often accommodate new development with fewer ecological costs. Well-designed infill and redevelopment projects in these areas, particularly along or adjacent to the transit access, can yield more balanced outcomes by pairing additional housing with green infrastructure from the outset.

The study also reveals the disproportionate ecological consequences of losing small but strategically important natural areas in zones with low baseline green space. These can include isolated woodlots, small wetlands, or narrow riparian buffers embedded within the urban fabric. Although individually modest, such patches provide vital regulating and habitat functions, especially in built-up areas with otherwise limited ecological resources. Existing planning policies in the Region tend to prioritise larger, contiguous natural areas, leaving smaller fragments more vulnerable to redevelopment or incremental loss (Region of Waterloo, 2015).

Together, these findings support a differentiated approach to intensification. The evidence suggests that urban intensification should be considered not only in terms of density targets but also ecological carrying capacity. Directing growth to areas with greater green infrastructure resilience, moderating development intensity in ecologically sensitive cores, and embedding UES considerations into planning legislation can reduce trade-offs while still meeting the Region's growth objectives. The 1 km × 1 km grid-based UES mapping framework developed for this study offers a practical tool for operationalising such an approach. It enables planners to spatially assess ecological conditions, identify priority zones for low-impact intensification, and monitor changes over time. Embedding this tool into regular planning processes—such as the Region's Official Plan or local municipal plans—would facilitate adaptive management in line with observed ecological trends.

This reframing of intensification as an ecologically responsive process aligns with the Region's policy direction while addressing the nuanced realities of local environmental performance. It moves away from the perception of density as a uniform planning objective and towards “sensitised intensification”—growth that is compact enough to protect regional landscapes, yet adaptive enough to sustain local ecosystem functions. Such an approach recognises that urban growth and environmental stewardship are not opposing goals; rather, they can be coordinated to produce resilient, liveable, and ecologically functional urban environments.

In practice, sensitised intensification calls for treating every patch of green—whether a large woodland or a modest urban verge—as a planning asset. It requires incorporating green infrastructure early in the design process rather than as an afterthought, and viewing ecological thresholds as guiding parameters for where and how growth should occur. The outcome is an urban form that meets housing and economic needs while also retaining the natural systems that moderate climate, support biodiversity, and enhance quality of life for residents.

As the Region of Waterloo continues to navigate growth pressures, the challenge lies in balancing its well-established commitment to compact development with an equally robust commitment to maintaining and enhancing UESs. By directing growth towards areas with higher ecological capacity, safeguarding vulnerable green spaces, and institutionalising ecological performance monitoring, the Region can continue to model a growth strategy that is not only spatially efficient but also ecologically sustainable.

Chapter 8

Conclusion

The analysis of urban ecosystem service patterns across the Region of Waterloo between 2013 and 2023 reveals that urbanisation induces complex, uneven, and scale-dependent ecological outcomes. While regional trends suggest an increase in provisioning services and a decline in regulating and cultural services, these generalisations mask substantial local variation shaped by land use, urban form, and spatial context. The ecosystem service bundles derived through spatial clustering offer critical insight into these patterns, revealing how ecological trade-offs and synergies emerge from specific combinations of urban typology, land cover, and development strategy.

A key finding of this study is the differentiated ecological response to varying forms of urban intensification. Regression analysis confirms that added built volume in already dense urban clusters has a more pronounced and concentrated impact on changes in UESs, particularly on regulating services such as air temperature, stormwater runoff, and water quality. In contrast, more dispersed or low-rise intensification shows diffused and widespread ecological impacts. This reinforces the importance of considering not just the amount of urban growth, but the spatial structure and density of development in understanding ecological outcomes.

Moreover, while green cover remains central to urban ecosystem health, its influence is highly context-sensitive. The quantity of green space alone is not sufficient; its spatial distribution, baseline availability, and ecological vulnerability strongly mediate its functional role. For example, even modest losses in already fragmented peri-urban green landscapes can trigger cascading service declines, whereas areas with higher baseline vegetation may exhibit delayed or masked impacts. These findings complicate assumptions that greater green cover necessarily ensures greater ecological stability.

Rural landscapes continue to support multifunctional services under relatively lower development pressure. However, emerging trade-offs—such as those between agricultural productivity and water-related services—indicate that even these areas are not immune to growing global ecological stress due to climate change. This underscores the importance of monitoring and managing non-urban areas alongside rapidly urbanising cores.

In conclusion, the findings point to the need for a spatially differentiated, systems-based approach to urban planning. Strategies for sustainable intensification must account for the form, density, and distribution of development rather than focusing solely on growth targets. Planning must also prioritise the quality and configuration of green infrastructure to buffer ecological stress. As urban regions continue to densify, a nuanced understanding of ecosystem service interdependencies will be essential for building long-term ecological resilience.

8.1 Limitations and Future Research

While this study offers valuable insights into the impacts of urban intensification on urban ecosystem services (UES) in the Region of Waterloo, it is essential to acknowledge several methodological and analytical limitations that may influence the interpretation of the results. These relate to data availability, spatial analysis techniques, and the theoretical framing of urban–ecological relationships.

A primary limitation lies in the temporal scope of the research. The analysis compares only two points in time—2013 and 2023—providing a decade-long snapshot of change. While this timeframe is adequate to identify notable shifts in UES provision, it cannot capture short-term fluctuations or longer-term trends. For instance, climatic anomalies, short-lived policy interventions, or sudden economic changes within the decade may have influenced ecosystem services in ways not accounted for in this study. A more continuous or multi-point temporal dataset could offer richer insights into the pace and nature of changes.

The research also depends heavily on available secondary datasets, including land use classifications, pollutant concentrations, and hydrological models. While these datasets are robust and widely used, they are subject to inherent uncertainties in measurement, classification, and spatial resolution. For example, the SOLRIS land cover dataset and municipal land use inventories may contain classification errors or temporal mismatches, while pollutant monitoring relies on a limited network of weather and air quality stations. In regions of sparse data coverage, interpolations may smooth over localised environmental variations.

Some parameters—such as air temperature—were derived from sparse and spatially clustered weather station data. Although kriging is generally preferred for its ability to incorporate spatial autocorrelation, its application in this study produced unreliable outputs. The Inverse Distance Weighting (IDW) method was therefore applied, as it generated spatial patterns that were more visually consistent with the intuitive impacts of the urban heat island effect. However, IDW assumes

that spatial influence reduces uniformly with distance and does not explicitly model spatial variability, which may lead to smoothing effects or the underrepresentation of localised extremes. This trade-off reflects a methodological compromise between statistical robustness and contextual plausibility.

Closely linked to data constraints is the simplification of ecosystem service indicators. In order to conduct a region-wide, multi-service assessment, each UES was represented through a set of measurable proxies—such as pollutant concentrations for air quality. While these proxies are scientifically defensible, they do not fully capture the complexity or multi-functionality of ecosystem services. For example, cultural services such as recreational value or aesthetic quality were not directly measured, yet they are often deeply affected by urban intensification and contribute significantly to community well-being.

The study also employed a Self-Organising Map (SOM) for clustering UES bundles. SOMs are effective in reducing multidimensional complexity and identifying non-linear relationships between variables, making them suitable for an integrated UES assessment. However, SOM outputs are sensitive to parameter settings, input normalisation, and the selection of training iterations. The method's reliance on unsupervised learning means that cluster interpretations require a degree of subjectivity, introducing researcher bias. While SOM was chosen for its ability to reveal emergent patterns without presupposing class boundaries, the resulting UES bundles should be viewed as analytical constructs rather than definitive typologies.

From a theoretical perspective, the study builds on the premise that urban intensification has measurable effects on UES at multiple scales. While this framing is supported by existing literature, it necessarily simplifies complex socio-ecological dynamics. UES are influenced not only by land use and spatial form but also by governance, cultural values, and temporal variability in environmental processes—factors not explicitly modelled here. Consequently, the findings should be interpreted as indicative patterns rather than deterministic causal relationships.

Spatial summarisation was conducted using a 1 km × 1 km grid. This resolution provided a consistent spatial framework for integrating diverse datasets and aligning with the scale of regional policy analysis. However, grid-based aggregation inevitably generalises local variability, particularly in heterogeneous zones with multiple urban forms. This may obscure fine-grained ecosystem service dynamics or overemphasise changes in areas where processes occur at smaller scales. Alternative

zoning units, such as administrative boundaries or land parcels, could capture some spatial nuances but would limit comparability across the study area.

In summary, while the chosen tools— such as SOM, IDW, and grid analysis—were well suited to the objectives and data realities of this research, each carries inherent limitations that shape the findings. The results should therefore be interpreted as indicative patterns within a broader conceptual and methodological framework, acknowledging that future research with denser datasets, alternative spatial units, and refined modelling approaches could yield more granular insights.

Despite these limitations, the study provides a strong foundation for integrating ecosystem service considerations into growth management strategies. Recognising these constraints not only clarifies the boundaries of the findings but also points to opportunities for future research—such as incorporating more temporal data points, exploring intermediate scales, refining ecosystem service indicators, and integrating social-ecological dimensions of planning. By doing so, subsequent work can build on this study’s insights and offer even more nuanced guidance for urban intensification policies that safeguard ecological and community well-being.

References

- Agriculture and Agri-Food Canada. (2023). *Annual crop inventory: Ontario (2013–2023)* [ACI]. Government of Canada. <https://open.canada.ca/data/en/dataset/7b5e5e34-2d4a-4f19-8283-7b9bd94d3573>
- Alavipanah, S., Haase, D., Lakes, T., & Qureshi, S. (2017). Integrating the third dimension into the concept of urban ecosystem services: A review. *ECOLOGICAL INDICATORS*, 72, 374–398. <https://doi.org/10.1016/j.ecolind.2016.08.010>
- Alberti, M. (2005). The Effects of Urban Patterns on Ecosystem Function. *International Regional Science Review*, 28(2), 168–192. <https://doi.org/10.1177/0160017605275160>
- Alvarado, O. A. (2020). A review of quantitative tools for measuring multiple urban ecosystem services. *Sustainable Mediterranean Construction*, 2020(4 Special Issue), 31–35. Scopus.
- Andrade, J. F., Cassman, K. G., Rattalino Edreira, J. I., Agus, F., Bala, A., Deng, N., & Grassini, P. (2022a). Impact of urbanization trends on production of key staple crops. *Ambio*, 51(5), 1158–1167. <https://doi.org/10.1007/s13280-021-01674-z>
- Andrade, J. F., Cassman, K. G., Rattalino Edreira, J. I., Agus, F., Bala, A., Deng, N., & Grassini, P. (2022b). Impact of urbanization trends on production of key staple crops. *Ambio*, 51(5), 1158–1167. <https://doi.org/10.1007/s13280-021-01674-z>
- Bennett, E. M., Peterson, G. D., & Gordon, L. J. (2009). Understanding relationships among multiple ecosystem services. *Ecology Letters*, 12(12), 1394–1404. <https://doi.org/10.1111/j.1461-0248.2009.01387.x>
- Berghauser Pont, M., Barthel, S., Colding, J., Gren, Å., Legeby, A., & Marcus, L. (2022). Editorial: Social-ecological urbanism: Developing discourse, institutions and urban form for the design of resilient social-ecological systems in cities. *Frontiers in Built Environment*, 8. <https://doi.org/10.3389/fbuil.2022.982681>
- Boehnke, D., Krehl, A., Mörmann, K., Volk, R., Lützkendorf, T., Naber, E., Becker, R., & Norra, S. (2022). Mapping Urban Green and Its Ecosystem Services at Microscale—A Methodological Approach for Climate Adaptation and Biodiversity. *Sustainability*, 14(15), Article 15. <https://doi.org/10.3390/su14159029>

- Bolund, P., & Hunhammar, S. (1999). Ecosystem services in urban areas. *Ecological Economics*, 29(2), 293–301. [https://doi.org/10.1016/S0921-8009\(99\)00013-0](https://doi.org/10.1016/S0921-8009(99)00013-0)
- Boutreux, T., Bourgeois, M., Bellec, A., Commeaux, F., & Kaufmann, B. (2024). Addressing the sustainable urbanism paradox: Tipping points for the operational reconciliation of dense and green morphologies. *Npj Urban Sustainability*, 4(1), 1–12. <https://doi.org/10.1038/s42949-024-00176-7>
- Bren d'Amour, C., Reitsma, F., Baiocchi, G., Barthel, S., Güneralp, B., Erb, K.-H., Haberl, H., Creutzig, F., & Seto, K. C. (2017). Future urban land expansion and implications for global croplands. *Proceedings of the National Academy of Sciences*, 114(34), 8939–8944. <https://doi.org/10.1073/pnas.1606036114>
- Brzoska, P., & Späße, A. (2020). From City- to Site-Dimension: Assessing the Urban Ecosystem Services of Different Types of Green Infrastructure. *Land*, 9(5), Article 5. <https://doi.org/10.3390/land9050150>
- Caldwell, W., Epp, S., Wan, X., Singer, R., Drake, E., & Sousa, E. C. (2022). Farmland Preservation and Urban Expansion: Case Study of Southern Ontario, Canada. *Frontiers in Sustainable Food Systems*, 6. <https://doi.org/10.3389/fsufs.2022.777816>
- Campagne, C. S., & Roche, P. (2018). May the matrix be with you! Guidelines for the application of expert-based matrix approach for ecosystem services assessment and mapping. *One Ecosystem*, 3, e24134. <https://doi.org/10.3897/oneeco.3.e24134>
- Cao, C., & Wang, J. (2025). The impact of urban expansion on food production: A bibliometric study of development, hotspots, and future prospects. *Frontiers in Sustainable Food Systems*, 9. <https://doi.org/10.3389/fsufs.2025.1550373>
- Caprioli, C., Bottero, M., & Mondini, G. (2020). Urban Ecosystem Services: A Review of Definitions and Classifications for the Identification of Future Research Perspectives. *Lecture Notes in Computer Science (Including Subseries Lecture Notes in Artificial Intelligence and Lecture Notes in Bioinformatics)*, 12253 LNCS, 332–344. Scopus. https://doi.org/10.1007/978-3-030-58814-4_23
- Chang, R., Wang, S., Fan, J., & Feng, P. (2024). Investigating the impacts of multi-dimensional urbanization processes on ecosystem services in mega-urban agglomerations at the grid scale.

International Journal of Environmental Science and Technology. <https://doi.org/10.1007/s13762-024-06063-z>

- Chen, T.-H. K., Qiu, C., Schmitt, M., Zhu, X. X., Sabel, C. E., & Prishchepov, A. V. (2020). Mapping horizontal and vertical urban densification in Denmark with Landsat time-series from 1985 to 2018: A semantic segmentation solution. *Remote Sensing of Environment*, 251, 112096. <https://doi.org/10.1016/j.rse.2020.112096>
- Chen, W. Y. (2017). Urban Nature and Urban Ecosystem Services. *Advances in 21st Century Human Settlements*, 181–199. Scopus. https://doi.org/10.1007/978-981-10-4113-6_9
- Cheng, X., Wei, B., Chen, G., Li, J., & Song, C. (2015). Influence of Park Size and Its Surrounding Urban Landscape Patterns on the Park Cooling Effect. *Journal of Urban Planning and Development*, 141(3), A4014002. [https://doi.org/10.1061/\(ASCE\)UP.1943-5444.0000256](https://doi.org/10.1061/(ASCE)UP.1943-5444.0000256)
- Colter, K. R., Middel, A. C., & Martin, C. A. (2019). Effects of natural and artificial shade on human thermal comfort in residential neighborhood parks of Phoenix, Arizona, USA. *Urban Forestry & Urban Greening*, 44, 126429. <https://doi.org/10.1016/j.ufug.2019.126429>
- Cortinovis, C., & Geneletti, D. (2018). Ecosystem services in urban plans: What is there, and what is still needed for better decisions. *Land Use Policy*, 70, 298–312. Scopus. <https://doi.org/10.1016/j.landusepol.2017.10.017>
- Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R. V., Paruelo, J., Raskin, R. G., Sutton, P., & van den Belt, M. (1997). The value of the world's ecosystem services and natural capital. *Nature*, 387(6630), 253–260. <https://doi.org/10.1038/387253a0>
- Croci, E., Lucchitta, B., & Penati, T. (2021). Valuing Ecosystem Services at the Urban Level: A Critical Review. *Sustainability*, 13(3), Article 3. <https://doi.org/10.3390/su13031129>
- Dadi, W., Mulugeta, M., & Semie, N. (2023). Impact of urbanization on the welfare of farm households: Evidence from Adama Rural District in Oromia regional state, Ethiopia. *Heliyon*, 10(1), e23802. <https://doi.org/10.1016/j.heliyon.2023.e23802>
- Das, M., & Das, A. (2019). Dynamics of Urbanization and its impact on Urban Ecosystem Services (UESs): A study of a medium size town of West Bengal, Eastern India. *Journal of Urban Management*, 8(3), 420–434. <https://doi.org/10.1016/j.jum.2019.03.002>

- de Groot, R. S., Alkemade, R., Braat, L., Hein, L., & Willemen, L. (2010). Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecological Complexity*, 7(3), 260–272. <https://doi.org/10.1016/j.ecocom.2009.10.006>
- Derkzen, M. L., van Teeffelen, A. J. A., & Verburg, P. H. (2015). REVIEW: Quantifying urban ecosystem services based on high-resolution data of urban green space: an assessment for Rotterdam, the Netherlands. *Journal of Applied Ecology*, 52(4), 1020–1032. <https://doi.org/10.1111/1365-2664.12469>
- Dey, S., Niyogi, J. G., & Das, D. (2024). Assessing the Role of Urban Form in Ecosystem Service Provisioning as a Part of Urban Resilience: A Case Study on Kolkata. In S. Ghosh, S. Majumdar, & A. Cheshmehzangi (Eds.), *Cities of Tomorrow: Urban Resilience and Climate Change Preparedness* (pp. 87–105). Springer Nature. https://doi.org/10.1007/978-981-97-9658-8_5
- Díaz, S., Demissew, S., Carabias, J., Joly, C., Lonsdale, M., Ash, N., Larigauderie, A., Adhikari, J. R., Arico, S., Báldi, A., Bartuska, A., Baste, I. A., Bilgin, A., Brondizio, E., Chan, K. M., Figueroa, V. E., Duraipah, A., Fischer, M., Hill, R., ... Zlatanova, D. (2015). The IPBES Conceptual Framework—Connecting nature and people. *Current Opinion in Environmental Sustainability*, 14, 1–16. <https://doi.org/10.1016/j.cosust.2014.11.002>
- Ding, Q., Wang, Y., & Zhuang, D. (2018a). Comparison of the common spatial interpolation methods used to analyze potentially toxic elements surrounding mining regions. *Journal of Environmental Management*, 212, 23–31. <https://doi.org/10.1016/j.jenvman.2018.01.074>
- Ding, Q., Wang, Y., & Zhuang, D. (2018b). Comparison of the common spatial interpolation methods used to analyze potentially toxic elements surrounding mining regions. *Journal of Environmental Management*, 212, 23–31. <https://doi.org/10.1016/j.jenvman.2018.01.074>
- Do Sustainable Urban Designs Generate More Ecosystem Services? A Case Study of Civano in Tucson, Arizona. (2015). *The Professional Geographer*, 67(2), 204–217. <https://doi.org/10.1080/00330124.2014.922021>
- Dobbs, C., Escobedo, F. J., & Zipperer, W. C. (2011). A framework for developing urban forest ecosystem services and goods indicators. *Landscape and Urban Planning* 99:196-206, 99, 196–206. <https://research.fs.usda.gov/treearch/39207>

- Dowdell, J. (2020, December 4). Ecosystems as a Framework for Urban Planning: Reconnecting. Rediscovering. Reinvigorating. *The Nature of Cities*.
<https://www.thenatureofcities.com/2020/12/04/ecosystems-as-a-framework-for-urban-planning-reconnecting-rediscovering-reinvigorating/>
- Duan, Y., Agrawal, S., Sanchez-Azofeifa, A., & Welegedara, N. (2024). *Urban Heat Island Effect in Canada: Insights from Five Major Cities* (SSRN Scholarly Paper 4965331). Social Science Research Network. <https://doi.org/10.2139/ssrn.4965331>
- Dushkova, D., Taherkhani, M., Konstantinova, A., Vasenev, V. I., & Dovletyarova, E. A. (2025). Understanding Factors Affecting the Use of Urban Parks Through the Lens of Ecosystem Services and Blue–Green Infrastructure: The Case of Gorky Park, Moscow, Russia. *Land*, 14(2), Article 2. <https://doi.org/10.3390/land14020237>
- Egerer, M., Fouch, N., Anderson, E. C., & Clarke, M. (2020). Socio-ecological connectivity differs in magnitude and direction across urban landscapes. *Scientific Reports*, 10(1), 4252. <https://doi.org/10.1038/s41598-020-61230-9>
- Eigenbrod, F., Bell, V. A., Davies, H. N., Heinemeyer, A., Armsworth, P. R., & Gaston, K. J. (2011). The impact of projected increases in urbanization on ecosystem services. *Proceedings of the Royal Society B: Biological Sciences*, 278(1722), 3201–3208. <https://doi.org/10.1098/rspb.2010.2754>
- Elliot, T., Babí Almenar, J., Niza, S., Proença, V., & Rugani, B. (2019). Pathways to Modelling Ecosystem Services within an Urban Metabolism Framework. *Sustainability*, 11(10), 2766. <https://doi.org/10.3390/su11102766>
- Elmqvist, T., Fragkias, M., Goodness, J., Güneralp, B., Marcotullio, P. J., McDonald, R. I., Parnell, S., Schewenius, M., Sendstad, M., Seto, K. C., & Wilkinson, C. (Eds.). (2013). *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities*. Springer Netherlands. <https://doi.org/10.1007/978-94-007-7088-1>
- Environment and Climate Change Canada. (2023). *Historical climate data*. Government of Canada. <https://climate.weather.gc.ca/>
- Escobedo, F. J., Giannico, V., Jim, C. Y., Sanesi, G., & Laforteza, R. (2019). Urban forests, ecosystem services, green infrastructure and nature-based solutions: Nexus or evolving

- metaphors? *Urban Forestry & Urban Greening*, 37, 3–12.
<https://doi.org/10.1016/j.ufug.2018.02.011>
- Fan, Y., Zheng, Y., Jia, C., & Song, Y. (2024a). The impact of China's urbanization on ecosystem service value from the perspective of gross ecosystem product: A case study of Beijing-Tianjin-Hebei region. *Scientific Reports*, 14(1), 15954. <https://doi.org/10.1038/s41598-024-64655-8>
- Fan, Y., Zheng, Y., Jia, C., & Song, Y. (2024b). The impact of China's urbanization on ecosystem service value from the perspective of gross ecosystem product: A case study of Beijing-Tianjin-Hebei region. *Scientific Reports*, 14(1), 15954. <https://doi.org/10.1038/s41598-024-64655-8>
- Fang, X., Li, J., & Ma, Q. (2023). Integrating green infrastructure, ecosystem services and nature-based solutions for urban sustainability: A comprehensive literature review. *Sustainable Cities and Society*, 98, 104843. <https://doi.org/10.1016/j.scs.2023.104843>
- Feld, C. K., Martins da Silva, P., Paulo Sousa, J., De Bello, F., Bugter, R., Grandin, U., Hering, D., Lavorel, S., Mountford, O., & Pardo, I. (2009). Indicators of biodiversity and ecosystem services: A synthesis across ecosystems and spatial scales. *Oikos*, 118(12), 1862–1871.
- Filion, P., Bunting, T., Pavlic, D., & Langlois, P. (2010). Intensification and Sprawl: Residential Density Trajectories in Canada's Largest Metropolitan Regions. *Urban Geography*, 31(4), 541–569. <https://doi.org/10.2747/0272-3638.31.4.541>
- Fischer, L. K., Honold, J., Botzat, A., Brinkmeyer, D., Cvejić, R., Delshammar, T., Elands, B., Haase, D., Kabisch, N., Karle, S. J., Laforteza, R., Nastran, M., Nielsen, A. B., van der Jagt, A. P., Vierikko, K., & Kowarik, I. (2018). Recreational ecosystem services in European cities: Sociocultural and geographical contexts matter for park use. *Ecosystem Services*, 31, 455–467. <https://doi.org/10.1016/j.ecoser.2018.01.015>
- Folke, C., Jansson, Å., Larsson, J., & Costanza, R. (1997). Ecosystem Appropriation by Cities. *Ambio*, 26(3), 167–172. <https://www.jstor.org/stable/4314576>
- Food and Agriculture Organization, United Nations. (2023). *The State of Food Security and Nutrition in the World 2023*. FAO; IFAD; UNICEF; WFP; WHO; <https://doi.org/10.4060/cc3017en>
- Gao, X., Shen, J., He, W., Zhao, X., Li, Z., Hu, W., Wang, J., Ren, Y., & Zhang, X. (2021). Spatial-temporal analysis of ecosystem services value and research on ecological compensation in Taihu

- Lake Basin of Jiangsu Province in China from 2005 to 2018. *Journal of Cleaner Production*, 317, 128241. <https://doi.org/10.1016/j.jclepro.2021.128241>
- Gauthier, P., & Gilliland, J. (2006). Mapping Urban Morphology: A Classification Scheme for Interpreting Contributions to the Study of Urban Form. *Urban Morphology*, 10(1), 41–50.
- Ghosh, S., & Sk., J. (2024). Impact of changing urban form and production-living-ecological space on changing ecosystem services of a smart city of Eastern India, Durgapur. *Environmental Monitoring and Assessment*, 196(9), 804. <https://doi.org/10.1007/s10661-024-12953-z>
- Gómez-Baggethun, E., & Barton, D. N. (2013). Classifying and valuing ecosystem services for urban planning. *Ecological Economics*, 86, 235–245. <https://doi.org/10.1016/j.ecolecon.2012.08.019>
- Grabowski, Z. J., Matsler, A. M., Thiel, C., McPhillips, L., Hum, R., Bradshaw, A., Miller, T., & Redman, C. (2017). Infrastructures as Socio-Eco-Technical Systems: Five Considerations for Interdisciplinary Dialogue. *Journal of Infrastructure Systems*, 23(4), 02517002. [https://doi.org/10.1061/\(ASCE\)IS.1943-555X.0000383](https://doi.org/10.1061/(ASCE)IS.1943-555X.0000383)
- Grafius, D. R., Corstanje, R., & Harris, J. A. (2018). Linking ecosystem services, urban form and green space configuration using multivariate landscape metric analysis. *Landscape Ecology*, 33(4), 557–573. <https://doi.org/10.1007/s10980-018-0618-z>
- Grafius, D. R., Corstanje, R., Warren, P. H., Evans, K. L., Hancock, S., & Harris, J. A. (2016). The impact of land use/land cover scale on modelling urban ecosystem services. *Landscape Ecology*, 31(7), 1509–1522. <https://doi.org/10.1007/s10980-015-0337-7>
- Grêt-Regamey, A., Galleguillos-Torres, M., Dissegna, A., & Weibel, B. (2020). How urban densification influences ecosystem services—A comparison between a temperate and a tropical city. *Environmental Research Letters*, 15(7), 075001. <https://doi.org/10.1088/1748-9326/ab7acf>
- Grimm, N. B., Faeth, S. H., Golubiewski, N. E., Redman, C. L., Wu, J., Bai, X., & Briggs, J. M. (2008). Global Change and the Ecology of Cities. *Science*, 319(5864), 756–760. <https://doi.org/10.1126/science.1150195>
- Grimm, N. B., Grove, J. M., Pickett, S. T. A., & Redman, C. L. (2000). Integrated Approaches to Long-Term Studies of Urban Ecological Systems. *BioScience*, 50(7), 571–584. [https://doi.org/10.1641/0006-3568\(2000\)050\[0571:IATLTO\]2.0.CO;2](https://doi.org/10.1641/0006-3568(2000)050[0571:IATLTO]2.0.CO;2)

- Grove, J. M., Cadenasso, M. L., Pickett, S. T. A., Machlis, G. E., Burch, W. R., & Ogden, L. A. (2015). *The Baltimore School of Urban Ecology: Space, scale, and time for the study of cities*. Yale University Press.
- Gu, C., Shi, Q., & Tongji, C. (2024). Evaluating the driving factors of the environmental injustice caused by the spatial mismatch of ecosystem services in mega cities in China. *Urban Ecosystems*, 27(4), 1349–1365. <https://doi.org/10.1007/s11252-024-01518-0>
- Guo, Q., Tian, Y., Zhang, Y., & Wang, Y. (2025). Spatial heterogeneity of multi-scale trade-off synergies in ecosystem services. *PeerJ*, 13, e19507. <https://doi.org/10.7717/peerj.19507>
- Herreros-Cantis, P., & McPhearson, T. (2021). Mapping supply of and demand for ecosystem services to assess environmental justice in New York City. *Ecological Applications*, 31(6), e02390. <https://doi.org/10.1002/eap.2390>
- Hoffhine Wilson, E., Hurd, J. D., Civco, D. L., Prisloe, M. P., & Arnold, C. (2003). Development of a geospatial model to quantify, describe and map urban growth. *Remote Sensing of Environment*, 86(3), 275–285. [https://doi.org/10.1016/S0034-4257\(03\)00074-9](https://doi.org/10.1016/S0034-4257(03)00074-9)
- Holt, A. R., Mears, M., Maltby, L., & Warren, P. (2015). Understanding spatial patterns in the production of multiple urban ecosystem services. *Ecosystem Services*, 16, 33–46. <https://doi.org/10.1016/j.ecoser.2015.08.007>
- Hou, D., Al-Tabbaa, A., O'Connor, D., Hu, Q., Zhu, Y.-G., Wang, L., Kirkwood, N., Ok, Y. S., Tsang, D. C. W., Bolan, N. S., & Rinklebe, J. (2023). Sustainable remediation and redevelopment of brownfield sites. *Nature Reviews Earth & Environment*, 4(4), 271–286. <https://doi.org/10.1038/s43017-023-00404-1>
- Howard, K., & Gerber, R. (2018). Impacts of urban areas and urban growth on groundwater in the Great Lakes Basin of North America. *Journal of Great Lakes Research*, 44(1), 1–13. <https://doi.org/10.1016/j.jglr.2017.11.012>
- Hu, H., Fu, B., Lü, Y., & Zheng, Z. (2015). SAORES: A spatially explicit assessment and optimization tool for regional ecosystem services. *Landscape Ecology*, 30(3), 547–560. <https://doi.org/10.1007/s10980-014-0126-8>
- Hu, Y., Xu, X., Huang, X., Li, Y., Cao, J., Yan, Y., Hu, X., & Wu, S. (2025a). Urban Spatial Management and Planning Based on the Interactions Between Ecosystem Services: A Case Study

- of the Beijing–Tianjin–Hebei Urban Agglomeration. *Remote Sensing*, 17(7), Article 7.
<https://doi.org/10.3390/rs17071258>
- Hu, Y., Xu, X., Huang, X., Li, Y., Cao, J., Yan, Y., Hu, X., & Wu, S. (2025b). Urban Spatial Management and Planning Based on the Interactions Between Ecosystem Services: A Case Study of the Beijing–Tianjin–Hebei Urban Agglomeration. *Remote Sensing*, 17(7), Article 7.
<https://doi.org/10.3390/rs17071258>
- Husain, N. N. (2024, May 22). Singapore: The compact city championing urban biodiversity. *Institute of Sustainability Studies*. <https://instituteofsustainabilitystudies.com/insights/lexicon/singapore-the-compact-city-championing-urban-biodiversity/>
- Hwang, W. H., Wiseman, P. E., & Thomas, V. A. (2017). Enhancing the energy conservation benefits of shade trees in dense residential developments using an alternative tree placement strategy. *Landscape and Urban Planning*, 158, 62–74. <https://doi.org/10.1016/j.landurbplan.2016.09.022>
- IPBES. (2019). *Summary for policymakers of the global assessment report on biodiversity and ecosystem services*. Zenodo. <https://doi.org/10.5281/zenodo.3553579>
- Jia, K., Huang, A., Yin, X., Yang, J., Deng, L., & Lin, Z. (2023). Investigating the Impact of Urbanization on Water Ecosystem Services in the Dongjiang River Basin: A Spatial Analysis. *MDPI*. <https://www.mdpi.com/2072-4292/15/9/2265>
- Kang, W., Chon, J., & Kim, G. (2020). Urban Ecosystem Services: A Review of the Knowledge Components and Evolution in the 2010s. *Sustainability*, 12(23), Article 23.
<https://doi.org/10.3390/su12239839>
- Kim, H., Oh, K., & Lee, D. (2021). The Contribution of Ecosystem Regulating Services Based on Their Interrelationship in the Urban Ecosystem. *Applied Sciences*, 11(20), Article 20.
<https://doi.org/10.3390/app11209610>
- Kim, J., de Leeuw, E., Harris-Roxas, B., & Sainsbury, P. (2023). Five urban health research traditions: A meta-narrative review. *Social Science and Medicine*, 336. Scopus.
<https://doi.org/10.1016/j.socscimed.2023.116265>
- Kremer, P., Hamstead, Z., Haase, D., McPhearson, T., Frantzeskaki, N., Andersson, E., Kabisch, N., Larondelle, N., Rall, E. L., Voigt, A., Baró, F., Bertram, C., Gómez-Baggethun, E., Hansen, R., Kaczorowska, A., Kain, J.-H., Kronenberg, J., Langemeyer, J., Pauleit, S., ... Elmqvist, T. (2016).

- Key insights for the future of urban ecosystem services research. *Ecology and Society*, 21(2).
<https://www.jstor.org/stable/26270402>
- Kroll, F., Müller, F., Haase, D., & Fohrer, N. (2012). Rural–urban gradient analysis of ecosystem services supply and demand dynamics. *Land Use Policy*, 29(3), 521–535.
<https://doi.org/10.1016/j.landusepol.2011.07.008>
- Li, B., Chen, N., Wang, Y., & Wang, W. (2018). Spatio-temporal quantification of the trade-offs and synergies among ecosystem services based on grid-cells: A case study of Guanzhong Basin, NW China. *Ecological Indicators*, 94, 246–253. <https://doi.org/10.1016/j.ecolind.2018.06.069>
- Li, J., & Heap, A. D. (2011). A review of comparative studies of spatial interpolation methods in environmental sciences: Performance and impact factors. *Ecological Informatics*, 6(3), 228–241.
<https://doi.org/10.1016/j.ecoinf.2010.12.003>
- Li, J., Wang, J., & Zhou, W. (2024). Different impacts of urbanisation on ecosystem services supply and demand across old, new and non-urban areas in the ChangZhuTan urban agglomeration, China. *Landscape Ecology*, 39(6), 107. <https://doi.org/10.1007/s10980-024-01900-5>
- Lin, Y., Chen, X., Huang, L., Zhu, C., Shahtahmassebi, A., Zhang, J., Shen, S., Peng, R., Deng, J., Wang, K., & Gan, M. (2021). Fine-scale mapping of urban ecosystem service demand in a metropolitan context: A population-income-environmental perspective. *Science of The Total Environment*, 781, 146784. <https://doi.org/10.1016/j.scitotenv.2021.146784>
- Liu, L., & Wu, J. (2022). Scenario analysis in urban ecosystem services research: Progress, prospects, and implications for urban planning and management. *Landscape and Urban Planning*, 224, 104433. <https://doi.org/10.1016/j.landurbplan.2022.104433>
- Liu, S., Zhao, J., Chen, Y., & Zhang, S. (2024). Urban Morphology Classification and Organizational Patterns: A Multidimensional Numerical Analysis of Heping District, Shenyang City. *Buildings*, 14(10), Article 10. <https://doi.org/10.3390/buildings14103157>
- Livesley, S. J., McPherson, E. G., & Calfapietra, C. (2016). The Urban Forest and Ecosystem Services: Impacts on Urban Water, Heat, and Pollution Cycles at the Tree, Street, and City Scale. *Journal of Environmental Quality*, 45(1), 119–124. <https://doi.org/10.2134/jeq2015.11.0567>
- Luederitz, C., Brink, E., Gralla, F., Hermelingmeier, V., Meyer, M., Niven, L., Panzer, L., Partelow, S., Rau, A.-L., Sasaki, R., Abson, D. J., Lang, D. J., Wamsler, C., & von Wehrden, H. (2015). A

- review of urban ecosystem services: Six key challenges for future research. *Ecosystem Services*, 14, 98–112. <https://doi.org/10.1016/j.ecoser.2015.05.001>
- Lyu, R., Zhang, J., Xu, M., & Li, J. (2018). Impacts of urbanization on ecosystem services and their temporal relations: A case study in Northern Ningxia, China. *Land Use Policy*, 77, 163–173. <https://doi.org/10.1016/j.landusepol.2018.05.022>
- Ma, Y., & Yang, J. (2025). A review of methods for quantifying urban ecosystem services. *Landscape and Urban Planning*, 253, 105215. <https://doi.org/10.1016/j.landurbplan.2024.105215>
- Mack, E. A., Akamagwuna, F. C., Murata, C., Materechera-Mitochi, F., Nnadozie, C. F., & Odume, O. N. (2024). Perceptions and knowledge of ecosystem services in urban river systems, Eastern Cape, South Africa. *Urban Ecosystems*, 27(5), 1905–1914. <https://doi.org/10.1007/s11252-024-01562-w>
- Marques, A. L., Alvim, A. T. B., & Schröder, J. (2022). Ecosystem Services and Urban Planning: A Review of the Contribution of the Concept to Adaptation in Urban Areas. *Sustainability*, 14(4), Article 4. <https://doi.org/10.3390/su14042391>
- McDonald, R. I., Mansur, A. V., Ascensão, F., Colbert, M., Crossman, K., Elmqvist, T., Gonzalez, A., Güneralp, B., Haase, D., Hamann, M., Hillel, O., Huang, K., Kahnt, B., Maddox, D., Pacheco, A., Pereira, H. M., Seto, K. C., Simkin, R., Walsh, B., ... Ziter, C. (2020). Research gaps in knowledge of the impact of urban growth on biodiversity. *Nature Sustainability*, 3(1), 16–24. <https://doi.org/10.1038/s41893-019-0436-6>
- McPhearson, T., Cook, E. M., Berbés-Blázquez, M., Cheng, C., Grimm, N. B., Andersson, E., Barbosa, O., Chandler, D. G., Chang, H., Chester, M. V., Childers, D. L., Elser, S. R., Frantzeskaki, N., Grabowski, Z., Groffman, P., Hale, R. L., Iwaniec, D. M., Kabisch, N., Kennedy, C., ... Troxler, T. G. (2022). A social-ecological-technological systems framework for urban ecosystem services. *One Earth*, 5(5), 505–518. <https://doi.org/10.1016/j.oneear.2022.04.007>
- Mehrotra, S., Bardhan, R., & Ramamritham, K. (2020). Urban form as policy variable for climate-sensitive area planning under heterogeneity: A geographically weighted regression approach. *Area Development and Policy*, 5(2), 167–188. <https://doi.org/10.1080/23792949.2019.1609368>
- Meyer, P. A., Brouwers, J. Martinus, & Martin, P. J. (2014). A three-dimensional groundwater flow model of the Waterloo Moraine for water resource management. *Canadian Water Resources*

- Journal / Revue Canadienne Des Ressources Hydriques*, 39(2), 167–180.
<https://doi.org/10.1080/07011784.2014.914800>
- Miao, C., Wang, J., & Wang, D. (2024). Research progress on urban forest ecosystem services and multifunctionality. *International Journal of Environmental Science and Technology*.
<https://doi.org/10.1007/s13762-024-06045-1>
- Miller, J. D., Kim, H., Kjeldsen, T. R., Packman, J., Grebby, S., & Dearden, R. (2014). Assessing the impact of urbanization on storm runoff in a peri-urban catchment using historical change in impervious cover. *Journal of Hydrology*, 515, 59–70.
<https://doi.org/10.1016/j.jhydrol.2014.04.011>
- Miller, M. D. (2012). The impacts of Atlanta’s urban sprawl on forest cover and fragmentation. *Applied Geography*, 34, 171–179. <https://doi.org/10.1016/j.apgeog.2011.11.010>
- Moarrab, Y., Salehi, E., Amiri, M. J., & Hovidi, H. (2022). Spatial–temporal assessment and modeling of ecological security based on land-use/cover changes (case study: Lavasanat watershed). *International Journal of Environmental Science and Technology*, 19(5), 3991–4006.
<https://doi.org/10.1007/s13762-021-03534-5>
- Mohammadyari, F., Zarandian, A., Mirsanjari, M. M., Suziedelyte Visockiene, J., & Tumeliene, E. (2023). Modelling Impact of Urban Expansion on Ecosystem Services: A Scenario-Based Approach in a Mixed Natural/Urbanised Landscape. *Land*, 12(2), Article 2.
<https://doi.org/10.3390/land12020291>
- Morgan, A., Branfireun, Brian, & Csillag, F. (2004). An Evaluation of the Contributions of Urbanization and Climatic Change to Runoff Characteristics in the Laurel Creek Watershed, Ontario. *Canadian Water Resources Journal / Revue Canadienne Des Ressources Hydriques*, 29(3), 171–182. <https://doi.org/10.4296/cwrj171>
- Mörtberg, U., Haas, J., Zetterberg, A., Franklin, J. P., Jonsson, D., & Deal, B. (2013). Urban ecosystems and sustainable urban development—Analysing and assessing interacting systems in the Stockholm region. *Urban Ecosystems*, 16(4), 763–782. <https://doi.org/10.1007/s11252-012-0270-3>

- Mumm, O., Zeringue, R., Dong, N., & Carlow, V. M. (2022a). Green Densities: Accessible Green Spaces in Highly Dense Urban Regions—A Comparison of Berlin and Qingdao. *Sustainability*, 14(3), Article 3. <https://doi.org/10.3390/su14031690>
- Mumm, O., Zeringue, R., Dong, N., & Carlow, V. M. (2022b). Green Densities: Accessible Green Spaces in Highly Dense Urban Regions—A Comparison of Berlin and Qingdao. *Sustainability (Switzerland)*, 14(3). Scopus. <https://doi.org/10.3390/su14031690>
- Naseri, N., & Mostafazadeh, R. (2023). Spatial relationship of Remote Sensing Ecological Indicator (RSEI) and landscape metrics under urban development intensification. *Earth Science Informatics*, 16(4), 3797–3810. <https://doi.org/10.1007/s12145-023-01119-z>
- Notte, A. L., Czúcz, B., Vallecillo, S., Polce, C., & Maes, J. (2022). Ecosystem condition underpins the generation of ecosystem services: An accounting perspective. *One Ecosystem*, 7, e81487. <https://doi.org/10.3897/oneeco.7.e81487>
- Ontario Ministry of Agriculture, Food and Rural Affairs. (2023). *Soil Survey Complex—Digital Soil Map of Ontario*. <https://www.lfo.ontario.ca>
- Ontario Ministry of the Environment, Conservation and Parks. (2023). *Air Quality Ontario: Historical monitoring data*. <https://www.airqualityontario.com/history/>
- Ontario Ministry of the Environment, Conservation and Parks. (2023). *Provincial Groundwater Monitoring Network (PGMN)*. <https://www.ontario.ca/environment-and-energy/map-groundwater-information>
- Ontario Ministry of the Environment, Conservation and Parks. (2023). *Water Well Information System (WWIS)*. <https://www.ontario.ca/environment-and-energy/map-groundwater-information>
- Ontario Ministry of Natural Resources and Forestry. (2021). *Southern Ontario Land Resource Information System (SOLRIS) Version 3.0*. Land Information Ontario. <https://www.ontario.ca/page/land-information-ontario>
- Ouyang, X., & Luo, X. (2022). Models for Assessing Urban Ecosystem Services: Status and Outlook. *Sustainability (Switzerland)*, 14(8). Scopus. <https://doi.org/10.3390/su14084725>

- Pan, H., Page, J., Cong, C., Barthel, S., & Kalantari, Z. (2021). How ecosystems services drive urban growth: Integrating nature-based solutions. *Anthropocene*, 35, 100297.
<https://doi.org/10.1016/j.ancene.2021.100297>
- Pickett, S. T. A., Cadenasso, M. L., Grove, J. M., Nilon, C. H., Pouyat, R. V., Zipperer, W. C., & Costanza, R. (2001). Urban Ecological Systems: Linking Terrestrial Ecological, Physical, and Socioeconomic Components of Metropolitan Areas. *Annual Review of Ecology and Systematics*, 32(1), 127–157. <https://doi.org/10.1146/annurev.ecolsys.32.081501.114012>
- Pickett, S. T., A, Cadenasso, M. L., Rosi-marshall, E. J., Belt, K. T., Groffman, P. M., Grove, J. M., Irwin, E. G., Kaushal, S. S., Ladeau, S. L., Nilon, C. H., Swan, C. M., & Warren, P. S. (2017). Dynamic heterogeneity: A framework to promote ecological integration and hypothesis generation in urban systems. *Urban Ecosystems*, 20(1), 1–14. <https://doi.org/10.1007/s11252-016-0574-9>
- Pinto, L., Ferreira, C. S. S., & Pereira, P. (2021). Environmental and socioeconomic factors influencing the use of urban green spaces in Coimbra (Portugal). *Science of The Total Environment*, 792, 148293. <https://doi.org/10.1016/j.scitotenv.2021.148293>
- Pinto, L. V., Inácio, M., Ferreira, C. S. S., Ferreira, A. D., & Pereira, P. (2022). Ecosystem services and well-being dimensions related to urban green spaces – A systematic review. *Sustainable Cities and Society*, 85, 104072. <https://doi.org/10.1016/j.scs.2022.104072>
- Potschin-Young, M., Haines-Young, R., Görg, C., Heink, U., Jax, K., & Schleyer, C. (2018). Understanding the role of conceptual frameworks: Reading the ecosystem service cascade. *Ecosystem Services*, 29, 428–440. <https://doi.org/10.1016/j.ecoser.2017.05.015>
- Privitera, R., Jelo, G., & La Rosa, D. (2024). Assessing Ecosystem Services Provided by Nature-Based Solutions Alongside Different Urban Morphologies. In A. Marucci, F. Zullo, L. Fiorini, & L. Saganeiti (Eds.), *Innovation in Urban and Regional Planning* (pp. 93–105). Springer Nature Switzerland. https://doi.org/10.1007/978-3-031-54096-7_9
- Pukowiec-Kurda, K. (2022). The urban ecosystem services index as a new indicator for sustainable urban planning and human well-being in cities. *Ecological Indicators*, 144, 109532.
<https://doi.org/10.1016/j.ecolind.2022.109532>
- Rahim, N. H., Razak, S. A., Chang, X., Chng Saun, F., Khan, M. N., Hamzah, S. N., Rohman, F., Ali, B., Kaplan, A., Iqbal, M., Ramazan Bozhuyuk, M., & Ercisli, S. (2024). Ecosystem Services by

- Urban Forest (UF)towards Climate Change Adaptation:A Review. *Polish Journal of Environmental Studies*, 33(4), 3503–3513. <https://doi.org/10.15244/pjoes/177147>
- Rahimi, L., Malekmohammadi, B., & Yavari, A. R. (2020). Assessing and Modeling the Impacts of Wetland Land Cover Changes on Water Provision and Habitat Quality Ecosystem Services. *Natural Resources Research*, 29(6), 3701–3718. <https://doi.org/10.1007/s11053-020-09667-7>
- Raudsepp-Hearne, C. (2009). *Managing Ecosystem Services: Tools and theory for understanding the dynamics of multiple ecosystem services on a landscape* [Doctoral Thesis, McGill University]. <https://escholarship.mcgill.ca/concern/theses/p2676w06g>
- Raudsepp-Hearne, C., Peterson, G. D., & Bennett, E. M. (2010). Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *Proceedings of the National Academy of Sciences*, 107(11), 5242–5247. <https://doi.org/10.1073/pnas.0907284107>
- Rees, W., & Wackernagel, M. (1996). Urban ecological footprints: Why cities cannot be sustainable—And why they are a key to sustainability. *Environmental Impact Assessment Review*, 16(4–6), 223–248. [https://doi.org/10.1016/S0195-9255\(96\)00022-4](https://doi.org/10.1016/S0195-9255(96)00022-4)
- Region of Waterloo. (2024, October 23). *About Water*. <https://www.regionofwaterloo.ca/en/living-here/about-water.aspx>
- Region of Waterloo. (2023). *Open data catalogue*. Region of Waterloo. <https://data.regionofwaterloo.ca/>
- Renard, D., Rhemtulla, J. M., & Bennett, E. M. (2016). *Data from: Historical dynamics of ecosystem services bundles* (Version 1, p. 261339 bytes) [Dataset]. Dryad. <https://doi.org/10.5061/DRYAD.G4590>
- Reyers, B., Biggs, R., Cumming, G. S., Elmqvist, T., Hejnowicz, A. P., & Polasky, S. (2013). Getting the measure of ecosystem services: A social–ecological approach. *Frontiers in Ecology and the Environment*, 11(5), 268–273.
- Ronchi, S. (2018a). Ecosystem Services and Planning. In S. Ronchi (Ed.), *Ecosystem Services for Spatial Planning: Innovative Approaches and Challenges for Practical Applications* (pp. 1–26). Springer International Publishing. https://doi.org/10.1007/978-3-319-90185-5_1

- Ronchi, S. (2018b). *Ecosystem Services for Spatial Planning: Innovative Approaches and Challenges for Practical Applications*. Springer International Publishing. <https://doi.org/10.1007/978-3-319-90185-5>
- Ronchi, S. (2018c). Methodology, Approaches and Innovative Experiences. In S. Ronchi (Ed.), *Ecosystem Services for Spatial Planning: Innovative Approaches and Challenges for Practical Applications* (pp. 27–76). Springer International Publishing. https://doi.org/10.1007/978-3-319-90185-5_2
- Ronchi, S. (2018d). Practical Integration of Ecosystem Services in the Planning and Assessment Process. In S. Ronchi (Ed.), *Ecosystem Services for Spatial Planning: Innovative Approaches and Challenges for Practical Applications* (pp. 77–128). Springer International Publishing. https://doi.org/10.1007/978-3-319-90185-5_3
- Ronchi, S. (2018e). The Adoption of RES Methodology in Strategic Environmental Assessment. In S. Ronchi (Ed.), *Ecosystem Services for Spatial Planning: Innovative Approaches and Challenges for Practical Applications* (pp. 129–147). Springer International Publishing. https://doi.org/10.1007/978-3-319-90185-5_4
- Ronchi, S., Pontarollo, N., & Serpieri, C. (2021). Clustering the built form at LAU2 level for addressing sustainable policies: Insights from the Belgium case study. *Land Use Policy*, 109, 105642. <https://doi.org/10.1016/j.landusepol.2021.105642>
- Ruckelshaus, M. H., Jackson, S. T., Mooney, H. A., Jacobs, K. L., Kassam, K.-A. S., Arroyo, M. T. K., Báldi, A., Bartuska, A. M., Boyd, J., Joppa, L. N., Kovács-Hostyánszki, A., Parsons, J. P., Scholes, R. J., Shogren, J. F., & Ouyang, Z. (2020). The IPBES Global Assessment: Pathways to Action. *Trends in Ecology & Evolution*, 35(5), 407–414. <https://doi.org/10.1016/j.tree.2020.01.009>
- Russo, A., & Cirella, G. T. (2021). Urban Ecosystem Services: Current Knowledge, Gaps, and Future Research. *Land*, 10(8), 811. <https://doi.org/10.3390/land10080811>
- Saidi, N. (2021). *Assessment of Ecosystem Services Bundles Contribution to decision-making at city scale* [Doctoral Thesis, University of Dundee]. <https://discovery.dundee.ac.uk/en/studentTheses/assessment-of-ecosystem-services-bundles>

- Sakke, N., Jafar, A., Dollah, R., Asis, A. H. B., Mapa, M. T., & Abas, A. (2023). Water Quality Index (WQI) Analysis as an Indicator of Ecosystem Health in an Urban River Basin on Borneo Island. *Water*, 15(15), Article 15. <https://doi.org/10.3390/w15152717>
- Sang, Å. O., Hagemann, F. A., Ekelund, N., & Svännel, J. (2021). Urban ecosystem services in strategic planning in Swedish municipalities. *Urban Ecosystems*, 24(6), 1343–1357. <https://doi.org/10.1007/s11252-021-01113-7>
- Sarı, E. N., & Bayraktar, S. (2023). The role of park size on ecosystem services in urban environment: A review. *Environmental Monitoring and Assessment*, 195(9), 1072. <https://doi.org/10.1007/s10661-023-11644-5>
- Schwarz, N., Hoffmann, F., Knapp, S., & Strauch, M. (2020). Synergies or Trade-Offs? Optimizing a Virtual Urban Region to Foster Plant Species Richness, Climate Regulation, and Compactness Under Varying Landscape Composition. *Frontiers in Environmental Science*, 8. <https://doi.org/10.3389/fenvs.2020.00016>
- Selbig, W. R., Loheide, S. P., II, Shuster, W., Scharenbroch, B. C., Coville, R. C., Kruegler, J., Avery, W., Haefner, R., & Nowak, D. (2022). Quantifying the stormwater runoff volume reduction benefits of urban street tree canopy. *Science of the Total Environment*, 806. Scopus. <https://doi.org/10.1016/j.scitotenv.2021.151296>
- Semeraro, T., Scarano, A., Buccolieri, R., Santino, A., & Aarrevaara, E. (2021). Planning of Urban Green Spaces: An Ecological Perspective on Human Benefits. *Land*, 10(2), 105. <https://doi.org/10.3390/land10020105>
- Shahtahmassebi, A. R., Wu, C., Blackburn, G. A., Zheng, Q., Huang, L., Shortridge, A., Shahtahmassebi, G., Jiang, R., He, S., Wang, K., Lin, Y., Clarke, K. C., Su, Y., Lin, L., Wu, J., Zheng, Q., Xu, H., Xue, X., Deng, J., & Shen, Z. (2018). How do modern transportation projects impact on development of impervious surfaces via new urban area and urban intensification? Evidence from Hangzhou Bay Bridge, China. *Land Use Policy*, 77, 479–497. <https://doi.org/10.1016/j.landusepol.2018.05.059>
- Shao, Q., Peng, L., Liu, Y., & Li, Y. (2023). A Bibliometric Analysis of Urban Ecosystem Services: Structure, Evolution, and Prospects. *Land*, 12(2), Article 2. <https://doi.org/10.3390/land12020337>

- Sharifi, F., Levin, I., M.Stone, W., & Nygaard, A. (2021). Green space and subjective well-being in the Just City: A scoping review. *Environmental Science & Policy*, 120, 118–126.
<https://doi.org/10.1016/j.envsci.2021.03.008>
- Shi, F., Zhou, B., Zhou, H., Zhang, H., Li, H., Li, R., Guo, Z., & Gao, X. (2022). Spatial Autocorrelation Analysis of Land Use and Ecosystem Service Value in the Huangshui River Basin at the Grid Scale. *Plants*, 11(17), Article 17. <https://doi.org/10.3390/plants11172294>
- Shoemaker, D. A., BenDor, T. K., & Meentemeyer, R. K. (2019). Anticipating trade-offs between urban patterns and ecosystem service production: Scenario analyses of sprawl alternatives for a rapidly urbanizing region. *Computers, Environment and Urban Systems*, 74, 114–125.
<https://doi.org/10.1016/j.compenvurbsys.2018.10.003>
- Söderman, T., Kopperoinen, L., Shemeikka, P., & Yli-Pelkonen, V. (2012). Ecosystem services criteria for sustainable development in urban regions. *Journal of Environmental Assessment Policy and Management*, 14(02), 1250008. <https://doi.org/10.1142/S1464333212500081>
- Sokolova, M. V., Fath, B. D., Grande, U., Buonocore, E., & Franzese, P. P. (2024). The Role of Green Infrastructure in Providing Urban Ecosystem Services: Insights from a Bibliometric Perspective. *Land*, 13(10), Article 10. <https://doi.org/10.3390/land13101664>
- Spatial Modeling and Assessment of Urban Form*. (n.d.). Retrieved November 6, 2024, from <https://books-scholarsportal-info.proxy.lib.uwaterloo.ca/en/read?id=/ebooks/ebooks5/springer5/2019-08-14/4/9783319542171>
- Steele, M. K., & Wolz, H. (2019). Heterogeneity in the land cover composition and configuration of US cities: Implications for ecosystem services. *Landscape Ecology*, 34(6), 1247–1261.
<https://doi.org/10.1007/s10980-019-00859-y>
- Sun, C., Xu, S., Qi, W., Chen, C., Deng, Y., Pei, N., & König, H. J. (2019). Biodiversity constraint indicator establishment and its optimization for urban growth: Framework and application. *Environmental Research Letters*, 14(12), 125006. <https://doi.org/10.1088/1748-9326/ab55aa>
- Sun, X., & Li, F. (2017). Spatiotemporal assessment and trade-offs of multiple ecosystem services based on land use changes in Zengcheng, China. *Science of The Total Environment*, 609, 1569–1581. <https://doi.org/10.1016/j.scitotenv.2017.07.221>

- Tan, B. A., Gaw, L. Y.-F., Masoudi, M., & Richards, D. R. (2021). Nature-Based Solutions for Urban Sustainability: An Ecosystem Services Assessment of Plans for Singapore’s First “Forest Town.” *Frontiers in Environmental Science*, 9, 610155. <https://doi.org/10.3389/fenvs.2021.610155>
- Tan, P. Y., & Abdul Hamid, A. R. B. (2014). Urban ecological research in Singapore and its relevance to the advancement of urban ecology and sustainability. *Landscape and Urban Planning*, 125, 271–289. <https://doi.org/10.1016/j.landurbplan.2014.01.019>
- Tan, P. Y., Zhang, J., Masoudi, M., Alemu, J. B., Edwards, P. J., Grêt-Regamey, A., Richards, D. R., Saunders, J., Song, X. P., & Wong, L. W. (2020). A conceptual framework to untangle the concept of urban ecosystem services. *Landscape and Urban Planning*, 200, 103837. <https://doi.org/10.1016/j.landurbplan.2020.103837>
- Tratalos, J., Fuller, R. A., Warren, P. H., Davies, R. G., & Gaston, K. J. (2007). Urban form, biodiversity potential and ecosystem services. *Landscape and Urban Planning*, 83(4), 308–317. <https://doi.org/10.1016/j.landurbplan.2007.05.003>
- Tv, R., Aithal, B. H., & Sanna, D. D. (2012). Insights to urban dynamics through landscape spatial pattern analysis. *International Journal of Applied Earth Observation and Geoinformation*, 18, 329–343. <https://doi.org/10.1016/j.jag.2012.03.005>
- US EPA, O. (2015, December 23). *Urbanization—Stormwater Runoff* [Data and Tools]. <https://www.epa.gov/caddis/urbanization-stormwater-runoff>
- Valencia Torres, A., Tiwari, C., & Atkinson, S. F. (2021). Progress in ecosystem services research: A guide for scholars and practitioners. *Ecosystem Services*, 49, 101267. <https://doi.org/10.1016/j.ecoser.2021.101267>
- Vauhkonen, J., & Ruotsalainen, R. (2017). Assessing the provisioning potential of ecosystem services in a Scandinavian boreal forest: Suitability and tradeoff analyses on grid-based wall-to-wall forest inventory data. *Forest Ecology and Management*, 389, 272–284. <https://doi.org/10.1016/j.foreco.2016.12.005>
- Veibiakkim, R., Shkaruba, A., & Sepp, K. (2025). A systematic review of urban ecosystem disservices and its evaluation: Key findings and implications. *Environmental and Sustainability Indicators*, 26. Scopus. <https://doi.org/10.1016/j.indic.2025.100612>

- Venter, Z. S., Hassani, A., Stange, E., Schneider, P., & Castell, N. (2024). Reassessing the role of urban green space in air pollution control. *Proceedings of the National Academy of Sciences*, 121(6), e2306200121. <https://doi.org/10.1073/pnas.2306200121>
- Vindigni, G., Mosca, A., Bartoloni, T., & Spina, D. (2021). Shedding Light on Peri-Urban Ecosystem Services Using Automated Content Analysis. *Sustainability*, 13(16), Article 16. <https://doi.org/10.3390/su13169182>
- Wang, B., Oguchi, T., & Liang, X. (2023). Evaluating future habitat quality responding to land use change under different city compaction scenarios in Southern China. *Cities*, 140, 104410. <https://doi.org/10.1016/j.cities.2023.104410>
- Wang, M., Zhao, Y., Shi, W., Yu, J., Chen, T., Bao, J., Song, W., & Chen, H. (2025). A modified curve number method for runoff prediction of different soil types in China. *Catena*, 254. Scopus. <https://doi.org/10.1016/j.catena.2025.108957>
- Wang, S., Li, J., Wu, D., Liu, J., Zhang, K., & Wang, R. (2009). The strategic ecological impact assessment of urban development policies: A case study of Rizhao City, China. *Stochastic Environmental Research and Risk Assessment*, 23(8), 1169–1180. <https://doi.org/10.1007/s00477-008-0290-y>
- Wang, W., Wu, T., Li, Y., Xie, S., Han, B., Zheng, H., & Ouyang, Z. (2020). Urbanization Impacts on Natural Habitat and Ecosystem Services in the Guangdong-Hong Kong-Macao “Megacity.” *Sustainability*, 12(16), Article 16. <https://doi.org/10.3390/su12166675>
- Wilson, F. D. (1984). Urban Ecology: Urbanization and Systems of Cities. *Annual Review of Sociology*, 10, 283–307. <https://www.jstor.org/stable/2083177>
- Wood, S. L. R., & Dupras, J. (2021). Increasing functional diversity of the urban canopy for climate resilience: Potential tradeoffs with ecosystem services? *Urban Forestry & Urban Greening*, 58, 126972. <https://doi.org/10.1016/j.ufug.2020.126972>
- Woodruff, S. C., & BenDor, T. K. (2016). Ecosystem services in urban planning: Comparative paradigms and guidelines for high quality plans. *Landscape and Urban Planning*, 152, 90–100. <https://doi.org/10.1016/j.landurbplan.2016.04.003>

- Yan, M., Chen, L., & Sun, R. (2023). Climatic effects on landscape multifunctionality in urban parks: A view for integrating ecological supply and human benefits. *Environmental Research Letters*, 19(1), 014032. <https://doi.org/10.1088/1748-9326/ad10d8>
- Yi, D., Guo, J., Pueppke, S. G., Han, Y., Ding, G., Ou, M., & Koomen, E. (2025). What dominates the variation of ecosystem services across different urban expansion patterns?—Evidence from the Yangtze River Delta region, China. *Environmental Impact Assessment Review*, 110, 107674. <https://doi.org/10.1016/j.eiar.2024.107674>
- Zhang, H., Deng, W., Zhang, S., Peng, L., & Liu, Y. (2022). Impacts of urbanization on ecosystem services in the Chengdu-Chongqing Urban Agglomeration: Changes and trade-offs. *Ecological Indicators*, 139, 108920.
- Zhang, S., Xiong, K., Qin, Y., Min, X., & Xiao, J. (2021). Evolution and determinants of ecosystem services: Insights from South China karst. *Ecological Indicators*, 133, 108437. <https://doi.org/10.1016/j.ecolind.2021.108437>
- Zhang, X., Han, L., Wei, H., Tan, X., Zhou, W., Li, W., & Qian, Y. (2022). Linking urbanization and air quality together: A review and a perspective on the future sustainable urban development. *Journal of Cleaner Production*, 346, 130988. <https://doi.org/10.1016/j.jclepro.2022.130988>
- Zhang, Y., Yang, Z., & Yu, X. (2015). Urban Metabolism: A Review of Current Knowledge and Directions for Future Study. *Environmental Science & Technology*, 49(19), 11247–11263. <https://doi.org/10.1021/acs.est.5b03060>
- Zhang, Z., Zhang, J., Liu, L., Gong, J., Li, J., & Kang, L. (2023). Spatial–Temporal Heterogeneity of Urbanization and Ecosystem Services in the Yellow River Basin. *Sustainability*, 15(4), Article 4. <https://doi.org/10.3390/su15043113>
- Zhao, C., & Sander, H. A. (2018). Assessing the sensitivity of urban ecosystem service maps to input spatial data resolution and method choice. *Landscape and Urban Planning*, 175, 11–22. Scopus. <https://doi.org/10.1016/j.landurbplan.2018.03.007>
- Zheng, D., Wang, Y., Hao, S., Xu, W., Lv, L., & Yu, S. (2020). Spatial-temporal variation and tradeoffs/synergies analysis on multiple ecosystem services: A case study in the Three-River Headwaters region of China. *Ecological Indicators*, 116, 106494. <https://doi.org/10.1016/j.ecolind.2020.106494>

- Zhou, L., Qin, Y., Cheng, J., Zhu, H., Li, M., Zhang, J., LeBleu, C., Shen, G., Chen, T., & Liu, Y. (2025). Urban ecosystem services, ecological security patterns and ecological resilience in coastal cities: The impact of land reclamation in Macao SAR. *Journal of Environmental Management*, 373, 123750. <https://doi.org/10.1016/j.jenvman.2024.123750>
- Zhou, L., Zhang, H., Bi, G., Su, K., Wang, L., Chen, H., & Yang, Q. (2022). Multiscale perspective research on the evolution characteristics of the ecosystem services supply-demand relationship in the chongqing section of the three gorges reservoir area. *Ecological Indicators*, 142, 109227. <https://doi.org/10.1016/j.ecolind.2022.109227>
- Zou, Z., Wu, T., Xiao, Y., Song, C., Wang, K., & Ouyang, Z. (2020). Valuing natural capital amidst rapid urbanization: Assessing the gross ecosystem product (GEP) of China's 'Chang-Zhu-Tan' megacity. *Environmental Research Letters*, 15(12), 124019. <https://doi.org/10.1088/1748-9326/abc2f8>

Appendix A

Deriving CN numbers: Overlaying MTO and SOLRIS CODES land use codes

LU_SOLRIS_Code	SOLRIS Land Use	Comment	CN_Code
90	Forest	Mapped to MTO Forest CN values	30
81	Forest	Mapped to MTO Forest CN values	30
82	Forest	Mapped to MTO Forest CN values	30
83	Forest	Mapped to MTO Forest CN values	30
91	Forest	Mapped to MTO Forest CN values	30
92	Forest	Mapped to MTO Forest CN values	30
93	Forest	Mapped to MTO Forest CN values	30
65	Forest	Mapped to MTO Forest CN values	30
191	Forest	Mapped to MTO Forest CN values	30
192	Forest	Mapped to MTO Forest CN values	30
250	Open Space (Grass cover)	Mapped to Open space with good grass cover	49
203	Urban Impervious	Mapped to Fully Impervious surfaces	98
1931	Row Crops - Corn	Mapped to Row Crops - Corn	67
1932	Row Crops - Wheat	Mapped to Row Crops - Wheat	65
1933	Row Crops - Soy	Mapped to Row Crops - Soy	66
1934	Row Crops - Other	Average of above row crops	66
202	Pasture	Mapped to Urban recreational spaces	49
250	Fallow	Mapped to Fallow/Idle Agricultural Land	64
131	Wetlands	Mapped to Wetlands (assumed high CN)	77
135	Wetlands	Mapped to Wetlands (assumed high CN)	77
140	Wetlands	Mapped to Wetlands (assumed high CN)	77
150	Wetlands	Mapped to Wetlands (assumed high CN)	77
160	Wetlands	Mapped to Wetlands (assumed high CN)	77
170	Water	Mapped to Open Water (CN=100)	100
64	Barren Land	Mapped to Barren/Exposed soils	76
201	Roads	Mapped to Roads/Transport surfaces	98