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# The role of urban biodiversity in green infrastructure multifuncionality and environmental justice

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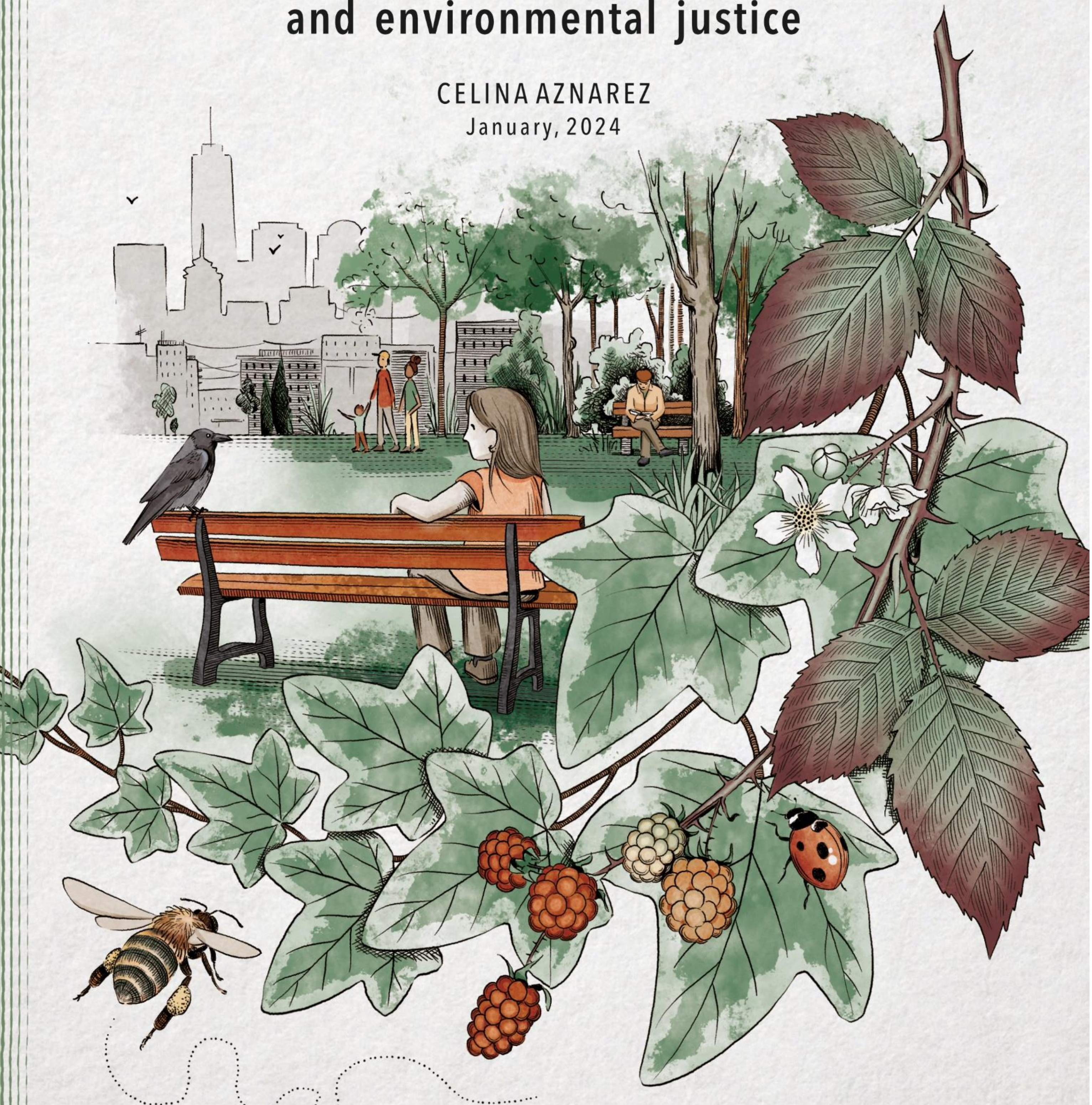

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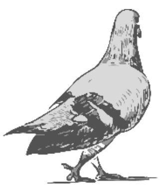
“Their visibility is part of the problem.
In contrast to badgers, polecats, deer, and other denizens of the borderlands of nature and culture, pigeons are not found on the periphery of cities.
They live in the most public, most visible places in the city.

In contrast to rats and cockroaches, they do not emerge only at nightfall, but exist within the city in the bright light of day.
They can fly away, and have always done so, which is why they cannot be banished indoors like cats or put on a leash like dogs. […]

The pigeon is a living metaphor for excess and communication, for insubordinate migration without a fatherland, and for producing solidarity in improbable places. […]

Where there are cities, there are also pigeons.
And where there are pigeons, there is resistance”

*Fahim Amir, 2020.*
*Pigeon politics. Being and Swine: The End of Nature (As We Knew It)*



# Contents

## Preface

This thesis is submitted following the requirements for the PhD in Environmental Sciences and Technoloy, at the Institute of Environmental Science and Technology (ICTA) of the Autonomous University of Barcelona (UAB). The thesis presented here summarises the main outcomes of my doctoral research, conducted within the La Caixa INPhINIT–INCOMING program, under the fellowship code (LCF/BQ/DI20/11780004). The research took place from October 2020 to December 2023, based at the Basque Centre for Climate Change (BC3) with ICTA doctoral program as primarily affiliation.

Supervised by Unai Pascual (BC3) and Francesc Baró (ICTA/Vrije Universiteit Brussels), this PhD also involved a close collaboration with the Ecoinformatics and Biodiversity section at Aarhus University, where I undertook a research stay from August to December 2021 under the guidance of Jens-Christian Svenning. The collaboration extended over the years through shorter visits. Additionally, in May 2023, I conducted a research stay at the Cosmopolis Centre for Urban Research, Vrije Universiteit Brussels, under the supervision of Francesc Baró.

Two out of the three empirical chapters in this dissertation have already been published, while the third is currently under review. Beyond the papers included in this thesis, I have co-authored five papers from previous collaborations and published one additional paper as the first author, along with a book chapter in the same capacity. This diverse body of work has allowed me to strengthen and expand collaborations and expertise in my field.

Additionally, I coordinated a project funded by the María de Maeztu Excellence Unit 2023-2027 to conduct the final empirical chapter of my PhD. This project facilitated collaboration with other research groups within BC3, particularly the integrated modeling group (ARIES - Artificial Intelligence for Environment & Sustainability), resulting in the development of new spatial prioritization tools for urban planning. While initially intended to deliver my last empirical chapter, the project fostered significant synergy, leading to the ongoing preparation of two additional research papers in which I am a co-author.

In terms of dissemination, progress updates were communicated to research groups such as BCNUEJ (Barcelona Lab for Urban Environmental Justice), Ecoinformatics and Biodiversity at Aarhus University, and CGIS (Brussels Centre for Urban Studies) through seminars covering topics like environmental justice, ecology, and urban climate. The core chapter results were presented at international conferences, including the Nordic Society Oikos (2021), Ecosystem Services Partnership conferences in the EU (2022), LAC (2023), and the upcoming European Geosciences Union (2024). Additionally, I co-hosted sessions at ESP EU (2022) and ESP LAC (2023) conferences, and I will also be co-hosting a session at the upcoming EGUconference (2024). Moreover, select outcomes were disseminated through ‘The Conversation’, a platform fostering collaboration between journalists and researchers, utilizing Creative Commons licenses to convey scientific findings to the general public. Finally, the thesis outcomes and implications were discussed in meetings with the Environmental Studies Centre in Vitoria-Gasteiz. During my PhD, I also contributed to the academic community by reviewing several publications for four different journals. By the end of 2023, I became a part of the editorial committee as an ECR for Sustainable Cities and Society.

## Because it takes a village to do a PhD: Thank you

The doctoral journey presented its own set of challenges, particularly the distance (over 10000 km away) from my family in Uruguay and living apart from my husband, amidst a pandemic. Missing out on holidays, birthdays, and virtually witnessing the passage of time in the lives of our family members was a constant reminder of the distance and the relative nature of time and life priorities. I started this journey because of their support, and for that, I am profoundly grateful. Thank you for standing by me in moments of joy and crisis, always pushing me forward with love.

First, I want to thank Juan Pablo, my husband. Words fall short in expressing how grateful I am for all the support and companionship I have received during these years. From relocating in the midst of COVID to Bilbao to be with me in this new chapter, completing his PhD in isolation in a different country from his workplace and friends, navigating the distance from our family, and for all the sacrifices afterwards to conciliate and preserve our paths together. Thank you for your invaluable support and help whenever I needed the most critical, yet constructive, and caring feedback. Thank you for reminding me when things could get confusing, to distinguish between the urgent and the important, and for the continuous growth of sharing life and new challenges together. Your unconditional love, care, and selflessness have painted it all yellow.

I'm grateful to my supervisors, Unai Pascual and Francesc Baró, for their trust, time, and dedication over the past three years. Their encouragement of my ideas has been crucial to my learning process. They provided the freedom for me to explore my own interests, connections and collaborations, never pushing back or imposing. I am beyond grateful for their responsiveness and support. Special thanks to my tutor, Isabelle, for her time and support with the less friendly and formal aspects of this journey. Her willingness to help, even under tight deadlines, and efforts to integrate me into the BCNUEJ activities have made this journey more enjoyable.

I take pride in being a first-gen graduate from the University of the Republic of Uruguay, and I am grateful for having received an excellent public and free education that allowed me to pursue this career path. Special acknowledgement goes to my supervisors and mentors in Uruguay, Isabel Gadino, Hugo Inda, Adriana Goñi, Carlos Iglesias, and Estela Delgado. Their support during my undergraduate training and the application process for the Caixa Fellowship, which ultimately led to this opportunity has been invaluable. Without their guidance and encouragement, this wouldn't have been possible in so many ways. I extend my gratitude to my friend and colleague German Taveira. Working together has been an honour, and I look forward to exploring new research avenues to collaborate in the future!

To Alba, Ana, Rini, and Itxaso, thank you for your companionship over the years. It's been a pleasure to work together side by side, finding ways to collaborate and support each other for the pleasure of it, as well as many adventures involving travelling, music, carnival, pintxoes, and many alfajores. Above all, I cherish the wonderful friendship that we have cultivated during this time.

Julia, Sean, and Guilhem, who were my bubble during the pandemic, starting from scratch in a place I didn't know, embarking on an equally unknown adventure for all. Thanks to them for discovering it together, even with a curfew in between.

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Ioar and Vicky, for your good advice, encouragement, and for being so straightforward and empathetic! Mila esker for your friendship and showing us the Basque Country.

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Special thanks to Jens-Christian Svenning in the Ecoinformatics section of Aarhus University for warmly welcoming me during my stay in Denmark. Thank you for being so generous with your time, patience, and attention and for all the good research advice. A warm thank you to Maya Pasgaard, for all the engaging and critical discussions, the literature suggestions, for your caring support, and for integrating me into the Sustainscapes group. It has been a pleasure working with you two! Thanks to the section colleagues for the warm welcome, lively breakfasts, 'crying cookies', DHL races and enjoyable times at Smagløs.

To Vrije Universiteit Brussels and the COSMPOLIS team, thank you for having me! I appreciate the good times and warm welcome from Elsa, Amy, Tao, Rob, and everyone else.

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My heart has expanded to embrace new people and places, and I'm grateful for the journey.

# Summary

As urban areas expand, societal contact with nature is increasingly confined to cityscapes, highlighting the need to explore the interactions between urban biodiversity, urban green infrastructure (UGI), and human well-being. The spatial distribution, accessibility, and quality of urban ecosystems are important aspects to consider when assessing these interactions. For instance, the uneven distribution and accessibility to UGI may cause many urban dwellers to live in poor environmental conditions with direct impacts on their health. Recognising the critical role of biodiversity in supporting UGI and the provision of ecosystem services (ES) is key. This requires an examination of the UGI spatial patterns and ecological characteristics that contribute to the production of ES. This dissertation examines the contribution of urban biodiversity to the multifunctionality of green infrastructure from an environmental justice perspective. The research unfolds through three standalone empirical chapters, each building on the outcomes of the previous one and corresponding to a specific research objective, using the "European Green Capital" Vitoria-Gasteiz, a middle-sized city and the capital of the Basque Country, as a case study. Chapter II assessed the urban wildness and habitat quality of UGI as biodiversity predictors adapted to urban ecosystems. To achieve this, I adapt spatial modelling tools traditionally used in areas with minimal human influence. This adaptation challenges the traditional perspective that considers urban land uses solely sources of threats to non-human-dominated landscapes, rather than recognising their potential as habitat sources. Results from Vitoria-Gasteiz reveal varying levels of relative habitat quality and wildness, with small and fragmented green spaces failing to effectively support urban biodiversity compared with larger and less fragmented periurban green spaces. Chapter III focuses on how socioeconomic and historical factors shape urban ecosystems and regulate ES provision from a distributive environmental justice perspective. Using remote sensing and census data, I examine luxury and legacy effects on urban biodiversity, vegetation cover, and ES, considering interactive effects among sociodemographic and biophysical variables of interest. Findings suggest that urban biodiversity is primarily driven by socioeconomic factors related to social "status" (*i.e.* high education attainment), whereas vegetation cover and regulating ES are influenced by management legacies in interaction with population density. In Chapter IV, I analyse the spatial patterns of temperature regulation ES supply and demand mismatches, considering sociodemographic and health factors driving distributional inequalities in exposure to urban heat as an environmental hazard. The results highlight the existence of disparities in access to nature-based urban cooling and hence the level of heat vulnerability, with increased exposure observed in areas inhabited by disadvantaged communities in contrast to more affluent areas. Together, these three empirical studies provide valuable insights into the influence of socioeconomic and health factors on urban ecosystem quality and distribution, and their cascading impact on human well-being and inequitable access to ES. These findings contribute to a better understanding of urban socio-ecological systems, providing actionable insights for advancing urban agendas towards equitable urban development in both Vitoria-Gasteiz and beyond.

## Resumen

A medida que las áreas urbanas se expanden, el contacto de la sociedad con la naturaleza se ve cada vez más limitado a los paisajes urbanos, resaltando la necesidad de explorar las interacciones entre la biodiversidad urbana, la infraestructura verde urbana (IVU) y el bienestar humano. La distribución espacial, accesibilidad y calidad de los ecosistemas urbanos son aspectos importantes a considerar al evaluar estas interacciones. Por ejemplo, la distribución desigual y la accesibilidad a la IVU pueden hacer que muchos habitantes urbanos vivan en condiciones ambientales deficientes con impactos directos en su salud. Reconocer el papel fundamental de la biodiversidad en el soporte de la IVU y la provisión servicios ecosistémicos (SE) es clave. Esto requiere un examen de los patrones espaciales de la IVU y las características ecológicas que contribuyen a la producción de SE. Esta tesis examina la contribución de la biodiversidad urbana a la multifuncionalidad de la infraestructura verde desde una perspectiva de justicia ambiental. La investigación se desarrolla a través de tres capítulos empíricos independientes, cada uno construyendo sobre los resultados del anterior y correspondiendo a un objetivo de investigación específico, utilizando Vitoria-Gasteiz, la "Capital Verde Europea", como estudio de caso, una ciudad de tamaño mediano y la capital del País Vasco. El Capítulo II evaluó la naturaleza silvestre urbana y la calidad del hábitat de la IVU como predictores de la biodiversidad adaptados a los ecosistemas urbanos.  Para lograr esto, adapto herramientas de modelado espacial tradicionalmente utilizadas en áreas con mínima influencia humana. Esta adaptación desafía la perspectiva tradicional que considera que los usos del suelo urbanos son solo fuentes de amenazas para paisajes no dominados por humanos, en lugar de reconocer su potencial como fuentes de hábitat. Los resultados de Vitoria-Gasteiz revelan niveles variables de calidad relativa del hábitat y naturaleza silvestre, con espacios verdes pequeños y fragmentados que no logran apoyar efectivamente la biodiversidad urbana en comparación con espacios verdes periurbanos más grandes y menos fragmentados. El Capítulo III se centra en cómo factores socioeconómicos e históricos dan forma a los ecosistemas urbanos y regulan la provisión de SE desde una perspectiva de justicia ambiental distributiva. Utilizando datos de teledetección y censos, examino los efectos de lujo y legado en la biodiversidad urbana, la cobertura vegetal y los SE, considerando efectos interactivos entre variables sociodemográficas y biofísicas de interés. Los hallazgos sugieren que la biodiversidad urbana está impulsada principalmente por factores socioeconómicos relacionados con el "estatus" social (es decir, una alta educación), mientras que la cobertura vegetal y los SE reguladores están influenciados por legados de gestión en interacción con la densidad de población. En el Capítulo IV, analizo los patrones espaciales de desajustes en la oferta y demanda de regulación térmica de SE, considerando factores sociodemográficos y de salud que impulsan desigualdades en la exposición al calor urbano como riesgo ambiental. Esto implica la integración de datos de teledetección, salud y sociodemográficos con herramientas de inteligencia artificial y sistemas de información geográfica. Los resultados destacan disparidades en la vulnerabilidad al calor, con una mayor exposición observada entre comunidades desfavorecidas en contraste con áreas más acomodadas. Los tres capítulos proporcionan conocimiento sobre la calidad ecológica de los espacios

urbanos, la influencia de los factores socioeconómicos en la naturaleza urbana y su impacto en cascada en el bienestar humano, a través de disparidades en el acceso a los SE. En conjunto, estos tres estudios empíricos ofrecen ideas valiosas sobre la influencia de factores socioeconómicos y de salud en la calidad y distribución de los ecosistemas urbanos, y su impacto cascada en el bienestar humano y el acceso desigual a los SE. Estos hallazgos contribuyen a una mejor comprensión de los sistemas socioecológicos urbanos, proporcionando ideas prácticas para avanzar en las agendas urbanas hacia un desarrollo urbano equitativo, tanto en Vitoria-Gasteiz como más allá.

## List of abbreviations and acronyms:

ARIES - Artificial Intelligence for Environment and Sustainability
CBD - Convention on Biological Diversity
CEA – Environmental Studies Centre [Centro de Estudios Ambientales]
COVID-19 – Coronavirus Disease 2019
CV – Coefficient of Variation
DBH – Diameter at Breast Height
DEM – Digital Elevation Model
EC –European Commission
EEA – European Environment Agency
ES – Ecosystem Services
EU – European Union
FAIR – Findable, Accessible, Interoperable, Reusable
GEE – Google Earth Engine
GIS – Geographic Information System
GLMs – Generalized Linear Models
HVI – Heat Vulnerability Index
HQ – Habitat quality
INVEST - Integrated Valuation of Ecosystem Services and Trade-offs
IPCC – Intergovernmental Panel on Climate Change
IUCN – International Union for Conservation of Nature
LST – Land Surface Temperature
NBS – Nature Based Solutions
NDVI – Normalized Difference Vegetation Index
NDWI – Normalized Difference Water Index
PCA – Principal Component Analysis
RF – Random Forest
SD – Standard Deviation
SDGs – Sustainable Development Goals
SIMPER – Similarity percentage
SMR – Standardised Mortality Ratio
TR-ES – Temperature regulating Ecosystem Services
UGBS – Urban Green and Blue Spaces
UGI – Urban Green Infrastructure
UGS – Urban Green Spaces
UHI – Urban Heat Island
UW – Urban Wildness

# Chapter I
# Introduction

## 1.1. Background

*Nature in urban environments*

With over half of the global population residing in urban areas, cities play a key role in shaping the functioning of coupled natural and social systems (Elmqvist *et al.*, 2021; Keeler *et al.*, 2019). This interaction brings ecological novelty to urban areas, with distinctive environments shaped by human activities, encompassing changes in disturbance regimes, alterations in resources and habitat conditions, among others (Murray *et al.*, 2022; Teixeira *et al.*, 2021; Andersson *et al.*, 2014; Hobbs *et al.*, 2006). However, anthropogenic impacts derived from urbanisation, such as impervious infrastructure development, intensified pollution, and ecological degradation, impact these socio-ecological interactions in several ways (Elqmvist *et al.*, 2021).

Population concentration in urban areas exacerbates alienation from nature, which is the decoupling between people and the perception of interdependence with the natural environment (Marvier *et al.*, 2023; Soga & Gaston, 2018; 2016). The growing confinement to urban landscapes limits people´s exposure to nature and thus influences their perceptions of what is desirable or acceptable from a natural environment (Soga & Gaston, 2018). This can be further amplified with time, since successive generations may come to normalise environmental degradation due to a lack of direct experience or historical information about past environmental conditions (Soga & Gaston, 2018). Moreover, promoting direct and meaningful experiences with nature is crucial for engaging with and maintaining a sense of personal connectedness to nature (Soga & Gaston, 2016). The role of urban nature in this connection is known as the “pigeon paradox”. Here, the pigeon, a common and widely distributed urban species, plays a key role on people’s experiences with urban nature, and thus helps in emphasising the human dependence on nature and the critical need for global conservation efforts (Dunn *et al.*, 2006).

*Socio-environmental challenges in urban areas*

Urban areas are socio-ecological systems under strong anthropogenic pressures, which impact their nature and thereby influence their capacity to provide multiple functions. Cities are, therefore, examples of the convergence of social–ecological–technological systems (*i.e.* including climatic, ecological, political, social, institutional, infrastructural, financial, and technological), all related to what has been described as the Anthropocene (Elqmvist *et al.*, 2021; McPhearson *et al.*, 2021; Steffen *et al.*, 2015). The Anthropocene is characterised by accelerated environmental changes derived from human activities, resulting in ecosystem degradation, climate change and biodiversity loss (McPhearson *et al.*, 2021; Steffen *et al.*, 2015). Furthermore, human-induced climate change has led to increased recurrence and intensity of extreme events such as heatwaves (IPCC, 2007). Social and environmental challenges, including climate change, biodiversity loss, resource availability, distributive inequity, and human and environmental health, are intertwined with the structure and function of urban ecosystems (Murray *et al.*, 2022; Keeler *et al.*, 2019; Nilon *et al.*, 2017). These growing and often compounding challenges in cities emphasise the urgency of promoting urban environments that can support human well-being.

The spatial configuration of urban ecosystems and biodiversity, influences the ecological structures that support ecosystem functions and services (Alberti, 2024; McPhearson *et al.*, 2016). Efforts to support these functions include urban greening, strategies that are becoming integral components of urban policy agendas, and aligning with global frameworks such as the Sustainable Development Goals (SDGs). Of particular relevance is SDG 11 on “Sustainable cities and communities” and specifically, target 11.7, which emphasises the commitment to universal access to green spaces. At the European level, the EU Commission’s Biodiversity Strategy for 2030 (European Commission, 2021) marks a significant milestone by explicitly incorporating urban nature and urging cities with over 20,000 inhabitants to implement ambitious greening strategies by 2030. These efforts are instrumental in shaping the discourse and concepts related to sustainable urban planning and management at the European level, as reflected in a growing body of policy and literature on urban green infrastructure (UGI), nature-based solutions, and ecosystem services (ES) (Seiwert & Rößler, 2020). Adopting these notions has led to initiatives such as the European Green Capital Award, a prize endorsed by the EU to recognise and support urban green and sustainable policies (European Commission, 2022).

Policies aimed at protecting and enhancing ES have evolved to align with post-2020 biodiversity goals (CBD, 2020), as highlighted in the EU Biodiversity Strategy 2030 and the new Ecological Restoration Directive (European Commission, 2022). The new restoration law proposes obligations for governments to stop the net losses of both green and blue spaces in urban environments by 2030 and increase them by 2040 and 2050. These strategies underscore the diverse benefits of urban nature for people and their dependence on biodiversity, usually by promoting green infrastructure (GI). Green infrastructure is the "network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of ecosystem services" (EEA, 2013). UGI can provide multiple ES, which are defined as the direct and indirect contributions of ecosystems to well-being (Potschin-Young *et al.*, 2018). Temperature regulation, stormwater runoff control, air and water quality improvement, and opportunities for recreation and social cohesion are just some examples of ES provided by UGI (Baró *et al.*, 2019; Gómez-Baggethun & Barton, 2013). Furthermore, UGI offer opportunities to reconnect with nature, addressing the prevailing alienation from nature, and provide spaces that promote the physical and mental well-being of urban residents, which emerged as a basic need during the COVID-19 pandemic where restrictions and lockdowns let people confined to their immediate surroundings (Honey-Rosés *et al.*, 2020; Ribeiro *et al.*, 2021). However, the capacity of UGI to provide ES is largely regulated by the biophysical attributes of the ecosystem, including biodiversity (Potschin-Young *et al.*, 2018).

*Urban biodiversity for ES support*

Biodiversity is a key component of ecosystems and refers to the variety and variability of living organisms in a certain area, whereas urban biodiversity specifically refers to the biodiversity in the surroundings of human settlements (Forman, 2014; CBD, 2012). Being a critical biophysical component of ecosystems, biodiversity plays a vital role in sustaining ES (Potschin-Young *et al.*, 2018) and thus enhance residents' well-being (Gómez-Baggethun & Barton, 2013). However, biodiversity does not support every ES in the same way, and its role is influenced by interactions with other biophysical and spatial attributes of the UGI (Marselle *et al.*, 2021; Schwarz *et al.*, 2017; Ziter, 2016). Landscape heterogeneity, including the distribution of impervious cover, heat islands, and vegetation cover, affects ecological and evolutionary patterns and processes, thereby influencing biodiversity patterns in urban areas. An integrated analysis of the interaction of biodiversity with other biophysical attributes and influencing socioeconomic drivers is key to understanding the role of UGI in providing ES for human well-being. The concept of wildness becomes relevant for this integrated analysis, since it allows the expression of landscape quality along a *continuum* of anthropogenic impact, from highly impacted urban patches to pristine areas (Noss, 2020; Müller *et al.*, 2018). Likewise, habitat quality (HQ) is a fundamental concept for biodiversity in urban contexts because it indicates the ecosystems´capacity to sustain wildlife, and thus, for biodiversity suitability and ecological functions (Hall *et al.*, 1997). Both indicators contribute to unravelling the complex relationship between biodiversity in UGI, ES, and human well-being.

Strategies promoting UGI for its multifunctional benefits face challenges in urban areas, due to land cover transformation and land use intensification, leading to habitat fragmentation and green space conversion (Stott *et al.*, 2015). Thus, the success of urban greening efforts relies on the complex interplay of social, ecological, and technological factors influencing UGI´s functioning (McPhearson *et al.*, 2021). Overlooking such processes could lead to unforeseen consequences on the ecosystem structures and functions that these solutions aim to preserve (Alberti, 2024; 2020; Des Roches *et al.*, 2020). This further implies that in the context of accelerated urban growth, it is key to understand how biodiversity is affected by rapidly expanding and heterogeneous urban environments, as this might affect the provision of ES along with biodiversity conservation. Particularly so since projected urban development is expected to progressively affect areas with higher biodiversity and even biodiversity hotspot areas (Lambert & Schell *et al.*, 2023; Seto *et al.*, 2012). Ecological research has mainly focussed on biophysical factors like climate and geography as primary drivers of local biodiversity (Leong *et al.*, 2018). Interdisciplinary research in urban ecology has increasingly recognised the interplay between biophysical and social factors (*i.e.* race, age, wealth) in shaping biodiversity (Lambert & Schell *et al.*, 2023; Hope, 2003).

*Effects of socio-ecological heterogeneity on ES support*

The socio-ecological heterogeneity of urban areas, including natural and artificial elements, is shaped by management policies and historical development. Thus, urban ecosystems and ES are co-produced by the interaction of the socioeconomic characteristics and behaviours of their societies (Alberti, 2024; Palomo *et al.*, 2016;

Grove *et al.*, 2014). Among these, social inequality is increasingly acknowledged as a driver of ecological heterogeneity (Lambert & Schell, 2023; Murray *et al.*, 2022; Schell *et al.*, 2020). Social inequality influences the distribution of land ownership and access to ecosystems and associated ES (Anguelovski *et al.*, 2020; Grove *et al.*, 2014). This privileges some individuals over others in terms of access to the ecosystem functions and services that sustain their well-being (Fig. 1) (Schell *et al.*, 2020; Langemeyer & Connolly., 2020).

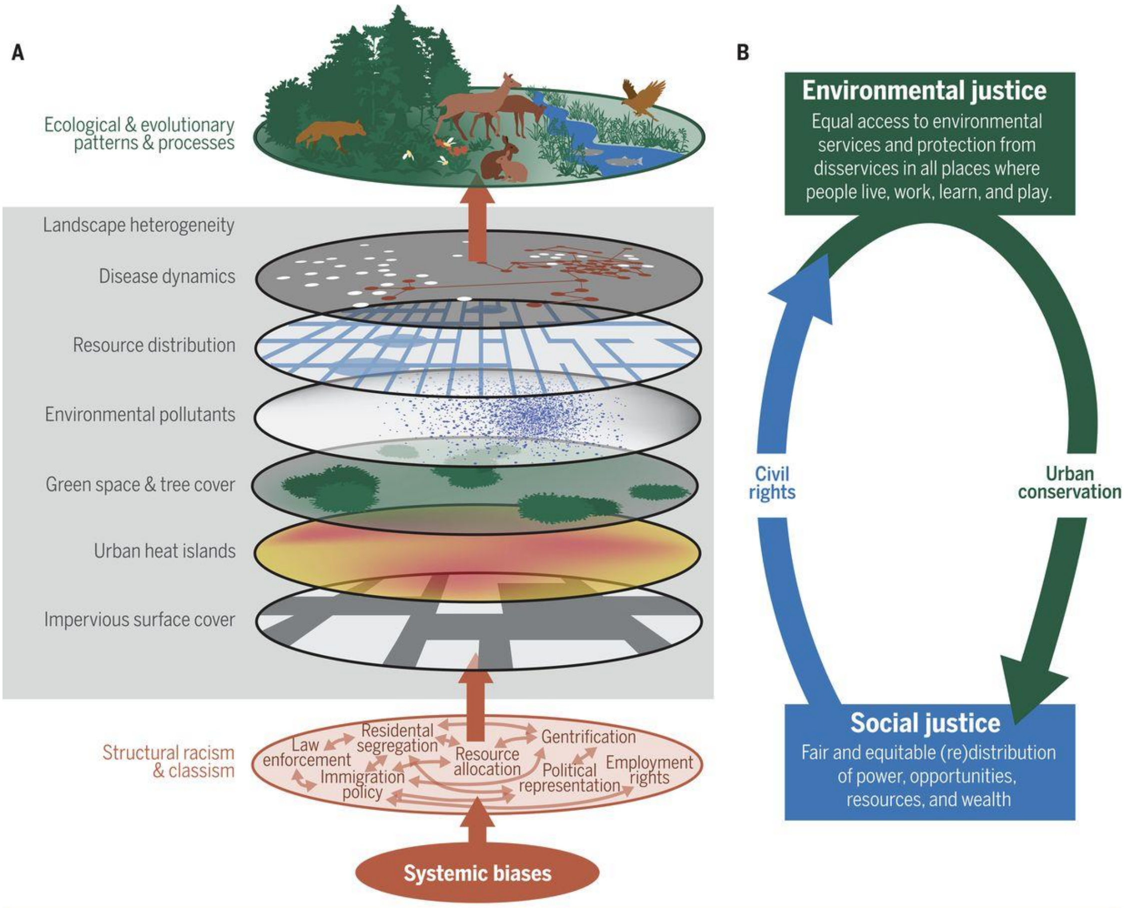


**Figure 1**. Systemic biases in structural racism and classism drive landscape heterogeneity in cities, including impervious surface cover, urban heat islands, green space, and tree cover. These influences the natural ecological and evolutionary patterns and processes of urban nature, thereby influencing the provision of ES. Incorporating environmental justice perspectives into ecology research promotes equal access to ES and positively impacts urban conservation and sustainability. Source: Schell *et al.*, 2020.

In addition, structural inequalities, or disparities in wealth, resources, and other outcomes resulting from discriminatory practises of institutions, profoundly shape the biophysical attributes of cities (Kuras *et al.*, 2020). This drives uneven patterns of vegetation cover and resource availability in cities through residential segregation, gentrification, and other systematic-biased based policies (Fig.1) (Schell *et al.*, 2020; Kuras *et al.*, 2020; Anguelovski *et al.*, 2020). This uneven allocation of resources and conditions has implications for both human and environmental health in urban ecosystems, like exposure to urban heat, environmental pollutants, and influences disease dynamics (Murray *et al.*, 2022; Schell *et al.*, 2020; Grove *et al.*, 2014; Kabisch & Haase, 2014).

*Role of management and environmental justice in ES provision*

Socioeconomic and historical management are also key factors in shaping urban biodiversity and GI, thereby influencing the distribution and access to ecosystem functions and services (Lambert & Schell, 2023; De Roches *et al.*, 2020; Smith, 2007). For instance, vegetation and associated ES are more accessible in affluent neighbourhoods, which often exhibit higher vegetation cover (Leong *et al.*, 2018; Hope *et al.*, 2003). This makes residents of affluent areas less vulnerable to environmental stressors such as heat, whereas disadvantaged neighbourhoods often face disproportionate burdens of negative environmental impacts (Nowak *et al*., 2022; Jenerette *et al.*, 2011). These patterns are intertwined with equity considerations because ecological outcomes exhibit distinct gradients of socioeconomically stratified environmental quality (Alberti *et al.*, 2024; Murray *et al.*, 2022; Kuras *et al.*, 2020).

Social justice, as a base for a fair and equitable social distribution, should be incorporated into the management of cities as socio-ecological systems to ensure equal access to ES and protection from negative environmental impacts (Schell *et al.*, 2020). Acknowledging environmental justice for urban ecosystem management positively impacts urban conservation and sustainability and further reinforces overall social justice (Fig. 1B, Schell *et al.*, 2020). Examining urban nature and biodiversity distribution requires a perspective of environmental justice, since it is key for understanding the role of urban nature in the production and allocation of ES among social groups (Lambert & Schell, 2023; Kuras *et al.*, 2020). Recent efforts in ES research underscore the need for integrating the environmental justice perspective to promote more equitable and just outcomes resulted from UGI interventions and greening policies (Calderón-Argelich *et al.*, 2021; Langemeyer & Connolly, 2020; Angelovski *et al.*, 2020; Meerow, 2020). This approach encompasses not only the positive contributions of ES but also potential pitfalls arising from their deployment in urban settings and amplifying factors contributing to green injustices (Calderón-Argelich *et al.*, 2021; Langemeyer & Connolly, 2020).

*Current research gaps in biodiversity and ES support*

The research efforts outlined above underscore critical gaps in our understanding of the role of biodiversity in interaction with the biophysical attributes of urban ecosystems, landscape heterogeneity and socioeconomic and historical management for ES provision and related human well-being. Despite acknowledging the general benefits of UGI for ES provision and human well-being, there is still little understanding about how biodiversity and related indicators (as HQ and wildness) influence ES provision in urban ecosystems (Ziter, 2016). The relevance of these studies has been emphasised to counteract the systematic oversight in GI planning and management policies of the role of biodiversity in sustaining UGI multifunctionality and well-being (Basnou *et al.*, 2020; Hansen *et al.*, 2014).

Likewise, the role of the socioeconomic and historical management of urban nature, influencing the provision and access to ES from an environmental justice perspective, is a developing research area that calls for further exploration. More research is particularly needed to explore how these processes influence ES provision and their cascading effects on human well-being (Leong *et al.*, 2018). Understanding how socioeconomic factors and past legacies contribute to shaping urban ecosystems, highlights the influence of structural socioeconomic inequalities on vegetation and biodiversity patterns, environmental quality, and the burdens affecting urban socio-ecological systems. Recognising and studying the consequences of luxury (wealth-related) and legacy effects (historical management) on the spatial and biophysical attributes of UGI can contribute to the development of equitable and just management practises and strategies for ES provision. This perspective further allows to evidence the cascading effect of the provision of ES on human well-being and protection from negative environmental impacts. Advancing knowledge about the role of biodiversity in UGI quality for ES provision is crucial for enhancing residents' equal access to ES and related well-being, as protection from environmental burdens emerging from the Anthropocene, such as urban heat hazards.

## 1.2. Research objectives and structure of the dissertation

Building on the background and research gaps highlighted above, the general aim of this dissertation is to analyse the role of biodiversity in sustaining the multifunctionality of urban green infrastructure in terms of the capacity to provide ecosystem services from an environmental justice perspective. This general goal was structured on the basis of three specific objectives, each of which was associated with a chapter (from II to IV).

**Objective 1 (Chapter II):** Evaluate the spatial patterns of urban wildness and habitat quality and their role in predicting urban biodiversity across different UGI types in a mid-sized city.

This, to answer the following research questions:

How are the spatial patterns of urban wildness and habitat quality associated with the biodiversity distribution of UGI across a mid-sized city?

What biophysical attributes of UGI influence spatial patterns of urban wildness and habitat quality?

Which spatial distribution and biophysical attributes of the UGI should be considered to enhance urban biodiversity?

**Objective 2 (Chapter III):** Examine how socioeconomic and historical factors shape UGI and ES from an environmental justice perspective.
This, to answer:

What are the roles of historical and socioeconomic factors in the distribution and current status of urban biodiversity and vegetation cover?

How do socioeconomic and historical factors impact the provision of ES, and how are they associated with additional factors such as population density, vegetation cover, and habitat quality?

**Objective 3 (Chapter IV):** Analyse the mismatches in supply and demand of ES, with a specific focus on temperature regulation provided by UGI, and identify the factors driving distributional green inequities.

This, to answer:

How do the biophysical attributes of the UGI affect the distribution of ES?

Which are the priority intervention areas arising from identifying spatial mismatches between the supply and demand of temperature regulating ES?

What socio-demographic and health factors contribute to mismatches between the supply and demand of urban ES, leading to environmental inequities?

These objectives are addressed sequentially in the three original stand-alone research papers included in this dissertation (Chapters II-IV). A summary of the research chapters is indicated in Table I, whose conceptual framing is presented within the general theoretical framework guiding this dissertation.

## 1.3. Theoretical framework

### 1.3.1. Urban ecology

Classical ecological studies have mainly focussed on "natural ecosystems" with low or minimal human influence to study the biological, biogeochemical, and evolutionary basis of the interactions between organisms and their environment (*sensu* Hackel 1866 in Kingsland 1991). These studies have focussed more on natural processes, portraying humans as external agents of the natural world and thus not on the direct interactions and interdependence between humans and the environment (Lambert & Schell, 2023; Liu *et al*., 2007). The more recent field of urban ecology has built on these classical studies by incorporating humans with a vital role in ecological processes, not merely as external agents but as integral contributors shaping system complexity (Pickett *et al*., 2013; Niemelä *et al.*, 2011). Urban ecology moves from a more classical ecological perspective of 'the ecology in cities', to acknowledging urban areas as complex socio-ecological systems (*i.e.*, 'the ecology of cities') (Pickett *et al.*, 2016; Grimm *et al.*, 2008). This approach advocates a comprehensive analysis of the urban system, integrating social structures, biophysical processes, and their interactions as the main drivers of urban structure and functioning (Alberti, 2024; De Roches *et al.*, 2020; Pickett *et al.*, 2016). This integrated study of urban environments as socio-ecological systems emphasises the fundamental role of human-nature interactions in shaping human well-being. Humans are key species in urban systems, and their motivations and perceptions regarding urban nature, as well as the dependence on nature for human well-being, will influence how nature is shaped in cities (De Roches *et al.*, 2020; Smith, 2007). Thus, the study of nature in culturally modified landscapes such as cities, can help to reconnect people with nature, avoid the extinction of the experience, and emphasise the role of nature for ES and human well-being while enhancing urban ecological quality and biodiversity conservation (Soga & Gaston, 2018; Kowarik, 2018). For instance, understanding how human-driven environmental changes (*e.g.*, intensified heatwaves) affect human well-being involves analysing the integrated biophysical factors, built infrastructure, social structures and human responses (Pickett, 2016; Pickett & Cadenasso 2008). This is associated with the integration of urban design and ecology to achieve urban sustainability and climate goals (*i.e.*, 'the ecology for cities') (Pickett *et al.*, 2016; Childers *et al.*, 2015) or into the broader goals of ecological integrity and environmental justice (Pickett *et al*., 2016; Rozzi *et al.*, 2015).

### 1.3.2. Ecosystem services cascade framework

The ES cascade is a useful analytical framework to connect into a process flow how the biophysical structures and processes of the ecosystems influence the provision of ES and their effects on human well-being (Potschin-Young *et al.*, 2018; Haines-Young & Potschin, 2010) (Fig. 2). This model emphasises the multifunctional character of UGI and allows us to understand how their design, distribution, structure, and maintenance influence ES provision and ultimately human well-being. This model consists of five interconnected components, providing a structured understanding of ecosystem dynamics: 1) the biophysical structure or processes, covering both biotic and abiotic elements shaping the ecosystem, 2) their transition into ecological functions, and sustained processes defining 3) the ecosystem's capacity to provide ES, 4) the positive impacts or benefits that contribute to the well-being and quality of life of residents, and 5) the values resulting from ES, influenced by economic, social, and cultural factors (Villamagna *et al.*, 2013; Haines-Young & Potschin, 2010). To result in human well-being and societal values, the ES provision must be related to the demand, referring to societal needs for ES (Dworczyk & Burkhard, 2021; Villamagna *et al.*, 2013).

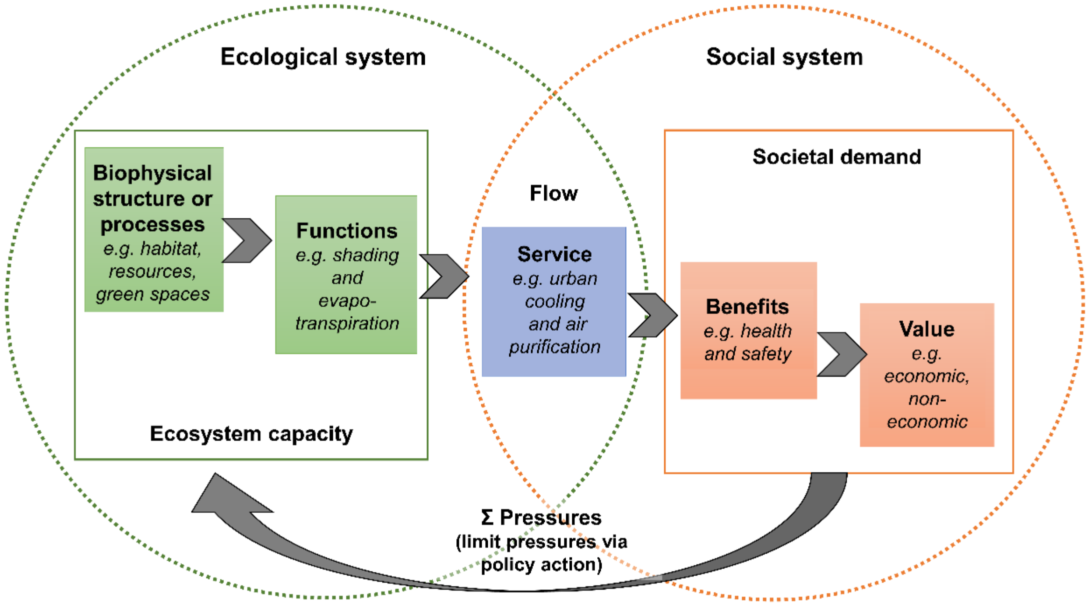


**Figure 2**. Ecosystem services cascade framework, from ecological to social systems: the ecosystem capacity influences the flow of ES to the societal demands of such services. Adapted from Potschin-Young *et al.* (2018).

The ES cascade framework reinforces the crucial role of ecological processes in ES provision while quantifying the relationship between supply and demand (Potschin-Young *et al.*, 2018; Schröter *et al.*, 2017). Furthermore, it integrates feedback mechanisms between its different components, *e.g.*, the value of a specific ES can influence decisions in UGI planning and management, thus affecting the biophysical structure and processes to increase the production of this ES (Haines-Young & Potschin, 2010). Therefore, the ES cascade is a systematic guide that can be used to unravel the complex relationships shaping urban ecosystems towards achieving human well-being and sustainability through effective ES management (Potschin & Haines-Young, 2018). This framework guides various aspects of ES assessment, including the mapping of ecosystem capacity, flow, and benefits (Maes *et al.*, 2012; 2016), making them useful for decision-making support, *e.g.*, to develop strategies for integrating multiple values that enhance human well-being (Pascual *et al.*, 2023; Baró *et al.*, 2015).

The connection between the ES capacity and flow indicates potential overuse when the flow exceeds the capacity (Schröter *et al.*, 2014). In contrast, considering the relationship between flow and demand provides information about potential unsatisfied demand if the demand exceeds the flow (Dworczyk & Burkhard, 2021; Geijzendorffer *et al.*, 2015). These potential imbalances contribute to the assessment of ES (mis)matches, indicating whether the intake is sustainable and whether the demand is met. ES mismatches arise from disparities in either the quality or quantity between ES capacity, flow, and demand (Geijzendorffer *et al.*, 2015). Despite the value of the ES cascade model in identifying mismatches, existing research has mostly focussed on the supply side, often overlooking the multifaceted nature of ES demand, including the diverse ways in which these services are valued (Langemeyer & Connolly, 2020; Basnou *et al.*, 2020; Wolff *et al.*, 2015).

Analysing mismatches between ES supply and demand within the ES cascade framework is key to inform strategic planning and management of UGI. This approach provides insight into the influential roles played by socioeconomic and historical contexts in shaping socio-ecological outcomes related to UGI. Examining the distributional patterns of ES mismatches allows to identify areas where certain communities may experience disparities in the quality and quantity of ES, thus revealing potential environmental injustices. This framework serves as a valuable tool for breaking down the processes that mediate these outcomes, enabling the integration of an environmental justice perspective into ES research (Calderón-Argelich *et al.*, 2021). Despite the relevance of such analysis, there remains a notable gap in the literature, with few studies examining spatial mismatches between ES supply and demand at an urban scale for UGI (Sebastiani *et al.*, 2021; Herreros-Cantis *et al.*, 2021; Liu *et al.*, 2020; Baró *et al.*, 2016). While some contributions have explored environmental justice dimensions using factors such as distance to UGI, green cover, and socioeconomic attributes (Herreros-Cantis *et al.*, 2021; Liotta *et al.*, 2020; Baró *et al.* 2019), there is still a need to understand these spatial disparities and their potential implications for environmental justice.

### 1.3.3. Biodiversity for ES provision and environmental justice

Biodiversity is a key biophysical attribute of ecosystems that determines ecosystem functionality, contributing to the provision of ES and hence societal well-being (Murray *et al.*, 2022). This relevance of biodiversity becomes evident in UGI, where it supports multiple functions that result in ES and associated human well-being in a closely interconnected socio-ecological urban environment. Furthermore, exposure to UGI with high biodiversity has been associated with increased psychological and restorative health benefits (Carrus *et al.*, 2015; Dearborn & Kark, 2010; Fuller *et al.*, 2007). However, the positive influence of biodiversity on ES and human well-being can be hindered by its uneven distribution in society, often associated with socioeconomic factors such as income or educational level (Kuras *et al*., 2020).

Biodiversity patterns are shaped by systematic biases in human behaviour and resources across urban systems, which can be directly and indirectly associated with socioeconomic status. Therefore, the spatial variability of socioeconomic status in a city can further illustrate the hierarchical divisions and power dynamics that drive biodiversity across spatio-temporal scales (Schell *et al.*, 2020; Kuras *et al.*, 2020). This distributional imbalance often particularly affects the most disadvantaged and vulnerable communities, hindering their societal resilience to external shocks such as those arising from climate change (Lambert & Schell, 2023; Tozer *et al.*, 2020; Andersson *et al.*, 2019; Kabisch & Haase, 2014). The incorporation of the environmental justice perspective into urban ES research remains underdeveloped (Calderón-Argelich, 2021; Langemeyer & Connolly, 2020; Enssle & Kabisch, 2020). In particular, how the distributive spatial patterns of urban biodiversity can sustain an equal ES provision from an environmental justice perspective has received insufficient attention. To bridge this gap, ES-focused research should address the role of biodiversity in both the capacity and provision of ES while incorporating key components of ES demand, such as accessibility and values contributing to a good quality of life (Langemeyer & Connolly, 2020; Pascual *et al.*, 2017). Understanding how socioeconomic factors and historical legacies contribute to shaping biodiversity and thus the capacity to provide ES is a key input for management and greening planning for equity.

Different structural socioeconomic and historic inequalities influence UGI and associated biodiversity patterns in cities, ultimately mediating ES benefits (Schell *et al.*, 2020).The 'luxury effect' is defined as the role of wealth in predicting higher biodiversity (*e.g.*, vegetation diversity) and green space quality (Leong *et al.*, 2018; Hope *et al.*, 2003). Affluent residents and neighbourhoods, with greater economic and political capital, tend to steward habitat and biodiversity, which also allows them to have greater access to nature-related benefits (Lambert & Schell, 2023; Clarke *et al.*, 2013). Frequently, these initiatives are driven by the desire to maintain a specific neighbourhood appearance or character, a concept referred to as the "Ecology of Prestige" (Grove *et al.*, 2014). Likewise, public investment in specific areas can elevate property value and draw in affluent residents, displacing economically vulnerable residents who are no longer capable of maintaining the elevated costs of living in such areas (Anguelovski *et al.*, 2020). In contrast, disadvantaged communities with fewer resources consistently face neglect in investments that could foster greater species diversity, perpetuating environmental injustices (Wolch *et al.*, 2014). Addressing these disparities requires comprehensive strategies to tackle socioeconomic inequalities in urban environmental management and promote equitable access to nature-related benefits.

Historical urban development is also a key driver in shaping the distribution of biodiversity and related ES. The 'legacy effect' refers to how past management practises shape the current urban landscape, notably by enhancing vegetation biodiversity in older neighbourhoods (Roman *et al.*, 2018; Clarke *et al.*, 2013). However, old towns in European cities, characterised by narrow streets and limited public space, can contradict this effect. Past patterns of inequality influence which species are affected by affluence, upholding luxury effects over time, resulting in historical additive responses (Leong *et al.*, 2018). For instance, legacies of colonialism or residential segregation may aggravate environmental injustices over time, by a differential investment in UGI that can widen biodiversity differences across socioeconomic groups (Kuras *et al.*, 2020; Lambert & Schell, 2023). Effective UGI management requires assessing the distribution of UGI and biodiversity across cities is crucial to guarantee equitable access to ES and human well-being. The success or failure of biodiversity enhancement policies across UGI and multiple ES is contingent on the equitable distribution of urban nature and its associated benefits (Cohen *et al.*, 2012; Strohbach *et al.*, 2009).

## 1.4. Methodological approach

### 1.4.1. Analytical perspective

This section outlines the general methodological approach used in this dissertation to address the defined questions and objectives. Although the three empirical chapters function as stand-alone papers, they are thematically interconnected, following the process of the ES cascade while maintaining a perspective of environmental justice across its components. Each chapter provides insights into the different components of the ES cascade, building on complementary objectives and methodological approaches. Chapter II focuses on the quantification and mapping of the ecological quality of UGI by establishing the relationships between biodiversity and urban wildness (UW), habitat quality (HQ), and vegetation structure. Thus, it addresses questions related to "how much?" and "where?" regarding biodiversity as a proxy for the ecosystem capacity to support ES (Fig. 2; Hamel *et al.*, 2021). This chapter is the basis for the subsequent biophysical analyses on biodiversity related to UGI in the following chapters. Chapter III analyses potential drivers of biodiversity in urban nature, specifically examining socioeconomic and historical factors (luxury and legacy effects) influencing greening processes and the potential provision of ES. Chapter IV analyses how the distribution of urban nature influences the capacity to supply ES and contrasts with the demand for such ES by focussing on temperature regulation ES. Chapters III and IV integrate the ES cascade components of ecological systems with the social systems, with a focus on the production of ES, the drivers that constitute pressures to influence the ecosystem capacity to support ES, and the particular benefits from a specific ES regarding its demand-supply mismatches (Fig. 2). These last chapters further include an equity perspective, examining "to whom?" UGI mostly benefits in terms of ES provision and adaptation to environmental impacts (*i.e.*, urban heat hazard).

### 1.4.2. Methods

The analytical chapters of this dissertation were based on complementary modelling of socioeconomic and environmental primary and secondary data for Vitoria-Gasteiz as a case study.

Chapter II involved a comparative analysis of the performance of urban wildness and habitat quality as biodiversity indicators. To this end, I adapted the Habitat Quality (HQ) InVEST module for urban settings. 'Wild' areas were identified based on their relative wildness along a human modification *continuum* and assessed using GIS-based composite indicators. The InVEST model was adapted to evaluate HQ as a proxy for urban biodiversity, estimating the variability of habitat and vegetation types, and their ecological integrity. In this process, I integrated expert-based knowledge for model parametrisation. This was performed through structured surveys to international experts on urban ecology, biodiversity, and urban environmental management. They were asked about habitat suitability and potential threats to ecological integrity regarding spatial data on UGI typologies. The validation involved testing the relationship between habitat quality and urban wildness using Generalised Linear Models (GLMs) and comparing estimated values with biodiversity data from the citizen science platform "ornitho" for butterflies, birds and mammals. The outcomes of these models were used as indicators for the biodiversity components supporting the ecosystem's capacity to provide ES in the subsequent chapters of this dissertation.

Chapter III examined the 'luxury' and 'legacy' effects on urban biodiversity, vegetation cover, and ES provision in Vitoria-Gasteiz. To this end, the HQ outcomes from Chapter II were incorporated as an additional measure of urban biodiversity, and complemented with local censuses for birds and public tree canopy data. Spatially explicit modelling methods were used to analyse the links between urban biodiversity, ES support and sociodemographic variables used as proxies for luxury and legacy effect. These methods included machine learning for classifying built and vegetation cover types through remote sensing or the construction of spatial indicators for census data on biodiversity and sociodemographic data at the neighbourhood level. Furthermore, process-based modelling approaches, such as the iTree Eco tool, were employed to quantify the flow of regulating ES derived from public canopy cover. The environmental and ES descriptors were related to the sociodemographic variables used as proxies for luxury and legacy effects by GLMs that evaluated the relationships and multiple interactions among the variables of interest.

Chapter IV analyses the potential mismatches in access to UGI – ES provision and the demand for these ES by focussing on urban heat hazards. This study further intends to identify environmental injustices derived from the unequal distribution of ES regarding social demand. In this study, the methods integrated GIS tools, using composite indicators to quantify temperature regulation ES supply, and the societal need to reduce heat-related vulnerability as the demand. For ES supply quantification, I combined the capacity and flow components from the ES cascade and remote sensing information (*e.g.*, canopy cover, and NDVI) and spatial models (HQ indicator from Chapter II). For ES demand, defined here as the societal need for vulnerability to urban heat reduction, composite indicators account for vulnerability as a demand measure and a function of exposure, sensitivity, and adaptive capacity. To operationalise the supply-demand mismatch assessment, a modelling workflow integrated into Artificial Intelligence for Environment and Sustainability (ARIES) is employed. GLMs and multivariate methods were used to examine identified mismatches, revealing the variables that contributed most to spatial mismatches and distributive green injustices.

### 1.4.3. Case study description

This dissertation was focussed on the middle-sized city of Vitoria-Gasteiz, located in the Basque Country on the northern Iberian Peninsula, as a case study (Fig. 3). Recognised as the 2012 European Green Capital, Vitoria-Gasteiz is home to a population of 253,672 inhabitants (INE, 2022). Located in a transitional biogeographic region, connecting the Mediterranean and Euro-Siberian zones, the city is surrounded by the Cantabria and Pyrenean mountain ranges, serving as biodiversity hotspots and contributors to the Pan-European Ecological Network (De la Hera, 2019). The city encompasses an extensive plain with urban and periurban green and blue spaces, featuring a 35-km-long and 731-ha green belt established in the 1990s to restore peripheral degraded areas while slowing urban expansion (Newman & Cabanek, 2020). This strategy aimed at enhancing biodiversity through habitat restoration and ecological connectivity. Features within the green belt, such as the Salburua wetlands, Zadorra River, and Vitoria Mounts, are safeguarded as community interest sites under the European Natura 2000 network. The city harbours more than 381 different species of trees and shrubs, with more than 130,000 trees distributed around the city and 12,160 shrub masses. The City Council of Vitoria-Gasteiz has a strategic Green Infrastructure plan (CEA, 2014) designed to enhance the city's urban green infrastructure and biodiversity. Furthermore, it has promoted initiatives and policies towards sustainable development, such as an ecological network connecting urban and peri-urban areas (Europarc, 2018). With 20 $m^2$ of urban green spaces per inhabitant, Vitoria-Gasteiz stands as a model "green" city (Neidig *et al.*, 2022), with a balanced distribution of green recreation areas that can be accessed within a radius of 250 m from any point within the residential grid (Orive & Lema, 2012). This sustainability effort and consistent strategy have been internationally recognised, being awarded as European Green Capital in 2012 (Albino & Dangelico, 2013). However, despite these efforts, according to the Biodiversity Conservation Strategy of the municipality, the biological diversity of the flora and fauna of the urban parks has decreased significantly (CEA, 2014). The city's commitment to ambitious greening strategies makes it an interesting research site, providing abundant environmental and sociodemographical data, including key spatial information for our methodological approaches.

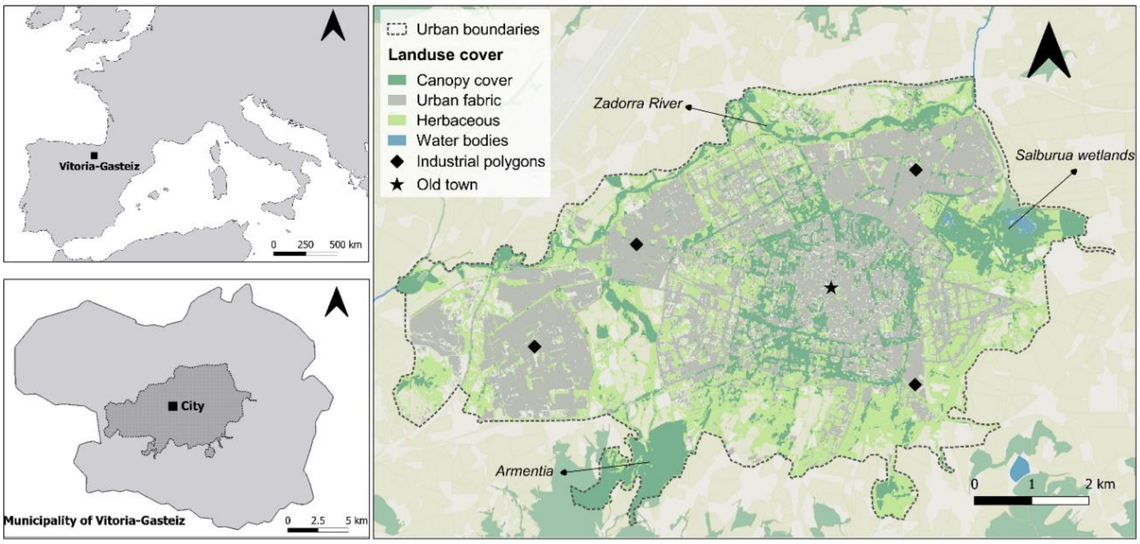


**Figure 3.** Study area of Vitoria-Gasteiz with its location within the urban and municipal boundaries and main land covers, such as urban fabric, canopy cover, herbaceous cover, and water bodies. Black diamonds indicate industrial polygons and the black star indicates the historical city centre. Arrows indicate the major natural landmarks of the green belt.

### 1.4.4. Dissertation structure

This dissertation is organised around three stand-alone research papers, each dedicated to addressing specific objectives (presented in section 1.1), aligned with the overarching aim of this dissertation. Presented individually in Chapters II-IV, these papers form the core contribution of the dissertation, with a final synthesis and discussion provided in Chapter V. While each chapter is independently readable, some partial overlap with the contents of this introductory chapter is expected. The empirical chapters are interconnected, forming a cohesive narrative that explores socio-ecological modelling purposes across different spatial scales. This approach provides a comprehensive exploration of urban landscapes as complex socio-ecological systems, with Vitoria-Gasteiz as a case study. At the time of submitting this dissertation, two research articles have been published (Chapters II and III), and Chapter IV is currently under review for potential publication. All chapters have been developed under my lead, with contributions from other collaborators. The publication status and the list of collaborators for each chapter are outlined on the first page of the respective chapters. An overview of the main characteristics across the three empirical chapters is provided in Table I.

In addition, the dissertation includes supporting information in Supplementary Materials 1, 2, and 3, which list additional research achievements and activities carried out during the PhD period at the end of this dissertation.

**Table I.** Research chapters' summary overview

| | **Chapter II** | **Chapter III** | **Chapter IV** |
|---|---|---|---|
| **Title** | Wildness and habitat quality drive spatial patterns of urban biodiversity | Luxury and legacy effects on urban biodiversity , vegetation cover and ecosystem services | Ecosystem services mismatches evidence inequalities in urban heat vulnerability |
| **Objective addressed** | Evaluate the spatial patterns of urban wildness and habitat quality, and their suitability to predict urban biodiversity across urban green infrastructure | Examine how socioeconomic and historical factors shape urban nature and ES integrating an environmental justice perspective | Analyse potential supply and demand mismatches in the access to ES provided by UGI and identify which factors are driving distributional injustices |
| **Geographical scope (spatial grain)** | City wide – urban landscape (Vitoria-Gasteiz) | Neighbourhood (Vitoria-Gasteiz) | Census tract (Vitoria-Gasteiz) |
| **Methodological approach** | Expert-based survey<br>Composite mapping<br>InVEST model<br>Generalised linear models | Machine learning<br>iTree Eco tool<br>InVEST model<br>Generalised linear models | Remote sensing<br>Composite mapping<br>InVEST model<br>ARIES modelling<br>Multivariate analyses |
| **Ecological structures, functions and ES assessed** | Biodiversity<br>Urban wildnesss<br>Habitat quality | Biodiversity<br>Vegetation cover<br>Temperature regulation<br>Air purification<br>Runoff control<br>Carbon sequestration | Vegetation cover<br>Habitat quality<br>Temperature regulation |
| **ES components assessed** | Capacity | Capacity and Flow<br>Pressures to ES capacity | Capacity, Flow (indirectly) and Demand |
| **Ethical considerations** | Ethical approval by BC3 management committee.<br>Informed consent and signatures collected | Ethical approval was not required | Ethical approval was not required |
| **Publication status** | Published in *Landscape and Urban Planning* | Published in *npj Urban Sustainability* | Under review in *Science of the Total Environment* |

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# CHAPTER II

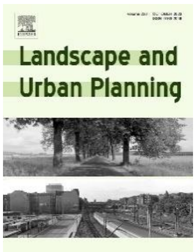
## Wildness and habitat quality drive spatial patterns of urban biodiversity


## Abstract:

Urban green and blue spaces (UGBS) are key for biodiversity conservation. Many studies focus on UGBS benefits for well-being, but how UGBS ecological and quality influence urban biodiversity is still poorly understood. We analysed the predictive accuracy of urban wildness (UW) and habitat quality (HQ) spatial patterns to biodiversity in the city of Vitoria-Gasteiz, Basque Country. Using GIS techniques, we mapped relative UW as a landscape quality, considering remoteness, challenging terrain, and perceived naturalness. We further evaluated HQ using the InVEST habitat quality module, including data on habitat sensitivity to threats (*e.g.* population density, light and noise pollution, accessibility) and suitability for biodiversity support, based on a parametrization by expert consultation. We compared UW and HQ to observed species richness obtained from crowd-sourced databases as a biodiversity proxy. UW and HQ models predicted general biodiversity urban patterns, being particularly adequate in UGBS. Peripheral UGBS were associated with higher UW and HQ and positively correlated to biodiversity, as opposed to the smaller-sized centrally located UGBS, more exposed to threats. Both predictors significantly explained biodiversity, and HQ better accounted for threat susceptibility in UGBS. Our findings suggest that small-sized UGBS, such as parks and squares, fail to effectively support urban biodiversity, due to their high exposure and vulnerability to threats, particularly in centric areas. Emphasizing efforts in larger centric UGBS with rewilding strategies (*e.g.* lowering management frequency) and reducing exposure to threats is essential to increase the habitat quality of UGBS and thus support urban biodiversity.



**Keywords:** Biodiversity conservation; green infrastructure; InVEST; Vitoria-Gasteiz, green and blue spaces; urban ecology



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## 2.1. Introduction

Given the continuous urban development around the world with anthropogenic activities increasingly shaping ecological processes and patterns, there is a growing need to enhance the multiple functions and benefits supplied by urban green and blue spaces (UGBS) (Mansur *et al.*, 2022; Reyes-Riveros *et al.*, 2021). UGBS supplies multiple functions and contributions to human well-being, including biodiversity support, urban cooling, runoff control, air and water quality improvement, opportunities for recreation and building social cohesion (Baró *et al.*, 2019; Díaz *et al.*, 2018; Gómez-Baggethun & Barton, 2013). There is growing evidence that exposure and access to UGBS areas encompass multiple health benefits: mental health, lowering disease and mortality risk, among others (Ribeiro *et al.*, 2021; Engemann *et al.*, 2019; Twohig-Bennett & Jones, 2018; Triguero-Mas *et al.*, 2017). By providing people with opportunities to connect with nature, UGBS are key to decreasing the 'extinction of experience' *sensu* Soga & Gaston (2016) and the alienation from nature, which are major drivers of biodiversity loss and environmental injustice (Noss, 2020; Lepczyk *et al.*, 2017; Soga *et al.*, 2016; Shanahan *et al.*, 2015; Dunn *et al.*, 2006).

UGBS have a key role in supporting biodiversity, by harbouring native and non-native species and thus promoting conservation both at regional and global scales (Lepczyk *et al.*, 2017). Yet, biodiversity patterns in cities rely on the size, quantity and quality characteristics of UGBS, ultimately influencing the provision of related health and well-being benefits (Houlden *et al.*, 2021; Marselle *et al.*, 2019; Lepczkyk *et al.*, 2017; Fuller *et al.*, 2007). Assessing urban biodiversity is, therefore, key to better understanding the role of UGBS in providing functions and contributions to well-being in urban socio-ecosystems. Even if biodiversity and UGBS are highly intertwined, the spatial and ecological characteristics of UGBS influencing urban biodiversity remain poorly understood (Marselle *et al.*, 2021; Schwarz *et al.*, 2017).

Urban planners usually perceive UGBS as multifunctional structures and it is often assumed that more UGBS will support more ecosystem services and host higher biodiversity (Schwarz *et al.*, 2017). But although urban areas have the potential to provide habitat and sustain wildlife (Russo & Holzer, 2021) as human-altered habitats, their ecological functioning is in continuous interaction with anthropogenic impacts. Moreover, the way societies conceive different ecological processes as beneficial or detrimental to human well-being depends on social-ecological contexts which ultimately determines how societies might value the conservation of urban biodiversity (Marselle *et al.,* 2021; Dunn *et al.,* 2006). Preserving and enhancing urban biodiversity and raising public awareness of its societal values requires an understanding of the drivers and effects of changes to urban biodiversity (Knapp *et al.,* 2021; Sallustio *et al.,* 2017; Terrado *et al.,* 2016). Thus, there is a clear need to analyse UGBS not only in terms of their quantity but also in UGBS quality attributes related to the conservation value of the naturalness present and their capacity to harbour urban biodiversity (Reyes-Riveros *et al.,* 2021; Lepczyk *et al.*, 2017).

UGBS may include areas with different management regimes, ranging from heavily managed formal UGBS such as urban parks to barely managed semi-natural and ruderal areas (Aguilera *et al.*, 2019). There is a growing interest in understanding how to reduce intensive management in UGBS and restore relatively 'wild', self-managing ecosystems to avoid and reverse urban biodiversity loss (Aguilera *et al.,* 2019; Müller *et al.*, 2018). Wildness is the relative quality of being wild or undomesticated, encompassing a broad spectrum of landscape contexts, anthropogenic influence and scales: from remnant patches of urban vegetation to vast pristine areas, disturbance regimes and biogeochemical processes (Noss, 2020). Wildness, as a landscape character can be spatially represented. Yet, it cannot be directly separated from non-wildness, but rather expressed from less to more 'wild' along a continuum of anthropogenic impacts (Zoderer *et al.*, 2020; Carver & Fritz, 2016). There have been many recent wildness mapping studies (Müller *et al.*, 2015) almost exclusively referred to non-urban landscapes, often minimally affected by human action (Zoderer *et al.*, 2020; Radford *et al.*, 2019; Carver *et al.*, 2012, 2013). Even though only one study has applied wildness mapping in cities (Müller *et al.,* 2018), wildness can be a cost-effective indicator for management to be used in urban contexts: to enhance biodiversity, screening for priority conservation areas, while improving human well-being in cities (Mansur *et al.*, 2022; Jalkanen *et al.*, 2020; Noss, 2020; Kowarik, 2018; Müller *et al.*, 2018).

In addition, habitat quality (hereafter HQ) indicates the capacity of an ecosystem to deliver the resources and conditions needed for wildlife and is a key determinant of biodiversity (Terrado *et al.*, 2016; Hall, 1997). As a way to assess HQ, ecological spatial models are gaining research interest to evaluate habitat attributes, such as key resources and the constraints impairing the use of resources (*i.e.* anthropogenic influence). One of these models is the Habitat Quality module from the Integrated Valuation of Environmental Services and Trade-offs (InVEST HQ) which has been used to analyse the degradation status of non-urban landscapes (Di Febbraro *et al.*, 2018; Terrado *et al.*, 2016), to identify specific conservation targets (Bhagabati *et al.*, 2014), and to study natural or protected areas (Moreira *et al.*, 2018; Sallustio *et al.*, 2017). As biodiversity patterns are inherently spatial (Sharp *et al.*, 2020) the use of spatially explicit indicators, such as urban wildness or HQ can play an important role in predicting the suitability of a landscape to host biodiversity and ecological functions. The use of these models is cost-effective, can be done remotely and can be linked to characteristics of the ecosystem's structure and functioning, such as naturalness (Müller *et al.*, 2018; Di Febbraro *et al.*, 2018). Although the application of InVEST HQ model has proved to be useful to assess biodiversity conservation status at large scales, it often oversimplifies the variability of habitat types at smaller scales as is the case of urban regions (Sallustio *et al.*, 2017). To our knowledge, InVEST HQ has not yet been applied to estimate HQ for urban land uses, which can be often considered in the model as threats to biodiversity instead of potential habitat sources (Wu *et al.*, 2019; Han *et al.*, 2019; Sallustio *et al.*, 2017). Although wildness and HQ has been extensively assessed in large scales and often low anthropized areas, there are limitations on their applications at a finer resolution in smaller geographical areas (Cao *et al.*, 2019; Sallustio *et al.*, 2017). Areas with high land cover heterogeneity such as urban areas are usually oversimplified in landscape assessments of wildness and HQ, by homogenizing land uses and habitats types at coarser scales (Cao *et al.*, 2019; Sallustio *et al.*, 2017). This may lead to the underrepresentation of locally important areas for biodiversity conservation and affect management opportunities to enhance naturalness and wildness. Cities are increasingly important areas worldwide and UGBS are key on the relationship societies – nature, but have traditionally been seen as opposite to wilderness. Therefore, new approaches are necessary to better address the opportunities and challenges associated with rewilding UGBS in cities. Identifying factors influencing the biodiversity and their relationship with wildness and habitat quality at smaller scales at the urban end of the wilderness *continuum* is a research avenue that calls for further exploration (Carver & Fritz, 2016; Dymond *et al.*,2003; Kowarik, 2011).

Research on urban ecology is usually mostly focused on large metropolitan regions and capital cities, despite the fast growth and dominance of small and medium-sized cities in many regions such as Europe (Borsekova *et al.*, 2018; Boulton *et al.*, 2018). Hence, the assessment of how UGBS and its capacity to harbour biodiversity in mid-sized cities is still understudied (Boulton *et al.*, 2018). Here, we analyse the suitability of the spatially-explicit HQ and wildness models, as predictors of biodiversity when adapted to a mid-size urban area. Particularly, we: (i) spatially assessed relative urban wildness as a landscape quality following Müller *et al.* (2018); (ii) adapted the InVEST habitat quality module for the assessment of habitat quality on the urban landscape, considering the effects of anthropogenic impacts on the urban landscape and particularly across UGBS; (iii) identified potential UGBS that may have a key role in terms of biodiversity maintenance by using the outputs from (i) and (ii); and (iv) analysed the correlation between wildness, HQ, and species richness as biodiversity proxies.

## 2.2. Materials and methods

### *2.2.1. Study area*

We conducted our study in Vitoria Gasteiz, a middle-sized European city, and the administrative capital of the Basque Country (248.087 inhabitants, Eustat, 2020), located in the North of the Iberian Peninsula (Fig.1). Vitoria-Gasteiz is situated at the centre of a transitional biogeographic region (Mediterranean and Eurosiberian) (De la Hera, 2019) with an extensive plain delimited by mountains connecting the Cantabria (West) and the Pyrenean (East) mountain ranges, two biodiversity hotspots maintaining the ecological connectivity of northern Iberia, and part of the Pan-European Ecological Network (CEA, 2014). Vitoria-Gasteiz has a remarkable share of urban and periurban green and blue spaces (Fig.1), including a 35 km-long green belt surrounding the core city (Orive & Lema, 2012). The green belt was created during the 1990s as a way to transform peripheral degraded areas into UGBS along with enhancing biodiversity by restoring habitats and increasing ecological connectivity (Newman & Cabanek, 2020). It consists of a system of peri-urban semi-natural green spaces, promenades, wetlands, streams, and ponds in the interface between the city and the countryside (Monclús, 2018). Because of their natural value, Salburua wetlands, Vitoria Mounts and the River Zadorra have been protected as community interest sites and included as part of the European Natura 2000 network (CEA, 2014, Fig.1). With 20 $m^2$ of UGBS per inhabitant, Vitoria-Gasteiz is considered a “green” city (CEA, 2014). Because of its pioneering and ambitious greening strategies, Vitoria-Gasteiz gained international recognition and was awarded in 2012 as a European Green Capital (CEA, 2014). In addition, Vitoria-Gasteiz offers a great test-bed for research given the high availability environmental data, including the spatial data required in our modelling approach.

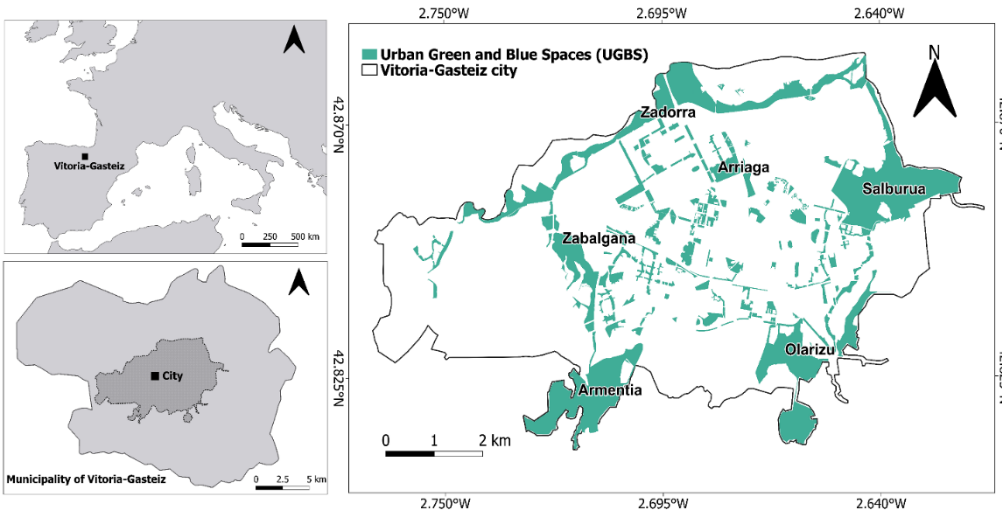


**Figure 1.** Location of Vitoria-Gasteiz municipality, city and main urban green and blue spaces within city boundaries defined by urban land uses and delimited by the green belt.

### *2.2.2. Urban wildness spatial model*

We mapped urban wildness (UW) at the city level along a *continuum* of historically human-modified landscapes, based on three parameters building on Carver *et al.* (2013), Müller *et al.* (2015), Müller *et al.* (2018), and Radford *et al.* (2019): i) Perceived naturalness, ii) Remoteness and iii) Challenging terrain.

Perceived naturalness was defined as the vegetation and land cover pattern created by land (un)management, appearing *natural* to the casual observer (Carver *et al.*, 2013). To estimate perceived naturalness for the whole study area, we joined the following spatial datasets: Urban Land Use Map (2020), Green Belt Land Use Map (2020), and Nature-Based Solutions Inventory (2020) (Table S1, Supplementary material) on a composite land use dataset (hereafter ‘land use map’) at a resolution of 10 m. Land uses were reclassified into 18 naturalness classes following Müller *et al.* (2018) (Table S2, Supplementary material). Naturalness values ranged from 1, representing ‘completely sealed areas’, to 18 as ‘land cover under the least human influence’. All datasets were provided by the Environmental Studies Centre (CEA), a public agency of the Vitoria-Gasteiz City Council. Brownfields and vacant land were classified based on the perceived naturalness as "Recreative areas" or "Relatively extensive open landscapes" according to the reconversion plan or use assigned for each area in the inventory of NBS provided by the City Council.

Remoteness was mapped by combining distance from mechanised access (shortest walking distances from main roads) and noise exposure to any pixel on the map as in Carver *et al.* (2013) and Müller *et al.* (2018). We reclassified the previous composite map based on the land use data into a cost surface building on Müller *et al.* 2018 (Table S3, Supplementary material). We estimated the time it takes a pedestrian to move across each pixel based on the assumed travel times for each land use type. The minimum time needed by a pedestrian to access any pixel in the study area from a vehicle access point was then calculated by the Path distance tool of ArcGIS. The weighted means of noise values (Lden) retrieved for the study area (CEA, 2017) were used for the calculation of noise exposure. The pixels without noise data were assigned 30 dB, comparable to a quiet garden (Müller *et al.*, 2018). We reclassified these into inverse values, so high values show low decibel values making sure that the indicator correlated positively to UW. Next, we combined both datasets, remoteness from mechanised access and noise exposure to create the final remoteness map.

Challenging terrain was depicted by combining terrain ruggedness index and occurrence of wetlands, again building on previous approaches (Müller *et al.*, 2018; Carver *et al.*, 2013). The ruggedness index was considered as the standard deviation (SD) of terrain curvature within a 250 m radius of the observer since it has been generally assumed that people perceive that area of their surroundings (Müller *et al.*, 2018; Scottish Natural Heritage, 2014; Carver *et al.*, 2012). We calculated the ruggedness index from a high-resolution (10 m cell size) digital elevation model (DEM) data available for the study site (CEA, 2017), with a higher score indicating steep and rough terrain. At this resolution, the DEM would also capture anthropogenic infrastructure (*e.g.* buildings), so when calculating the SD of the terrain curvature such buildings would tend to return high values (Müller *et al.*, 2018). To avoid the misleading effect of anthropogenic infrastructure on the index, we first excluded all pixels containing built infrastructure (*e.g.* roads, railways, buildings), from the dataset, and then the SD was calculated for each pixel using a 250 m neighbourhood buffer. As a previously identified limitation, the method captures ruggedness rather than the challenging character of the terrain more generally, so flatter areas such as wetlands and bogs tend to be underestimated (Müller *et al.*, 2018; Scottish

Natural Heritage, 2014). Thus, to cover all challenging terrain in terms of topographic attributes, we considered the occurrence of wetlands from the land use map to the terrain ruggedness index. Therefore, if a pixel cell led within a wetland, we added the average SD of terrain ruggedness to the pixel cell value (Müller *et al.*, 2018).

We obtained the final relative UW map by Vitoria-Gasteiz by combining these three indicator maps (perceived naturalness, remoteness, and challenging terrain) using equal-weighted simple addition. We then rescaled each indicator map and the resulting map to a 0–1 range using Eq. (1) following Carver *et al.* (2013) and Müller *et al.* (2018):

$$Sij = \frac{(Xij - OmV)*(NMV - NmV)}{(OMV - OmV + NMV)} \quad (1)$$

where *Sij* refers to the standardized value of cell *j* in map *i*, *Xij* to the current value of cell *j* in the map *I*, *OmV* to the old minimum value, *OMV* to the old maximum value, *NmV* to the new minimum value, and *NMV* to the new maximum value.

We performed a Jack-knife analysis to assess the sensitivity - robustness and weigh the uncertainty of the final UW model by checking the effect that leaving one input variable out of the model would have on the results (Quenouille, 1956). We calculated three alternative maps by excluding one input map, respectively. Then we calculated the correlation coefficients of the three resulting maps when compared to the original (full) UW map by using the r.regression.line module in GRASS 7.8.5 (QGIS.org, 2021). All the spatial analysis was performed in QGIS 3.18.2 (QGIS.org, 2021) and ArcGIS 10.7.1 (ESRI, 2019).

### *2.2.3. Habitat Quality spatial model*

We calculated the indicator for habitat quality (HQ) using the spatially-explicit InVEST Habitat Quality Model (v.3.8.9; Stanford University, Stanford, CA, USA) which estimates the relative degradation extent and status of different habitat types in a given region. This model is based on the relative sensitivity to different threats, distance to these potential threats, and location of protected areas. The HQ approach assumed that habitats with higher quality will support higher biodiversity and is a general estimator of biodiversity suitability (Sharp *et al.*, 2020). Therefore, we estimated HQ as a function of i) the suitability (*Hj*) of each land use to provide habitat for biodiversity (as local species richness), ii) anthropogenic threats which might impair HQ, and iii) the sensitivity of each land use type to each threat. HQ models can be used for a particular species or general patterns of biodiversity associated to habitat quality, regardless any particular taxonomic or functional group, which makes it a suitable rapid tool for general biodiversity assessment studies (Sharp *et al.*, 2020). Since not all habitats are affected by different threats, in the same way, we characterised the sensitivity of habitat types to various threats. We considered habitat suitability to be affected by: i) the relative impact of each threat (*Wr*, weighted relative importance of each threat), ii) the distance between habitat and the source of the threat (including the maximum threshold distance, *Max.D*), and iii) the relative sensitivity of each habitat type to the threat (*Sjr*). Further details on the InVEST HQ model and parameterisation can be found in Sharp *et al.* (2020), Sallustio *et al.* (2017), and Terrado *et al.* (2016).

Since our work was based on an urban context, the HQ models were adapted from the InVEST approach by considering the spatial distribution of habitats including a broad range of managed land use classes as potential habitat providers to urban biodiversity rather than merely threats (Sharp *et al.*, 2020). To define the potential habitats, we used the categorization of the 18 naturalness classes described in section 2.2. (Table S2, supplementary data). Instead of a binary (*i.e.* 'natural' vs 'unnatural') approach (Müller *et al.*, 2018), we considered the relative habitat suitability score for each land use class organized along a naturalness *continuum*. To account for threats, we used spatial information of railways and traffic intensity sorted by main, secondary, and residential roads (Vitoria-Gasteiz City Council, 2020). We also included raster information on noise pollution decomposed by dB intensity (Vitoria-Gasteiz City council, 2018) and population density data (Vitoria-Gasteiz city council, 2016) as additional potential threats.

A relative habitat suitability score *Hj* (scaled from 0 to 1), was assigned to each land use class, where 1 indicates land use classes with the highest suitability for biodiversity (Sharp *et al.*, 2020). *Hj* and all threat parameters were determined through expert consultation (following Kuhnert *et al.*, 2010). This information was gathered from a purposedly designed structured survey administered to 21 international experts, on urban ecology, biodiversity conservation, and urban environmental management who suggested values to parametrize our urban landscape model (see Questionnaire S1, in Supplementary material) (Tables S5 and S6).

Since the list of 18 naturalness classes was too extensive for an expert survey, we grouped these classes into 8 categories of urban habitats based on urban green infrastructure typology according to Hansen *et al.* (2017) (Table S4, Supplementary material). During the administering of the survey, we did not share any results or feedback among the group of experts as in Terrado *et al.* (2016), so our method relied on experts having a good understanding of the questions. To estimate the model uncertainty and to determine parametrization scores from each descriptor variable, we first calculated the mean (μ), standard deviation (SD), and coefficient of variation (CV). We then identified and excluded from the analysis extreme values in the valuation of any descriptors from the expert survey by z-score deviation. The HQ score uncertainty was calculated with the Zonal Statistic tool (QGIS) as the average HQ and their coefficient of variation for each habitat typology class (Di Febbraro *et al.*, 2018). The final values used as input parameters to build the HQM are reported in Tables S5 and S6 from the supplementary material.

The total threat level value in each raster cell $x$ of habitat typology $j$ was given by $D_{xj}$ (Eq. (2)), where $y$ indexes all grid cells on $r$'s map and $Y_r$ corresponds to the set of raster cells of $r$'s map. If $S_{jr}$ (relative sensitivity of each habitat type to the threat) equals 0, then $D_{xj}$ is not a function of threat $r$. Threats were normalized so the sum of all considered threats equals 1 (Sharp *et al.*, 2020):

$$D_{xj} = \sum_{r=1}^{R} \sum_{y=1}^{Yr} \left( \frac{Wr}{\sum_{r=1}^{R} Wr} \right) r_y r_{xi} \beta_x S_{jr} \quad (2)$$

The raster cell's degradation score was then translated to habitat quality ($Q_{xj}$) scores along with $H_j$ using scaling parameters ($z$ and $k$) in Eq. (3) (Sharp *et al.*, 2020). $Q_{xj}$ is equal to 0 if $H_j$ is equal to 0. $Q_{xj}$ can never be greater than 1. As raster cells degradation score increases, habitat quality decreases and vice versa:

$$Q_{xj} = H_j \left( 1 - \left( \frac{D_{xj}^z}{D_{xj}^z + k^z} \right) \right) \quad (3)$$

### 2.2.4. Spatial correlation with urban biodiversity data

We evaluated the spatial correlation of the UW and HQ models with observed biodiversity data considering the correlation coefficient (R), the adjusted coefficient of determination ($R^2$), and the regression of the residuals, with the 'r.regression.line' and 'r.regression.multi' modules in GRASS 7.8.5 (QGIS.org, 2021). Biodiversity data relates to observed species richness (as a proxy of alpha biodiversity, *i.e.* mean species diversity at a local scale (Wittaker, 1972)) from occurrence records of mammals, birds and butterflies obtained from the 'Ornitho.eus' database (www.ornitho.eus) for the period January 1994 to December 2020. Birds and butterflies are common core indicators of biodiversity according to the City Biodiversity Index (SCBD, 2012), and all groups selected are well represented across citizen science and monitoring programs in the city (Albaina *et al.*, 2020; De la Hera, 2019). Each record included a species occurrence at a specified geographic location along with the date and number of individuals. We first cleaned the species occurrence records retrieved from the 'ornitho' dataset by removing geospatial errors, duplicated, and museum registers. To correlate observed species richness with HQ and UW values, we first joined the cleaned species occurrences to a grid cell polygon layer of 100 x 100 m covering the whole study area. We then extracted the total count of species occurrences, species richness, and the mean values of our explanatory variables (*i.e.* UW and HQ) matching this grid cell scale. The responses of species richness to UW and HQ were analysed separately for each taxonomic group by Gaussian generalized linear models (GLM) first, for the whole city and then spatially constrained to UGBS. We selected the best models based on their statistical significance (F and p values $<0.05$) and goodness of fit (adjusted $R^2$), checking for normality and homoscedasticity through visual inspection of the residuals. To achieve a better fit for the whole city scale model, species richness was log10-transformed. To estimate the strength of the relationship between species richness, UW and HQ, we performed non-parametric Spearman's correlation tests. To test how different or complimentary were UW and HQ at predicting urban species richness, *i.e.* if the model had a detectable effect on richness, we regressed that variable (*e.g.* UW) against the residuals of related variables (*i.e.* richness as a function of habitat quality) and vice versa. All the statistical analyses were conducted in RStudio 1.4.1717 (RStudio Team 2021) and GRASS 7.8.5 (QGIS.org, 2021).

## 2.3. Results

We found that UGBS with higher UW and HQ values had the highest biodiversity values for all considered taxonomic groups. These UGBS were generally represented by large-sized peripheral areas, with high perceived naturalness, remoteness, challenging terrain (for UW), and high habitat suitability (for HQ). Both UW and HQ estimators were positively correlated to each other and associated with biodiversity.

### *2.3.1. Spatial patterns of urban wildness*

UW mapping showed that peripheral areas were associated with higher wildness values, and in general lower values towards the city centre (Fig. 2). Areas with higher UW corresponded to challenging terrain zones or with barriers to access, particularly wetlands, streams, foothill forests, and hills mostly located at the edges of the city and distant from the urban centre (*i.e.* Ramsar´s Salburua wetlands in the East and mountains to the South). Several large-sized UGBS, like large parks and freshwater ecosystems (*i.e.* rivers and wetlands), also showed relatively high UW even when located near the city centre (Fig. 2). Build-up infrastructure from residential and industrial areas was mapped as least wild, corresponding to 'completely sealed' or 'mostly sealed areas' according to Müller *et al.* (2018) classification. UW decreased towards higher impervious coverage and was particularly low in large industrialized with few or absent UGBS.


**Figure 2.** Urban wildness map: relative wildness for Vitoria-Gasteiz city on a 10 m resolution.

The sensitivity analysis of the UW model, comparing the differences in UW between the original (full inputs) and three alternative maps after excluding one of the three input parameters (perceived naturalness, remoteness and challenging terrain), showed a stable mapping trend and high correlation (correlation when excluding: Naturalness R = 0.75; Remoteness R = 0.97; Ruggedness R= 0.97) (Table S7, Supp Mat.).

### *2.3.2. Spatial patterns of urban habitat quality*

There was a clear urban-rural gradient in relation to the HQ spatial pattern, where most urban core areas were associated with low HQ values, while peripheral and large-sized UGBS showed the highest HQ to support biodiversity (Fig. 3). These areas mostly corresponded to the Ramsar wetland Salburua (to the East, Fig. 3), foothill forests and hills (to the South and West, Fig. 3). Streams showed high HQ values but were mostly restricted to the watercourse and not in the surrounding environment (*i.e.* floodplain). Furthermore, these ecosystems were particularly affected towards the city centre depicting very low HQ (Fig. 3). Vitoria-Gasteiz showed a large coverage of UGBS, mostly related to green corridors or parks near the city centre (*e.g.* Zabalgana and Park Arriaga, Fig. 1) that were however associated with low HQ values, some of them ranging between 0.12 to 0.15 (Fig. 3).

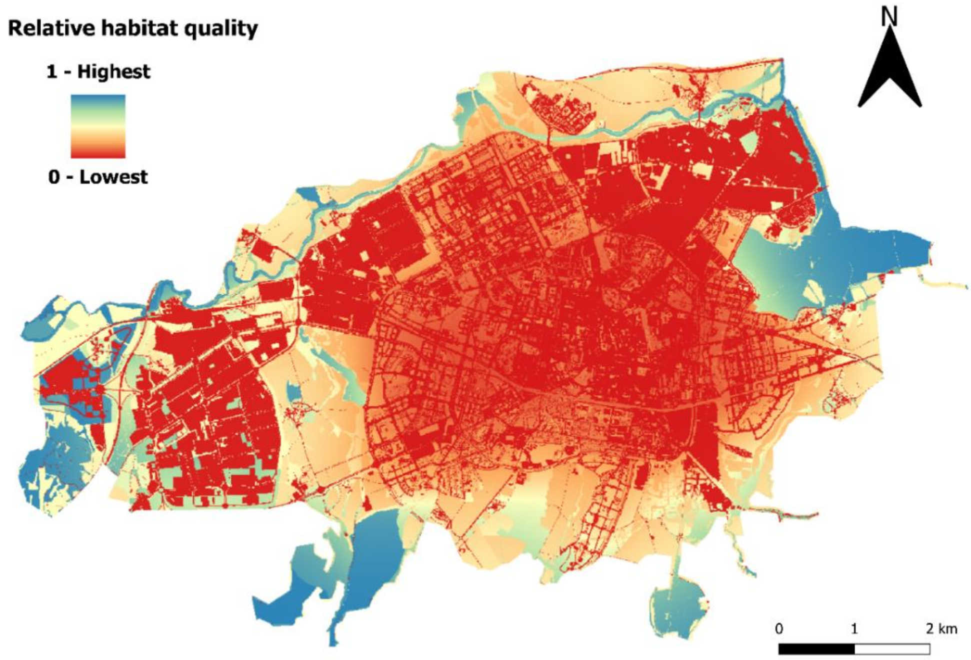


**Figure 3.** Urban habitat quality (HQ) map for Vitoria-Gasteiz city: Landscape-scale relative HQ mapping on a 10 m resolution. Habitat quality is based on a continuous scale from 0 to 1, where the lowest values indicate lower habitat quality (warm colours) and the higher values indicate higher habitat quality (cold colours).

The coefficients of variation (CV) that expressed the uncertainty for the expert-based parametrization of habitat suitability for HQ, showed that freshwater ecosystems, natural, semi-natural areas and feral areas were generally perceived as more natural and similarly valuated (CV freshwater ecosystems = 14%, CV natural, semi-natural areas and feral areas= 11%, Appendix S5). Whereas, the habitat suitability valuation uncertainty was much higher for more anthropized areas, and the highest level of uncertainty corresponded to predominantly sealed areas (CV= 238 %) (Appendix S5). Other land uses such as agricultural land, parks or grasslands were associated with intermediate habitat suitability CV values of 30%, 24% and 30% respectively (Appendix Table S5). Furthermore, the coefficient of variation concerning the threats for HQ was highly variable depending on the land use class and type of threat (Table S6).

### *2.3.3. Compared wildness and habitat quality spatial patterns*

Even if UW and HQ were positively correlated (Adjusted $R^2$= 65%) and explained a similar amount of biodiversity variation, they showed spatial differences for biodiversity estimations in particular areas, with some areas with a significantly higher valuation for UW than for HQ (in red Fig. 6) and vice versa (in blue Fig. 6).

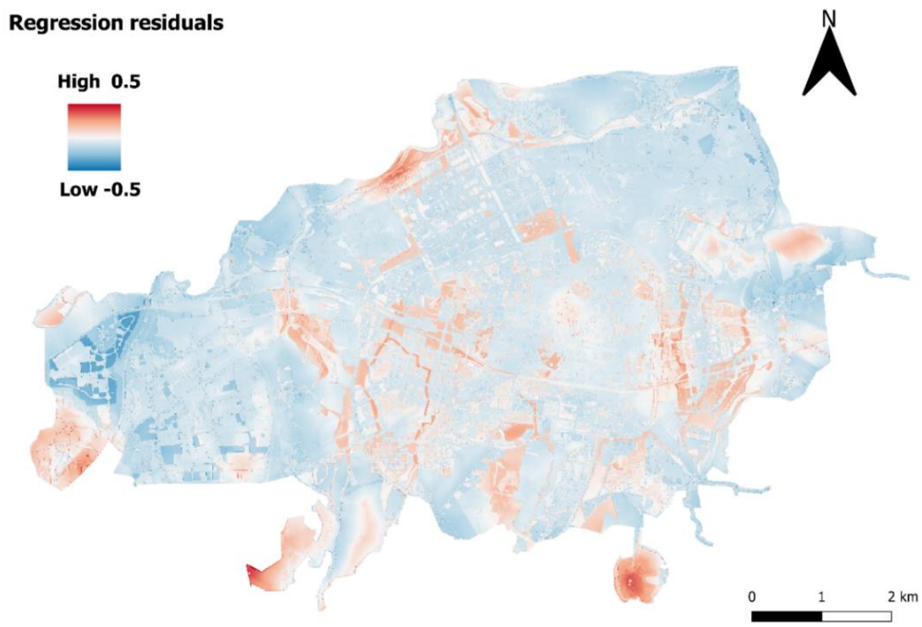


**Figure 6.** Comparative mapping of urban wildness (UW) and habitat quality (HQ) explaining urban biodiversity patterns. Red areas indicate higher UW values compared to HQ, while blue areas indicate higher HQ values compared to UW. The higher the colour intensity the higher the differences between UW and HQ. Residual map based on the regression of UW as a function of habitat quality (HQ).

UW depicted higher values than HQ in large-sized green spaces and peripheral areas including forests, hills, wetlands and rivers but also in centrally-located areas such as urban parks had higher UW values compared to HQ (in red Fig. 6). Particularly, freshwater ecosystems (river and wetlands) and the surrounding areas depicted higher UW compared to HQ (Fig. 7 in red). While in the case of HQ, peripheral and industrial areas showed higher values than UW (Fig. 7 in blue).

### *2.3.4. Wildness and habitat quality as proxies for biodiversity support*

The 'ornitho' occurrence registers for Vitoria-Gasteiz consisted of 50168 ind of 250 species of birds, 2721 ind of 48 species of mammals and 2063 ind of 97 species of butterflies. The most frequently observed bird species were mostly generalists and songbirds like *Pica pica* (Eurasian Magpie) or *Turdus merula* (Eurasian Blackbird), and the worldwide dominant *Passer domesticus* (House sparrow) (Fig. 4). The most common butterflies were *Pararge aegeria* (Speckled Wood), *Pieris rapae* (Small White) and *Colias crocea* (Clouded yellow). Among the most common mammals, we found *Sus scrofa* (Wild Boar), *Oryctolagus cuniculus* (European Rabbit) and *Vulpes vulpes* (Red Fox), mostly mid-sized generalists and omnivores. However, other mammals observed in the case study area include the endangered *Arvicola sapidus* (Southern Water Vole) and the critically endangered *Mustrela lutreola* (European mink) according to the IUCN red list (IUCN, 2021).

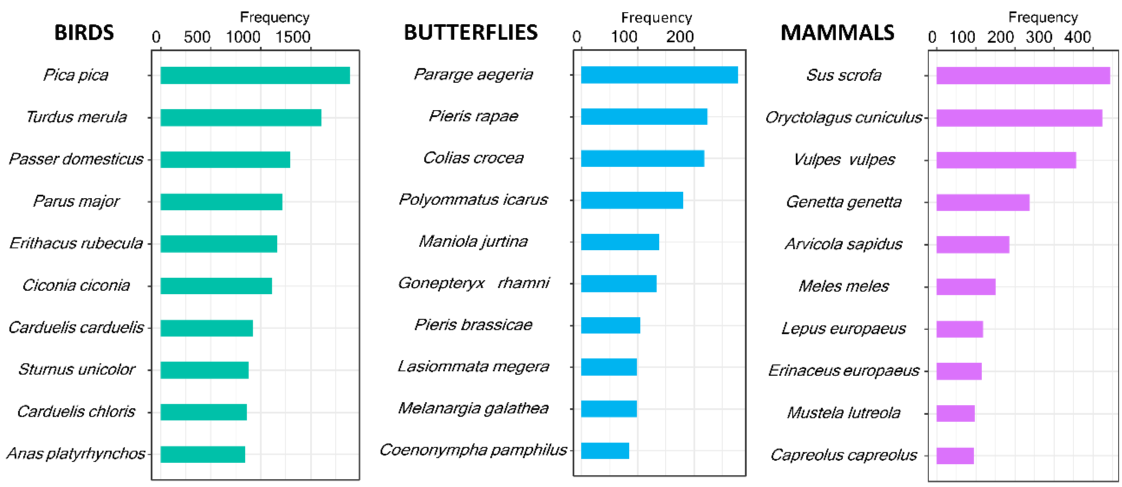


**Figure 4.** Ordered list of the 10 most common species of birds, butterflies and mammals, in the 'ornitho' biodiversity dataset as occurring in Vitoria-Gasteiz (year: 1994-2020).

Species richness of the different considered taxonomic groups (birds, butterflies and mammals) was positively correlated to both, UW and HQ values for the whole study area (Fig. 5 and Table S7 – Supplementary material). The Spearman's correlation for the strength of the relationship between species richness and UW, showed a strong positive correlation for birds ($p < 0.001$, $R = 0.26$), butterflies ($p < 0.01$, $R = 0.33$) and mammals ($p < 0.001$, $R = 0.34$), respectively. Similarly, the Spearman's correlation for richness and HQ, was also strongly and positively correlated for birds ($p < 0.001$, $R^2= 0.32$), butterflies' ($p < 0.01$, $R^2= 0.29$) and mammals ($p < 0.001$, $R^2= 0.37$).

The goodness of fit increased substantially when focusing our analysis on UGBS instead of the whole study area (Fig 5). Here, the relationship between species richness and both modelled predictors UW and HQ tested positive for all the considered taxonomic groups showing considerably higher correlation values and significance (Fig. 5).

The Spearman's correlation of richness against UW was positively correlated for birds ($p < 0.001$, $R^2= 0.39$), butterflies ($p < 0.001$, $R^2= 0.79$), and mammals ($p < 0.01$, $R^2= 0.61$). Likewise, species richness and HQ were positively correlated for birds ($p < 0.05$, $R = 0.2$), butterflies ($p < 0.001$, $R = 0.65$) and mammals ($p < 0.01$, $R = 0.62$). Overall, the richness of all considered taxonomic groups showed a consistent positive response to UW and HQ.

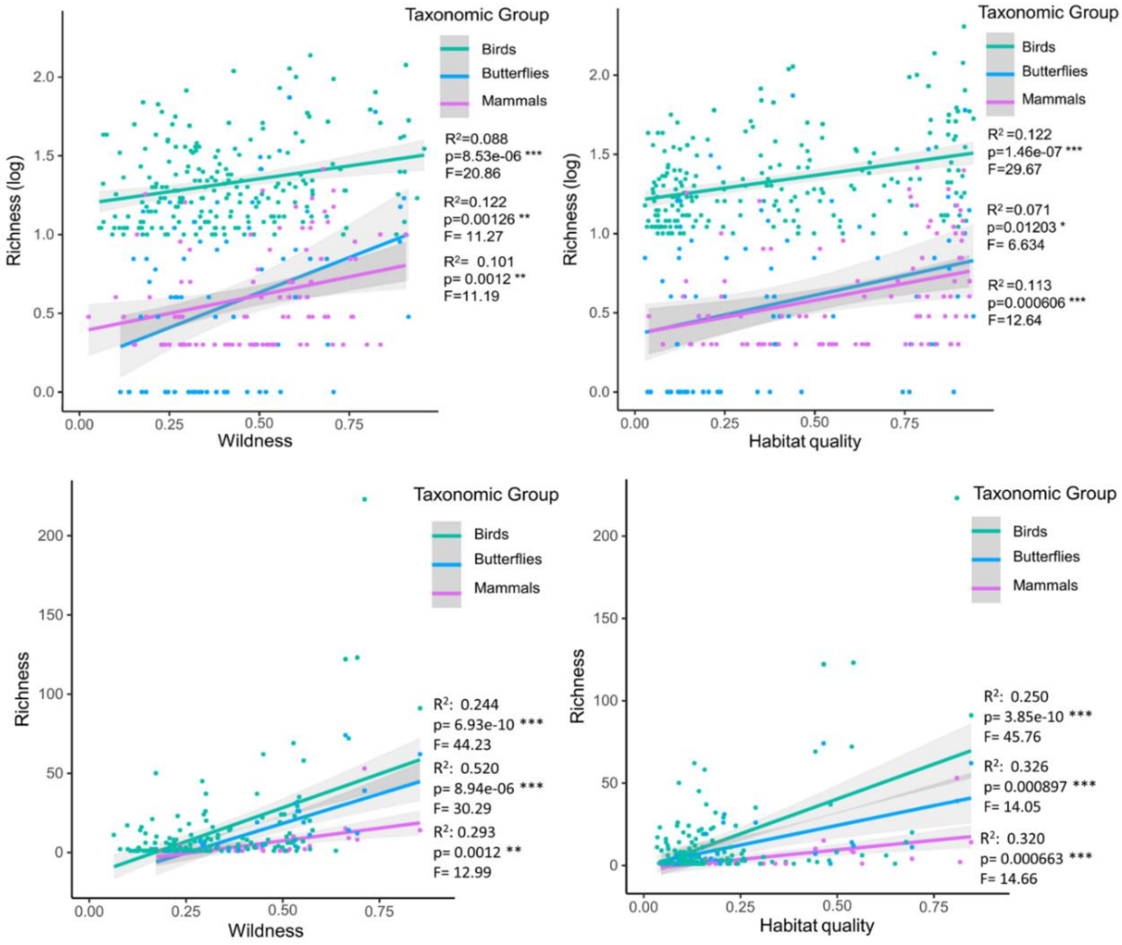


**Figure 5**. Upper panel: Whole-city patterns, regression analysis of richness against UW (left) and HQ (right), and fitted linear model results for each taxonomic group. Lower panel: Regression analysis of richness against UW and HQ constrained to UGBS and fitted linear model results for each taxonomic group ($R^2$ and significance values of $p < 0.05$ and confidence intervals (shaded area).

Residual regressions analysis showed that UW and HQ were not influencing richness independently, but rather capturing the same relations (Table S8, Supplementary material). All the different taxonomic groups used to test the relationship between HQ, UW and richness, showed a clear trend to increase as our indicators increased.

## 2.4. *Discussion*

### *2.4.1. Habitat quality and wildness to biodiversity*

Our results suggest that both UW and HQ models can be adequately adapted to urban environments to capture an area's capacity to support biodiversity. UW and HQ were significantly positively correlated with the biodiversity for all the considered taxonomic groups, and thus correlated to each other. The selection of these taxonomic groups was appropriate since they are common in urban environments and easily identifiable, making them suitable for citizen science programs. The most frequent species within each group, particularly in the case of generalists birds and small to medium-sized mammals, are in general highly recorded in crowdsourced databases due to their visibility and widespread presence in UGBS (Sultana & Storch, 2021). However, the biodiversity in UGBS is largely conditioned by the spatial and vegetal conformation of the UGBS (Reyes-Riveros *et al.*, 2021; Lepczyk *et al.*, 2017; Turner & Gardner, 2015). Some of the most frequent bird species found in the city such as magpies and blackbirds are commonly associated with weedy patches and native shrubs in central areas within the city and for which the amount of vegetation is a major determinant of density, diversity and distribution (Forman, 2014). Butterflies are usually the least species-rich in areas with intensive management and design such as the case of urban parks (Aguilera *et al.*, 2019; Forman, 2014). The list of frequent butterfly species observed in our case study, coincided partially with previous research showing the decline of species richness in butterflies related to management intensity (Aguilera *et al.*, 2019). In the case of large wild mammals, despite being less likely to be observed on an urban scale compared to birds and butterflies, the presence of mid-sized generalists within the city may be indicating that part of the UGBS may act as wildlife corridors, entering the city (Forman, 2014).

We found that UGBS located in the periphery of the city, generally large-sized, were associated with similar high range values for both, UW and HQ and therefore the highest potential to support urban biodiversity. These areas were mostly represented by, freshwater ecosystems, large-sized parks, and foothills, where human access and threats were lower, and they are currently protected as community interest sites in the Natura 2000 network, such as the Ramsar wetland Salburua and the Zadorra river (CEA, 2014). Parks and other residential UGBS, patchier, small-sized and centric (Fig. 1) were characterized by low values of both proxies of biodiversity support UW and HQ, being particularly low when considering HQ. We found contrasting patterns of UW and HQ values between these areas, suggesting that not only the location but the size and shape of UGBS may have an important role in influencing biodiversity (Fig 2 and 3). These findings are consistent with previous research highlighting the positive relation between UGBS characteristics on species richness, as size and habitat quality of UGBS in cities are key local factors to biodiversity (Aronson *et al.*, 2017; Lepczyk *et al.*, 2017; Goddard *et al.*, 2010). Larger UGBS with a lower perimeter:area ratio have less border effect, *i.e.* are less exposed to threats and effects of the surrounding cover types (Turner & Gardner, 2015). This may explain why larger areas had both higher UW and HQ: the larger the UGBS were, the more distance from the central areas to the edge, *i.e.* the higher the distance to potential threats. Contrastingly, smaller-sized or edged shapes with a higher perimeter:area ratio, UGBS are potentially more exposed to threats and the interference cover of human activities (higher edge effect) (Turner & Gardner, 2015). Studies in large-sized cities evidenced that the perimeter:area ratio and indicators related to shape and connectedness to surrounding areas (edge effect) are better predictors of bird biodiversity than merely the size of UGBS (Garízabal-Carmona & Mancera-Rodríguez, 2021; Shih, 2018). Despite the positive effect of UGBS size on species richness, it remains poorly understood how size thresholds on individual patch size may influence biodiversity conservation in urbanized landscapes (Lepczyk *et al.*, 2017). There are further current discussions about how land sharing (extensive sprawling urbanization) vs. land sparing (intensive and compact urbanization) urbanisation initiatives along a landscape fragmentation *continuum* may influence biodiversity in urban contexts (Stott *et al.*, 2015). Therefore, effective strategies to increase biodiversity in urban contexts should account not only for the size but for the shape and connectedness of UGBS (Turner & Gardner, 2015). As increasing the size of UGBS in consolidated cities can be challenging, due to land availability constraints (Stott *et al.*, 2015), previous research suggests that increasing the tree coverage and vegetation stratum complexity is a good alternative to improve UGBS functionality (Garízabal-Carmona & Mancera-Rodríguez, 2021; Müller *et al.*, 2018; Sändstrom *et al.*, 2006). However, such measures should be carefully considered as management for one taxonomic group may be detrimental to others (*e.g.* increasing tree coverage would generally benefit birds but not butterflies, as these tend to prefer semi-open rather than shaded spaces) (Warren *et al.*, 2021; Sändstrom *et al.*, 2006).

Both proxies, UW and HQ were in general higher in the periphery of the urban areas indicating a higher potential to support biodiversity. Some centric areas (*e.g.* centric parks Fig. 2) were identified with high UW but low HQ, and in general, HQ values were high exclusively in areas located in the periphery of the city. These differences may be due to the inputs used to estimate both indicators. While UW takes into account structural descriptors of land use and terrain (perceived naturalness, remoteness and challenging terrain) (Müller *et al.*, 2018, 2015), HQ includes assessing the expert perception of the habitat suitability of different land uses and their susceptibility to different anthropogenic impacts. Thus, HQ provides not only structural descriptors that can be obtained from land use maps and digital elevation models but incorporate functional descriptors of habitat suitability and vulnerability to threats to spatially describe the capacity to support biodiversity (Sharp *et al.*, 2020; Di Febbraro *et al.*, 2018; Sallustio *et al.*, 2017). HQ seems to better consider potential threats (*i.e.* roads, night light, population density, railways and noise), their distribution and potential impact on the capacity of the different land use classes to harbour biodiversity. Contrastingly, other areas were identified with higher UW than HQ. For instance, industrial areas with low UW values showed medium HQ values since the spatial distribution and addition of threats had a lower influence in these areas as they are far from the city centre (*e.g.* nightlight sources were fewer and population density was considerably lower than towards the city centre).

In general, we found high UW and HQ values in freshwater ecosystems (*e.g.* Zadorra river) indicating an overall high capacity to support biodiversity. Preserved wetlands, streams and surrounding areas (floodplains) typically support high biodiversity and are ecological corridors that may connect UGBS to larger ecosystems (*e.g.* peripheral lakes or mountains) (Mansur *et al.*, 2022). However, these ecosystems were highly vulnerable in urban areas (Stroud *et al.*, 2022) and were in our study identified as with higher UW than HQ due to the high vulnerability of these ecosystems to potential impacts. This vulnerability seemed to be particularly high for rivers possibly due to their higher edge effect which lowers the distance to threats (Sharp *et al.*, 2020). Mapping of HQ allows identifying how these potential threats can affect the habitat suitability to support biodiversity as well as the connectivity of these ecological corridors.

### *2.4.2. Methodological considerations*

Both UW and HQ are simple indicators, yet robust in describing the spatial patterns of biodiversity in the urban context, though they have different potential uses. UW seems to be more suitable when functional data is not available as it incorporates indicators related to the landscape structure and the input requirement is relatively low (*i.e.* digital elevation model, land use, mechanized access). Besides, both models may partially address some limitations associated with NDVI/NDWI indicators, which are widely used in urban environmental studies to quantify exposure to UGBS but are less adequate for measuring their quality. Such indicators are sensitive to seasonal variability and do not differentiate between green space typologies as considered in this study, thus limiting their applicability in the context of urban biodiversity conservation (Trethewey & Reynolds, 2021).

On the other hand, HQ is a more robust model to evaluate impacts affecting the functionality of the landscape, since it includes information on threats and their relative impact on biodiversity conservation objectives. These differences in UW and HQ, provide more accurate information to managers by identifying areas that can be categorized with high UW that are not so likely to support high biodiversity due to the influence of threats. Furthermore, our outcomes allow identifying key areas for urban biodiversity protection, (un)suitable places for new urban development or where efforts need to focus on reducing threats impacts. However, both models do not integrate size or connectivity descriptors that are highly associated to habitat quality for biodiversity, which would be a good improvement to further model development. Acquisition of extensive information on (urban) biodiversity status is a time and resource-consuming endeavour (Jalkanen *et al.*, 2020) and since the input requirements to assess general biodiversity proxies in our spatial models were relatively low, they are particularly suitable for places where biodiversity information is deficient or monitoring efforts are scarce. The combination with expert consultation, a widely-used method to obtain information when data is limited (Terrado *et al.*, 2016; Kuhnert *et al.*, 2010), makes this approach cost-effective to evaluate potential urban biodiversity support while incorporating habitat suitability and sensitivity to threats to the different habitats.

These models have however some limitations based on their different assumptions on biodiversity support, data quality, and potential modelling uncertainties, as happens with other biophysical models (*e.g.* Aznarez *et al.*, 2021; Bagstad *et al.*, 2013). UW was in general a robust indicator of biodiversity, and this was supported by the results of the Jack-knife test, where it remained with stable values even when excluding one of its three basic components (perceived naturalness, remoteness and challenging terrain) (Appendix A). However, from these three indicators, the maps excluding perceived naturalness had lower performances, indicating that perceived naturalness is a key input contributing the most to UW character. Yet, other studies suggest that remoteness was more important in influencing UW (Müller *et al.*, 2018), which indicates that the potential role of each indicator may vary depending on the habitat distribution of the study area.

Expert consultation was effective when lacking threat data to enable spatial comparisons and use relative scores to rank habitat suitability and quality indicators. However, different expertise backgrounds from the consulted experts, considering the suitability of the urban habitats to different types of biodiversity, along with the different anthropogenic threats to habitat suitability may influence the results (Di Febbraro *et al.*, 2018). Despite the subjectivity and variability in expert considerations (Sallustio *et al.*, 2017) introducing moderate uncertainty in our HQ predictions, the overall assessment proved to be consistent with the relative UW outcomes and coherent with the species richness indicators. The parametrization scores obtained from the international expert consultation can be further tested to any other urban context, as the survey was designed for urban biodiversity and land uses in general. The crowd-sourced data of the occurrence of taxonomic groups used to assess potential associations with UW and HQ indicators may have different biases that hinder their suitability as predictions of species richness as overrepresented data in easily accessible UGBS, highly populated areas or circulation ways (Sultana & Storch, 2021; Petersen *et al.*, 2021; Callaghan *et al.*, 2020). For instance, bird richness was higher due to an artefact in the greater number of observations (Fig 5) since it is a more charismatic group, widely used in citizen science and reported in 'ornitho.eus'. However, these data allow studying large spatial scales based on large databases from multiple observations which overcome at least part of these biases and made these data suitable for urban biomonitoring programs (Callaghan *et al.*, 2020).

UW and HQ were positively correlated and robust proxies to predict biodiversity for all considered taxa in our study. This suggests that at least with the data availability we had for biodiversity, land use, habitat suitability and potential threats, in a middle-sized city such as Vitoria Gasteiz, these databases were reliable to estimate biodiversity at urban scales. However, there is a substantial unexplained variation between our proxies and biodiversity that is attributable at least partially to other non-contemplated factors, *e.g.* seasonal variability, unassessed threats or habitat conditions, biogeographical filters, the effect of invasive species. Yet, further research considering small (*e.g.* micro-scales) and unconventional habitats with aggregated distribution patterns in the environment should consider possible limitations in the use of these models developed for larger areas (Knapp *et al.*, 2021; Soanes *et al.*, 2019). This approach should allow to improve the currently limited availability of biodiversity data and inventory completeness with an adequate aggregation scale for different taxonomic groups across the urban landscape, ultimately assessing spatial patterns at fine scale resolution. Recognizing and measuring the value of small spaces and unconventional habitats to urban biodiversity (*i.e.* from brownfields and cemeteries to cavities within buildings and infrastructure) calls for exploration along with novel conservation opportunities for management (Knapp *et al.*, 2021; Soanes *et al.*, 2019). Our results suggest that even if the obtained outcomes may be sensitive to biases, which is often the case in ecological modelling, they provide valuable and robust insight into identifying urban areas with high potential to support biodiversity and to pinpoint conservation areas.

### *2.4.3. Implications for urban biodiversity management*

In our study, UGBS close to the city centre showed low and very low UW and HQ values, suggesting that cities such as Vitoria-Gasteiz, with 20 $m^2$ UGBS per inhabitant, do not necessarily encompass high HQ to biodiversity as these UGBS may be ecologically very poor, although still providing nature contributions to people. Coinciding with other studies (Mansur *et al.*, 2022; Soanes *et al.*, 2019; Turner & Gardner, 2015; Forman, 2014) our results suggest that although small and fragmented UGBS, are important for species persistence, biodiversity would benefit from decreasing perimeter:area ratio, along with increasing the size, quality and connectivity of UGBS from peri-urban areas to core areas. As the Vitoria-Gasteiz municipal Biodiversity Conservation Strategy points out, the overall biological diversity of the flora and fauna in urban parks has decreased significantly due to intensive management practices (*i.e.* the frequency with which the grass is being cut or pruned and dead trees removed) (CEA, 2014). Another reason adding to the low HQ for biodiversity support is the specific and structural simplicity of UGBS, which mostly consists of the vertical structure of green spaces (*i.e.* lawn and trees) (Garízabal-Carmona & Mancera-Rodríguez, 2021; Aronson *et al.*, 2017; CEA, 2014). Previous studies suggest that classical parks with lawns and tall deciduous trees usually harbour few breeding species (Forman, 2014). Increasing HQ for biodiversity in urban socio-ecosystems is critically needed (Soanes *et al.*, 2019), which is also correlated to UW according to our findings. This includes actions aiming to improve the conditions for human-wildlife coexistence, by accommodating dynamic natural processes (Mansur *et al.*, 2022), providing resources such as refuge sites (*e.g.* holes in tree chunks, adding shrub cover), feeding sources or reducing the impact of different threats impact to habitat suitability, such as night light or fragmentation (Soanes *et al.*, 2019; Forman, 2014). In other words, to increase biodiversity, cities should bring some wildness back. Reducing management (through rewilding *sensu* Perino *et al.* 2019), without affecting people's accessibility to UGBS could be a cost-effective way to enhance HQ in the city and therefore general biodiversity (Mansur *et al.*, 2022; Müller *et al.*, 2018). A variety of wildness-friendly actions such as reducing pruning or mowing would be suitable to increase the environmental quality of UGBS (Kowarik, 2018) while reducing maintenance costs, ultimately benefitting urban dwellers through encouraging interaction with nature. Vacant land and brownfields may likewise be an opportunity for increasing the network of green infrastructure in urban landscapes (Kabisch & Haase, 2013). Due to the scattered and low density of urban residential developments, Vitoria-Gasteiz has a considerable share of vacant and abandoned land, which has been included in the municipal Green Strategy. Yet, previous research warns that the repurposing of such areas may have undesired consequences for biodiversity (Macgregor *et al.*, 2022; Broughton *et al.*, 2020). Therefore, the ecological value of vacant land and brownfields along with its potential for urban wildness for increasing both, local biodiversity and recreational purposes, should be carefully considered (Macgregor *et al.*, 2022; Kabisch & Haase, 2013).

As the size and connectivity of UGBS influence biodiversity, it is also closely related to public access to recreation in natural environments (Senetra *et al.*, 2018). Experiencing nature in cities is a necessary condition for conservation since people with more exposure and access to nature are more interested in its conservation (Callaghan *et al.*, 2020). As an increasing majority of the human population lives in cities, urban biodiversity is becoming the main people's interaction with nature (Dunn *et al.*, 2006). However, considering the link between contact with nature and conservation measures, it is important to address the inequitable distribution of urban nature, and specifically, biodiversity, as it may contribute to lower levels of participation from minorities in environmental leadership initiatives (Mansur *et al.*, 2022; Dunn *et al.*, 2006). Here, our results may be also used to indicate areas within the city with little or no UGBS, and relate it to the demand side for such UGBS (*i.e.* through population density at the neighbourhood scale) as areas with very low UW and HQ values coincided with highly impervious areas. Further research should address how urban biodiversity patterns are distributed among vulnerable urban dwellers (Mansur *et al.*, 2022). However, understanding the way that urban dwellers value and experience nature among contrasting perspectives is key to identifying socio-ecological trade-offs and feedback (Mansur *et al.*, 2022; Hill *et al.*, 2021). For instance, urban dwellers often perceive rewilding actions (*i.e.* low maintenance regime) negatively in the city, seeing such areas as abandoned, particularly when applied to urban contexts (Kowarik, 2018; Botzat *et al.*, 2016). On the other hand, some species present in urban surroundings can be seen as pests (*e.g.* wild boar, seagulls, pigeons), leading to new human-wildlife conflicts or reinforcing existing ones (Forman, 2014; Dunn *et al.*, 2006). Management actions should be carefully considered and designed to successfully communicate that unmanaged areas are intended by planners and not the product of neglect (Kowarik, 2018; Müller *et al.*, 2018; Botzat *et al.*, 2016). Since most of the world population including decision-makers and financial resources reside in urban areas, urban biodiversity conservation actions will ultimately rely on the ability of the population to experience and maintain the connection with urban nature (Callaghan *et al.*, 2020; Soga & Gaston, 2016; Dunn *et al.*, 2006).

## 2.5. Conclusion

UW and HQ indicators have been found to be appropriate for predicting the potential for biodiversity support in an urban context and therefore be integrated as key layers into urban planning. However, it should be noted that their use and individual suitability will depend on the data availability and study focus. UW is more suitable when focused on landscape structure indicators if functional data is not available, while HQ adds to this spatial information the effects of anthropogenic impacts on biodiversity and landscape functionality. Overall, this research has developed easily adaptable and replicable indicators that can provide information at a fine urban scale, accounting for different categories of UGBS and considering the anthropogenic effects on the quality of UGBS. The methodology used was based on open-access software, public data and crowdsourced data for all the phases of the modelling approach. This approach can provide a detailed level of information on urban biodiversity conservation in cities with limited monitoring capacities. Our study suggests that strategies aiming to increase biodiversity in urban landscapes should be based on improving wildness by enhancing structural dimensions of UGBS (*i.e.* perimeter:area ratio, size and connectivity), reducing management frequency and intensity along with evaluating potential anthropogenic pressures on different habitats.

Urban dwellers would benefit from enhanced contact with nature by increasing these drivers of urban biodiversity, but this would rely on the capacity to adequately communicate the aim and intentions behind rewilding actions. We, highlight that UGBS planning should focus not only on access and quantity but also on the ecological quality and particularly on the support for biodiversity that will ultimately enhance urban dwellers' experiences in their nearby nature.

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# Chapter III

## Luxury and legacy effects on urban biodiversity, vegetation cover and ecosystem services

**Abstract:**

Socioeconomic and historical drivers shape urban nature distribution and characteristics, as luxury (wealth-related) and legacy (historical management) effects. Using remote sensing and census data on biodiversity and socioeconomic indicators, we examined these effects on urban biodiversity and vegetation cover in Vitoria-Gasteiz (Basque Country). We also tested the luxury and legacy hypotheses on regulating ecosystem services (ES) and explored predictor interactions. Higher educational attainment positively correlated with urban biodiversity, confirming the luxury effect, but had no effect on vegetation cover or ES. Older areas had higher vegetation cover and ES evidencing a legacy effect with an inverse response on biodiversity, attributable to more recent management strategies promoting biodiversity in green spaces. Habitat quality amplified the luxury effect, while population density strengthened the legacy effect. Our results suggest that urban biodiversity is mainly driven by socioeconomic factors, while vegetation cover and ES are influenced by management legacies in interaction with population density.

**Keywords:** *Urban ecology; Urban biodiversity; Urban ecosystem services; Environmental justice; Nature-based solutions*

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## 3.1. Introduction

Urban green spaces (UGS) and their associated biodiversity are key to urban residents' health and quality of life (Clarke et al., 2013; Magle et al., 2021). Green infrastructure, including parks and street trees, is increasingly considered in urban planning as nature-based solutions (NbS) for climate change adaptation and mitigation (Goodwin et al., 2023; Kabisch, 2019; Oke et al., 2021). UGS provide multiple ecosystem services (ES) and functions, such as habitat maintenance, local climate regulation, air quality improvement, runoff control, and spaces for people to relax, exercise, socialise, among others (Anderson et al., 2021; Baró et al., 2019; Gómez-Baggethun & Barton, 2013). These benefits are influenced by the characteristics of the UGS, such as their structure, size, connectivity, and biodiversity (Aznarez et al., 2022; Houlden et al., 2021). For instance, tree diversity and vegetation structure can affect the amount of shade reducing the urban heat island effect, its quantity can influence the amount of carbon sequestration, and connectivity can promote wildlife movement in the landscape (Forman, 2014). Additionally, different taxonomic groups can further contribute to plant pollination, providing food and other resources, improving soil and water quality, and control pests.

Classical approaches in ecological research have largely focused on climatic and other natural or semi-natural factors as local biodiversity determinants (Leong et al., 2018). However, socioeconomic factors related to wealth or urban form and development are increasingly being recognised as key drivers of biodiversity patterns in cities (Chamberlain et al., 2019; Leong et al., 2018; Pickett et al., 2016). The effect of urban development and social dynamics in cities lead to heterogeneous urban landscapes by influencing the spatial distribution and proportion of impervious surfaces, availability of resources, and environmental quality factors like water, soil or air quality (Magle et al., 2021; Schell et al., 2020). In this context, urban residents with higher socioeconomic status, have a greater capacity to allocate resources towards vegetation and habitat, thereby influencing the overall UGS dynamics (Grove et al., 2014; Magle et al., 2021; Schell et al., 2020). In addition, wealthier residents can have greater influence and agency over public and private decisions, including land use planning and investments in their neighbourhoods, due to their lobbying capacity, as well as access to decision-makers, including public and private investors (Baró et al., 2019; Grove et al., 2014). Their political connections, social capital, knowledge, and access to information further strengthen their ability to advocate for their own interests, which can ultimately shape urban biodiversity and vegetation patterns over time (Locke et al., 2021; Grove et al., 2014; Clarke et al., 2013; Werner & Zahner., 2010). Driven by the lifestyle choices, social status, and ability to invest in environmental management options, the influence of wealthier residents can ultimately contribute to environmentally driven neighbourhood differentiation (Grove et al., 2014; Boone et al., 2010). These factors collectively contribute to the complex dynamics of urban ecosystems and their biodiversity (Clarke et al., 2013; Boone et al., 2010). Thus, urban residents may have differential access to nature and its associated benefits, including situations where disadvantaged communities may often experience deprived access to those benefits (Turner et al., 2007). Human-driven uneven distributional patterns of vegetation and biodiversity can shape ecological patterns underpinning ecosystem functions and ES in urban contexts (Schell et al., 2020). People's exposure to nature and its related benefits has been increasingly constrained to urban contexts with advancing urbanisation (Aznarez et al., 2022; Dunn et al., 2006). Since urban nature and biodiversity tend to be unevenly distributed across social groups, it is key to address its distribution in urban contexts from an environmental justice dimension (Tozer et al., 2020; Baró et al., 2019).

Wealthiest areas have been spatially related to higher biodiversity, a pattern defined as ‘luxury effect’, and proxied by plants (Leong et al., 2018; Davis et al., 2012; Hope et al., 2003), birds (Howes & Reynolds, 2021; Chamberlain et al., 2019; Davis et al., 2012) and to mammals, lizards and arthropods to a lesser extent (Magle et al., 2021; Chamberlain et al., 2019;Blicharska et al., 2017). This pattern has also been analysed regarding urban green cover inequalities using remote sensing (Schwarz et al., 2015; Jenerette et al., 2013;Grove et al., 2006). Thus, lower household income and individual educational attainment have been associated with low abundance of trees within urban areas (Kirkpatrick et al., 2011). However, the exclusive use of conventional economic indicators, such as income, as proxies for social status in studies of the luxury effect is a topic of debate. These indicators oversimplify complex social concepts, making accurate quantification challenging. Recent meta-analysis has indicated that income might be an incomplete predictor of ecological communities’ patterns (Kuras et al., 2020). Therefore there is a strong need to include alternative indicators that consider individual development opportunities, capabilities, and other factors for a comprehensive understanding of the luxury effect. In this regard, education is an alternative descriptor that may capture a broader range of socioeconomic variation and improve the ability to explain the luxury effect (Magle et al., 2021; Kuras et al., 2020; Kendal et al., 2012; Hope et al., 2003). Previous research has shown that people with higher educational attainment are more likely to prefer and have the economic means to live in areas with more UGS in cities (Kendal et al., 2012; Luck et al., 2009;Heynen & Lindsey, 2003). Such wealthier groups may also exert a higher influence on urban environmental management, tending to demand higher environmental quality in their neighbourhoods, *e.g*. by supporting vegetation cover in neighbourhoods where they live (Kendal et al., 2012). Although income and higher educational attainment have been associated with urban biodiversity patterns in the literature, their correlation does not imply a causation. This is because urban socio-ecological systems are shaped by a myriad of biophysical and social factors, and their interactions, including population dynamics, biogeographical filters, socio-cultural aspects, habitat fragmentation and land uses (Aznarez et al., 2022; Pickett et al., 2016; Forman, 2014; Pickett et al., 2008).

The way vegetation cover and biodiversity are spatially distributed in urban landscapes is further influenced by the legacies of past land use policies, including UGS management practices and urban planning strategies (Locke et al., 2021; Schell et al., 2020; Roman et al., 2018; Hope et al., 2003). This is defined as 'legacy effect' and is often explained by higher plant diversity in older neighbourhoods (*i.e*. areas with older housing and urban development), and reflects the longer-term trajectory of management practices (Clarke et al., 2013). For instance, longer development periods will allow for a higher tree diversity due to extended successional times, several establishment of different tree species by multiple managers, and adequate timeframes for species with long lifespans to reach their full size (Clarke et al., 2013; Boone et al., 2010; Hope et al., 2003). Since legacy effect can uphold biodiversity patterns driven by luxury effect over time, both luxury and legacy effects may result in additive responses (Leong et al., 2018).

Understanding the consequences of luxury and legacy effects on spatial patterns of urban biodiversity and their cascading effects on human well-being may contribute to better management practices and strategies towards environmental justice. Hence, we here analyse how luxury and legacy effects influence the spatial distribution of vegetation cover, biodiversity and their associated regulating ES in urban landscapes. Our study contributes to the limited existing knowledge about the role of luxury and legacy effects on the provision of ES in urban landscapes. We focus our study on the mid-size European city of Vitoria-Gasteiz (248,087 inhabitants, Eustat, 2020), located in the Basque Country. In the 1990s, Vitoria-Gasteiz implemented an ecological restoration initiative to create a green belt in its degraded peripheral areas. In 2012, the city was awarded as the European Green Capital, recognising its greening efforts. Following that, during 2014, a green infrastructure plan was designed to further increase UGS towards the urban core, promote urban wildlife, and enhance ES (Aznarez et al., 2022; CEA, 2014). We here: i) test for the luxury and legacy effects by assessing the relationship between urban residents' high educational attainment and urban development age, respectively, regarding biodiversity and vegetation cover; and ii) assess how luxury and legacy effects influence the supply of regulating ES, by focusing on those provided by public urban trees. We expect to evidence the presence of luxury and legacy effects, which are associated with increased biodiversity and vegetation cover, thereby enhancing the capacity to provide regulating ES. However, we expect these effects to be influenced by additional socio-environmental factors and their interaction, such as population density, vegetation cover, and habitat quality, which are often overlooked in the analysis of these effects. By integrating these factors and their interactions, our research contributes to a more comprehensive understanding of the complex dynamics shaping urban environments.

## 3.2. Methods

### *3.2.1. Study site*

Vitoria-Gasteiz is a middle-sized city (248.087 inhabitants (Eustat, 2020)), located in the Basque Country and is internationally recognised as the 2012 European Green Capital. Located in the North of the Iberian Peninsula, Vitoria-Gasteiz has a considerable share of publically-accessible green infrastructure, including urban parks, forests, wetlands and canopy (Fig. 1) (Aznarez et al., 2022). From the mid-1980s, the city committed to a consistent urban greening strategy encompassing a planned urban network of green infrastructure (Neidig et al., 2022). The latter included the restoration of multiple ecosystems towards the urban fringes of the city which aimed at limiting urban expansion and slowing industrial activity. The most remarkable outcome of these greening policies was a 731-ha and 35 km long greenbelt delimiting the urban core. The urban green areas of Vitoria-Gasteiz harbour more than 381 different species of trees and shrubs with more than 130,000 trees distributed around the city as well as 12,160 shrub masses (CEA, 2014). Vitoria-Gasteiz is one of the European cities with a greener surface area (20 $m^2$) per inhabitant. Despite the consistent efforts towards greening the city, smaller-sized centrally located UGS has been recently associated with low habitat quality and urban biodiversity, as opposed to larger areas towards the outskirts of the city (Aznarez et al., 2022). Vitoria-Gasteiz offers a unique and suitable case study to test if luxury and legacy effects influence the spatial distribution of urban vegetation, biodiversity and ES at the neighbourhood scale. This is due to its fine-scale urban public canopy coverage data (*i.e.* urban tree inventory), resulting from the implementation of a recent urban green infrastructure planning strategy (CEA, 2017; CEA, 2014), as well as a bird census database resulting from a consistent biodiversity monitoring program.

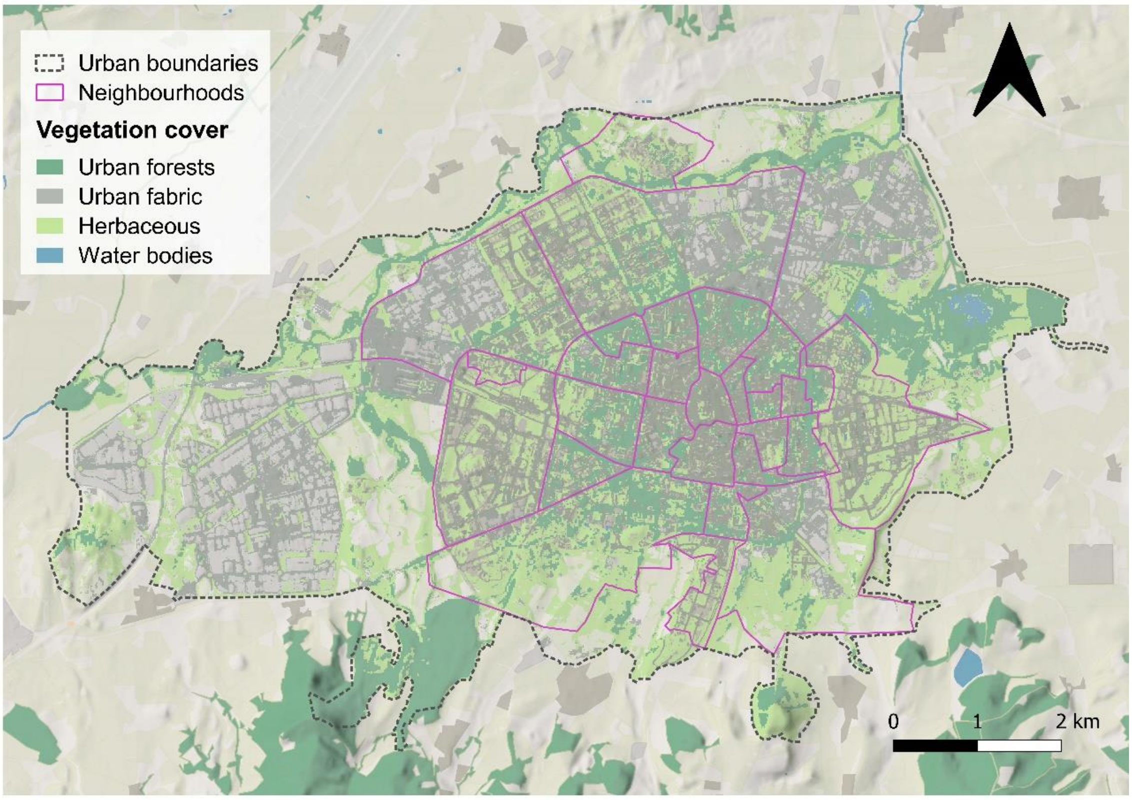


**Figure 1. Study area.** Location of Vitoria-Gasteiz city, neighbourhood delimitation and main land uses cover within urban boundaries defined by urban fabric, forests, water bodies and herbaceous cover. Own elaboration based on GEE random forest classifier and Sentinel -2 imagery (see Section 3.2.2. for methodology details). Background from © MapTiler (www.maptiler.com) and © OpenStreetMap (CC BY-SA 2.0).

### 3.2.2. *Indicators for luxury and legacy effects*

We combined public biodiversity datasets with a remote sensing approach and socio-demographic variables to assess the luxury and legacy effects. We first compiled two biodiversity datasets as a measure of species richness: an inventory of trees located in public land including individual tree information provided by the City Council (CEA, 2017) and a bird census (10016 observations) consisting of 100 sampling points distributed across the public urban green space between 2017 to 2020 by the NGO SEO/BirdLife (Fernández Calvo *et al*., 2020), with the data being provided by the City Council. We also used mean habitat quality values per neighbourhood as a complementary indicator for biodiversity, whose spatially explicit data was obtained from previous research in the study area (Aznarez et al., 2022). Here, habitat quality refers to the capacity of urban ecosystems to provide the resources and conditions needed for wildlife. This indicator is influenced by the proximity to and the intensity of human land uses and is a continuous variable ranging from 0 – low to 1 – high.

To distinguish between different vegetation cover types (*i.e*. herbaceous and canopy cover), we described 'herbaceous' as all non-woody vegetation (Clarke et al., 2013), while 'canopy' describes the layers of leaves and woody vegetation that cover the ground when viewed from above (Locke et al., 2021). We used 'vegetation cover' to describe the combination of the above-mentioned cover types. To integrate vegetation cover from both, public and private spaces, we complemented the above-mentioned datasets with a land cover classification using Google Earth Engine (GEE) and Random Forest classifier, a machine-learning method for satellite imagery-based land use classification (Breiman, 2001). A key advantage of the Random Forest classifier includes its high accuracy, robustness and efficient handling of noise or overfitting (Rodriguez-Galiano et al., 2012). Besides, RF is a non-parametric method, so it does not need input variables following a particular statistic distribution (Puissant et al., 2014). We used the Sentinel-2 (A level-2A) Multi-Spectral Instrument (MSI) with two imageries averaged from 2017/07/11 and 2017/07/18 and between 10 and 20 m spatial resolution. We used the following MSI spectral bands: B2, B3, B4, B5, B6, B7, B8, B8A, B11, and B12. Before using the imagery, we performed an atmospheric correction using the QA60 quality band from Sentinel-2 to mask the clouds and select the images with the least cloud cover. To differentiate build areas from vegetated areas, we first calculated the Urban Index (UI) (bands B12 and B8A) and Normalised Difference Vegetation Index (NDVI) (bands B8 and B4) as inputs to be included in our land use classification. Then, we added to our dataset a Digital Elevation Model (DEM, 30 m resolution) from SRTM V3/USGS available for our study site and in the products catalogue of GEE. The DEM was used to assess aspect, slope, and hill shade variables. We then defined 4 different land use classes to be used in our study area: urban forests, urban fabric, herbaceous and water bodies. Following previous studies' recommendations, we selected 100 decision trees (n = 100) to run the classification model. The RF was then trained to map the vegetation cover distribution in Vitoria-Gasteiz and validate the classification accuracy. The validation data was then used to calculate a confusion matrix and assess the method's overall accuracy and kappa index to quantify the performance of RF.

To assess the role of vegetation cover in terms of luxury and legacy effects on ES, we used our land cover supervised classification outcomes (10 m resolution) including both herbaceous and canopy coverage. The resulting spatial pattern of herbaceous and canopy coverage from GEE is shown in Supplementary Fig. 1 and green cover percentage at the neighbourhood level is shown in Supplementary Fig. 2. To complement the dataset containing the public canopy inventory, we focused on canopy and herbaceous cover results.

Once we had the vegetation cover from our land use classification mapping (Fig. 1), we selected socio-demographic variables indicating socioeconomic status to test them against the vegetation cover and biodiversity data. We collected all the variables for the 2015 year from the Basque Statistics Office (Eustat, 2016) at the neighbourhood level. As yearly median household income was not available for the year 2015, we used census data on the percentage of residents with high educational attainment by neighbourhood over the total population (Kendal et al., 2012; Cilliers et al., 2011). Specifically, 'high educational attainment' is defined as the completion of tertiary education, including university studies, higher engineering and similar, as well as postgraduate, master's, doctoral and specialization studies (Eustat, 2015). This variable is used as a suitable proxy for socioeconomic status given that it is a key human capital marker (and hence also of income levels) (Mirowsky & Ross, 2003;Lynch & Kaplan, 2000) (Supplementary Fig. 5), which in turn has been shown elsewhere to influence both species diversity and vegetation cover (Clarke et al., 2013; Kendal et al., 2012; Kirkpatrick et al., 2011). Given that education and income tend to be highly correlated, in most cases it is difficult to isolate the influence of both variables which results in most studies considering only income and excluding education level, leading to findings only related to income (Magle et al., 2021; Chamberlain et al., 2019; Clarke et al., 2013; Kendal et al., 2012; Hope et al., 2003). Since information on educational attainment is relatively easily accessible, using this variable may be particularly adequate for case studies with low census data availability.

We also included residential housing by construction year to build our neighbourhood age index and population density (inhabitants/ha) as control variables for the statistical modelling. To account for legacy effects, we built a neighbourhood development age indicator (see Eq. 1):

$$\text{Neighbourhood age} = \sum(\text{at} * \%t) \quad \text{(Eq.1)}$$

The neighbourhood age is calculated by summing the age of the transformation (at) at different time periods, ranging from 1800 to 2015, multiplied by the percentage of the area built by neighbourhood and time period (%t) (*i.e.* built housing, refer to Supplementary Table 2). We then rescaled the indicator values to a 0-1 range. Therefore, the higher the indicator, the older the neighbourhood and land transformation. The spatial distribution of the considered socio-demographic variables is shown in Supplementary Fig. 2 and Supplementary Table 2 for the development age per neighbourhood.

### *3.2.3. Ecosystem services modelling of urban tree canopy*

We quantified four ES for our case study: urban temperature regulation, runoff control, air purification, and carbon offsetting. We used data from the city council that were analysed using the i-Tree Eco software program (v.6, www.itreetools.org). The i-Tree Eco tool is a process-based suite of models developed by the US Forest Service and designed to quantify urban forest structure and functions, including its ES supply (Nowak, 2000). This tool operates at a local scale and requires standardised field data of individual trees (*i.e.* species name, tree height, diameter at breast height (DBH), land use, crown size and health indicators) from either complete inventories or plot-based sampling (Baró et al., 2019; Lerman et al., 2014). It then combines the tree measures with hourly air pollution and meteorological data to assess the forest structure and quantify ES provision (Baró et al., 2015; Nowak, 2000). The outcomes of i-Tree assessments have been addressing the value of urban canopy to improve the life quality of urban residents through the supply of multiple ES such as air quality regulation, carbon storage, runoff control and heat mitigation (*e.g.* Baró et al., 2019; Nowak, 2013). Further, i-Tree applications have contributed to exploring the relationship between urban forest management and environmental quality, along with identifying urban environmental justice issues (Baró et al., 2019; Driscoll et al., 2012; Nowak et al., 2006).

From the complete inventory of urban public trees provided by the Vitoria-Gasteiz city council (originally containing 135,560 trees), we excluded trees from the modelling due to inaccuracies such as missing geolocation and structural variables, trees located in rural areas, inaccurate identification and removed trees, which finally resulted in 89,001 individuals for a clean version of the dataset. We confirmed the structural variable measures used for the assessment with an expert on tree physiology, (see Supplementary Table 4).

Given the limitations of i-Tree Eco to update the system's pollutant concentrations and precipitation data, such measures had to correspond to 2015 as the most recent year and be derived from the average values of four monitoring stations. Pollution removal is processed by i-Tree Eco for ozone (O3), nitrogen dioxide (NO2), carbon monoxide (CO), sulphur dioxide (SO2), and particulate matter of fewer than 2.5 microns (PM2.5) (Nowak et al., 2006). The i-Tree Eco indicators are presented in Table 1. Our modelling assessed urban temperature regulation by considering transpiration as an indicator of tree canopies' cooling effect on air temperature (USDA, 2018). We estimated runoff control by considering rainfall infiltration, evaporation and water intercepted by the tree canopy as the total amount of avoided runoff (Bowler et al., 2010). Meanwhile, we estimated air purification from the dry deposition of air pollutants following Nowak *et al*. (2008) approach.We stratified the analysis of all the indicators at the neighbourhood level (n=28) to allow statistical analyses with the sociodemographic variables. Then, given that the distributive patterns of the four ES mapped were co-occurrent, we aggregated the ES outcomes to one single ES index value, following Baró *et al*. 2019. To this, we re-scaled each one of the four regulating ES values in a range from 0 to 100 as the minimum and maximum values and then summarised the result without the application of weights. The final result was also rescaled to the 0 to 100 range.

**Table 1**. **ES indicators quantified by i-Tree Eco and ES index considered in the assessment of public canopy.**

| Ecosystem services (ES) | Indicator | Units |
|---|---|---|
| Temperature regulation | Transpiration | $m^3$/ha/year |
| Runoff control | Avoided runoff | $m^3$/ha/year |
| Air purification | Pollution removal is the sum of the removal rates:<br>$O^3$<br>$NO^2$<br>CO<br>$SO^2$<br>$PM_{2\cdot5}$ | kg/ha/year |
| Carbon sequestration | Carbon storage | ton/ha |
| Urban ES Index | The scaled sum of the four combined indicators | 0 to 100 |

### *3.2.4. Data analysis*

The scale of our statistical analyses was defined at the neighbourhood level (n = 28) as it was the finest scale for which the considered socioeconomic data was available. The socioeconomic data used was from the 2015 year to pair it with i-Tree Eco modelling. To account for legacy and luxury effects, we examined the contributions of both, high education attainment and urban development age (*i.e.* as explanatory variables), on biodiversity, vegetation cover and the four regulating ES along with other control variables such as population density, habitat quality and neighbourhood area (in ha) (Supplementary Table 1).

Initially, we computed all pairwise Spearman correlations (Supplementary Fig. 4) to see how our selected variables correlated with each other using the 'corrplot' package in R (Wei & Simko, 2017). Then, we tested for spatial autocorrelation of the considered variables using Global Moran's I in ArcGIS v.10.7.1 (Supplementary Fig. 2). We used generalised linear models (GLMs) to assess biodiversity, vegetation cover and ES in relation to our selected explanatory and control variables. For each explanatory variables considered (*i.e.* high educational attainment, neighbourhood development age, habitat quality, vegetation cover, population density), we performed a separate linear regression. When finding strong and significant correlations, we tested for interactive effects between variables such as the case for i) high educational attainment and habitat quality, ii) neighbourhood age and population density, and iii) neighbourhood age and vegetation cover. We selected the models based on statistical significance (p values $<0.05$) and F-test and adjusted $R^2$ for goodness of fit, following the principle of parsimony. We assessed the performance of the selected models by testing for normality and good fit, influential observations and visual inspection of diagnostic plots. Statistical analyses were conducted using R v.4.2.2 and the ‘tidyverse’ (Wickham et al., 2019), ‘performance’ (Lüdecke et al., 2021) and ‘SjPlot’ (Lüdecke, 2022) R packages.

## 3.3. Results

### *3.3.1. Luxury and legacy effects on biodiversity*

Biodiversity, as proxied by tree and bird species richness, was positively associated with high educational attainment, as an indicator of the luxury effect ($R^2$=0.25, F= 10.01, $p<0.01$, Fig. 2, Table 2). Conversely, biodiversity was negatively associated with the indicator of legacy effect: neighbourhood development age ($R^2$=0.22, F= 8.61, $p<0.01$, Fig. 2, Table 2). Neighbourhoods with higher biodiversity corresponded to those which underwent significant urban development between the late 1970s and the late 1990s (See Supplementary Table 2). Furthermore, species richness was positively correlated to habitat quality ($R^2$=0.45, F= 22.85, $p<0.001$, Fig. 2, Table 2) with a logarithmic response, vegetation cover ($R^2$=0.16, F=6.35, $p<0.05$, Fig. 2, Table 2) and negatively to neighbourhood´s population density ($R^2$=0.34, F = 14.92, $p<0.001$, Fig. 2, Table 2).

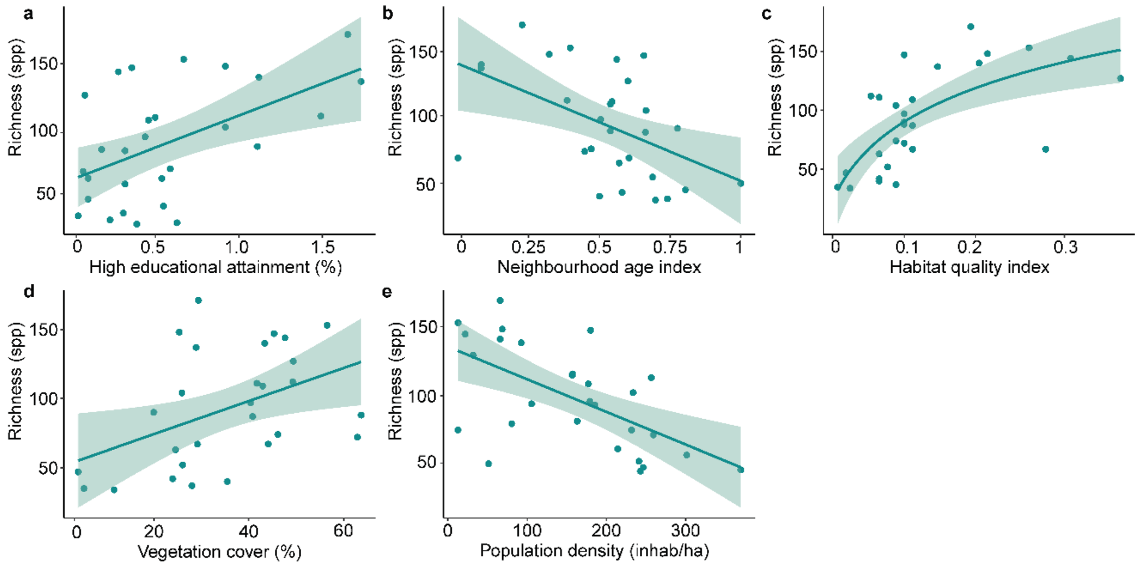


**Figure 2. Urban biodiversity responses to socio-environmental characteristics.** Linear models of biodiversity, expressed as tree and bird species richness, in response to a) high educational attainment (% of the population with high education attainment by neighbourhood over the total population, proxy of luxury effect), b) neighbourhood development age index (values ranging from 0 – newest to 1 – oldest), c) habitat quality index (values ranging from 0- lowest to 1- highest), d) vegetation cover (% by neighbourhood) and e) neighbourhood population density (inhab/ha). Shaded areas represent the 95% confidence interval. Model information provided in Table 1.

**Table 2. Linear models of urban biodiversity and vegetation cover to socio-environmental characteristics.** Model and parameters of urban biodiversity (tree and bird species richness) and vegetation cover (% per neighbourhood) in response to: high educational attainment (% of the population with high education attainment by neighbourhood over the total population, proxy of luxury effect), neighbourhood development age index (values ranging from 0 – newest to 1 – oldest), habitat quality index (values ranging from 0- lowest to 1- highest), population density (inhab/ha), vegetation cover (only for biodiversity), and interactions of high educational attainment with habitat quality and neighbourhood age with population density. Model formula for each predictive variable, adjusted variance explained ($R^2$), F-statistic (F), significance level (p) and standard error (Std. Error). N= 28, n.s.: non-significant.

| **Biodiversity** *(Species richness)* | | | | |
|---|---|---|---|---|
| **Model** | **Adj. $R^2$** | **F** | **p** | **Std.Error** |
| *Richness = 66.05 + 46.32 * High education* | 0.25 | 10.01 | <0.01 | 36.09 |
| *Richness = 140.68 - 91.70 * Neighbourhood age* | 0.22 | 8.61 | <0.01 | 36.82 |
| *Richness = 202.40 + 52.89 * log(Habitat quality)* | 0.45 | 22.85 | <0.001 | 30.99 |
| *Richness = 50.34 + 1.19 * Vegetation cover* | 0.16 | 6.35 | <0.05 | 38.08 |
| *Richness = 133.87 - 0.26 * Population density* | 0.34 | 14.92 | <0.001 | 33.86 |
| *Richness = 63.44 + 355.11 * High education:Habitat quality* | 0.53 | 31.09 | <0.001 | 28.67 |
| *Richness = 124.64 - 0.33 * Neighbourhood age:Population density* | 0.38 | 17.92 | <0.001 | 32.68 |
| **Vegetation cover** (%) | | | | |
| *Vegetation cover = 39.07 - 6.35 * High education* | 0.00 | 1.03 | n.s. | 15.44 |
| *Vegetation cover = 50.84 - 29.46 * Neighbourhood age* | 0.15 | 5.97 | <0.05 | 14.2 |
| *Vegetation cover = 69.04 + 16.21 * log(Habitat quality)* | 0.29 | 12.23 | <0.01 | 12.98 |
| *Vegetation cover = 51.68 - 0.10 * Population density* | 0.39 | 18.36 | <0.001 | 12.05 |
| *Vegetation cover = 48.53 - 0.14 * Neighbourhood age:Population density* | 0.48 | 26 | <0.001 | 11.13 |
| *Vegetation cover = 35.44 - 0.33 * High education:Habitat quality* | -0.04 | <0.001 | n.s. | 15.74 |

Apart from these direct effects, we also found significant effects on the interaction between variables correlated to species richness. Urban biodiversity was correlated with the interaction of high educational attainment and habitat quality, where biodiversity increased with high educational attainment, particularly so in neighbourhoods with higher habitat quality ($R^2$ = 0.53, F= 31.09, p<0.001, Fig. 3 A, Table 2). In addition, urban biodiversity was negatively correlated with the interaction between neighbourhood development age and population density, where biodiversity decreased with neighbourhood age, particularly so in more densely populated neighbourhoods ($R^2$ = 0.38, F= 17.92, p<0.001, Fig. 3 B, Table 2).

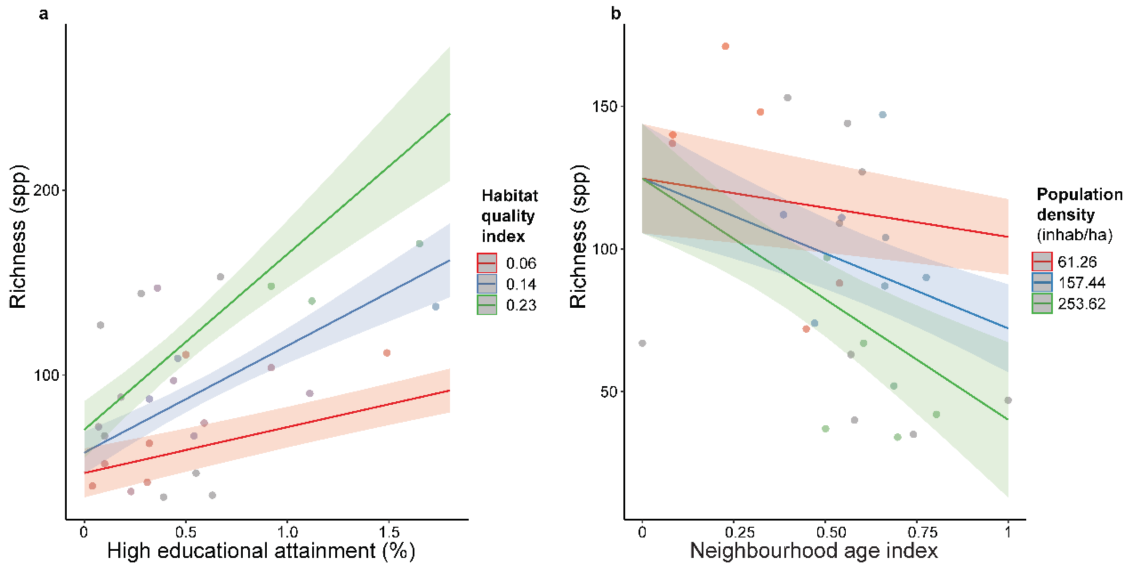


**Figure 3. Urban biodiversity responses to proxies of luxury and legacy effect.** Linear models of biodiversity, expressed as tree and bird species richness, in response to a) high educational attainment (% of the population with high education attainment by neighbourhood over the total population, proxy of luxury effect), by levels of habitat quality index (values ranging from 0 - lowest to 1 - highest), and b) neighbourhood age index (values ranging from 0 – newest to 1 – oldest, proxy of legacy effect) by levels of population density (inhab/ha). Shaded areas represent a 95% confidence interval. Habitat quality and population density are grouped into three categories defined by the mean value and +/-1 standard deviation (Aiken & West, 1991). Model information provided in Table 1.

Furthermore, neighbourhood age and population density were highly correlated in Supplementary Fig. 4 (R = 0.64; p < 0.01).

### *3.3.2. Luxury and legacy effects on vegetation cover*

Vegetation cover, including both herbaceous and canopy cover, showed no significant relation to higher education attainment, as an estimator of the luxury effect (Table 2). Furthermore, it was positively correlated with habitat quality ($R^2$=0.29, F= 12.23, p<0.01, Table 2), showing a logarithmic response, and negatively correlated with population density at neighbourhood scale ($R^2$=0.39, F = 18.36, p<0.001, Table 2). Yet, vegetation cover decreased with neighbourhood age, as an estimator of the legacy effect, particularly so, in areas with higher population density, where older and more populated neighbourhoods tend to have lower vegetation cover ($R^2$ = 0.15, F= 5.97, p<0.05, Fig. 4, Table 2). Newly developed neighbourhoods typically with lower population densities showed a high share of vegetation cover (Supplementary Fig. 2 and Supplementary Table 1). Habitat quality also showed a positive logarithmic response to vegetation cover ($R^2$= 0.29, F = 12.23, <0.01).

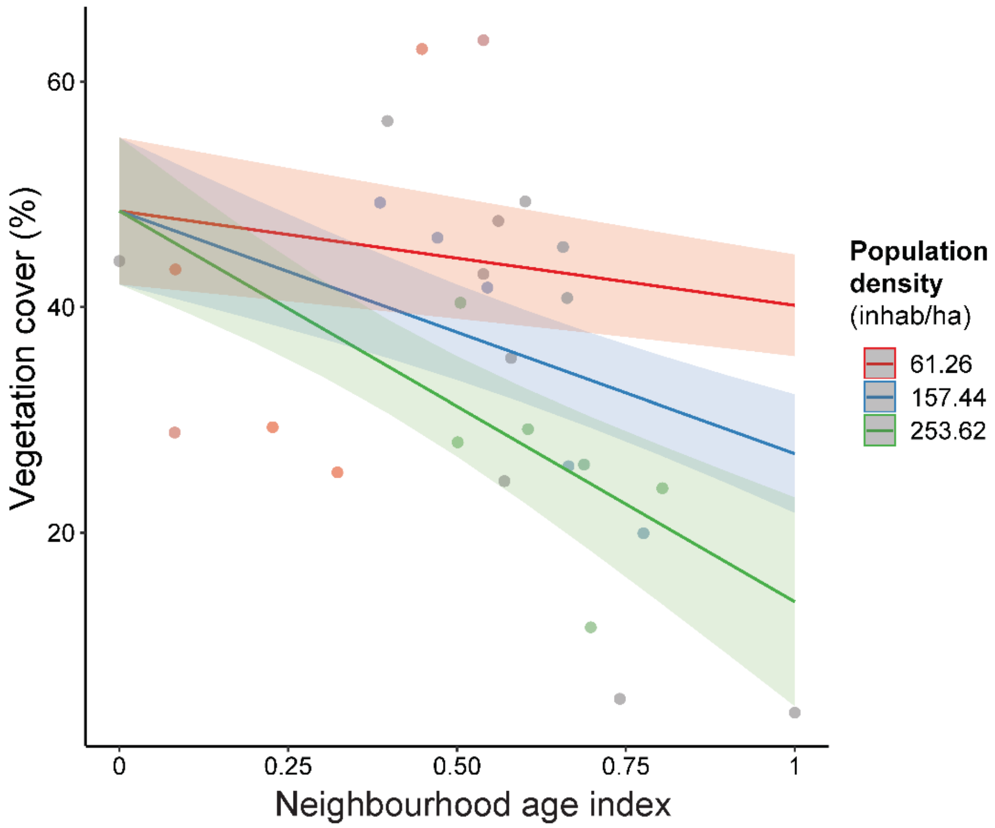


**Figure 4. Vegetation cover (% of neighbourhood area) linear models in response to the interaction of neighbourhood age index and population density.** Shaded areas represent the 95% confidence interval. Population density (inhab/ha) categories grouped by three terms defined by the mean value and +/-1 standard deviation (Aiken & West, 1991). Model information provided in Table 1.

### 3.3.3. *Luxury and legacy effects on ecosystem services*

The estimated provision of regulating ES based based on the application of the i-Tree tool (Nowak, 2000) for  in Vitoria-Gasteiz in 2015, indicated that urban trees accounted for 185,145 $m^3$/yr. of transpired water, 30,652 $m^3$/yr. of avoided runoff, 14.1 ton/yr. of removed air pollutants and 617.0 ton/yr. of carbon sequestration respectively (Supplementary Table 3). Since these four regulating ES were spatially co-ocurrent, we aggregated them into a single ES index with a value ranging from 0 (no provision) to 100 (highest provision). Moran's I showed no significant autocorrelation in the spatial distribution of the aggregated ES index (Moran's I: z-score = 0.34, p = 0.72).

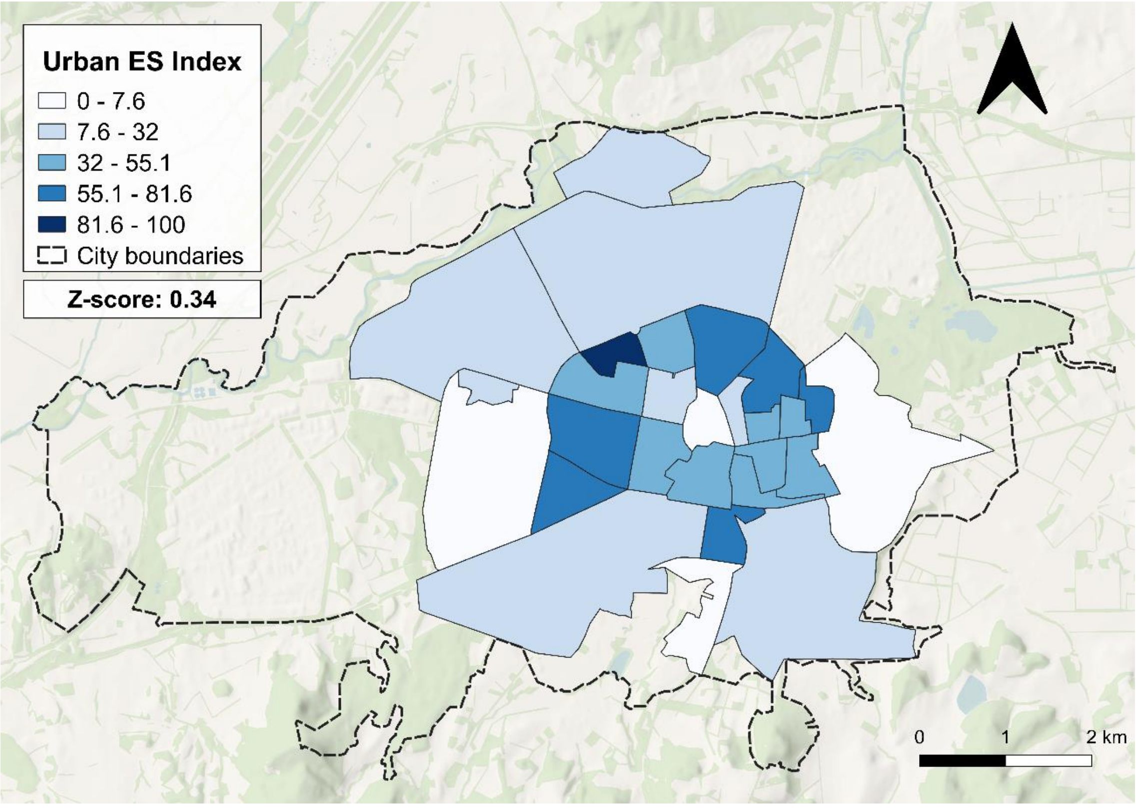


**Figure 5. Spatial distribution of the regulating ES index supplied by urban trees at the neighbourhood scale.** Index values were classified using natural breaks for spatial representation: by maximising variance between classes and minimising variance within each class. Background from © MapTiler (www.maptiler.com) and © OpenStreetMap (CC BY-SA 2.0).

The regulating ES index was higher in older neighbourhoods surrounding the historical mediaeval neighbourhood of Casco Viejo (old town) (Fig. 5) with higher canopy cover (Supplementary Fig. 1 and Fig. 2). Conversely, the lowest share of the urban tree canopy, and thus the lowest provision of regulating ES was registered in Casco Viejo, and newer neighbourhoods located on the outskirt's areas of the city where tree-based green infrastructure is more recent (Fig. 5).

The ES index was positively correlated with vegetation cover, including trees and herbaceous vegetation (Table 3). We found no direct correlation between the ES index regarding luxury or legacy effect predictors alone, *i.e.* high educational attainment and neighbourhood age respectively (Table 3). Yet, we found a legacy effect on regulating ES mediated by vegetation cover. That is, the regulating ES index was positively correlated with neighbourhood age when interaction with vegetation cover is included, where older neighbourhoods provide more regulating ES, especially in neighbourhoods with more vegetation cover ($R^2$= 0.57, F=37.2, p<0.001, Fig. 6, Table 3).

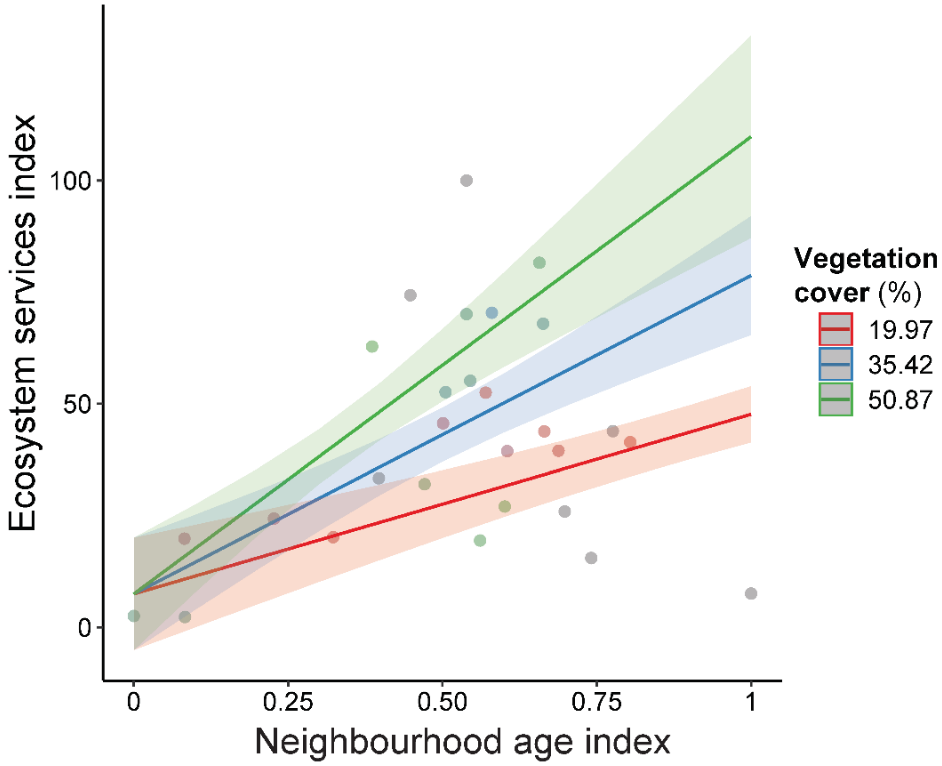


**Figure 6. Ecosystem services (ES) linear models in response to the interaction of neighbourhood age index and vegetation cover (% per neighbourhood).** Shaded areas represent the 95% confidence interval. Neighbourhood development age is grouped by three terms defined by the mean value and +/-1 standard deviation (Aiken & West, 1991). Model information provided in Table 2.

**Table 3. Linear models of regulating ecosystem services index to socio-environmental characteristics.** Model and parameters of regulating ecosystem services index (ES) in response to: high educational attainment (% of the population with high education attainment by neighbourhood over the total population, proxy of luxury effect), neighbourhood development age index (values ranging from 0 – newest to 1 – oldest), habitat quality index (values ranging from 0 - lowest to 1 - highest), vegetation cover (% per neighbourhood), population density (inhab/ha), and interactions of vegetation cover with population density. Model formula for each predictive variable, adjusted variance explained ($R^2$), F-statistic (F), significance level (p) and standard error (Std. Error). N= 28, n.s.: non-significant.

| Ecosystem Services | | | | |
|---|---|---|---|---|
| **Model** | **Adj. $R^2$** | **F** | **P** | **Std.Error** |
| *ES = 50.95 - 15.87 * High education* | 0.05 | 2.65 | n.s | 24.05 |
| *ES = 27.05 + 28.22 * Neighbourhood age* | 0.03 | 1.85 | n.s | 24.39 |
| *ES = 57.50 - 108.29 * Habitat quality* | 0.10 | 4.03 | n.s | 23.49 |
| *ES = 15.11 + 0.75 * Vegetation cover* | 0.19 | 7.39 | <0.05 | 22.28 |
| *ES = 37.84 + 0.02 * Population density* | -0.02 | 0.25 | n.s | 25.12 |
| *ES = 7.47 + 2.0 * Neighbourhood age:Vegetation cover* | 0.57 | 37.2 | <0.001 | 16.19 |

## 3.4. Discussion

Our results support the 'luxury effect' hypothesis, which suggests that urban biodiversity is associated with wealthier neighbourhoods, which tend to have higher education attainment (Schell et al., 2020;Kirkpatrick et al., 2011). Species richness is positively correlated with high educational attainment (Table 2, Fig. 2). This correlation is particularly strong in neighbourhoods with higher levels of educational attainment and higher-quality habitats, emphasising the luxury effect. Previous research suggests that this luxury effect can be attributed to human and other species' preferences for environmentally desirable areas while avoiding environmental burdens such as pollution (Magle et al., 2021; Schell et al., 2020). Wealthier social groups, proxied as those group with higher educational attainment in this study, hold significant influence over local UGS investments and have the capacity to shape land use planning according to their interests (Grove et al., 2014). The greater resources of wealthier households enable them to allocate more towards private green spaces, resulting in enduring legacy effects that reinforce the luxury effect and bring about long-term changes in UGS composition (Leong et al., 2018; Roman et al., 2018).

We also found an inverse legacy effect on urban biodiversity, with declining species richness as neighbourhood development age increases (Fig.2). This is likely attributable to the dynamic influence of neighbourhood development on the quality of urban ecosystems. UGS are inherited in the landscape, reflecting legacies of past management, greening movements, and changing socioeconomic conditions, that affect neighbourhood development and people's behavioural changes (Roman et al., 2018). The historical decisions and urban planning employed in the past have a lasting effect on current vegetation cover patterns, which may not be easily predicted based on present data (Kendal et al., 2012; Kirkpatrick et al., 2011; Boone et al., 2010). Vitoria-Gasteiz has historically prioritized the conservation of older neighbourhoods while preserving the design and medieval architecture of the old town, with the creation of UGS being a more recent priority (since the 1990s). Evidence from the city shows that, although it has a relatively long history of green planning, UGS close to the urban core, where older neighbourhoods are located, has resulted in lower ecological quality, but still provides multiple ES such as recreation, cooling or runoff control (Aznarez et al., 2022; CEA, 2014). Other landscape descriptors were found to influence urban biodiversity, in particular, habitat quality, vegetation cover, and population density at neighbourhood scale (Fig. 2). These are key factors that had an effect on the magnitude of both luxury and legacy effects: while the luxury effect on biodiversity was stronger in areas of high habitat quality, the inverse legacy effect was more pronounced with increased population density. This implies complex socio-ecological interactions that influence luxury and legacy effects, including discourses, narratives about environmental management, and the role of social power and political influence. Hence, it is essential to understand the socio-political context of the city in order to gain insight into the ideological motivations behind greening efforts (Neidig et al., 2022; Pickett et al., 2016).

Our findings provide useful information on how both effects can be further influenced by ecological and social descriptors. In order to improve habitat quality and greening to enhance biodiversity, urban management should focus on the ecological status of older neighbourhoods in the urban core. Given the logarithmic response of habitat quality to species richness (Fig. 2), increases of habitat quality at the lowest ranges would have comparatively higher effects on species richness than at higher ranges. The most biodiverse neighbourhoods (Fig. 3), were mainly developed between the late 1970s and early 2000s (Supplementary Fig. 1 and Fig. 3). During this period, urban policies were implemented to drive greening strategies to reverse the damage from industrial activities and urban expansion (Supplementary Table 2) (Neidig et al., 2022). This suggests that past management practices and planning history continue to strongly influence present outcomes. Time is a key factor in urban biodiversity dynamics, particularly for species with slower colonisation times (Clarke et al., 2013). Thus, legacy effects must be considered in a broader context, incorporating land-use management history to understand how past management actions has shaped current landscape characteristics (Roman et al., 2018).

Our findings also indicate that, when solely considering vegetation cover (in both public and private land), there is no evidence to support the luxury effect hypothesis observed in previous studies (Jenerette et al., 2013; Kendal et al., 2012; Kirkpatrick et al., 2011; Hope et al., 2003). This suggest that greener neighbourhoods do not necessarily attract residents with higher educational attainment, nor do they have increased habitat quality due to their 'green' character (Table 2). Hence, we infer that biodiversity was a better indicator than vegetation cover when analysing the luxury effect. This could be attributed to other factors not taken into account here, such as management choices involving the selection of native or non-native plant species, or preferences of some species over others.

We found that population density negatively affects species richness (Fig. 2) and exacerbates the negative effects of neighbourhood age on biodiversity and vegetation cover (Fig. 3). Population density and neighbourhood age are correlated (Supplementary Fig. 4), suggesting that older neighbourhoods tend to be more densely populated, having a detrimental effect on biodiversity and vegetation cover. This can be attributed to the design of historical old towns, like our case study, characterised by narrow streets and compact forms, compared to later developments like suburbs and urban sprawl areas, which tend to be less dense (Adelfio et al., 2018). Population density influence vegetation trends over time, by affecting available canopy space and limiting the growth of new individuals of trees (Locke & Grove, 2016; Clarke et al., 2013). The relationship between population density and vegetation cover is complex, with both positive and negative associations observed (Locke & Grove, 2016; Grove et al., 2014). For instance, while population density and income have been negatively correlated with vegetation cover (Clarke et al., 2013), vegetation cover has been positively linked to more housing units (Grove et al., 2014). These mixed outcomes suggest the importance of considering multiple factors and their interactions, such as education, land use management legacies, and income, when examining population density. Our study highlights that population density not only directly affects biodiversity negatively but also amplifies the negative impact of neighbourhood age on biodiversity and vegetation cover (Figs. 2, 3, 4).

We also found an inverse legacy effect between vegetation cover and neighbourhood age, that as per species richness was mediated by population density (see Fig. 4), implying that older and more populated neighbourhoods have less vegetation cover. Additionally, there was a decreasing trend of herbaceous vegetation cover towards the urban core (Supplementary Fig. 1), possibly due to urban densification (Clarke et al., 2013; Hope et al., 2003). Less populated areas, with lower housing density maintain a relatively high percentage of both herbaceous and tree cover (Fig. 4). Given these findings, urban planners should prioritise strategies that conserve and enhance vegetation in older and more compact areas. This is particularly crucial considering the effects of climate change in cities, which lead to environmental burdens like air pollution and heat stress, exacerbating environmental and climate injustices (Lenton et al., 2023). Implementing interventions such as green roofs, green walls, and street trees becomes key, especially in densely populated urban areas with limited space (Langemeyer et al., 2020). These interventions not only mitigate the challenges but also contribute to tackle the unequal distribution of greening. Moreover, prioritising green over impervious cover is vital to prevent the negative impacts of urban densification on biodiversity and vegetation, which worsen environmental injustices. To guide evidence-based strategies in sustainable urban development, further research is needed to understand how population density, land use, and neighbourhood age influence biodiversity and vegetation cover.

Moving on to the effects on regulating urban ES, our research tested the luxury and legacy hypotheses for ES for the first time. Higher educational attainment and neighbourhood development age were not associated with regulating ES provision when considered as single terms (Table 3). However, we observed a strong association between vegetation cover and regulating ES provided by urban public trees. Nevertheless, there was no evidence that increased urban biodiversity leads to a larger flow of these regulating ES. The distribution of regulating ES seemed to be influenced by management and land use legacies, indicated by neighbourhood development age, in interaction with vegetation cover (Fig. 6). Our models that included richness and habitat quality were statistically non-significant (Table 3). This suggests that urban tree planting schemes in the past may have been focused at providing shading or aesthetic purposes rather than enhancing plant diversity. Our results support that older neighbourhoods with higher canopy cover yield more regulating ES, even if they have less biodiversity (Baró et al., 2019). The i-Tree Eco accounts for tree structural traits (*e.g*. DBH, height, and crown size), which have a greater influence on regulating ES than tree diversity (Graça et al., 2017). Urban trees typically have an average lifespan of between 19 and 28 years, and the mortality rate of young trees is generally high (Roman & Scatena, 2011). As such, older neighbourhoods with larger, more mature trees provide higher regulating ES when compared to newer neighbourhoods with a higher proportion of young trees and herbaceous cover (Fig. 5). This pattern may be influenced by age, suggesting that the older and greener the neighbourhood, the stronger the relationship with regulating ES provision (Fig. 6). This could be due to the long-term establishment of trees (Clarke et al., 2013), with large old trees being carefully maintained for their cultural and heritage values (Lindenmayer & Laurance, 2017; Thaiutsa et al., 2008). Our findings suggest that ES provision is concentrated in areas surrounding the urban core, where a higher proportion of public older trees are located. However, this also implies that newer neighbourhoods offer potential opportunities to enhance green cover, biodiversity conservation and ES provision in the future, as tree canopy cover increases with age**.** Together, these findings suggest that UGS identity is markedly different between older and newer neighbourhoods, reflecting relevant legacy effects patterns.

This research illustrates the multidimensional character of the socioeconomic attributes associated with urban biodiversity, vegetation and ES. While the associations described here are correlative and do not infer direct causation, our findings emphasise the relevance of the luxury effect phenomenon. We have also shown that the legacy of prior investment and management of UGS has a considerable effect on how benefits from the urban canopy are distributed within a city. Urban planning and the implementation of green infrastructure must consider these effects to address climate adaptation, biodiversity conservation, and environmental justice. This is especially important as the people capable to pay the increased property and rent prices associated with more biodiverse and green areas, and thus investing in higher adaptive capacity to climate change, are high-income residents (Anguelovski et al., 2019, 2020; Clarke et al., 2013; Mansfield et al., 2005).

Our results support the notion that past land use planning and development, with or without planning, influence the availability and quality of UGS, thus impacting the potential effectiveness of green infrastructure, including the exposure and access to regulating ES, which are key for climate adaptation (Goodwin et al., 2023). This has consequences for residential areas, which often reflect lifestyles associated with group identities and social status (Grove et al., 2014; Pickett et al., 2011; Boone et al., 2010). Management decisions, such as allocating resources for urban green infrastructure, can further drive neighbourhood differentiation and exacerbate existing environmental injustices (Anguelovski et al., 2020). This highlights the importance of considering luxury and legacy effects in urban planning strategies from an environmental justice perspective to achieve a better balance in the distribution of UGS and associated ES. This has important implications for environmental justice, as the way urban planning for green infrastructure is implemented will determine who ultimately benefits from urban investments. This includes recognising and accounting for the socioeconomic structure embedded in urban planning practices, which can drive systemic and asymmetric power relationships that reproduce environmental injustices (Anguelovski et al., 2020).

Expanding and conserving UGS as part of urban green infrastructure without prioritising access to their benefits for vulnerable groups can exacerbate environmental injustices and trigger luxury effects (Wolch et al., 2014). It is crucial to prioritise access to these benefits for those who need them the most, particularly for individuals and communities who are most exposed to hostile environmental conditions. Achieving this requires considering socio-environmental variables such as population density, which can affect the responses of luxury and legacy effects on the distribution of UGS and associated ES. Therefore, these variables should be included in management strategies to better predict the impact of these effects on the distribution of UGS and ES (Clarke et al., 2013). This, in turn, can have an effect on the success or failure of biodiversity and ecological quality enhancement policies across urban green infrastructure and multiple ES (Cohen et al., 2012; Strohbach et al., 2009). Urban managers often perceive biodiversity conservation as a co-benefit of other actions (*e.g.* opportunities for recreation, or mitigating extreme events), rather than a strategic priority systematically planned to enhance ecological integrity in the urban context (Oke et al., 2021).

Planning urban green infrastructure becomes a key determinant of biodiversity patterns and ecosystem functions, as well as providing access to and exposure of urban nature and its associated ES. Older neighbourhoods with early designed UGS have distinct structural and natural features that influence their ability to support biodiversity and provide ES. Thus, UGS should be evaluated based on their specific characteristics and capacity to support ES, which are influenced by historical perceptions of green spaces. In the context of Vitoria-Gasteiz, the city's urban form and historical characteristics present a challenge when it comes to integrating green infrastructure into older areas. This demands the implementation of green-grey infrastructure integration approaches. However, in newer areas of the city, there is an opportunity to prioritize a higher coverage of trees to facilitate a greater flow of ES in the future (Baró et al., 2019). By doing so, both the legacy effect and the population density effect can be taken into account, leading to a more equitable distribution of UGS and their associated ES throughout the city.

To reduce inequalities, strategies should focus on increasing learning opportunities in and about nature, amplifying its social value, investing in long-term biodiversity enhancement plans, and implementing conservation and restoration actions to protect and improve vegetation cover. These measures could foster community engagement in the creation and preservation of UGS, while also mitigating the impacts of population density and supporting the success of ecological quality enhancement policies. However, it is important to acknowledge that social inequalities may persist. Residents in economically disadvantaged areas may have limited resources to allocate towards private green spaces, hindering their participation in green initiatives. Addressing resource constraints alongside educational efforts is crucial to achieve equitable access to UGS and mitigate green inequalities. Further research is needed to identify the specific types of regulating ES provided by different tree species, which could help prioritize tree planting schemes in coordination with urban biodiversity enhancement plans. With these considerations in mind, our study opens up new avenues to consider luxury and legacy effects and unravel the relationship between urban biodiversity, ecological functions, and ES, along with its cascading effects on urban dwellers' well-being.

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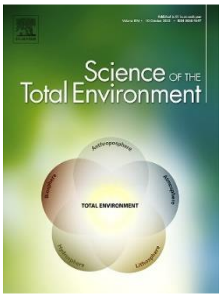
# Chapter IV

## Ecosystem service mismatches evidence inequalities in urban heat vulnerability

### Abstract

Exposure to heat poses a pressing challenge in cities, with uneven health and environmental impacts across the urban fabric. To assess disparities in heat vulnerability and its environmental justice implications, we model supply-demand mismatches for the ecosystem service (ES) urban temperature regulation. We integrated remote sensing, health, and socio-demographic data with Artificial Intelligence for Environment and Sustainability (ARIES) and geographical information system tools. We computed composite indicators at the census tract level for urban cooling supply, and vulnerability to heat as a measure of demand. We do so in the context of the mid-size city of Vitoria-Gasteiz, Basque Country (Europe). We mapped relative mismatches after identifying and analysed their relationship with socio-demographic and health factors. Our findings show disparities in heat vulnerability, with increased exposure observed among socioeconomically disadvantaged communities, the elderly, and people with health issues. Areas associated with higher income levels show lower ES mismatches, indicating higher temperature regulation supply and reduced heat vulnerability. The results point at the need for nature-based heat mitigation interventions that especially focus on the more socioeconomically disadvantaged communities.

**Keywords:** *Urban heat island; Ecosystem services mismatch; Vulnerability; Environmental justice*

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## 4.1. Introduction

Over the past decades, heatwaves have increased in frequency, intensity, and duration (Cowan *et al*., 2014; Fischer & Schär, 2010), with expectations of further intensification due to accelerated global warming (Ballester *et al*., 2023). Urban areas are particularly vulnerable to rising global temperatures due to the urban heat island (UHI) effect, which results in substantially higher temperatures in dense urban environments compared to suburban-rural areas (Marando *et al*., 2022; Massaro *et al*., 2023). The synergistic effects of heatwaves and UHIs affects human health and the urban environment (Grimm *et al*., 2008; Chen *et al*., 2023), constituting a major hazard for urban residents with far-reaching health and well-being impacts (IPCC, 2021; Kuras *et al*., 2015). For instance, elevated urban temperatures can reduce work productivity, impair cognitive performance and learning capacity (Lenton *et al*., 2023), while also contributing to increased morbidity, mortality rates, and maternal health risks (Ballester *et al*., 2023; Lenton *et al*., 2023).

With the expansion and densification of urban environments, there is a an increased interest in the provision of urban ecosystem services (ES) (Liu *et al*., 2020) through the implementation of urban green infrastructure strategies, particularly regarding urban heat mitigation (Francoeur *et al*., 2021; Langemeyer *et al*., 2020; Wolch *et al*., 2014). Urban green infrastructure (UGI) helps reduce land surface and air temperatures during hot periods by providing shade, limiting the absorption of solar radiation in artificial surfaces, and promoting evapotranspiration (Errea *et al*., 2023; Massaro *et al*., 2023).

Urban development shapes the distribution and balance between green and grey infrastructure, influencing heat absorption and release (Schell *et al*., 2020; Marando *et al*., 2019). Inequities in exposure and access to UGI and associated ES can also be driven by socioeconomic and urban planning factors whose interaction further exacerbate disparities (Aznarez *et al*., 2023; Baró *et al*., 2019). These disparities hinder the equitable provision of urban temperature regulating ecosystem service (TR-ES) while recreating so-called “luxury effects” on canopy cover (Jenerette *et al*., 2013). In turn, this can contribute to a complex mosaic of temperatures within cities that may expose residents to a *continuum* of heat, ranging from thermal comfort to potential hyperthermia (Lenton *et al*., 2023; Jenerette *et al*., 2011). Social inequities amplify heat-related disparities and their health implications, particularly among socioeconomically disadvantaged communities (Wang *et al*., 2023; Schell *et al*., 2020; Chakraborty *et al*., 2019).

Equitable access to UGI for mitigating heat vulnerabilities requires evaluating the distribution of the supply of TR-ES and the societal demand for such ES. We refer to 'supply' as the capacity of urban ecosystems to provide TR-ES, and 'demand' as the societal need to reduce heat-related vulnerability and risk through UGI (Burkhard & Maes, 2017). Despite increasing recognition, understanding and mapping ES demand remain challenging due to this multifaceted nature, influenced by factors like access, technological influences, socioeconomic conditions or demographics (Dworczyk & Burkhard, 2021; González-García *et al*., 2020). Unlike traditional ES assessments that rely on indirect inferences of demand by considering the magnitude of pressures and assessing the population exposed to these pressures (Baker *et al*., 2021; Wolff *et al*., 2015), our research integrates the multifaceted character of demand. We contribute to addressing this gap by integrating the demand side of TR-ES into our assessment, considering that heat vulnerability is unevenly distributed among urban residents, reflecting different TR-ES needs. By adopting the heat vulnerability assessment framework from the IPCC (2007), encompassing exposure, sensitivity, and adaptive capacity, we quantify heat vulnerability as a proxy for estimating TR-ES demand (Langemeyer *et al*., 2020; Mallen *et al*., 2019; Bradford *et al*., 2015).

Through spatially explicit assessments of TR-ES supply-demand mismatches, we aim to provide evidence-based information that can guide urban planning, addressing environmental inequities and identifying priority intervention areas for reducing heat-related vulnerabilities. This aligns with existing literature emphasizing the importance of such assessments in guiding effective urban interventions (Baró *et al*., 2015; Herreros-Cantis & McPhearson, 2021). We refer to TR-ES mismatch as the difference in either the quality or quantity between TR-ES supply and demand (Sebastiani *et al*., 2021; Geijzendorffer *et al*., 2015). We specifically aim to i) develop an integrated modelling approach that combines remote sensing, field data, and socio-demographic information to assess TR-ES supply and demand; ii) analyse the spatial distribution of the TR-ES supply-demand mismatches as a way to target priority intervention areas; iii) identify the sociodemographic and health factors that influence TR-ES supply–demand mismatches.

## 4.2. Methods

### *4.2.1. Study area*

Our study focused on the mid-sized city of Vitoria-Gasteiz (Basque Country), located in the North of the Iberian Peninsula (Fig. 1) and internationally known by its ambitious urban greening policies, as reflected by the 2012 European Green Capital award. Vitoria-Gasteiz has numerous green and blue spaces designed as part of its publicly accessible UGI, including a 35 km-long green belt from the 1990s (Fig. 1). The city, with a population of 253,672 inhabitants (INE, 2022), experiences a temperate and humid climate. The average annual temperatures have ranged between 4.4 ºC and 17.1 ºC over the period from 1991 to 2021 (Errea *et al*., 2023). Surface UHI has been rising from approximately 5°C in 2012 to 10.9°C in 2022 (Olazabal *et al*., 2012; Errea *et al*., 2023). The natural value of some features within the green belt, notably the Salburua wetlands, the Zadorra River and Vitoria Mounts, were included in the Natura 2000 network and protected as community interest sites. In addition, the current local urban green infrastructure planning aims at the creation of nature-based solutions that also contribute to mitigating urban heat in strategically selected areas (CEA, 2014).

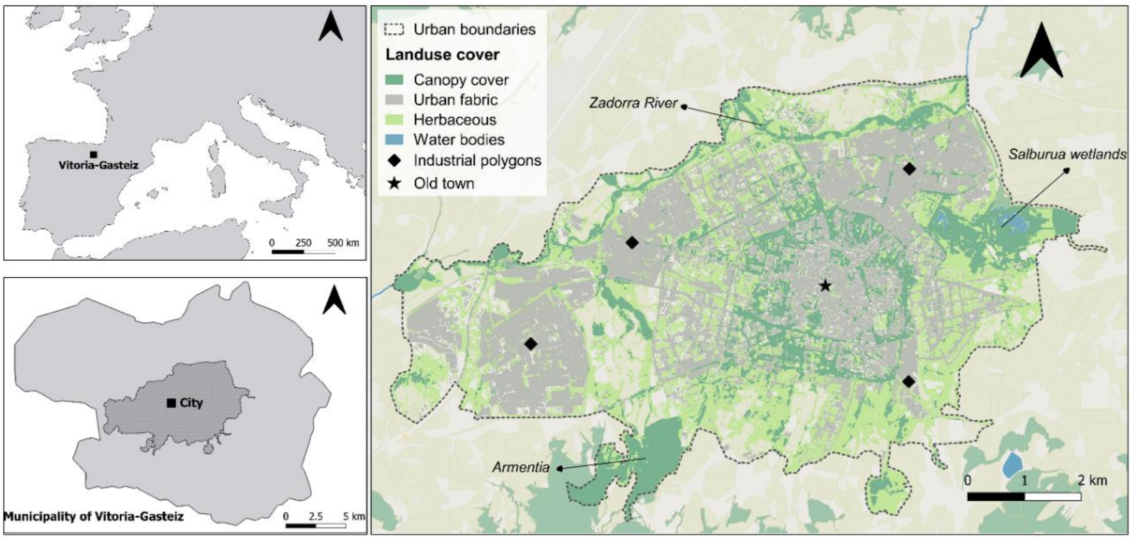


**Figure 1.** Study area of Vitoria-Gasteiz with its location within the urban and municipal boundaries and main land covers, such as urban fabric, canopy cover, herbaceous cover, and water bodies. Black diamonds indicate industrial polygons and the black star indicates the historical city centre. Arrows indicate the major natural landmarks of the green belt.

### *4.2.2. Analytical framework*

To address compatibility and transferability challenges in ES assessment models (Balbi *et al*., 2022), we adopted an integrated modelling approach based on FAIR (Findable, Accessible, Interoperable, and Reusable models and data) principles (Wilkinson *et al*., 2016). By combining the Artificial Intelligence for Environment & Sustainability (ARIES) modelling platform (Villa *et al*., 2014, 2017) with GIS tools, we generated results that are accessible to urban planners. ARIES modelling was performed on k.LAB, an advanced modelling web-based software that connects spatial data and model components through semantics and machine reasoning (Marquez-Torres *et al*., 2023; Capriolo *et al*., 2020; Villa *et al*., 2014).

Our integrated modelling approach was designed to comprehensively assess the spatial distribution of TR-ES supply and demand (Fig. 2). We used composite indicator that combines the ecological, biophysical and societal aspects of TR-ES supply and demand into a unitless, normalized value (Alam *et al*., 2016). To quantify the supply of urban TR-ES, we examined the ecological properties and functions of UGI (Fusaro *et al*., 2023; Sebastiani *et al*., 2021), including habitat quality as indicator of ecological integrity, and vegetation cover as a proxy for shading and evapotranspiration capacity. For assessing ES demand, we followed the IPCC AR5 framework (IPCC, 2014), which considers risk as the combination of urban heat hazard, exposure and associated vulnerability (Wang *et al*., 2023; Hsu *et al*., 2021). To assess these components of ES demand, we first relied on remotely sensed data to generate a Land Surface Temperature (LST) map, as input for assessing the spatial distribution of heat hazards. We then assessed heat exposure by overlaying the spatial distribution of urban heat hazard (*i.e*. LST) with population density and land cover data at the census tract level. Then we built a Heat Vulnerability Index (HVI), as a function of exposure, sensitivity and adaptive capacity, in order to account for vulnerability to urban heat. Lastly, the assessment of TR-ES mismatches involved subtracting the supply from the demand at the census tract level. To operationalize this workflow (Fig. S1, Supp. Mat.), we developed different modules within the ARIES platform, along with an associated ontology and semantics. We also incorporated different data types to ARIES including environmental and socio-demographic data across different spatial scales.


**Figure 2.** Conceptual framework on urban temperature regulation ES (TR-ES) supply and demand, and their mismatches based on composite indicators. TR-ES supply is evaluated based on the ecological attributes of urban green infrastructure and TR-ES demand is based on urban heat vulnerability assessments. Spatial mismatches among TR-ES supply-demand underscores its significance for (distributive) environmental justice.

### *4.2.3. Urban temperature regulation (TR-ES) supply*

To quantify TR-ES supply we employed a composite indicator approach at the census tract level (Alem *et al*., 2016). We based our assessment on three indicators: i) the first indicator refers to the canopy cover, as the proportion of ground area covered by layers of leaves and woody vegetation when viewed from above (Locke *et al*., 2021), expressed as a percentage and used as a proxy for the vegetation shading effect. The indicator was estimated from PNOA LiDAR data (2015-2023) at 1m resolution (IDEE, 2023) (Fig. S2, Supp. Mat.); ii) the second indicator is the mean Normalized Difference Vegetation Index (NDVI) and was obtained from Landsat 8 images for the summer period of 2021-2022. NDVI values above 0.25 were considered and validated by spatial overlay with orthophotos and urban land use maps at a resolution of 10m (CEA, 2020). NDVI has been previously used as a proxy of TR-ES provision (Sebastiani *et al*., 2021; Marando *et* *al*., 2019); iii) the third indicator is based on a 10m-resolution raster map of the spatial distribution of UGI habitat quality (Aznarez *et al*. 2022). Higher habitat quality indicates higher productivity and ecological integrity, while lower habitat quality suggests habitat fragmentation and the influence of threats and edge effects, reducing the evapotranspiration and hence cooling potential of vegetation (Aznarez *et al*., 2022; Ewers & Banks-Leite, 2013). TR-ES supply for urban temperature regulation is calculated by Eq.2 adapted from Sebastiani *et al*. (2021):

$$TRES = Cc + NDVI + HQ \quad (2)$$

where Cc represents the percentage of canopy cover by census tract, NDVI refers to the mean NDVI calculated for vegetation from 2022 summer Landsat 8 imageries and HQ represents the mean habitat quality of UGI.

### *4.2.4. Urban temperature regulation (TR-ES) demand*

The societal demand for urban temperature regulation was estimated based on the spatial distribution of the heat vulnerability index (HVI). To identify priority areas for heat-related risk reduction, we used a unitless characterization of social factors. We considered vulnerability to heat as a subtype of social vulnerability, defined by IPCC AR5 as the predisposition of social groups to adverse impacts including the sensitivity to harm and lack of capacity to adapt (Wisner, 2016). We built and calculated the HVI for Vitoria-Gasteiz for each census tract following Reid *et al*. (2009). To this, we summed up the exposure level ($E_i$) and sensitivity level ($S_i$) to heat and then subtracted the adaptive capacity level ($A_i$) as given in Eq. (1):

$$HVI = Ei + Si \mp Ai \qquad (1)$$

For the development of the HVI specific to Vitoria-Gasteiz, we used spatial data on socioeconomic and demographic variables at the census tract level which was the finest resolution at which data was accessible and is generally the relevant spatial scale to urban planning and policies (Table S1, Supp. Mat.).

Heat exposure refers here to the urban population being exposed to high temperatures and related health impacts. This was assessed using LST (Fig. S4, Supp. Mat.) and population density as indicators (Wang *et al*., 2023). High-density urban areas with elevated temperatures are known to pose increased heat exposure risks (Manoli *et* *al*., 2019). The spatial distribution of exposure was mapped (Table S2, Supp. Mat.) by incorporating four overlapping spatial layers into our analysis: i) a raster map of population density to map the hotspots of anthropogenic heat, serving both as a measure for heat exposure and an aggravating factor for the UHI effect (Manoli *et al*., 2019; Jin *et al*., 2019); ii) NDVI to depict the density of vegetation, with areas with significantly low NDVI (<0.1) indicating bare soil, which contributes to LST intensity and therefore higher heat exposure (Marando *et al*., 2019; Tesfamariam *et al*., 2023); iii) LST to estimate the extent and magnitude of urban heat; and iv) land cover data to identify urban impervious surfaces.

To calculate sensitivity, we considered factors that could either amplify or mitigate the impact of heat exposure. These factors encompassed residents' age and health aspects (Mallen *et al*., 2019; Kuras *et al*., 2015; Reid *et al*., 2009). Our data included age-related metrics (percentage of the population aged ≤14 and ≥65) and health indicators, including respiratory diseases, diabetes and pulmonary conditions in both, male and female populations (see detailed information in Table S1, Supp. Mat.). For the health variables, we employed the Standardized Mortality Ratio (SMR), which measures the likelihood of higher-than-expected mortality rates within a specific group compared to a reference population. The SMR data were obtained from the Basque Mortality Atlas for the period 2013 - 2017, and the age-related metrics were sourced from the EUSTAT, the Basque Bureau of Statistics (2021).

Finally, to map heat-related adaptive capacity, we examined the capacities and resources accessible to city residents to cope with heat exposure. We identified education attainment, income, and foreign origin as potential factors influencing adaptive capacity to heat (Kuras *et al*., 2015; Reid *et al*., 2009). Specifically, above the median annual household gross income (€34,473) and high educational attainment (*i.e*. residents with and above tertiary education) were assumed to be factors that can enhance adaptive capacity, potentially leading to reduced exposure to heat (Wang *et al*., 2023). Conversely, lower income and education attainment (*i.e*. residents with primary or no formal education), along with foreign origin (*i.e*. from Asia, Africa and America), were expected to indicate a lower adaptive capacity. The latter can be attributed due to difficulties in getting sufficient support from educational and healthcare institutions, as well as linguistic barriers for accessing knowledge and information (Keeler *et al*., 2019; Nayak *et al*., 2018).

To allow for comparisons, we standardized the initial set of vulnerability-related variables to have an average of 0 and a standard deviation of 1. We then assessed the correlation between the selected variables by calculating pairwise Spearman's correlation coefficients (Fig. S1, Supplementary data) using the "ggally" package in R (Schloerke *et al*., 2022). To simplify the interpretation and mitigate the influence of outliers, we categorized each variable into six distinct scores based on their standard deviations, *i.e*. z-score values (Table S3, Supp. Mat.). The specific scores were assigned to each category, ranging from 1 (*i.e*. minimal vulnerability) to 6 (*i.e*. highest vulnerability).

Drawing from Reid *et al*. (2009), and given the limited understanding of the individual impacts of each identified factor on vulnerability to urban heat, we assumed an equal weight and aggregated the assigned scores for each variable. This aggregation produced a cumulative HVI value for each census tract. To ensure that higher values align with increased vulnerability, the considered variables were adjusted. If a variable had a positive impact suggesting lower vulnerability, it was computed by subtracting it to contribute to the cumulative measure of increased vulnerability. To spatially represent the HVI values at the census tract level, these were normalized between 0 to 1 to also make them more easily comparable.

#### *4.2.5. Spatial TR-ES supply-demand mismatch analysis*

Each census tract was assigned TR-ES supply and demand values ranging from 0 (lowest) to 1 (highest). Supply indicated the normalized estimation of TR-ES attributed to UGI, while demand reflected the societal need for exposure reduction and the calculated HVI. To map TR-ES supply-demand mismatches, we adopted a pragmatic approach involving the subtraction of the supply from the demand index to estimate a TR-ES budget for each census tract (following Burkhard *et al*., 2012; Herreros-Cantis & McPhearson, 2021). The resulting values spanned from 1 to -1, with a value of 1 representing the highest potential for mismatch, indicating high TR-ES demand in the face of low supply. Lower values indicated areas where the supply of TR-ES exceeded potential demand, with values near 0 suggesting a balance between potential demand and supply. To identify the patterns driving supply–demand mismatches, we used the Getis-Ord Gi* hotspot analysis within ArcMap 10.7.1 (ESRI). This analysis consists in grouping census tracts into (seven) clusters based on their z-scores and p-values where high or low values exhibited spatial clustering based on their initial supply-demand mismatch value. These clusters represent cold (very low values) and hot (very high values) spots with specific confidence intervals (90%, 95%, and 99%), and statistically non-significant mismatches.

To discern the individual influences of the selected variables comprising the HVI index and identify the key socio-demographic and health variables influencing the spatial mismatches in TR-ES, we used generalized linear models (GLMs). The mismatch values were associated with the combination of these variables by a stepwise regression approach (Table S1, Fig.S3, Supp. Mat.). We then selected the model with the strongest overall fit and most parsimonious, and checked the model assumptions by visual inspection of the residual plots. To describe how socio-demographic and health-related variables were associated with the generated clusters (cold and hot spots), we applied a principal component analysis (PCA). We then identified the contribution and significance of each variable that differentiated the cold and hot clusters performing a similarity percentage analysis (SIMPER). These analyses were conducted using R, version 1.4.1717 (R Core Team 2023), with specific packages dedicated to: 'vegan' for multivariate statistics, 'tidyverse' for data handling, 'sjPlot' for representation, and 'performance' for model validation.

## 4.3. Results

### *4.3.1. Spatial patterns of urban exposure to heat hazard*

To analyse heat spatial distribution, we mapped summer LST. Temperature ranged between 24.93 - 55.80 degrees Celsius (Fig. S4, Supp. Mat). The zones with higher temperatures were primarily observed in industrial areas that are characterized by heat-absorbing materials and sparse vegetation cover. The heat exposure map (see Fig. S5, Supp. Mat.), shows moderate to high values (ranging from 0.4 to 1) in areas surrounding the city centre, including industrial and residential areas. These areas correspond to higher urban densities, compactness and LST. Heat exposure values decreased towards periurban areas.

### *4.3.2. Spatial patterns of urban temperature regulation (TR-ES) supply and demand*

Figure 3 shows the spatial distribution of (a) TR-ES supply and (b) TR-ES demand in Vitoria-Gasteiz at the census tract level. Higher TR-ES supply values (0.6 to 1) were predominantly found within and around large urban parks, forests, wetlands and foothills in the green belt, which constituted 12.04% of the census tracts. The green belt is characterized by land uses such as forests, wetlands, and foothills extending toward the periurban border. Lower TR-ES supply values (ranging from 0 to 0.2) were primarily focused in the urban core and industrial zones. These zones (Fig. 1), often featured industrial polygons and older developments with sparse vegetation cover and higher impervious surface densities.

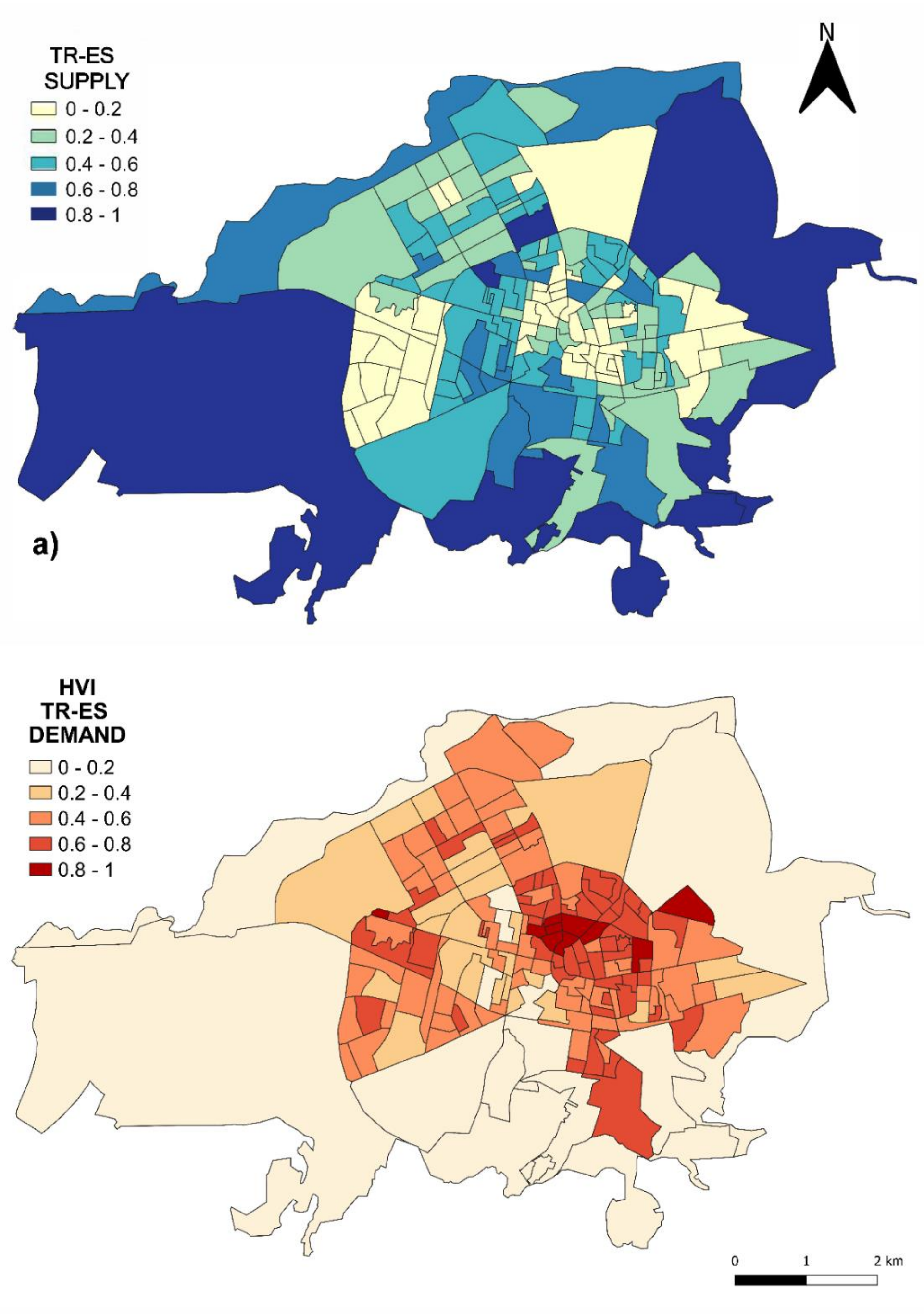


**Figure 3**. Spatial distribution of urban temperature regulation (TR-ES) supply and vulnerability to heat index (as demand) indicators at the census tract level. Interpretation values refer to Very low (0 – 0.2), Low (0.2 – 0.4), Moderate (0.4 – 0.6), High (0.6 – 0.8) and Very high (0.8 – 1).

Regarding ES demand, 65 out of 166 census tracts (39.16%), displayed either high (55 census tracts) or very high (10 census tracts) levels of vulnerability to heat. The old town of Vitoria-Gasteiz (see Fig. 1), central areas and the recently developed periurban areas, exhibited relatively higher heat vulnerability. Consequently, residents living in these areas face increased heat exposure (Figure S5, supp. Mat), which result in a greater demand for TR-ES.

#### *4.3.3. Spatial patterns of TR-ES mismatches*

The average general TR-ES supply–demand mismatch index was 0.21, with positive values (*i.e*. supply < demand) prevalent in 75% of the census tracts (Figure 4a). A higher mismatch (55% of census tracts), were observed in the city centre, industrial zones, and recent urban developments. Conversely, areas with the low mismatch values were primarily found in residential census tracts located around the green belt and to the Southern part of the city.

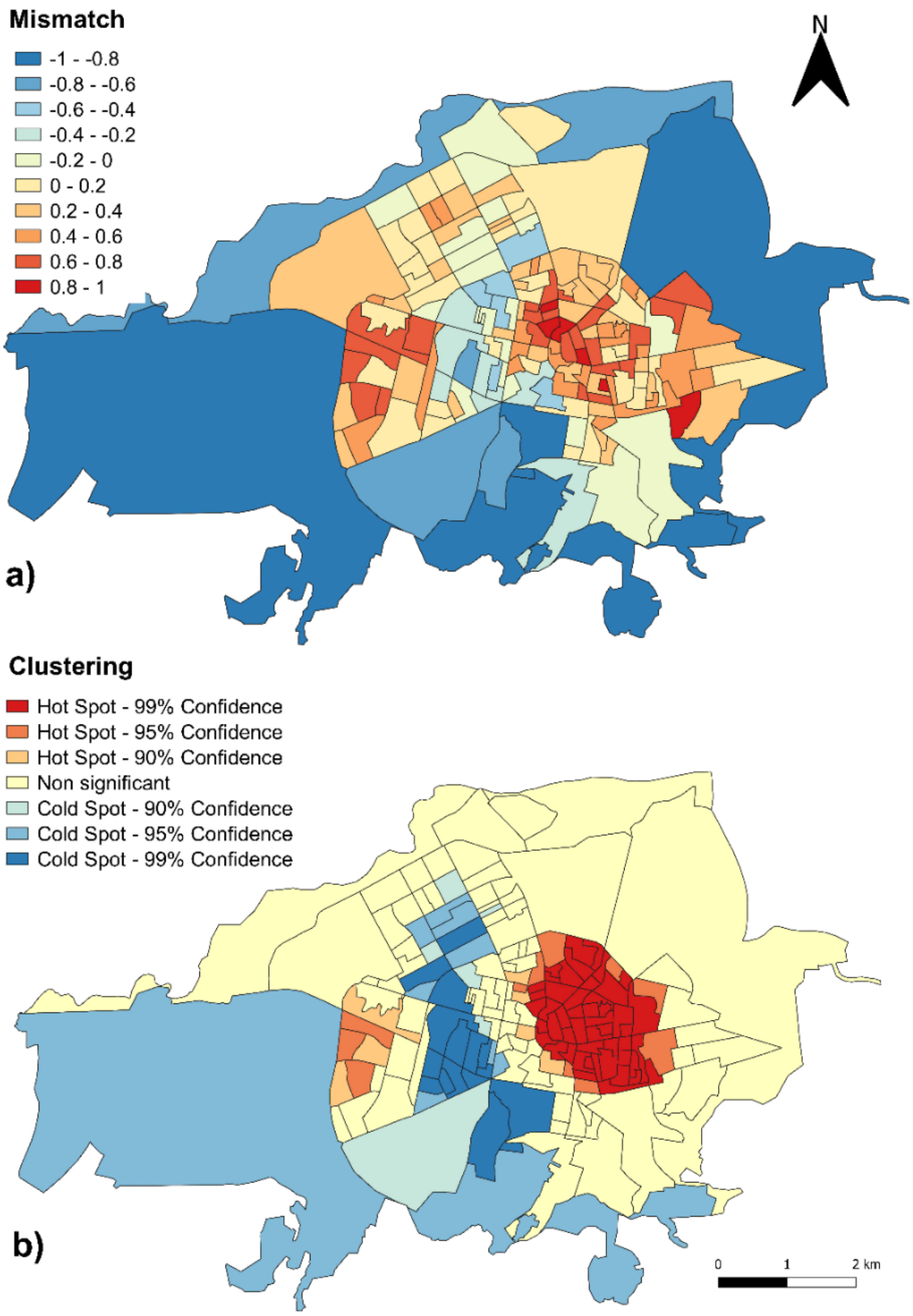


**Figure 4**. Spatial distribution of a) Accumulated supply-demand mismatch values at the census tract level, ranging from -1 (supply > demand) to 1 (supply < demand), with values closer to 0 indicating balance (supply = demand). b) Spatial clustering analysis (Getis Ord Gi*) of supply-demand mismatch values at the census tract level. Clusters are categorized based on z-scores and p-values (p > 0.05) from the cluster analysis. Coldspots (supply > demand) are represented in shades of blue, while hotspots (supply < demand) are depicted in shades of red and orange at 90%, 95%, and 99% confidence intervals. Census tracts in yellow indicate no significant spatial clustering.

Hotspot analysis of TR-ES mismatches values (Fig. 4b), showed clustering patterns based on their z-scores. The top 10% of z-scores (hot spots) accounted for 39.76% of the census tracts suggest unsatisfied demand for TR-ES in these areas, as also shown in the HVI (Fig. 3). Hot spots were associated with a higher average exposure (0.74) and a lower (than the median) household income (avg.: €32,663) compared to cold spots (Table S4, Supp. Mat). They also had a higher proportion of individuals with low educational attainment (40.68%) and born in the American continent (9.22%). Conversely, the lowest 10% confidence interval supply–demand balance z-score values (cold spots) covered 18.07% of the census tracts, suggesting satisfied demand for TR-ES. Cold spots showed considerably lower median heat exposure (0.54) and higher (than median) household income (avg.: €44,918). Furthermore, 42.17% of census tracts displayed no significant spatial clustering.

TR-ES mismatches were associated in 76.3% of the variance, negatively with household income and positively with SMR of lung cancer in men, population below the age of 14, population born in America, and exposure to heat (Table 2). Household income had a low but significant negative effect on the TR-ES mismatch, while the other health and demographic variables showed positive and statistically significant effects, although relatively small, except for heat exposure, which had a medium effect (Fig S7, Supp. Mat.).

**Table 2**. GLM model outputs for the effects of heat exposure, population below the age of 14, SMR of lung cancer in men, population born in America, and household income on model-estimated ES supply-demand mismatch values in Vitoria-Gasteiz.

| **Mismatch = Income + M_lung_cancer + ≤14 age + American + Heat exposure** | | | | | |
|---|---|---|---|---|---|
| | *Estimates* | *std. Error* | *CI* | *F* | *p* |
| (Intercept) | 0.21 | 0.01 | 0.18 – 0.24 | 14.52 | < 2e-16 |
| Income | -0.12 | 0.02 | -0.15 – -0.08 | -6.53 | 8.21e-10 |
| M_lung_cancer | 0.08 | 0.02 | 0.05 – 0.11 | 4.88 | 2.60e-06 |
| ≤ 14 age | 0.13 | 0.02 | 0.10 – 0.17 | 7.90 | 4.31e-13 |
| American | 0.08 | 0.02 | 0.04 – 0.11 | 4.01 | 9.27e-05 |
| Heat exposure | 0.21 | 0.02 | 0.17 – 0.24 | 12.20 | < 2e-16 |
| Observations | 166 | | | | |
| $R^2$ adjusted | 0.763 | | | | |

The PCA accounted for 53% of the total variance on its first two components and showed a differentiated association of cold spot from hot spot clusters regarding the socio-environmental and health variables (Fig. 5). Cold spot clusters were positively associated with income, high education attainment, and female SMR for respiratory diseases (these variables were negatively associated with hot clusters). Contrastingly, hot clusters were positively associated with heat exposure, elderly population, foreign origin, low education attainment, and male and female SMR for diabetes and lung cancer.

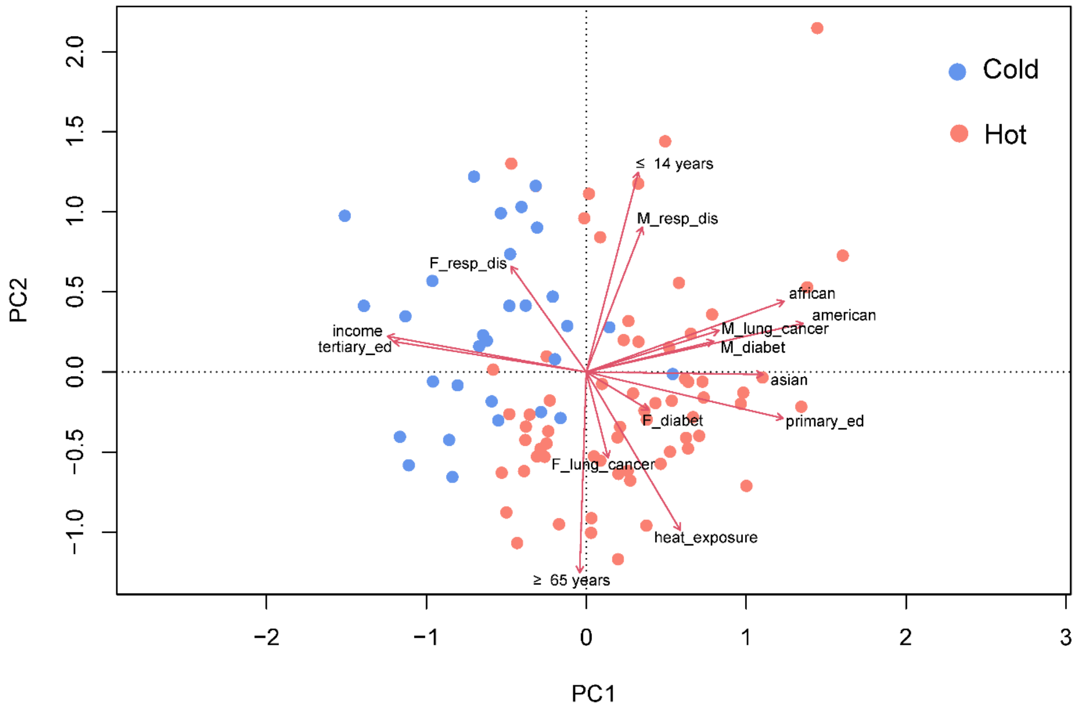


**Figure 5**. Principal Component Analysis (PCA) ordination biplot for hot (red dots) and cold spots (blue dots) clustering units and socioeconomic and health variables used as indicators of heat vulnerability. Abbreviations: Tertiary education (tertiary_ed), respiratory disease in women (F_resp_dis), children (≤ 14 years), elderly (≥ 65 years), diabetes in women (F_diabet), respiratory disease in men (M_resp_dis), diabetes in men (M_diabet), primary education (primary_ed), lung cancer in men (M_lung_cancer), lung cancer in women (F_lung_cancer). The first two axes components (axes) for 53% of the variance (Axis 1: 32.5%, Axis 2: 20%).

The SIMPER analysis identified that the main variables associated with the difference between hot and cold spots were higher heat exposure, lower education attainment, SMR of respiratory diseases in men, SMR of lung cancer in women, and residents of American origin in hot spot clusters. In contrast, cold spots were mainly characterized by higher income, high education attainment, and SMR of respiratory diseases in women (Table 3).

**Table 3.** SIMPER analysis results for variables that contributed to the dissimilarity between a) HOT spots and b) COLD spots clustering patterns. Bold variables are significant and organised by relative contribution.

| Variable | Dissimilarity (%) | SD | Contribution (%) | p | Ratio | Abundance |
|---|---|---|---|---|---|---|
| **Heat exposure** | 2.58 | 2.36 | 14.50 | **0.001** | 1.089 | HOT > COLD |
| M_lung_cancer | 1.96 | 1.49 | 25.50 | 0.189 | 1.315 | HOT > COLD |
| **Income** | 1.54 | 1.49 | 34.10 | **0.001** | 1.033 | HOT < COLD |
| **Tertiary_ed** | 1.46 | 1.02 | 42.30 | **0.001** | 1.436 | HOT < COLD |
| **F_resp_dis** | 1.45 | 1.18 | 50.40 | **0.013** | 1.221 | HOT < COLD |
| **Primary_ed** | 1.43 | 0.92 | 58.50 | **0.001** | 1.553 | HOT > COLD |
| ≥ 65 years | 1.42 | 1.00 | 66.50 | 0.127 | 1.419 | HOT > COLD |
| **M_resp_dis** | 1.30 | 0.97 | 73.80 | **0.010** | 1.340 | HOT > COLD |
| **F_lung_cancer** | 1.24 | 0.93 | 80.70 | **0.010** | 1.325 | HOT > COLD |
| F_diabet | 0.87 | 0.68 | 85.60 | 0.067 | 1.283 | HOT > COLD |
| M_diabet | 0.82 | 0.66 | 90.20 | 0.246 | 1.235 | HOT > COLD |
| **American** | 0.56 | 0.44 | 93.40 | **0.005** | 1.275 | HOT > COLD |
| ≤ 14 years | 0.55 | 0.56 | 96.50 | 0.557 | 0.9741 | HOT > COLD |
| African | 0.50 | 0.54 | 99.30 | 0.375 | 0.9324 | HOT > COLD |
| Asian | 0.13 | 0.12 | 100 | 0.179 | 1.0481 | HOT < COLD |

## 4.4. Discussion

### *4.4.1. Novel contributions in the assessment of TR-ES supply and demand*

Central areas, like the medieval quarter, and residential census tracts near industrial sites (Fig. 3), exhibit high values of heat vulnerability marked by increased heat hazard (Fig S4, Supp. Mat.) and low values of TR-ES supply, suggesting unsatisfied demand for this ES. These areas are characterized by heat-absorbing materials, compact structures, high population density and sparse vegetation. These characteristics exacerbate local temperatures, leading to UHI effects and TR-ES mismatches. These mismatches also result from the city's exposure to heat and the limited capacity to increase TR-ES supply, thereby increasing heat-related vulnerability among residents. Our findings align with previous research in Vitoria-Gasteiz that has identified central and industrial areas as the most unfavourable, characterized by elevated temperatures and insufficient UGI, making them particularly prone to urban heat impacts (Olazabal *et al*., 2012; Rodríguez-Gómez *et al*., 2022; Errea *et al*., 2023).

Our study introduces a novel approach by integrating heat vulnerability indicators into TR-ES demand assessments (Fig. 2), highlighting the multifaceted nature of ES demand often overlooked when assessments rely solely on population distribution (Baker *et al*., 2021; Wolff *et al*., 2015). The HVI index (Fig. 3 and Fig. S6, Supp. Mat.) highlights the importance of identifying heat-vulnerable communities, emphasizing the need for tailored interventions that integrate both public health and greening initiatives (Langemeyer *et al*., 2020; Nayak *et al*., 2018; Reid *et al*., 2009). This approach is a valuable tool for priority-setting of UGI projects and promoting equity within urban management plans.

We have also mapped urban TR-ES supply, based on UGI quantity, quality and its ecological functions (Park *et al*., 2017; Yang & Wang, 2017). By using ecological indicators like habitat quality, canopy cover and NDVI, we have expanded upon traditional assessments of UGI capacity to provide cooling effects solely based on vegetation indexes while providing comprehensive, easily adaptable and replicable estimators (Aznarez *et al*., 2022; Wang *et al*., 2023). Factors like ecological integrity and connectivity influence the supply of temperature regulation by urban ecosystems (Luo & Fu, 2023). Recent studies emphasize that vegetation cover, species composition, and diversity are key drivers of microclimate temperature regulation (Gillerot *et al*., 2022; Zhang *et al*., 2022). Moreover, research considering habitat fragmentation and edge effects suggest that vegetation patches exposed to external microclimatic conditions can limit forests capacity to supply TR-ES (Ewers & Banks-Leite, 2013).

We have found that areas with extensive and less fragmented green spaces, such as the green belt and large urban parks (Figure 3), are associated with higher levels of TR-ES supply. In contrast, industrial and central areas, which have limited TR-ES supply exhibit higher ecological fragmentation (Aznarez *et al*., 2022). This emphasizes the relevance of considering land-use patterns, particularly fragmentation, when examining TR-ES dynamics in urban landscapes. Understanding how compact and extensive urbanization models influence ES offer valuable insights for optimizing urban land management for effective TR-ES provision (Stott *et al*., 2015). Our study highlights the role of ecological integrity and fragmentation in shaping temperature regulation ES in urban areas.

#### *4.4.2. Implications of TR-ES mismatches for urban environmental justice*

Our analysis of TR-ES supply-demand mismatches (Fig. 4) has revealed significant urban climate inequalities shaped by socio-demographic factors, including health aspects, reflecting various vulnerability dimensions. TR-ES mismatches are primarily associated with variables related to health condition, age, foreign origin, and increased heat exposure, leading to higher unsatisfied demand for TR-ES in areas with these characteristics (Table 2, Fig S7, Supp. Mat.). This finding aligns with previous research indicating that vulnerable groups, such as the elderly, children, migrants, people with chronic diseases, and those with limited exposure to UGI face elevated health risks from extreme heat (*e.g*. Amorim-Maia *et al*., 2023; Keeler *et al*., 2019; Nayak *et al*., 2018; Bradford *et al*., 2015; Reid *et al*., 2009). Notably, high-income neighbourhoods in Vitoria-Gasteiz, characterized by higher vegetation cover and lower population density, exhibit lower TR-ES mismatch values.

Spatially clustering of TR-ES mismatch values into hot spot and cold spot areas can (Figs. 4b and 5) provide valuable information for priority-setting and designing of UGI interventions addressing the unique needs of these areas. Hot spots in Vitoria-Gasteiz, associated with socioeconomically disadvantaged areas with higher heat vulnerability, contrast with cold spots primarily found in more affluent areas (Table 3; Table S4. Supp. Mat.). This finding aligns with prior research highlighting the uneven distribution of canopy cover in cities, which is influenced by socioeconomic factors, ultimately contributing to the stratification of environmental stressors, such as the UHI effect (Chakraborty *et al*., 2019; Jenerette *et al*., 2011). These disparities hinder equitable access to regulating ES, as shown by luxury effects related to canopy cover (Schell *et al*., 2020; Jenerette *et al*., 2013). Higher-income areas benefit from reduced urban heat vulnerability and better TR-ES supply, while lower-income areas being more exposed to heat and rely more on public resources to cope with it (Jenerette *et al*., 2011).

Our findings also highlight the heat risks for residents in hot spots, especially in areas with high LST (*i.e*. heat hazard), limited vegetation, and high population density (Figures S4 and S5, Supp. Mat.). We thus emphasize the need to explicitly integrate environmental and climate justice principles into UGI development plans to address distributional environmental injustices and mitigate heat-related risks by those who are more vulnerable. The uneven distribution of UGI contributing to TR-ES mismatches in Vitoria-Gasteiz also aligns with prior research on environmental injustices and increased climate vulnerability (Herreros-Cantis & McPhearson, 2021). This finding complements prior research in Vitoria-Gasteiz that suggests that management and land use legacies are key factors in shaping the distribution of UGI-based regulating ES, as is the case of older areas with well-established canopy (Aznarez *et al*., 2023). However, the socioeconomic status also influences UGI accessibility, quality, and quantity (Schell *et al*., 2020). Our results further indicate that the majority of residents Vitoria-Gasteiz (75% of census tracts), lack access to UGI that can supply TR-ES, in contrast to those living in higher-income areas. This observation holds significance, especially considering the extensive city-wide green space coverage, where 31% census tracts in the city have over 30% canopy cover (Fig. S2, Supp. Mat.).

#### *4.4.3. Considerations for urban planning and policy*

Addressing the challenges posed by the rising temperatures in urban landscapes requires strategic interventions. Areas with the highest heat exposure, such as the central and nearby industrial areas in Vitoria-Gasteiz (Fig S5, Supp. Mat.), offer opportunities for long-term interventions. These may involve extensive tree planting, enhancing canopy cover, and allowing vegetation to mature (Aznarez *et al.*, 2023). In locations where greening measures may not be feasible, the use of heat-reflective materials could be considered (Mallen *et al.*, 2019).

However, areas characterized by high vulnerability to heat, particularly those with inequalities in access to TR-ES (Figure 4), require establishing adaptation strategies that address multifaceted and intersecting factors associated with increased demand for TR-ES (Table 2; Fig. 5). Potential approaches would include greening initiatives designed to enhance TR-ES supply and the establishment of climate shelters. These shelters can be outdoor settings integrated within UGI, providing shade and canopy cover, or indoor facilities offering refuge and assistance during extreme heat events. Both approaches aim to minimize outdoor exposure through favourable environmental conditions and access to drinking water, particularly for vulnerable groups like the elderly (Amorim-Maia *et al.*, 2023; Keeler *et al.*, 2019; Bradford *et al.*, 2015). Effective implementation of these measures relies on understanding the spatial variability of heat-related vulnerability, as highlighted by our TR-ES mismatch mapping approach (Fig. 4). Therefore, enhancing adaptive capacities within urban landscapes goes beyond increasing overall green cover; it involves targeted efforts that address specific population needs, community-based resilience actions, awareness-raising, and measures beyond green spaces expansion.

Increasing urban vegetation mitigates UHI effects and reduces local temperatures (Marando *et al.*, 2022, 2019). However, compact cities like Vitoria-Gasteiz must balance effective urban development with UGI preservation and expansion. Urban densification, while resource-efficient, may hinder TR-ES provision (Mascarenhas *et al.*, 2019;

Sebastiani *et al*., 2021; Aznarez *et al*., 2023). Given the limited space for expanding greenery, an integrated approach that combines grey-green infrastructure is key (Langemeyer *et al*., 2020). While recognizing that UGI benefits are not linearly correlated with vegetation quantity (Whitlow *et al*., 2014), enhancing canopy coverage can help achieve a better TR-ES supply-demand balance. Optimizing UGI spatial distribution through green roofs, green walls, pocket parks, and larger green spaces where feasible, all interconnected for efficient resource use, is seen as crucial. Diversifying vegetation structure within existing UGI, beyond traditional lawns, can also offer a cost-effective approach for addressing urban heat hazards, especially in areas where tree planting is not feasible due to physical constraints or management considerations (Francoeur *et al*., 2021). This approach can help balance TR-ES supply and demand while also providing habitat for biodiversity, with synergies with other urban ES like pollination and storm water mitigation (Francoeur *et al*., 2021; Kremer *et al*., 2016).

UGI should be biodiverse and multifunctional, considering tree species that are resilient to climate change (Lanza & Stone, 2016). While deciduous tree species are effective for urban heat mitigation due to their shading and evapotranspiration (Sebastiani *et al*., 2021), a sole focus on increasing TR-ES supply may pose risks, including increased vulnerability to climate change-related impacts like pests and diseases. Evaluating tree-planting schemes for their impact on urban biodiversity and ecological functions and services is crucial, emphasizing the importance of maintaining a diverse plant species range for supporting biodiversity. Successful implementation of these strategies, especially in socially vulnerable areas, requires collaborative efforts, legislative support, and community involvement. Recognizing that UGI design and management are also influenced by cultural and aesthetic aspects, these factors should be considered in urban planning (Francoeur *et al*., 2021; Grove *et al*., 2014).

#### *4.4.4. Limitations and future research*

Our study has various limitations. The reliance on relative values for TR-ES (supply, demand, and mismatches) (Figs. 3 and 4), may limit applicability and comparability across other cities and locations. The use of standardized indicators for TR-ES supply and demand, lacking a common framework may introduce subjectivity (Sebastiani *et al*., 2021), which is also related to the incommensurability of considered variables. Partially addressing this incommensurability, we used unitless composite indicators which integrates multiple dimensions of different variables (Alam *et al*., 2016).

The decision to assign equal weights in our HVI index simplified the modelling process but potentially overlooked variation in the actual contributions of the individual variables. To address this, GLM and PCA helped discern the individual influences of variables, enhancing the understanding of their roles in TR-ES mismatches. Additionally, our analysis of TR-ES demand does not consider residents' preferences and values, which can significantly affect their subjective vulnerability to heat. For instance, individuals prioritizing air conditioning or engaging less with urban green spaces may experience higher heat risks. Integrating these insights into the analysis is essential for targeted interventions. However, collecting representative data on people's preferences and values can be time-consuming and resource-intensive. Integrating the HVI index in TR-ES demand quantification, may offer a cost-effective way to account for intersecting factors shaping resident's needs, improving current methods.

Additionally, advancing our understanding of urban TR-ES mismatches can benefit from incorporating multi-temporal data, including seasonal variations in LST or NDVI indicators and addressing spatial and temporal variability (Yao *et al*., 2020). Satellite-derived LST is a practical way for mapping ambient temperature indicators (Fig. S4, Supp. Mat.), but it also has limitations. Future research may benefit from integrating air temperature data from urban weather monitoring stations, closely associated with human perceptions of heat, comfort, and health (Wang *et al*., 2023; Chakraborty *et al*., 2022). In our case, remote sensing seems suitable for capturing the spatial heterogeneity of urban climate variability. Our combination of satellite-derived LST with NDVI and population density yielded consistent results (Figs. 3, S4 and S5, Supp. Mat.), aligning with previous research in the study area (Rodríguez-Gómez *et al*., 2022).

It should also be noted that equal weighting of TR-ES demand indicators may overlook variations in their contribution to heat vulnerability, requiring further refinement to consider the distinct impacts of each individual indicator (Wang *et al*., 2022). Data limitations led to the exclusion of factors like access to air conditioning, and living alone, which can exacerbate vulnerability, particularly among socially isolated elderly individuals who may not recognize heat-related symptoms or seek help (Bradford *et al*., 2015). Addressing heat-related vulnerabilities from a distributive justice perspective requires examining the multifaceted barriers faced by the most heat-vulnerable groups, including inadequate housing, limited access to urban cooling, and socioeconomic inequalities (Amorim-Maia *et al*., 2023). We thus emphasize the need for further research to address urban heat risks and the interplay between socioeconomic and biophysical factors in vulnerability assessments, calling for exploration and resolution of methodological and data-related challenges.

## 4.5. Conclusions

Exposure to urban heat poses a pressing challenge in the face of climate crisis, with uneven health and environmental impacts in cities. UGI can play a key role in mitigating these impacts by reducing ambient temperatures. Our study offers valuable insights into the complex relationships among urban heat, the spatial distribution of TR-ES, and heat vulnerability in urban areas. Through a comprehensive assessment of TR-ES mismatches in the Vitoria-Gasteiz case, combining remote sensing, field data, and socio-demographic data we have identified key priorities for UGI planning at the intersection of environmental justice and climate adaptation.

To plan effective urban heat mitigation interventions, it is crucial to understand the social and health determinants contributing to unequal heat risks. Our research contributes a replicable workflow and modelling approach that serves as a monitoring and spatial prioritization tool. We have identified the locations of vulnerable communities affected by heat hazards, enabling evidence-based recommendations tailored to specific needs for mitigating heat-related vulnerabilities. Addressing these challenges is key to ensuring the well-being of urban residents as cities confront growing climate impacts. Integrating vulnerability considerations and ES distribution analyses into practical urban planning and management strategies is crucial. This represents not only a shift towards urban sustainable and green transitions, but more importantly, towards just and green transitions. By optimizing UGI and addressing local community needs, urban adaptation plans can work toward mitigating urban heat while promoting equity and creating sustainable urban environments.

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# Chapter V

# Discussion and conclusions

## 5.1. Synthesis

This dissertation provides empirical evidence on how biodiversity-related key structural and functional attributes of UGI are related to their capacity to provide ES and support human well-being from an environmental justice perspective. This was achieved through three complementary analytical approaches in Vitoria-Gasteiz , a mid-sized European city implementing remarkable greening strategies. In these studies, urban biodiversity was a key biophysical attribute in driving the capacity of UGI to support ES and thus influence their social distribution, benefits, and potential protection from environmental burdens. While biodiversity is usually highlighted by its influence on the ecosystem capacity to provide ES, this role is rarely directly analysed in cities through specific indicators of biodiversity (Marselle *et al*., 2021; Ziter, 2016). This dissertation builds on this by approaching biodiversity as a key driver of spatial heterogeneity in ES capacity in cities, its relation to socioeconomic and historical management of urban greening, and the related effects on the protection from human-driven environmental burdens. Furthermore, this research was developed considering an environmental justice perspective, where urban biodiversity patterns and related ES were analysed regarding spatial and socioeconomic distribution in the city. In this way, it acknowledged how biodiversity patterns are socially co-produced in this city, while providing relevant empirical knowledge and new data to support UGI management strategies that promote an equal social distribution of the benefits produced by urban nature.

## 5.2. Contributions and main findings

This dissertation makes significant contributions at the intersection of three knowledge domains: urban ecology, ecosystem services, and environmental justice, with a specific focus on UGI as the main unit of analysis.

### 5.2.1 Theoretical contributions

*Biodiversity and ecosystem capacity to provide ES*

Biodiversity supports the ecosystem´s capacity to provide ES, and the results suggest that uneven patterns of biodiversity distribution in the case study city drive unequal patterns of access to their related ES and overall human well-being. Biodiversity patterns were explored in terms of species richness for different taxonomic groups and related to other associated indicators, including habitat characteristics. Both biodiversity and related indicators, habitat quality (HQ) and urban wildness (UW), were largely influenced by the UGI typology. Small and highly fragmented spaces, typically in central areas of the city, provide limited support for biodiversity compared with larger and more connected peripheral areas (*e.g.*, urban green belt). These uneven spatial patterns of biodiversity are associated with unequal access to biodiversity-related benefits by the local population. Such a distribution is partly explained by socioeconomic and historical processes influencing urban greening (Chapter III) and results in the uneven distribution of benefits and protection from environmental burdens such as urban heat regulation (Chapter IV). The results suggest potential improvements in UGI structural dimensions and management to better support biodiversity: larger sized areas with higher connectivity, lower perimeter:area ratio, reduced management (*e.g.*, less mowing and pruning), and mitigate the influence of threats (*e.g.* noise, light pollution, mechanised access and urban densification).

The biodiversity-related indicators used in this study, HQ and UW, allowed the integration of functional characteristics of the environment such as naturalness, remoteness, challenging terrain, habitat suitability, and vulnerability to threats (*i.e.*, pressures exerted by human activity) that further describe the biophysical attributes of an ecosystem related to its capacity to provide ES. This complemented purely biodiversity data as species richness and allowed the integration of metrics related to the functioning of ecosystems. This complementary analysis of different biophysical factors addresses an identified bias in ES studies: while most assessments of ES provision consider levels of habitats or land-use type, studies that link an ES to biodiversity usually only focus on one functional or taxonomic group (Ziter, 2016). **Chapter II**, including indicators of habitat such as HQ and UW along with biodiversity, aligns with the identified need for more comprehensive studies that allow us to expand our knowledge of the biological underpinnings of urban ES provision (Ziter, 2016). Therefore, my findings contribute to better clarifying the association between the biophysical attributes related to biodiversity that influence the ecosystem capacity to provide ES, as well as the cascading effects on human well-being (Schell *et al*., 2020; Baró *et al*., 2019; Leong *et al*., 2018; Ziter *et al*., 2016).

The combined use of habitat indicators as HQ, wildness along with biodiversity data also provide a way to conceptualise the city as a habitat source, identifying areas that should be prioritised for improving ecological integrity (Müller *et al*., 2018). My findings underscore the significance of larger, less fragmented green spaces with high naturalness, remoteness, challenging terrain, and habitat suitability (*i.e*. such as the green belt and expansive urban parks) in supporting biodiversity, particularly in terms of HQ and UW. These findings agree with studies that highlight the size and habitat quality of urban green spaces as key local determinants of biodiversity (Aronson *et al*., 2017; Lepczyk *et al*., 2017). These results emphasise the potential of urban ecosystems to support biodiversity, with increased ecological functions and ES in alignment with the multifunctional character of UGI, thus challenging prevailing notions of urban systems as biological deserts (Spotswood *et al*., 2021; Hall *et al*., 2016).

The assessment of the ecological quality of urban green and blue spaces in Vitoria-Gasteiz forms the basis for spatially evaluating the capacity of urban ecosystems to supply ES by assessing how comprehensive UGI biophysical attributes predict the distribution of urban biodiversity. By adapting spatial modelling tools traditionally used in areas with minimal human influence, **Chapter II** addresses the overlooked role of urban ecosystems as habitat sources.

*Socioeconomic and historical legacies in ES*

Socioeconomic and historical management shapes biodiversity and UGI spatial heterogeneity, thus influencing ES distribution in cities (Lambert & Schell, 2023; De Roches *et al.*, 2020; Smith, 2007). Thus, the results of **Chapter III** supports the 'luxury effect' hypothesis, where urban biodiversity is associated with affluent neighbourhoods, which tend to be strongly correlated with higher socioeconomic status and education attainment (Schell *et al*., 2020; Kirkpatrick *et al*., 2011). This positive association was particularly strong in neighbourhoods with higher levels of educational attainment and higher-quality habitats, reinforcing the luxury effect. As HQ improved, this correlation intensified, potentially establishing a feedback loop where biodiversity benefits are accentuated in affluent areas. This pattern is attributed to the shared preferences of humans and other species for environmentally desirable areas while avoiding environmental burdens (Magle *et al*., 2021; Schell *et al*., 2020). This supports the notion that affluent households tend to allocate more towards green spaces and steward habitat and biodiversity (Lambert & Schell, 2023; Grove *et al*., 2014; Clarke *et al*., 2013).

However, while our results support the the existence of a luxury effect regarding biodiversity indicators, we found no evidence supporting the luxury effect when solely considering vegetation cover (in both public and private land), contrary to what was expected based on previous studies (Jenerette *et al*., 2013; Kendal *et al*., 2012; Hope *et al*., 2003). This could be attributed to other factors not included here, such as management choices involving the selection of native or non-native plant species, or preferences of some species over others (Kendal *et al*., 2012).

In addition to these contemporary socioeconomic factors, **Chapter III** further explores the effect of historical legacies in shaping the spatial heterogeneity of biodiversity and UGI. By exploring how past investment and UGI management practises shape the current distribution of associated benefits, this study underscores the enduring effects of past land use planning and development decisions on UGI distribution and biodiversity. The time of development was a key factor in explaining the distribution and biodiversity of the UGI. In the case study of Vitoria-Gasteiz, management historically prioritised conserving older neighbourhoods and the mediaeval architecture of the old town, typically with low green cover and biodiversity. The most biodiverse neighbourhoods were the ones more recently developed (1970 – 2000) with greening efforts becoming a focus since the 1990s as a strategy to reverse the damage of industrial activities and urban expansion. This supports the idea that legacy effects must incorporate land-use management history to better understand how past management actions have shaped current landscape characteristics (Roman *et al*., 2018). However, in addition to green cover, enhanced biodiversity in urban ecosystems requires sufficient time to allow ecological successions and colonisation, particularly for slow-growing species, which makes time of development a key factor in urban biodiversity dynamics (Clarke *et al*., 2013).

Regarding the provision of regulating ES, I found no clear evidence for the luxury or legacy effect when solely considering socioeconomical and historical indicators of luxury or legacy effect, since higher educational attainment and neighbourhood development age showed no direct association with regulating ES provision. Likewise, there was no evidence that increased urban biodiversity leads to a larger flow of these regulating ES. This contradicts other research that highlights how the luxury effect is closely related to the benefits that people obtain from nature (Schell *et al*., 2020; Leong *et al*., 2018). Nonetheless, the provision of regulating ES was higher in areas with more vegetation cover, particularly in older neighbourhoods. This indicates a more complex provision of regulating ES associated with green cover and land use legacies, which partially supports the existence of a legacy effect. The results further support that older neighbourhoods with higher canopy cover yield more regulating ES, even if they have less biodiversity (Baró *et al*., 2019). This suggests that urban tree planting schemes in the past may have focussed on providing shading or aesthetic purposes rather than enhancing plant diversity. As such, older neighbourhoods with larger, more mature trees provide higher regulating ES than newer neighbourhoods with a higher proportion of young trees and herbaceous cover (Roman *et al*., 2018). This pattern may be influenced by age, suggesting that the older and greener the neighbourhood, the stronger the relationship with regulating ES provision. These findings suggest that UGI identity is markedly different between older and newer neighbourhoods, reflecting relevant legacy effects patterns.

Our findings contribute to the ES cascade model (Potschin-Young *et al*., 2018), by addressing the supply of ES and the underlying drivers (*i.e*. luxury and legacy effects) that constitute pressures that influence the ecosystem's capacity to provide ES. Consequently, these areas enhance the UGI's ability to deliver essential ecosystem functions and processes that support the provision of ES.

Notably, this study represents the first explicit test of the luxury and legacy effect hypotheses in relation to ES by examining how affluence and historical legacies influence the provision and distribution of ES in urban areas. By integrating the traditionally ecologically approached luxury and legacy effects into the inherently anthropocentric ES framework, this dissertation contributes to theoretically bridging ecological and anthropocentric perspectives for urban ES research. Likewise, it contributes to the identification of underlying factors that co-produce ecosystem capacity to support ES as well as to contextualise these processes and related cascading effects within the framework of urban ecology and ES (Alberti, 2024; Lambert & Schell, 2023; De Roches *et al.*, 2020; Roman *et al*., 2018).

*Environmental justice for ES assessments*

This dissertation analysed the mismatches in supply and demand of ES, with a specific focus on temperature regulation provided by UGI, and identified the factors driving distributional green inequities. The results suggest that heat vulnerability is unevenly distributed among urban residents, reflecting different needs for temperature regulation ES. The city centre, with higher urban density, and industrial areas with heat-absorbing materials and sparse vegetation cover were the areas with higher temperatures and lower temperature regulation. These areas were more vulnerable to heat as a consequence of higher heat exposure. However, large urban parks, forests, wetlands, and foothills in the green belt were the ones that mostly supported the ES of temperature regulation. The results indicate a predominantly higher demand compared with the supply of ES, with a prevalence in demand in 75% of the considered areas. This mismatch was higher in the city centre industrial zones, and recent urban developments and lower in residential areas around large green areas. Hotspots of mismatches in the supply-demand of ES were positively associated with health-related and demographic variables: lower income, children (<14), elderly (>65), low educational attainment, higher proportion of foreign born population, and higher mortality ratio due to lung cancer and diabetes. Cold spots showed lower median heat exposure and were associated with higher household income, higher education attainment, and higher mortality due to respiratory diseases among women.

The results highlight disparities in heat vulnerability, with increased exposure observed among socioeconomically disadvantaged communities, foreign born populations, the elderly, and those with health issues. Conversely, affluent areas with higher income levels exhibited lower ES mismatches, indicating better ES supply and reduced heat vulnerability. These results emphasise the association between the provision of ES and underlying sociodemographic and health factors in driving distributional injustices in exposure to environmental burdens. These findings align with previous research highlighting the socioeconomic stratification of environmental stressors (Chakraborty *et al.*, 2019; Jenerette *et al.*, 2011), and the role of canopy cover in cities, as discussed in **Chapter III**.

An environmental justice perspective for ES assessments requires fair and equitable social distribution of the benefits obtained from ecosystems and protection from negative environmental impacts (Schell *et al.*, 2020). In this sense, this dissertation connects the ecosystem capacity to sustain ES with societal demands for these ES by analysing the mismatches in temperature regulation supply–demand in an urban environment. Thus, it aligns with what has been underscored by recent research: the need for integrating an environmental justice perspective to promote more equitable and just outcomes resulting from UGI interventions and greening policies (Calderón-Argelich *et al.*, 2021; Langemeyer & Connolly, 2020; Angelovski *et al.*, 2020; Meerow, 2020). By connecting direct indicators on the capacity to support ES and on the demand for ES, this approach contributes to the existing gap between the conceptual and practical aspects of environmental justice and ES research acknowledged by Calderón-Argelich *et al.* (2021) and Langemeyer & Connolly (2020).

The equity perspective of this dissertation examines "to whom?" UGI mostly benefits in terms of ES provision and adaptation to environmental impacts by focussing on the specific ES of protection from urban heat hazard. This approach adds to the research in environmental justice by underscoring that environmental injustices result from the interplay of various factors that need to be recognised (Lambert & Schell, 2023). Thus, it emphasises the need to integrate equity principles into urban planning by understanding the spatial distribution of UGI and ES within the urban landscape, along with environmental injustices as emerging properties of such patterns. Moreover, it integrates the diverse needs of different sociodemographic groups in relation to heat hazards.

The integration of the heat vulnerability index enhances the comprehensiveness of ES demand assessments, improving our understanding of the scale and distribution of ES needs and deficits in urban areas. This approach is crucial when relying on proxies like population density, which may oversimplify the diverse needs of communities (Baker *et al*., 2021; Wolff *et al*., 2015). By adopting the heat vulnerability assessment framework from the IPCC (2007), encompassing exposure, sensitivity, and adaptive capacity, I quantified heat vulnerability as a proxy for estimating ES demand, thus contributing to better connect demand according to different ES needs (Langemeyer *et al*., 2020; Mallen *et al*., 2019; Bradford *et al*., 2015). Unlike traditional ES assessments, which rely on indirect inferences of demand by considering the magnitude of pressures and assessing the population exposed to these pressures (Baker *et al*., 2021; Wolff *et al*., 2015), this research recognises and integrates the multifaceted nature of demand. For instance, some populations such as the elderly, individuals with pre-existing health conditions, or lower-income communities may be more vulnerable to heat-related hazards (Keeler *et al*., 2019; Nayak *et al*, 2018; Reid *et al*., 2009). The integration of these factors into the heat vulnerability index adds nuance to the assessment, aligning with the ES cascade theory by recognising and incorporating the diverse and specific demands for ES across demographic groups. Furthermore, it is key to identify the factors that influence the demand for specific ES provided by UGI and to acknowledge the potential tensions arising from contrasting expectations regarding these demands (Andersson *et al.*, 2014).

### 5.2.2. Methodological contributions

*Adaptation and generation of new models for urban environments*

**Chapter II** introduces a novel modelling approach challenging the conventional view of urban land uses as threats to 'natural' landscapes. This approach adopts spatial models originally designed for non-urban settings using versatile, adaptable, and replicable indicators that require low data input. Complementing previous work using InVEST HQ models and wildness assessments in urban landscapes (Hamel *et al*., 2021; Müller *et al*., 2018), the integration of UW and HQ indicators enhances our understanding of their relationship with biodiversity and the biophysical structure of UGI. Furthermore, both models partially address the limitations associated with widely used NDVI/NDWI indicators in urban environmental studies to quantify exposure to green and blue spaces, but are less adequate for measuring their quality. Such indicators are sensitive to seasonal variability and do not differentiate between green space typologies, as discussed in **Chapter II**, thus limiting their applicability in the context of urban biodiversity conservation (Trethewey & Reynolds, 2021). This, in turn, contributes to the prediction of biodiversity patterns and provides novel tools for urban biodiversity assessments that can be globally scaled to mid-sized cities.

While UW and HQ indicators share similar capabilities in capturing biodiversity patterns, the choice between these indicators depends on data availability and the specific focus of the study. UW considers the structural descriptors of land use and terrain (*i.e.*, perceived naturalness, remoteness, and challenging terrain) (Müller *et al*., 2018). In contrast, HQ includes assessing the expert perception of the habitat suitability of different land uses and their susceptibility to different anthropogenic impacts, providing both structural and functional descriptors of habitat suitability and vulnerability to threats (Sharp *et al*., 2020; Di Febbraro *et al*., 2018; Sallustio *et al*., 2017). For understanding landscape structure, UW proves effective as a predictor, even in the absence of functional data (Müller *et al*., 2015). HQ complements UW by integrating information about anthropogenic impacts, offering a more comprehensive perspective on how these factors collectively influence biodiversity and landscape functionality. HQ is more robust in considering potential impacts affecting landscape functionality, incorporating information on threats (*i.e*., roads, night light, population, density, railways, and noise), and their relative impact on the capacity of different land use classes to harbour biodiversity. Differences between UW and HQ yield more precise information for managers, identifying areas characterised by high UW that may not necessarily support high biodiversity due to the influence of threats.

Furthermore, the results facilitate the identification of key areas for urban biodiversity conservation, unsuitable places for new urban development, or pinpoint locations where efforts should concentrate on reducing the impacts of threats. Despite the time and resource-intensive nature of acquiring extensive information on urban biodiversity status (Jalkanen *et al*., 2020), the relatively low input requirements of our spatial models make them particularly suitable for places where biodiversity information is lacking or monitoring efforts are scarce. The combination with expert consultation, a widely used method to obtain information when data are limited (Terrado *et al*., 2016; Kuhnert *et al*., 2010), makes this approach cost-effective for evaluating potential urban biodiversity support while incorporating habitat suitability and sensitivity to threats to different habitats.

Despite being applied in a context with high data availability, this approach is cost-effective and holds promise for data-scarce regions with limited resources for monitoring and running complex models (Hamel *et al*., 2021). Particularly, cities in the Global South are expected to undergo a rapid urbanisation process. In summary, these easily adaptable and replicable proxies can be integrated as spatial prioritisation tools, informing urban planning at a fine scale and accounting for different UGI typologies along with the effects of anthropogenic impacts on their quality and multifunctionality.

*Integration analysis of biodiversity and socioeconomic drivers*

**Chapter III** contributes to a nuanced understanding of interactions and mutual reinforcement in urban biodiversity and vegetation patterns, a dimension often overlooked in studies addressing similar questions (Schell *et al*., 2020; Leong *et al*., 2018). Beyond individual factors, this study integrates other landscape descriptors, notably HQ, vegetation cover, and population density, as influential factors shaping urban biodiversity. Notably, it introduces novelty by considering interactions and cumulative effects, shedding light on how these factors contribute to the magnitude of both luxury and legacy effects. The luxury effect on biodiversity was found to be stronger in areas of high HQ, whereas the inverse legacy effect was more pronounced with increased population density. Notably, population density exacerbates the detrimental effect of neighbourhood age on biodiversity and vegetation cover. The correlation between population density and neighbourhood age suggests that older neighbourhoods tend to be more densely populated, leading to adverse effects on biodiversity and vegetation cover. Integrating these variables is critical for a more accurate assessment of luxury or legacy effects, contributing to capture the contribution of such factors to the analysed phenomena. Furthermore, this study focuses on the temporal influence of population density on vegetation trends, which affects available canopy space and restricts the growth of new tree individuals over time (Locke *et al*., 2016; Clarke *et al*., 2013). Thus, the research not only establishes that population density directly impacts biodiversity negatively but also amplifies the adverse effects of neighbourhood age on biodiversity and vegetation cover.

*Spatial prioritisation for environmental justice in greening initiatives*

Grounded in FAIR principles, **Chapter IV** presents an analytical framework that is adaptable to any city, leveraging insights from a data-rich case study. It extends to the development of new models within the broader Artificial Intelligence for Environment & Sustainability (ARIES) framework. This integrated modelling allows to assess the spatial distribution of ES supply and demand through composite indicators, combining ecological, biophysical, and societal variables into a unitless normalised value relatable to ecological properties and functions (Fusaro *et al*., 2023; Sebastiani *et al*., 2021; Park *et al*., 2017; Yang & Wang, 2017; Alam *et al*., 2016).

Scale is critical when analysing different spatial explicit models, and while in **Chapter III** the analysis of luxury and legacy effect on ES was performed on a neighbourhood scale, the analysis of ES supply-demand in **Chapter IV** was performed at the finest available scale, the census tract level. This adjustment allowed us to analyse the sociodemographic and health-related variables available at the census tract level, aligning with the idea of smaller scale requirements for urban planning and UGI research (Yok-Tan & Samsudin, 2017). **Chapter IV** proposes an alternative to complex ES modelling tools such as iTree Eco, addressing challenges related to data requirements and technical expertise for resource intensive implementation (Xie *et al*., 2023; Nowak *et al*., 2018). Indicators used for composite mapping in **Chapter IV** (*i.e*. NDVI, percentage of canopy cover and HQ), require fewer resources, making them easily adaptable to various contexts, particularly in areas with limited data inputs (Alam *et al*., 2016).

Notably, the integration of the HQ as an indicator of ecological integrity for temperature regulation supply is a conceptual and methodological contribution. It expands upon traditional assessments of UGI capacity to provide temperature regulation ES, complementing more commonly used proxies such as canopy cover and NDVI. This aligns with recent research focussing on factors such as ecological integrity and connectivity that influence temperature regulation by urban ecosystems (Luo & Fu, 2023). Other studies underscore the critical role of vegetation structure, patch size, species composition, and diversity as primary drivers of microclimate temperature regulation (Kraemer & Kabisch, 2022; Gillerot *et al*., 2022; Zhang *et al*., 2022). In this context, a higher HQ indicates relatively higher productivity and ecological integrity, whereas a lower HQ indicates higher exposure to anthropogenic threats, particularly to urban densification and mechanised access, contributing to habitat fragmentation and the influence of threats and edge effects, reducing the evapotranspiration and hence cooling potential of vegetation (Ewers & Banks-Leite, 2013).

This chapter introduces a novel approach by integrating heat vulnerability indicators into temperature regulation demand assessments, highlighting the multifaceted nature of ES demand, which is often overlooked when assessments rely solely on population distribution (Baker *et al*., 2021; Wolff *et al*., 2015). Through spatially explicit assessments of the ES supply-demand mismatches, it contributes evidence-based information to support urban planning by addressing environmental inequities and identifying priority intervention areas for reducing heat-related vulnerabilities. This aligns with the existing literature that emphasises the importance of such assessments in guiding effective urban interventions (Baró *et al*., 2015; Herreros-Cantis & McPhearson, 2021).

*Contributions to Open Science*

All contributions in this dissertation have addressed open science principles. The two articles already published as part of this thesis, corresponding to **Chapters II** and **III**, are available in open-access journals. The modelling and data used in these articles come from public repositories and institutions or were generated using open source or freely available software. The forthcoming publication of the model developed in ARIES

(Artificial Intelligence for Environment and Sustainability) for assessing ES mismatches and urban heat vulnerability assessment, grounded in FAIR (Findable, Accessible, Interoperable, Reusable) principles, will be made publicly available. This openness enables researchers and practitioners to adapt the model to their specific needs, ensuring widespread accessibility and replicability in various contexts. Some of the outcomes were further disseminated through ‘The Conversation’, a platform that fosters collaboration between journalists and researchers, using Creative Commons licences to communicate scientific findings to the general public. The scientific results of each core chapter of this dissertation were presented at international conferences spanning diverse academic backgrounds, including the Nordic Society Oikos (2021), Ecosystem Services Partnership regional conferences in the EU (2022), LAC (2023), and the upcoming European Geosciences Union (2024). In addition, over the three-year duration of this thesis, progress was communicated to various research groups (BCNUEJ –Barcelona Lab for Urban Environmental Justice, Ecoinformatics and Biodiversity –Aarhus University, CGIS – Brussels Centre for Urban Studies) through seminars covering different topics such as environmental justice, ecology and biogeography, and urban climate. These seminars aimed to encourage discussions around the research presented in this thesis. Finally, the outcomes and implications of the thesis were shared in meetings with the Environmental Studies Centre (City council) in Vitoria-Gasteiz.

## 5.3. Limitations and caveats

Focusing on the ecological and utilitarian aspects of UGI for human well-being, this dissertation acknowledges the potential oversight of non-utilitarian values and cultural perspectives in UGI. This aligns with broader challenges in ES literature, emphasising the need for a more comprehensive understanding of values and social structures shaping biodiversity, UGI, and associated ES (Pascual *et al.*, 2023; Langemeyer & Connolly, 2020; Ernstson, 2013). While analytically useful, the focus of this dissertation may limit our understanding of how people perceive and distribute actual benefits (Langemeyer & Connolly, 2020). For instance, the analysis of temperature regulation ES in **Chapter IV** could benefit from a complementary study on residents' preferences and values, which influence their subjective vulnerability to heat, but such an endeavour is time-consuming and resource-intensive for a 3-year PhD. The integration of the heat vulnerability index partially addresses this limitation by considering the multifaceted aspects of ES demand. Despite these challenges, theis dissertation explores socioeconomic and historical factors influencing UGI (**Chapter III**), addressing greening inequities in relation to ES demands (**Chapters III – IV**).

Modelling ES approaches as the used in this dissertation, although effective, usually have inherent limitations tied to theoretical assumptions, data quality, and potential modelling uncertainties (Bagstad *et al*., 2013). Expert consultation in **Chapter II**, while a valuable method, is prone to biases stemming from diverse expertise backgrounds, influencing HQ parametrisation (Di Febbraro *et al*., 2018; Sallustio *et al*., 2017). To overcome this, we included consultation from experts in different areas and then identified and excluded extreme values in the valuation of the descriptors from the analysis. Thus, the HQ results aligned with the relative UW spatial patterns and species richness.

Further limitations arise from the use of secondary data. The reliability of crowdsourced data in assessing potential associations with UW and HQ indicators is influenced by biases arising from overrepresented data in easily accessible green and blue spaces, highly populated areas, or circulation ways, as noted in other studies (Sultana & Storch, 2021; Petersen *et al.*, 2021; Callaghan *et al*., 2020). For instance, the popularity of birds in citizen science, evident in increased observations on the 'ornitho.eus' platform, may lead to an overestimation of bird richness. Despite these biases, such data are invaluable for studying large spatial scales and provide a large amount of data, thereby minimising errors and rendering them suitable for inclusion in urban biomonitoring programmes (Callaghan *et al*., 2020). Within the context of a middle-sized city such as Vitoria Gasteiz, the positive correlation between UW and HQ results in robust proxies for predicting biodiversity across various taxa. This suggests that, with the available data on biodiversity, land use, habitat suitability, and potential threats, these databases prove reliable for estimating urban biodiversity. However, there remains unexplained variation between these proxies and biodiversity, attributed in part to factors like seasonal variability, unassessed threats, habitat conditions, biogeographical filters, and the impact of invasive species, which were not addressed in this study.

Further biases were derived from data availability. In the case of **Chapter III**, the iTree-Eco tool was only applicable to public canopy cover, lacking data on private canopy cover. I partially addressed this limitation by integrating remote sensed vegetation cover classifications, although it was not possible to estimate the ES for this private canopy cover. In addition, it is important to acknowledge the absence of biodiversity data on aquatic species in both Chapters II and III, which restrict the use of these models when biodiversity data for a certain type of ecosystem is underrepresented. Similarly, this data limitation restricted the analysis of the luxury effect for aquatic ecosystems in **Chapter III**, leaving a research gap in luxury effect assessments. Likewise, we could not consider the influence of aquatic ecosystems on temperature regulation ES modelling in **Chapter IV** as the relationship with these ecosystems is still not well understood (Sebastiani *et al*., 2021). Therefore, the lack of data availability for aquatic ecosystems is a consistent gap that restricts their use for ES studies and maintains a restricted knowledge on related processes, such as the relationship with the luxury effect or urban heat.

The challenges outlined in **Chapter IV** extend to quantifying and mapping temperature regulation indicators by relying on model-based proxies through composite indicators. The identified limitations associated with using relative values for supply, demand, and mismatches underscore the need for caution when interpreting and generalising findings to other cities. The use of standardised indicators in composite mapping introduces subjectivity and uncertainties regarding the process of building the composite indicator and the comparability of considered variables. While unitless composite indicators address incommensurability to some extent, they assign equal weights to the HVI index, potentially overlooking variations in the actual contributions of individual variables. The combination of heterogeneous data may oversimplify the underlying complexity of the phenomenon under assessment. To address this limitation and as a potential avenue for future research, refining methods to obtain empirical data at the census tract level and aligning spatial scales more precisely could enhance the accuracy and reliability of ES mapping in urban areas.

## 5.4. Future research

The research outlined in this dissertation provides insights into urban ecology, environmental justice, and biodiversity conservation and highlights critical gaps in current modelling approaches. These approaches, predominantly developed in the data-rich contexts of the Global North, face challenges when applied to the specific needs of urban environments in the Global South (Hamel *et al*., 2021). While the methodology used here aligns with the Global North context, using complex models designed for abundant data, as demonstrated in **Chapter III** using iTree Eco, its applicability in the Global South faces challenges due to data accessibility constraints. This dissertation takes steps to bridge this gap by offering modelling approaches in **Chapters II** and **IV** that are both easily adaptable and replicable, requiring relatively low data inputs.

Future research must address such potential implementation gaps in the Global South, considering the prevalent inequalities and environmental injustices that constitute one of the main characteristics of UGI in these regions (Pauleit *et al.*, 2021). The call is for a nuanced understanding of the socio-ecological dynamics in the Global South, enabling the development of models that capture the complexity of these urban environments. This approach would not only advance our understanding of urban ecology patterns but also contribute to actionable insights for sustainable urban development, promoting equitable outcomes, which becomes capital in markedly unequal regions.

Future research should investigate the impact of the uneven distribution of urban nature on the perceptions, experiences, and engagement of urban dwellers (Marvier *et al.*, 2023; Dunn *et al.*, 2006). Adopting a pluralistic view of urban biodiversity, recognising the value of urban ecosystems beyond societal utility, offers opportunities to incorporate co-production practises into conservation initiatives (Lambert & Schell, 2023; Pascual *et al.*, 2023; 2021). This shift towards co-production is facilitated when there is a reduced emphasis on procedural and taxonomical concerns and a greater focus on understanding the "why" behind conservation efforts. Shifting away from strict biodiversity definitions allows an examination of the reasoning behind conserving particular species and the intended recipients of these conservation efforts (Lambert & Schell, 2023). Furthermore, research should examine the impact of strategies such as creating or restoring green spaces in underserved communities, which can offer valuable insights into enhancing positive connections with nature. Assessing both the social and ecological outcomes of these strategies could clarify the dynamics between environmental justice, UGI planning, and conservation.

The urgent need to understand how socioeconomic factors are linked to biodiversity patterns in Global South countries is also emphasised, as luxury and legacy effects have predominantly been studied in Global North countries, resulting in a significant knowledge gap regarding biodiversity drivers in diverse contexts (Kuras *et al.*, 2020; Chamberlain *et al.*, 2019). Exploring the ecological processes underpinning the luxury effect, such as the impacts of affluent and diverse neighbourhoods on urban resilience and adaptation to climate change, requires further investigation. Future efforts should focus on elucidating the specific mechanisms linking affluence to biodiversity, recognising that biological communities respond not merely to affluence itself but to the presence, quality, or absence of intermediate resources or conditions often represented by affluence (Kuras *et al*., 2020).

It is key to determine how these observed patterns extend to aquatic biodiversity and ecosystem functions and benefits, a facet not addressed in this study due to data constraints, aligning with previously identified research gaps (Leong *et al*., 2018). Furthermore, the biodiversity found in various urban commons beyond common binary distinctions, such as private or public, formal or informal, calls for exploration, as these are likely shaped by a diverse set of individuals and conditions. Unlike more controlled and regulated environments, some urban commons like ponds, community gardens, or forest remnants, may exhibit a less clear distinction in terms of management and ownership, blurring the lines between what is considered private or public and formal or informal in the management of these spaces (Kuras *et al*., 2020; Seto *et al*., 2010).

Building on the insights provided by this dissertation, particularly the exploration of luxury and legacy effects and their connection to urban biodiversity, UGI, and ES, future research should address imbalances in ecological resources and landscape features within urban socio-ecological systems. Identifying factors influencing the demand for and co-production of ES among diverse social groups (Haase *et al*., 2014; Andersson *et al*., 2014), will contribute to more informed and equitable urban conservation strategies.

## 5.5. Recommendations for urban policy and practise

This dissertation highlights the crucial role of ecological quality in supporting urban biodiversity and enhancing the connection between urban dwellers and nature, moving beyond considerations of access and quantity. The recommendations presented here focus on UGI to guide policy implementation in urban greening and conservation, aligning with a broader shift towards just and green urban planning. By optimising UGI and carefully addressing local community needs, urban planning can effectively contribute to overall environmental sustainability, creating green spaces that are both biodiverse and socially equitable.

Preserving larger components of UGI, especially in peripheral areas, is crucial for safeguarding urban biodiversity, considering factors such as size, connectivity, and perimeter:area ratio (*e.g.*, **Chapter II**). This becomes relevant for ongoing discussions about the impact of extensive or intensive and compact urbanisation models on biodiversity (Stott *et al*., 2015). It is necessary to target the increase in biodiversity and ensure species persistence, by enhancing size, quality, and connectivity while mitigating edge effects in small, fragmented spaces by reducing the perimeter:area ratio (Turner & Gardner, 2015). Any UGI development scheme, beyond providing multiple benefits, should be carefully considered for its impact on urban biodiversity as a basic biophysical attribute influencing the provision of ES. Urban planning should prioritise rewilding strategies in larger central urban green spaces, minimising exposure to threats such as transportation networks, light pollution and urban densification, while increasing HQ for effective biodiversity support.

Yet, the implications of enhancing biodiversity and wildness in urban areas warrant careful consideration. Perception studies (Samus *et al*., 2022; Hofmann *et al*., 2012) often suggest that the management regime of UGI consistently influences its valuation. Urban residents tend to favour intensively managed areas over wilder ones (Kowarik, 2018). Nevertheless, residents express a willingness to embrace 'wild' environments if minimal maintenance and accessibility are provided, along with visible signs of care (Kowarik, 2018; Jorgensen & Tylecote, 2007). Therefore, the design of urban wild areas should carefully communicate that any perceived abandonment is intentional and planned, rather than a result of neglect (Müller *et al*., 2018; Botzat *et al*., 2016). Greening interventions should first target areas with low HQ and little or no UGI, particularly focussing on central areas where these conditions are prevalent (**Chapters II-IV**). The logarithmic response of biodiversity to HQ identified in **Chapter III** emphasises that enhancing HQ at lower ranges may have a greater impact on species richness than at higher ranges. Therefore, targeted efforts to improve and maintain HQ, especially in areas with lower initial HQ, can substantially contribute to biodiversity enhancement. The findings emphasise the need for a context-specific biodiversity management approach, directing resources and interventions towards improving UGI quality in less affluent or degraded neighbourhoods. Rather than applying uniform strategies, planners and conservationists should consider the existing HQ and tailor interventions accordingly. Understanding the non-linear relationship between HQ and species richness enables urban planners to allocate resources efficiently, maximising the positive impact on biodiversity in a more targeted manner. These findings call for a shift from uniform conservation strategies to context-specific actions, considering the existing HQ for effective and sustainable urban biodiversity conservation.

In compact cities where urban densification hinder ES provision (Sebastiani *et al*., 2021; Mascarenhas *et al*., 2019), there is a greater need to merge grey–green infrastructure for the implementation of UGI. As land availability constraints challenge UGI extension, increasing the vegetation stratum complexity and canopy coverage becomes a useful alternative to improve the functionality of the already existing UGI (Francoeur *et al*., 2021; Müller *et al*., 2018; Stott *et al*., 2015). In addition, optimising UGI spatial distribution through various components, such as green roofs, green walls, pocket parks, and larger green spaces, interconnected for efficient resource use, is deemed crucial (Langemeyer *et al*., 2020).

Our findings underscore the relevance of enhancing adaptive capacities within urban landscapes through targeted greening efforts that address specific population needs, allowing for assessment and spatial mapping (**Chapter IV**). Achieving equitable access to ES requires management efforts targeting the ecological status of older neighbourhoods with less UGI, enhancing both HQ and greening for improved biodiversity. The positive correlation between high educational attainment and species richness, particularly pronounced in affluent neighbourhoods and better HQ, underscores the luxury effect. As HQ improves, this correlation strengthens, potentially creating a reinforcing feedback loop that accentuates biodiversity benefits in affluent areas. To foster more equitable urban biodiversity outcomes, conservation strategies should prioritise interventions targeting HQ in less affluent or degraded neighbourhoods, breaking the reinforcing loop. The expansion and conservation of UGI must prioritise access to their benefits for vulnerable groups to avoid worsening environmental injustices and luxury effects. Hence, it is critical to prioritise access to these benefits for those most exposed to hostile environmental conditions, particularly individuals and communities.

Greening efforts must consider accessibility and social equity to guard against potential unintended consequences such as green gentrification, which can uphold or even worsen existing inequalities (Nesbitt *et al*., 2023; Anguelovski *et al*., 2020). Addressing these challenges requires a multifaceted policy strategy that integrates greening efforts and environmental enhancement with social equity considerations. Measures such as affordable housing, community land trusts, and rent control are key to preventing community displacement (Oscilowicz *et al*., 2021). Prioritising the repurposing of underutilised spaces, creating provisional green spaces on vacant lands, and enhancing existing UGI in underserved areas should take over a sole focus on creating new green spaces (Oscilowicz *et al*., 2021). This would contribute to more equitable greening initiatives and sustainable urban development, aligning biodiversity conservation with broader urban development goals.

## 5.6. Conclusions

This dissertation underscores the fundamental role of biodiversity in providing ES in cities and the drivers affecting its uneven distribution. Biodiversity supports the capacity of urban ecosystems to provide ES. However, this association encompasses not only the direct indicators of biodiversity such as species richness, but also key biophysical indicators of habitat functionality to sustain biodiversity: HQ and UW. Thus, integrating these complementary habitat and wildness indicators is key for a comprehensive analysis of the multifunctionality of UGI and, in particular, to better understand the factors determining its capacity to provide ES. This research also highlights that the distribution of biodiversity in cities, including HQ, is shaped by social inequalities and historical management legacies associated with luxury and legacy effects. This uneven distribution of nature, ultimately determines the differential access to ES among the urban population and therefore shapes the relationship between urban dwellers and nature. This was evidenced regarding the ES of temperature regulation, where such inequalities were reflected in profound mismatches between the supply and demand, particularly for groups undergoing different situations rendering them vulnerable to urban heat hazards.

This inequality in the access to the benefits of urban nature differently affects various socioeconomic and demographic groups, emphasising injustices in ES distribution and impacting the direct effects of ES on human well-being. Consequently, UGI management becomes a determining factor in ES provision and must be approached equitably to counteract environmental inequities. This ensures a comprehensive and equitable approach to urban biodiversity conservation, which is critical for fostering just and sustainable urban development as well as the overall goal of environmental justice. This dissertation calls for greening strategies that emphasises not only the expansion of UGI, but that also enhance ecological integrity to support higher biodiversity and related ES. Such strategies, should always be based on the equitable social distribution of urban nature and related benefits, from an from an environmental justice perspective that allows to better face current and future environmental challenges.

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# Supplementary material Chapter II

**Table S1.** Joined datasets used to define study's area land use and naturalness classes. Source: Environmental Studies Centre (as CEA in Spanish), an autonomous organization that is associated with the Vitoria-Gasteiz City Council.

| Datasets | Description | Data Format |
|---|---|---|
| **Urban Land Use Map (2020)** | Land uses included in the city of Vitoria-Gasteiz (excluding the green belt) | *Shapefile* |
| **Green Belt Land UseMap (2020)** | Parks and land uses included in the green belt of Vitoria-Gasteiz | *Shapefile* |
| **Nature-Based Solutions Inventory (2020)** | Green infrastructure actions and Nature-Based Solutions (NBS). Includes details on the intervention, NBS type and description | *Shapefile* |

**Table S2**. Reclassification of the land uses from Urban Land Use Map (2020); Green Belt Land Use Map (2020) and Nature-Based Solutions Inventory (2020) adapted from Müller *et al*. 2018. Land uses source: Environmental Studies Centre (as CEA in Spanish).

| Naturalness value | Description | Original land use classes |
|---|---|---|
| **1** | Completely sealed areas | Buildings, bus lanes, highways, pavement and asphalt paths, pavement areas, parking lot, warehouse, main road, secondary road, residential roads |
| **2** | Mostly sealed, but there could also be some unsealed parts | City centre, bike lanes, industrial sites, open pool, water fountains, water pumping, treatment plant |
| **3** | Sealed and green areas mixed | Building greens, ground-based and façade green walls, low built-up, sidewalks, trails, green roundabouts, extensive green roofs, green interventions on pavement areas |
| **4** | Highly intensive agriculture and extraction sites | Pot plants, resource extraction and brownfields |
| **5** | Intensive agriculture | Agriculture, crop fields, afforestation |
| **6** | Less intensive and extensive agriculture, horticulture | Horticulture, greenhouse, umbraculum, other agricultural land use |
| 7 | Sports-grounds grassland | centre |
| **8** | Recreative areas | Areas for recreative purposes, large urban parks, pocket park, botanical garden, cemetery and churchyard, camping areas, decorative greenery, public area, recreation area |
| **9** | Orchards, allotments and community gardens, biodynamic farming | Orchards, edible forest, nursery garden, composting area. Community gardens allotments. |
| **10** | Permanent grassland with normal yields | Permanent grass normal yield. |
| **11** | Permanent grassland with low yields | Permanent grass, low yield, used grasslands with water drilling |
| **12** | Meadows, dry meadows, lawns | Meadow and dry meadow (also includes garden/park grasslands, grassland along roads) |
| **13** | Non-native trees, afforestation | Re-afforestation area with an environmental focus, state-afforestation |
| **14** | Relatively extensive open landscapes, trees and bushes also in urban areas, native evergreen species | Bushes and trees (also along roads, in hedges), shrub land patches, abandoned sites with spontaneous vegetation, naturalized industrial border, evergreen oak, |
| **15** | Fallows, forests with also non-native deciduous species, (wet) grasslands often less intensely cut | Area with environmental focus – relict vegetation, fallow in marginal zones, freshwater meadow, Maple, plain (dry grassland), stream bank, wet meadow |
| **16** | Deciduous native trees, forest | Oak, riparian forest, juniper forest, application for environmental assurances, not specified deciduous tree |
| **17** | Area presumably under little human influence | Nature areas, open nature in a protected forest, foothills-scrubland, rich pond, wetland |
| **18** | Land cover presumably under least human influence | High mountain, lake, stream |

**Table S3.** Cost surface for the calculation of remoteness from mechanized access (*i.e*. values are the seconds needed to pass through one 10 m pixel). Some of the land use classes were summarized into broader classes according to Müller *et al*. (2018). Some land use classes as freshwater ecosystems were assigned NoData as travel time since they were handled as not passable barriers. Cost values follows Müller *et al*. (2018).

| **Land Use Classes** | **Travel time in km/h** | **Cost value in s** |
|---|---|---|
| Roads (secondary, residential), bus lanes, trails | 15 | 2 |
| grasslands, agriculture, areas for recreative purposes, vacant land, open nature in protected forest, public area, plain, beach, parking lot, harbour, technical area, city centre, industry, recreation area, sports ground, horticulture, cemetery, coast, ceased field, marginal zone, nature and nature like areas | 5 | 7 |
| orchards, fallows, bushes and trees, stream banks | 4 | 9 |
| any kind of forest or tree plantation, scrub, heather | 3 | 12 |
| mire/bog, wetland | 2 | 18 |
| lake, ruin/barrow, rich pond, highway, secondary highway, railway 1 track, railway ≥ 2 tracks, basin, low built-up, high built-up, building, resource extraction, sea, stream, closed system annual | NoData (barrier) | NoData |

**Table S4**: Reclassification and aggregation of Land Use Classes database to Urban Green Space Typology, based on Hansen *et al*. (2017). This classification was used for the expert-based survey to parametrize the InVEST HQM.

| Land Use Classes | Urban Green Space Typology |
|---|---|
| Roads (secondary, residential), bus lanes, trails | Sealed areas |
| Building greens, ground-based and façade green walls, low built-up, sidewalks, trail, green roundabouts, extensive green roof, green interventions on pavement areas | Building greens |
| Agriculture, crop fields, afforestation. Horticulture, greenhouse, umbraculum, other agricultural land use. Permanent grass normal yield. Permanent grass, low yield, used grasslands with water drilling. | Agricultural land |
| Meadow and dry meadow (also includes garden/park grasslands, grassland along roads). Sports ground, playgrounds, campus, sports centre. | Private, commercial, or institutional green space |
| Areas for recreative purposes, large urban parks, pocket park, botanical garden, cemetery and churchyard, camping areas, decorative greenery, public area, recreation area | Parks and recreation |
| Orchards, edible forest, nursery garden, composting area. Community gardens allotments. | Community gardens |
| Oak, riparian forest, forest, application for environmental assurances, not specified deciduous tree, nature areas, open nature in protected forest, foothills-scrubland, rich pond, wetland, high mountains | Natural, semi-natural and feral areas |
| lake, sea, stream | Waters |

**Questionnaire S1:** Questionnaire on the definition of the input indicators of the InVEST Habitat Quality Model for the Urban context. Double click on the figure to open the pdf containing the full questionnaire.

11/2/21, 9:39 AM    Parametrization of habitat quality models for urban areas

# Parametrization of habitat quality models for urban areas

This form aims to collect qualified opinions from experts about how different land uses and covers may influence the habitat to support biodiversity in urban contexts.

Here we defined and adapted our LULC classes from the green infrastructure typology developed by Hansen et al. (2017) for green infrastructure planning. This typology is made up of more than 40 types of green spaces and other land uses clustered into 8 groups.

You will be asked to rate the potential impacts, positive or negative, of the different land uses – land covers on supporting urban biodiversity in its broader sense (not restricted to any particular taxonomic group or species).

The weighted indicators obtained will be used to adapt the InVEST (Integrated Valuation of Ecosystem Services and Trade-offs) Habitat Quality Model to urban contexts whose outcome will be two maps representing urban habitat quality and of urban habitat degradation.

Your inputs will be valuable contributions to our research.

More info about InVEST: http://releases.naturalcapitalproject.org/invest-userguide/latest/habitat_quality.html

*Obligatorio

1. Correo *

2. What is your field of expertise? *

3. I have read and I accept the conditions stated on the informed consent and data protection document: https://docs.google.com/forms/d/e/1FAIpQLSdFT4H7a9KdKn4L011lQbqcsEOupAmhJ0xr2s0SOsPiLkgG4g/viewform?usp=sf_link *

*Selecciona todos los que correspondan.*

☐ Yes

https://docs.google.com/forms/d/1YEI2CHhnBYYZlpHvLluLI7KZWs-dVjWKsvmjooODctA/edit    1/14

See also: https://drive.google.com/file/d/1x-BQLRbjNVlgX7PsotpVVjlXxwzfeNBg/view?usp=sharing

**Table S5.** Input values of Habitat suitability (Hj), mean (μ), standard deviation (STD) and the coefficient of variation (CV) per each LULC class, resulting from the experts' survey.

| **LULC Types** | **Habitat suitability (*Hj*) μ (STD, CV)** |
|---|---|
| **Predominantly sealed areas** | 0.04 (0.09, 2.38) |
| **Grey and green areas mixed** | 0.44 (0.13, 0.31) |
| **Agricultural land** | 0.49 (0.15, 0.30) |
| **Private or institutional green areas** | 0.45 (0.15, 0.33) |
| **Extensive agricultural lands** | 0.61 (0.15, 0.24) |
| **Parks and recreation** | 0.61 (0.15, 0.24) |
| **Grasslands** | 0.49 (0.15, 0.30) |
| **Orchards, allotments and community gardens** | 0.66 (0.20, 0.30) |
| **Afforestation** | 0.45 (0.15, 0.33) |
| **Natural, semi-natural and feral areas** | 0.95 (0.10, 0.11) |
| **Water bodies** | 0.89 (0.12, 0.14) |

**Table S6.** Input values for the sensitivity to threats ($S_{jr}$ ), their relative weights, ($W_r$) and their maximum influence distance (*Max.D*), (mean ($\mu$), standard deviation (STD) and the coefficient of variation (CV)) per each LULC class, resulting from the experts' survey.

| Threats / LULC | Predominantly sealed areas | Grey and green areas mixed | Agricultural land | Private or institutional GS | Extensive agricultural lands | Parks and recreation | Grasslands | Orchards, allotments and community gardens | Afforestation | Natural, semi-natural and feral areas | Water bodies | Wr (0-1) | Max. D (km) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Railways | 0.14 (0.22, 1.59) | 0.18 (0.20, 1.09) | 0.37 (0.27, 0.72) | 0.32 (0.20, 0.63) | 0.47 (0.29, 0.62) | 0.47 (0.29, 0.62) | 0.37 (0.27, 0.72) | 0.44 (0.31, 0.71) | 0.32 (0.20, 0.63) | 0.63 (0.27, 0.44) | 0.52 (0.32, 0.62) | 0.49 (0.18, 0.38) | 0.5 (0.4, 0.8) |
| Nightlight | 0.20 (0.20, 1.02) | 0.57 (0.24, 0.42) | 0.55 (0.25, 0.46) | 0.50 (0.25, 0.50) | 0.62 (0.19, 0.30) | 0.62 (0.19, 0.30) | 0.55 (0.25, 0.46) | 0.65 (0.23, 0.35) | 0.50 (0.25, 0.50) | 0.75 (0.21, 0.28) | 0.70 (0.24, 0.35) | 0.64 (0.24, 0.37) | 1.7 (1.9, 1.1) |
| Roads 1: Primary roads. | 0.29 (0.35, 1.17) | 0.50 (0.28, 0.56 | 0.66 (0.23, 0.35) | 0.63 (0.24, 0.39) | 0.75 (0.13, 0.17) | 0.75 (0.13, 0.17) | 0.66 (0.23, 0.35) | 0.71 (0.20, 0.29) | 0.63 (0.24, 0.39) | 0.87 (0.18, 0.21) | 0.73 (0.25, 0.34) | 0.92 (0.12, 0.13) | 0.8 (0.3, 0.4) |
| Roads 2: Secondary roads. | 0.15 (0.23, 1.52) | 0.28 (0.18, 0.64) | 0.45 (0.14, 0.30) | 0.47 (0.25, 0.53) | 0.53 (0.09, 0.16) | 0.53 (0.09, 0.16) | 0.45 (0.14, 0.30) | 0.47 (0.17, 0.37) | 0.47 (0.25, 0.53) | 0.66 (0.20, 0.30) | 0.64 (0.22, 0.35) | 0.69 (0.14, 0.20) | 0.5 (0.3, 0.6) |
| Roads 3: Residential roads. | 0.13 (0.16, 1.20) | 0.15 (0.15, 1.05) | 0.23 (0.17, 0.73) | 0.28 (0.18,0.64) | 0.34 (0.12, 0.36) | 0.34 (0.12, 0.36) | 0.23 (0.17, 0.73) | 0.29 (0.18, 0.62) | 0.28 (0.18,0.64) | 0.49 (0.22,0.46) | 0.48 (0.28, 0.58) | 0.46 (0.20, 0.44) | 0.4 (0.4, 1.0) |
| Noise pollution (65 to 70 dB) | 0.23 (0.26, 1.11) | 0.42 (0.24, 0.59) | 0.40 (0.25, 0.62) | 0.40 (0.20, 0.50) | 0.43 (0.15, 0.34) | 0.43 (0.15, 0.34) | 0.40 (0.25, 0.62) | 0.45 (0.22, 0.48) | 0.40 (0.20, 0.50) | 0.57 (0.18, 0.31) | 0.43 (0.27, 0.63) | 0.47 (0.22, 0.46) | 0.7 (0.8, 1.1) |
| Harmful noise pollution (>70 dB) | 0.23 (0.26, 1.11) | 0.62 (0.25, 0.40) | 0.60 (0.25, 0.41) | 0.57 (0.27, 0.47) | 0.68 (0.20, 0.29) | 0.68 (0.20, 0.29) | 0.60 (0.25, 0.41) | 0.65 (0.28, 0.43) | 0.57 (0.27, 0.47) | 0.78 (0.23, 0.29) | 0.62 (0.34, 0.55) | 0.69 (0.22, 0.32) | 1.3 (1.1, 0.9) |
| Low Population density | 0.32 (0.35, 1.10) | 0.39 (0.25, 0.65) | 0.63 (0.25, 0.39) | 0.60 (0.24, 0.40) | 0.73 (0.20, 0.27) | 0.73 (0.20, 0.27) | 0.63 (0.25, 0.39) | 0.58 (0.28, 0.43) | 0.60 (0.24, 0.40) | 0.82 (0.18, 0.22) | 0.88 (0.13, 0.15) | 0.76 (0.20, 0.26) | 0.9 (0.9, 1.1) |
| Medium Population density | 0.32 (0.35, 1.10) | 0.39 (0.25, 0.65) | 0.63 (0.25, 0.39) | 0.60 (0.24, 0.40) | 0.73 (0.20, 0.27) | 0.73 (0.20, 0.27) | 0.63 (0.25, 0.39) | 0.58 (0.28, 0.43) | 0.60 (0.24, 0.40) | 0.82 (0.18, 0.22) | 0.88 (0.13, 0.15) | 0.76 (0.20, 0.26) | 1.8 (2.5, 1.4) |
| High Population density | 0.32 (0.35, 1.10) | 0.39 (0.25, 0.65) | 0.63 (0.25, 0.39) | 0.60 (0.24, 0.40) | 0.73 (0.20, 0.27) | 0.73 (0.20, 0.27) | 0.63 (0.25, 0.39) | 0.58 (0.28, 0.43) | 0.60 (0.24, 0.40) | 0.82 (0.18, 0.22) | 0.88 (0.13, 0.15) | 0.76 (0.20, 0.26) | 5.2 (7.9, 1.5) |

**Table S7**. GLM models with non-normal data distribution (untransformed data) for the whole study area showing: p, F and adjusted $R^2$ values.

| Treatment | F statistic | P-value | Adjusted $R^2$ |
|---|---|---|---|
| **Avian richness ~ HQ mean** | 27.95 | 3.18e-07 *** | 0.116 |
| **Avian richness ~ Wildness mean** | 22.27 | 4.39e-06 *** | 0.094 |
| **Butterfly richness ~ HQ mean** | 3.538 | 0.064 | 0.033 |
| **Butterfly richness ~ Wildness mean** | 8.466 | 0.00479 ** | 0.092 |
| **Mammal richness ~ HQ mean** | 9.549 | 0.00266 ** | 0.089 |
| **Mammal richness ~ Wildness mean** | 10.46 | 0.00171 ** | 0.094 |

**Table S8.** Residuals from the relationship of wildness and habitat quality with richness (Birds, Mammals, and Butterflies), respectively, regressed against related variables. Results (no influence nor significant models) indicate that the tested variables are not influencing richness independently richness, meaning that they have identical relations to richness.

| **Tested model** | | **Taxon** | | |
|---|---|---|---|---|
| **Residual** | **Variables** | **Birds** | **Butterfly** | **Mammals** |
| **Wildness mean~Richness** | HQ mean | Adj $R^2$ = 0.007<br>P-value= 0.11 | Adj $R^2$ = 0.013<br>P-value= 0.97 | Adj $R^2$ = 0.013<br>P-value= 0.97 |
| **Mean HQ ~Richness** | Wildness mean | Adj $R^2$ = -0.004<br>P-value= 0.86 | Adj $R^2$ = 0.006<br>P-value= 0.23 | Adj $R^2$ = -0.008<br>P-value= 0.65 |

## Sensitivity analysis

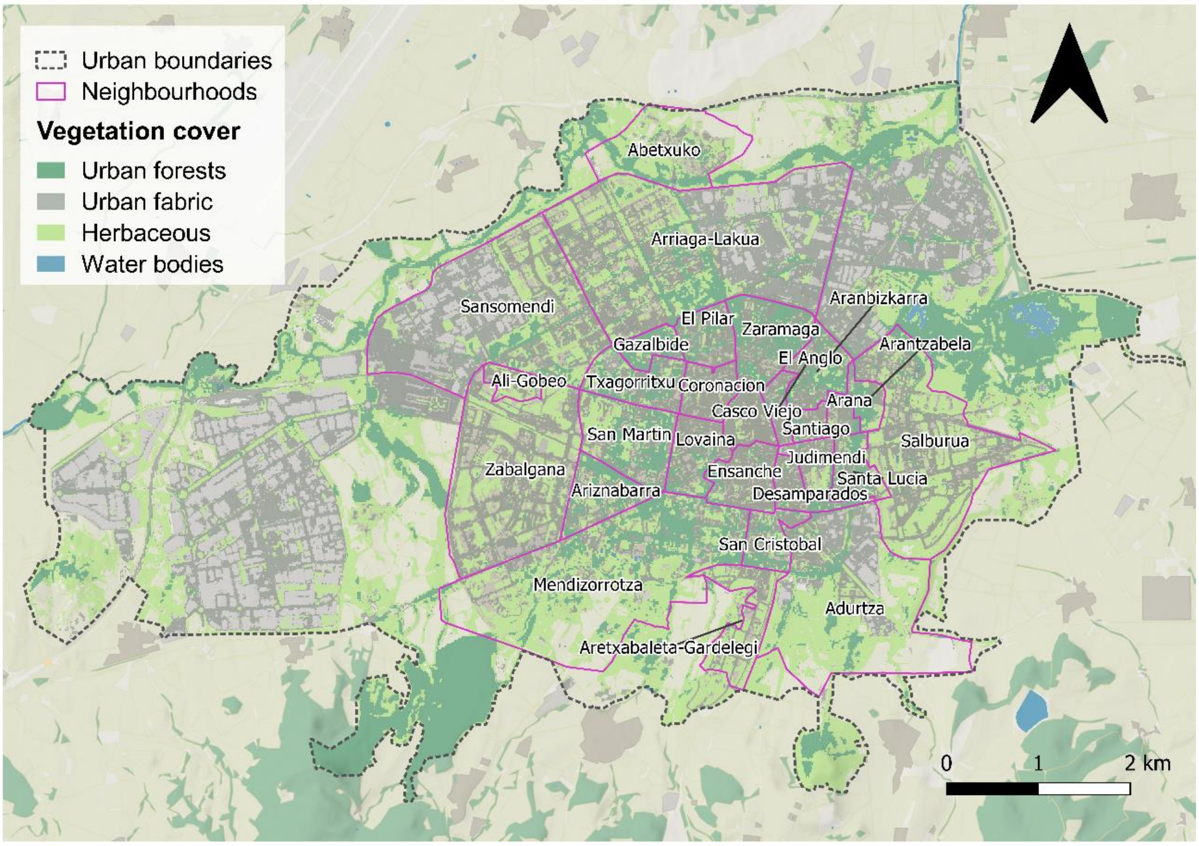
We examined the robustness of our relative urban wildness (UW) mapping by using a Jack-knife, or leave one out approach, excluding one of the indicator maps, respectively and evaluating the effect of removing one indicator input from the original UW map. When accounting for the overall uncertainty of our three indicators inputs, the coefficient of variation was the highest, by 130% for perceived naturalness while 70% and 80% for remoteness and ruggedness, respectively. The mean differences were higher (17.2) when removing perceived naturalness, while there was a modest increase (-8.4) in the mean values after removing remoteness and almost no variation (0.5) when removing ruggedness. Compared to the map including all the indicators, our results suggest that perceived naturalness is the input most influencing the final UW spatial outcomes (Table S7). The correlation coefficients of the full UW map, and the three alternative UW maps (Table S7) considering only two inputs, respectively, all showed R correlation values >0.7, indicating a positive linear relationship. Even after removing one indicator from the map, the general pattern in pixel values remained stable

**Table S7.** Comparison of the alterations in UW values between the original (full map including all the three indicators) and three alternative UW maps after jack-knifing (including only two indicators each)

| Indicator excluded | SD | μ difference in UW values | CV | Correlation coefficient R |
|---|---|---|---|---|
| **Perceived naturalness** | 26.2 | 17.2 | 1.3 | 0.748 |
| **Remoteness** | 45.3 | -8.4 | 0.7 | 0.966 |
| **Ruggedness** | 49.9 | 0.5 | 0.8 | 0.967 |

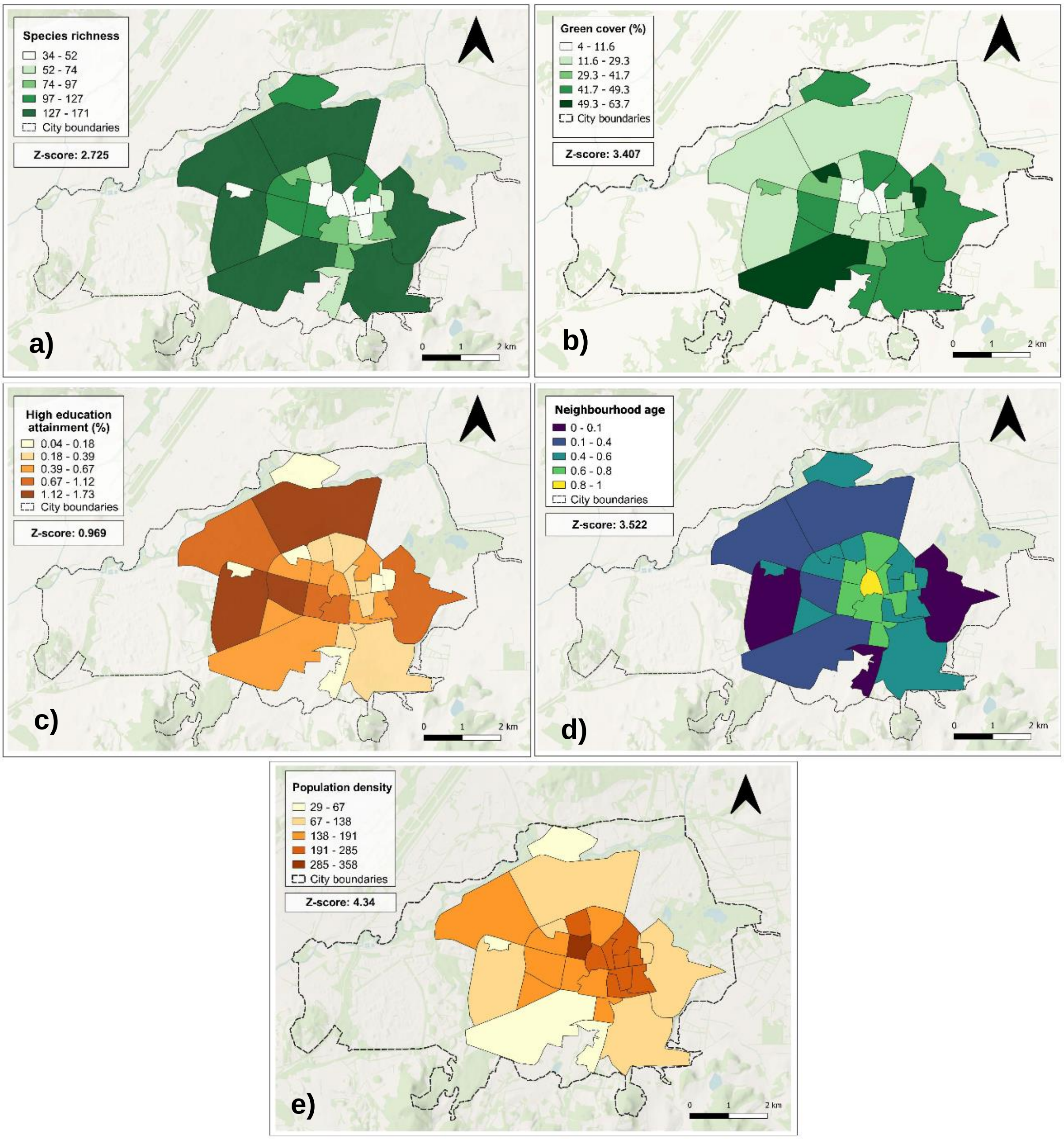
# Supplementary Material III

Herbaceous land cover is abundant in newer neighbourhoods located towards the outskirts of the urban core (see appendix) such as Salburua (36.45%) or Aretxabaleta-Gardelegi (34.33%) (Fig. S1), while canopy cover is higher towards older neighbourhoods such as Gazalbide (63.69%) or Zaramaga (35.32%) in the urban core. The neighbourhoods with the highest percentage of vegetation cover (*i.e.* herbaceous and canopy cover) were Gazalbide, Arantzabela and Mendizorrotza (Fig S1), with 63.69%, 62.91% and 56.52% respectively, while the neighbourhoods with the lowest percentage of vegetation cover were Casco Viejo, Coronación and El Anglo with 4.01%, 5.24% and 11.59%, respectively.




**Supplementary figure 1.** Spatial distribution of the neighbourhoods, their delimitation (in pink) and urban boundaries (black). Background from © MapTiler (www.maptiler.com) and © OpenStreetMap (CC BY-SA 2.0).

**Supplementary figure 2.** Spatial distribution of predictor variables at the neighbourhood level: a) Species richness (total count), b) Vegetation cover (%), c) High education attainment (%), d) Neighbourhood age (range 0- newer developments to 1 –older developments) and e) Population density (per hectare, mean values). The predictor values were classified using the natural breaks method. Species richness, green cover and neighbourhood age show a statistically significant clustering given the Moran's Index z-score values. Background from © MapTiler (www.maptiler.com) and © OpenStreetMap (CC BY-SA 2.0).



**Supplementary Table 1**. Description of the different considered variables for each analysis unit

Note:
* Neighbourhood age is represented by an index ranging from 0 (newer) to 1 (older) (see Table S1 – Supp. Mat S1)
** High education attainment represents the percentage of population with tertiary education level over the total population.
*** Habitat quality represents the average value of habitat quality per neighbourhood

| ID | Neighbourhood | Neighbourhood age * | High education (%) ** | Population density (inhab/ha) | Area (ha) | Species richness (count) | Vegetation cover (%) | Canopy cover (%) | Herbaceous cover (%) | Habitat quality *** |
|---|---|---|---|---|---|---|---|---|---|---|
| **1** | CASCO VIEJO | 1 | 0.55 | 301.51 | 28.56 | 47 | 4.01 | 1.71 | 2.3 | 0.05 |
| **2** | ENSANCHE | 0.776 | 1.11 | 179.08 | 45.643 | 90 | 19.94 | 15.38 | 4.56 | 0.12 |
| **3** | LOVAINA | 0.665 | 0.92 | 177.26 | 44.074 | 104 | 25.89 | 19.4 | 6.49 | 0.11 |
| **4** | CORONACION | 0.741 | 0.63 | 369.73 | 32.517 | 35 | 5.24 | 3.66 | 1.58 | 0.04 |
| **5** | EL PILAR | 0.57 | 0.32 | 259.35 | 35.357 | 63 | 24.57 | 14.74 | 9.83 | 0.09 |
| **6** | GAZALBIDE | 0.539 | 0.18 | 105.71 | 21.897 | 88 | 63.69 | 48.85 | 14.84 | 0.12 |
| **7** | TXAGORRITXU | 0.545 | 0.5 | 157.02 | 51.632 | 111 | 41.73 | 27.84 | 13.89 | 0.09 |
| **8** | SAN MARTIN | 0.386 | 1.49 | 157.33 | 78.754 | 112 | 49.27 | 30.49 | 18.78 | 0.08 |
| **9** | ZARAMAGA | 0.657 | 0.36 | 180.18 | 66.343 | 147 | 45.33 | 35.32 | 10.01 | 0.12 |
| **10** | EL ANGLO | 0.698 | 0.39 | 243.12 | 17.231 | 34 | 11.59 | 10.82 | 0.77 | 0.055 |
| **11** | ARANTZABELA | 0.448 | 0.07 | 80.63 | 18.644 | 72 | 62.91 | 44.05 | 18.85 | 0.12 |
| **12** | SANTIAGO | 0.501 | 0.23 | 246.58 | 14.026 | 37 | 28.02 | 21.38 | 6.64 | 0.11 |
| **13** | ARANBIZKARRA | 0.539 | 0.46 | 256.67 | 43.431 | 109 | 42.93 | 37.17 | 5.76 | 0.13 |
| **14** | ARANA | 0.688 | 0.1 | 214.45 | 14.531 | 52 | 26.04 | 20.9 | 5.14 | 0.1 |
| **15** | DESAMPARADOS | 0.605 | 0.54 | 231.86 | 25.803 | 67 | 29.18 | 25.23 | 3.95 | 0.13 |
| **16** | JUDIMENDI | 0.804 | 0.31 | 241.21 | 23.547 | 42 | 23.94 | 15.1 | 8.84 | 0.09 |
| **17** | SANTA LUCIA | 0.505 | 0.44 | 233.75 | 33.216 | 97 | 40.38 | 30.45 | 9.93 | 0.12 |
| **18** | ADURTZA | 0.561 | 0.28 | 21.91 | 281.006 | 144 | 47.65 | 24.29 | 23.37 | 0.32 |
| **19** | SAN CRISTOBAL | 0.663 | 0.32 | 185.55 | 32.538 | 87 | 40.8 | 31.73 | 9.07 | 0.13 |
| **20** | MENDIZORROTZA | 0.397 | 0.67 | 12.85 | 356.586 | 153 | 56.52 | 33.4 | 23.13 | 0.27 |
| **21** | ARIZNABARRA | 0.471 | 0.59 | 163.22 | 48.028 | 74 | 46.15 | 35.78 | 10.37 | 0.11 |
| **22** | ALI-GOBEO | 0.58 | 0.04 | 51.39 | 18.705 | 40 | 35.48 | 2.59 | 32.9 | 0.09 |
| **23** | SANSOMENDI | 0.323 | 0.92 | 68.7 | 307.12 | 148 | 25.34 | 6.87 | 18.47 | 0.22 |
| **24** | ARRIAGA-LAKUA | 0.227 | 1.65 | 65.93 | 424.384 | 171 | 29.36 | 9.39 | 19.97 | 0.2 |
| **25** | ABETXUKO | 0.601 | 0.08 | 31.94 | 106.775 | 127 | 49.37 | 19.97 | 29.4 | 0.38 |
| **26** | ZABALGANA | 0.082 | 1.73 | 92.64 | 241.232 | 137 | 28.88 | 3.14 | 25.74 | 0.16 |
| **27** | SALBURUA | 0.083 | 1.12 | 66.08 | 237.728 | 140 | 43.34 | 6.88 | 36.45 | 0.21 |
| **28** | ARETXABALETA-GARDELEGI | 0 | 0.1 | 12.64 | 59.404 | 67 | 44.09 | 9.76 | 34.33 | 0.29 |

**Supplementary Table 2:** Magnitudes of dwellings according to time of construction by neighbourhood in percentage from 1900 to 2015. Source: City council of Vitoria-Gasteiz (2015). Municipal statistical indicators of land use and residential building.

| ID | STRATUM | Before 1900 | 1901-1950 | 1951-1960 | 1961-1970 | 1971-1980 | 1981-1990 | 1991-2000 | 2001-2010 | 2011-2015 |
|---|---|---|---|---|---|---|---|---|---|---|
| 1 | CASCO VIEJO | 35.45 | 12.27 | 7.24 | 11.84 | 8.27 | 4.97 | 13.04 | 6.39 | 0.51 |
| 2 | ENSANCHE | 14.7 | 11.93 | 8.45 | 14.62 | 23.28 | 12.97 | 7.41 | 3.98 | 2.66 |
| 3 | LOVAINA | 4.55 | 8.65 | 26.26 | 14.25 | 12.01 | 8.68 | 19.3 | 6.07 | 0.22 |
| 4 | CORONACION | 0 | 8.28 | 16.69 | 60.42 | 12.18 | 0.13 | 1.25 | 1.05 | 0 |
| 5 | EL PILAR | 0 | 0 | 0.55 | 21.51 | 69.53 | 8.41 | 0 | 0 | 0 |
| 6 | GAZALBIDE | 0 | 0 | 0 | 0 | 93.56 | 6.44 | 0 | 0 | 0 |
| 7 | TXAGORRITXU | 0 | 0 | 2.35 | 13.22 | 73.8 | 0.79 | 9.84 | 0 | 0 |
| 8 | SAN MARTIN | 0 | 0 | 0.35 | 1.43 | 4.5 | 81.57 | 6.25 | 5.91 | 0 |
| 9 | ZARAMAGA | 0 | 0.53 | 7.71 | 55.01 | 36.73 | 0 | 0 | 0.02 | 0 |
| 10 | EL ANGLO | 1.24 | 10.86 | 17.78 | 22.59 | 35.47 | 3.94 | 6.83 | 1.28 | 0 |
| 11 | ARANTZABELA | 0 | 0 | 0 | 0 | 58.03 | 28.87 | 0 | 13.11 | 0 |
| 12 | SANTIAGO | 0 | 1.96 | 0 | 17.43 | 20.91 | 59.7 | 0 | 0 | 0 |
| 13 | ARANBIZKARRA | 0 | 0.16 | 1.18 | 10.61 | 69.93 | 16.92 | 0.39 | 0.81 | 0 |
| 14 | ARANA | 0 | 0 | 0 | 93.6 | 6.4 | 0 | 0 | 0 | 0 |
| 15 | DESAMPARADOS | 0.43 | 4.3 | 4.66 | 42.5 | 20.84 | 21.27 | 1.31 | 2.95 | 1.74 |
| 16 | JUDIMENDI | 0 | 19.53 | 15.34 | 48.83 | 12.84 | 2.27 | 0.82 | 0 | 0.36 |
| 17 | SANTA LUCIA | 0 | 0 | 1.01 | 1.69 | 65.19 | 31.16 | 0.96 | 0 | 0 |
| 18 | ADURTZA | 0.03 | 2.99 | 14.83 | 31.71 | 13.82 | 18.77 | 7.45 | 4.91 | 5.49 |
| 19 | SAN CRISTOBAL | 0 | 8.24 | 17.58 | 19.36 | 41.18 | 9.16 | 3.75 | 0.73 | 0 |
| 20 | MENDIZORROTZA | 0.88 | 4.01 | 3.13 | 0.06 | 14.04 | 41.53 | 15.99 | 18.53 | 1.83 |
| 21 | ARIZNABARRA | 0 | 3.81 | 4.16 | 18.54 | 9.08 | 31.58 | 31.61 | 1.22 | 0 |
| 22 | ALI-GOBEO | 1.21 | 12.38 | 25.73 | 4.61 | 20.39 | 0.73 | 0.97 | 30.1 | 3.88 |
| 23 | SANSOMENDI | 0 | 0 | 0 | 0 | 28.26 | 7.47 | 47.74 | 16.53 | 0 |
| 24 | ARRIAGA-LAKUA | 0.04 | 0 | 0.04 | 0.23 | 5.96 | 17.14 | 27.62 | 48.66 | 0.32 |
| 25 | ABETXUKO | 0.21 | 0.07 | 27.04 | 44.71 | 3.58 | 0 | 6.51 | 17.6 | 0.29 |
| 26 | ZABALGANA | 0.05 | 0 | 0 | 0.01 | 0 | 0 | 0 | 74.03 | 25.92 |
| 27 | SALBURUA | 0.19 | 0.02 | 0 | 0 | 0 | 0 | 0 | 72.39 | 27.4 |
| 28 | ARETXABALETA-GARDELEGI | 0 | 0 | 0 | 0 | 0.24 | 0 | 0 | 0 | 99.76 |

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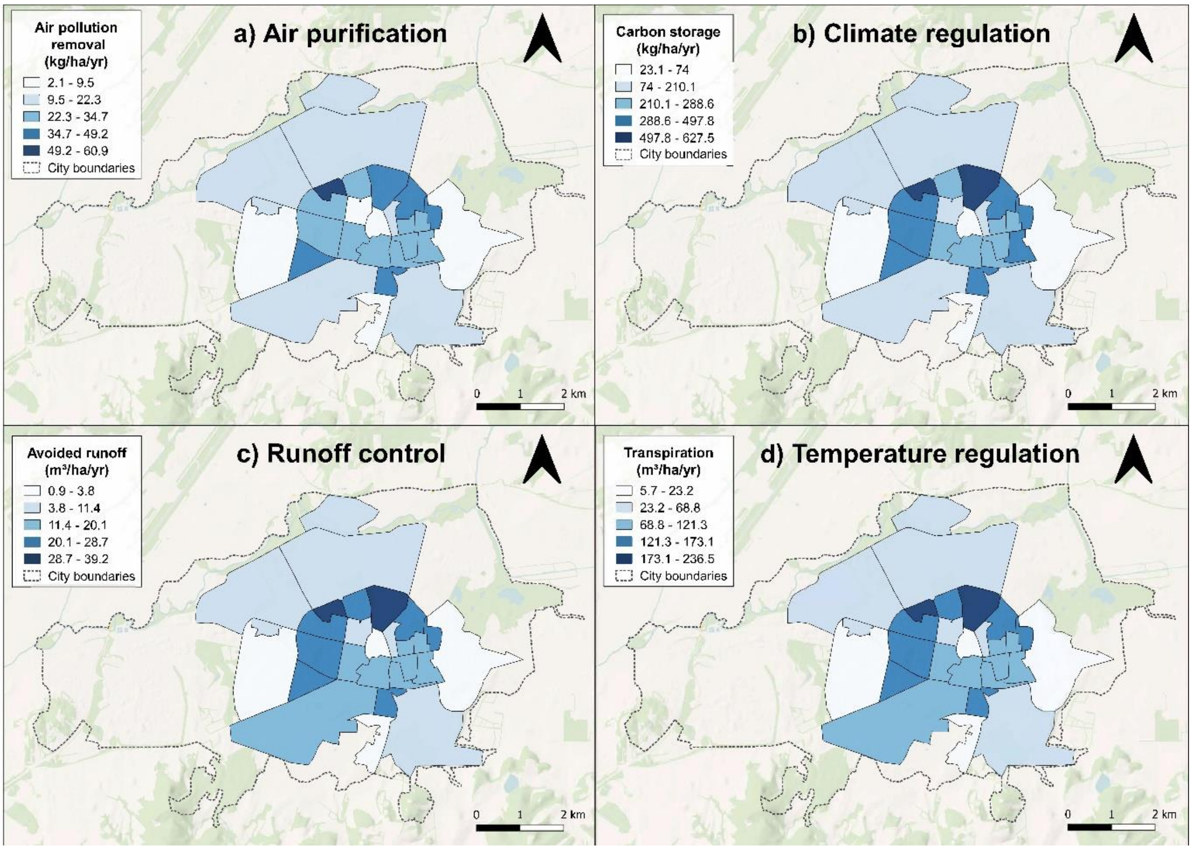
**Supplementary Table 3.** Ecosystem services values and ES index by neighbourhood (public tree inventory of the city of Vitoria-Gasteiz, the year 2015). Note: ES INDEX total value is the mean value for the city

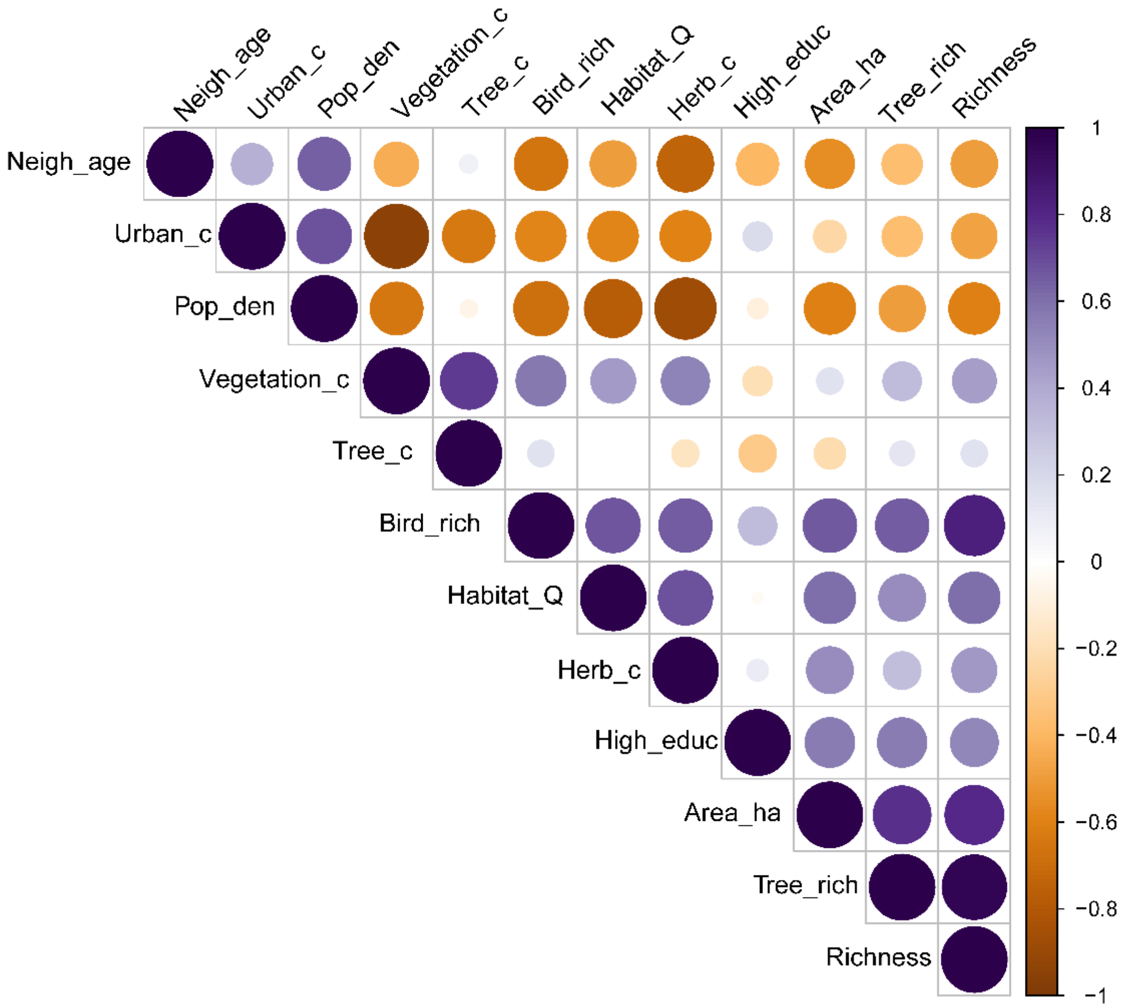
| ID | Stratum | Pollution Removal (ton/yr) | Pollution Removal (kg/ha/yr) | Avoided runoff (m3/yr) | Avoided runoff (m3/ha/yr) | Transpiration ($m^3$/yr) | Transpiration ($m^3$/ha/yr) | C Storage (ton/yr) | C Storage (kg/ha/yr) | ES INDEX |
|---|---|---|---|---|---|---|---|---|---|---|
| **1** | CASCO VIEJO | 0.05 | 6.04 | 109.72 | 3.84 | 662.72 | 23.20 | 2.33 | 74.01 | 7.56 |
| **2** | ENSANCHE | 0.38 | 30.61 | 822.87 | 18.03 | 4970.28 | 108.90 | 12.6 | 250.45 | 43.87 |
| **3** | LOVAINA | 0.35 | 31.50 | 768.79 | 17.44 | 4643.62 | 105.37 | 12.56 | 258.55 | 43.81 |
| **4** | CORONACION | 0.09 | 9.49 | 188.41 | 5.80 | 1138.02 | 35.01 | 6.06 | 169.10 | 15.51 |
| **5** | EL PILAR | 0.37 | 33.88 | 806.13 | 22.80 | 4869.17 | 137.74 | 10.68 | 274.08 | 52.48 |
| **6** | GAZALBIDE | 0.39 | 60.92 | 857.24 | 39.16 | 5177.91 | 236.54 | 15.14 | 627.45 | 100.00 |
| **7** | TXAGORRITXU | 0.53 | 31.80 | 1148.25 | 22.24 | 6935.68 | 134.33 | 21.49 | 377.60 | 55.14 |
| **8** | SAN MARTIN | 0.90 | 34.67 | 1970.83 | 25.03 | 11904.20 | 151.16 | 38.69 | 445.70 | 62.83 |
| **9** | ZARAMAGA | 1.01 | 42.67 | 2197.06 | 33.12 | 13270.68 | 200.04 | 41.03 | 561.08 | 81.58 |
| **10** | EL ANGLO | 0.09 | 18.43 | 196.17 | 11.39 | 1184.93 | 68.77 | 2.89 | 152.16 | 25.93 |
| **11** | ARANTZABELA | 0.24 | 49.16 | 534.24 | 28.66 | 3226.91 | 173.12 | 9.44 | 459.43 | 74.31 |
| **12** | SANTIAGO | 0.12 | 29.76 | 258.74 | 18.46 | 1562.87 | 111.47 | 4.46 | 288.59 | 45.64 |
| **13** | ARANBIZKARRA | 0.56 | 42.40 | 1212.91 | 27.93 | 7326.20 | 168.69 | 21.54 | 449.94 | 70.09 |
| **14** | ARANA | 0.10 | 28.10 | 214.05 | 14.73 | 1292.88 | 88.98 | 4.4 | 274.72 | 39.49 |
| **15** | DESAMPARADOS | 0.19 | 27.07 | 407.19 | 15.78 | 2459.48 | 95.33 | 7.11 | 250.00 | 39.41 |
| **16** | JUDIMENDI | 0.18 | 27.75 | 383.69 | 16.30 | 2317.55 | 98.45 | 7.14 | 275.16 | 41.41 |
| **17** | SANTA LUCIA | 0.31 | 31.96 | 666.63 | 20.07 | 4026.59 | 121.25 | 14.02 | 382.98 | 52.60 |
| **18** | ADURTZA | 1.05 | 14.75 | 2284.27 | 8.13 | 13797.44 | 49.10 | 42.1 | 135.92 | 19.43 |
| **19** | SAN CRISTOBAL | 0.39 | 47.97 | 850.86 | 26.16 | 5139.35 | 157.99 | 14.22 | 396.56 | 67.93 |
| **20** | MENDIZORROTZA | 2.12 | 22.31 | 4621.68 | 12.96 | 27915.90 | 78.29 | 82.58 | 210.09 | 32.04 |
| **21** | ARIZNABARRA | 0.56 | 46.66 | 1212.91 | 25.26 | 7326.20 | 152.57 | 26.35 | 497.80 | 70.38 |
| **22** | ALI-GOBEO | 0.07 | 12.61 | 149.19 | 7.98 | 901.16 | 48.19 | 3.72 | 180.47 | 20.17 |
| **23** | SANSOMENDI | 1.30 | 15.71 | 2837.89 | 9.24 | 17141.42 | 55.82 | 70.85 | 209.29 | 24.33 |
| **24** | ARRIAGA-LAKUA | 1.99 | 19.22 | 4349.47 | 10.25 | 26271.68 | 61.91 | 96.89 | 207.12 | 27.05 |
| **25** | ABETXUKO | 0.39 | 14.78 | 858.54 | 8.04 | 5185.78 | 48.57 | 17.46 | 148.35 | 19.85 |
| **26** | ZABALGANA | 0.16 | 3.05 | 341.32 | 1.41 | 2061.64 | 8.55 | 14.48 | 54.45 | 2.30 |
| **27** | SALBURUA | 0.16 | 2.90 | 346.97 | 1.46 | 2095.79 | 8.82 | 15.73 | 60.03 | 2.53 |
| **28** | ARETXABALETAGARDELEGI | 0.03 | 2.14 | 56.08 | 0.94 | 338.70 | 5.70 | 1.51 | 23.06 | 0.00 |
| | TOTAL | 14.08 | 738.31 | 30652.10 | 2733.85 | 185144.75 | 2733.85 | 7694.14 | 1137.66 | 40.63* |

**Supplementary Table 4**: Tree structural variables for i-Tree Eco processing

| Perimeter at breast height (cm) | Total height (m) | DBH (cm) |
|---|---|---|
| **<14** | <3-5 | 2.39 |
| **14-20** | 3-5 | 5.41 |
| **20-40** | 3-12 | 9.55 |
| **40-60** | 8-12 | 15.92 |
| **60-80** | 12-18 | 22.29 |
| **80-100** | 12-18 | 28.66 |
| **100-120** | 12-25 | 35.03 |
| **120-150** | 12-25 | 42.99 |
| **150-200** | 12-25 | 55.73 |
| **180-250** | 12-30 | 68.47 |
| **250-300** | 12-40 | 87.58 |
| **>300** | 12-40 | 100.32 |

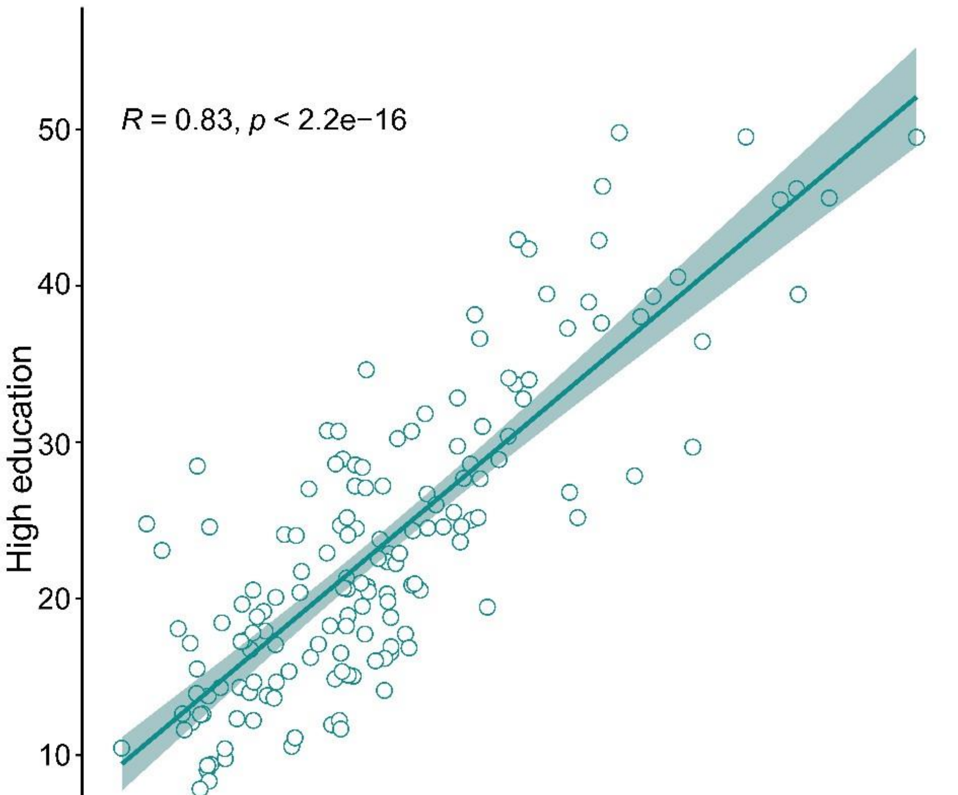



**Supplementary figure 3.** Spatial distribution of regulating ecosystem services iTree model outcomes provided by public trees at the neighbourhood scale: a) Air purification, b) Climate regulation, c) Runoff control and d) Temperature regulation. The values were classified using the natural breaks method. Background from © MapTiler (www.maptiler.com) and ©OpenStreetMap (CC BY-SA 2.0).




**Supplementary figure 4**. Spearman's rank correlation matrix plot showing the relationship between a) Neighbourhood development age; b) Urban cover; (c) Population density, d) Total vegetation cover per neighbourhood (%); e) Tree cover (%); f) Bird richness;
g) Habitat quality (i.e. mean value of UGS per neighbourhood); h) Herbaceous cover (%); i) High education attainment (%); j) Neighbourhood area (ha); k) Tree richness; l) Total richness (i.e. bird and tree richness combined). Correlations from negative to positive are indicated in orange to purple respectively. The strength of the correlation is indicated by dot size and colour saturation. Note that only significant correlations are shown ($p < 0.05$).

Based on data from the Vitoria-Gasteiz municipality for 2020 (uploaded to the Basque Statistics Office webpage the 24th October 2022) we performed a Pearson's correlation analysis between tertiary education (high education attainment) and income level. Both variables are strongly correlated, but data on income were not available for the period corresponding to the other socio- environmental variables used in analysis, and even these data for 2020 were only available at the end of 2022 with a different aggregation than the available for other variables used.




**Supplementary figure 5**. Pearson correlation plot depicting the linear relationship between high education attainment and income. R and significance value ($p < 0.05$) for the linear model, where the shaded area indicates the confidence interval of 95%

# Supplementary Material Chapter IV

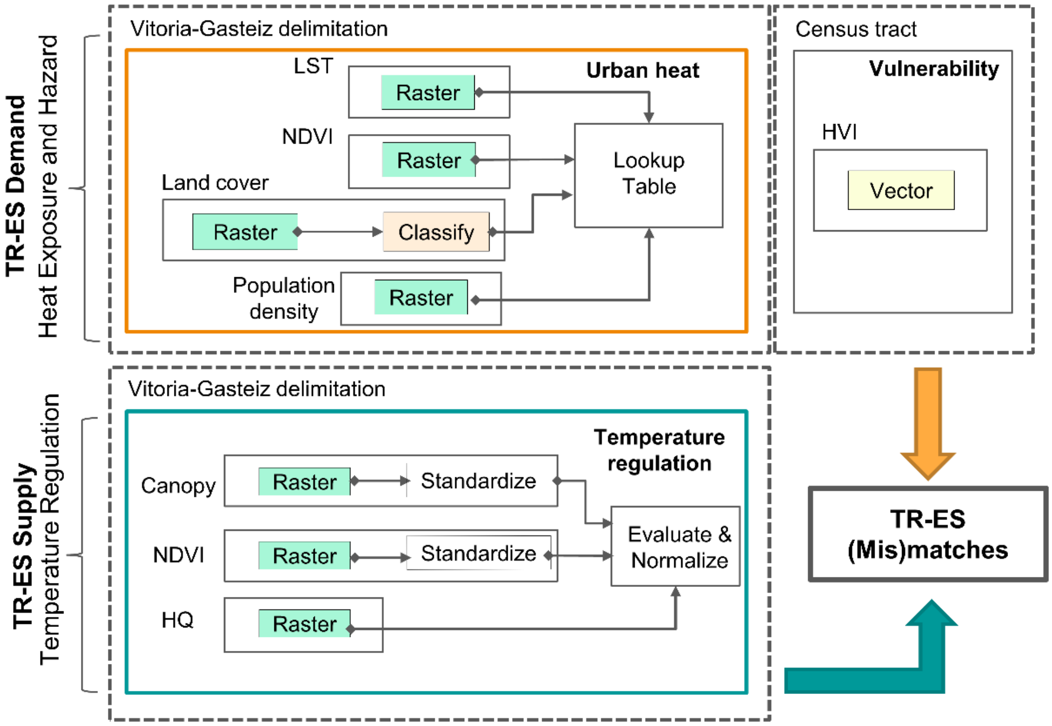



**Figure S1.** Dataflow of TR-ES Supply and Demand model created on ARIES platform. Note: The look-up table refers to the four spatial layers i) LST; ii) NDVI; iii) Land cover; iv) Population density in connection to Table S2.

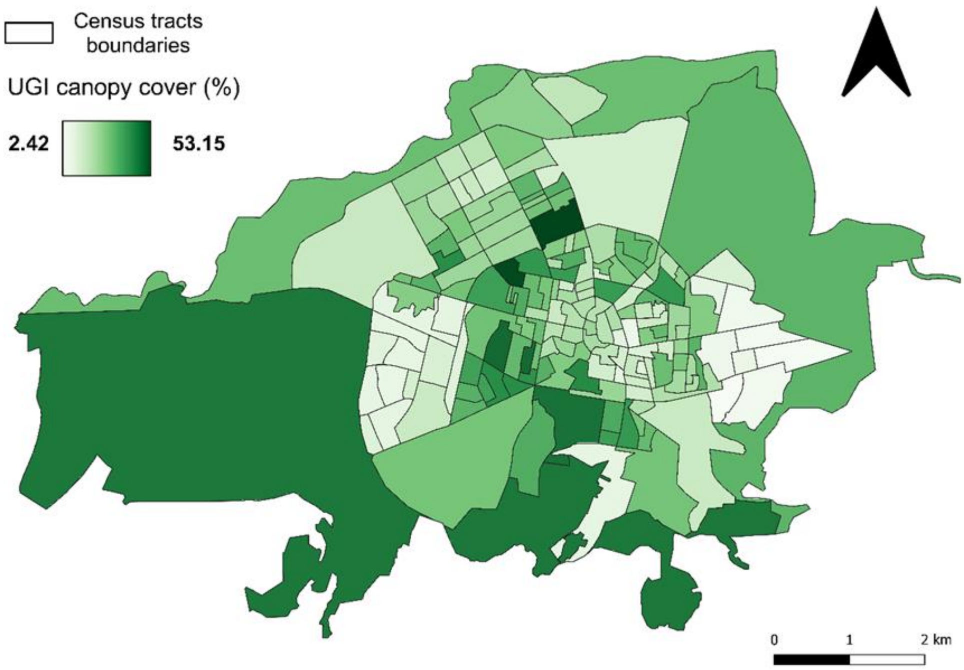



**Figure S2.** Spatial patterns of overall UGI canopy cover at the census tract level. Indicator values classified using pretty breaks method. The indicator is spatially auto correlated (z-score > 3). Source: own elaboration based on LiDAR data.

**Table S1.** Data sources and description of final the different considered variables per analysis unit (census tract level).

| Data source | Variable definition | Vulnerability component | Mean (SD) | Minimum Maximum | Median |
|---|---|---|---|---|---|
| Estadística de padrón continuo (2021) | % population that was born in America | Adaptive Capacity | 7.58 – 3.90 | 1.88 – 22.21 | 6.44 |
| | % population that was born in Africa | | 4.695 – 4.33 | 0.22 – 29.74 | 3.4 |
| | % population that was born in Asia | | 1.12 – 0.95 | 0 – 5.93 | 0.8 |
| | % low education attainment | | 37.80 – 9.24 | 9.24 – 60.12 | 37.3 |
| | % high education attainment | | 23.43 – 9.87 | 4.03 – 48.81 | 21.21 |
| | % population over 65 years | Sensitivity | 23.55 – 12.82 | 1.63 – 47.26 | 26.14 |
| | % population below 14 years | | 13.37 – 6.40 | 4.31 – 33.06 | 10.81 |
| INE (2020) | Median household income | Adaptive Capacity | 36102 - 9489 | 22278 - 89731 | 34473 |
| Atlas Mortalidad Pais Vasco (2012-2017) | SMR* Women – Diabetes | Sensitivity | 0.38 – 0.07 | 0.24 – 0.63 | 0.38 |
| | SMR Women – Lung Cancer | | 0.39 – 0.09 | 0.22 – 0.70 | 0.37 |
| | SMR Women – Respiratory disease | | 0.42 – 0.11 | 0.13 – 0.76 | 0.42 |
| | SMR Men – Diabetes | | 0.33 – 0.06 | 0.20 – 0.55 | 0.32 |
| | SMR Men– Lung Cancer | | 0.46 – 0.15 | 0.18 – 0.93 | 0.44 |
| | SMR Men – Respiratory disease | | 0.12 – 0.43 | 0.15 – 0.89 | 0.42 |
| Own elaboration | % vegetation cover | Exposure | 15.01 – 17.55 | 0 – 83.19 | 8.66 |
| | LST (ºC)** | | 37.78 – 3.75 | 24.23 – 55.80 | 37.54 |
| Centro de Estudios Ambientales (2020) | Land use land cover | | N/A | N/A | N/A |
| Centro de Estudios Ambientales (2016) | Population density*** | | 209.23 – 105.36 | 10.98 – 446.41 | 200.95 |
| Own elaboration | Habitat quality | N/A | 0.09 – 0.06 | 0.03 – 0.45 | 0.07 |

Notes:
*% population that was born in America includes the whole continent.
** For health assessment, we employed the Standardized Mortality Ratio (SMR), which assesses the likelihood of higher-than-expected mortality rates within a specific group compared to a reference population (SMR > 1). For example, an SMR value greater than 0.5 (or 50%) indicates an elevated probability of the SMR exceeding 1. This implies a heightened risk of mortality associated with the disease in the context of urban heat.
***Mean and Median were calculated based on the average LST values per census tract.
****Descriptive statistics were calculated based on the average population density values per hectare at the census tract level.

**Table S2.** Cross-tabulation matrix for assessing the exposure dimension of the demand for temperature regulation ES at the census tract level. Population density and temperature break values were extracted and adapted from (Herreros-Cantis et al. 2021; Di Napoli et al., 2020; Baró et al., 2016).

| Population density (inhabitants/ha) | Temperature (ºC) | | | | | |
|---|---|---|---|---|---|---|
| | <20 | 20-25 | 25-30 | 30-35 | 35-40 | >=40 |
| **<=5** | 0 | 0 | 0 | 0 | 0 | 0 |
| **5-50** | 0 | 0,2 | 0,2 | 0,4 | 0,4 | 0,6 |
| **50-100** | 0 | 0,2 | 0,4 | 0,4 | 0,6 | 0,8 |
| **100-200** | 0 | 0,4 | 0,4 | 0,6 | 0,6 | 0,8 |
| **200-400** | 0 | 0,4 | 0,6 | 0,6 | 0,8 | 0,8 |
| **>400** | 0 | 0,6 | 0,8 | 0,8 | 0,8 | 1 |

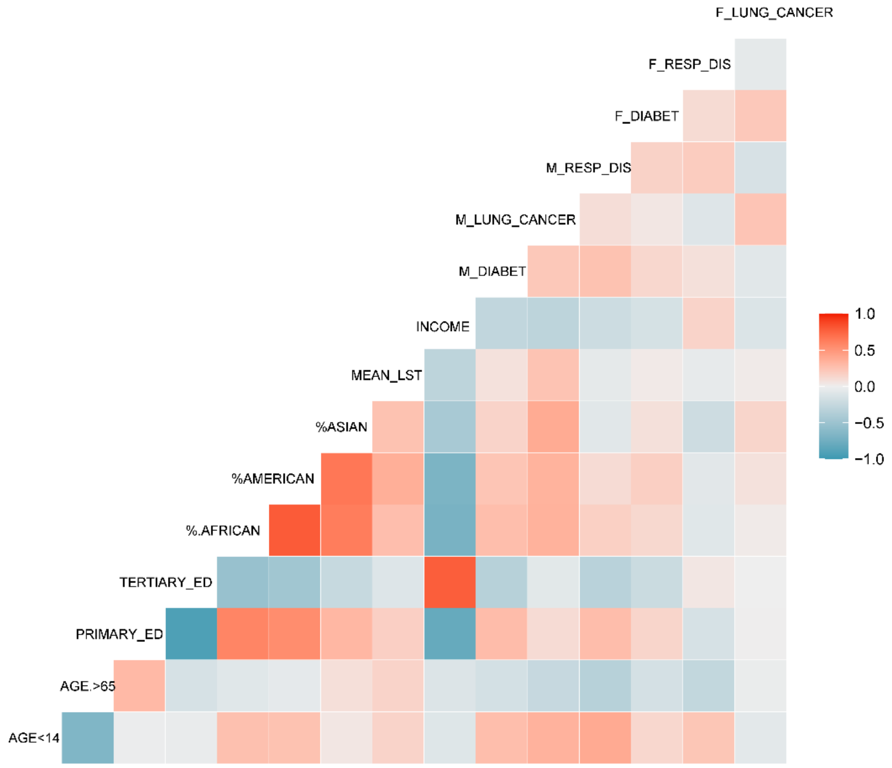



**Figure S3**. Spearman's pairwise correlation rank matrix of heat-related vulnerability variables. The plot illustrates the relationships between a) Age < 14; b) Age > 65; c) Low education attainment (up to primary education %), d) High education attainment (tertiary education %); e) Population born in African countries (%); f) Population born in American countries (%); g) Population born in Asian countries (%); h) Mean LST (ºC); i) Income (EUR); j) SMR Diabetes men; k) SMR Lung cancer men; l) SMR respiratory diseases men; m) SMR Diabetes women; n) SMR respiratory diseases women; o) SMR Lung cancer women. Correlations range from negative (depicted in blue) to positive (shown in red), with colour saturation indicating the strength of the correlation. Only significant correlations ($p < 0.05$) are displayed.

**Table S3.** HVI component scoring mechanism based on variables z-score values. Drawing on Reid et al. (2009).

| Z-score range | HVI component score |
|---|---|
| -2 or lower | 1 |
| -2 to -1 | 2 |
| -1 to 0 | 3 |
| 0 to 1 | 4 |
| 1 to 2 | 5 |
| 2 or higher | 6 |

### Spatial distribution of urban heat hazard:

To measure urban heat hazard, we adapted an open-source code by McCartney et al. (2020) in Google Earth Engine (GEE) to map the mean Land Surface Temperature (LST) at daytime for each pixel in the entire municipality of Vitoria-Gasteiz. LST is commonly used as a proxy for UHI and is positively correlated with the density of impervious surfaces and negatively associated with green infrastructure (Rodríguez-Gómez et al., 2022). The LST values were derived from surface temperature time series using Landsat 8 and 9 Level-2 imageries, considering cloud cover (<20%) from summer season (19th June to 19th September) in 2021-2022. Yet, it should be noted that LST alone may not encompass all the significant factors contributing to heat exposure, including population density and vegetation cover.


**Figure S4.** Land surface temperature (LST) in Vitoria-Gasteiz, highlighting localized heat-intensive zones indicated by industrial and dense urban structures surrounded by heat-absorbing materials in aerial photographs (a) and (b). he LST variations on the map are depicted from 24.94ºC (in blue shades) to a maximum of 55.8ºC (in red shades), representing the temperature hazard across the city.

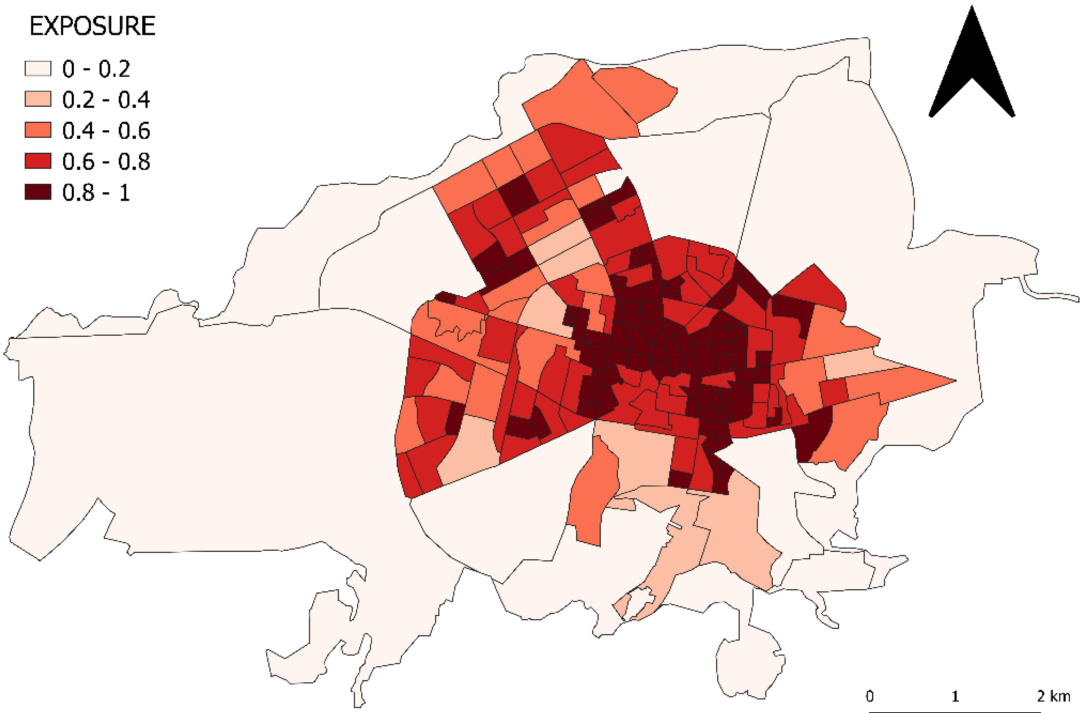



**Figure S5.** Urban heat Exposure map integrating population density, vegetation cover, and LST to assess areas most susceptible to heat hazard (i.e. as depicted in Figure S3) in Vitoria-Gasteiz. The darkest shades of red highlight regions with elevated heat exposure.

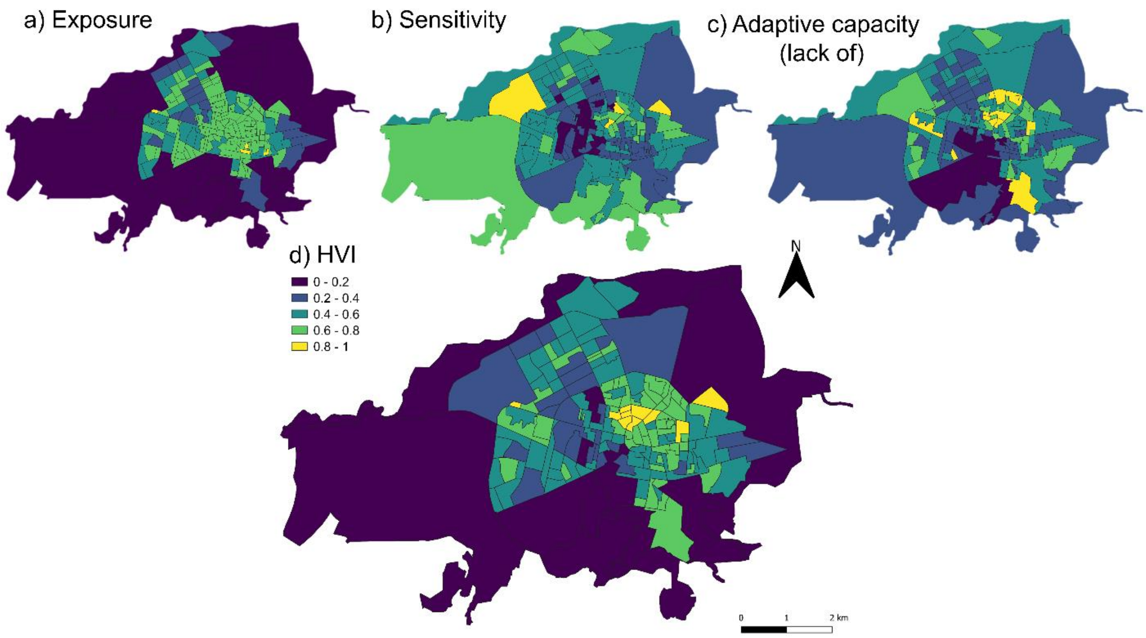



**Figure S6.** Heat Vulnerability Index and its sub-components: exposure, sensitivity, and lack of adaptive capacity. This map illustrates the HVI scores derived from Table S2 using z-scores from composing variables, subsequently normalized on a scale of 0 to 1. The lighter shades of yellow and green highlight areas with the highest cumulative vulnerability components.

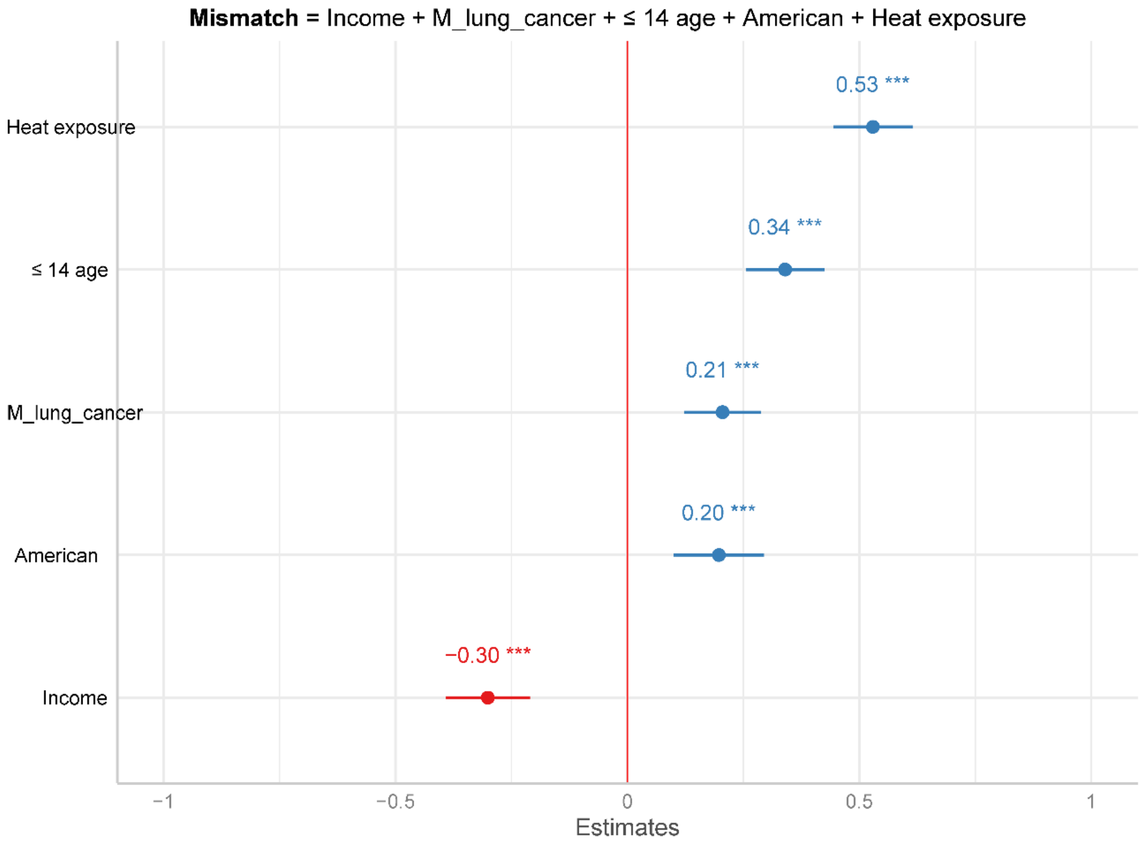



**Fig. S7.** GLM model outputs for the effects of heat exposure, population below the age of 14, SMR of lung cancer in men, population born in America, household income on model-estimated ES supply-demand mismatch values in Vitoria-Gasteiz. Standardized Beta coefficients and confidence intervals for the variables are presented to show their effect size. Standardized beta coefficients and confidence intervals indicate effect sizes. Significant effects are non-overlapping confidence intervals with zero.

**Table S4.** Descriptive mean values of HOT and COLD spots for significant variables driving clustering patterns.

| | | Exposure | Income | Tertiary | F_resp_di | Primary | M_resp_di | F_lung_c | American |
|---|---|---|---|---|---|---|---|---|---|
| HOT | Average | 0.74 | 32663 | 21.32 | 0.38 | 40.68 | 0.41 | 0.44 | 9.22 |
| | Median | 0.79 | 32901 | 19.8 | 0.39 | 41.08 | 0.41 | 0.45 | 8.36 |
| | SD | 0.1 | 5931.5 | 8.4 | 0.1 | 8.4 | 0.08 | 0.09 | 4.4 |
| COLD | Average | 0.56 | 44918 | 30.59 | 0.46 | 30.11 | 0.4 | 0.46 | 4.74 |
| | Median | 0.64 | 39930 | 30.71 | 0.44 | 29.91 | 0.4 | 0.44 | 4.7 |
| | SD | 0.24 | 14373.8 | 10.24 | 0.11 | 7.55 | 0.12 | 0.11 | 1.98 |

As urban areas expand, societal contact with nature is increasingly confined to cityscapes, highlighting the need to explore the interactions between urban biodiversity, urban green infrastructure (UGI) and human well-being. The spatial distribution, accessibility, and quality of urban ecosystems are important aspects to consider when assessing these interactions. Recognising the critical role of biodiversity in UGI ecosystem services (ES) provision is key, requiring an examination of UGI spatial patterns and ecological characteristics. This dissertation investigates the contribution of urban biodiversity to the multifunctionality of UGI from an environmental justice perspective. Comprising three empirical studies, each building upon the previous one, this research focuses on Vitoria-Gasteiz, the 'European Green Capital' of the Basque Country. These studies collectively offer valuable insights into the influence of socioeconomic and health factors on urban ecosystem quality and distribution, their cascading impact on human well-being, and the inequitable access to ES. The findings contribute to a better understanding of the complex dynamics within urban socio-ecological systems, providing actionable insights for advancing urban agendas toward equitable urban development in both Vitoria-Gasteiz and beyond.

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