The impacts of urban nature-based solutions: An integrated spatial vulnerability perspective

David Camacho-Caballero

Ph.D. dissertation

Directors: Prof. Dr. Johannes Langemeyer, Prof. Dr. Gara Villalba, Dr. David N. Barton. Tutor: Prof. Dr. Gara Villalba

Ph.D. Program in Environmental Science and Technology

Institut de Ciència i Tecnologia Ambientals (ICTA)

Universitat Autònoma de Barcelona (UAB)

June 2024

UAB Universitat Autònoma de Barcelona

ICTA Institut de Ciència i Tecnologia Ambientals - UAB

URBAG Integrated System Analysis of Urban Vegetation and Agriculture

EXCELENCIA MARÍA DE MAEZTU

TABLE OF CONTENTS

ACKNOWLEDGMENTS 1

SUMMARY 3

LIST OF FIGURES 5

LIST OF TABLES 7

CHAPTER I. INTRODUCTION 8

I.1. Background and research gap 8

I.2. Research objectives and research design 12

I.3. Nature-based solutions in urban environments: an overview 17

I.3.1. Nature-based solutions: conceptualization and understanding of its impacts 17

I.3.2. Current approaches for evaluating the impacts of Nature-based solutions 18

I.3.3. Case studies 29

I. References 30

CHAPTER II. EXPLORING THE CURRENT UNDERSTANDING OF NATURE-BASED SOLUTIONS IMPACTS IN COMPLEX URBAN ENVIRONMENTS 38

II. Abstract 38

II.1. Introduction 39

II.2. Sustainability, resilience and equity as urban challenges and their relation to Nature-based solutions 41

II.2.1. Sustainability 41

II.2.2. Resilience 43

II.2.3. Equity 44

II.2.4. Urban challenges: core interactions 45

II.3. Vulnerability assessments: an opportunity for better evaluating Nature-based solutions in urban environments 49

II.4. Conclusions 52

II. References 53

CHAPTER III. ASSESSING NATURE-BASED SOLUTIONS IN THE FACE OF URBAN VULNERABILITIES: A MULTI-CRITERIA DECISION APPROACH 65

III. Abstract 65

III.1. Introduction 67

III.2. Nature-based solutions vulnerability framework 70

III.2.1. Conceptual approach 70

III.2.2. Stepwise approach of the Nature-based solutions vulnerability framework 71

III.3. Case study: Urban Agriculture in the Metropolitan Area of Barcelona 73

III.4. Methodology 74

III.4.1. Development of scenarios 74

III.4.2. Selection of vulnerabilities and mapping of indicators 75
III.4.3. Normalization of indicators 79
III.4.4. Aggregation of indicators for single vulnerabilities 80
III.4.5. Stakeholder weighting 80
III.4.6. Aggregation of single vulnerabilities for a combined vulnerability 80
III.5. Results 81
III.5.1. Combined vulnerability 81
III.5.2. Vulnerability of lack of local food 83
III.5.3. Vulnerability to heat 85
III.5.4. Vulnerability of lacking recreational space 87
III.5.5. Vulnerability of loss of biodiversity 89
III.6. Discussion 91
III.6.1. Land-use changes in the Metropolitan Area of Barcelona shift vulnerabilities unevenly 91
III.6.2. Advancing Nature-based solutions planning through an integrated vulnerability assessment 92
III.6.3. Considerations for the future application of the Nature-based solutions vulnerability framework 94
III.7. Conclusions 96
III. References 97
CHAPTER IV. UNDERSTANDING NATURE-BASED SOLUTIONS IMPACTS WITHIN AND BEYOND URBAN LIMITS: AN INTEGRATED VULNERABILITY ASSESSMENT 104
IV. Abstract 104
IV.1. Introduction 106
IV.2. Nature-based solutions vulnerability framework 108
IV.2.1. Conceptual considerations 108
IV.2.2. Stepwise approach 110
IV.3. Case study: green roofs in Oslo municipality 111
IV.4. Methodology 115
IV.4.1. Development of scenarios 115
IV.4.2. Selection of local-scale vulnerabilities and mapping of indicators 116
IV.4.3. Selection of broad-scale vulnerabilities and calculation of indicators 117
IV.4.4. Normalization of local-scale indicators 117
IV.4.5. Normalization of broad-scale indicators 117
IV.4.6. Aggregation of indicators for single local-scale vulnerabilities 118
IV.4.7. Aggregation of indicators for single broad-scale vulnerabilities 118
IV.4.8. Stakeholder weighting 118
IV.4.9. Development of a most favorable scenario 118
IV.5. Results 119

IV.5.1. Vulnerability to lack of habitats for pollinators 121
IV.5.2. Vulnerability to heavy rainfall events 121
IV.5.3. Vulnerability to heat 123
IV.5.4. Vulnerability to air pollution 124
IV.5.5. Vulnerability to lack of opportunities for interacting with natural environments 125
IV.5.6. Broad-scale vulnerabilities 126
IV.5.7. Most favorable scenario based on stakeholder and equal weighting schemes 127
IV.6. Discussion 129
IV.6.1. Cross-scale understanding of green roofs impacts reveal both synergies and tradeoffs 129
IV.6.2. Integrating cross-scale impacts of Nature-based solutions can support urban planning 130
IV.6.3. An improved methodological approach with considerations for future applications 132
IV.7. Conclusion 134
IV. References 134
CHAPTER V. SCIENTIFIC CONTRIBUTIONS AND FUTURE RESEARCH 142
V.1. Nature-based solutions in the face of sustainability, resilience and equity 142
V.2. A vulnerability approach for assessing the impacts of Nature-based solutions 144
V.3. Nature-based solutions impact shift vulnerabilities across spatial scales, creating both synergies and tradeoffs 146
V.4. Recommendations for policy and practice 147
V.5. Limitations and caveats 149
V.6. Recommendations for future research 150
V.7. Final thoughts 151
V. References 152
APPENDICES 155
Appendix 1. Supplementary data for Chapter III 156
1-A. Supplementary scheme 156
1-B. Methodological details 159
1-C. Supplementary maps 179
1-D. Supplementary table 215
Appendix 2. Supplementary data for Chapter IV 216
2-A. Methodological details 216
2-B. LCA impacts and inventory 258
2-C. Supplementary maps 266

ACKNOWLEDGMENTS

I would like to thank my supervisors, Johannes Langemeyer, Gara Villalba and David N. Barton, for their scientific support and guidance throughout my research. Their expertise and constructive feedback have been essential in developing this thesis. I appreciate their patience and the time they invested in helping me refine my work. Thank you for your support and for providing valuable insights during this journey.

I would also like to thank the URBAG team for their scientific collaboration. The integrative approach of my work builds upon several of their individual findings, which have been crucial in applying the vulnerability framework developed in this dissertation.

I am also grateful for the opportunity to conduct a research stay at NINA Oslo and for the warm reception I received from everyone there. Special thanks to David, Zander, Zofie, Helene and Anders for their support and hospitality. Thank you for providing such a welcoming and collaborative environment.

Additionally, I would like to express my gratitude to the municipal entities that were open to collaborating on the research case studies. Specifically, I extend my thanks to the PDU from the Metropolitan Area of Barcelona for their valuable participation and provision of data. I am also very grateful to the Oslo Municipality for their active involvement in the stakeholder processes and for offering valuable insights into the development of green roofs within the city.

In the same vein, I extend my appreciation to all the stakeholders who participated in the participatory and co-creative workshops essential for integrating the various facets of urban NBS. Your invaluable input was instrumental in making this work possible.

A great deal of gratitude goes out to the ICTA community for fostering a stimulating environment where I had the privilege to interact with not only brilliant minds but also compassionate individuals.

Thank you, Nora, so much, for being my main support throughout most of this PhD journey. Your patience, insights, understanding, love and kos were indispensable for arriving at this point. I extend my profound appreciation to Hege and Atle for your interest and hospitality during all these years. I am indebted to you all.

Finally, I would like to express my gratitude to all my friends and family who have been essential throughout this entire process, particularly in recent months. To my friends, your kindness, companionship, and support have meant the world to me. Thank you Svea, Laura, Sarah, Judith, Elsa, Jose, Violena, Joaquín, Coll, Michael, Karen and Josh. A mi familia, quienes incluso desde Caracas y Virginia, me cuidan, me animan y me hacen sentir el amor más profundo. Gracias miles Mamá, Papá y Tía Noe.

My work during this dissertation was made possible through funding from the Spanish Ministry of Science, Innovation and University FPI grant PRE2020-095791 and the URBAG research group (ERC Consolidator Grant 818002-URBAG).

Summary

This thesis tackles the pressing need to enhance our comprehension of how urban nature-based solutions (NBS) can have a wide array of desired and undesired impacts. Despite NBS being promoted in urban agendas as versatile tools to address urban and global challenges, the assessment of their impacts remains understudied. This deficiency results in an incomplete understanding of the potential synergies and trade-offs that NBS could be producing both within and beyond urban spaces. For this matter, this thesis critically examines the current state of assessments of urban NBS impacts and proposes a path forward for their improvement.

I first portray the scope of NBS impacts within complex urban environments, showing their connections to sustainability, resilience, and equity challenges. I illustrate how the interactions between these challenges can lead to both desired and undesired outcomes, highlighting the complexity of anticipating NBS impacts. Furthermore, I review existing evaluation approaches and identify the key limitations among them. Based on these, I argue for adopting a vulnerability-focused approach to enhance NBS impact evaluation.

Next, I introduce a framework for assessing NBS impacts that go beyond traditional approaches centered on environmental impacts and ecosystem services. This framework evaluates the extent to which NBS alter local-scale vulnerabilities (within urban areas), employing spatial indicators (exposure/sensitivity) and multi-criteria decision analysis to integrate them. To demonstrate its effectiveness, I apply this framework to the case study of the Metropolitan Area of Barcelona, assessing the impacts of increasing (peri-)urban agriculture on critical vulnerabilities. The results reveal diverse spatial outcomes and trade-offs in urban vulnerabilities, influenced by both the quantity and location of (peri-)urban agriculture

Building upon this, I extend the vulnerability framework to assess impacts across different spatial scales. This involves evaluating NBS impacts on local-scale vulnerabilities and on broad-scale vulnerabilities. The latter are assessed by considering their effects on planetary boundaries. I apply this extended framework to the case study of extensive green roofs in Oslo. This approach offers a novel and integrated understanding of NBS cross-scale trade-offs and synergies and allows to produce spatially explicit outcomes depicting optimal NBS configurations where the desired NBS impacts are maximized while undesired minimized.

This research contributes to enhancing NBS planning dynamics by offering both theoretical and practical insights into how urban NBS can simultaneously produce desired and undesired effects within and beyond urban environments. Through linking NBS impacts to vulnerabilities, the tested NBS-vulnerability framework presents a versatile and replicable methodology for further assessing urban NBS. This novel approach holds value for urban policy and planning as it enables an integrated, cross-scale, and site-specific assessment of NBS aligned with urban agendas, thereby reducing uncertainties and bridging the gap between short-term and long-term impacts.

Keywords: Nature-based solutions; Vulnerability assessment; Urban vulnerability; Urban planning; Planetary boundaries; Sustainability; Resilience; Equity

LIST OF FIGURES

Chapter I

1. Graphical abstract of Ph.D. dissertation, based on specific objectives and chapter distribution.
2. Research gap, question, objectives and chapter distribution of Ph.D. dissertation.
3. Classification of nature-based solutions. Adapted from Eggermont et al. (2015).
4. Visual representation of the capabilities of current approaches for evaluating the impacts of nature-based solutions.
5. Land use maps and some statistics of the Municipality of Oslo (and surroundings) and the Metropolitan Area of Barcelona.

Chapter II

1. Examples of nature-based solutions characteristics supporting urban challenges.
2. Shift in urban vulnerabilities before and after the implementation of nature-based solutions. Figure adapted from Dumitru & Lourido (2022).

Chapter III

1. Graphical representation of the current approach for assessing the impacts of nature-based solutions (NBS) in urban environments versus the NBS-vulnerability framework for assessing local-scale vulnerabilities.
2. Stepwise approach of the Nature-based solutions vulnerability framework, along with the objective of each step and its expected outcomes.
3. Current land uses and proposed development scenarios for the Metropolitan Area of Barcelona with percentages of land used by agricultural lands, other green spaces and built-up areas.
4. Spatial distribution of the Combined vulnerability and changes across scenarios with stakeholder's weights.
5. Spatial distribution of Vulnerability of lack of local food and changes across scenarios.
6. Spatial distribution of Vulnerability to heat and changes across scenarios.
7. Spatial distribution of changes between Scenario 3 and Scenario 0 in normalized indicators Heatwave day temperatures and Heatwave night temperatures, and aggregated exposure of Vulnerability to heat.
8. Spatial distribution of Vulnerability of lacking recreational space and changes across scenarios.
9. Spatial distribution of Vulnerability of loss of biodiversity and changes across scenarios.

Chapter IV

1. Graphical representation of the nature-based solutions (NBS) vulnerability framework for assessing local and broad-scale vulnerabilities.
2. Stepwise approach of the Nature-based solutions vulnerability framework, along with steps descriptions and outcomes.
3. Proposed scenarios for estimating green roofs impacts on local and broad-scale vulnerabilities, depicting the number of green roofs, their total extension, average size and percentage occupation out of the total potential green roofs.
4. Spatial distribution of Vulnerability to lack of habitats for pollinators and changes across scenarios.

5. Spatial distribution of Vulnerability to heavy rainfall events and changes across scenarios.
6. Spatial distribution of Vulnerability to heat and changes across scenarios.
7. Spatial distribution of Vulnerability to air pollution and changes across scenarios.
8. Spatial distribution of Vulnerability to lack of opportunities for interacting with natural environments and changes across scenarios.
9. Priority areas for green roof implementation based on a most favorable scenario where desired impacts are maximized and undesired minimized. Two weighting schemes are considered, along with their respective green roof extensions and percentage occupation out of the total potential green roofs.

LIST OF TABLES

Chapter I

1. List of frameworks for the evaluation of Nature-based solutions with its main characteristics.

Chapter III

1. Vulnerabilities, indicators, average/sum of absolute indicator values before normalization, thresholds and weights from the assessment of urban agriculture in the Metropolitan Area of Barcelona.
2. Percentage change (compared to scenario S0) of single vulnerabilities and combined vulnerability under both weighting schemes.

Chapter IV

1. Vulnerabilities, indicators, average/sum of absolute exposure and sensitivity indicator values before normalization, thresholds and weights from the assessment of green roofs in Oslo.
2. Single vulnerability values for each scenario and percentage change (compared to scenario S0).

CHAPTER I. INTRODUCTION

I.1. Background and research gap

Currently, over 50% of the world's population resides in cities, and projections suggest that by 2030—coinciding with the target year for achieving the UN Sustainable Development Goals—this figure will increase to 60% (United Nations, 2020). Cities serve as hubs for crucial services like healthcare and commerce, driving 75% of the global GDP, and despite occupying a small fraction of the Earth's surface, urban environments are responsible for the majority of the global environmental impacts (e.g., energy demand and carbon emissions) (Elmqvist et al., 2019).

Consequently, urban environments are confronted with significant challenges. Cities worldwide are increasingly vulnerable to environmental hazards such as droughts, floods, and heatwaves due to the escalating impact of climate change (Filho et al., 2019). Pollution and other disturbances, like noise, generated within cities directly and sometimes dramatically affect the health of urban populations (Nieuwenhuijsen et al., 2022). Furthermore, social inequality and aging infrastructure are also emerging as pressing issues requiring solutions within urban spaces (Kloss, 2022; Wei & Ewing, 2018).

Within this context of significant challenges faced by urban environments, cities are increasingly recognized as the focal point for studying and implementing actions to address societal and environmental concerns. Global initiatives, such as the 11th Sustainable Development United Nations goal (United Nations, 2015) and the Global Covenant of Mayors for Climate and Energy (Global Covenant of Mayors for Climate & Energy, 2023), aim to make cities inclusive, safe, resilient, and sustainable. These efforts coincide with the emergence of concepts like 'green cities' and 'eco-cities' in urban planning agendas, which advocate for sustainable management and expansion of urban green spaces to mitigate local and global challenges. Within this narrative, Nature-based solutions (NBS) have emerged as a comprehensive umbrella term covering various ecosystem-based approaches.

NBS are understood as "actions to protect, conserve, restore, sustainably use and manage natural or modified terrestrial, freshwater, coastal and marine ecosystems which address social, economic and environmental challenges effectively and adaptively, while simultaneously providing human well-being, ecosystem services, resilience and biodiversity benefits" (United Nations, 2022). The concept of NBS has gained momentum through global promotion by inter-governmental bodies such as the World Bank (MacKinnon et al., 2008), the International Union for Conservation of Nature (IUCN, 2009), and notably the European Union and the European Commission (Bauduceau et al., 2015). The European

Commission aims to establish Europe as a leader in NBS research and innovation, as well as a global hub for NBS exchange, collaboration, and promotion (Zwierzchowska et al., 2019). For this matter, efforts have been made to promote and disseminate relevant knowledge and best practices within Europe, evident in various EU documents, funding initiatives, and notably within projects under Horizon 2020 (EU-funded Research and Innovation) focusing on diverse NBS interventions.

The support for these initiatives is based on the assumption that NBS provide benefits such as improving the connection between nature and human society (Hanson et al., 2020; Randrup et al., 2020), developing resilient urban environments (Cohen-Shacham et al., 2016; Faivre et al., 2017; Kabisch et al., 2016), and enhancing the quality of life, promoting health, and fostering wellbeing (Dick et al., 2019; Panno et al., 2017).

While NBS have gained popularity for their social and ecological benefits, it is also acknowledged that NBS interventions in urban areas can have unintended consequences. For instance, large greening projects may require excessive and unsustainable water usage (Pereira et al., 2023), vegetation pollen can heighten allergy risks (McInnes et al., 2017), and greenhouse gases may be emitted during construction and maintenance in urban and peri-urban environments (e.g., Giama et al., 2021; Lulovicova & Bouissou, 2024). Moreover, the unequal distribution of NBS impacts across spatial scales and social groups can exacerbate trade-offs and potential social, racial, and health inequities arising from the access to the benefits provided by NBS (Wolch et al., 2014). Such disparities can result in forms of societal marginalization, such as green gentrification (Anguelovski et al., 2022).

Considering that NBS offer a variety of both desired and undesired effects and that NBS exhibit a context-sensitive and a site-specific nature (Raparthi & Vedamuthu, 2022), their implementation can result in the generation of synergies or tradeoffs among various NBS impacts. Because of this, NBS projects have been described as complex endeavors due to the multifactorial elements playing a role in their development, i.e., NBS types, implementation scales, conflicting interests, and lack of data (Rödl & Arlati, 2022). This complexity has posed challenges in establishing a clear methodology for comprehensively assessing the multifaceted implications of urban NBS (Dumitru et al., 2020). Such assessments must encompass a wide range of impacts, experienced both within and beyond urban boundaries, to mitigate unforeseen tradeoffs across spatial scales concerning sustainability, which ensures long-term viability, resilience, for enhancing capacity to adapt and recover from shocks, and equity, which promotes inclusive and just distribution of risks and benefits (United Nations, 2015).

While this is the case, the understanding of urban NBS impacts remains understudied (Kabisch et al., 2016; Rödl & Arlati, 2022), as urban regeneration and NBS planning is still dominated by silo-thinking, which tends to treat social and ecological challenges as distinct and occasionally in conflict with one another (Dumitru et al., 2020). NBS planning is still lacking integrative approaches that provide a clearer view of the effectiveness of NBS across multiple societal challenges. There continue to persist four great knowledge gaps when assessing the effectiveness of NBS (Kabisch et al., 2016): first, incomplete evidence base concerning the effectiveness of NBS relating to its short and long-term impacts, including trade-offs and synergies provided by NBS in terms of climate change mitigation/adaptation and biodiversity, human health, or social aspects; second, lack of understanding of people's perceptions and societal reactions to NBS effects; third, lack of clear information systems to assess the viability of NBS implementation; and fourth, insufficient knowledge of how to best design and implement NBS to best serve multiple and simultaneous purposes.

Furthermore, there is a need for a deeper understanding of the spatial dynamics influencing the effectiveness of NBS (Langemeyer et al., 2020), which should encompass how the location, design, and overall presence of NBS contribute to various impacts (Pereira et al., 2023). In the case of urban environments, this has been explored to some extent, but the focus often remains on assessing only the desired outcomes of NBS (i.e., net provision of ecosystem services), often overlooking potential undesired consequences of NBS implementation (Seddon, 2022).

In this same line, it is also relevant to consider that, due to the complexity and interconnectedness of urban environments (McPhearson et al., 2016), the impacts of urban NBS can also extend beyond their immediate urban surroundings. For instance, urban NBS are able to support regional-scale ecological connectivity (Molné et al., 2023). Also, the production and use of fertilizers for maintaining urban NBS (e.g., green roofs) can increase freshwater and marine eutrophication risks in non-urban environments (Tang et al., 2023). The broad scope of impacts poses a challenge to current NBS evaluations, as there is no defined method for comprehensively assessing their effects across different spatial scales.

Neglecting to fully grasp the diverse impacts stemming from different NBS configurations may lead to overlooking potential undesired effects, resulting in incomplete assessments of NBS effectiveness. To counter this, a more nuanced understanding of NBS impacts, including their spatial dynamics, is essential. Such an approach can help illustrate the various tradeoffs and synergies inherent in NBS, enhancing their assessment in urban settings. By adopting this nuanced approach, we can better monitor and evaluate NBS performance over time, while also gaining insights to optimize, scale up, and replicate these strategies (Dumitru et al., 2020).

To recap, NBS has the potential to aid urban areas in meeting their social and ecological challenges. However, our understanding of the impacts of urban NBS (both intended and unintended) remains limited. Current assessments tend to only consider net NBS effects, overlooking possible synergies and tradeoffs, and fail to fully account for the spatial distribution of NBS impacts or their broader influence beyond city boundaries. A more comprehensive evaluation of NBS considering these aspects is needed to allow for a better strategic and effective planning of NBS. This research gap leads to formulating the research question driving this Ph.D. dissertation:

How to better assess the impacts of Nature-based solutions from a spatial planning perspective to enhance their benefits and reduce their unintended consequences?

To begin addressing this question, I propose considering the premise that adopting a vulnerability-focused approach could offer a useful perspective in assessing NBS. Vulnerabilities, which can be broadly defined as the susceptibility to harm (Cutter, 2016) of both social and ecological systems, are spatially explicit and spatially heterogeneous. Vulnerabilities can be related to NBS impacts, as these impacts can either exacerbate or mitigate them, depending on the location and design of the NBS (Pereira et al., 2023). In doing so, shifting vulnerabilities may present an opportunity to observe possible synergies and tradeoffs arising from NBS impacts, under the assumption that desired NBS impacts would reduce vulnerabilities and undesired impacts would increase them. Moreover, vulnerability assessments have the capacity to gauge the extent and spatial distribution of hazards, providing a means for identifying spatial disparities in the ability to cope with them (Baró et al., 2021). This allows for a spatially explicit assessment of the potential impacts of NBS on vulnerabilities in urban environments.

Additionally, based on the previous premises, a vulnerability approach could also provide a pathway for understanding NBS impacts beyond urban limits, as systems and populations are not solely affected by the nearby presence of NBS, and vulnerabilities can extend beyond local boundaries through cascading effects (Little, 2010). Approaching NBS from this perspective allows a better assessment of the interplay between NBS impacts across spatial scales, providing further understanding of the tradeoffs and synergies arising from their overall impacts.

Vulnerability assessments have proven effective in strategically guiding urban decision-making and actions (Fekete, 2009; Nahiduzzaman et al., 2015; Rigillo & Cervelli, 2014). However, they remain an underexplored approach when it comes to assessing NBS impacts, providing a window of opportunity for proposing a way forward to better evaluating the heterogenous ways in which urban NBS behave, and how these can shape the urban and non-urban spaces in both desirable and undesirable ways.

I.2. Research objectives and research design

Building upon the background and research gaps highlighted above, the general aim of this thesis is to ***examine how various impacts linked to the implementation NBS in urban environments can be assessed and integrated by employing a vulnerability approach***. This general goal is structured in three specific objectives (see **Fig. 1**), each associated with one or more chapters of the thesis (see **Fig. 2**). Below is a description of each specific objective.

- Objective 1: To examine the relationship between complex urban environments and the impacts of NBS. Specific objectives:
    - To explore the multiple implications of NBS impacts in complex urban environments by describing how these relate to the urban challenges of Sustainability, Resilience and Equity.
    - To examine how adopting a vulnerability-focused approach can improve the understanding of NBS impacts in complex urban environments.
- Objective 2: To assess the impacts of urban NBS on local-scale vulnerabilities. Specific objectives:
    - To develop a stepwise, multi-criteria, and integrated assessment framework that allows for the evaluation of NBS impacts on local-scale vulnerabilities.
    - To relate desired and undesired impacts of urban NBS with shifts in local-scale vulnerabilities, portraying both synergies and tradeoffs.
    - To test the framework's effectiveness by assessing how different potential (peri)urban agriculture scenarios could shift local-scale vulnerabilities in the Metropolitan area of Barcelona.
- Objective 3: To evaluate the influence of urban NBS on both local and broad-scale vulnerabilities. Specific objectives:
    - To develop a stepwise, multi-criteria, and integrated assessment framework that allows for the assessment of NBS impacts on local and broad-scale vulnerabilities.

- To relate desired and undesired impacts of urban NBS with shifts in local and broad-scale vulnerabilities, portraying synergies and trade-offs across spatial scales.
- To test the framework's effectiveness by assessing how different potential green roof scenarios in the municipality of Oslo could shift both local and broad-scale vulnerabilities.

To address these objectives, this dissertation is structured in five chapters. The introductory chapter (Chapter I) presents the existing research gaps and sets out the overall research objectives. It also establishes a common understanding of the concept of NBS and the current state of the evaluation of their impacts. Additionally, I will outline the two distinct case studies employed in this dissertation, explaining their main differences and significance in allowing the development and validation of the NBS-vulnerability framework.

Following, Chapter II addresses Research objective 1 (see **Fig. 1**), by exploring the multifaceted impacts of implementing NBS in complex urban environments, shedding light on both expected and unexpected impacts across the different urban challenges of Sustainability, Resilience, and Equity. Furthermore, it examines how adopting a vulnerability-focused approach could provide a way forward for tackling the current shortcomings inherent in existing approaches to NBS assessments and allow a better understanding of the benefits and detriments produced by urban NBS and the extent of their impacts within and beyond urban limits.

Next, Chapter III portrays a more practical endeavor. It focuses on addressing Research objective 2 by developing a framework able to assess the extent to which NBS impacts can alter local-scale vulnerabilities. The framework is based on relating ecosystem services and urban metabolism impacts arising from NBS to spatially explicit vulnerabilities, and it is tested on the case study of increasing (peri-)urban agriculture in the Metropolitan Area of Barcelona. The assessment was able to successfully assess the impacts of peri-urban agriculture in terms of critical vulnerabilities experienced within the urban boundaries of the city.

In Chapter IV, and building upon the NBS-vulnerability framework developed in Chapter III, I further developed the assessment of NBS impacts by considering those that affect vulnerabilities beyond the urban limits where NBS are implemented, addressing Research objective 3. This approach was applied to the case study of green roofs in Oslo, to assess their impacts on both local-scale vulnerabilities and broad-scale vulnerabilities. Results were employed for creating an optimal green roof configuration providing the best combinations of impacts, displaying results that challenge the common notion of urban green maximization. The chapter contends that a more nuanced understanding of NBS impacts can be achieved by observing their cross-scale trade-offs and synergies, which can be a valuable aspect for creating more sustainable, resilient and equitable urban environments.

Chapter V synthesizes and discusses the main findings and theoretical contributions to the academic and urban planning themes related to the assessment and implementation of NBS impacts. Additionally, this chapter outlines potential areas for further research, highlights the main conceptual and methodological limitations, and offers recommendations for further improving the assessment of NBS.

This Ph.D. thesis is embedded in the European Research Council (ERC) Consolidator project: integrated System Analysis of Urban Vegetation and Agriculture (818002-URBAG). This research project aims to determine to what extent green infrastructures can be effective in contributing to make cities more resilient to climate change and more sustainable in terms of water management, food production, air quality, human-well-being and biodiversity. This project aims to develop an integrated, spatially resolved framework for quantifying food-water-energy interaction and quantitative analysis and simulation.

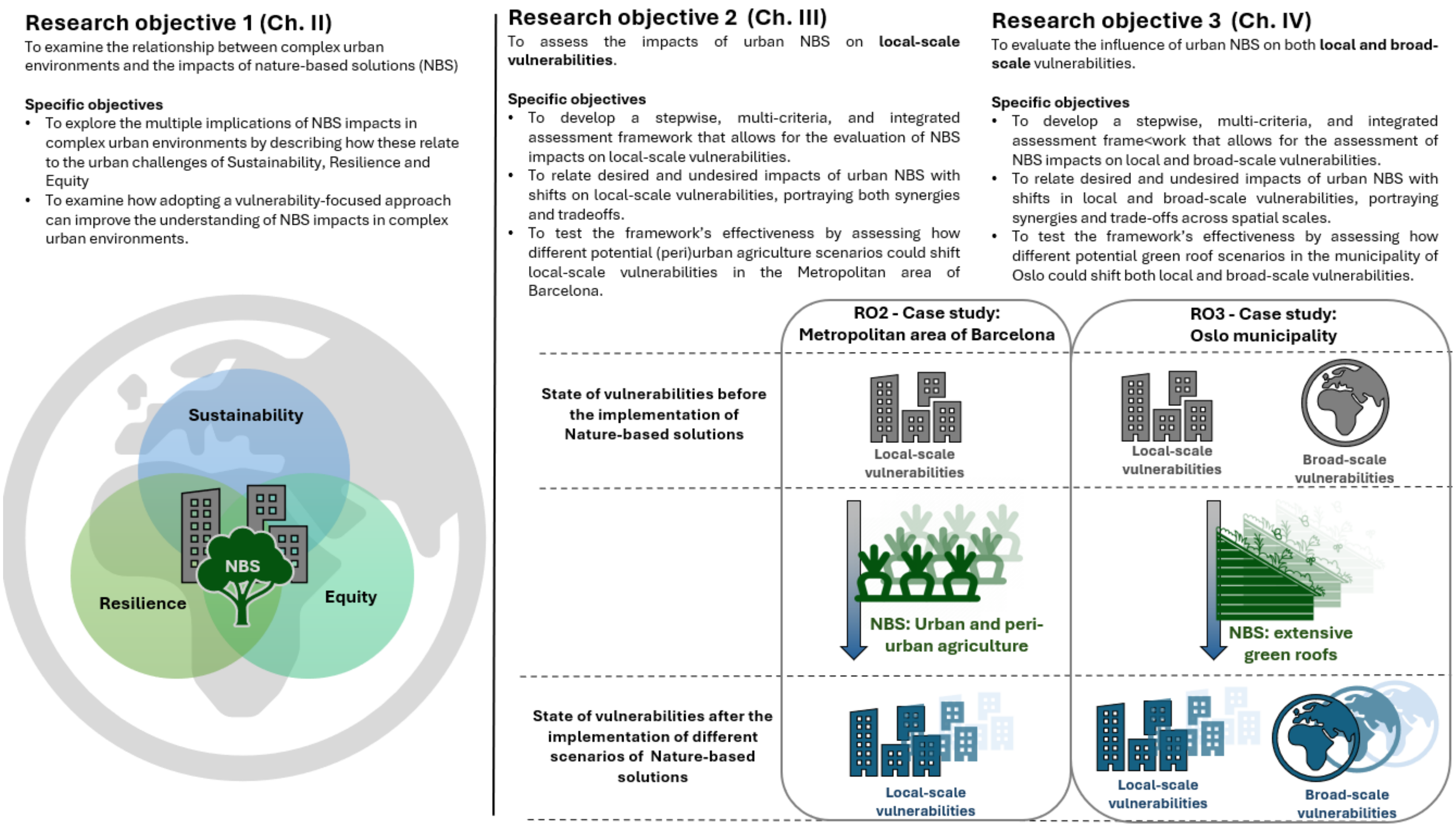


Figure 1. Graphical abstract of Ph.D. dissertation, based on specific objectives and chapter distribution

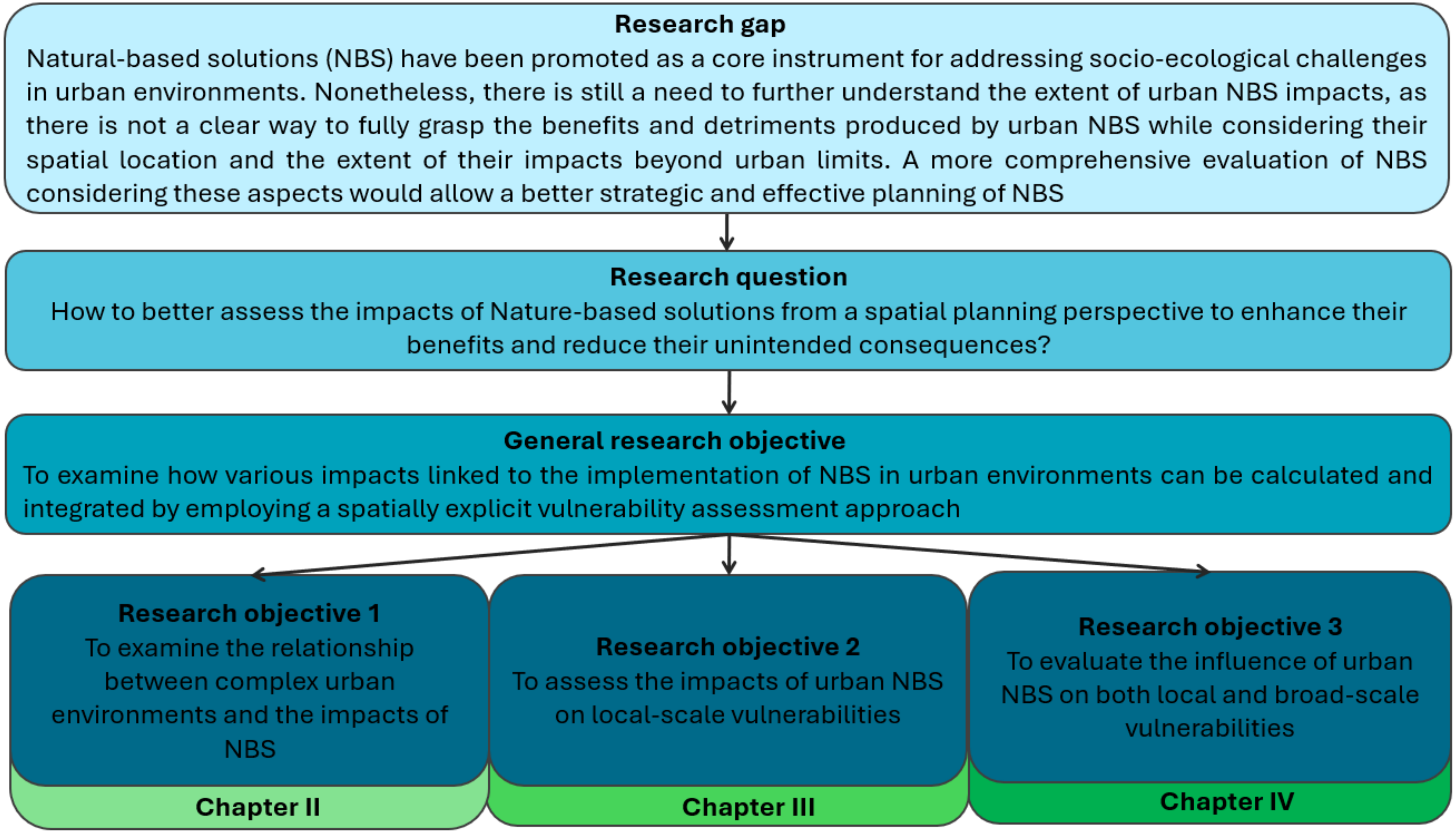


Figure 2. Research gap, question, objectives and chapter distribution of Ph.D. dissertation

I.3. Nature-based solutions in urban environments: an overview

I.3.1. Nature-based solutions: conceptualization and understanding of its impacts

The latest and most widespread definition of NBS describes them as "actions to protect, conserve, restore, sustainably use and manage natural or modified terrestrial, freshwater, coastal and marine ecosystems which address social, economic and environmental challenges effectively and adaptively, while simultaneously providing human well-being, ecosystem services, resilience and biodiversity benefits" (United Nations, 2022). Additionally, and aligned with this definition, a set of implementation principles aids in elucidating the concept and scope of NBS (Cohen-Shacham et al., 2016). First, NBS should adhere to nature conservation norms. Second, NBS can be deployed independently or in conjunction with other solutions, such as technological and engineering approaches, to tackle societal challenges. Third, NBS strategies should be tailored to the specific natural and cultural contexts of each site, encompassing traditional, local, and scientific knowledge. Fourth, NBS should yield societal benefits equitably and transparently, fostering broad participation. Fifth, NBS must safeguard biological and cultural diversity, as well as the adaptability of ecosystems over time. Sixth, NBS interventions are implemented on a landscape scale. Seventh, NBS acknowledges the trade-offs between immediate economic gains and the preservation of future options for ecosystem services. Eighth, NBS should be integrated into the overarching design of policies, measures, or actions aimed at addressing specific challenges.

While sharing a common definition, NBS has been divided into typologies based on the required level of engineering and development involved in its implementation (See **Fig. 3**). Three types of NBS typologies are suggested (Cohen-Shacham et al., 2016):

1. Type 1: solutions that involve making better use of existing natural or protected ecosystems, with no or minimal intervention (e.g., protected urban forests)
2. Type 2: solutions that involve enhancing or diversifying existing ecosystems (e.g., restoration of urban agricultural areas)
3. Type 3: solutions that involve creating new ecosystems (e.g. creation of green roofs)

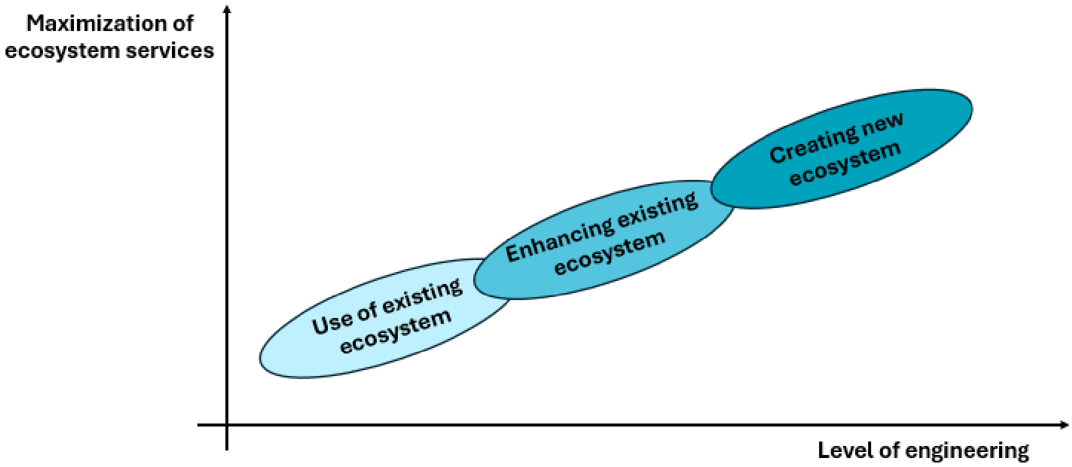


Figure 3. Classification of Nature-based Solutions. Adapted from Eggermont et al. (2015)

Whereas all types of NBS exist in urban environments, Types 1 and 2 face significant threats from the expansion of urban areas and the strain of increasing population (Moschetto et al., 2021). Specifically, the advance of urban sprawl disproportionately affects semi-natural grasslands, neglected agricultural lands, and forests. This expansion also contributes to the fragmentation of landscapes, exacerbating land degradation (Ren et al., 2022; Wang & Wang, 2022). A lack of consideration of these dynamics can create inconsistencies when promoting the creation of urban NBS, in cases where Type 3 NBS, which requires more investment and maintenance, unintentionally substitute existing Type 1 and 2 NBS (Pereira et al., 2023). In this regard, the intrinsic characteristics associated with NBS typologies and their interactions with urban environments require further consideration when assessing their impacts, as they may play a role in the benefits and detriments provided by NBS.

To have a clearer understanding of the categorization and assessment of impacts arising from NBS implementation, the next section will review the existing frameworks and methodologies that are currently being employed for evaluating the effects of NBS.

I.3.2. Current approaches for evaluating the impacts of Nature-based solutions

As previously mentioned, the understanding of urban NBS impacts is still limited. This is primarily due to the tendency to address social and ecological issues separately, sometimes seeing them in conflict with each other. Additionally, there is a lack of integrated methodologies to fully grasp how NBS can effectively tackle diverse urban ang global challenges (see **Fig. 4**).

For instance, Jezzini et al. (2023) conducted a comprehensive literature review focusing on the various models and methods used to quantify the specific effects of green infrastructures, a prominent type of NBS in urban environments. Their study identified 25 distinct impacts, such as air purification, recreational opportunities, water consumption, and gentrification. They provided detailed methodologies for calculating each impact, tailored to the specific type of green infrastructure. Similarly, the EU Handbook for evaluating the impacts of NBS (European Commission, 2021) provide decision-makers with a robust set of single methodologies to assess the impacts of NBS across 12 societal challenge areas ranging from climate resilience and biodiversity enhancement to social justice and cohesion. Even when individual methods and models offer valuable insights for understanding the level of achievement of an NBS objective, these are not enough for grasping the full extent of NBS impacts in urban environments. This is related to the fact that the NBS location, design and overall presence provide a variety of impacts that can change the state of urban conditions in both intended and unintended ways (Pereira et al., 2023). In this sense, it is recommended to develop holistic approaches to effectively balance the benefits and challenges associated with NBS (Jezzini et al., 2023).

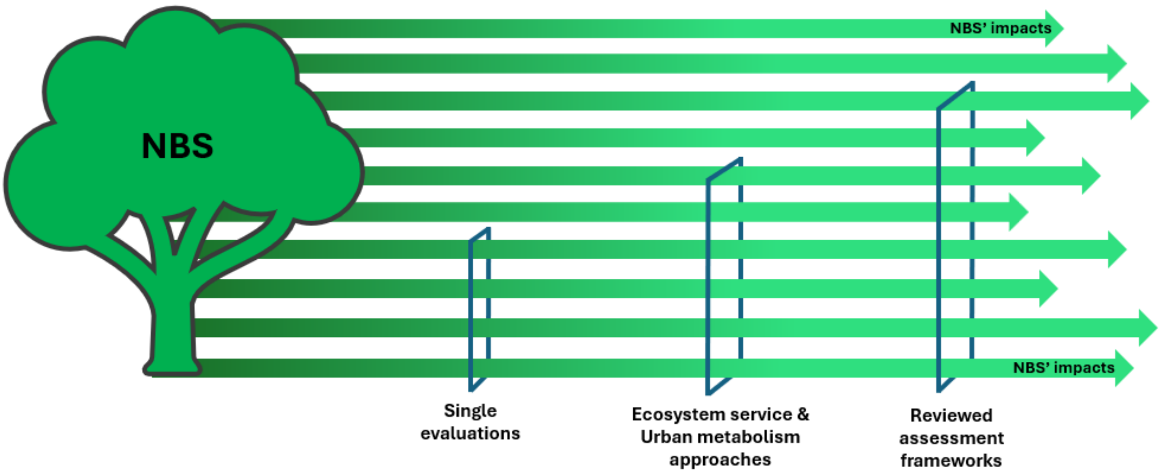


Figure 4. Visual representation of the capabilities of current approaches for evaluating the impacts of nature-based solutions (NBS). Arrows depict the diverse and simultaneous desired and undesired impacts generated by NBS, while windows symbolize the capabilities of the existing evaluation approaches to comprehend these impacts.

More comprehensive strategies for understanding the impacts of NBS include the Ecosystem Service (ES) and Urban Metabolism (UM) approaches, which, instead of employing single methodologies for assessing NBS, can include one or more methods to assess a range of NBS impacts. ES refers to the diverse ways in which humans derive benefits from natural ecosystems (Fisher et al., 2009). NBS can deliver these benefits and co-benefits across environmental, social, and economic dimensions. ES delivered by NBS comprises regulating services such as temperature regulation and flood control. Provisioning services entail the supply of food, water, and raw materials. Cultural services encompass recreational opportunities, aesthetic value, and the preservation of cultural heritage. Finally, supporting services involve aspects such as soil formation, nutrient cycling, and biodiversity conservation (Gómez-Baggethun et al., 2013). The ES approach recognizes that NBS provide benefits in a spatially explicit way (Herreros-Cantis & McPhearson, 2021) and the location of NBS determines the ES distribution.

UM, on the other hand, describes the change in the flow of resources and energy associated with the functioning of the NBS under study in urban environments throughout its lifetime. Under this perspective, NBS has been examined to understand their operational requirements and the impacts they have in urban environments (e.g., Wang et al., 2020) demonstrating that NBS can improve resource efficiency by reducing energy consumption, water consumption, and waste production. Nevertheless, UM has also been employed for studying NBS' undesired impacts (e.g., excessive water consumption, and emission of air pollutants). In this sense, UM provides valuable insights in regard to NBS functioning. Although undesired NBS impacts have been recognized in the literature (e.g., Pereira et al., 2023; Roman et al., 2020), their consideration when assessing the overall impacts of NBS are still underdeveloped (Perrotti & Stremke, 2020). Furthermore, UM approaches lack clearer consideration of geographically explicit NBS impacts, which has been deemed necessary for a more differentiated understanding of NBS (Mendoza Beltran et al., 2022).

Despite both ES and UM aim to comprehend the impacts of NBS over the long term, both disciplines diverge significantly in their utilization of concepts, assessment approaches, analytical tools, and models (Cárdenas-Mamani & Perrotti, 2022). Even so, both of these approaches provide a more comprehensive understanding of the impacts of NBS compared to the single assessment techniques previously mentioned. The ES approach acknowledges multifunctionality, recognizing NBS' ability to simultaneously deliver multiple benefits. This perspective considers the dynamic nature of NBS, understanding that synergies and trade-offs may emerge from their impacts. The UM approach also takes into account various impacts of NBS, encompassing alterations in the energy flow of a city, as well as the production and emission of materials. This perspective extends beyond urban limits, considering factors such as carbon emissions associated with the production and transportation of elements like green roofs to the city.

While both ES and UM approaches offer valuable insights into the study of NBS, they have some limitations. For instance, the ES approach, while adept at capturing changes in resource use in urban environments through factors like urban agriculture's impact on food production, overlooks critical aspects such as water irrigation and fertilizer use. Conversely, the UM approach neglects NBS impacts unrelated to resource and energy flows, such as the recreational potential of urban parks. Even though, a combined approach integrating spatial considerations of ES impacts with UM's capacity to assess NBS impacts beyond urban environments, while overcoming the limitations of net assessment of NBS impacts, could significantly enhance our understanding of NBS by portraying a clearer way to fully grasp the benefits and detriments produced by NBS. However, the joint assessment of NBS under these two approaches remains partial (Elliot et al., 2022), even though efforts to combine these approaches are being developed (see Cárdenas-Mamani & Perrotti, 2022).

Furthermore, novel assessment methods for NBS have emerged. These approaches seek to broaden the evaluation of NBS beyond singular assessments, incorporating pre-existing methods like the ES approach to enhance the comprehension of NBS impacts. **Table 1** provides a compilation of key frameworks used to assess the impacts of NBS, evaluated based on essential characteristics outlined in the literature for a comprehensive NBS assessment. Further elaboration on these characteristics will be provided in the subsequent paragraphs. Many of these frameworks are endorsed in the EU Handbook for evaluating the impacts of NBS (Dumitru et al., 2020), while additional frameworks identified in the existing literature have also been incorporated.

The review consistently identifies a multi-criteria assessment of NBS impacts across all frameworks. This implies the simultaneous evaluation of various NBS impacts, even in those cases where specific impact categories are the primary focus (e.g., Mahmoud et al., 2021) concentrates solely on the multiple social impacts of NBS). Multi-criteria assessment is a useful tool for developing holistic assessments of urban NBS (Venter et al., 2021), as it enables the integration of quantitative and qualitative data, discordant information, and stakeholders' considerations into decision-making processes, and allows the comparison of various alternatives by weighting different evaluation criteria (Marttunen et al., 2017). Such an approach has been described as necessary for the evaluation of NBS (Dumitru et al., 2020) and for the comprehensive assessments of urban land changes (Langemeyer et al., 2016) in the face of urban sustainability policies (Kalantari et al., 2019). Successful applications of multi-criteria assessment for the spatially explicit evaluation of NBS in urban environments can be found in the literature (Asare et al., 2024; Langemeyer et al., 2020; Venter et al., 2021), although these are usually focused only on the possible benefits expected from NBS within the urban limits of where these are implemented.

Moreover, while the multi-criteria approach involves separately evaluating different impacts of NBS by addressing them individually, the integrated assessment considers the cumulative effects of multiple impacts simultaneously. This approach aims for a holistic understanding of the overall impact of NBS. In this sense, most of the frameworks did not capture this quality, except the one proposed by Langemeyer et al. (2020). In this case, the NBS desired impacts are assessed by assigning weights through stakeholder input. However, this approach missed the actual quantification of the expected benefits and did not consider the possible undesired impacts arising from the NBS under study.

In contrast, stakeholder participation in the assessment of NBS was present in most of the frameworks. This is a crucial aspect as stakeholder engagement helps navigate the complexity of NBS planning by incorporating diverse urban perspectives (Nesshöver et al., 2017). Such engagement aids in comprehending the contextual nuances of NBS development, a valuable aspect given that NBS are context-specific and site-sensitive. Moreover, the participation of diverse stakeholders, especially those belonging to the communities where NBS are implemented, can enhance equity in the NBS planning and assessment (Sekulova et al., 2021), and provide a window of opportunity for recognizing which NBS impacts can be desired or undesired based on participants experience, background and expectations (Kiss et al., 2022). Because of these traits, stakeholder engagement is encouraged for NBS planning (Zingraff-Hamed et al., 2020).

Furthermore, some frameworks have been developed to assess specific types of NBS impacts. For instance, Altamirano et al., (2021) and Caroppi et al., (2024) only evaluate the effects of NBS on hydrological systems, while Mahmoud et al., (2021) concentrates on the social impacts of NBS. While these approaches provide valuable information on the very specific impacts in their respective areas, they are not relevant for a comprehensive assessment of possible synergies or tradeoffs that NBS impacts could have among the various challenges faced by urban environments (Raymond et al., 2017; Samuelsson et al., 2018)

In this same line of thought, only two of the frameworks (Raymond et al., 2017 and Gómez Martín et al., 2020) explicitly consider the possible impacts that NBS could have beyond the site-specific location where they are implemented. The exclusion of these aspects can result in a partial assessment of NBS impacts, considering that urban environments are a dynamic and interconnected network of biophysical and social elements that interact across multiple scales and affect the flow and use of critical natural, socio-economic, and cultural resources (Redman et al., 2004) even outside of their physical environments (Seto et al., 2012). While there have been efforts to analyze and quantify these impacts (e.g., Benis & Ferrão, 2017; Gargari et al., 2016), such assessment focuses on single impacts without consideration of the wider implication of NBS (e.g., synergies and tradeoffs).

The undesired impacts of NBS were missing across most of the studied frameworks (except for Gómez Martín et al., 2020). For clarification, some of the approaches considered the financial implications of developing and maintaining NBS (e.g., Somarakis et al., 2019), however, this aspect is not being considered within this dissertation. The lack of consideration for the undesired effects of NBS has been raised in the past, particularly in the context of addressing impacts stemming from ineffective ecosystem management (Eggermont et al., 2015). To gain a comprehensive understanding of how NBS can influence urban environments, it's crucial to examine both desired and undesired effects to have a clearer understanding of which are the synergies and tradeoffs associated with NBS implementations. This holistic perspective is essential for effectively addressing and either enhancing or mitigating urban challenges (Gómez Martín et al., 2020).

Finally, a few frameworks of NBS evaluation (e.g., Mahmoud et al., 2021; Altamirano et al., 2021, Caroppi et al., 2024) conducted comparisons between the existing conditions and the anticipated conditions after the implementation of NBS. Evaluating the impacts of NBS and comparing them to the current state of urban settings is recommended to gauge the extent of these impacts and ensure the effectiveness of NBS (Mussinelli et al., 2021). This process aligns with the theory of change, ensuring interventions follow clear causal pathways from actions to outcomes (Mahmoud et al., 2021) and the ex-ante approach which can provide key findings for the evaluation of NBS impacts (Langemeyer et al., 2016).

This review has shown that, while efforts are being made to better assess the NBS impacts in urban environments, existing NBS evaluation frameworks fall short of relevant aspects deemed necessary for a proper assessment. For instance, several proposals focus on assessing single NBS desired impacts, without consideration of the wider implications of NBS implementation. Conversely, more developed approaches may consider multiple impacts of NBS but tend to focus solely on desired or undesired net impacts, failing to integrate a more holistic assessment. Furthermore, considerations regarding the spatial distribution of NBS impacts, both within and beyond their implementation sites, remain underdeveloped, resulting in a lack of clarity regarding potential synergies and trade-offs. All this leaves room for improving NBS assessments to provide a more accurate evaluation of NBS' expected and unexpected impacts, as well as the possible synergies and trade-offs arising from them. To address this deficiency, I propose to adopt a spatially explicit vulnerability-focused approach, which will be further developed in the upcoming chapters.

Table 1. List of frameworks for the evaluation of Nature-based solutions with its main characteristics

Reference	Objective of the framework	Methodology	Multi-criteria assessment (Y/N)	Integrated assessment (Y/N)	Stakeholder participation (Y/N)	Focus on specific type(s) impact (Y/N)	NBS impacts beyond urban limits (Y/N)	Considers undesired impacts of NBS (Y/N)	Urban conditions pre- and post-NBS (Y/N)
Jeuken et al. (2020)	To build a system of multiscale and multi-thematic Urban Performance Indicators for the assessment of Urban Challenges and NBS	1) Definition of individual urban challenges and sub-challenges, categorized under topics of climate, environment, resources, economy, and social. 2) Selection of relevant urban performance indicators for the assessment of NBS based on urban challenges	YES	NO	NO	NO	NO	NO	NO
Maia et al. (2021)	To help in exploring or examining the contributions that NBS can make towards meeting urban sustainability challenges	1) Selection of NBS to study 2) Selection the relevant sustainability challenges to address by NBS (10 challenges) 3) Analysis of the expected benefits arising from the selected NBS (based on the pre-assessed scores provided by Naturvation) 4) Comparison of the expected benefits of the selected NBS to those from alternative NBS	YES	NO	YES	YES	NO	NO	NO
Somarakis et al. (2019)	To provide a step-wise guide to lead the successful implementation of NBS in urban environments	1) Definition of challenges to be addressed by NBS 2) Identification of relevant stakeholders to be considered for the assessment of NBS 3) Definition of objectives to be achieved by NBS 4) Definition of scenarios portraying different NBS configurations 5) Evaluation the scenarios by using multiple performance criteria 6) Developing of a financing scheme for NBS 7) Implementation of NBS 8) Monitoring	YES	NO	YES	NO	NO	NO	YES
Dumitru & Lourido (2022)	To support cities in developing and successfully implementing robust monitoring and evaluation plans that can deliver systematic and comparable evidence as to NBS effectiveness.	1) Matching NBS expected impacts to the city´s strategic objectives 2) Choosing appropriate indicators 3) Developing a data plan for impact evaluation	YES	NO	YES	NO	NO	NO	NO

Table 1. List of frameworks for the evaluation of Nature-based solutions with its main characteristics (continuation)

Reference	**Objective of the framework**	**Methodology**	**Multi-criteria assessment (Y/N)**	**Integrated assessment (Y/N)**	**Stakeholder participation (Y/N)**	**Focus on specific type(s) impact Y/N**	**NBS impacts beyond urban limits (Y/N)**	**Considers undesired impacts of NBS (Y/N)**	**Urban conditions pre- and post-NBS (Y/N)**
Raymond et al. (2017)	To assess the performance of NBS in dealing with challenges related to climate resilience in urban areas	1) Select of NBS to study 2) identification of expected impact of NBS on societal challenges 3) Selection of indicators for assessing the impacts of NBS 4) Assessment of potential impacts of NBS on societal challenges	YES	NO	YES	NO	YES	NO	NO
Altamirano et al. (2021)	To characterize the process of value capture from the supply side (service providers through the implementation of the NBS) to the demand side (service beneficiaries), and to identify potential business models for NBS	1) Identification of the conditions of the environments where NBS will be implemented 2) Quantification of the desired impacts of NBS 3) Discussion of desired conditions with decision-makers/stakeholders 4) Development of impact assessment for assessing environmental conditions with and without NBS. 4) Implementation of NBS	YES	NO	YES	YES	NO	NO	YES
Baldacchini (2019)	To gain knowledge on the cost-effectiveness of NBS, establish a replicable model for assessing them and to quantify their benefits to society	1) Select of NBS to study 2) Select the relevant challenges to address by NBS (4 challenges) 3) Data collection on NBS impacts (spatial and experimental) 4) Yearly monitoring of NBS impacts	YES	NO	YES	NO	NO	NO	NO
Eiter et al. (2022)	To select the relevant indicators to assess the effects of Edible City Solutions.	1) Application of suggested indicators for a single evaluation of the impacts of NBS	NO	NO	NO	YES	NO	NO	NO

Table 1. List of frameworks for the evaluation of Nature-based solutions with its main characteristics (continuation)

Reference	**Objective of the framework**	**Methodology**	**Multi-criteria assessment (Y/N)**	**Integrated assessment (Y/N)**	**Stakeholder participation (Y/N)**	**Focus on specific type(s) impact Y/N**	**NBS impacts beyond urban limits (Y/N)**	**Considers undesired impacts of NBS (Y/N)**	**Urban conditions pre- and post-NBS (Y/N)**
van der Jagt et al. (2023)	To assess NBS by the inclusive participation of relevant stakeholders	1) Perform stakeholder mapping 2) Selection of indicators aligned with locally relevant societal challenge areas 3) (pre)selection of indicators from the portfolio based on the co-defined monitoring goals and objectives 4) Conducting and indicator appraisal workshop 5) Assessment of NBS	YES	NO	YES	NO	NO	NO	NO
Mahmoud et al. (2021)	To identify macro categories of evaluation, along with sub-sectors of indicators that could be transversal for the evaluation of NBS in different cities	1) Scoping and gathering information of the relevant social impacts of NBS 2) Pre selection of indicators for assessing social impacts 3) conducting verification workshops 4) Questionnaires development 5) Data collection and analysis	YES	NO	YES	YES	NO	NO	YES
Caroppi et al. (2024)	To analyze the performance of NBS for hydro-meteorological risk management	1) selection and application of key performance indicators for the (co)benefits and costs associated with the implementation of NBS 2) Stakeholder weighting of (co)benefits 3) aggregation of indicators	YES	NO	YES	YES	NO	NO	YES
Langemeyer et al. (2020)	To support decision analysis of the location of the most suitable NBS (green roofs) for optimized ecosystem service provision in Barcelona	1) Selection of NBS alternatives 2) Selection of ecosystem service relevant for the study area 3) Stakeholder workshop for determine the ES that should be prioritized, the capacity of NBS to provide such NBS and the feasibility of NBS 4) Mapping of ecosystem service demands 5) Development of a supply-demand model of ecosystem 6) Aggregation of indicators	YES	YES	YES	NO	NO	NO	NO

Table 1. List of frameworks for the evaluation of Nature-based solutions with its main characteristics (continuation)

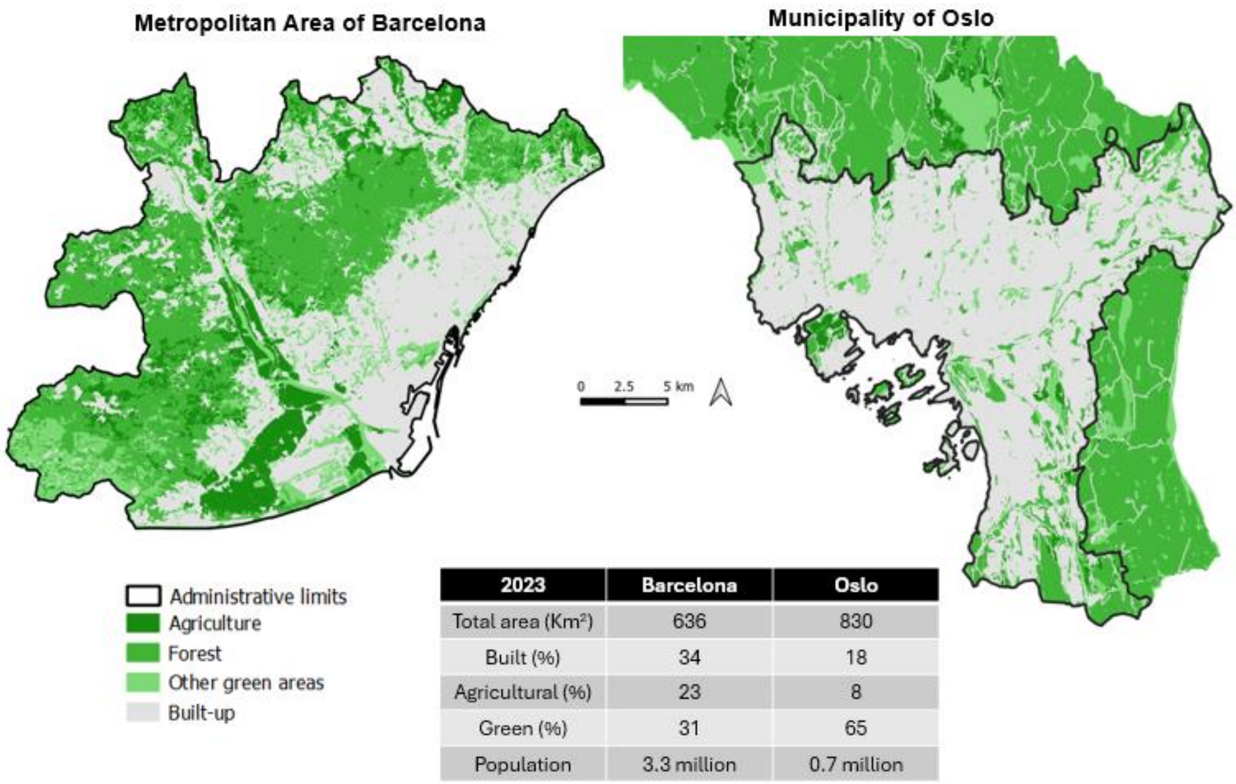
Reference	Objective of the framework	Methodology	Multi-criteria assessment (Y/N)	Integrated assessment (Y/N)	Stakeholder participation (Y/N)	Focus on specific type(s) impact Y/N	NBS impacts beyond urban limits (Y/N)	Considers undesired impacts of NBS (Y/N)	Urban conditions pre- and post-NBS (Y/N)
Padró et al. (2020)	to integrate social metabolism variables into land planning, through the quantification of the metabolic flows of the green infrastructure land uses	1) Definition of NBS scenarios 2) Selection of socioecological indicators 3) Conduction of cartographic and statistical analyses based on indicators	YES	NO	NO	NO	NO	NO	NO
Calliari et al. (2019)	To assess the direct benefits/costs and co-benefits/costs of NBS via ex-ante approach	1) Define the baseline environment to be improved by NBS 2) Set concrete goals to be achieved by NBS 3) Recognizing external factors that can affect NBS performance 4) Identifying traditional, nature-based or hybrid alternatives 5) Climate-proof alternatives for anticipating Climate change impacts 6) Map expected (in)direct effects of alternatives 7) Set the criteria to evaluate alternatives 8) Evaluate alternatives	YES	NO	YES	NO	NO	NO	NO
García-Blanco et al. (2023)	To assess the NBS capacity to address climate change impacts	1) selection of indicators for the evaluation of NBS 2) Calculation and mapping of indicators 3) Weighting of indicators 4) aggregation of indicators	YES	NO	NO	YES	NO	NO	YES
Gómez Martín et al. (2020)	To provide an easy-to-use classification scheme focusing on hydrological extreme events	1) Define the level of human intervention within the NBS implementation 2) Select risk to be addressed by NBS 3) Select the type of area for NBS implementation 4) Identify both ecosystem services and disservices provided by NBS 5) Identify the scale of the impacts of NBS	YES	NO	NO	YES	YES	YES	NO

I.3.3. Case studies

Two urban environments were employed as case studies for developing the vulnerability framework proposed in this Ph.D. dissertation: The metropolitan area of Barcelona (Spain), and the Municipality of Oslo (Norway). These cities have been chosen due to their contrasting characteristics in terms of geography, urban structure, green infrastructure and population density (See **Fig. 5**). On one hand, Barcelona is one of the most densely populated urban areas in Europe, with a small fraction of green areas, but a rich variety of agricultural lands. On the other hand, Oslo is a city with extensive green coverage, a smaller built environment and lower population density. Both of these cities are the case study of the project URBAG, a project that has gathered and produced data which has been employed for the developing of this thesis.

Oslo and Barcelona have ongoing municipal plans for improving the presence and distribution of NBS in the city. For instance, Barcelona has intentions to further develop urban green spaces and urban agricultural areas, as described in its urban master plan (Barcelona Regional and AMB-PDU, 2020), while Oslo plans to increase the presence of green roofs and green facades in the upcoming years (Oslo Kommune, 2022). In addition, both cities have recognized urban challenges that can be addressed by NBS, such as temperature regulation for upcoming heatwaves (Barcelona Metropolitan Area - AMB, 2018; Oslo Kommune, 2020), lack of urban biodiversity conditions (Barcelona City Council, 2013; Oslo Kommune, 2023) and increase in the amounts of recreational opportunities in natural environments (Barcelona Regional and AMB-PDU, 2020; Oslo Kommune, 2015).

These cases provide a great opportunity for testing the feasibility and versatility of the NBS-vulnerability framework proposed in the dissertation. On one hand, developing this approach on two different urban environments allows to better understand how versatile the proposed approach is, while providing enough insights to understand its applicability limitations, thus establishing a robust methodological foundation for its further implementation. Furthermore, given that the assessment aligns with urban agendas and objectives, the findings could serve as valuable inputs for informing local policymaking concerning the development of NBS.



2023	Barcelona	Oslo
Total area (Km²)	636	830
Built (%)	34	18
Agricultural (%)	23	8
Green (%)	31	65
Population	3.3 million	0.7 million

Figure 5. Land use maps and some statistics of the Municipality of Oslo (and surroundings) and the Metropolitan Area of Barcelona

I. References

Altamirano, M.A., de Rijke, H., Basco Carrera, L., Arellano Jaimerena, B. (2021). *Handbook for the Implementation of Nature-based Solutions for Water Security: guidelines for designing an implementation and financing arrangement*. NAIAD Project

Anguelovski, I., Connolly, J. J. T., Cole, H., Garcia-Lamarca, M., Triguero-Mas, M., Baró, F., Martin, N., Conesa, D., Shokry, G., del Pulgar, C. P., Ramos, L. A., Matheney, A., Gallez, E., Oscilowicz, E., Máñez, J. L., Sarzo, B., Beltrán, M. A., & Minaya, J. M. (2022). Green gentrification in European and North American cities. *Nature Communications 2022 13:1*, *13*(1), 1–13. https://doi.org/10.1038/s41467-022-31572-1

Anguelovski, I., Shi, L., Chu, E., Gallagher, D., Goh, K., Lamb, Z., Reeve, K., & Teicher, H. (2016). Equity Impacts of Urban Land Use Planning for Climate Adaptation: Critical Perspectives from the Global North and South. Journal of Planning Education and Research, 36(3), 333-348. https://doi.org/10.1177/0739456X16645166

Asare, P., Atun, F., & Pfeffer, K. (2024). Spatial Multi-Criteria Analysis for Discovering Nature-Based Solutions Location for Urban Flood Mitigation in Accra. *Applied Spatial Analysis and Policy*, *17*(1), 207–239. https://doi.org/10.1007/S12061-023-09541-Y/FIGURES/10

Baldacchini, C. (2019). *Monitoring and Assessment Plan, Deliverable No. 4.1, proGIreg*. European Commission

Bauduceau, N., Berry, P., Cecchi, C., Elmqvist, T., Fernandez, M., Hartig, T., Krull, W., Mayerhofer, E., N, S., Noring, L., Raskin-Delisle, K., Roozen, E., Sutherland, W., & Tack, J. (2015). *Towards an EU Research and Innovation Policy Agenda for Nature-based Solutions & Re-*

naturing Cities: Final Report of the Horizon 2020 Expert Group on 'Nature-based Solutions and Re-naturing Cities'. Publications Office of the European Union. https://doi.org/10.2777/765301

Barcelona City Council. (2013). Barcelona green infrastructure and biodiversity plan 2020. In *Barcelona, Spain (in Spanish, English summary)*. https://ajuntament.barcelona.cat/ecologiaurbana/sites/default/files/Barcelona green infrastructure and biodiversity plan 2020.pdf

Barcelona Metropolitan Area - AMB. (2018). *Climate and Energy Plan 2030*. https://www.amb.cat/es/web/ecologia/actualitat/publicacions/detall/-/publicacio/plan-clima-y-energia-2030/7155151/11818

Barcelona Regional and AMB-PDU. (2020). *Initial Strategic Document. Advance of the Urbanisitic Metropolitan Master Plan* (1st ed., Vol. 1). AMB.

Baró, F., Camacho, D. A., Pérez Del Pulgar, C., Triguero-Mas, M., & Anguelovski, I. (2021). School greening: Right or privilege? Examining urban nature within and around primary schools through an equity lens. *Landscape and Urban Planning*, *208*, 104019. https://doi.org/10.1016/J.LANDURBPLAN.2020.104019

Benis, K., & Ferrão, P. (2017). Potential mitigation of the environmental impacts of food systems through urban and peri-urban agriculture (UPA) – a life cycle assessment approach. *Journal of Cleaner Production*, *140*, 784–795. https://doi.org/10.1016/J.JCLEPRO.2016.05.176

Calliari, E., Staccione, A., & Mysiak, J. (2019). An assessment framework for climate-proof nature-based solutions. *Science of The Total Environment*, *656*, 691–700. https://doi.org/10.1016/J.SCITOTENV.2018.11.341

Cárdenas-Mamani, Ú., & Perrotti, D. (2022). Understanding the contribution of ecosystem services to urban metabolism assessments: An integrated framework. *Ecological Indicators*, *136*, 108593. https://doi.org/10.1016/J.ECOLIND.2022.108593

Caroppi, G., Pugliese, F., Gerundo, C., De Paola, F., Stanganelli, M., Urciuoli, G., Nadim, F., Oen, A., Andrés, P., & Giugni, M. (2024). A comprehensive framework tool for performance assessment of NBS for hydro-meteorological risk management. *Journal of Environmental Planning and Management*, *67*(6), 1231–1257. https://doi.org/10.1080/09640568.2023.2166818

Cohen-Shacham, E., Walters, G., Janzen, C., & Maginnis, S. (2016). Nature-based solutions to address global societal challenges. *IUCN: Gland, Switzerland*, *97*, 2016-2036.

Cutter, S. L. (2016). Resilience to What? Resilience for Whom? *The Geographical Journal*, *182*(2), 110–113. https://doi.org/10.1111/GEOJ.12174

Dick, J., Miller, J. D., Carruthers-Jones, J., Dobel, A. J., Carver, S., Garbutt, A., Hester, A., Hails, R., Magreehan, V., & Quinn, M. (2019). How are nature based solutions contributing to priority societal challenges surrounding human well-being in the United Kingdom: A systematic map protocol. *Volume 8, Issue 1*, *8*(1). https://doi.org/10.1186/s13750-019-0180-4

Dumitru, A., Frantzeskaki, N., & Collier, M. (2020). Identifying principles for the design of robust impact evaluation frameworks for nature-based solutions in cities. *Environmental Science & Policy*, *112*, 107–116. https://doi.org/10.1016/J.ENVSCI.2020.05.024

Dumitru, A., & Lourido, D. T. (2022). Impact assessment guidebook. Connecting Nature Guidebook.

Eiter, S., Fjellstad, W., Metselaar, K., & Teixeira da Silva, R. (2022). *Prioritised indicators and baseline (D5.5)*. Zenodo. https://doi.org/10.5281/ZENODO.5895956

Eggermont, H., Balian, E., Azevedo, J. M. N., Beumer, V., Brodin, T., Claudet, J., Fady, B., Grube, M., Keune, H., Lamarque, P., Reuter, K., Smith, M., Van Ham, C., Weisser, W. W., & Le Roux, X. (2015). Nature-based Solutions: New Influence for Environmental Management and Research in Europe. *GAIA - Ecological Perspectives for Science and Society*, *24*(4), 243–248. https://doi.org/10.14512/GAIA.24.4.9

Elliot, T., Torres-Matallana, J. A., Goldstein, B., Babí Almenar, J., Gómez-Baggethun, E., Proença, V., & Rugani, B. (2022). An expanded framing of ecosystem services is needed for a sustainable urban future. *Renewable and Sustainable Energy Reviews*, *162*, 112418. https://doi.org/10.1016/J.RSER.2022.112418

Elmqvist, T., Andersson, E., Frantzeskaki, N., McPhearson, T., Olsson, P., Gaffney, O., Takeuchi, K., & Folke, C. (2019). Sustainability and resilience for transformation in the urban century. *Nature Sustainability*, *2*(4), 267–273. https://doi.org/10.1038/s41893-019-0250-1

Enssle, F., & Kabisch, N. (2020). Urban green spaces for the social interaction, health and well-being of older people— An integrated view of urban ecosystem services and socio-environmental justice. *Environmental Science & Policy*, *109*, 36–44. https://doi.org/10.1016/J.ENVSCI.2020.04.008

European Commission. (2021). *Evaluating the impact of nature-based solutions: A handbook for practitioners*.

Faivre, N., Fritz, M., Freitas, T., de Boissezon, B., & Vandewoestijne, S. (2017). Nature-Based Solutions in the EU: Innovating with nature to address social, economic and environmental challenges. *Environmental Research*, *159*, 509–518. https://doi.org/10.1016/J.ENVRES.2017.08.032

Fekete, A. (2009). Validation of a social vulnerability index in context to river-floods in Germany. *Natural Hazards and Earth System Sciences*, *9*(2), 393–403. https://doi.org/10.5194/NHESS-9-393-2009

Filho, W. L., Balogun, A. L., Olayide, O. E., Azeiteiro, U. M., Ayal, D. Y., Muñoz, P. D. C., Nagy, G. J., Bynoe, P., Oguge, O., Yannick Toamukum, N., Saroar, M., & Li, C. (2019). Assessing the impacts of climate change in cities and their adaptive capacity: Towards transformative approaches to climate change adaptation and poverty reduction in urban areas in a set of developing countries. *Science of The Total Environment*, *692*, 1175–1190. https://doi.org/10.1016/J.SCITOTENV.2019.07.227

Fisher, B., Turner, R. K., & Morling, P. (2009). Defining and classifying ecosystem services for decision making. *Ecological Economics*, *68*(3), 643–653. https://doi.org/10.1016/j.ecolecon.2008.09.014

García-Blanco, G., Navarro, D., & Feliu, E. (2023). Adopting Resilience Thinking through Nature-Based Solutions within Urban Planning: A Case Study in the City of València. *Buildings 2023, Vol. 13, Page 1317*, *13*(5), 1317. https://doi.org/10.3390/BUILDINGS13051317

Gargari, C., Bibbiani, C., Fantozzi, F., & Campiotti, C. A. (2016). Environmental Impact of Green Roofing: The Contribute of a Green Roof to the Sustainable use of Natural Resources in a Life Cycle Approach. *Agriculture and Agricultural Science Procedia*, *8*, 646–656. https://doi.org/10.1016/J.AASPRO.2016.02.087

Giama, E., Papageorgiou, C., Theodoridou, I., & Papadopoulos, A. M. (2021). Life Cycle Analysis and Life Cycle Cost Analysis of green roofs in the Mediterranean climatic conditions. *Energy Sources, Part A: Recovery, Utilization, and Environmental Effects*. https://doi.org/10.1080/15567036.2021.1914782

Global Covenant of Mayors for Climate & Energy. (2023). *Global Covenant of Mayors: Common Reporting Framework*. https://www.globalcovenantofmayors.org/wp-content/uploads/2023/11/CRF7-0-2023-09-14-final.pdf

Gómez Martín, E., Máñez Costa, M., & Schwerdtner Máñez, K. (2020). An operationalized classification of Nature Based Solutions for water-related hazards: From theory to practice. *Ecological Economics*, *167*, 106460. https://doi.org/10.1016/J.ECOLECON.2019.106460

Gómez-Baggethun, E., Gren, Å., Barton, D. N., Langemeyer, J., McPhearson, T., O'farrell, P., Andersson, E., Hamstead, Z., & Kremer, P. (2013). Urban ecosystem services. In *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities: A Global Assessment* (pp. 175–251). Springer Netherlands. https://doi.org/10.1007/978-94-007-7088-1_11

Hanson, H. I., Wickenberg, B., & Alkan Olsson, J. (2020). Working on the boundaries—How do science use and interpret the nature-based solution concept? *Land Use Policy*, *90*, 104302. https://doi.org/10.1016/J.LANDUSEPOL.2019.104302

Herreros-Cantis, P., & McPhearson, T. (2021). Mapping supply of and demand for ecosystem services to assess environmental justice in New York City. *Ecological Applications*, *31*(6), e02390. https://doi.org/10.1002/EAP.2390

IUCN (2009). *No time to lose: Make full use of nature-based solutions in the post-2012 climate change regime.* In *Position Paper on the Fifteenth Session of the Conference of the Parties to the United Nations Framework Convention on Climate Change (COP 15)* (pp. 1-5). Gland, Switzerland: International Union for Conservation of Nature.

Jezzini, Y., Assaf, G., & Assaad, R. H. (2023). Models and Methods for Quantifying the Environmental, Economic, and Social Benefits and Challenges of Green Infrastructure: A Critical Review. *Sustainability 2023, Vol. 15, Page 7544*, *15*(9), 7544. https://doi.org/10.3390/SU15097544

Jeuken, Y. R. H., Breukers, S., Elie, S., & Rugani, M. (2020). Nature4Cities: Nature Based Solutions Projects Implementation Handbook. https://doi.org/10.3030/730468

Kabisch, N., Frantzeskaki, N., Pauleit, S., Naumann, S., Davis, M., Artmann, M., Haase, D., Knapp, S., Korn, H., Stadler, J., Zaunberger, K., & Bonn, A. (2016). Nature-based solutions to climate change mitigation and adaptation in urban areas: Perspectives on indicators, knowledge gaps, barriers, and opportunities for action. *ECOLOGY AND SOCIETY*, *21*, 39. https://doi.org/10.5751/ES-08373-210239

Kalantari, Z., Santos Ferreira, C. S., Page, J., Goldenberg, R., Olsson, J., & Destouni, G. (2019). Meeting sustainable development challenges in growing cities: Coupled social-ecological systems modeling of land use and water changes. *Journal of Environmental Management*, *245*, 471–480. https://doi.org/10.1016/J.JENVMAN.2019.05.086

Kiss, B., Sekulova, F., Hörschelmann, K., Salk, C. F., Takahashi, W., & Wamsler, C. (2022). Citizen participation in the governance of nature-based solutions. *Environmental Policy and Governance*, *32*(3), 247–272. https://doi.org/10.1002/EET.1987

Kloss, P. (2022). The Interaction Between Resilience and Intelligence of Cities. *Smart Cities*, 23–34. https://doi.org/10.1201/9781003272199-3

Langemeyer, J., Gómez-Baggethun, E., Haase, D., Scheuer, S., & Elmqvist, T. (2016). Bridging the gap between ecosystem service assessments and land-use planning through Multi-Criteria Decision Analysis (MCDA). *Environmental Science and Policy*, *62*(2015), 45–56. https://doi.org/10.1016/j.envsci.2016.02.013

Langemeyer, J., Wedgwood, D., McPhearson, T., Baró, F., Madsen, A. L., & Barton, D. N. (2020). Creating urban green infrastructure where it is needed – A spatial ecosystem service-based decision analysis of green roofs in Barcelona. *Science of The Total Environment*, *707*, 135487. https://doi.org/10.1016/J.SCITOTENV.2019.135487

Little, R. G. (2010). Controlling Cascading Failure: Understanding the Vulnerabilities of Interconnected Infrastructures. *Http://Dx.Doi.Org/10.1080/106307302317379855*, *9*(1), 109–123. https://doi.org/10.1080/106307302317379855

Lulovicova, A., & Bouissou, S. (2024). Life cycle assessment as a prospective tool for sustainable agriculture and food planning at a local level. *Geography and Sustainability*, *5*(2), 251–264. https://doi.org/10.1016/J.GEOSUS.2024.01.008

MacKinnon, K., Sobrevila, C., & Hickey, V. (2008). *Biodiversity, climate change, and adaptation: nature-based solutions from the World Bank portfolio* (No. 46726, pp. 1-112). The World Bank.

Maia, S., Almasy, D., Pinter, L., Werner, A., Hörschelmann, K., Dammers, E., Bulkeley, H. (2021). *Urban nature navigator: silver level implementation guide*

Mahmoud, I. H., Morello, E., Vona, C., Benciolini, M., Sejdullahu, I., Trentin, M., & Pascual, K. H. (2021). Setting the social monitoring framework for nature-based solutions impact: Methodological approach and pre-greening measurements in the case study from clever cities milan. *Sustainability (Switzerland)*, *13*(17), 9672. https://doi.org/10.3390/SU13179672/S1

Marttunen, M., Lienert, J., & Belton, V. (2017). Structuring problems for Multi-Criteria Decision Analysis in practice: A literature review of method combinations. *European Journal of Operational Research*, *263*(1), 1–17. https://doi.org/10.1016/J.EJOR.2017.04.041

McInnes, R. N., Hemming, D., Burgess, P., Lyndsay, D., Osborne, N. J., Skjøth, C. A., Thomas, S., & Vardoulakis, S. (2017). Mapping allergenic pollen vegetation in UK to study environmental exposure and human health. *Science of The Total Environment*, *599–600*, 483–499. https://doi.org/10.1016/J.SCITOTENV.2017.04.136

McPhearson, T., Haase, D., Kabisch, N., & Gren, Å. (2016). Advancing understanding of the complex nature of urban systems. *Ecological Indicators*, *70*, 566–573. https://doi.org/10.1016/J.ECOLIND.2016.03.054

Mendoza Beltran, A., Jepsen, K., Rufí-Salís, M., Ventura, S., Madrid Lopez, C., & Villalba, G. (2022). Mapping direct N2O emissions from peri-urban agriculture: The case of the Metropolitan Area of Barcelona. Science of The Total Environment, 822, 153514. https://doi.org/10.1016/J.SCITOTENV.2022.153514

Molné, F., Donati, G. F. A., Bolliger, J., Fischer, M., Maurer, M., & Bach, P. M. (2023). Supporting the planning of urban blue-green infrastructure for biodiversity: A multi-scale prioritisation framework. *Journal of Environmental Management*, *342*, 118069. https://doi.org/10.1016/J.JENVMAN.2023.118069

Moschetto, F. A., Ribeiro, R. B., & De Freitas, D. M. (2021). Urban expansion, regeneration and socioenvironmental vulnerability in a mangrove ecosystem at the southeast coastal of São Paulo, Brazil. *Ocean & Coastal Management*, *200*, 105418. https://doi.org/10.1016/J.OCECOAMAN.2020.105418

Mussinelli, E., Tartaglia, A., Castaldo, G., & Cerati, D. (2021). Project and evaluation of nature-based solutions for the regeneration of public space. *Smart Innovation, Systems and Technologies*, *178 SIST*, 1342–1351. https://doi.org/10.1007/978-3-030-48279-4_125/COVER

Nahiduzzaman, K. M., Aldosary, A. S., & Rahman, M. T. (2015). Flood induced vulnerability in strategic plan making process of Riyadh city. *Habitat International*, *49*, 375–385. https://doi.org/10.1016/J.HABITATINT.2015.05.034

Nesshöver, C., Assmuth, T., Irvine, K. N., Rusch, G. M., Waylen, K. A., Delbaere, B., Haase, D., Jones-Walters, L., Keune, H., Kovacs, E., Krauze, K., Külvik, M., Rey, F., van Dijk, J., Vistad, O. I., Wilkinson, M. E., & Wittmer, H. (2017). The science, policy and practice of nature-based solutions: An interdisciplinary perspective. In *Science of the Total Environment* (Vol. 579, pp. 1215–1227). Elsevier B.V. https://doi.org/10.1016/j.scitotenv.2016.11.106

Nieuwenhuijsen, M. J., Barrera-Gómez, J., Basagaña, X., Cirach, M., Daher, C., Pulido, M. F., Iungman, T., Gasparrini, A., Hoek, G., De Hoogh, K., Khomenko, S., Khreis, H., De Nazelle, A., Ramos, A., Rojas-Rueda, D., Pereira Barboza, E., Tainio, M., Thondoo, M., Tonne, C., … Mueller, N. (2022). Study protocol of the European Urban Burden of Disease Project: a health impact assessment study. *BMJ Open*, *12*(1), e054270. https://doi.org/10.1136/BMJOPEN-2021-054270

Oslo kommune. (2015). *Oslo 2030: Smart, trygg og grønn*.

Oslo Kommune. (2020). *Climate change vulnerability analysis for Oslo*.

Oslo Kommune. (2022). *Strategi for grønne tak og fasader*. https://www.oslo.kommune.no/getfile.php/13452654-1654694941/Tjenester%20og%20tilbud/Plan%2C%20bygg%20og%20eiendom/Byggesaksveiledere%2C%20normer%20og%20skjemaer/Strategi%20for%20gr%C3%B8nne%20tak%20og%20fasader.pdf

Oslo Kommune. (2023). *Handlingsplan for biologisk mangfold i Oslo 2023–2030*.

Padró, R., La Rota-Aguilera, M. J., Giocoli, A., Cirera, J., Coll, F., Pons, M., Pino, J., Pili, S., Serrano, T., Villalba, G., & Marull, J. (2020). Assessing the sustainability of contrasting land use scenarios through the Socioecological Integrated Analysis (SIA) of the metropolitan green infrastructure in Barcelona. *Landscape and Urban Planning*, *203*, 103905. https://doi.org/10.1016/J.LANDURBPLAN.2020.103905

Panno, A., Carrus, G., Lafortezza, R., Mariani, L., & Sanesi, G. (2017). Nature-based solutions to promote human resilience and wellbeing in cities during increasingly hot summers. *Volume 159, Pages 249 - 256*, *159*, 249–256. https://doi.org/10.1016/j.envres.2017.08.016

Pereira, P., Yin, C., & Hua, T. (2023). Nature-based solutions, ecosystem services, disservices, and impacts on well-being in urban environments. *Current Opinion in Environmental Science & Health*, *33*, 100465. https://doi.org/10.1016/J.COESH.2023.100465

Perrotti, D., & Stremke, S. (2020). Can urban metabolism models advance green infrastructure planning? Insights from ecosystem services research. *Environment and Planning B: Urban Analytics and City Science*, *47*(4), 678–694. https://doi.org/10.1177/2399808318797131/ASSET/IMAGES/LARGE/10.1177_2399808318797131-FIG2.JPEG

Randrup, T. B., Buijs, A., Konijnendijk, C. C., & Wild, T. (2020). Moving beyond the nature-based solutions discourse: introducing nature-based thinking. *Urban Ecosystems*, *23*(4), 919–926. https://doi.org/10.1007/S11252-020-00964-W/FIGURES/1

Raparthi, K., & Vedamuthu, R. (2022). Assessing the Role of Nature-Based Solutions in Urban Resilience and Climate Change Adaptation. *Lecture Notes in Civil Engineering*, *191*, 13–21. https://doi.org/10.1007/978-981-16-5839-6_2

Raymond, C.M., Berry, P., Breil, M., Nita, M.R., Kabisch, N., de Bel, M., Enzi, V., Frantzeskaki, N., Geneletti, D., Cardinaletti, M., Lovinger, L., Basnou, C., Monteiro, A., Robrecht, H., Sgrigna, G., Munari, L. and Calfapietra, C. (2017) *An Impact Evaluation Framework to Support Planning and Evaluation of Nature-based Solutions Projects*. Centre for Ecology & Hydrology, Wallingford, United Kingdom

Redman, C. L., Grove, J. M., & Kuby, L. H. (2004). Integrating social science into the Long-Term Ecological Research (LTER) Network: Social dimensions of ecological change and ecological dimensions of social change. Ecosystems, 7(2), 161–171. https://doi.org/10.1007/S10021-003-0215-Z/FIGURES/3

Ren, Q., He, C., Huang, Q., Shi, P., Zhang, D., & Güneralp, B. (2022). Impacts of urban expansion on natural habitats in global drylands. *Nature Sustainability 2022 5:10*, *5*(10), 869–878. https://doi.org/10.1038/s41893-022-00930-8

Rigillo, M., & Cervelli, E. (2014). Mapping Urban Vulnerability: The Case Study of Gran Santo Domingo, Dominican Republic. *Advanced Engineering Forum*, *11*, 142–148. https://doi.org/10.4028/WWW.SCIENTIFIC.NET/AEF.11.142

Rödl, A., & Arlati, A. (2022). A general procedure to identify indicators for evaluation and monitoring of nature-based solution projects. *Ambio*, *51*(11), 2278–2293. https://doi.org/10.1007/S13280-022-01740-0/TABLES/3

Roman, L. A., Conway, T. M., Eisenman, T. S., Koeser, A. K., Ordóñez Barona, C., Locke, D. H., Jenerette, G. D., Östberg, J., & Vogt, J. (2020). Beyond 'trees are good': Disservices, management costs, and tradeoffs in urban forestry. *Ambio*, 615–630. https://doi.org/10.1007/s13280-020-01396-8

Samuelsson, K., Giusti, M., Peterson, G. D., Legeby, A., Brandt, S. A., & Barthel, S. (2018). Impact of environment on people's everyday experiences in Stockholm. *Landscape and Urban Planning*, *171*, 7–17. https://doi.org/10.1016/J.LANDURBPLAN.2017.11.009

Seddon, N. (2022). Harnessing the potential of nature-based solutions for mitigating and adapting to climate change. *Science*, *376*(6600), 1400–1416. https://doi.org/10.1126/SCIENCE.ABN9668/ASSET/BF73657E-636C-4029-BB0F-386F6D3AD3CE/ASSETS/IMAGES/LARGE/SCIENCE.ABN9668-F3.JPG

Sekulova, F., Anguelovski, I., Kiss, B., Kotsila, P., Baró, F., Palgan, Y. V., & Connolly, J. (2021). The governance of nature-based solutions in the city at the intersection of justice and equity. In *Cities* (Vol. 112). Elsevier Ltd. https://doi.org/10.1016/j.cities.2021.103136

Seto, K. C., Reenberg, A., Boone, C. G., Fragkias, M., Haase, D., Langanke, T., Marcotullio, P., Munroe, D. K., Olah, B., & Simon, D. (2012). Urban land teleconnections and sustainability. *Proceedings of the National Academy of Sciences of the United States of America*, *109*(20), 7687–7692. https://doi.org/10.1073/pnas.1117622109

Somarakis, G., Stagakis, S., Chrysoulakis, N., Mesimäki, M., & Lehvävirta, S. (2019). ThinkNature nature-based solutions handbook.

Tang, S., Yan, X., Jiang, J., Zheng, Y., Yang, Y., Xu, P., & Shang, F. (2023). Catchment-scale life cycle impacts of green infrastructures and sensitivity to runoff coefficient with stormwater modelling. *Science of The Total Environment*, *904*, 166736. https://doi.org/10.1016/J.SCITOTENV.2023.166736

United Nations. (2015). *Goal 11: Make cities inclusive, safe, resilient and sustainable*. https://www.un.org/sustainabledevelopment/cities/

United Nations. (2020). *Policies on Spatial Distribution and Urbanization Have Broad Impacts on Sustainable Development*. https://www.un.org/development/desa/pd/sites/www.un.org.development.desa.pd/files/undes_pd_2020_popfacts_urbanization_policies.pdf

United Nations. (2022). *Resolution adopted by the United Nations Environment Assembly on 2 March 2022*.

van der Jagt, A. P. N., Buijs, A., Dobbs, C., van Lierop, M., Pauleit, S., Randrup, T. B., & Wild, T. (2023). An action framework for the participatory assessment of nature-based solutions in cities. *Ambio*, *52*(1), 54–67. https://doi.org/10.1007/S13280-022-01772-6/FIGURES/4

Venter, Z. S., Barton, D. N., Martinez-Izquierdo, L., Langemeyer, J., Baró, F., & McPhearson, T. (2021). Interactive spatial planning of urban green infrastructure – Retrofitting green roofs where ecosystem services are most needed in Oslo. *Ecosystem Services*, *50*(June), 101314. https://doi.org/10.1016/j.ecoser.2021.101314

Wang, Q., & Wang, H. (2022). Spatiotemporal dynamics and evolution relationships between land-use/land cover change and landscape pattern in response to rapid urban sprawl process: A case study in Wuhan, China. *Ecological Engineering*, *182*, 106716. https://doi.org/10.1016/J.ECOLENG.2022.106716

Wang, Y., Ni, Z., Hu, M., Li, J., Wang, Y., Lu, Z., Chen, S., & Xia, B. (2020). Environmental performances and energy efficiencies of various urban green infrastructures: A life-cycle assessment. *Journal of Cleaner Production*, *248*, 119244. https://doi.org/10.1016/j.jclepro.2019.119244

Wei, Y. D., & Ewing, R. (2018). Urban expansion, sprawl and inequality. *Landscape and Urban Planning*, *177*, 259–265. https://doi.org/10.1016/J.LANDURBPLAN.2018.05.021

Wolch, J. R., Byrne, J., & Newell, J. P. (2014). Urban green space, public health, and environmental justice: The challenge of making cities 'just green enough.' *Landscape and Urban Planning*, *125*, 234–244. https://doi.org/10.1016/J.LANDURBPLAN.2014.01.017

Zingraff-Hamed, A., Hüesker, F., Lupp, G., Begg, C., Huang, J., Oen, A., Vojinovic, Z., Kuhlicke, C., & Pauleit, S. (2020). Stakeholder Mapping to Co-Create Nature-Based Solutions: Who Is on Board? *Sustainability 2020, Vol. 12, Page 8625*, *12*(20), 8625. https://doi.org/10.3390/SU12208625

Zwierzchowska, I., Fagiewicz, K., Poniży, L., Lupa, P., & Mizgajski, A. (2019). Introducing nature-based solutions into urban policy – facts and gaps. Case study of Poznań. *Land Use Policy*, *85*, 161–175.

Chapter II. Exploring the current understanding of nature-based solutions impacts in complex urban environments

II. Abstract

Complex urban environments are facing sustainability (i.e., long-term viability), resilience (i.e., capacity to adapt and recover from shocks) and equity (i.e., inclusive and just distribution of risks and benefits) challenges, with Nature-based solutions (NBS) emerging as potential instruments to address them. However, urban NBS often yield unexpected synergies and tradeoffs due to their multiple impacts, challenging existing evaluation frameworks. This chapter aims to explore the implications of NBS impacts in complex urban settings, linking them to urban challenges and proposing a vulnerability-focused approach to improve evaluation. It was found that urban challenges can be interconnected, and that NBS can either strengthen or weaken them depending on their planning and management. For example, NBS resilience practices can bolster a city's overall sustainability by providing stability and continuity in the face of unforeseen challenges. Yet, they may exacerbate patterns of unequal exposure to climate change impacts if resilient capacities are treated as private competitive advantages. Furthermore, the study also reveals that, to navigate these complexities, a vulnerability approach proves valuable. It allows for the consideration of both desired and undesired NBS impacts, providing a framework to integrate and anticipate changes in urban challenges. These findings underscore the need for more nuanced assessment methods to understand urban NBS synergies and tradeoffs, while highlighting a theoretical potential of vulnerability assessment to bridge this gap that is yet to be tested.

Keywords: Sustainability; Resilience; Equity; Nature-based solutions; Vulnerability; Cities; Urban

II.1. Introduction

Urban environments are complex systems (Batty, 2008; Bettencourt, 2013; Bettencourt et al., 2007) consisting of interconnected and interdependent components displaying intricate and sometimes unpredictable behaviors characterized by feedback loops and nonlinearity (McPhearson et al., 2016). Such complexity becomes an obstacle when trying to understand and govern urban systems, especially when tackling resilience challenges while achieving sustainable development (Rodríguez-Rodríguez et al., 2015). What is more, urban environments face increased pressures from social and environmental hazards, such as hurricanes, flooding, heatwaves, rapid urbanization and an aging infrastructure (IPCC, 2022). In many cases, impacts arising from such threats have been, and will likely continue to be, unevenly distributed among urban populations, as the economically vulnerable and socially marginalized endure the most severe consequences, as evidenced in the aftermath of wildfires (Davies et al., 2018), heatwaves (Gronlund, 2014) and hurricanes (Garciá-López, 2018).

Considering this, there is a need for addressing resilience, equity, and sustainability challenges in urban environments (Berbés-Blázquez et al., 2023; United Nations, 2015). Reaching these objectives demands transformative changes in form of deep and fundamental modifications in the human relationship with global ecological systems, in order not to exceed planetary boundaries while meeting human needs (Gillard et al., 2016; Patterson et al., 2017).

Nature-based solutions (NBS) have been promoted as a core instrument for addressing societal challenges such as climate change, natural disasters, food security, water security and social development (Bush & Doyon, 2019; Cohen-Shacham et al., 2016). Because of this, NBS have great potential when pursuing urban transformations that contribute to 'planetary well-being' (Kortetmäki et al., 2021) (i.e., maintaining Earth's integrity for species and humans to thrive). Furthermore, and unlike most technical solutions, NBS show strong capabilities in addressing several urban challenges simultaneously (Haase et al., 2017), while providing multiple co-benefits — or ecosystem services (ES) (Albert et al., 2020; Calliari et al., 2019; Loiseau et al., 2016; Song et al., 2019; Wendling et al., 2018).

Nonetheless, NBS exhibit a context-sensitive and a site-specific nature (Raparthi & Vedamuthu, 2022). This, coupled with the complexity of urban environments and the interconnectedness of its dynamics, results in the generation of synergies or tradeoffs among various impacts of NBS. For instance, urban agriculture offers an opportunity for the production of food in cities and the improvement of food security in periods of crisis by reducing the dependence on global food sources, therefore, creating more resilient cities (Barthel & Isendahl, 2013; Gulyas & Edmondson, 2021). At the same time, urban agriculture can be of low efficiency in terms of their use of material and labor resources, as shown by their emergy measures and benefit-to-cost ratios (McDougall et al., 2019), an aspect with disadvantageous implications from a sustainability point of view.

In this regard, there is still a need to further understand the extends urban NBS impacts (Kabisch et al., 2016; Rödl & Arlati, 2022) and the possible synergies and tradeoffs that can arise when trying to address urban challenges. Current evaluations predominantly focus on assessing the net impacts of NBS, often overlooking undesired effects and offering only limited insight into impacts beyond the immediate implementation sites. To build a solid evidence base for the performance of NBS in cities, it's vital to clearly define their impacts, understand synergies and trade-offs, and create efficient monitoring and evaluation methods (Dumitru et al., 2020). Embracing a more integrative approach would bolster urban policy makers' ability to gauge the effectiveness of NBS in addressing urban challenges, considering that urban regeneration planning is still dominated by silo-thinking, which tends to treat social and ecological challenges as distinct and occasionally in conflict with one another (Dumitru et al., 2020).

In this context, taking a vulnerability-focused approach could provide an additional dimension to the evaluation of the impact of NBS. The impact of NBS on urban vulnerabilities can either exacerbate or mitigate them, depending on the location and design of the NBS (Pereira et al., 2023). However, vulnerability assessments remain an underexplored avenue when it comes to assessing NBS, even though they have proven effective in strategically guiding urban decision-making and actions (Fekete, 2009; Nahiduzzaman et al., 2015; Rigillo & Cervelli, 2014). These assessments have the capacity to gauge the extent and distribution of the impacts caused by hazards while identifying disparities in the ability to cope with them (Baró et al., 2021). Such attributes can be instrumental in depicting the synergies and trade-offs resulting from the interplay between NBS and urban challenges.

Building upon the recognition of the complexity of urban environments and the urgent challenges they face, this chapter aims to grasp the extents of NBS impacts within these intricate urban landscapes and the way that these impacts can be understood from a vulnerability point of view. To achieve this, two main objectives are pursued: (1) to explore the multiple implications of NBS impacts in complex urban environments by describing how these relate to the urban challenges of Sustainability, Resilience and Equity, and (2) to examine how adopting a vulnerability-focused approach can improve the understanding of NBS impacts in complex urban environments.

II.2. Sustainability, resilience and equity as urban challenges and their relation to Nature-based solutions

The UN's Sustainable Development Goal 11 aims to create "inclusive, safe, resilient, and sustainable cities and human settlements". NBS have been recognized as a valuable strategy for advancing this objective due to their ability to address the fundamental challenges of sustainability, resilience, and equity in urban development (European Commission, 2019; United Nations, 2015, 2022). These challenges encompass different dimensions of urban complexity and play a pivotal role in shaping desired urban futures.

To better understand the relevance of these urban challenges, this section provides explicit definitions for each challenge, describes its core interactions and examines their relations with NBS impacts (see **Fig. 1**).

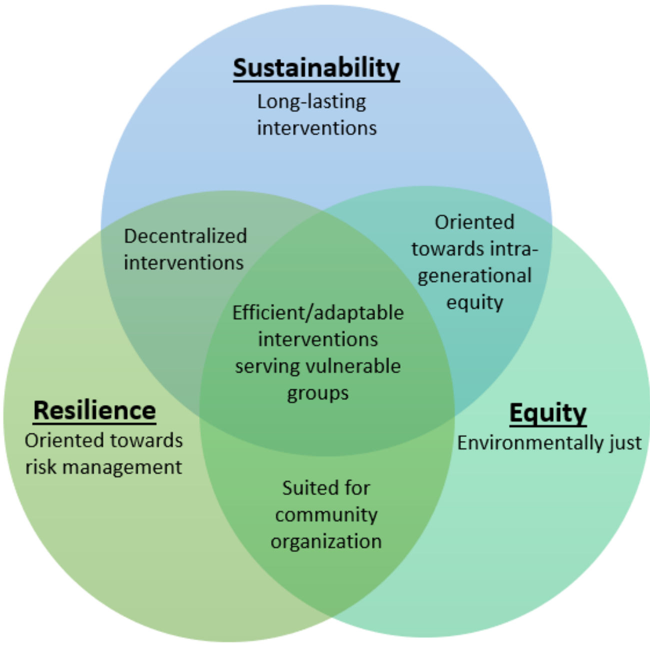


Figure 1. Examples of NBS characteristics supporting urban challenges

II.2.1. Sustainability

In the urban context, Sustainability can be described as an integrative and coevolutionary process among the city subsystems (economic, social, physical and environmental) securing the population with a long-lasting level of well-being without compromising the development of surrounding areas while reducing the harmful effect of society on earth's biosphere (Camagni, 1998). Metabolic perspectives have been employed to further explain urban sustainability, describing it as a state in which the utilization of resources and generation of waste by urban areas stay within the ecological carrying capacity of their support systems (including the broader Earth ecosystem), all the while guaranteeing the ability to uphold a satisfactory quality of life, as deemed acceptable by present and future societies (Davidson, 2010). For example, the vital energy, food, and water systems necessary to sustain current and future urban areas, along with the management of resulting waste, strongly underscore the pivotal role that cities should assume in sustainability efforts (Romero-Lankao et al., 2017). Sustainability is often treated as a normative concept, representing the vision for future urban environments and associated with efficient and optimized systems (Elmqvist et al., 2019).

NBS are often described as beneficial for improving sustainability in urban environments through the provision of benefits (Abson et al., 2014; Andersson et al., 2015; Commission & Innovation, 2015; Dorst et al., 2019). One compelling example is urban green spaces and their capacity for improving resource efficiency. They reduce the energy consumption required for cooling buildings by providing natural shade and lowering the urban heat island effect (Shao & Kim, 2022), thus decreasing greenhouse gas emissions and energy costs (Zhang et al., 2014). Additionally, green spaces act as natural air filters, enhancing air quality and reducing the health burdens associated with pollution (Gourdji, 2018; Speak et al., 2012). Moreover, green roofs and vegetated swales can efficiently capture and filter rainwater, reducing water consumption and minimizing the strain on municipal water supply systems (Shafique et al., 2018).

Nonetheless, NBS also produce unexpected impacts that should be understood in order to anticipate possible feedback loops affecting the sustainability of the system where these are developing (Frantzeskaki et al., 2021). For example, the selection of suitable tree species and genotypes are important factors for improving the lifespan and quality of trees in urban environments (Sæbø et al., 2003), but the selection could backlash if its associated side effects are not considered, such as the emission of biogenic volatile organic compounds and allergens (Cariñanos et al., 2020; Ren et al., 2014). In a different example, green roofs could improve the sustainability of cities by reducing the energy building demand (Susca et al., 2011), while construction materials of the green roofs, such as expanded clay, require high amounts of energy for its production (Tams et al., 2022), an aspect that can counteract the actual sustainability of green roofs.

II.2.2. Resilience

Resilience describes the capacity of an urban area to adapt and re-arrange itself when facing abrupt, gradual or chronic changes (Alberti & Marzluff, 2004; Pickett et al., 2004), prosper on those periods of stability while organizing and adapting when change or disruption times emerge (Gardner, 2019). Resilience is associated with the diversity and redundancies among the components providing services within a system. Unlike sustainability, resilience is a non-normative concept, and it could be desired and undesired (Elmqvist et al., 2019). Urban poverty, for example, might be considered as undesired resilience, while resilience against climate change is generally considered as desired resilience. Resilience is then understood as a buffer against undesired changes — whereby desire is context and preference dependent — that alter the social-ecological functioning of and human wellbeing urban environments.

Historically, many cities have endured disasters (natural and human-induced), displaying its resilience capacities (Bettencourt et al., 2007). However, in the Anthropocene era, cities face new risks from global environmental changes (Bai et al., 2018). The UN has noted that urban areas are increasingly vulnerable due to their high population density, infrastructure connectivity, and interdependence (United Nations, 2019). This makes them susceptible to cascading system failures during natural disasters, resulting in more significant fatalities and economic losses than rural areas (Elmqvist et al., 2019).

NBS have been praised for their potential contribution to building urban resilience (Bush & Doyon, 2019; Langemeyer et al., 2021). For example, green corridors crossing the urban areas and connecting green infrastructures provide multiple communications routes aside from the already existing ones (Zhang et al., 2019), therefore, improving resilience by bringing social connectivity. Infiltration trenches and bioswales can effectively mitigate the flooding effects of high-frequency precipitations (Huang et al., 2020), improving city's capacity for dealing with pluvial floodings. Furthermore, NBS also offer spaces for people to encounter and interact, thus promoting social learning (McPhearson et al., 2015), participation (Cohen-Shacham et al., 2019), recreation and social cohesion (Jennings & Bamkole, 2019). These elements can lead to processes of participation and local organization, which has been described as positive for improving the resilience of cities (Biggs et al., 2012; Quigley et al., 2018).

However, the misuse of NBS can result in maladaptation to the environments in which these are implemented (Seddon, 2022). NBS with low diversity value, such as those that do not involve native species or rely heavily in monocultures, are vulnerable to environmental changes in the long term, affecting its capacity to provide benefits (Seddon, 2022), and therefore, its contributions to improving resilience in both urban and non-urban environments. Another example describes that the capacity of woodlots for protecting homes from windstorms can be compromised if tree species are not able to withstand droughts and storms (Wood et al., 2017). Neglecting to take into account that NBS can themselves be vulnerable to the very environmental conditions they are intended to mitigate may undermine their effectiveness in enhancing resilience (Turner et al., 2022).

II.2.3. Equity

Equity is widely described as the even distribution of risks, vulnerabilities and benefits across demographic groups, space and time (Hampton, 1999; Harlan et al., 2006; Langemeyer & Connolly, 2020; Moreno-Jiménez et al., 2016; Tonne et al., 2018). Equity is usually understood under three principles: distributional, recognitional and procedural (Schlosberg, 2004). Distributional equity focuses on the equitable distribution of material resources to all individuals within a society (Meerow et al., 2019). Social goods may be distributed to explicitly improve the welfare of the disadvantaged (Schlosberg, 2007), therefore differing with equalitarianism, which focus on the equal allocation of goods for all people. Recognitional injustices involve social structures (e.g., norms, beliefs, language) that give rise to disparities among groups and govern uneven resource allocations (Dumm, 1992). Lastly, procedural equity is described as the fairness and inclusivity of the methods employed by those in power to attain particular results or make decisions (Hanson & Alkan Olsson, 2022). As described by Meerow et al. (2019), procedural equity is closely connected to both recognitional and distributional equity, as an individual or group's membership and participation in decision-making is integral to the equitable distribution of material goods. Without procedures of recognition, an individual or group is unable to participate in the community; without such participation, their unique needs for social goods cannot be recognized either. Examples of inequities include the way that specific urban areas have higher flooding risks (Norman et al., 2012) or poor air quality (Grineski et al., 2007), and that some populations, such as children, are more vulnerable to environmental injustices (Landrigan et al., 2010).

When talking about NBS relation to equity, the literature has mostly focused on the distribution of urban vegetation and its associated benefits (Nesbitt et al., 2018). According to Nesbitt, this collection of research emphasizes the exploration and comprehension of the interconnections between city greenery and economic aspects. Examples of distributional approaches to NBS include the identification areas that might benefit from NBS implementation (e.g., Baró et al., 2021; Li et al., 2020; Meerow & Newell, 2017) or the anticipated provision of ecosystem services from NBS in regions with high ecosystem service demand (Baró et al., 2019; Langemeyer et al., 2020; Venter et al., 2021). The distributional approach typically presupposes that urban vegetation represents desirable or at the very least, harmless assets or advantages, and that limited accessibility signifies the existence of inequality.

Nonetheless, green equity can also be comprehended through the lenses of procedural recognitional equity. Examples of procedural and recognitional inequities has been observed in Baltimore (USA) and small towns in South Africa, where current distributions of urban vegetation and access to it has been related to the lack of access to financial and social services of minority communities in the past (Boone et al., 2009; Matthew McConnachie & Shackleton, 2010). However, procedural and recognitional aspects of equity have not been considered as thoroughly as the distributional approach when planning for NBS (Kato-Huerta & Geneletti, 2023).

II.2.4. Urban challenges: core interactions

As discussed in the previous section, each of the challenges posed by Sustainability, Resilience, and Equity, approaches urban issues from distinct perspectives. Within this context, urban interventions like NBS, which aim to enhance urban conditions, can cause both tradeoffs and synergies among these challenges. The outcome depends on whether these interventions are planned holistically, considering their potential impacts on all these challenges, or if they are designed solely to address one of them. In the following section, we will explore some of the fundamental interactions that can emerge between these urban challenges, illustrating how NBS interventions can influence the intricate dynamics of urban environments.

II.2.4.1. Resilience and Sustainability

Resilience and sustainability are two concepts that have been widely described in urban regeneration programs (Elmqvist et al., 2019). Still, these two concepts are often used interchangeably in international policy documents (Elmqvist, 2017). For example, in the New Urban Agenda (United Nations, 2017), resilience and sustainability are often employed in the same sentence. Even though described as different concepts, they seem to be presented as positively correlated (Elmqvist et al., 2019), even when this may not be the case. An example of how these two approaches differ is on how they portray the governance of urban practices. On one hand, sustainability calls for top down governance practices (Aina et al., 2019) to enforce regulations and promote urban practices that ensure consistency in achieving sustainability targets across the entire urban area. Resilience, on the other hand, calls for bottom-up and local scale governance (Mehmood, 2015) as means for local communities to customize strategies and responses to the specific challenges they face, allowing flexibility and adaptation to changing urban conditions. When considering NBS, a sustainability approach rooted in top-down governance can be illustrated by initiatives like green roof incentives implemented by Brussels municipality (City of Brussels, 2021). In contrast, a more resilient governance focused NBS approach could entail the establishment of locally managed urban community gardens.

Moreover, sustainability practices call for high efficiency, optimal systems while resilience calls for redundancy, which involves an excess in capacity and back-up systems that allow a city to maintain its functionality in case of crisis (Elmqvist, 2017). As mentioned before, these two aspects, if not examined carefully, can be at odds with one another, or, in contrast, complement each other if resilience supports a path for achieving sustainability targets (Elmqvist et al., 2019).

In this same line, it is important to understand that sustainability and resilience do not exclude each other. For example, while sustainability practices emphasize high efficiency and optimal systems, resilience, with its focus on redundancy and backup systems, can enhance the overall sustainability of a city by providing stability and continuity in the face of unexpected challenges (Elmqvist et al., 2019). For instance, an NBS such as green roofs help to reduce energy consumption by insulating buildings and regulating indoor temperatures, leading to lower energy consumption (Jaffal et al., 2012), which positively impacts sustainability of city and, in periods of energy shortage, can also enhance a city's resilience. Moreover, green roofs effectively manage rainfall, alleviating stress on traditional stormwater systems during floods. This not only improves resilience but also reduces the necessity for expanding and maintaining conventional stormwater infrastructures, thereby contributing to sustainability. This illustrates how urban practices can address multiple challenges by enhancing resource efficiency and fortifying a city's capacity to adapt to and recover from unforeseen adversities.

II.2.4.2. Sustainability and Equity

Scholars argue that achieving genuine sustainability in urban development requires the comprehensive mitigation of various forms of inequities (Dsouza et al., 2023). They contend that sustainability endeavors that fail to prioritize equity will only prolong economic and social disparities, as well as the unequal distribution of power and political dynamics (Meerow et al., 2019). The notion of equity is inherently inferred in worldwide interpretations of sustainability. For instance, the Brundtland (1987) definition, which revolves around "satisfying the needs of the current generation without jeopardizing the ability of future generations to fulfill their own needs," relies on the principle of intergenerational fairness (Stavins et al., 2003). Nevertheless, the integration of equity into sustainable urban development initiatives is not consistent, primarily because there is a stronger emphasis on end results rather than the equitable distribution of these outcomes (Bozeman et al., 2022; Dsouza et al., 2023; Nesbitt et al., 2019; Yeeles et al., 2018). For example, implementing policies to promote clean energy sources, like solar power or wind turbines, can reduce greenhouse gas emissions and enhance environmental sustainability. However, the upfront costs of such technologies may be prohibitive for disadvantaged communities, leading to an unequal distribution of benefits (Sovacool et al., 2022). From a NBS perspective, redeveloping urban areas to enhance sustainability, such as creating green spaces, can lead to rising property values and gentrification (Anguelovski et al., 2022). While these improvements contribute to a more sustainable urban environment, they may displace lower-income residents, exacerbating inequality and social exclusion.

Moreover, the sustainability and equity relationship can also be expanded outside of the physical urban environments if considering *urban land teleconnections* (Seto et al., 2012), which refers to the flow of economic goods, people, services and land use change that drive and respond to urbanization and cannot be solved in a single geographical location. This aspect is also relevant for NBS, as they do not only influence the conditions of urban environments where they are being developed but also its surroundings and even distant locations — a notion that has become increasingly relevant as economies became more globalized. An example of this is the demand of water by urban vegetation (Darrel Jenerette et al., 2011; Pataki et al., 2011) a major trade-off in water-limited environments where urban, agricultural, and in-stream uses compete for water that will be further exacerbated by future droughts caused by climate change (Liang et al., 2017).

The existing tensions between urban sustainability and equity do not necessarily prevent potential synergies (see **Fig. 1**) For example, promoting urban agriculture and community gardens in underserved neighborhoods can increase access to fresh, locally grown food (Wang et al., 2014). Simultaneously, these initiatives can improve the urban sustainability by reducing the energy required for transportation of fresh produce (Zumkehr & Campbell, 2015) and food losses due to long supply chains (Tonini et al., 2022).

The relationship between urban sustainability and equity highlights the critical importance of addressing both environmental and social dimensions in urban development. By doing so, it is possible to navigate the complexities of urban sustainability and equity to create inclusive and sustainable urban environments.

II.2.4.3. Resilience and Equity

At global scale, low-income countries are usually more vulnerable, for instance, to climate change related drought and flooding (Füssel, 2010), while showing lower capacities to benefit from resilient practices such as climate adaptation and mitigation efforts (Anguelovski et al., 2016). At urban level, Shokry et al. (2018) highlights that resilience building in urban environments is focused mostly on addressing vulnerabilities to natural disasters, leaving aside the fact that many of these vulnerabilities are created and maintained by longstanding unsustainable and unjust urban development patterns, while disadvantaged groups often lack the political power to promote resilient solutions to environmental threats (Raddatz & Mennis, 2013). For example, attempts aimed at improving flood resilience in urban areas such as Manila and Jakarta have resulted in the compulsory displacement of informal settlements, causing disruptions to the residents' means of making a living (Anguelovski et al., 2016; Meerow, 2017).

When implementing NBS, such tensions can also be found. In cases where NBS planning fails to counteract for inequalities, preexisting market mechanisms might even exacerbate the existing patterns of unequal exposure to climate change impacts and other social-ecological challenges (Anguelovski et al., 2016; Whitehead, 2013). Examples of such tensions can be found in the private development of flood-protected “resilience zones” as described for the city of Boston, USA (Brugmann, 2014), where real state firms are engaging in the flood risk management by employing NBS, and treat resilience as a competitive advantage, risking to further increase uneven exposure to risks (Teicher, 2018). In Detroit, USA, city greening strategies foster the implementation of green roofs primarily in the affluent areas of Detroit's urban core in order to mitigate the heat island effect, while a large proportion of low-income residents do not have accessible green infrastructures within walking distance (Sanchez & Reames, 2019).

In response to these dynamics, resilience planning in cities is paying more attention to equity considerations. In a study performed by Fastiggi et al. (2020), it was found that out of 20 north American cities, 19 were considering equity and prioritizing vulnerable communities in some capacity within their urban resilience governance. In a related context, White-Newsome & Slay (2022) elaborate on the substantial disparities faced by low-income communities in the US, experiencing notably heightened impacts attributable to climate change compared to other demographic groups. In response to this pressing issue, non-governmental organizations and community-based organizations have mobilized to implement resilience planning initiatives (White-Newsome & Slay, 2022). These efforts aim to tackle urban flooding by fostering leadership capacity in the realms of climate and equity. The overarching goal is to propel equitable planning strategies that empower low-income communities to effectively adapt to the challenges posed by urban flooding.

In this context, NBS can enhance urban resilience and reduce inequalities when explicitly tailored to the specific needs of communities (Meerow & Newell, 2017). By promoting equitable impact distribution (Langemeyer & Connolly, 2020; Meerow et al., 2019) and ensuring meaningful participation in decision-making, along with acknowledging social, cultural, and political differences (Meerow et al., 2019), NBS can effectively address both technical and social dimensions of urban resilience. For instance, creating flood mitigation strategies, like rain gardens, bioretention basins, and green roofs, can be made equitable by assessing the most vulnerable areas for flood extremes. This involves considering not only the exposed areas to flooding, but also by evaluating the presence of sensitive populations. This approach ensures a comprehensive consideration of distributional issues. In addition, and as for the consideration of recognitional and recognitional equity, NBS planners could acknowledge local dynamics and engage in participative and co-creative approaches involving stakeholders or representatives of different backgrounds associated with the targeted areas (Sekulova et al., 2021).

Equitable considerations are essential when striving to enhance the resilience of a city. If resilience efforts are not distributed fairly in an urban environment, it can give rise to scenarios of inequity, ultimately compromising resilience in the long run (Meerow & Newell, 2019).

II.3. Vulnerability assessments: an opportunity for better evaluating Nature-based solutions in urban environments

As previously stated in Chapter I, there is a lack of understanding regarding the overall impacts of NBS including its possible desired and undesired impacts, both within and beyond urban limits. The review of urban challenges has provided a further overview of the possible synergies and tradeoffs involved in the functioning of urban NBS, highlighting the necessity to better comprehend these in order for a better strategic and effective planning of NBS.

For this matter, a vulnerability approach, where vulnerability is defined as the susceptibility to harm (Cutter, 2016; Liverman, 2001) of both social and ecological systems (Pörtner et al., 2022), is a valuable concept to consider for a better understanding the possible impacts of urban NBS. To start, vulnerability approaches have already been employed for assessing the complexity of urban environments. For instance, Fistola et al. (2020) propose understanding urban systems through the state, interactions, and imbalance of urban risks, which together derive integrated vulnerabilities. This approach allows for the development of strategies that improve urban resilience. Furthermore, vulnerability assessments have been employed to study the demand for ecosystem services provided by NBS in urban environments, with a focus on equity aspects (Baró et al., 2016; Meerow & Newell, 2017). Such a strategy aims to identify hotspots where implementing NBS could effectively address the greatest number of vulnerabilities simultaneously.

These methodologies provide evidence that vulnerability assessments are valuable for assessing dynamics within complex urban environments and are worth further exploration in understanding the impacts that NBS could have in these contexts. This effort aims to address the existing gap in NBS assessment regarding the lack of recognition of potential synergies and tradeoffs that may arise in complex urban environments

A way forward in this direction entails identifying urban vulnerabilities directly associated with challenges and susceptible to influence by NBS. As mentioned earlier, this perspective is already evident in studies examining the demand for ecosystem services in urban environments. Expanding this existing framework would involve considering not only the demand for ecosystem services but also the impacts of NBS on vulnerabilities and, consequently, on urban challenges. While existing assessments of NBS are already considering urban challenges, such as resilience and sustainability (e.g., Jeuken et al., 2020; Maia et al., 2021) these lack a more nuanced consideration of possible synergies and tradeoffs among them, nor have explored how these may be affecting general vulnerabilities.

Additionally, adopting a vulnerability-oriented approach can support the comprehensive evaluation of various impacts of urban NBS. This approach enables the depiction of multiple dynamics that arise from the interaction of NBS with urban environments, offering insights into potential trade-offs and synergies. As a way of example, when analyzing the thermal regulation effects of NBS, focusing on vulnerabilities involves comparing existing urban temperatures (considered a hazard) with projected temperatures post-NBS implementation. Likewise, when evaluating the creation of spaces for social cohesion within a city, a vulnerability approach would encompass factors such as the scarcity of public spaces areas for social encounters and the potential of NBS to enhance these. By examining these potential shifts, we can gain a deeper understanding of the capabilities and efficacy of urban NBS.

Beyond the consideration of multiple NBS impacts, a vulnerability framing would allow for its integration by relating them to a shared concept. For instance, given the difficulties in understanding of urban conditions through the joint understanding of ecosystem service and urban metabolism approach (Elliot et al., 2019), a vulnerability assessment could help integrate different NBS impacts by simplifying its interpretation: instead of assessing NBS impacts from its type of impact (e.g., ecosystem service or metabolic change), these could be assessed simply by looking at which vulnerabilities are affecting, and how much are these being affected. Such integration could be supported by stakeholder participation, revealing context-specific root causes of vulnerabilities (Schneiderbauer et al., 2017), expanding the assessment beyond quantitative measurements (Salter et al., 2010) and promoting equity aspects around NBS discussion (Sekulova et al., 2021). An approach of this nature would provide a common ground to simultaneously assess a wider range of impacts arising from the implementation and functioning of NBS.

Furthermore, undesired impacts of NBS can be considered as increased vulnerabilities due to the implementation of NBS. An example of this occurs in NBS policies that utilize afforestation with non-native monocultures. This practice may lead to a decrease in biodiversity conditions, as it lacks the diversity of plant species typically found in natural environments and displaces native species (Seddon et al., 2020), consequently increasing the vulnerability to biodiversity loss. In addition, and considering that undesired effects are not limited to urban environments, further equity and sustainability considerations could be integrated by assessing the impacts of NBS implementations beyond city limits. For instance, while the installation of green roofs might aid in sequestering greenhouse gases within the urban areas where they are situated (Konopka et al., 2021), its construction and transportation to urban areas could contribute to greenhouse gas emissions outside of city limits, thereby yielding a nuanced effect on the global vulnerability to climate change. In this context, achieving a clearer and shared understanding of the undesired and cross-scale impacts of NBS would facilitate assessing potential trade-offs (Pereira et al., 2023; Seddon et al., 2020). This aspect has been recognized as a limitation of sustainability efforts in urban environments (Bozeman et al., 2022; Dsouza et al., 2023), and its consideration provides a chance to challenge the discourse of maximizing green spaces in urban areas (Roman et al., 2020).

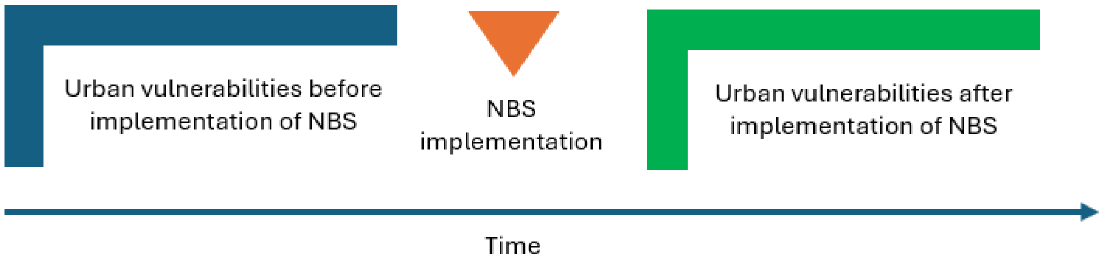


Figure 2. Shift in urban vulnerabilities before and after the implementation of Nature-based Solutions. Figure adapted from Dumitru & Lourido (2022)

Finally, and according to European Commission (2021), an appropriate impact evaluation method for NBS should clearly state and use reference conditions and baseline data for comparison in order to determine change(s) attributable to NBS implementation (see **Fig. 2**). Shifting vulnerabilities can provide insights on how urban conditions would look like before and after the implementation of NBS. Its effectiveness is evident from the limited instances in which such an approach has been implemented to observe the change in individual vulnerabilities following the creation of NBS (e.g., López-Valencia, 2019). Employing this strategy with the consideration of the spatially explicit characteristics of NBS impacts would expand the assessment of NBS beyond the evaluation of net impacts, thereby enabling a nuanced understanding of NBS impacts across space and time.

II.4. Conclusions

This chapter aims to enhance the understanding of NBS within complex urban environments and provide a way forward in better assessing their impacts. The analysis illustrates how the key urban challenges of sustainability, resilience, and equity can be interconnected, and how NBS can either strengthen or weaken them depending on their planning and management, producing both synergies and tradeoffs.

In this context, a thorough assessment of NBS portraying such dynamics could contribute to maximizing the desired effects and minimizing undesired impacts. To address this, it is proposed to adopt a vulnerability approach for NBS assessment, considering that vulnerability premises have already been employed to evaluate complex urban environments and are partially related to the assessment of NBS.

Vulnerability assessments can incorporate many desired traits for NBS evaluation that allow for a better understanding of NBS tradeoffs and synergies across spatial scales, such as considering and integrating multiple impacts, assessing both desired and undesired effects, and anticipating changes in urban conditions. However, it is important to note that the favorable outcomes of such an approach are yet to be confirmed, as it has not been thoroughly tested.

By providing a nuanced understanding of the complexities surrounding urban NBS impacts and advocating for more comprehensive evaluations, this chapter lays the theoretical foundations for the subsequent development of the integrated NBS-vulnerability framework, which will be developed in the following chapters of this thesis.

II. References

Abson, D. J., von Wehrden, H., Baumgärtner, S., Fischer, J., Hanspach, J., Härdtle, W., Heinrichs, H., Klein, A. M., Lang, D. J., Martens, P., & Walmsley, D. (2014). Ecosystem services as a boundary object for sustainability. *Ecological Economics*, *103*, 29–37. https://doi.org/10.1016/J.ECOLECON.2014.04.012

Aina, Y. A., Wafer, A., Ahmed, F., & Alshuwaikhat, H. M. (2019). Top-down sustainable urban development? Urban governance transformation in Saudi Arabia. *Cities*, *90*, 272–281. https://doi.org/10.1016/J.CITIES.2019.03.003

Albert, C., Brillinger, M., Guerrero, P., Gottwald, S., Henze, J., Schmidt, S., Ott, E., & Schröter, B. (2020). Planning nature-based solutions: Principles, steps, and insights. *Ambio*. https://doi.org/10.1007/s13280-020-01365-1

Alberti, M., & Marzluff, J. M. (2004). Ecological resilience in urban ecosystems: Linking urban patterns to human and ecological functions. *Urban Ecosystems*, *7*(3), 241–265. https://doi.org/10.1023/B:UECO.0000044038.90173.c6

Andersson, E., Tengö, M., McPhearson, T., & Kremer, P. (2015). Cultural ecosystem services as a gateway for improving urban sustainability. *Ecosystem Services*, *12*, 165–168. https://doi.org/10.1016/J.ECOSER.2014.08.002

Anguelovski, I., Connolly, J. J. T., Cole, H., Garcia-Lamarca, M., Triguero-Mas, M., Baró, F., Martin, N., Conesa, D., Shokry, G., del Pulgar, C. P., Ramos, L. A., Matheney, A., Gallez, E., Oscilowicz, E., Máñez, J. L., Sarzo, B., Beltrán, M. A., & Minaya, J. M. (2022). Green gentrification in European and North American cities. *Nature Communications 2022 13:1*, *13*(1), 1–13. https://doi.org/10.1038/s41467-022-31572-1

Anguelovski, I., Shi, L., Chu, E., Gallagher, D., Goh, K., Lamb, Z., Reeve, K., & Teicher, H. (2016). Equity Impacts of Urban Land Use Planning for Climate Adaptation: Critical Perspectives from the Global North and South. *Journal of Planning Education and Research, 36(3), 333-348*. https://doi.org/10.1177/0739456X16645166

Bai, X., Dawson, R. J., Ürge-Vorsatz, D., Delgado, G. C., Salisu Barau, A., Dhakal, S., Dodman, D., Leonardsen, L., Masson-Delmotte, V., Roberts, D. C., & Schultz, S. (2018). Six research priorities for cities and climate change. *Nature*, *555*(7694), 23–25. https://doi.org/10.1038/d41586-018-02409-z

Baró, F., Calderón-Argelich, A., Langemeyer, J., & Connolly, J. J. T. (2019). Under one canopy? Assessing the distributional environmental justice implications of street tree benefits in

Barcelona. *Environmental Science & Policy*, *102*, 54–64. https://doi.org/10.1016/J.ENVSCI.2019.08.016

Baró, F., Camacho, D. A., Pérez Del Pulgar, C., Triguero-Mas, M., & Anguelovski, I. (2021). School greening: Right or privilege? Examining urban nature within and around primary schools through an equity lens. *Landscape and Urban Planning*, *208*, 104019. https://doi.org/10.1016/J.LANDURBPLAN.2020.104019

Baró, F., Palomo, I., Zulian, G., Vizcaino, P., Haase, D., & Gómez-Baggethun, E. (2016). Mapping ecosystem service capacity, flow and demand for landscape and urban planning: A case study in the Barcelona metropolitan region. *Land Use Policy*, *57*, 405–417. https://doi.org/10.1016/J.LANDUSEPOL.2016.06.006

Barthel, S., & Isendahl, C. (2013). Urban gardens, agriculture, and water management: Sources of resilience for long-term food security in cities. *Ecological Economics*, *86*, 224–234. https://doi.org/10.1016/J.ECOLECON.2012.06.018

Batty, M. (2008). The Size, Scale, and Shape of Cities. *Science*, *319*(5864), 769–771. https://doi.org/10.1126/SCIENCE.1151419

Berbés-Blázquez, M., Cook, E. M., Grimm, N. B., Iwaniec, D. M., Mannetti, L. M., Muñoz-Erickson, T. A., & Wahl, D. (2023). Assessing resilience, equity, and sustainability of future visions across two urban scales. *Sustainability Science*, *18*(6), 2549–2566. https://doi.org/10.1007/S11625-023-01396-Z/FIGURES/9

Bettencourt, L. M. A. (2013). The origins of scaling in cities. *Science*, *340*(6139), 1438–1441. https://doi.org/10.1126/SCIENCE.1235823/SUPPL_FILE/BETTENCOURT.SM.PDF

Bettencourt, L. M. A., Lobo, J., Helbing, D., Kühnert, C., & West, G. B. (2007). Growth, innovation, scaling, and the pace of life in cities. *Proceedings of the National Academy of Sciences of the United States of America*, *104*(17), 7301–7306. https://doi.org/10.1073/PNAS.0610172104/SUPPL_FILE/INDEX.HTML

Biggs, R., Schlüter, M., Biggs, D., Bohensky, E. L., Burnsilver, S., Cundill, G., Dakos, V., Daw, T. M., Evans, L. S., Kotschy, K., Leitch, A. M., Meek, C., Quinlan, A., Raudsepp-Hearne, C., Robards, M. D., Schoon, M. L., Schultz, L., & West, P. C. (2012). *Toward Principles for Enhancing the Resilience of Ecosystem Services.* Annual Review Of Environment And Resources Volume 37, 2012. https://doi.org/10.1146/annurev-environ-051211-123836

Boone, C. G., Buckley, G. L., Grove, J. M., & Sister, C. (2009). Parks and People: An Environmental Justice Inquiry in Baltimore, Maryland. Annals of the Association of American Geographers, 99(4), 767–787. https://doi.org/10.1080/00045600903102949

Bozeman, J. F., Nobler, E., & Nock, D. (2022). A Path Toward Systemic Equity in Life Cycle Assessment and Decision-Making: Standardizing Sociodemographic Data Practices. *Environmental Engineering Science*, *39*(9), 759–769. https://doi.org/10.1089/EES.2021.0375/ASSET/IMAGES/EES.2021.0375_FIGURE1.JPG

Brugmann, J. (2014). *Building resilient cities: from risk assessment to redevelopment*. Ceres.

Brundtland, G. (1987). *Report of the World Commission on Environment and Development: Our Common Future*. https://sustainabledevelopment.un.org/content/documents/5987our-common-future.pdf

Bush, J., & Doyon, A. (2019). Building urban resilience with nature-based solutions: How can urban planning contribute? *Cities*, *95*, 102483. https://doi.org/10.1016/J.CITIES.2019.102483

Calliari, E., Staccione, A., & Mysiak, J. (2019). An assessment framework for climate-proof nature-based solutions. *Science of The Total Environment*, *656*, 691–700. https://doi.org/10.1016/J.SCITOTENV.2018.11.341

Camagni, R. (1998). Sustainable urban development: Definition and reasons for a research programme. *International Journal of Environment and Pollution*, *10*(1), 6–26. https://doi.org/10.1504/IJEP.1998.002228

Cariñanos, P., Ruiz-Peñuela, S., Valle, A. M., & de la Guardia, C. D. (2020). Assessing pollination disservices of urban street-trees: The case of London-plane tree (Platanus x hispanica Mill. ex Münchh). *Science of The Total Environment*, *737*, 139722. https://doi.org/10.1016/J.SCITOTENV.2020.139722

City of Brussels (2021). *Bonus for a green roof.* Brussels. https://www.brussels.be/bonus-green-roof

Cohen-Shacham, E., Andrade, A., Dalton, J., Dudley, N., Jones, M., Kumar, C., Maginnis, S., Maynard, S., Nelson, C. R., Renaud, F. G., Welling, R., & Walters, G. (2019). Core principles for successfully implementing and upscaling Nature-based Solutions. *Environmental Science & Policy*, *98*, 20–29. https://doi.org/10.1016/J.ENVSCI.2019.04.014

Cohen-Shacham, E., Janzem, C., Walters, G., & Maginnis, S. (2016). Nature-based solutions to address global societal challenges. In *Nature-based solutions to address global societal challenges* (IUCN, 2016). IUCN International Union for Conservation of Nature. https://doi.org/10.2305/iucn.ch.2016.13.en

Commission, E., & Innovation, D.-G. for R. and. (2015). *Towards an EU research and innovation policy agenda for nature-based solutions & re-naturing cities : final report of the Horizon 2020 expert group on 'Nature-based solutions and re-naturing cities': (full version).* Publications Office. https://doi.org/doi/10.2777/479582

Cutter, S. L. (2016). Resilience to What? Resilience for Whom? *The Geographical Journal*, *182*(2), 110–113. https://doi.org/10.1111/GEOJ.12174

Darrel Jenerette, G., Harlan, S. L., Stefanov, W. L., & Martin, C. A. (2011). Ecosystem services and urban heat riskscape moderation: water, green spaces, and social inequality in Phoenix, USA. *Ecological Applications*, *21*(7), 2637–2651. https://doi.org/10.1890/10-1493.1

Davidson, D. J. (2010). The Applicability of the Concept of Resilience to Social Systems: Some Sources of Optimism and Nagging Doubts. Society and Natural Resources, 23(12), 1135–1149. https://doi.org/10.1080/08941921003652940

Davies, I. P., Haugo, R. D., Robertson, J. C., & Levin, P. S. (2018). The unequal vulnerability of communities of color to wildfire. *PLOS ONE*, *13*(11), e0205825. https://doi.org/10.1371/JOURNAL.PONE.0205825

Dorst, H., van der Jagt, A., Raven, R., & Runhaar, H. (2019). Urban greening through nature-based solutions – Key characteristics of an emerging concept. *Sustainable Cities and Society*, *49*, 101620. https://doi.org/10.1016/J.SCS.2019.101620

Dsouza, N., Devadason, A., Senerat, A. M., Watanatada, P., Rojas-Rueda, D., & Sebag, G. (2023). Sustainability and Equity in Urban Development (S&EUD): A Content Analysis of "Bright

Spots" from the Accelerating City Equity (ACE) Project. *Sustainability (Switzerland)*, *15*(9), 7318. https://doi.org/10.3390/SU15097318/S1

Dumitru, A., Frantzeskaki, N., & Collier, M. (2020). Identifying principles for the design of robust impact evaluation frameworks for nature-based solutions in cities. *Environmental Science & Policy*, *112*, 107–116. https://doi.org/10.1016/J.ENVSCI.2020.05.024

Dumitru, A., & Lourido, D. T. (2022). Impact assessment guidebook. Connecting Nature Guidebook

Dumm, T. L. (1992). Justice and the Politics of Difference. By Iris Marion Young. Princeton: Princeton University Press, 1990. 286p. $45.00 cloth, $10.95, paper. *American Political Science Review*, *86*(2), 520–521. https://doi.org/DOI: 10.2307/1964259

Elliot, T., Almenar, J. B., Niza, S., Proença, V., & Rugani, B. (2019). Pathways to Modelling Ecosystem Services within an Urban Metabolism Framework. *Sustainability 2019, Vol. 11, Page 2766*, *11*(10), 2766. https://doi.org/10.3390/SU11102766

Elmqvist, T. (2017). Development: Sustainability and resilience differ. *Nature*, *546*(7658), 352.

Elmqvist, T., Andersson, E., Frantzeskaki, N., McPhearson, T., Olsson, P., Gaffney, O., Takeuchi, K., & Folke, C. (2019). Sustainability and resilience for transformation in the urban century. *Nature Sustainability*, *2*(4), 267–273. https://doi.org/10.1038/s41893-019-0250-1

European Commission. (2019). *The European green deal*.

European Commission. (2021). *Evaluating the impact of nature-based solutions: A handbook for practitioners*.

Fastiggi, M., Meerow, S., & Miller, T. R. (2020). Governing urban resilience: Organisational structures and coordination strategies in 20 North American city governments. Urban Studies, 58(6), 1262-1285. https://doi.org/10.1177/0042098020907277

Fekete, A. (2009). Validation of a social vulnerability index in context to river-floods in Germany. *Natural Hazards and Earth System Sciences*, *9*(2), 393–403. https://doi.org/10.5194/NHESS-9-393-2009

Fink, H. S. (n.d.). *Human-Nature for Climate Action: Nature-Based Solutions for Urban Sustainability*. https://doi.org/10.3390/su8030254

Fistola, R., Gargiulo, C., & La Rocca, R. A. (2020). Rethinking vulnerability in city-systems: A methodological proposal to assess "urban entropy." *Environmental Impact Assessment Review*, *85*, 106464. https://doi.org/10.1016/J.EIAR.2020.106464

Frantzeskaki, N., McPhearson, T., & Kabisch, N. (2021). Urban sustainability science: prospects for innovations through a system's perspective, relational and transformations' approaches: This article belongs to Ambio's 50th Anniversary Collection. Theme: Urbanization. *Ambio*, *50*(9), 1650–1658. https://doi.org/10.1007/s13280-021-01521-1

Füssel, H. M. (2010). How inequitable is the global distribution of responsibility, capability, and vulnerability to climate change: A comprehensive indicator-based assessment. *Global Environmental Change*, *20*(4), 597–611. https://doi.org/10.1016/J.GLOENVCHA.2010.07.009

Garciá-López, G. A. (2018). The Multiple Layers of Environmental Injustice in Contexts of (Un)natural Disasters: The Case of Puerto Rico Post-Hurricane Maria. Environmental Justice, 11(3), 101-108. https://doi.org/10.1089/ENV.2017.0045

Gardner, J. (2019). The inclusive healthy places framework: A new tool for social resilience and public infrastructure. *Biophilic Cities Journal*, *2*(2), 10–15.

Gillard, R., Gouldson, A., Paavola, J., & Van Alstine, J. (2016). Transformational responses to climate change: beyond a systems perspective of social change in mitigation and adaptation. *WIREs Clim Change*, *7*, 251–265. https://doi.org/10.1002/wcc.384

Gourdji, S. (2018). Review of plants to mitigate particulate matter, ozone as well as nitrogen dioxide air pollutants and applicable recommendations for green roofs in Montreal, Quebec. *Environmental Pollution*, *241*, 378–387. https://doi.org/10.1016/J.ENVPOL.2018.05.053

Grineski, S., Bolin, B., & Boone, C. (2007). Criteria Air Pollution and Marginalized Populations: Environmental Inequity in Metropolitan Phoenix, Arizona*. *Social Science Quarterly*, *88*(2), 535–554. https://doi.org/10.1111/J.1540-6237.2007.00470.X

Gronlund, C. J. (2014). Racial and socioeconomic disparities in heat-related health effects and their mechanisms: a review. *Current Epidemiology Reports*, *1*(3), 165. https://doi.org/10.1007/S40471-014-0014-4

Gulyas, B. Z., & Edmondson, J. L. (2021). Increasing City Resilience through Urban Agriculture: Challenges and Solutions in the Global North. *Sustainability 2021, Vol. 13, Page 1465*, *13*(3), 1465. https://doi.org/10.3390/SU13031465

Haase, D., Kabisch, S., Haase, A., Andersson, E., Banzhaf, E., Baró, F., Brenck, M., Fischer, L. K., Frantzeskaki, N., Kabisch, N., Krellenberg, K., Kremer, P., Kronenberg, J., Larondelle, N., Mathey, J., Pauleit, S., Ring, I., Rink, D., Schwarz, N., & Wolff, M. (2017). Greening cities – To be socially inclusive? About the alleged paradox of society and ecology in cities. *Habitat International*, *64*, 41–48. https://doi.org/10.1016/J.HABITATINT.2017.04.005

Hampton, G. (1999). Environmental equity and public participation. *Policy Sciences*, *32*(2), 163–174. https://doi.org/10.1023/A:1004591620163

Hanson, H. I., & Alkan Olsson, J. (2022). The Link Between Urban Green Space Planning Tools and Distributive, Procedural and Recognition Justice. *Human-Nature Interactions*, 285–295. https://doi.org/10.1007/978-3-031-01980-7_23

Harlan, S. L., Brazel, A. J., Prashad, L., Stefanov, W. L., & Larsen, L. (2006). Neighborhood microclimates and vulnerability to heat stress. *Social Science & Medicine*, *63*(11), 2847–2863. https://doi.org/10.1016/J.SOCSCIMED.2006.07.030

Huang, Y., Tian, Z., Ke, Q., Liu, J., Irannezhad, M., Fan, D., Hou, M., & Sun, L. (2020). Nature-based solutions for urban pluvial flood risk management. *Wiley Interdisciplinary Reviews: Water*, *7*(3), e1421. https://doi.org/10.1002/WAT2.1421

IPCC, 2022: Climate Change 2022: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [H.-O. Pörtner, D.C. Roberts, M. Tignor, E.S. Poloczanska, K. Mintenbeck, A. Alegría, M. Craig, S. Langsdorf, S. Löschke, V. Möller, A. Okem, B. Rama (eds.)]. Cambridge University Press. Cambridge University Press, Cambridge, UK and New York, NY, USA, 3056 pp., doi:10.1017/9781009325844

Jaffal, I., Ouldboukhitine, S. E., & Belarbi, R. (2012). A comprehensive study of the impact of green roofs on building energy performance. *Renewable Energy*, *43*, 157–164. https://doi.org/10.1016/J.RENENE.2011.12.004

Jennings, V., & Bamkole, O. (2019). The Relationship between Social Cohesion and Urban Green Space: An Avenue for Health Promotion. *International Journal of Environmental Research and Public Health 2019, Vol. 16, Page 452*, *16*(3), 452. https://doi.org/10.3390/IJERPH16030452

Jeuken, Y. R. H., Breukers, S., Elie, S., & Rugani, M. (2020). Nature4Cities: Nature Based Solutions Projects Implementation Handbook. https://doi.org/10.3030/730468

Kabisch, N., Frantzeskaki, N., Pauleit, S., Naumann, S., Davis, M., Artmann, M., Haase, D., Knapp, S., Korn, H., Stadler, J., Zaunberger, K., & Bonn, A. (2016). Nature-based solutions to climate change mitigation and adaptation in urban areas: Perspectives on indicators, knowledge gaps, barriers, and opportunities for action. *Ecology and Society*, *21*(2). https://doi.org/10.5751/ES-08373-210239

Kato-Huerta, J., & Geneletti, D. (2023). *Analysing the treatment of environmental justice and nature-based solutions in the Urban Climate Action Plans of Latin American metropolitan areas*. https://doi.org/10.1080/13549839.2023.2221431

Konopka, J., Heusinger, J., & Weber, S. (2021). Extensive Urban Green Roof Shows Consistent Annual Net Uptake of Carbon as Documented by 5 Years of Eddy-Covariance Flux Measurements. *Journal of Geophysical Research: Biogeosciences*, *126*(2), e2020JG005879. https://doi.org/10.1029/2020JG005879

Kortetmäki, T., Puurtinen, M., Salo, M., Aro, R., Baumeister, S., Duflot, R., Elo, M., Halme, P., Husu, H.-M., Huttunen, S., Hyvönen, K., Karkulehto, S., Kataja-aho, S., Keskinen, K. E., Kulmunki, I., Mäkinen, T., Näyhä, A., Okkolin, M.-A., Perälä, T., … Kotiaho, J. S. (2021). Planetary well-being. *Humanities and Social Sciences Communications*, *8*(1). https://doi.org/10.1057/S41599-021-00899-3

Landrigan, P. J., Rauh, V. A., & Galvez, M. P. (2010). Environmental justice and the health of children. *The Mount Sinai Journal of Medicine, New York*, *77*(2), 178–187. https://doi.org/10.1002/MSJ.20173

Langemeyer, J., & Connolly, J. J. T. (2020). Weaving notions of justice into urban ecosystem services research and practice. *Environmental Science and Policy*, *109*(April), 1–14. https://doi.org/10.1016/j.envsci.2020.03.021

Langemeyer, J., Madrid-Lopez, C., Mendoza Beltran, A., & Villalba Mendez, G. (2021). Urban agriculture — A necessary pathway towards urban resilience and global sustainability? *Landscape and Urban Planning*, *210*, 104055. https://doi.org/10.1016/J.LANDURBPLAN.2021.104055

Langemeyer, J., Wedgwood, D., McPhearson, T., Baró, F., Madsen, A. L., & Barton, D. N. (2020). Creating urban green infrastructure where it is needed – A spatial ecosystem service-based decision analysis of green roofs in Barcelona. *Science of The Total Environment*, *707*, 135487. https://doi.org/10.1016/J.SCITOTENV.2019.135487

Li, L., Uyttenhove, P., & Van Eetvelde, V. (2020). Planning green infrastructure to mitigate urban surface water flooding risk – A methodology to identify priority areas applied in the city of Ghent. *Landscape and Urban Planning*, *194*. https://doi.org/10.1016/j.landurbplan.2019.103703

Liang, L. L., Anderson, R. G., Shiflett, S. A., & Jenerette, G. D. (2017). Urban outdoor water use and response to drought assessed through mobile energy balance and vegetation greenness measurements. *Environmental Research Letters*, *12*(8), 084007. https://doi.org/10.1088/1748-9326/AA7B21

Liverman DM. (2001). Vulnerability to global environmental change. In J. X. Kasperson & R. E. Kasperson (Eds.), *Global Environmental Risk* (1st ed., pp. 201–217). Routledge.

Loiseau, E., Saikku, L., Antikainen, R., Droste, N., Hansjürgens, B., Pitkänen, K., Leskinen, P., Kuikman, P., & Thomsen, M. (2016). Green economy and related concepts: An overview. *Journal of Cleaner Production*, *139*, 361–371. https://doi.org/10.1016/J.JCLEPRO.2016.08.024

López-Valencia, A. P. (2019). Vulnerability assessment in urban areas exposed to flood risk: methodology to explore green infrastructure benefits in a simulation scenario involving the Cañaveralejo River in Cali, Colombia. *Natural Hazards*, *99*(1), 217–245. https://doi.org/10.1007/S11069-019-03736-8/TABLES/13

Maia, S., Almasy, D., Pinter, L., Werner, A., Hörschelmann, K., Dammers, E., Bulkeley, H. (2021). *Urban nature navigator: silver level implementation guide*

Matthew McConnachie, M., & Shackleton, C. M. (2010). Public green space inequality in small towns in South Africa. *Habitat International*, *34*(2), 244–248. https://doi.org/10.1016/J.HABITATINT.2009.09.009

McDougall, R., Kristiansen, P., & Rader, R. (2019). Small-scale urban agriculture results in high yields but requires judicious management of inputs to achieve sustainability. *Proceedings of the National Academy of Sciences of the United States of America*, *116*(1), 129–134. https://doi.org/10.1073/PNAS.1809707115

McPhearson, T., Andersson, E., Elmqvist, T., & Frantzeskaki, N. (2015). Resilience of and through urban ecosystem services. *Ecosystem Services*, *12*, 152–156. https://doi.org/10.1016/J.ECOSER.2014.07.012

McPhearson, T., Haase, D., Kabisch, N., & Gren, Å. (2016). Advancing understanding of the complex nature of urban systems. *Ecological Indicators*, *70*, 566–573. https://doi.org/10.1016/J.ECOLIND.2016.03.054

Meerow, S. (2017). Double exposure, infrastructure planning, and urban climate resilience in coastal megacities: A case study of Manila. Environment and Planning A: Economy and Space, 49(11), 2649-2672. *https://doi.org/10.1177/0308518X17723630*

Meerow, S., & Newell, J. P. (2017). Spatial planning for multifunctional green infrastructure: Growing resilience in Detroit. *Landscape and Urban Planning*, *159*, 62–75. https://doi.org/10.1016/j.landurbplan.2016.10.005

Meerow, S., & Newell, J. P. (2019). Urban resilience for whom, what, when, where, and why? *Urban Geography*, *40*(3), 309–329. https://doi.org/10.1080/02723638.2016.1206395

Meerow, S., Pajouhesh, P., & Miller, T. R. (2019). Social equity in urban resilience planning. *Local Environment*, *24*(9), 793–808. https://doi.org/10.1080/13549839.2019.1645103

Mehmood, A. (2015). Of resilient places: planning for urban resilience. European Planning Studies, 24(2), 407–419. https://doi.org/10.1080/09654313.2015.1082980

Moreno-Jiménez, A., Cañada-Torrecilla, R., Vidal-Domínguez, M. J., Palacios-García, A., & Martínez-Suárez, P. (2016). Assessing environmental justice through potential exposure to air pollution: A socio-spatial analysis in Madrid and Barcelona, Spain. *Geoforum*, *69*, 117–131. https://doi.org/10.1016/J.GEOFORUM.2015.12.008

Nahiduzzaman, K. M., Aldosary, A. S., & Rahman, M. T. (2015). Flood induced vulnerability in strategic plan making process of Riyadh city. *Habitat International*, *49*, 375–385. https://doi.org/10.1016/J.HABITATINT.2015.05.034

Nesbitt, L., Meitner, M. J., Girling, C., & Sheppard, S. R. J. (2019). Urban green equity on the ground: Practice-based models of urban green equity in three multicultural cities. *Urban Forestry & Urban Greening*, *44*, 126433. https://doi.org/10.1016/J.UFUG.2019.126433

Nesbitt, L., Meitner, M. J., Sheppard, S. R. J., & Girling, C. (2018). The dimensions of urban green equity: A framework for analysis. *Urban Forestry & Urban Greening*, *34*, 240–248. https://doi.org/10.1016/J.UFUG.2018.07.009

Norman, L. M., Villarreal, M. L., Lara-Valencia, F., Yuan, Y., Nie, W., Wilson, S., Amaya, G., & Sleeter, R. (2012). Mapping socio-environmentally vulnerable populations access and exposure to ecosystem services at the U.S.–Mexico borderlands. *Applied Geography*, *34*, 413–424. https://doi.org/10.1016/J.APGEOG.2012.01.006

Pataki, D. E., Boone, C. G., Hogue, T. S., Jenerette, G. D., Mcfadden, J. P., & Pincetl, S. (2011). Socio-ecohydrology and the urban water challenge. *Ecohydrology*, *4*(2), 341–347. https://doi.org/10.1002/ECO.209

Patterson, J., Schulz, K., Vervoort, J., van der Hel, S., Widerberg, O., Adler, C., Hurlbert, M., Anderton, K., Sethi, M., & Barau, A. (2017). Exploring the governance and politics of transformations towards sustainability. *Environmental Innovation and Societal Transitions*, *24*, 1–16. https://doi.org/https://doi.org/10.1016/j.eist.2016.09.001

Pereira, P., Yin, C., & Hua, T. (2023). Nature-based solutions, ecosystem services, disservices, and impacts on well-being in urban environments. *Current Opinion in Environmental Science & Health*, *33*, 100465. https://doi.org/10.1016/J.COESH.2023.100465

Pickett, S. T. A., Cadenasso, M. L., & Grove, J. M. (2004). Resilient cities: meaning, models, and metaphor for integrating the ecological, socio-economic, and planning realms. *Landscape and Urban Planning*, *69*(4), 369–384. https://doi.org/10.1016/J.LANDURBPLAN.2003.10.035

Pörtner, H.-O., Roberts, D., Tignor, M., Poloczanska, E., Mintenbeck, K., Alegría, A., Craig, M., Langsdorf, S., Löschke, S., Möller, V., Okem, A., Rama, B., Belling, D., Dieck, W., Götze, S., Kersher, T., Mangele, P., Maus, B., Mühle, A., & Weyer, N. (2022). *Climate Change 2022: Impacts, Adaptation and Vulnerability Working Group II Contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. https://doi.org/10.1017/9781009325844.

Quigley, M., Blair, N., & Davison, K. (2018). Articulating a social-ecological resilience agenda for urban design. *Https://Doi.Org/10.1080/13574809.2018.1440176*, *23*(4), 581–602. https://doi.org/10.1080/13574809.2018.1440176

Raddatz, L., & Mennis, J. (2013). Environmental Justice in Hamburg, Germany. *Professional Geographer*, *65*(3), 495–511. https://doi.org/10.1080/00330124.2012.700500

Raparthi, K., & Vedamuthu, R. (2022). Assessing the Role of Nature-Based Solutions in Urban Resilience and Climate Change Adaptation. *Lecture Notes in Civil Engineering*, *191*, 13–21. https://doi.org/10.1007/978-981-16-5839-6_2

Ren, Y., Ge, Y., Gu, B., Min, Y., Tani, A., & Chang, J. (2014). Role of management strategies and environmental factors in determining the emissions of biogenic volatile organic compounds from urban greenspaces. *Environmental Science and Technology*, *48*(11), 6237–6246. https://doi.org/10.1021/ES4054434/SUPPL_FILE/ES4054434_SI_001.PDF

Rigillo, M., & Cervelli, E. (2014). Mapping Urban Vulnerability: The Case Study of Gran Santo Domingo, Dominican Republic. *Advanced Engineering Forum*, *11*, 142–148. https://doi.org/10.4028/WWW.SCIENTIFIC.NET/AEF.11.142

Rödl, A., & Arlati, A. (2022). A general procedure to identify indicators for evaluation and monitoring of nature-based solution projects. *Ambio*, *51*(11), 2278–2293. https://doi.org/10.1007/S13280-022-01740-0/TABLES/3

Rodríguez-Rodríguez, D., Kain, J. H., Haase, D., Baró, F., & Kaczorowska, A. (2015). Urban self-sufficiency through optimised ecosystem service demand. A utopian perspective from European cities. *Futures*, *70*, 13–23. https://doi.org/10.1016/J.FUTURES.2015.03.007

Roman, L. A., Conway, T. M., Eisenman, T. S., Koeser, A. K., Ordóñez Barona, C., Locke, D. H., Jenerette, G. D., Östberg, J., & Vogt, J. (2020). Beyond 'trees are good': Disservices, management costs, and tradeoffs in urban forestry. *Ambio*, 615–630. https://doi.org/10.1007/s13280-020-01396-8

Romero-Lankao, P., McPhearson, T., & Davidson, D. J. (2017). The food-energy-water nexus and urban complexity. *Nature Climate Change 2017 7:4*, *7*(4), 233–235. https://doi.org/10.1038/nclimate3260

Sæbø, A., Benedikz, T., & Randrup, T. B. (2003). Selection of trees for urban forestry in the Nordic countries. *Urban Forestry & Urban Greening*, *2*(2), 101–114. https://doi.org/10.1078/1618-8667-00027

Salter, J., Robinson, J., & Wiek, A. (2010). Participatory methods of integrated assessment—a review. *Wiley Interdisciplinary Reviews: Climate Change*, *1*(5), 697–717. https://doi.org/10.1002/WCC.73

Sanchez, L., & Reames, T. G. (2019). Cooling Detroit: A socio-spatial analysis of equity in green roofs as an urban heat island mitigation strategy. *Urban Forestry & Urban Greening*, *44*, 126331. https://doi.org/10.1016/J.UFUG.2019.04.014

Schlosberg, D. (2004). Reconceiving Environmental Justice: Global Movements And Political Theories. *Environmental Politics*, *13*(3), 517–540. https://doi.org/10.1080/0964401042000229025

Schlosberg, D. (2007). Defining Environmental Justice: Theories, Movements, and Nature. *Defining Environmental Justice: Theories, Movements, and Nature*, *9780199286294*, 1–256. https://doi.org/10.1093/ACPROF:OSO/9780199286294.001.0001

Schneiderbauer, S., Calliari, E., Eidsvig, U., & Hagenlocher, M. (2017). *The most recent view of vulnerability* (pp. 68–82).

Seddon, N. (2022). Harnessing the potential of nature-based solutions for mitigating and adapting to climate change. *Science*, *376*(6600), 1400–1416. https://doi.org/10.1126/SCIENCE.ABN9668/ASSET/BF73657E-636C-4029-BB0F-386F6D3AD3CE/ASSETS/IMAGES/LARGE/SCIENCE.ABN9668-F3.JPG

Seddon, N., Chausson, A., Berry, P., Girardin, C. A. J., Smith, A., & Turner, B. (2020). Understanding the value and limits of nature-based solutions to climate change and other global challenges. In *Philosophical Transactions of the Royal Society B: Biological Sciences* (Vol. 375, Issue 1794). Royal Society Publishing. https://doi.org/10.1098/rstb.2019.0120

Sekulova, F., Anguelovski, I., Kiss, B., Kotsila, P., Baró, F., Palgan, Y. V., & Connolly, J. (2021). The governance of nature-based solutions in the city at the intersection of justice and equity. In *Cities* (Vol. 112). Elsevier Ltd. https://doi.org/10.1016/j.cities.2021.103136

Seto, K. C., Reenberg, A., Boone, C. G., Fragkias, M., Haase, D., Langanke, T., Marcotullio, P., Munroe, D. K., Olah, B., & Simon, D. (2012). Urban land teleconnections and sustainability. *Proceedings of the National Academy of Sciences of the United States of America*, *109*(20), 7687–7692. https://doi.org/10.1073/pnas.1117622109

Shafique, M., Kim, R., & Kyung-Ho, K. (2018). Green Roof for Stormwater Management in a Highly Urbanized Area: The Case of Seoul, Korea. *Sustainability 2018, Vol. 10, Page 584*, *10*(3), 584. https://doi.org/10.3390/SU10030584

Shao, H., & Kim, G. (2022). A Comprehensive Review of Different Types of Green Infrastructure to Mitigate Urban Heat Islands: Progress, Functions, and Benefits. *Land 2022, Vol. 11, Page 1792*, *11*(10), 1792. https://doi.org/10.3390/LAND11101792

Shokry, G., Connolly, J., & Anguelovski, I. (2018). From critical climate adaptation to green resilience gentrification and new social-ecological riskscapes. *Conference Proceedings. Retrieved from: Https://Www. Researchgate. Net/Publication/329910152_From_critical_climate_adaptation_to_g Reen_resilience_gentrification_and_new_social-Ecological_riskscapes*.

Song, Y., Kirkwood, N., Maksimović, Č., Zhen, X., O'Connor, D., Jin, Y., & Hou, D. (2019). Nature based solutions for contaminated land remediation and brownfield redevelopment in cities: A review. *Science of The Total Environment*, *663*, 568–579. https://doi.org/10.1016/J.SCITOTENV.2019.01.347

Sovacool, B. K., Barnacle, M. L., Smith, A., & Brisbois, M. C. (2022). Towards improved solar energy justice: Exploring the complex inequities of household adoption of photovoltaic panels. *Energy Policy*, *164*, 112868. https://doi.org/10.1016/J.ENPOL.2022.112868

Speak, A. F., Rothwell, J. J., Lindley, S. J., & Smith, C. L. (2012). Urban particulate pollution reduction by four species of green roof vegetation in a UK city. *Atmospheric Environment*, *61*, 283–293. https://doi.org/10.1016/J.ATMOSENV.2012.07.043

Stavins, R. N., Wagner, A. F., & Wagner, G. (2003). Interpreting sustainability in economic terms: dynamic efficiency plus intergenerational equity. *Economics Letters*, *79*(3), 339–343. https://doi.org/10.1016/S0165-1765(03)00036-3

Susca, T., Gaffin S.R., & Dell'Osso G.R. (2011). Positive effects of vegetation: Urban heat island and green roofs. *Environmental Pollution*, *159*(8–9), 2119–2126. https://doi.org/10.1016/J.ENVPOL.2011.03.007

Tams, L., Nehls, T., & Calheiros, C. S. C. (2022). Rethinking green roofs- natural and recycled materials improve their carbon footprint. *Building and Environment*, *219*, 109122. https://doi.org/10.1016/J.BUILDENV.2022.109122

Teicher, H. M. (2018). Practices and pitfalls of competitive resilience: Urban adaptation as real estate firms turn climate risk to competitive advantage. *Urban Climate*, *25*(January 2017), 9–21. https://doi.org/10.1016/j.uclim.2018.04.008

Tonini, P., Odina, P. M., Orsini, F., & Durany, X. G. (2022). Economic benefit and social impact derived by a food loss prevention strategy in the vegetable sector: A spatial and temporal analysis at the regional level. *Frontiers in Sustainable Food Systems*, *6*, 1043591. https://doi.org/10.3389/FSUFS.2022.1043591/BIBTEX

Tonne, C., Milà, C., Fecht, D., Alvarez, M., Gulliver, J., Smith, J., Beevers, S., Ross Anderson, H., & Kelly, F. (2018). Socioeconomic and ethnic inequalities in exposure to air and noise pollution in London. *Environment International*, *115*, 170–179. https://doi.org/10.1016/J.ENVINT.2018.03.023

Turner, B., Devisscher, T., Chabaneix, N., Woroniecki, S., Messier, C., & Seddon, N. (2022). The Role of Nature-Based Solutions in Supporting Social-Ecological Resilience for Climate Change Adaptation. *Annual Review of Environment and Resources*, *47*(Volume 47, 2022), 123–148. https://doi.org/10.1146/ANNUREV-ENVIRON-012220-010017/1

United Nations. (2015). *Goal 11: Make cities inclusive, safe, resilient and sustainable*. https://www.un.org/sustainabledevelopment/cities/

United Nations. (2017). *The New Urban Agenda*. https://habitat3.org/the-new-urban-agenda/

United Nations. (2019). *United Nations Human Settlements Programme*. https://unhabitat.org/urban-themes/resilience/

United Nations. (2022). *Resolution adopted by the United Nations Environment Assembly on 2 March 2022*.

Venter, Z. S., Barton, D. N., Martinez-Izquierdo, L., Langemeyer, J., Baró, F., & McPhearson, T. (2021). Interactive spatial planning of urban green infrastructure – Retrofitting green roofs where ecosystem services are most needed in Oslo. *Ecosystem Services*, *50*(June), 101314. https://doi.org/10.1016/j.ecoser.2021.101314

Wang, H., Qiu, F., & Swallow, B. (2014). Can community gardens and farmers' markets relieve food desert problems? A study of Edmonton, Canada. *Applied Geography*, *55*, 127–137. https://doi.org/10.1016/J.APGEOG.2014.09.010

Wendling, L. A., Huovila, A., zu Castell-Rüdenhausen, M., Hukkalainen, M., & Airaksinen, M. (2018). Benchmarking Nature-Based Solution and Smart City Assessment Schemes Against the Sustainable Development Goal Indicator Framework. *Frontiers in Environmental Science*, *0*(JUL), 69. https://doi.org/10.3389/FENVS.2018.00069

Whitehead, M. (2013). Neoliberal Urban Environmentalism and the Adaptive City: Towards a Critical Urban Theory and Climate Change. *Urban Studies*, *50*(7), 1348–1367. https://doi.org/10.1177/0042098013480965

White-Newsome, J. L., & Slay, J. A. (2022). *Learning to Lead with Equity: Advancing Climate Resilience Planning to Address Urban Flooding Across Multiple Sectors and Scales*. 97–116. https://doi.org/10.1007/978-3-030-73939-3_6

Wood, B. T., Quinn, C. H., Stringer, L. C., & Dougill, A. J. (2017). Investigating Climate Compatible Development Outcomes and their Implications for Distributive Justice: Evidence from Malawi. *Environmental Management*, *60*(3), 436–453. https://doi.org/10.1007/S00267-017-0890-8/FIGURES/4

Yeeles, A., Sosalla-Bahr, K., Ninete, J., -, al, Schultz, B., Brockington, D., Friedman, R. S., Law, E. A., Bennett, N. J., Ives, C. D., R Thorn, J. P., & Wilson, K. A. (2018). How just and just how? A systematic review of social equity in conservation research. *Environmental Research Letters*, *13*(5), 053001. https://doi.org/10.1088/1748-9326/AABCDE

Zhang, B., Xie, G. di, Gao, J. xi, & Yang, Y. (2014). The cooling effect of urban green spaces as a contribution to energy-saving and emission-reduction: A case study in Beijing, China. *Building and Environment*, *76*, 37–43. https://doi.org/10.1016/J.BUILDENV.2014.03.003

Zhang, Z., Meerow, S., Newell, J. P., & Lindquist, M. (2019). Enhancing landscape connectivity through multifunctional green infrastructure corridor modeling and design. *Urban Forestry & Urban Greening*, *38*(July 2018), 305–317. https://doi.org/10.1016/j.ufug.2018.10.014

Zumkehr, A., & Campbell, J. E. (2015). The potential for local croplands to meet US food demand. *Frontiers in Ecology and the Environment*, *13*(5), 244–248. https://doi.org/10.1890/140246

CHAPTER III. ASSESSING NATURE-BASED SOLUTIONS IN THE FACE OF URBAN VULNERABILITIES: A MULTI-CRITERIA DECISION APPROACH

III. Abstract

Nature-based solutions (NBS) are increasingly employed to address urban challenges. Typically, NBS planning emphasizes environmental impacts and ecosystem services, often overlooking their role in addressing vulnerabilities. My objective is to develop a framework assessing the extent to which NBS alter urban vulnerabilities. For this, I relate ecosystem service and urban metabolism analyses to spatially explicit vulnerabilities. The framework relies on multi-criteria decision analysis to integrate diverse impacts. It follows a stepwise approach including the development of land-use scenarios, selection of vulnerabilities and indicators, normalization and aggregation of indicators, and stakeholder weighting. I apply the framework to the Metropolitan Area of Barcelona to assess the impacts of increasing (peri-)urban agriculture in terms of critical vulnerabilities: heat, lack of recreational space, biodiversity loss, and lack of local food. Results show that agricultural expansion decreased the vulnerability of lack of local food, increased the vulnerability of biodiversity loss, and increased the heat vulnerability in terms of night temperatures for sensitive areas. Results reveal diverse spatial outcomes and trade-offs in urban vulnerabilities due to shifts in (peri-)urban agriculture. The framework innovatively evaluates NBS impacts by linking multiple evaluation methods through spatially explicit vulnerabilities, fostering the strategic planning of NBS at the urban metropolitan scale.

Keywords: Nature-based solutions; Ecosystem services; Urban metabolism; Vulnerability assessment; Urban vulnerability; Urban planning

This chapter corresponds to the following published article:

Camacho-caballero, D., Langemeyer, J., Segura-barrero, R., Ventura, S., Beltran, A.M., & Villalba, G. (2024). Assessing nature-based solutions in the face of urban vulnerabilities: a multi-criteria decision approach. Sustainable Cities and Society, 105257. https://doi.org/10.1016/j.scs.2024.105257

List of abbreviations

AMB: Metropolitan Area of Barcelona

ES: Ecosystem service

MCDA: multi-criteria decision analysis

NBS: Nature-based solutions

PDU: Urban Master Plan of the Metropolitan Area of Barcelona

S0: Urban agriculture scenario "Current"

S1: Urban agriculture scenario "Trending"

S2: Urban agriculture scenario "Alternative"

S3: Urban agriculture scenario "Potential"

UA: Urban agriculture

UM: Urban metabolism

III.1. Introduction

Nature-based solutions (NBS) are being increasingly advocated to bolster urban resilience and sustainability (Cohen-Shacham et al., 2019). NBS are understood as "actions to protect, conserve, restore, sustainably use and manage natural or modified terrestrial, freshwater, coastal and marine ecosystems which address social, economic and environmental challenges effectively and adaptively, while simultaneously providing human well-being, ecosystem services, resilience and biodiversity benefits" (United Nations, 2022). NBS, such as urban and peri-urban forests, green roofs and walls, pervious pavements, and urban agriculture, exemplify a multifunctional, solution-oriented approach to enhancing urban sustainability (Dorst et al., 2019). In particular, urban agriculture serves as a key illustration of NBS, and it will be a focal point throughout this study. Typically, planning NBS involves analyzing different alternatives and their projected outcomes in terms of the direct and indirect contributions to human well-being, or ecosystem service (ES) provision (Raymond et al., 2017). However, there is often insufficient attention given to how NBS address specific vulnerabilities in spatially heterogeneous urban landscapes (Langemeyer et al., 2020) (see **Fig. 1**). Considering such an aspect could enhance the evaluation of NBS by broadening the perspectives included in its assessment, contributing to a more comprehensive evaluation of NBS (Dumitru et al., 2020).

Vulnerability can be broadly defined as the susceptibility to harm (Cutter, 2016) of both social and ecological systems. Urban vulnerabilities are spatially heterogeneous and encompass two key dimensions: exposure and sensitivity. Exposure relates to the proximity of systems to hazards, while sensitivity describes the extent to which a system is impacted by hazards (Thiault et al., 2018). For instance, vulnerability analyses provide insights into the extent and patterns of people's exposure to climate-related risks and the inequalities in coping with these impacts (Baró et al., 2021). Yet, despite important advances toward a differentiated understanding of urban vulnerabilities (Herreros-Cantis & McPhearson, 2021), NBS planning is still widely based on the assessment of net ES.

The ES framework highlights the various ways in which humans can benefit from natural ecosystems (Fisher et al., 2009). NBS can bring these benefits and co-benefits in environmental, social, and economic terms (Cohen-Shacham et al., 2019). However, ES are spatially explicit (Herreros-Cantis & McPhearson, 2021) and the location of NBS determines its distribution. Depending on the ES type, its provision and beneficiaries may differ (Basnou et al., 2020) (e.g., food can be transported, temperature regulation cannot). This creates urban areas with ES deficits (Langemeyer et al., 2020), commonly referred to as areas with ES demands. These spatial (mis)matches need to be considered when planning NBS (Basnou et al., 2020) since the distribution of ES across various scales and groups is crucial from a socio-environmental perspective, not least under equity considerations (Langemeyer & Connolly, 2020).

Compared to ES, the environmental impacts of NBS have received less attention. Environmental impacts are generally analyzed through urban metabolism (UM) approaches, understood as the sum of processes that an urban system needs to maintain itself by importing, producing and exporting materials, while also emitting waste (Kennedy et al., 2007). UM offers essential evaluation techniques for sustainable city planning (Perrotti & Stremke, 2020). Under this perspective, NBS can be examined to understand their impacts on energy consumption/reduction, water consumption and greenhouse gas emissions. For instance, urban agriculture can contribute to nutrient discharges from fertilizer use, leading to eutrophication, and potentially impacting biodiversity (Firbank et al., 2007). Similar to the ES perspective, UM focuses on analyzing net impacts within a system by assessing the balance between inputs and outputs. Considering such impacts in a geographically explicit manner has been deemed necessary for a more differentiated understanding of NBS (Mendoza Beltran et al., 2022).

Given the significance of spatial analysis in comprehending both ES and UM, adopting a vulnerability approach provides an advanced perspective for studying NBS impacts because of its spatially explicit characteristics. Vulnerability analyses have a well-established tradition in the disaster and risk literature (Liang & Xie, 2022), yet they have not been widely integrated into NBS planning, with few exceptions such as the case of Lehmann et al. (2023). Instead, ES demand approaches are trending (Pan et al., 2021). For example, ES demand has identified areas with insufficient green spaces that could benefit from green interventions to enhance heat and recreational conditions (Meerow & Newell, 2017). However, broader vulnerability considerations in NBS research are limited, as ES demand approaches often overlook potential changes arising from NBS implementation. A more comprehensive vulnerability assessment should thus consider changes in urban exposure and sensitivity due to NBS interventions, providing a novel understanding of NBS impacts. For UM, spatial analyses have become more widespread and are deemed necessary for enhancing land-use planning (Bahers et al., 2022). However, the spatial metabolic effects of NBS have been overlooked (Chrysoulakis et al., 2021). According to Otero Peña et al., 2022, UM has yet to consider urban vulnerabilities, which offer opportunities for enhancing resource efficiency in urban environments.

Furthermore, a spatially explicit integration between ES, UM and urban vulnerabilities can support the planning of NBS in urban environments. This involves considering both intended and unintended NBS impacts simultaneously (Dumitru et al., 2020), as well as their (dis)joint effects on urban vulnerabilities, as vulnerabilities may increase or decrease similarly or oppositely (Zuniga-Teran et al., 2020). These impacts can be related to land-use changes resulting from NBS implementations (Fernandes & Guiomar, 2018). The relevance of this aspect becomes apparent as the integration of NBS evaluation with urban policies remains partial (Pan et al., 2021).

Stakeholder involvement can boost the effectiveness of such an evaluation scheme. Stakeholder engagement aids in addressing the complexity of planning NBS by considering diverse urban perspectives (Nesshöver et al., 2017). Such collaborative approaches can foster urban resilience and sustainability, while also promoting acceptance of NBS (Mees et al., 2015). Because of these traits, stakeholder engagement is encouraged for NBS planning (Zingraff-Hamed et al., 2020). Stakeholders have been involved in NBS assessments to evaluate feasibility and estimate the provision and ranking of ES (Langemeyer et al., 2020; Venter et al., 2021). Consequently, involving stakeholders provides an opportunity for enhancing NBS assessments through urban vulnerabilities.

The objective of this study is to develop a stepwise, multi-criteria and integrated assessment framework capable of evaluating how and to what degree NBS change urban vulnerabilities. This framework will anticipate and evaluate potential intended and unintended consequences arising from NBS implementation resulting in varying degrees of exposure to risks, an aspect that ES and UM assessments fail to contemplate and that can enhance the planning of NBS in urban environments. I hence propose to link the ES and UM analyses to spatially explicit vulnerabilities (see **Fig. 1**) for assessing NBS scenarios representing various land-use configurations while considering stakeholders' inputs. To demonstrate the effectiveness of this approach, I apply it to the case study of (peri-) urban agriculture in the Metropolitan Area of Barcelona (AMB, for its acronym in Catalan). Urban agriculture is a nature-based solution that plays a significant role in shaping the various land-use scenarios outlined in the Urban Master Plan of the AMB, providing valuable insights into the future development of the area.

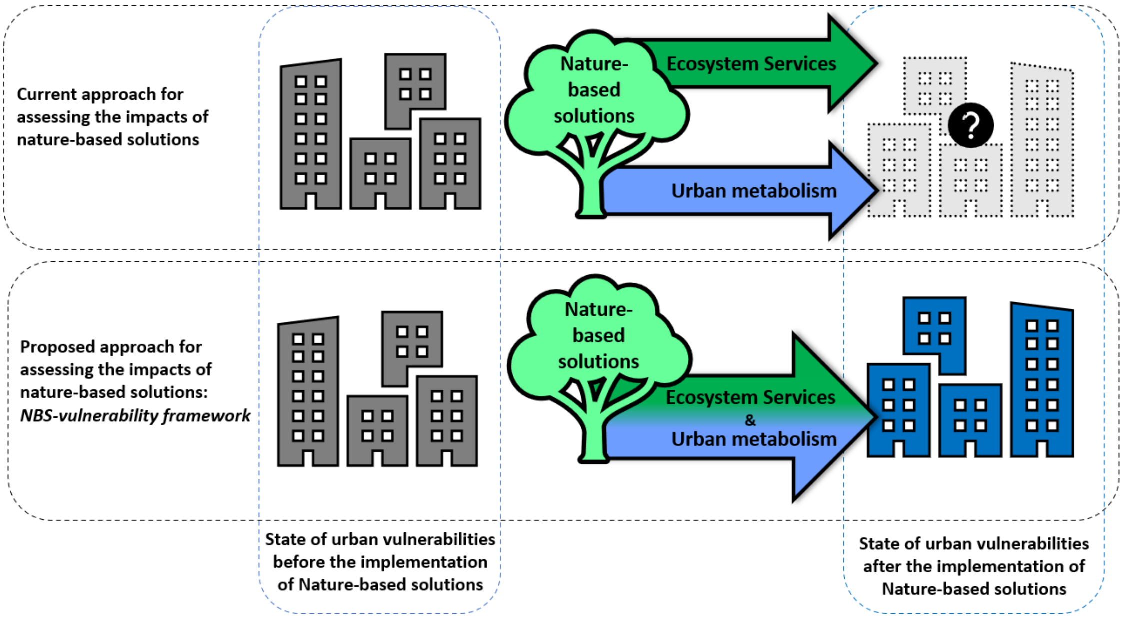


Figure 1. Graphical representation of the current approach for assessing the impacts of nature-based solutions (NBS) in urban environments versus the NBS-vulnerability framework. Arrows represent NBS impacts in urban environments. The current approach focuses on assessing impacts either via ecosystem services or urban metabolism perspective, often overlooking their role in addressing vulnerabilities. The NBS-vulnerability framework suggests linking the ecosystem services and urban metabolism analyses to urban vulnerabilities, elucidating how NBS impacts can affect the latter

III.2. Nature-based solutions vulnerability framework

III.2.1. Conceptual approach

This study approaches urban environments as socio-ecological systems: a dynamic and interconnected network of biophysical and social elements that interact across multiple scales and affect the flow and use of critical natural, socio-economic, and cultural resources (Redman et al., 2004). Within these systems, NBS are designed to address both biophysical and social factors and their interrelationships (Tzoulas et al., 2021). The location, design and overall presence of NBS provide a variety of impacts that can change urban vulnerabilities in both intended and unintended ways (Pereira et al., 2023). Urban vulnerabilities can thus be enhanced or reduced by NBS (Herreros-Cantis & McPhearson, 2021). This study is based on the premise that NBS effects can result from changes in ES supply (e.g., altering temperature regulation through green areas) or from a change in UM (e.g., shifts in urban energy demands due to cooling building requirements). These two effects are often interlinked (e.g., expanding green spaces typically reduces temperatures, lowering the city's cooling energy demand) (Shao & Kim, 2022), causing changes in both the UM of energy and the ES supply of thermal regulation. My suggestion is to comprehensively evaluate these effects, understanding how they influence urban exposures from a spatially explicit perspective. Then, relating these to sensitive urban areas, like places with low-income populations living in buildings with poor energy performance. In essence, I aim to develop a method that determines how the direct and indirect effects of NBS transform urban vulnerabilities.

To do so, I propose linking NBS impacts to spatially explicit changes in the exposures to hazards while considering sensitivities as a static variable. Furthermore, it is understood that changes in vulnerabilities cause both joint and disjoint effects (Zuniga-Teran et al., 2020), which need to be assessed simultaneously to capture the synergies or tradeoffs incurred by NBS. Joint effects happen when NBS impact several vulnerabilities similarly: vulnerabilities either increase or decrease jointly. For instance, green areas have reduced vulnerabilities to air pollution and urban heat islands (Meerow & Newell, 2017). Conversely, disjoint urban vulnerabilities are those in conflict with one another, leading to trade-offs (i.e., increasing one vulnerability while decreasing another). For example, urban agriculture offers recreational opportunities (Langemeyer et al., 2021) reducing the vulnerability of lacking recreational spaces, but may negatively impact ecological vulnerability if non-organic fertilizers are used (Potter & LeBuhn, 2015). To operationalize the joint and disjoint effects on vulnerability, multi-criteria decision analysis (MCDA) premises are considered.

MCDA is a useful tool for developing holistic assessments of urban NBS (Venter et al., 2021), as it enables the integration of quantitative and qualitative data, discordant information, and stakeholders' considerations into decision-making processes, and allows the comparison of various alternatives by weighting different evaluation criteria (Marttunen et al., 2017). Relying on MCDA's capacity to compare different alternative scenarios and accommodate the diverse perspectives within urban environments, I propose a stepwise approach to examine the multidimensional impacts of NBS on vulnerabilities.

III.2.2. Stepwise approach of the Nature-based solutions vulnerability framework

The framework integrates urban ES and UM assessments to spatially explicit vulnerabilities into a structured approach consisting of several key steps, as shown in **Figure 2**.

First, NBS scenarios are developed to represent various land-use configurations specific to the urban environment under study. Within the MCDA methodology, scenarios — or "alternatives" — are useful for exploring potential future states of the environment in situations marked by uncertainty (Marttunen et al., 2017). The proposed framework starts by developing potential configurations of NBS in the form of land-use-change maps, to later contrast how the vulnerabilities shift when compared to a reference scenario. These maps require an appropriate resolution to accommodate vulnerabilities with different spatial patterns. The next step involves identifying and selecting the social-ecological vulnerabilities affected by the NBS. Vulnerabilities are chosen based on urban challenges and agendas, allowing for tailormade NBS planning adapted to local necessities. I propose for each vulnerability to be evaluated by at least one exposure and one sensitivity indicator. Exposure indicators are calculated for each scenario, while sensitivity indicators are calculated once and remain static for all scenarios. Indicators need to be spatially explicit and their resolution compatible with the defined land-use scenarios. The product of this step is a map for each indicator, for each scenario.

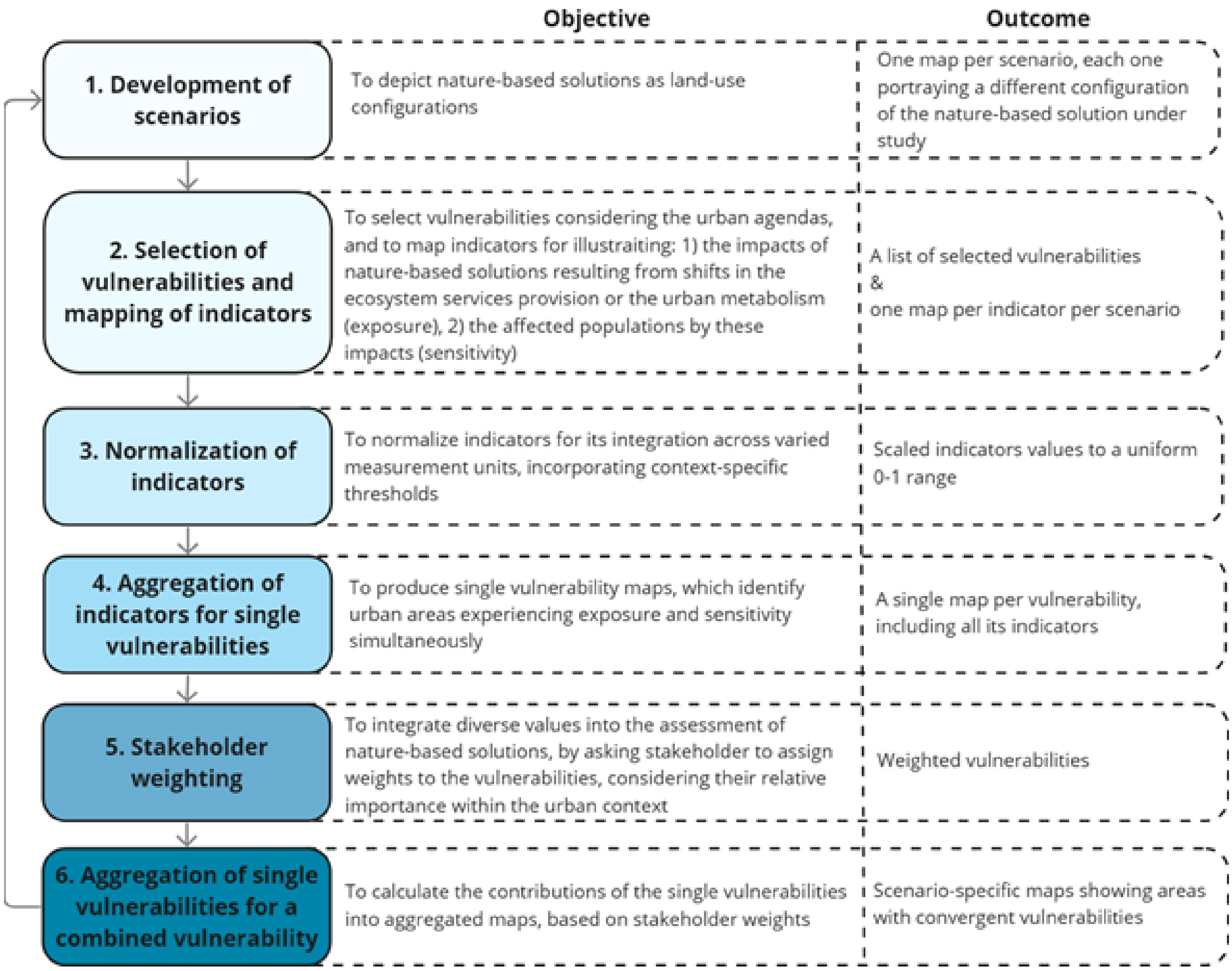


Figure 2. Stepwise approach of the Nature-based solutions vulnerability framework, along with the objective of each step and its expected outcomes

The third stage involves normalizing absolute values of exposure and sensitivity indicators to create a unified scale, enabling integration across different measurement units.

Thresholds are included to determine the magnitude of the NBS impacts based on the selected indicators. Thresholds serve as cutoff values and are established based on scientific knowledge or urban objectives. Thresholds are context-specific, reflecting the urban environment where NBS are situated, enabling risk differentiation based on local conditions. For instance, the threshold for what is considered a heatwave can vary by region due to differing meteorological conditions (Kovats & Kristie, 2006). By the end of this stage, all indicator absolute values are transformed to a uniform scale ranging from 0 to 1. In the fourth step, the normalized exposure and sensitivity indicators of each vulnerability are aggregated to obtain single vulnerability maps, which identify urban areas experiencing exposure and sensitivity simultaneously. Aggregation is employed for representing multidimensional realities through single indexes (OECD & European Union, 2008). In our case, aggregation is necessary to sum the indicators per vulnerability, resulting in a single map.

In the fifth stage, stakeholders are asked to assign weights to the vulnerabilities, considering their relative importance within the urban context. Stakeholder engagement integrates diverse values to the evaluation of the vulnerabilities (Reed, 2008) and has previously been used for assessing NBS. For example, Langemeyer et al., (2020) conveyed stakeholders to rank different ES demands, identifying urban areas where green infrastructure benefits are most needed. Stakeholder-assigned weights are subsequently employed to calculate the contributions of vulnerabilities defined in step 4 into aggregated maps. The outcome consists of scenario-specific maps showing areas with convergent vulnerabilities.

III.3. Case study: Urban Agriculture in the Metropolitan Area of Barcelona

Urban and peri-urban agriculture (UA), an example of NBS, can address urban challenges by improving food security, regulating urban temperatures, promoting social cohesion and enhancing pollination (Wilhelm & Smith, 2018). Yet, the omission of its multifunctionality regarding ES supply, coupled with metabolic impacts, within the context of specific social-ecological vulnerabilities, is a recognized barrier to its promotion (Langemeyer et al., 2021).

Previous metabolism studies indicate UA's significant influence on the inter-related flows of food, water and energy in cities, resulting in impacts on the environment. For example, local crop production can reduce the energy required for transporting fresh produce (Zumkehr & Campbell, 2015), food losses due to long supply chains (Tonini et al.,

2022) and water requirements by optimizing irrigation systems (Parada et al., 2021). Metabolism analyses highlight the potential for lowering environmental impacts through resource circularity, such as reusing waste nutrients like phosphorus as fertilizers (Rufí-Salís et al., 2020).

Yet, assessments including both ES and UM perspectives need to be jointly considered (Perrotti & Stremke, 2020). While some studies have assessed various UA impacts, including ecological and social functions (Padró et al., 2020), there is an opportunity to further enhance the understanding of UA impacts by examining how vulnerabilities are being altered. As it will be shown, vulnerability assessment supports the integration of ES and UM, providing a common ground for interpreting the impacts of UA within the spatially explicit context.

The AMB —our case study area (see **Fig. 3**) — comprises the municipality of Barcelona and other 35 surrounding municipalities with a total population of 3.3 million people (IDESCAT, 2020). The AMB, characterized by high compactness and population density (Baró et al., 2014), faces exacerbated vulnerabilities by climate change impacts (AMB, 2018). For this matter, the AMB plans to enhance resilience by creating green spaces, including UA, as part of the Urban Master Plan (PDU). This plan prioritizes ES and aims to protect agricultural land for local food production while preserving the natural system (Barcelona Regional and AMB-PDU, 2023).

III.4. Methodology

III.4.1. Development of scenarios

We apply the NBS-vulnerability framework to evaluate how four scenarios with various degrees of UA address the vulnerabilities of the AMB. The scenarios were developed by the office of the PDU (Barcelona Regional and AMB-PDU, 2023) to foresee possible land-use changes in the region. The scenarios are Current (S0), which serves as the reference state, relying on the URBAG land-use map land-use map (Mendoza Beltran et al., 2022); Trending (S1) representing a business-as-usual approach with urban expansion and a reduction of green areas; Alternative (S2) converting urban parks into agricultural areas; and Potential (S3), restoring agricultural lands to their 1956 state. **Figure 3** offers a scenario overview. For detailed descriptions, consult Padró et al. (2020).

We used QGIS 3.28.0-Firenze and ArcGIS 10.8.1 to produce and manage all maps and indicators. Scenarios and indicators were transformed into a 50x50m grid that allowed to (1) detail land-use changes across scenarios while considering the extension of the AMB, (2) aggregate various indicators, and (3) manage datasets within data processing capabilities.

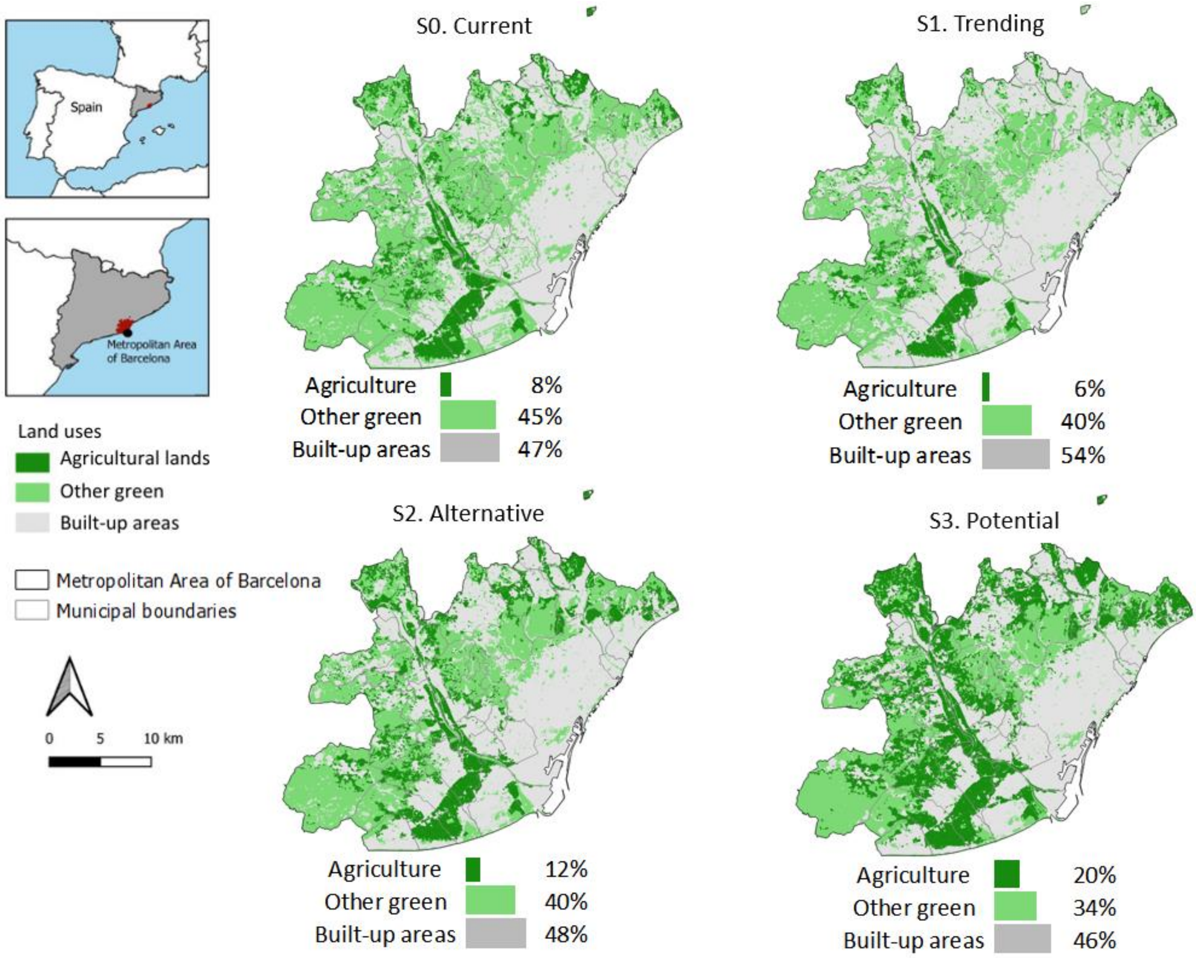


Figure 3. Current land uses and proposed development scenarios for the Metropolitan Area of Barcelona with percentages of land used by agricultural lands, other green spaces and built-up areas

III.4.2. Selection of vulnerabilities and mapping of indicators

The second step in the NBS-vulnerability framework involves selecting the vulnerabilities and the spatially explicit exposure and sensitivity indicators that most appropriately define them. For this study, four vulnerabilities were selected based on AMB future objectives and policies (described in the next section): (1) *vulnerability of lack of local food*, (2) *vulnerability to heat*, (3) *vulnerability of lacking recreational space* and (4) *vulnerability of loss of biodiversity* (see **Table 1**). Each vulnerability is described by at least one exposure and one sensitivity indicator (see **Appendix 1-A**). Indicators were chosen through a literature review and discussion among the interdisciplinary team of authors participating in the assessment. In some cases, the same sensitivity indicator (i.e. population density) is applied to various vulnerabilities because it is the most appropriate way of reflecting urban susceptibility. No double counting arises from these situations because exposure values are always different, and the multiplication of exposure and sensitivity values results in diverse vulnerability maps.

Indicators were converted into a 50x50m grid, allowing for the integration of different resolutions. Henceforth, each of these grid cells will be referred to as pixels.

Table 1. Vulnerabilities, indicators, average/sum of absolute indicator values before normalization, thresholds and weights from the assessment of urban agriculture in the Metropolitan Area of Barcelona

Vulnerability	Indicator	Unit	Exposure/ Sensitivity	Average/sum of absolute indicator values before normalization					Threshold value for normalization	Weights for single vulnerability aggregation	Stakeholder weights for combined vulnerability
				S0.	**S1.**	**S2.**	**S3.**	**Average/sum**			
Vulnerability of lack of local food	Diversity of crops	Index	Exposure	0.018	0.015	0.022	0.036	Average	No threshold value	0.5	48%
	Production of vegetables in the AMB	Ton	Exposure	39,148	34,369	49,014	64,984	Sum	No exposure ≥ 57,348 Ton of vegetables produced for the whole AMB per year	0.25	
	Production of fruits in the AMB	Ton	Exposure	9,284	7,767	12,138	23,104	Sum	No exposure ≥ 59,088 Ton of fruit produced for the whole AMB per year	0.25	
	Population density	Hab./Km²	Sensitivity	5,061	N/A	N/A	N/A	Average		1	
Vulnerability to heat	Heatwave day temperatures	°C	Exposure	29.05	29.07	29.04	29.02	Average	No exposure ≤ 32°C	0.5	14%
	Heatwave night temperatures	°C	Exposure	24.25	24.27	24.24	24.26	Average	No exposure ≤ 23°C	0.5	
	Population density	Hab./Km²	Sensitivity	5,061	N/A	N/A	N/A	Average		0.5	
	Elderly population density	Hab./Km²	Sensitivity	980	N/A	N/A	N/A	Average		0.5	
Vulnerability of lacking recreational space	Areas with accessibility to green spaces at less than 300m, less than 1000m and more than 1000m	Km²	Exposure	54.7	65.1	55.7	53.6	Sum of Km² with accessibility to green spaces at more than 1000m	No exposure ≤ 300m High exposure ≥ 1000m	1	9%
	Population density	Hab./Km²	Sensitivity	5,061	N/A	N/A	N/A	Average		1	
Vulnerability of loss of biodiversity	Phosphorous discharges from fertilizer use	Ton	Exposure	21	19	28	38	Sum	No exposure ≤ 363.43 Ton/year for the whole region	1	29%
	Functional diversity	Composed Index	Sensitivity	0.23	N/A	N/A	N/A	Average		0.5	
	Singular biodiversity	Composed Index	Sensitivity	0.35	N/A	N/A	N/A	Average		0.5	

4.2.1. Vulnerability of lack of local food

Urban expansion and land abandonment in the AMB caused a significant reduction in agricultural land, from 24,600 hectares in 1956 to 5,400 hectares in 2009 (IERMB, 2016). To address this, the AMB aims to enhance food security via urban policies, including the protection of urban agricultural spaces (Barcelona Regional and AMB-PDU, 2023).

I define lack of local food as a region's ability to meet its residents' food demand, a vulnerability affected by UA's role in increasing and diversifying food production (Langemeyer et al., 2021). To assess this vulnerability, the exposure indicators are (a) diversity of crops, as diversity is linked to improved yield and disease management (He et al., 2019), (b) production of vegetables and (c) fruits in the AMB as a proxy for food supply assessment. The sensitivity indicator is the overall population density, representing areas with higher food demand (for detailed indicator descriptions, justification for its selection and calculations, see **Appendix 1-B**, Section 1).

4.2.2. Vulnerability to heat

In general, vegetation regulates temperatures during heatwaves (Shao & Kim, 2022) by absorbing solar radiation, enabling transpiration, and providing shade. Regarding UA, irrigation offers daytime cooling through evapotranspiration (Kueppers et al., 2007). Given the future projection of more intense and frequent heat waves at the AMB (del Río et al., 2007), the AMB has recognized the *vulnerability to heat* as an urgent challenge to address (Barcelona Metropolitan Area - AMB, 2018).

To evaluate exposure to heat, two indicators are included: daytime temperatures (13h-16h) and nighttime temperatures (21h-7h), as observed during a heatwave in the AMB (June 20th 2015 - July 25th 2015). Both indicators have been correlated with health problems (Heaviside et al., 2016). Sensitivity indicators are based on (a) overall population density and (b) elderly population density, both employed for assessing population at risk (for detailed indicator descriptions, justification for its selection and calculations, see **Appendix 1-B**, Section 2).

4.2.3. Vulnerability of lacking recreational space

AMB's high compactness and limited green spaces (Baró et al., 2014) lead to a high demand for outdoor recreational areas, a valuable factor for residents' physical and mental well-being (Triguero-Mas et al., 2015). The AMB plans to improve NBS accessibility to fulfil this need (Barcelona Regional and AMB-PDU, 2023). Peri-urban farmland offers a wide range of recreation opportunities (Langemeyer et al., 2021), and can thus address the *vulnerability of lacking recreational space*.

To assess this vulnerability's exposure, the indicator selected was accessibility to green spaces at less than 300m, less than 1000m and more than 1000m (Grunewald et al., 2017), while the sensitivity indicator consists of overall population density (for detailed indicator descriptions, justification for its selection and calculations, see **Appendix 1-B**, Section 3).

4.2.4. Vulnerability of loss of biodiversity

The AMB's diverse urban environments foster a variety of species, while nearby forests provide a stable habitat for adapted species (Langemeyer & Gómez-Baggethun, 2017). The AMB plans to enhance biodiversity in parks and coastal regions (AMB, 2018). Despite its benefits, UA can negatively impact ecosystems if non-organic fertilizers are used (Potter & LeBuhn, 2015). To evaluate the *vulnerability of loss of biodiversity*, the exposure indicator phosphorous discharges from fertilizer is used as a proxy for potential eutrophication affecting biodiversity conditions (Firbank et al., 2007). The sensitivity indicators include (a) functional biodiversity and (b) singular biodiversity, providing insights into the relationships between biodiversity and ecosystem functioning (Basnou et al., 2020) (for detailed indicator descriptions, justification for its selection and calculations, see **Appendix 1-B**, Section 4).

III.4.3. Normalization of indicators

The third step in the NBS-vulnerability framework is to normalize the exposure and the sensitivity indicators so that they can be compared on the same scale (see **Appendix 1-A**). All indicators' absolute values were scaled to 0-1 using min-max normalization (see **Appendix 1-B**, Section 5), where 0 indicates no exposure/sensitivity and 1 indicates the highest exposure/sensitivity.

First, min-max values for the exposure indicators are defined according to threshold values provided by the literature (Inèdit, 2022; Díaz et al., 2015; Royé, 2017; Stigsdotter et al., 2010; Vos et al., 2022; Bauwelinck et al., 2021; Grazuleviciene et al., 2014; Paquet et al., 2013; Reid et al., 2017; European Environmental Agency, 2020). Thresholds representing no exposure are included as minimum values, while those indicating high critical exposure are set as maximum values. For example, the no-exposure threshold for the *Heatwave Day temperatures* indicator is 32ºC (Díaz et al., 2015) - below this temperature, the exposure to heat is deemed insignificant and consequently, there is no vulnerability. Likewise, the no-exposure threshold for the indicator *Phosphorous discharges from fertilizer use* is 363.43 tonnes/year for the AMB (European Environmental Agency, 2020). Below this value, the exposure is not deemed critical for biodiversity and therefore no *vulnerability of loss of biodiversity* is given. The thresholds selected for this study are described in **Table 1** (for detailed normalization calculations, see **Appendix 1-B**).

Indicators *Phosphorous discharges from fertilizer use* and production of vegetables/fruits included extra steps in the normalization to provide a more accurate representation of the final value (e.g., production of vegetables was normalized to consider both pixel-level production and overall production in AMB). This is because certain impacts can only be accurately assessed at the AMB level. For instance, the production of vegetables threshold is based on the target amount of locally produced vegetables that AMB residents should consume, while *Heatwave day temperatures* rely on a fixed temperature that may or may not occur in many areas simultaneously.

III.4.4. Aggregation of indicators for single vulnerabilities

The next step is to aggregate the normalized indicators into a single exposure and a single sensitivity for each vulnerability (see equation in **Appendix 1-B**, Section 6.1). For this, the relative weights of the indicators were equally distributed (see **Table 1**). Next, and for each of the vulnerabilities, the single exposure and single sensitivity were aggregated (see equation in **Appendix 1-B**, Section 6.2). This allowed to obtain a single vulnerability that effectively summarizes its exposures and sensitivities. Additionally, there were incorporated calculations of the sum of pixel values and their relative change between scenarios for each of the vulnerabilities to depict the magnitude of each vulnerability in the AMB and its behavior across scenarios.

III.4.5. Stakeholder weighting

Next, stakeholder participation is held to determine the weight of vulnerabilities towards calculating an overall score for each NBS scenario. A workshop was carried out on November 25th, 2022 (URBAG, 2022), where stakeholders ranked relevant vulnerabilities for future UA planning in the AMB. Values from the ranking are displayed in **Table 1**. For details about this dynamic, please see **Appendix 1-B**, Section 7. For photographs of the workshop, please see **Appendix 1-A**, Fig A.2, A.3 and A.4.

III.4.6. Aggregation of single vulnerabilities for a combined vulnerability

Based on the weights established by stakeholders, the *single vulnerabilities* were aggregated via a weighted sum (see **Appendix 1-A**). By doing this, it was produced a *Combined vulnerability* including all indicators from all vulnerabilities (see equation in **Appendix 1-B**, Section 8). This final aggregation was repeated using equal weights to understand the robustness of the analysis and whether vulnerabilities were impacted by different weighting schemes. Similar to *Single Vulnerabilities*, the sum of pixel values and their relative change between scenarios were calculated for the *Combined Vulnerability*.

III.5. Results

Before presenting the results of the assessment, it is pertinent to analyze the stakeholder weighting outcomes, which were used to calculate the *Single Vulnerabilities* into the *Combined Vulnerability values*. The stakeholders ranked the *single vulnerabilities,* from most to least relevant, resulting in: *vulnerability of lack of local food*, *vulnerability of loss of biodiversity*, *vulnerability to heat*, *vulnerability of lacking recreational space*. The weights are shown in **Table 1**, where vul*nerability of lack of local food* was attributed 48% and *vulnerability of lacking recreational space* 9%.

III.5.1. Combined vulnerability

The spatial distribution of the *Combined Vulnerability* in scenario S0, considering stakeholder weights, is primarily concentrated in the southeast of the AMB (see **Fig. 4a**), where the Barcelona municipality is located. In this region, pixels exhibiting the highest vulnerability levels were identified, peaking at 0.42 on the scale between 0 and 1 (0 represents no vulnerability and 1 represents the maximum theoretical *Combined vulnerability*, indicating the concentration of all *Single vulnerabilities* at their maximum levels). This region gathers most of the *vulnerability of lack of local food*, *vulnerability to heat* and *vulnerability of lacking recreational space*, as it concentrates the highest population density in all AMB, making it the most sensitive area for the aforementioned vulnerabilities (see **Appendix 1-C**, section 1.12, 2.11, 3.4). Characterized by extensive built-up areas with limited UA and green spaces (see **Fig. 3**), the Barcelona municipality experiences higher exposure levels in contrast to more vegetated zones. Similarly, the southwestern AMB also presents vulnerability concentrations, although less pronounced (pixel values reaching 0.31) and less widely spread. Similar to Barcelona municipality, this area maintains consistent population densities; however, it differs in having smaller built-up areas and higher prevalence of UA and green spaces. In contrast, regions lacking sensitivity, such as the eastern parts of the AMB, primarily consisting of UA and other green areas, experience low or no vulnerability.

Examining changes across scenarios, it is observed that as UA expands, *Combined vulnerability* decreases. As shown in **Table 2**, when applying stakeholder weights, S3 - featuring the highest UA coverage - reduces vulnerability by 14.9% while S2 does so by 6%. Conversely, S1, with the smallest UA coverage, increases the *Combined vulnerability* by 3.1%. This trend persists when applying equal weights as a robustness analysis, albeit the changes between scenarios are less significant (S3 decreases by 11%, S2 by 4.4% and S1 increases by 2.5%). This common behavior can primarily be attributed to the reduced *vulnerability of lacking recreational space* and *vulnerability of lack of local food*, which outweighs the increases in *vulnerability to heat* and *vulnerability of loss of biodiversity* observed for S2 and S3 compared to S0.

Table 2. Percentage change (compared to scenario S0) of single vulnerabilities and combined vulnerability under both weighting schemes. Calculation is based on the difference in the sum of pixel values between scenarios.

	Evaluation schemes	**S1-S0**	**S2-S0**	**S3-S0**
Combined vulnerability	Stakeholder weighting	3.1%	-6.0%	-14.9%
	Equal weighting	2.5%	-4.4%	-11.0%

	Vulnerability	**S1-S0**	**S2-S0**	**S3-S0**
Single vulnerabilities	Vulnerability of lack of local food	3.5%	-7.0%	-17.6%
	Vulnerability to heat	1.0%	0.2%	0.4%
	Vulnerability of lacking recreational space	0.5%	-0.3%	-2.1%
	Vulnerability of loss of biodiversity	-19.4%	67.2%	210.0%

From a spatial perspective (see **Fig. 4**), vulnerability reductions under S3 concentrate in AMB's southeastern and southwestern areas (see **Fig. 4d**). As previously mentioned, these areas compress higher sensitivities than other AMB sections, making them more susceptible to exposure changes. Thus, decreases in exposure arising from UA expansions in these areas have a more significant impact on its vulnerability (e.g., the highest vulnerability in S3 reaches 0.36). Moreover, the expansion of UA in other sections (e.g., northern areas) also reduces vulnerabilities in these southeastern and southwestern regions. This is related to the *vulnerability of lack of local food*, as land-use changes at both the pixel level and the overall AMB influence its exposure and, consequently, its vulnerability. Yet, S3 also displays increased vulnerabilities, especially in the north-east, west and center-south of the AMB. This can be related to the concentration of *vulnerability of loss of biodiversity* in these areas, exacerbated by the substitution of other types of green areas by UA and associated phosphorous discharges increasing the exposure level. S2 shows a resembling spatial pattern to S3 but is less pronounced (see **Fig. 4c**), as fewer UA areas substitute other green spaces. Meanwhile, S1 exhibited the opposite spatial behavior (see **Fig. 4b**), confirming the link between exposure changes and UA: reductions in UA, both the local and overall AMB levels, led to increased exposure. The spatial distribution of the changes in *Combined vulnerability* under equal weights follows a similar trend to that with stakeholder weights but with a larger proportion of areas remaining unchanged (see **Appendix 1-C**, section 5).

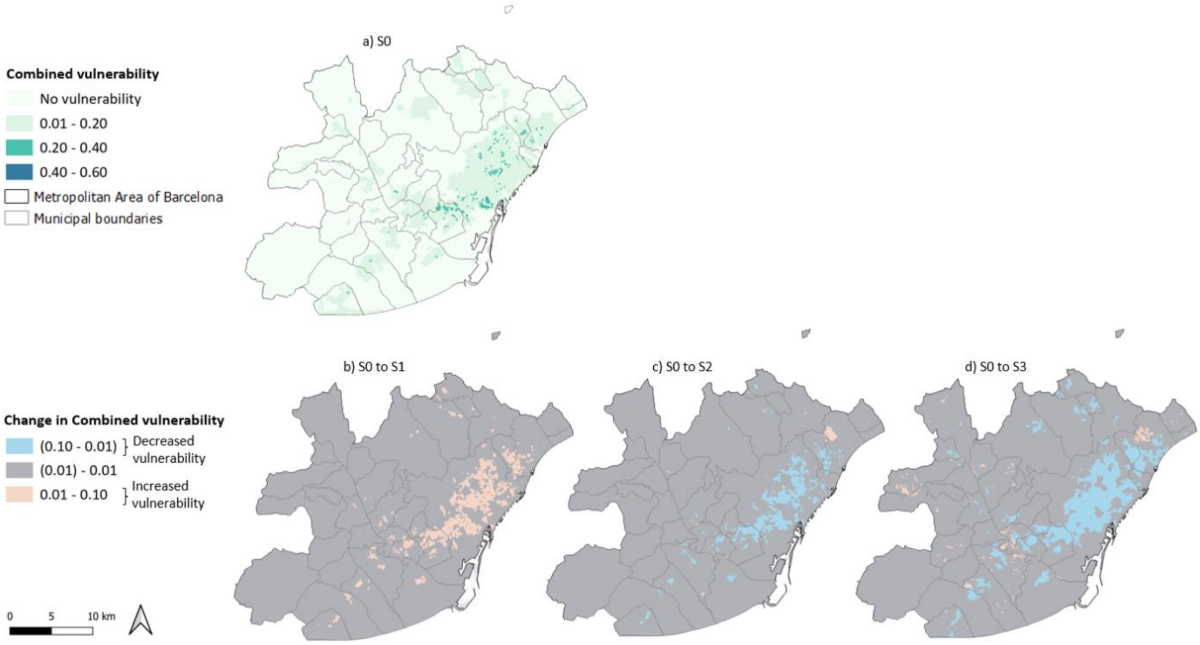


Figure 4. Spatial distribution of the Combined vulnerability and changes across scenarios with stakeholder's weights. Gray areas represent no vulnerability changes between scenarios.

III.5.2. Vulnerability of lack of local food

Vulnerability of lack of local food is the most dominant vulnerability in the AMB under the assumptions of the study (i.e., sum of pixel values; see **Appendix 1-D**). Its spatial distribution for scenario S0 (see **Fig. 5a**) concentrates the highest vulnerabilities in the southeastern AMB, where the Barcelona municipality is located. This area experiences the highest sensitivity and exposure in the AMB, resulting in pixel values of 0.79 on a 0-1 scale. The area's high sensitivity is due to its dense population, while exposure is defined by limited crop diversity and lack of local fruit/vegetable production (see **Appendix 1-C**, sections 1.1, 1.2, 1.3) as well as to the overall fruit/vegetable production at the AMB for S0. Similar conditions are observed in small patches in the north-eastern and south-western AMB. From a land-use perspective, these AMB areas are densely built up and lack UA compared with less vulnerable sections. Meanwhile, areas with similar exposure, like the southern AMB (see **Appendix 1-C**, section 1.10), do not face *vulnerability of lack of local food* due to experiencing the lowest population density in the AMB, and thus, exhibit minimal sensitivity (see **Appendix 1-C**, section 1.12).

Overall, the *vulnerability of lack of local food* is reduced by the expansion of UA (see **Table 2**). In scenarios S2 and S3, where UA is more prevalent compared to S0, vulnerability decreases by 7% and 17.6%, respectively. S1, which decreases UA by expanding built-up areas, increases vulnerability by 3.5%. This same pattern is observed for the exposure indicators (see **Table 1**). For example, the production of vegetables and fruits in the AMB significantly increases under S3 by 25,836t (65.9%) and 13,820t (148.8%) respectively.

When analyzing how these vulnerability changes are distributed across space, concentrations in sensitive areas were observed, including the south-east (Barcelona municipality), north, south-west and center (see **Fig. 5b, c, d**). In the AMB center, S1 showed increased vulnerability (see **Fig. 5b**), primarily linked to heightened exposure resulting from UA losses. Meanwhile, the Barcelona municipality experiences greater sensitivity and comparatively smaller UA reductions, witnessing extensive vulnerability increases. By contrast, S2 and S3 experienced vast vulnerability decreases in the Barcelona municipality (see **Fig. 5c, d**) even with limited increases in UA for this area. Changes in vulnerability within Barcelona municipality are also impacted by UA transformations in other areas which alter the overall exposure of the AMB. A similar trend is observed in the central-north and north-west sections, also sensitive areas, where UA increased minimally or not at all, yet vulnerability decreased for both S2 and S3. Conversely, in areas with lack of sensitivity, such as the western AMB, vulnerabilities remained unchanged despite substantial local and overall exposure changes due to UA expansion (see **Appendix 1-C**, section 1.11). Finally, *vulnerability of lack of local food* can be decreased even when UA locations do not coincide with sensitivity areas, highlighting the significance of UA quantity over its specific location.

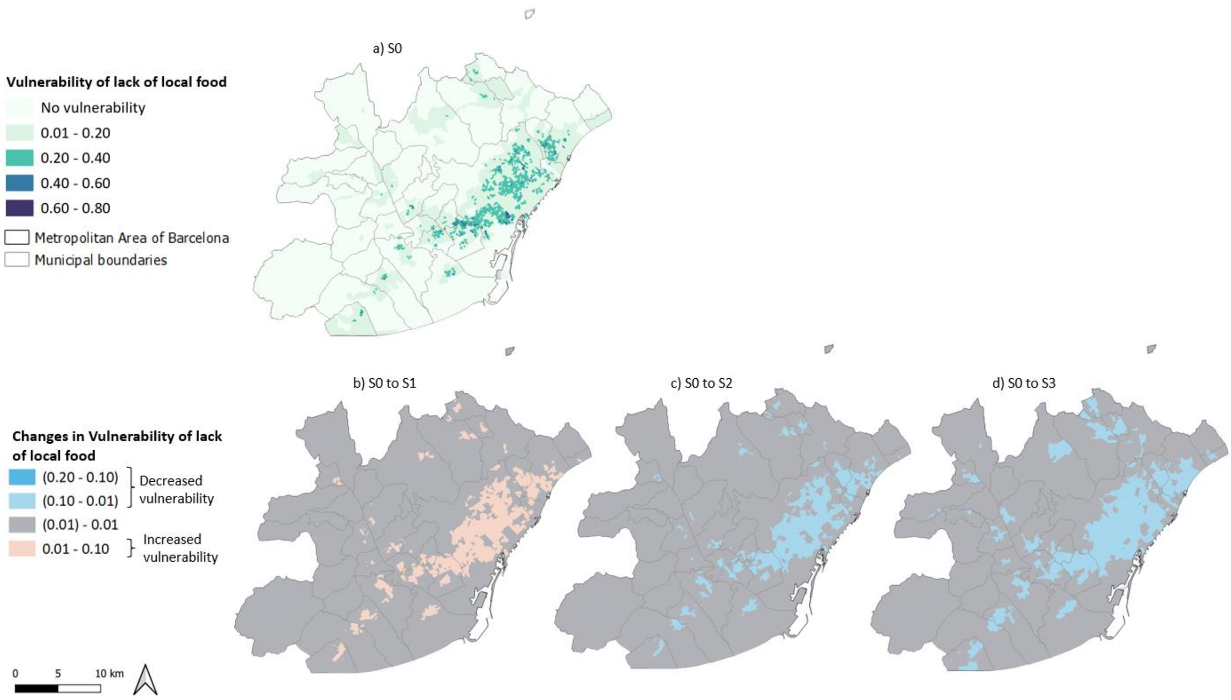


Figure 5. Spatial distribution of vulnerability of lack of local food and changes across scenarios. Gray areas represent no vulnerability changes between scenarios.

III.5.3. Vulnerability to heat

Vulnerability to heat is the second most pressing vulnerability in the AMB (see **Appendix 1-D**). For S0, this vulnerability is concentrated in the southeastern AMB (Barcelona municipality) (see **Fig. 6a**), a region characterized mostly by built-up areas and scarce green spaces compared to other AMB sections. In the southwestern AMB, some vulnerability patches are found, though they are less prominent. This is because exposure values in this area are lower (see **Appendix 1-C**, section 2.4), primarily due to fewer built-up areas. The northern AMB, characterized by urban areas and green spaces, experiences the highest exposure levels due to its lower altitude and distance from the sea, which prevents it from accessing cooling sea breezes. Remarkably, due to its lack of sensitive areas, no vulnerability is observed. In contrast, the southeastern AMB, with lower exposure values, exhibits maximum vulnerability (0.31) due to its high population density, particularly among the elderly (see **Appendix 1-C**, sections 2.9, 2.10). This area has some of the highest heatwave night temperatures in the AMB, while daytime temperatures are not as extreme (see **Appendix 1-C**, sections 2.1 and 2.2). In addition, as most day temperatures stay below their threshold (32°C), the primary factor affecting overall exposure is elevated nighttime temperatures consistently exceeding their 23°C threshold (see **Table 1**).

All potential future scenarios result in increased vulnerability: S1, marked by urban expansion and reduced UA, leads to a higher vulnerability compared to S0 (1%) (see **Table 2**). In S3 and S2, where UA expands while other green spaces decrease, vulnerability also increases, but to a lower degree (0.4% and 0.2% respectively). Examining spatial shifts across scenarios reveals an overarching trend: reductions in exposure fail to align with sensitivity hotspots within the AMB, resulting in minimal overall changes in the vulnerability across scenarios (cf. **Fig. 6b, c, d**). Despite observed temperature reductions in S2 and S3 (e.g., average day temperatures decreased by 0.01ºC in S2 and 0.03 ºC in S3; see **Table 1**), their impact on the city's overall vulnerability remained limited, as these temperature reductions did not align with sensitive zones. An example is seen in **Fig. 7c**, where exposure to heat decreases in the northern area of the AMB for scenario S3 due to the cooling effect of the irrigated agricultural fields. However, there is no sensitive population in that area, thus the expansion of UA does not result in reducing vulnerability.

Another reason S3 does not result in reducing *vulnerability to heat* as much as might be expected is because the cooling effect of the additional vegetation remains local during the day, while at night the temperature reductions are more widespread throughout the AMB. Thus, night temperatures have a more influential role in shaping exposure than day-time temperatures. This is illustrated in **Fig. 7**, which captures the differences between S0 and S3 for normalized *heatwave day* *temperatures*, *heatwave night temperatures*, and aggregated exposure. While the changes in normalized day temperatures are localized, shifts in normalized night temperatures are more evenly distributed (**Fig. 7a, b**). Despite both indicators having similar average value variations (-0.03°C and +0.01°C respectively; see **Table 1**), fluctuations in night temperatures are the primary drivers of exposure changes (see **Fig. 7c**). However, reducing exposure does not reduce vulnerability for the main reason mentioned before in this section: the reductions do not affect sensitive areas. In addition to the location of the sensitive population, the threshold value for exposure also plays an important role in vulnerability. Although reductions in absolute nighttime temperatures did occur, even within built-up areas, these remained below the 23°C threshold and consequently did not reduce the exposure.

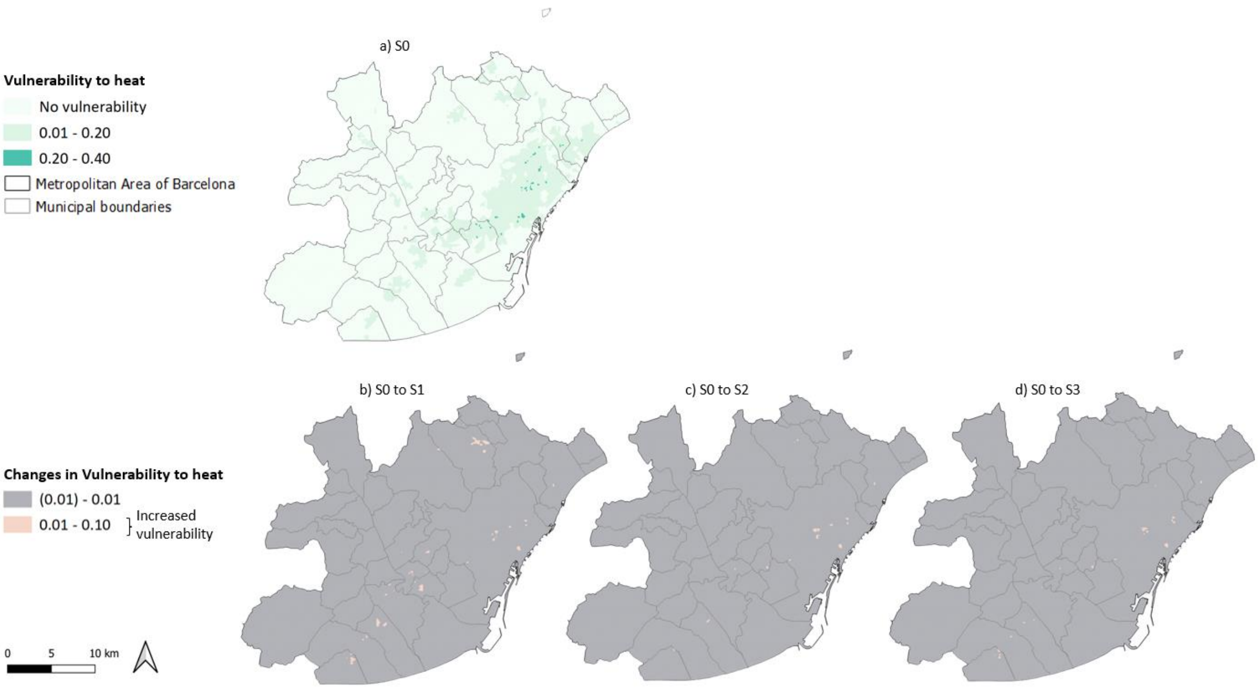


Figure 6. Spatial distribution of vulnerability to heat and changes across scenarios. Gray areas represent no vulnerability changes between scenarios.

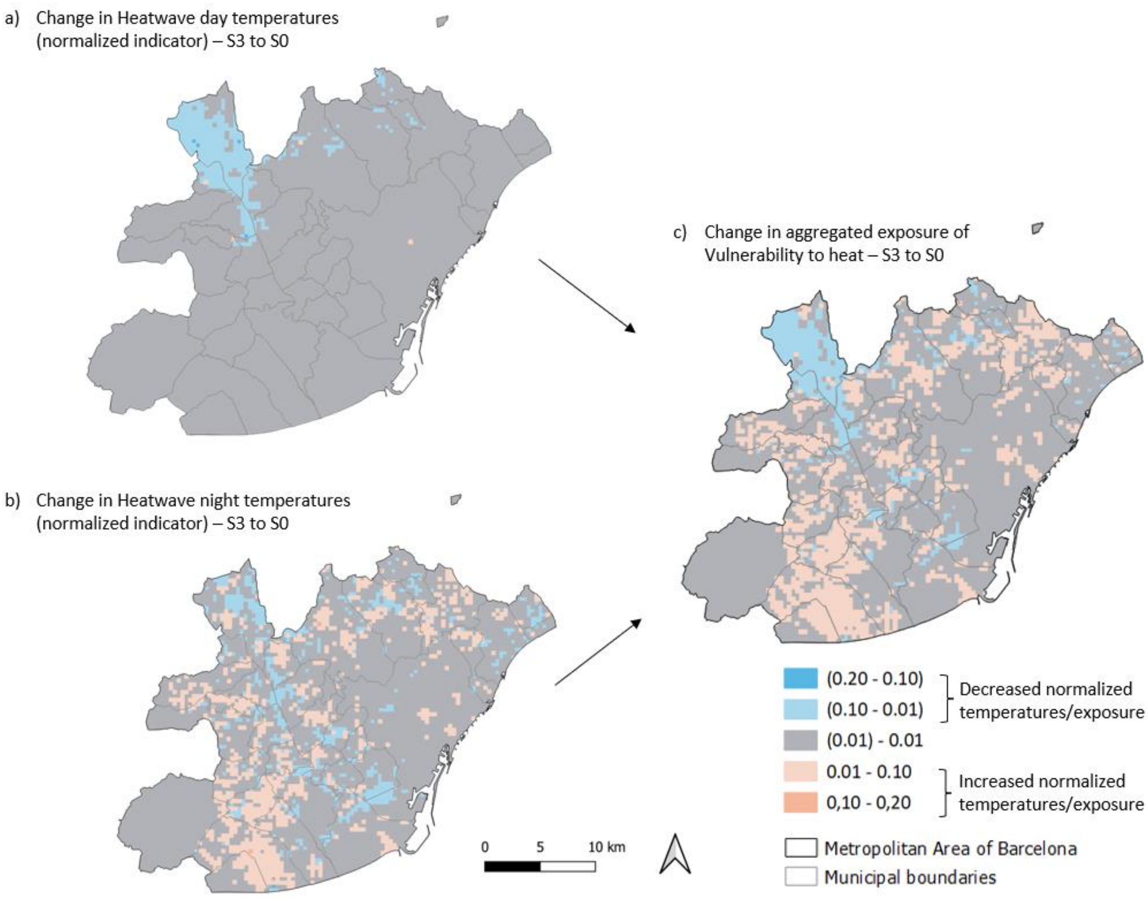


Figure 7. Spatial distribution of changes between Scenario 3 and Scenario 0 in normalized indicators Heatwave day temperatures and Heatwave night temperatures, and aggregated exposure of vulnerability to heat. Gray areas represent no change in normalized temperatures/exposure.

III.5.4. Vulnerability of lacking recreational space

Vulnerability of lacking recreational space is the third most prominent vulnerability in the AMB (see **Appendix 1-D**). In terms of location, vulnerability in S0 is concentrated in the southeastern part of the AMB (see **Fig. 8a**), where the Barcelona municipality is located. Here, pixel values reach 0.74, due to exceptionally high sensitivity attributed to the dense population and limited green spaces within 300m (see **Appendix 1-C**, section 3.1, 3.4). This vulnerability pattern owes itself to the significant presence of built-up areas and the limited availability of green spaces in comparison with other areas with lower exposure. Interestingly, the southern parts of the AMB experience higher exposure than the Barcelona municipality but lower population density, preventing this vulnerability.

We find that UA increases correlate with reductions in *vulnerability of lacking recreational space* (see **Table 2**). Notably, S2 and S3, experiencing increases in UA, decreased the vulnerability by 0.3% and 2.1% respectively. Conversely, S1, which expands built-up areas and decreases UA, increased the vulnerability by 0.5%. These trends are consistent with exposure values (cf. **Table**

1), where the total area with green spaces accessible beyond 1000m shifts from 54.7 km^2 in S0 to 65.1 km^2 in S1 (indicating increased exposure), while S3 shifts to 53.6 km^2 (the most substantial reduction in exposure). However, S2 presents an exception, with an exposure value increase (55.7 km^2 compared to 54.7 km^2 from S0). This illustrates that *vulnerability of lacking recreational space* can be reduced even when exposure increases.

The spatial changes of *vulnerability of lacking recreational space* show uneven distribution across the AMB. In S1 (see **Fig. 8b**), vulnerabilities increased in the Barcelona municipality, the most sensitive area of the AMB. These arise from an increased exposure because of the reduction in accessible green spaces associated with the expansion of built-up areas near the municipality. While spatially limited, these land uses notably affect vulnerability due to the high population density in the area. Interestingly, vulnerabilities within the Barcelona municipality decreased in S2 and S3 (see **Fig. 8c, d**) due to the strategic replacement of built-up areas with UA, leading to a vulnerability reduction despite an increased overall exposure in S2. Furthermore, the south-center region, another sensitive AMB section, displays less dispersed vulnerability changes across scenarios, attributed to more extensive land-use alterations compared to the Barcelona municipality. In S3, this area's vulnerability diminishes due to reduced exposure produced by a UA expansion replacing built-up areas. However, it is worth noting that not all reductions in built-up spaces that modify green areas lead to vulnerability shifts. For instance, the northern AMB experienced exposure reductions from increased UA in S2 and S3 (see **Appendix 1-C**, section 3.3); yet, these changes do not correspond to any sensitive area that would translate into vulnerability changes. Thus, highlighting the significance of NBS locations over their quantity.

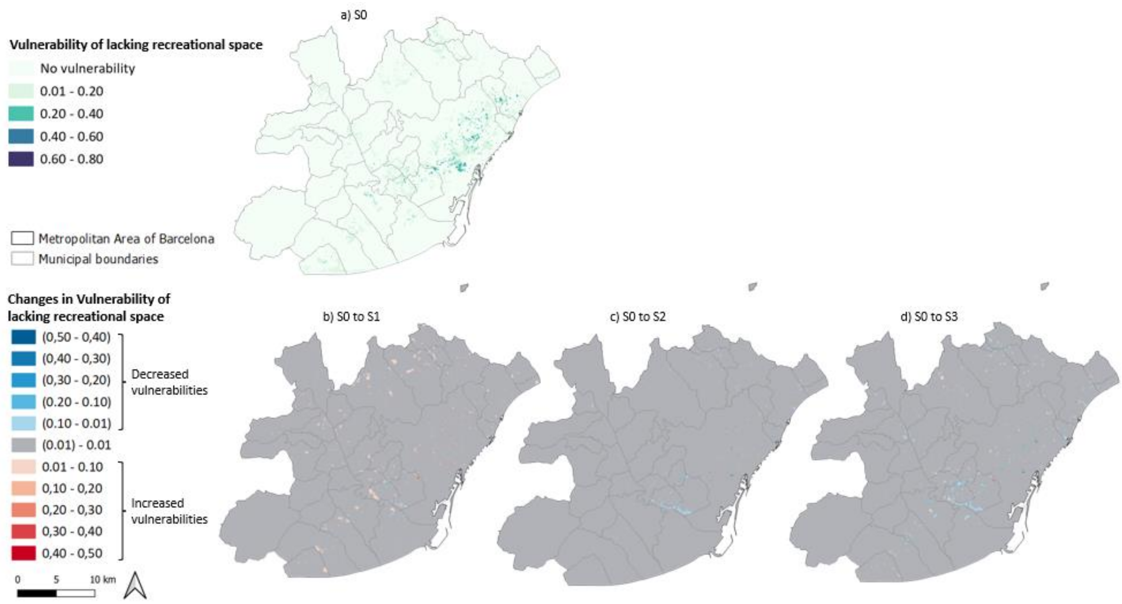


Figure 8. Spatial distribution of vulnerability of lacking recreational space and changes across scenarios. Gray areas represent no vulnerability changes between scenarios

III.5.5. Vulnerability of loss of biodiversity

Vulnerability of loss of biodiversity emerges as the least pronounced vulnerability (see **Appendix 1-D**). The spatial distribution of this vulnerability in S0 is concentrated in the southwestern AMB (see **Fig. 9a**), with pixel values reaching 0.02. This area has the highest exposure in the AMB as it concentrates the greatest amount of *Phosphorous discharges from fertilizer use* (see **Appendix 1-C**, section 4.1). However, after normalization, this indicator is limited to a pixel value of 0.05, as the overall phosphorous discharge from AMB is far from its threshold value (see **Table 1**). Regarding sensitivity, this location has a pixel value of 0.6, while the most sensitive areas typically score 0.8 (see **Appendix 1-C**, section 4.6). The eastern AMB encounters comparable exposure and sensitivity, resulting in a similar vulnerability, albeit over a smaller area. Meanwhile, the northeastern AMB experiences similar exposure values, but vulnerability does not manifest due to the lack of sensitivity. From a land-use perspective, regions exhibiting vulnerability coincide exclusively with UA areas.

Generally, this vulnerability increases as UA expands (see **Table 2**): the *vulnerability of loss of biodiversity* increases by 67.2% and 210% for S2 and S3, respectively. Conversely, S1, which reduces UA in the AMB, reduces vulnerability by -19.4%. This trend is also reflected in the exposure value *Phosphorous discharges from fertilizer use* (see **Table 1**), which escalates from 21 tons in S0 to 38 tons in S3. Despite the percentage increases in this vulnerability being larger compared to other vulnerabilities, the actual extent of these changes is limited because exposure values are relatively low when compared to its threshold. This can be appreciated as the biggest exposure value observed is 0.1 in S3 (see **Appendix 1-C**, section 4.2).

Even with limited exposure, the vulnerability's spatial distribution was altered. In S1 (see **Fig. 9b**), vulnerability is primarily reduced in the southern AMB due to an exposure reduction coming from the substitution of UA by built-up areas. Similar changes occur in the northeastern AMB. Conversely, increased vulnerability in S2 is mainly concentrated on the eastern AMB (see **Fig. 9c**). These changes arise from increased exposures linked to UA substituting other green areas, intersecting with sensitive areas. In S3, a similar land-use dynamic increased vulnerabilities in the central and north-central regions of the AMB (see **Fig. 9d**).

Yet, not all increases in UA uniformly impact vulnerabilities. In S3, northeastern AMB shifts other green areas to UA, raising its exposure. Similarly, the southeastern area, within the Barcelona municipality, experiences higher exposure as UA expands by diminishing built-up areas. However, as these UA expansions do not align with sensitive zones, vulnerability remains unchanged. This demonstrates that UA expansions can occur without escalating this vulnerability, provided they happen in low-sensitivity areas. The Barcelona municipality exemplifies this, with one of the lowest sensitivities in the AMB. These reduced sensitivities coincide mostly with the built-up areas of the AMB and its surroundings, highlighting an opportunity for UA expansion that does not increase the *vulnerability of loss of biodiversity.*

Figure 9. Spatial distribution of vulnerability of loss of biodiversity and changes across scenarios. Gray areas represent no vulnerability changes between scenarios.

III.6. Discussion

III.6.1. Land-use changes in the Metropolitan Area of Barcelona shift vulnerabilities unevenly

The NBS-vulnerability framework revealed how various agricultural configurations influenced vulnerability. Generally, UA expansions reduced vulnerabilities (e.g., S2, S3), and UA contractions raised vulnerabilities (e.g., S1). This direct relationship between enhanced UA and decreased vulnerabilities was especially evident for *vulnerability of lack of local food* which, compared to S0, was reduced by 17.6% in S3 by an increase in the UA area of 12%. The opposite was true for *vulnerability of loss of biodiversity*, as the agriculture expansion caused a 210% increase due to phosphate discharges into areas with critical biological status. More discrete changes were observed for *vulnerability to Heat* with an increase of 0.4% in S3, and for *vulnerability of lacking recreational space*, which decreased by 2.1% in S3.

Consistent with previous research on the ES socio-spatial distribution (Herreros-Cantis & McPhearson, 2021), the impacts of increased UA are influenced by their locations. Beyond the overall UA expansion, vulnerabilities are shaped by the spatial distribution of these increases. *Vulnerability of lacking recreational space* illustrates this point. In S2, exposure levels exceed those in S0 due to the decreased greenery in the AMB. Despite this, the vulnerability is reduced. This can be attributed to the redistribution of green areas, as specific built-up areas are substituted with UA across sensitive regions. Furthermore, the impact of UA locations is also present in the *vulnerability of loss of biodiversity,* which significantly increases in S3 compared to S0, despite the rise in exposure due to phosphorous discharges being somewhat smaller. This disproportionate vulnerability surge is linked to the convergence of exposure increases within sensitive areas, which intensifies its impact. *vulnerability of lack of local food,* however, presents an exception regarding how UA locations change vulnerabilities. As observed in the Barcelona municipality, the most sensitive area of the AMB, significant vulnerability shifts occurred across all scenarios despite experiencing minimal UA changes. These shifts were mostly driven by UA changes in other sections of the AMB. This outcome is attributed to the normalization method of the exposure values of this vulnerability, enabling exposure changes driven by UA shifts to affect sensitivities even when these are not geographically aligned.

This study also reveals that vulnerability changes are not always as expected, as observed in *vulnerability to heat.* While the literature agrees on the heat mitigation abilities of NBS (Shao & Kim, 2022), the impacts of the UA scenarios remain inconclusive. *Vulnerability to heat* increased in all scenarios; however, the most substantial increase occurred in S1, the scenario with the smallest amount of UA. Vulnerability increases were less in S2 and S3, where UA is more prevalent than in S0, implying that UA changes alone do not homogeneously impact this vulnerability. From a land-use perspective, the northeastern AMB experienced vulnerability increases in S1 when built-up areas replaced UA and green spaces, and in S2 when UA increased by reducing other green spaces. Similarly, the Barcelona municipality saw increased vulnerability in all scenarios, either when substituting green spaces with built-up areas or UA. These cases indicate that expanding built-up areas and converting green spaces to UA heighten the *vulnerability to heat* alike. However, this deduction requires careful interpretation, as the dynamics between land-use and temperature are influenced by various factors, including green space types, irrigation practices, wind patterns and building configurations (Segura et al., 2021). Additionally, calculating vulnerability is highly sensitive to the threshold values chosen. This is especially evident with temperature changes: an increase in nighttime temperature above the threshold significantly increased the *vulnerability to heat*, while daytime temperatures, in general, were less likely to exceed their threshold and had a comparatively smaller effect in reducing vulnerability. Slightly changing these thresholds could change these vulnerability calculations significantly.

In short, land-use changes have differentiated impacts on vulnerability. *Vulnerability of lack of local food* decreases when UA expands in high or low population density areas. For *vulnerability to heat,* new UA does not reduce the vulnerability, regardless of whether these expansions match or not with sensitive areas. Yet, vulnerability does increase if UA expansions reduce other green spaces within built-up areas. For *vulnerability of lacking recreational space*, both UA and other green space expansions are more effective at reducing vulnerability in regions with higher population density than in low or uninhabited areas. Finally, for *vulnerability of loss of biodiversity*, the creation of new UA areas does not increase it when happening within built-up environments. However, the vulnerability does increase when UA expansions happen in less urbanized regions.

III.6.2. Advancing Nature-based solutions planning through an integrated vulnerability assessment

The complexity around how to distribute NBS effectively has been recognized as a major challenge in urban NBS planning (Langemeyer et al., 2020) and yet, the integration of NBS evaluation with spatial urban planning remains partial (Pan et al., 2021). The proposed framework advances NBS planning on three main aspects: (a) NBS-vulnerability integration,

(b) spatially and context-specific impact assessment, and (c) multi-dimensional ex-ante assessment of NBS impacts.

First, it was introduced a unique interdisciplinary framework that integrates UM, ES, and spatially explicit vulnerabilities. This approach diverges from previous work by simultaneously considering these dimensions for evaluating NBS. To my knowledge, no interdisciplinary approach of this kind has been developed. Traditionally, researchers have focused on identifying vulnerable areas for NBS implementation (e.g., Baró et al., 2021) or studying the relationship between vulnerable regions and the anticipated ES supply from NBS (e.g., Langemeyer et al., 2020). Other researchers have addressed the relationship between ES demand and supply (e.g., Basnou et al., 2020) or NBS environmental impacts through UM approaches (e.g., Mendoza Beltran et al., 2022). Some studies have related UM impacts to ES or benefits from NBS (Padró et al., 2020). The simultaneous consideration of diverse outlooks has been described as necessary for NBS evaluation (Dumitru et al., 2020) and for the comprehensive assessments of land-use changes regarding urban sustainability policies (Kalantari et al., 2019). The framework meets these demands by calculating diverse NBS impacts through MCDA, a useful approach for the holistic assessments of NBS (Venter et al., 2021). This streamlines and enhances the overall understanding of NBS effects, improving the NBS planning process.

Second, the proposed framework establishes a coherent spatial integration between the fields of ES, UM and vulnerability. The case study underscores the importance of this comprehensive approach, revealing different spatial vulnerability patterns, and expanding our understanding of how NBS influence urban contexts. Furthermore, the framework focuses on context-specific objectives to identify local vulnerabilities (e.g., AMB acknowledges *vulnerability to heat* as a pressing challenge), while also considering local thresholds when calculating exposure values (e.g., excess of heat during nighttime). This approach helps avoid using standardized measurements detached from the specific context, which can lead to misleading interpretations (Kuhlicke et al., 2011) and ineffective NBS implementations. Moreover, involving stakeholders in weighting vulnerabilities enhances the framework's ability to generate customized outcomes for the local context, enabling the consideration of unique challenges and priorities of the region. This is crucial for minimizing uncertainties about NBS impacts in urban settings (Nesshöver et al., 2017).

Third, the framework aims to aid NBS planning by foreseeing various impacts (intended and unintended) via ex-ante assessments of different NBS scenarios. The ex-ante approach, advised for ensuring NBS effectiveness (Mussinelli et al., 2021), remains a critical knowledge gap in urban planning, especially at the intersection of NBS and vulnerabilities. The framework addresses this by incorporating the underlying principles of vulnerability assessments, recognizing that systems exposed to hazards manifest multiple dimensions with spatial and temporal variations (IPCC, 2012). Based on this, the framework converges diverse vulnerabilities and projects them through various NBS-driven land-use scenarios, allowing for the foresight of potential vulnerability changes.

The proposed framework constitutes an important advancement for NBS planning, offering a spatially and context-specific, ex-ante assessment approach to urban vulnerabilities. Employing vulnerabilities as a shared analytical language to interpret NBS impacts within socio-ecological systems has significant potential to help evaluate trade-offs and reduce uncertainties in NBS implementation in urban environments. Moreover, through collaborative comprehension of the various impacts of NBS on vulnerabilities, the framework allows for strategic planning to enhance urban resilience against hazards (e.g., mitigating *vulnerability to heat*) and promote sustainability (e.g., addressing the *vulnerability of loss of biodiversity*). This integrated approach positions the framework as a valuable tool for urban planners and policymakers seeking to promote effective NBS within the urban metropolitan scale.

Considering these advancements, I want to raise some methodological considerations that can enhance the future uptake of the proposed framework.

III.6.3. Considerations for the future application of the Nature-based solutions vulnerability framework

The proposed framework innovatively integrates ES and UM into a vulnerability analysis, providing spatially explicit results at different levels of detail (indicators, single vulnerabilities and combined vulnerability). This aspect represents a desirable trait for NBS assessments (Mendoza Beltran et al., 2022) that allows a differentiated understanding of its outcomes. However, the implementation of the framework highlights aspects for future improvement.

First, scenarios cannot fully capture vulnerabilities as systems and populations are not solely affected by nearby hazards. Vulnerabilities can extend beyond local boundaries through cascading effects (Little, 2010), which relates to the extent of ES supply (Metzger et al., 2005) and UM impacts (Kissinger & Stossel, 2021). For instance, Phosphorous discharges from fertilizer use associated with the UA expansion within the AMB could cause water eutrophication beyond the region, impacting the *vulnerability of loss of biodiversity* in such areas. While the NBS-vulnerability framework is limited by its spatial scope, it does allow for the contextualized consideration of vulnerabilities within this area. For instance, the normalization of the indicator *Production of vegetables in the AMB* considered the food production at the pixel level (local scale) and at the overall AMB level (regional scale) enabling the assessment of part of the cascading effects within the urban system. However, delving deeper into these dynamics could improve our understanding of NBS effects on (peri-)urban vulnerabilities.

Second, vulnerabilities cannot be grasped only by quantitative sources (Salter et al., 2010), so stakeholders' involvement is essential to reveal context-specific root causes (Schneiderbauer et al., 2017). The proposed framework incorporates participatory methods only for the weighting of vulnerabilities. A similar approach could also be applied to the weighting of indicators: instead of assigning equal weights, engaging a stakeholder panel to evaluate their relevance could offer a more robust justification for their significance in the urban context. Nevertheless, Madruga De Brito et al. (2018) suggest broader participatory approaches throughout the entire vulnerability process, not just limited to weighting stages. This would ensure the accuracy of factors like vulnerability selection and data standardization, thus enhancing the feasibility of the selected measures. For the proposed assessment, this aspect gains relevance as the selection of vulnerabilities and weights significantly impacts the results. Consequently, stakeholder input can further enhance the framework's reliability. Taking the UA evaluation as an example, stakeholders in the AMB often highlight water scarcity as a relevant concern (Pratt et al., 2019), which could complement the current assessment. However, assessing water scarcity as a vulnerability is not straightforward as UA is vulnerable to water scarcity while also being a major stressor. Engaging stakeholders in this discussion can help clarify the treatment of these vulnerabilities.

Third, the proposed assessment supports a better grasp of urban environments' complexity and their relation to NBS by placing greater emphasis on environmental justice considerations. According to Kato-Huerta & Geneletti (2023), a closer link between environmental justice principles and urban planning tools is necessary to enhance the evaluation of areas needing green interventions. The distributive equity approach relies on understanding the spatial location of environmental risks, amenities and social disadvantages. In this application, social disadvantages were not highlighted: social sensitivity indicators were represented by population densities, without considering more sophisticated demographics or the intersectionality in the sensitivity to hazards (i.e., Anguelovski et al., 2020). Furthermore, adaptative capacity proxies were not included in the assessment. Adaptive capacities can reduce sensitivities (e.g., higher household income can improve sensitivity to heat by the utilization of air conditioning) (Ortiz et al., 2022), introducing another level of complexity to the assessment. Moreover, procedural and recognitional justice aspects, centered on diverse social and cultural values and equitable engagement spaces, are crucial for ensuring environmentally just cities and effective NBS (Langemeyer & Connolly, 2020), and yet, they have not been fully integrated into the NBS evaluation frameworks (Kato-Huerta & Geneletti, 2023).

Finally, while the presented framework enhances our comprehension of NBS impacts in urban settings, there is room for enhancing its capabilities. This involves considering broader cascading effects, expanding stakeholder involvement, and further integrating environmental justice considerations. Subsequent research and applications can explore these aspects, bolstering the framework's effectiveness in addressing NBS planning in urban environments.

III.7. Conclusions

This study aimed to develop a framework for assessing NBS' impact on urban vulnerabilities, advancing beyond the net-impact assessments seen in ES and UM research. The framework employs a stepwise approach based on MCDA to estimate shifts in urban vulnerabilities across diverse land-use scenarios driven by NBS interventions. By bridging ES, UM and spatially explicit vulnerabilities analyses, the proposed assessment broadens the evaluative space for NBS in urban planning.

The application of this framework in the UA case study within the AMB showcased its effectiveness in gaining a differentiated and spatially specific comprehension of NBS impacts. It was observed that vulnerabilities exhibited multifaceted outcomes and trade-offs in their spatial distribution when responding to UA changes (e.g., agricultural expansions decreased the *vulnerability of lack of local food*, even when happening far from sensitive areas, and increased *vulnerability of loss of biodiversity*, except when confined within built-up areas).

The collaborative nature of the presented approach is expected to enhance sustainable and resilient practices in urban environments by providing a spatially explicit foresight into potential changes in socio-ecological vulnerabilities associated with NBS implementation. These estimations, characterized by their spatial specificity and alignment with context-specific objectives, foster the strategic planning of NBS at the urban metropolitan scale.

As I explore future applications of the framework for the evaluation of different types of NBS and at different urban scales, I acknowledge potential improvements that need to be considered, such as further cascading vulnerability effects, extending stakeholder involvement beyond weighting stages, and integrating further environmental justice considerations.

III. References

Anguelovski, I., Brand, A. L., Connolly, J. J. T., Corbera, E., Kotsila, P., Steil, J., Garcia-Lamarca, M., Triguero-Mas, M., Cole, H., Baró, F., Langemeyer, J., del Pulgar, C. P., Shokry, G., Sekulova, F., & Argüelles Ramos, L. (2020). Expanding the Boundaries of Justice in Urban Greening Scholarship: Toward an Emancipatory, Antisubordination, Intersectional, and Relational Approach. Annals of the American Association of Geographers, *110(6), 1743–1769. https://doi.org/10.1080/24694452.2020.1740579*

Bahers, J. B., Athanassiadis, A., Perrotti, D., & Kampelmann, S. (2022). The place of space in urban metabolism research: Towards a spatial turn? A review and future agenda. *Landscape and Urban Planning*, *221*, 104376. https://doi.org/10.1016/J.LANDURBPLAN.2022.104376

Barcelona Metropolitan Area - AMB. (2018). *Climate change adaptation plan for the Barcelona Metropolitan Area*.

Barcelona Regional and AMB-PDU. (2023). *Strategic environmental study*. https://urbanisme.amb.cat/informar-se/document-aprovacio-inicial

Baró, F., Camacho, D. A., Pérez Del Pulgar, C., Triguero-Mas, M., & Anguelovski, I. (2021). School greening: Right or privilege? Examining urban nature within and around primary schools through an equity lens. *Landscape and Urban Planning*, *208*, 104019. https://doi.org/10.1016/J.LANDURBPLAN.2020.104019

Baró, F., Chaparro, L., Gómez-Baggethun, E., Langemeyer, J., Nowak, D. J., & Terradas, J. (2014). Contribution of ecosystem services to air quality and climate change mitigation policies: The case of urban forests in Barcelona, Spain. *Ambio*, *43*(4), 466–479. https://doi.org/10.1007/S13280-014-0507-X/TABLES/4

Basnou, C., Baró, F., Langemeyer, J., Castell, C., Dalmases, C., & Pino, J. (2020). Advancing the green infrastructure approach in the Province of Barcelona: integrating biodiversity, ecosystem functions and services into landscape planning. *Urban Forestry & Urban Greening*, *55*, 126797. https://doi.org/10.1016/J.UFUG.2020.126797

Bauwelinck, M., Casas, L., Nawrot, T. S., Nemery, B., Trabelsi, S., Thomas, I., Aerts, R., Lefebvre, W., Vanpoucke, C., Van Nieuwenhuyse, A., Deboosere, P., & Vandenheede, H. (2021). Residing in urban areas with higher green space is associated with lower mortality risk: A census-based cohort study with ten years of follow-up. Environment International, 148, 106365. https://doi.org/10.1016/J.ENVINT.2020.106365

Chrysoulakis, N., Somarakis, G., Stagakis, S., Mitraka, Z., Wong, M.-S., & Ho, H.-C. (2021). Monitoring and Evaluating Nature-Based Solutions Implementation in Urban Areas by

Means of Earth Observation. *Remote Sensing 2021, Vol. 13, Page 1503*, *13*(8), 1503. https://doi.org/10.3390/RS13081503

Cohen-Shacham, E., Andrade, A., Dalton, J., Dudley, N., Jones, M., Kumar, C., Maginnis, S., Maynard, S., Nelson, C. R., Renaud, F. G., Welling, R., & Walters, G. (2019). Core principles for successfully implementing and upscaling Nature-based Solutions. *Environmental Science & Policy*, *98*, 20–29. https://doi.org/10.1016/J.ENVSCI.2019.04.014

Cutter, S. L. (2016). Resilience to What? Resilience for Whom? *The Geographical Journal*, *182*(2), 110–113. https://doi.org/10.1111/GEOJ.12174

del Río, S., Fraile, R., Herrero, L., & Penas, A. (2007). Analysis of recent trends in mean maximum and minimum temperatures in a region of the NW of Spain (Castilla y León). *Theoretical and Applied Climatology*, *90*(1–2), 1–12. https://doi.org/10.1007/S00704-006-0278-9/METRICS

Díaz, J., Carmona, R., Mirón, I. J., Ortiz, C., León, I., & Linares, C. (2015). Geographical variation in relative risks associated with heat: Update of Spain's Heat Wave Prevention Plan. *Environment International*, *85*, 273–283. https://doi.org/10.1016/J.ENVINT.2015.09.022

Dorst, H., van der Jagt, A., Raven, R., & Runhaar, H. (2019). Urban greening through nature-based solutions – Key characteristics of an emerging concept. *Sustainable Cities and Society*, *49*, 101620. https://doi.org/10.1016/J.SCS.2019.101620

Dumitru, A., Frantzeskaki, N., & Collier, M. (2020). Identifying principles for the design of robust impact evaluation frameworks for nature-based solutions in cities. *Environmental Science & Policy*, *112*, 107–116. https://doi.org/10.1016/J.ENVSCI.2020.05.024

Fernandes, J. P., & Guiomar, N. (2018). Nature-based solutions: The need to increase the knowledge on their potentialities and limits. *Land Degradation & Development*, *29*(6), 1925–1939. https://doi.org/10.1002/LDR.2935

Firbank, L. G., Petit, S., Smart, S., Blain, A., & Fuller, R. J. (2007). Assessing the impacts of agricultural intensification on biodiversity: a British perspective. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *363*(1492), 777–787. https://doi.org/10.1098/RSTB.2007.2183

Fisher, B., Turner, R. K., & Morling, P. (2009). Defining and classifying ecosystem services for decision making. *Ecological Economics*, *68*(3), 643–653. https://doi.org/10.1016/J.ECOLECON.2008.09.014

Grazuleviciene, R., Dedele, A., Danileviciute, A., Vencloviene, J., Grazulevicius, T., Andrusaityte, S., Uzdanaviciute, I., & Nieuwenhuijsen, M. J. (2014). The influence of proximity to city parks on blood pressure in early pregnancy. International Journal of Environmental Research and Public Health, 11(3), 2958–2972. https://doi.org/10.3390/IJERPH110302958

European Environmental Agency. (2020). *Is Europe living within the limits of our planet?* https://doi.org/10.2800/890673

Grunewald, K., Richter, B., Meinel, G., Herold, H., & Syrbe, R. U. (2017). Proposal of indicators regarding the provision and accessibility of green spaces for assessing the ecosystem service "recreation in the city" in Germany. *International Journal of Biodiversity Science, Ecosystem Services and Management*, *13*(2), 26–39. https://doi.org/10.1080/21513732.2017.1283361

He, H. ming, Liu, L. na, Munir, S., Bashir, N. H., Wang, Y., Yang, J., & LI, C. yun. (2019). Crop diversity and pest management in sustainable agriculture. *Journal of Integrative Agriculture*, *18*(9), 1945–1952. https://doi.org/10.1016/S2095-3119(19)62689-4

Heaviside, C., Vardoulakis, S., & Cai, X. M. (2016). Attribution of mortality to the urban heat island during heatwaves in the West Midlands, UK. *Environmental Health: A Global Access Science Source*, *15*(1), 49–59. https://doi.org/10.1186/S12940-016-0100-9/TABLES/3

Herreros-Cantis, P., & McPhearson, T. (2021). Mapping supply of and demand for ecosystem services to assess environmental justice in New York City. *Ecological Applications*, *31*(6), e02390. https://doi.org/10.1002/EAP.2390

IDESCAT. (2020). *Population in the municipalities in 1st January of 2020*.

IERMB Institut d'Estudis Regionals i Metropolitans de Barcelona. (2016). *AGRICULTURA METROPOLITANA - Agricultura urbana i periurbana a l'àmbit metropolità de Barcelona: beneficis econòmics, socials i ambientals. Actuació 3.1.1*.

Inèdit. (2022). *Petjada de carboni de l'alimentació a Barcelona*. https://ajuntament.barcelona.cat/premsa/wp-content/uploads/2021/11/Presentatio_PC_Alimentacio_RDP081121-final.pdf

IPCC. (2012). *Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation: A Special Report of Working Groups I and II of the Intergovernmental Panel on Climate Change*.

Kalantari, Z., Santos Ferreira, C. S., Page, J., Goldenberg, R., Olsson, J., & Destouni, G. (2019). Meeting sustainable development challenges in growing cities: Coupled social-ecological systems modeling of land use and water changes. *Journal of Environmental Management*, *245*, 471–480. https://doi.org/10.1016/J.JENVMAN.2019.05.086

Kato-Huerta, J., & Geneletti, D. (2023). *Analysing the treatment of environmental justice and nature-based solutions in the Urban Climate Action Plans of Latin American metropolitan areas*. https://doi.org/10.1080/13549839.2023.2221431

Kennedy, C., Cuddihy, J., & Engel-Yan, J. (2007). The Changing Metabolism of Cities. *Journal of Industrial Ecology*, *11*(2), 43–59. https://doi.org/10.1162/JIE.2007.1107

Kissinger, M., & Stossel, Z. (2021). An integrated, multi-scale approach for modelling urban metabolism changes as a means for assessing urban sustainability. *Sustainable Cities and Society*, *67*, 102695. https://doi.org/10.1016/J.SCS.2020.102695

Kovats, R. S., & Kristie, L. E. (2006). Heatwaves and public health in Europe. *European Journal of Public Health*, *16*(6), 592–599. https://doi.org/10.1093/EURPUB/CKL049

Kueppers, L. M., Snyder, M. A., & Sloan, L. C. (2007). Irrigation cooling effect: Regional climate forcing by land-use change. *Geophysical Research Letters*, *34*(3). https://doi.org/10.1029/2006GL028679

Kuhlicke, C., Scolobig, A., Tapsell, S., Steinführer, A., & de Marchi, B. (2011). Contextualizing social vulnerability: Findings from case studies across Europe. *Natural Hazards*, *58*(2), 789–810. https://doi.org/10.1007/S11069-011-9751-6/TABLES/7

Langemeyer, J., & Connolly, J. J. T. (2020). Weaving notions of justice into urban ecosystem services research and practice. *Environmental Science and Policy*, *109*(April), 1–14. https://doi.org/10.1016/j.envsci.2020.03.021

Langemeyer, J., & Gómez-Baggethun, E. (2017). *Urban biodiversity and ecosystem services* (pp. 36–53). https://doi.org/10.9774/gleaf.9781315402581_4

Langemeyer, J., Madrid-Lopez, C., Mendoza Beltran, A., & Villalba Mendez, G. (2021). Urban agriculture — A necessary pathway towards urban resilience and global sustainability?

Landscape and Urban Planning, *210*, 104055. https://doi.org/10.1016/J.LANDURBPLAN.2021.104055

Langemeyer, J., Wedgwood, D., McPhearson, T., Baró, F., Madsen, A. L., & Barton, D. N. (2020). Creating urban green infrastructure where it is needed – A spatial ecosystem service-based decision analysis of green roofs in Barcelona. *Science of The Total Environment*, *707*, 135487. https://doi.org/10.1016/J.SCITOTENV.2019.135487

Lehmann, S., Szopinska-Mularz, M., García-Blanco, G., Navarro, D., & Feliu, E. (2023). Adopting Resilience Thinking through Nature-Based Solutions within Urban Planning: A Case Study in the City of València. *Buildings 2023, Vol. 13, Page 1317*, *13*(5), 1317. https://doi.org/10.3390/BUILDINGS13051317

Liang, H., & Xie, Q. (2022). System Vulnerability Analysis Simulation Model for Substation Subjected to Earthquakes. *IEEE Transactions on Power Delivery*, *37*(4), 2684–2692. https://doi.org/10.1109/TPWRD.2021.3114279

Little, R. G. (2010). Controlling Cascading Failure: Understanding the Vulnerabilities of Interconnected Infrastructures. *Http://Dx.Doi.Org/10.1080/106307302317379855*, *9*(1), 109–123. https://doi.org/10.1080/106307302317379855

Madruga De Brito, M., Evers, M., Delos, A., & Almoradie, S. (2018). Participatory flood vulnerability assessment: a multi-criteria approach. *Hydrol. Earth Syst. Sci*, *22*, 373–390. https://doi.org/10.5194/hess-22-373-2018

Marttunen, M., Lienert, J., & Belton, V. (2017). Structuring problems for Multi-Criteria Decision Analysis in practice: A literature review of method combinations. *European Journal of Operational Research*, *263*(1), 1–17. https://doi.org/10.1016/J.EJOR.2017.04.041

Meerow, S., & Newell, J. P. (2017). Spatial planning for multifunctional green infrastructure: Growing resilience in Detroit. *Landscape and Urban Planning*, *159*, 62–75. https://doi.org/10.1016/j.landurbplan.2016.10.005

Mees, H. L. P., Driessen, P. P. J., & Runhaar, H. A. C. (2015). "Cool" governance of a "hot" climate issue: public and private responsibilities for the protection of vulnerable citizens against extreme heat. *Regional Environmental Change*, *15*(6), 1065–1079. https://doi.org/10.1007/S10113-014-0681-1/METRICS

Mendoza Beltran, A., Jepsen, K., Rufí-Salís, M., Ventura, S., Madrid Lopez, C., & Villalba, G. (2022). Mapping direct N2O emissions from peri-urban agriculture: The case of the Metropolitan Area of Barcelona. *Science of The Total Environment*, *822*, 153514. https://doi.org/10.1016/J.SCITOTENV.2022.153514

Metzger, M. J., Leemans, R., & Schröter, D. (2005). A multidisciplinary multi-scale framework for assessing vulnerabilities to global change. *International Journal of Applied Earth Observation and Geoinformation*, *7*(4), 253–267. https://doi.org/10.1016/J.JAG.2005.06.011

Mussinelli, E., Tartaglia, A., Castaldo, G., & Cerati, D. (2021). Project and evaluation of nature-based solutions for the regeneration of public space. *Smart Innovation, Systems and Technologies*, *178 SIST*, 1342–1351. https://doi.org/10.1007/978-3-030-48279-4_125/COVER

Nesshöver, C., Assmuth, T., Irvine, K. N., Rusch, G. M., Waylen, K. A., Delbaere, B., Haase, D., Jones-Walters, L., Keune, H., Kovacs, E., Krauze, K., Külvik, M., Rey, F., van Dijk, J., Vistad, O. I., Wilkinson, M. E., & Wittmer, H. (2017). The science, policy and practice of nature-based solutions: An interdisciplinary perspective. In *Science of the Total Environment* (Vol. 579, pp. 1215–1227). Elsevier B.V. https://doi.org/10.1016/j.scitotenv.2016.11.106

OECD, & European Union. (2008). *Handbook on Constructing Composite Indicators: Methodology and User Guide*. OECD. https://doi.org/10.1787/9789264043466-en

Ortiz, L., Gamarro, H., Gonzalez, J. E., & McPhearson, T. (2022). Energy burden and air conditioning adoption in New York City under a warming climate. *Sustainable Cities and Society*, *76*, 103465. https://doi.org/10.1016/J.SCS.2021.103465

Otero Peña, D., Perrotti, D., & Mohareb, E. (2022). Advancing urban metabolism studies through GIS data: Resource flows, open space networks, and vulnerable communities in Mexico City. *Journal of Industrial Ecology*, *26*(4), 1333–1349. https://doi.org/10.1111/JIEC.13261

Padró, R., La Rota-Aguilera, M. J., Giocoli, A., Cirera, J., Coll, F., Pons, M., Pino, J., Pili, S., Serrano, T., Villalba, G., & Marull, J. (2020). Assessing the sustainability of contrasting land use scenarios through the Socioecological Integrated Analysis (SIA) of the metropolitan green infrastructure in Barcelona. *Landscape and Urban Planning*, *203*, 103905. https://doi.org/10.1016/J.LANDURBPLAN.2020.103905

Pan, H., Page, J., Cong, C., Barthel, S., & Kalantari, Z. (2021). How ecosystems services drive urban growth: Integrating nature-based solutions. *Anthropocene*, *35*, 100297. https://doi.org/10.1016/J.ANCENE.2021.100297

Paquet, C., Orschulok, T. P., Coffee, N. T., Howard, N. J., Hugo, G., Taylor, A. W., Adams, R. J., & Daniel, M. (2013). Are accessibility and characteristics of public open spaces associated with a better cardiometabolic health? *Landscape and Urban Planning*, *118*, 70–78. https://doi.org/10.1016/J.LANDURBPLAN.2012.11.011

Parada, F., Gabarrell, X., Rufí-Salís, M., Arcas-Pilz, V., Muñoz, P., & Villalba, G. (2021). Optimizing irrigation in urban agriculture for tomato crops in rooftop greenhouses. *Science of The Total Environment*, *794*, 148689. https://doi.org/10.1016/J.SCITOTENV.2021.148689

Pereira, P., Yin, C., & Hua, T. (2023). Nature-based solutions, ecosystem services, disservices, and impacts on well-being in urban environments. *Current Opinion in Environmental Science & Health*, *33*, 100465. https://doi.org/10.1016/J.COESH.2023.100465

Perrotti, D., & Stremke, S. (2020). Can urban metabolism models advance green infrastructure planning? Insights from ecosystem services research. *Environment and Planning B: Urban Analytics and City Science*, *47*(4), 678–694. https://doi.org/10.1177/2399808318797131

Potter, A., & LeBuhn, G. (2015). Pollination service to urban agriculture in San Francisco, CA. *Urban Ecosystems*, *18*(3), 885–893. https://doi.org/10.1007/S11252-015-0435-Y/FIGURES/2

Pratt, T., Allen, L. N., Rosenberg, D. E., Keller, A. A., & Kopp, K. (2019). Urban agriculture and small farm water use: Case studies and trends from Cache Valley, Utah. *Agricultural Water Management*, *213*, 24–35. https://doi.org/10.1016/J.AGWAT.2018.09.034

Raymond, C. M., Frantzeskaki, N., Kabisch, N., Berry, P., Breil, M., Nita, M. R., Geneletti, D., & Calfapietra, C. (2017). A framework for assessing and implementing the co-benefits of nature-based solutions in urban areas. *Environmental Science & Policy*, *77*, 15–24. https://doi.org/10.1016/J.ENVSCI.2017.07.008

Reed, M. S. (2008). Stakeholder participation for environmental management: A literature review. *Biological Conservation*, *141*(10), 2417–2431. https://doi.org/10.1016/j.biocon.2008.07.014

Redman, C. L., Grove, J. M., & Kuby, L. H. (2004). Integrating social science into the Long-Term Ecological Research (LTER) Network: Social dimensions of ecological change and ecological dimensions of social change. *Ecosystems*, *7*(2), 161–171

Reid, C. E., Clougherty, J. E., Shmool, J. L. C., & Kubzansky, L. D. (2017). Is All Urban Green Space the Same? A Comparison of the Health Benefits of Trees and Grass in New York City. *International Journal of Environmental Research and Public Health 2017, Vol. 14, Page 1411*, *14*(11), 1411. https://doi.org/10.3390/IJERPH14111411

Royé, D. (2017). The effects of hot nights on mortality in Barcelona, Spain. *International Journal of Biometeorology*, *61*(12), 2127–2140. https://doi.org/10.1007/S00484-017-1416-Z/METRICS

Rufí-Salís, M., Brunnhofer, N., Petit-Boix, A., Gabarrell, X., Guisasola, A., & Villalba, G. (2020). Can wastewater feed cities? Determining the feasibility and environmental burdens of struvite recovery and reuse for urban regions. *Science of The Total Environment*, *737*, 139783. https://doi.org/10.1016/J.SCITOTENV.2020.139783

Salter, J., Robinson, J., & Wiek, A. (2010). Participatory methods of integrated assessment—a review. *Wiley Interdisciplinary Reviews: Climate Change*, *1*(5), 697–717. https://doi.org/10.1002/WCC.73

Schneiderbauer, S., Calliari, E., Eidsvig, U., & Hagenlocher, M. (2017). *The most recent view of vulnerability* (pp. 68–82).

Segura, R., Badia, A., Ventura, S., Gilabert, J., Martilli, A., & Villalba, G. (2021). Sensitivity study of PBL schemes and soil initialization using the WRF-BEP-BEM model over a Mediterranean coastal city. *Urban Climate*, *39*, 100982. https://doi.org/10.1016/J.UCLIM.2021.100982

Shao, H., & Kim, G. (2022). A Comprehensive Review of Different Types of Green Infrastructure to Mitigate Urban Heat Islands: Progress, Functions, and Benefits. *Land 2022, Vol. 11, Page 1792*, *11*(10), 1792. https://doi.org/10.3390/LAND11101792

Stigsdotter, U. K., Randrup, T. B., Ekholm, O., Schipperijn, J., Toftager, M., & Kamper-Jørgensen, F. (2010). Health promoting outdoor environments--associations between green space, and health, health-related quality of life and stress based on a Danish national representative survey. Scandinavian Journal of Public Health, 38(4), 411–417. https://doi.org/10.1177/1403494810367468

Thiault, L., Marshall, P., Gelcich, S., Collin, A., Chlous, F., & Claudet, J. (2018). Mapping social–ecological vulnerability to inform local decision making. *Conservation Biology*, *32*(2), 447–456. https://doi.org/10.1111/COBI.12989

Tonini, P., Odina, P. M., Orsini, F., & Durany, X. G. (2022). Economic benefit and social impact derived by a food loss prevention strategy in the vegetable sector: A spatial and temporal analysis at the regional level. *Frontiers in Sustainable Food Systems*, *6*, 1043591. https://doi.org/10.3389/FSUFS.2022.1043591/BIBTEX

Triguero-Mas, M., Dadvand, P., Cirach, M., Martínez, D., Medina, A., Mompart, A., Basagaña, X., Gražulevičiene, R., & Nieuwenhuijsen, M. J. (2015). Natural outdoor environments and mental and physical health: Relationships and mechanisms. *Environment International*, *77*, 35–41. https://doi.org/10.1016/J.ENVINT.2015.01.012

URBAG. (2022). Agricultural perspectives in the Metropolitan Area of Barcelona.https://urbag.eu/wp-content/uploads/2023/03/UA-AMB_Report_01_EN.pdf. Accessed October 1, 2023

Tzoulas, K., Galan, J., Venn, S., Dennis, M., Pedroli, B., Mishra, H., Haase, D., Pauleit, S., Niemelä, J., & James, P. (2021). A conceptual model of the social–ecological system of nature-based solutions in urban environments. *Ambio*, *50*(2), 335–345. https://doi.org/10.1007/S13280-020-01380-2/TABLES/3

United Nations. (2022). *Resolution adopted by the United Nations Environment Assembly on 2 March 2022*.

Vos, S., Bijnens, E. M., Renaers, E., Croons, H., Van Der Stukken, C., Martens, D. S., Plusquin, M., & Nawrot, T. S. (2022). Residential green space is associated with a buffering effect on stress responses during the COVID-19 pandemic in mothers of young children, a prospective study. Environmental Research, 208, 112603. https://doi.org/10.1016/J.ENVRES.2021.112603

Venter, Z. S., Barton, D. N., Martinez-Izquierdo, L., Langemeyer, J., Baró, F., & McPhearson, T. (2021). Interactive spatial planning of urban green infrastructure – Retrofitting green roofs where ecosystem services are most needed in Oslo. *Ecosystem Services*, *50*(June), 101314. https://doi.org/10.1016/j.ecoser.2021.101314

Wilhelm, J. A., & Smith, R. G. (2018). Ecosystem services and land sparing potential of urban and peri-urban agriculture: A review. *Renewable Agriculture and Food Systems*, *33*(5), 481–494. https://doi.org/10.1017/S1742170517000205

Zingraff-Hamed, A., Hüesker, F., Lupp, G., Begg, C., Huang, J., Oen, A., Vojinovic, Z., Kuhlicke, C., & Pauleit, S. (2020). Stakeholder Mapping to Co-Create Nature-Based Solutions: Who Is on Board? *Sustainability 2020, Vol. 12, Page 8625*, *12*(20), 8625. https://doi.org/10.3390/SU12208625

Zumkehr, A., & Campbell, J. E. (2015). The potential for local croplands to meet US food demand. *Frontiers in Ecology and the Environment*, *13*(5), 244–248. https://doi.org/10.1890/140246

Zuniga-Teran, A. A., Gerlak, A. K., Mayer, B., Evans, T. P., & Lansey, K. E. (2020). Urban resilience and green infrastructure systems: towards a multidimensional evaluation. *Current Opinion in Environmental Sustainability*, *44*, 42–47. https://doi.org/10.1016/J.COSUST.2020.05.001

CHAPTER IV. UNDERSTANDING NATURE-BASED SOLUTIONS IMPACTS WITHIN AND BEYOND URBAN LIMITS: AN INTEGRATED VULNERABILITY ASSESSMENT

IV. Abstract

Nature-based solutions (NBS) are strongly promoted for improving socio-ecological challenges in urban environments. Yet, multi-scale socio-ecological impacts of NBS remain understudied, and hence unconsidered in NBS implementation. To tackle this research and planning gap, it was developed a stepwise multi-criteria analysis approach to assess cross-scale NBS impacts on the capacity of NBS to alter social-ecological vulnerabilities. The framework was applied to the NBS case study of green roofs in Oslo. NBS impacts were calculated based on spatially explicit green roofs' scenarios and related to local-scale vulnerabilities – heat, heavy rainfall, lack of habitat for pollinators, air pollution, and lack of nature interaction – via spatial indicators (exposure/sensitivity) and to broad-scale vulnerabilities – climate change, stratospheric ozone depletion, novel entities and changes in biochemical flows – by considering their effects on planetary boundaries. Impacts on vulnerabilities were then weighed by stakeholders for the development of an NBS configuration where the desired impacts are maximized while the undesired are minimized. Results show that, while green roofs were effective in reducing local-scale vulnerabilities (e.g., to heatwave and extreme rainwater events), they increased broad-scale vulnerabilities (e.g., to climate change) due to the environmental impacts associated with their construction, implementation, and removal. This approach provides a novel and integrated framework to examine cross-scale trade-offs and synergies of NBS, supporting planning for sustainable, resilient, and equitable cities with both a local to global perspective.

Keywords: Nature-based solutions; Vulnerability assessment; Urban vulnerability; Urban greening; Planetary boundaries; Risk; Cross-scale

List of abbreviations

LCA: Life-cycle assessment

NBS: Nature-based solutions

MCDA: Multi-criteria decision analysis

PB: Planetary boundaries

S0: Green roof scenario “Current”

S1: Green roof scenario “Green roof strategy”

S2: Green roof scenario “Ambitious”

S3: Green roof scenario “Maximization”

S4: Green roof scenario “Most favorable”

IV.1. Introduction

Nature-based solutions (NBS) are quickly gaining prominence in policy and planning for addressing diverse urban socio-ecological challenges (Bush & Doyon, 2019; Cohen-Shacham et al., 2016), not least in the context of climate change adaptation (Hanson et al., 2020; Randrup et al., 2020). Yet, NBS are primarily planned and implemented with a local focus, with their potential broader impacts, such as their requirements for water and materials, often overlooked (Taguchi et al., 2020). As urban NBS are increasingly mainstreaming, this omission might cause unexpected and unintended negative impacts of global scale (Rödl & Arlati, 2022). The example of urban agriculture can illustrate this omission. Urban agriculture serves as an NBS for local food security in cities, enhancing urban resilience (Gulyas & Edmondson, 2021). In addition, urban agriculture produces multiple ecosystem services, such as runoff mitigation, urban cooling, social cohesion and community empowerment (Langemeyer et al., 2021). However, it may also exhibit negative environmental impacts related to their use of material resources (e.g., fuel, fertilizer, water), posing sustainability challenges at regional to global scale (Hawes et al., 2024; McDougall et al., 2019). To better understand the various impacts of NBS at different spatial scales, more comprehensive NBS assessments are necessary (Dumitru et al., 2020). Considering impacts being experienced within and beyond urban boundaries promises to prevent unforeseen tradeoffs across spatial scales with regard to the urban and global challenges of sustainability (i.e., long-term viability), resilience (i.e., capacity to adapt and recover from shocks) and equity (i.e., inclusive and just distribution of risks and benefits) (United Nations, 2015).

In this study, I assess urban NBS by introducing a novel multi-criteria decision analysis framework that considers the cross-scale impacts of NBS on both local and global vulnerability. Here, vulnerability is defined as the sociological and ecological susceptibility to harm (Cutter, 2016). While most NBS assessments are still widely based on the evaluation of ecosystem services provision or demand, efforts to relate NBS assessment to vulnerability is a more recent endeavor. Shah et al. (2020), for instance, selected and designed different types of NBS for their capacity to reduce flood vulnerabilities based on stakeholder knowledge. Herreros-Cantis & McPhearson (2021) conducted a spatially explicit assessment of areas where risk reduction could be addressed through the provision of ecosystem services. Furthermore, Camacho-Caballero et al. (2024) calculated possible shifts in urban vulnerabilities based on urban agriculture interventions.

Different from ‘net-benefit’ ecosystem services assessments, the vulnerability approach operates on the premise that NBS can either heighten or diminish vulnerabilities Herreros-Cantis & McPhearson (2021). For instance, local-scale vulnerabilities, such as heat exposure or a lack of recreational space, can be decreased or increased by the presence of NBS (e.g., Panno et al., 2017; van den Bosch & Ode Sang, 2017). Similarly, broad-scale vulnerabilities, like global warming and eutrophication, can also be negatively or positively affected by NBS (e.g., Álvarez-Rogel et al., 2020; Canadell & Raupach, 2008). Although there is growing recognition of vulnerabilities at the local level in NBS research, the broader-scale impacts of NBS continue to be largely disregarded. Furthermore, there appears to be a deficiency in an integrated approach that considers both local and broader-scale impacts of NBS (Raymond et al., 2017).

Impacts of urban NBS at broader scales often depend on material flows well beyond the city limits. These include carbon stocks (Keith et al., 2021), water, energy and materials (Bellezoni et al., 2021), which have been explored using different approaches. For instance, Wang et al. (2017) assessed and mapped the carbon capture and storage capabilities of global forests. An emerging field of research addresses environmental impacts produced by NBS using life-cycle assessments (LCA), tracking the sources and origins of elements involved in the NBS creation, maintenance and dismantling (e.g., Giama et al., 2021). While being promising in providing a novel understanding on NBS impacts at larger scales, LCA has not yet been linked to, nor combined with NBS vulnerability assessments at local scale.

For a combined (local-global) NBS vulnerability assessment, I suggest understanding larger scale vulnerabilities in the context of Planetary Boundaries (PB). The latter is defined as the “safe operating limits within which humanity can operate to maintain a stable and resilient global environment” (Rockström et al., 2009). I argue that the exceedance of a PB can be interpreted as a vulnerability, as it increases the risk of both social and ecological systems to safely develop. There are nine specific boundaries, including, for example, 350 ppm of atmospheric carbon dioxide for Climate change and 276 dobson units of ozone concentration for Stratospheric ozone depletion. Out of the nine PBs, six are considered to be currently exceeded (Richardson et al., 2023). PBs have been previously used, for instance, for assessing how the environmental impacts of EU production and consumption are transgressing Earth's ecological limits and carrying capacity (e.g. Sala et al., 2020). In the case of NBS, global sustainable forest management and conservation efforts have been assessed based on their impacts on PBs (Zhang et al., 2021). Here, the application of PB will be expanded to evaluate how urban NBS affect different vulnerabilities at large scales.

However, the integration of various vulnerabilities across different scales introduces significant complexity, which may hinder its incorporation into policymaking and planning processes. Multi-criteria decision analysis (MCDA) has been utilized to manage this complexity in planning and policymaking for NBS in a structured manner (cf. Langemeyer et al. 2016; Saarikoski et al. 2016). Integrated vulnerability assessments building on MCDA approaches have been shown to provide useful insights for decision makers by breaking down intricate decision problems (e.g. Abdullah et al., 2021; Chisholm et al., 2022). It promises to allow for a simultaneous understanding of NBS synergies and trade-offs across spatial scales – a recommended premise for NBS evaluations (European Commission, 2021). Moreover, by translating stakeholder preferences into weights, MCDA provides a systematic approach to incorporate subjective decision-making factors in a structured manner (Thokala & Madhavan, 2018), a valuable aspect to consider as stakeholders' varying perceptions can lead to different interpretations of NBS impacts (Santoro et al., 2019).

The objective of this research is to enhance the assessment of urban NBS by developing an MCDA framework capable of considering and integrating NBS impacts both within and beyond urban limits through the lens of vulnerability. My aim is that the framework is able to answer the following questions: *how various impacts linked to the implementation of NBS in urban environments can be calculated and integrated by employing a vulnerability assessment approach?* and *Which are the synergies and tradeoffs arising from the cross-scale impacts of urban NBS?* For doing so, NBS impacts arising from diverse spatially explicit land-use scenarios will be linked to multi-dimensional and multi-scalar vulnerabilities, to understand the cross-scale tradeoffs and synergies arising from their implementation. To demonstrate the effectiveness of this approach, I apply it to the case study of green roofs in the municipality of Oslo to assess how different NBS configurations affect both local and broad-scale vulnerabilities, and how these impacts relate to each other.

IV.2. Nature-based solutions vulnerability framework

IV.2.1. Conceptual considerations

I consider that both local and broad-scale vulnerabilities can be influenced by the implementation of NBS. Local-scale vulnerabilities refer to those affected by NBS within the specific (urban) area in which they are implemented, such as heat and air pollution. Broad-scale vulnerabilities are influenced by NBS beyond the confines of the urban area where they are implemented (see **Fig. 1**), such as climate change and stratospheric ozone depletion.

When evaluating the impacts of NBS on local-scale vulnerabilities, I start from the premise that the location and design of NBS can lead to the alteration of urban socio-ecological vulnerabilities in both intended and unintended ways (Pereira et al., 2023). Therefore, I propose a thorough assessment of these impacts from a spatially explicit perspective, aiming to comprehend two aspects: first, the NBS influence on urban exposures, these being the proximity of systems to hazards (e.g., temperatures during heatwaves, lack of green spaces, floodable areas); and second, the way exposure overlap with sensitive urban areas, these being the extent to which a system is impacted by hazards. For example, areas with a high presence of elderly and children, and low-income households may be more sensitive to environmental hazards, such as heatwaves.

We consider that reaching or crossing the global PBs presented by (Rockström et al., 2009) represents a broad-scale vulnerability, given the risk implications it poses for the safe development of both social and ecological systems. I argue that systems and populations are not solely affected by the nearby presence of NBS, and vulnerabilities can extend beyond local boundaries through cascading effects (Little, 2010). Furthermore, and as reflected by the urban land teleconnections approach (Seto et al., 2012), the impacts arising from urban dynamics cannot be solved in a single geographical location, as they not only shift the conditions of urban environments where they are implemented but also their surrounding areas and even distant locations. An example of this is the water demand by urban vegetation, an important trade-off in water-limited environments facing future droughts resulting from climate change (Segura et al., forthcoming). Since the broad-scale vulnerabilities apply at larger scales, they are not represented in a geographically explicit way as are the local-scale vulnerabilities. Likewise, broad-scale vulnerabilities do not consider the sensitivity dimension of vulnerability since it is assumed that global impacts such as global warming and ozone depletion affect global populations and ecosystems as a whole. While I do acknowledge that broad-scale vulnerabilities are not equally distributed (Füssel, 2010), determining their spatial distribution is beyond the scope of this study.

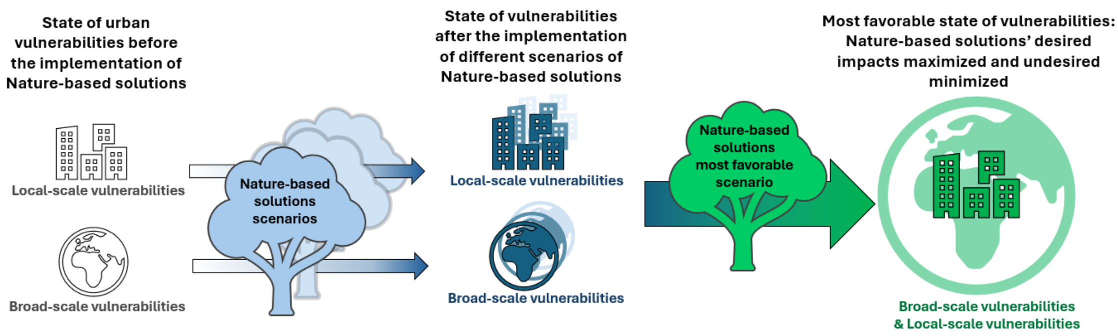


Figure 1. Graphical representation of the nature-based solutions (NBS) vulnerability framework. Arrows represent NBS impacts on vulnerabilities. The NBS-vulnerability framework proposes to consider the NBS' impacts on both local and broad-scale vulnerabilities

IV.2.2. Stepwise approach

The proposed framework for evaluating NBS follows a structured stepwise approach combining the assessment of local and broad-scale vulnerabilities, applying concepts of MCDA such as stakeholder weighting and converging discordant information, while including stakeholders' input (see **Fig. 2**). The following paragraphs briefly outline the steps. For a more comprehensive explanation of the steps, please refer to **Appendix 2-A** in the supporting information.

The first step consists of the creation of scenarios portraying potential urban NBS configurations, which will later be compared to a reference scenario to determine shifts in vulnerabilities. Second, relevant vulnerabilities potentially affected by NBS are selected. Local-scale vulnerabilities are chosen based on urban agendas and calculated by spatially explicit exposure and sensitivity indicators, while broad-scale vulnerabilities are selected based on applicable PB and calculated based on non-spatially indicators of exposure (for the full list of PBs, please see **Appendix 2-A**). The outcome of this stage is a map of each NBS scenario, reflecting local-scale vulnerabilities, along with an impact value assigned to each broad-scale vulnerability.

Third, the vulnerability values are normalized for aggregation. Local-scale indicators are normalized using contextualized threshold values to determine the magnitude of the NBS impacts, converting values to a uniform scale of 0-1. Broad-scale indicators, on the other hand, are normalized by adopting the boundary values set for each PB. This enables the evaluation of the percentage of contribution or reduction that NBS have within each PB. For example, the quantity of phosphate flows from fertilizers used in urban agriculture can be quantified and compared against the PB value for Biochemical flows to ascertain the percentage contribution from NBS. By the end of this stage, local-scale indicator values are transformed to a uniform scale ranging from 0 to 1, while broad-scale indicators are expressed as a percentage value.

Fourth, aggregation of local-scale vulnerabilities and broad-scale vulnerabilities is carried out, resulting in a single map for each single local-scale vulnerability, and a single percentage value for each broad-scale vulnerability, for every scenario.

The fifth step is stakeholder weighting, where relevant actors are presented with the calculated impacts of NBS on vulnerabilities and asked to engage in the discussion and weighting of all single vulnerabilities (i.e., local and broad-scale vulnerabilities). The objective of this dynamic exercise is to quantify the relevance of each of the vulnerabilities when considering the possible tradeoffs and synergies happening within and across spatial scales. The result of this process is a set of weights assigned to all vulnerabilities.

The final and sixth step involves employing the stakeholder weights for developing a most favorable scenario for the implementation of NBS, based on the premise that the amount of desired impacts on vulnerabilities is maximized, while the undesired impacts are minimized. The outcome of this step is the optimal scenario for the implementation of NBS.

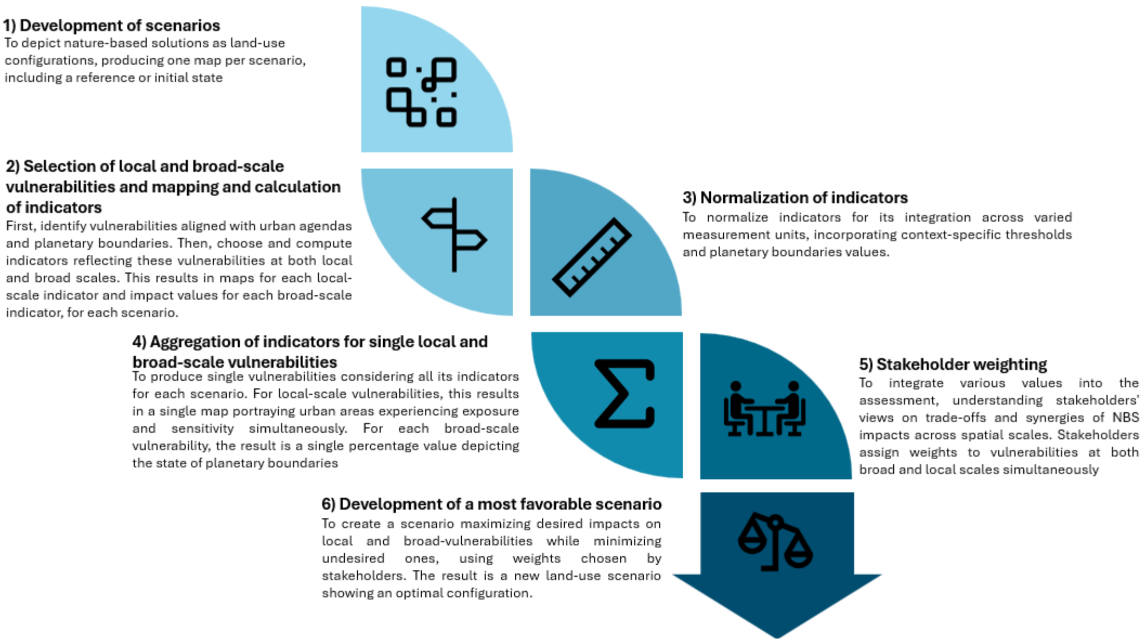


Figure 2. Stepwise approach of the Nature-based solutions vulnerability framework, along with steps descriptions and outcomes

IV.3. Case study: green roofs in Oslo municipality

Green roofs, as NBS, offer ecosystem services within urban environments, aiding in vulnerability mitigation. These services include thermal regulation, runoff mitigation, and provision of habitats for pollinators (e.g., Gilabert et al., 2021; Johannessen et al., 2017;

Langemeyer et al., 2020; Venter et al., 2021). Despite some studies linking the provision of ecosystem services in urban areas to their broader environmental impacts on a global scale (e.g., Gargari et al., 2016), there's still a need for a more detailed assessment. Specifically, the spatially explicit impacts of different green roof configurations in urban environments, especially regarding off-site (non-urban) impacts, remain understudied.

Green roofs have gained popularity due to their capacity to be created in underutilized spaces while also avoiding the acquisition of new land or changes to existing plots (Vijayaraghavan, 2016), gaining relevance within the urban policy arena (Liberalesso et al., 2020). Oslo, the capital of Norway, with a population of 709,037 residents (Strand, 2023), is actively promoting green roof creation. The city has garnered attention as a green capital (Oslo Kommune, 2020) for efforts to reduce its carbon footprint, air pollution, and improve urban mobility (Oslo Kommune, 2021a, 2021c). Currently, Oslo holds the highest green space coverage among major European capitals (Oslo Kommune, 2020), though this is gradually diminishing due to population growth and urban densification (Kruse et al., 2022; Oslo Kommune, 2018). The city has set a target to establish 2030 green roofs and facades by 2030, as outlined in its municipal strategy (Oslo Kommune, 2022). Therefore, it was chosen to evaluate green roofs in the Oslo municipality as the case study for applying the NBS-vulnerability framework.

Table 1. Vulnerabilities, indicators, average/sum of absolute exposure and sensitivity indicator values before normalization, thresholds and weights from the assessment of green roofs in Oslo

Local/Broad scale vulnerability	**Vulnerability**	**Indicator**	**Unit**	**Exposure/ Sensitivity**	**Average/sum of absolute indicator values before normalization**				**Average/ sum**	**Threshold value for normalization**	**Weights for single vulnerability aggregation**	**Stakeholder weights for combined vulnerability**
					S0.	**S1.**	**S2.**	**S3.**				
Local-scale	Vulnerability to lack of habitats for pollinators	Pollinator habitat suitability	Index (0-1)	Exposure	0.31	0.31	0.31	0.34	Average	Increased exposure if proximity to high-traffic roads ≤ 200m	1	18%
		Precautionary zones for honeybee keeping	Km2	Sensitivity	62.35	N/A	N/A	N/A	Sum	N/A	0.5	
		Areas with presence of red listed bee species	Km2	Sensitivity	5.6	N/A	N/A	N/A	Sum	N/A	0.5	
	Vulnerability to heavy rainfall events	Runoff coefficient	Liters/second	Exposure	44.8	44.7	44.5	40.9	Average	No exposure ≤ 3.5 l/s per ha	1	24%
		Areas with presence of critical infrastructure	Km2	Sensitivity	56.9	N/A	N/A	N/A	Sum	N/A	0.25	
		Population density	Hab./Km2	Sensitivity	4,659	N/A	N/A	N/A	Average	N/A	0.25	
		Elderly population density (75yo<)	Hab./Km2	Sensitivity	532	N/A	N/A	N/A	Average	N/A	0.25	
		Low-income households	% of households	Sensitivity	18	N/A	N/A	N/A	Average	N/A	0.25	
	Vulnerability to heat	Outdoor heatwave day temperatures	◦C	Exposure	28.5	28.5	28.5	28.5	Average	No exposure ≤ 30°C	0.25	13%
		Outdoor heatwave night temperatures	◦C	Exposure	21.0	21.0	21.0	21.0	Average	No exposure ≤ 20 °C	0.25	
		Indoor heatwave day temperatures	◦C	Exposure	28.5	28.3	28.3	27.7	Average	No exposure ≤ 26°C	0.5	
		Population density	Hab./Km2	Sensitivity	4,659	N/A	N/A	N/A	Average	N/A	0.33	
		Elderly population density (75yo<)	Hab./Km2	Sensitivity	532	N/A	N/A	N/A	Average	N/A	0.33	
		Low-income households	% of households	Sensitivity	18	N/A	N/A	N/A	Average	N/A	0.33	
	Vulnerability to air pollution	Particulate matter 10 (PM_{10})	Ton/year	Exposure	439.7	439.5	439.47	435.87	Sum	No exposure ≤ 402.19 Ton/year for the whole region	0.5	7%
		Population density	Hab./Km2	Sensitivity	4,658.60	N/A	N/A	N/A	Average	N/A	0.33	
		Children population density	Hab./Km2	Sensitivity	778.76	N/A	N/A	N/A	Average	N/A	0.33	
		Low-income households	% of households	Sensitivity	18.01	N/A	N/A	N/A	Average	N/A	0.33	
	Vulnerability to lack of opportunities for interacting with natural environments	Share of green areas	% of area	Exposure	44.8	45.0	45.1	50.7	Average	No exposure ≤ 30% of area	0.5	18%
		Green Gini coefficient	Index (0-1)	Exposure	12.3	12.2	12.1	9.2	Average	No threshold value	0.5	
		Population density	Hab./Km2	Sensitivity	4,659	N/A	N/A	N/A	Average	N/A	0.33	
		Children population density	Hab./Km2	Sensitivity	779	N/A	N/A	N/A	Average	N/A	0.33	
		Low-income households	% of households	Sensitivity	18	N/A	N/A	N/A	Average	N/A	0.33	

Table 1. Vulnerabilities, indicators, average/sum of absolute indicator values before normalization, thresholds and weights from the assessment of green roofs in Oslo (continuation)

Local/Broad scale vulnerability	**Vulnerability**	**Indicator**	**Unit**	**Exposure/Sensitivity**	**Average/sum of absolute indicator values before normalization**				**Average/ sum**	**Threshold value for normalization**	**Weights for single vulnerability aggregation**	**Stakeholder weights for combined vulnerability**
					S0.	**S1.**	**S2.**	**S3.**				
Broad-scale	Vulnerability to climate change	Net emissions of greenhouse gases	kg CO2 eq.	Exposure	1.11E+04	2.44E+04	4.27E+04	6.09E+05	Sum	Planetary boundary allocation ≤ 6.98E+08	1	10%
	Vulnerability to changes in biogeochemical flows	Acidification potential	mol H+ eq	Exposure	5.32E+02	1.16E+03	2.04E+03	2.91E+04	Sum	Planetary boundary allocation ≤ 1.03E+08	0.25	5%
		Eutrophication potential (marine)	kg N eq	Exposure	1.25E+02	2.74E+02	4.81E+02	6.85E+03	Sum	Planetary boundary allocation ≤ 2.06E+07	0.25	
		Eutrophication potential (freshwater)	kg P eq	Exposure	1.75E+01	3.83E+01	6.73E+01	9.59E+02	Sum	Planetary boundary allocation ≤ 5.96E+05	0.25	
		Eutrophication potential (terrestrial)	mol N eq	Exposure	1.88E+03	4.11E+03	7.22E+03	1.03E+05	Sum	Planetary boundary allocation ≤ 6.29E+08	0.25	
	Vulnerability to novel entities	Human toxicity potential (cancer)	CTUh	Exposure	7.01E-05	1.53E-04	2.69E-04	3.83E-03	Sum	Planetary boundary allocation ≤ 9.86E+01	0.20	5%
		Human toxicity potential (non-cancer)	CTUh	Exposure	1.87E-03	4.10E-03	7.19E-03	1.02E-01	Sum	Planetary boundary allocation ≤ 4.20E+02	0.20	
		Photochemical ozone formation	kg NMVOC eq	Exposure	4.08E+02	8.93E+02	1.57E+03	2.23E+04	Sum	Planetary boundary allocation ≤ 4.17E+07	0.20	
		Potential ecotoxicity (freshwater)	CTUe	Exposure	3.68E+04	8.05E+04	1.41E+05	2.01E+06	Sum	Planetary boundary allocation ≤ 1.35E+10	0.20	
		Ionizing radiation	kBq U-235 eq	Exposure	8.92E+03	1.95E+04	3.42E+04	4.88E+05	Sum	Planetary boundary allocation ≤ 5.4E+10	0.20	
	Vulnerability to stratospheric ozone depletion	Ozone depleting substances	kg CFC11 eq	Exposure	7.31E-03	1.60E-02	2.81E-02	4.00E-01	Sum	Planetary boundary allocation ≤ 5.53E+04	1	2%

IV.4. Methodology

In the following sections it will be described how each of the steps of the NBS-vulnerability framework has been applied to the case study of green roofs in the Oslo Municipality. For more detailed information on each step, please consult **Appendix 2-A.**

IV.4.1. Development of scenarios

Four scenarios are proposed to understand the possible growth trends of green roofs in the city of Oslo, which were co-created with stakeholders with experience in managing this and other types of NBS (URBAG, 2021). Scenarios portray the single land-use change from rooftops without green roofs to rooftops with green roofs. The scenarios are: Current (S0) which serves as the reference state and is based on an aerial photo-survey conducted by the Oslo municipality in 2017 (Oslo Kommune, 2021b), identifying 928 green roofs covering 18 hectares; Green roof strategy (S1), which aligns with the objectives outlined in the municipal strategy for the increase of green roofs and facades by 2030 (Oslo Kommune, 2022), projecting 2030 green roofs and covering 41 hectares; Ambitious (S2), representing an optimistic implementation of green roofs in the municipality, larger in scale than scenario S1, with 3,550 green roofs covering 72 hectares; and Maximization (S3), representing the creation of green roofs in all available rooftops of the city with an area bigger than 10m2 and a slope below 30°, resulting in 56,786 green roofs covering 1,039 hectares. **Figure 3** offers a scenario overview.

Location and size of new green roofs for S1, S2 were chosen based on the premise of maintaining the spatial distribution and average size of green roofs found in S0.

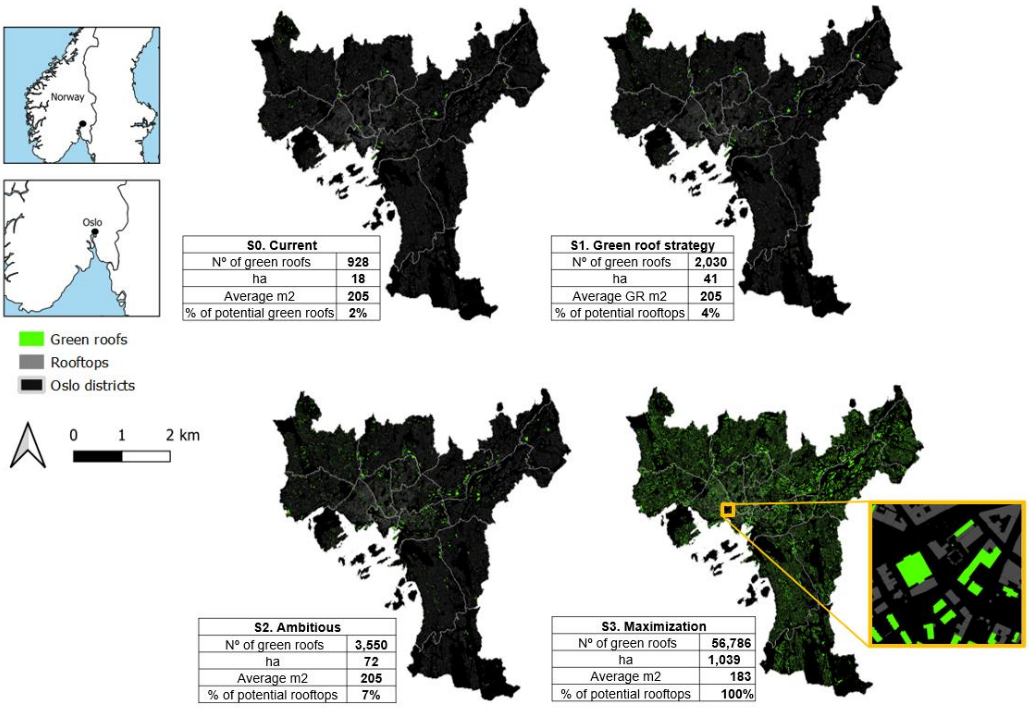


Figure 3. Proposed scenarios for estimating green roofs impacts on local and broad-scale vulnerabilities, depicting the number of green roofs, their total extension, average size and percentage occupation out of the total potential green roofs

IV.4.2. Selection of local-scale vulnerabilities and mapping of indicators

Five local-scale vulnerabilities were chosen based on Oslo's future objectives and policies for improving the city's livability and resilience (Oslo Kommune, 2012, 2021c, 2023a), where the adoption of green roofs can offer assistance, including: *vulnerability to lack of habitats for pollinators*, *vulnerability to heavy rainfall events*, *vulnerability to heat*, *vulnerability to air pollution* and *vulnerability to lack of opportunities for interacting with natural environments*. Each one is described by at least one exposure and sensitivity indicator (see **Table 1**), which were selected through a literature review and collaborative discussions within the interdisciplinary team involved in the assessment. In certain cases, identical sensitivity indicators (e.g., population density) were applied to various vulnerabilities because it is the most appropriate way of portraying urban susceptibility. There is no risk of redundancy in these cases, as exposure values consistently vary, and the combination of exposure and sensitivity offer varied vulnerability maps.

Finally, to facilitate integration across different resolutions, I transformed indicators into a 50x50m grid. Each of these grid cells will be referred to as pixels from this point forward.

IV.4.3. Selection of broad-scale vulnerabilities and calculation of indicators

Each broad-scale vulnerability represents a single PB that could be potentially affected by the implementation of green roofs in desired or undesirable ways. For selecting these vulnerabilities, first, a life-cycle assessment (LCA) was carried out addressing the environmental impacts associated with the construction, installation, use and disposal of 1m2 of an extensive green roof over a year. Each impact category in the LCA was then linked to a PB based on established connections found in prior research literature (e.g., Ryberg et al., 2018; Sala et al., 2020; Sandin et al., 2015) (see **Table 1**). After doing so, four broad-scale vulnerabilities were identified: *vulnerability to climate change, vulnerability to changes in biogeochemical flows, vulnerability to novel entities* and *vulnerability to stratospheric ozone depletion.*

IV.4.4. Normalization of local-scale indicators

For creating a unified scale across the local-scale indicators, the exposure and sensitivity values were scaled to 0-1 using min-max normalization where 0 indicates no exposure/sensitivity and 1 indicates the highest exposure/sensitivity. For exposure, normalization was conducted by employing threshold values that represent levels of low or high exposure. For instance, for the runoff coefficient, the no-exposure threshold is 3.5 l/s per ha (Oslo Kommune, 2023a) – below this runoff level, the exposure to heavy rainfall events is considered irrelevant, translating into a lack of vulnerability. Thresholds can be found in **Table 1**.

IV.4.5. Normalization of broad-scale indicators

To normalize the broad-scale indicators, first, PB values were downscaled to match the geographical scale of Oslo based on its population. For doing this, it was employed the equality allocation method, which assumes equal rights to ecological space for all individuals (Häyhä et al., 2016), thereby allocating downscaled PB values to Oslo, allowing for a more nuanced assessment of green roofs impacts aligned with their implementation scale. This process was carried out using the methodology and reference values outlined by Sala et al. (2020) and Sanye Mengual & Sala (2023), where PBs were directly linked to LCA impact categories and downscaled to per capita reference values based on the global population. These per capita values were then multiplied by Oslo's population (Oslo Kommune, 2023a) to derive scaled values adapted to the case study, resulting in single threshold values for each of the broad-scale indicators (i.e., LCA impact categories; see **Table 1**). Subsequently, thresholds were employed to normalize each of the broad-scale indicators. This was done by calculating the percentage utilization that each indicator had within each of the downscaled PB (i.e., out of the total downscaled PB adapted to Oslo, green roofs impacts occupy a certain share of the PB).

IV.4.6. Aggregation of indicators for single local-scale vulnerabilities

Normalized local-scale indicators were first combined into single exposures and sensitivities for each vulnerability, using equal weights (see **Table 1**). These were then aggregated to derive single local-scale vulnerabilities that summarize both exposures and sensitivities. I calculated the sum of pixel values and their relative changes between scenarios for each vulnerability, helping depict each vulnerability's behavior across scenarios.

IV.4.7. Aggregation of indicators for single broad-scale vulnerabilities

Normalized broad-scale indicators were aggregated employing equal weights (see **Table 1**) for obtaining an overall impact of each scenario in each of the broad-scale vulnerabilities. This value is expressed in a percentage, representing the share of green roofs impacts within the total downscaled PB for Oslo.

IV.4.8. Stakeholder weighting

An online workshop was held on January 25th, 2024, where participants were invited to jointly consider the vulnerability shifts associated with the green roof's impacts, seeking out potential tradeoffs and synergies between local-scale and broad-scale vulnerabilities. There, stakeholders simultaneously weighted the relevance of each of the single vulnerabilities when assessing the impacts of implementing green roofs in Oslo. The weights obtained from this dynamic are shown in **Table 1**. During this workshop, participants were also invited to discuss the utility of the NBS-vulnerability framework for supporting the planning and policymaking around NBS in Oslo. Discussion points of this chapter are partly informed by this workshop dynamic.

IV.4.9. Development of a most favorable scenario

Finally, employing the weights defined in the stakeholder workshop, a most favorable scenario for the implementation of green roofs in Oslo was developed. The aim was to determine the quantity of green roofs and their spatial allocation to maximize desired impacts on vulnerabilities while minimizing undesired impacts. Additionally, an alternative most favorable scenario was developed employing equal weights instead of stakeholder weights for assessing its sensitivity to weighting schemes.

IV.5. Results

First, I present the impacts of each scenario on local-scale vulnerabilities, describing their shifts and spatial dynamics, and then on the broad-scale vulnerabilities, describing the percentage of contribution or reduction that green roofs have within each PB. Next, the results for the most favorable scenario for green roof implementation are shown based on the two weighting schemes.

Table 2. Single vulnerability values for each scenario and percentage change (compared to scenario S0). For local-scale vulnerabilities, calculation is based on the sum of pixel values, where higher values represent a greater vulnerability. For broad-scale vulnerabilities, calculation is based on the share of green roofs impacts within the total downscaled PB for Oslo city, where 0 implies that impacts occupy none of the PB and 1 that impacts take all the PB.

	Vulnerability	**Values expressed in**	**S0**	**S1**	**S2**	**S3**	**S4 Stakeholder weights**	**S4 Equal weights**
Local-scale	Vulnerability to lack of habitats for pollinators	Sum of pixel values	9,778	9,753	9,687	9,329	9,383	9,654
		Difference vs. S0 (%)		-0.3%	-0.9%	-4.6%	-4.0%	-1.3%
	Vulnerability to heavy rainfall events	Sum of pixel values	4,247	4,239	4,229	3,898	3,946	4,175
		Difference vs. S0 (%)		-0.2%	-0.4%	-8.2%	-7.1%	-1.7%
	Vulnerability to heat	Sum of pixel values	4,035	3,913	3,820	3,332	3,438	3,742
		Difference vs. S0 (%)		-3.0%	-5.3%	-17.4%	-14.8%	-7.3%
	Vulnerability to air pollution	Sum of pixel values	1,021	1,021	1,021	1,008	1,009	1,012
		Difference vs. S0 (%)		0.0%	-0.1%	-1.3%	-1.2%	-0.9%
	Vulnerability to lack of opportunities for interacting with natural environments	Sum of pixel values	532	517	501	183	186	219
		Difference vs. S0 (%)		-2.8%	-5.8%	-65.6%	-65.1%	-58.8%
Broad-scale	Vulnerability to climate change	Share within planetary boundary	1.6E-05	3.5E-05	6.1E-05	8.7E-04	6.0E-04	1.4E-04
		Difference vs. S0 (%)		120%	300%	5,300%	3,700%	800%
	Vulnerability to changes in biogeochemical flows	Share within planetary boundary	1.1E-05	2.4E-05	4.2E-05	6.0E-04	4.1E-04	9.2E-05
		Difference vs. S0 (%)		120%	300%	5,300%	3,700%	800%
	Vulnerability to novel entities	Share within planetary boundary	3.6E-06	7.9E-06	1.4E-05	2.0E-04	1.3E-04	3.0E-05
		Difference vs. S0 (%)		120%	300%	5,300%	3,700%	800%
	Vulnerability to stratospheric ozone depletion	Share within planetary boundary	1.3E-07	2.9E-07	5.0E-07	7.2E-06	4.9E-06	1.1E-06
		Difference vs. S0 (%)		120%	300%	5,300%	3,700%	800%

IV.5.1. Vulnerability to lack of habitats for pollinators

Vulnerability to lack of habitats for pollinators emerges as the most pressing for Oslo, based on the premises of the study (i.e., sum of pixel values; see **Table 2**). It is spatially concentrated across the whole bay area, and in single patches in the center, north and south of Oslo (see **Fig. 4a**). All of these areas are considered sensitive due to the presence of red-listed species and their designation as precautionary zones for beekeeping. This coincides with widespread high exposure due to the high presence of built-up areas (i.e., lack of pollinator habitats), resulting in pixel values reaching up to 1 on a 0-1 scale.

This vulnerability consistently decreases with green roofs expansion, with reductions of 0.3%, 0.9%, and 4.6% in S1, S2, and S3, respectively (see **Table 2**), attributed to reduced exposure due to the increased pollinator habitats provided by green roofs (see **Table 1**).

Spatially speaking, these reductions align with increased green roof presence in sensitive areas like the bay area, center, and north of Oslo for all scenarios (See **Fig. 4b, c, d**). In this sense, S1 and S2 miss the opportunity to address this vulnerability more effectively, as new green roofs are mostly located in the northwest of the city, areas that exhibit very low or nonexistent sensitivity values (see **Appendix 2-C**, sections 1.4 – 1.6).

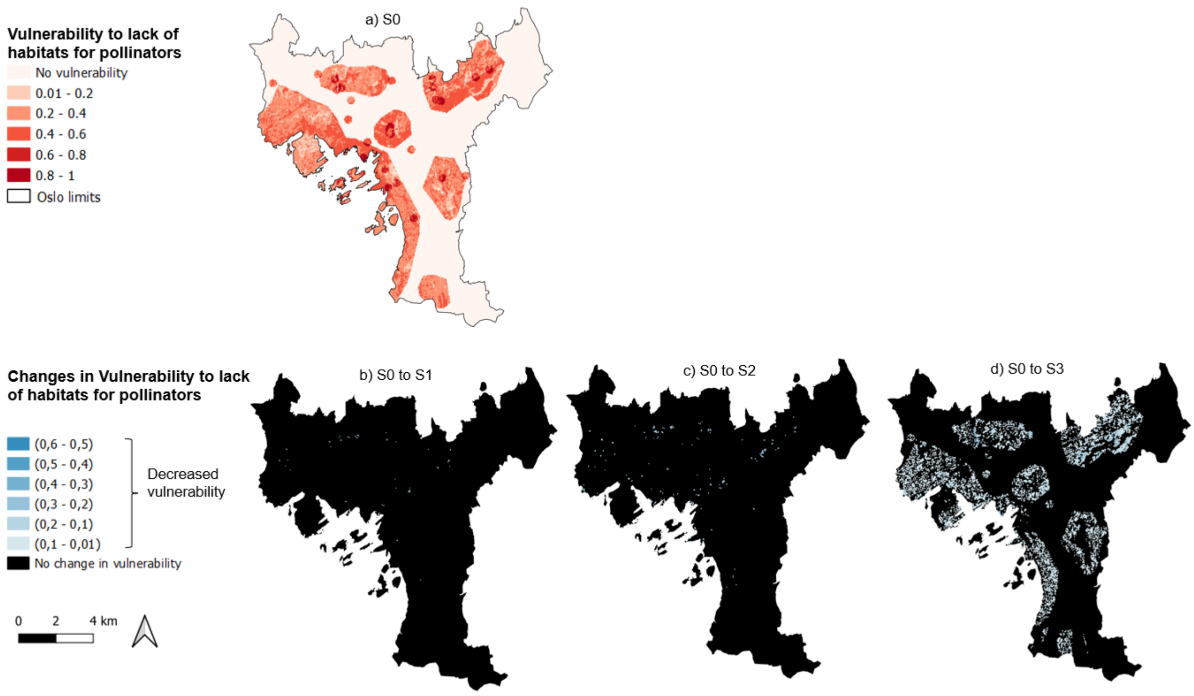


Figure 4. Spatial distribution of vulnerability to lack of habitats for pollinators and changes across scenarios. Black areas represent no vulnerability changes between scenarios

IV.5.2. Vulnerability to heavy rainfall events

Vulnerability to heavy rainfall events is found to be the second most significant vulnerability for Oslo (see **Table 2**). From a spatial perspective, it is mostly concentrated at the city center (see **Fig. 5a**), where vulnerability values arise up to 1 on a 0-1 scale, explained by high runoff coefficients (i.e., exposure) and high population densities, low-income households, and critical infrastructures (i.e., sensitivities) (see **Appendix 2-C**, section 2.4 – 2.8).

Across the scenarios, *vulnerability to heavy rainfall events* experienced reductions when compared to S0 (see **Table 2**). In S3, where the highest green roof presence is observed, this vulnerability diminishes by 8.2%, while S1 and S2 reduce it by 0.2% and 0.4% respectively, coinciding with similar reductions in runoff coefficients across the scenarios (see **Table 1**).

From a spatial perspective, S1 and S2 present reductions only in very specific areas located mostly in the urban center (see **Fig. 5b, c**). S3, on the other hand, provides widespread reductions that extend across the entire municipality (see **Fig. 5d**), following sensitive areas with high population densities (center of the city) and critical infrastructures (e.g., as main roads) (see **Appendix 2-C –** 2.4, 2.5, 2.6).

It is important to mention that, despite runoff reductions often occurring in sensitive areas, they do not always lead to more pronounced reductions in vulnerabilities. This is because, in many cases, these reductions are minimal compared to the volume of runoff experienced, which translates in very small changes in the exposure to heavy rainfall events.

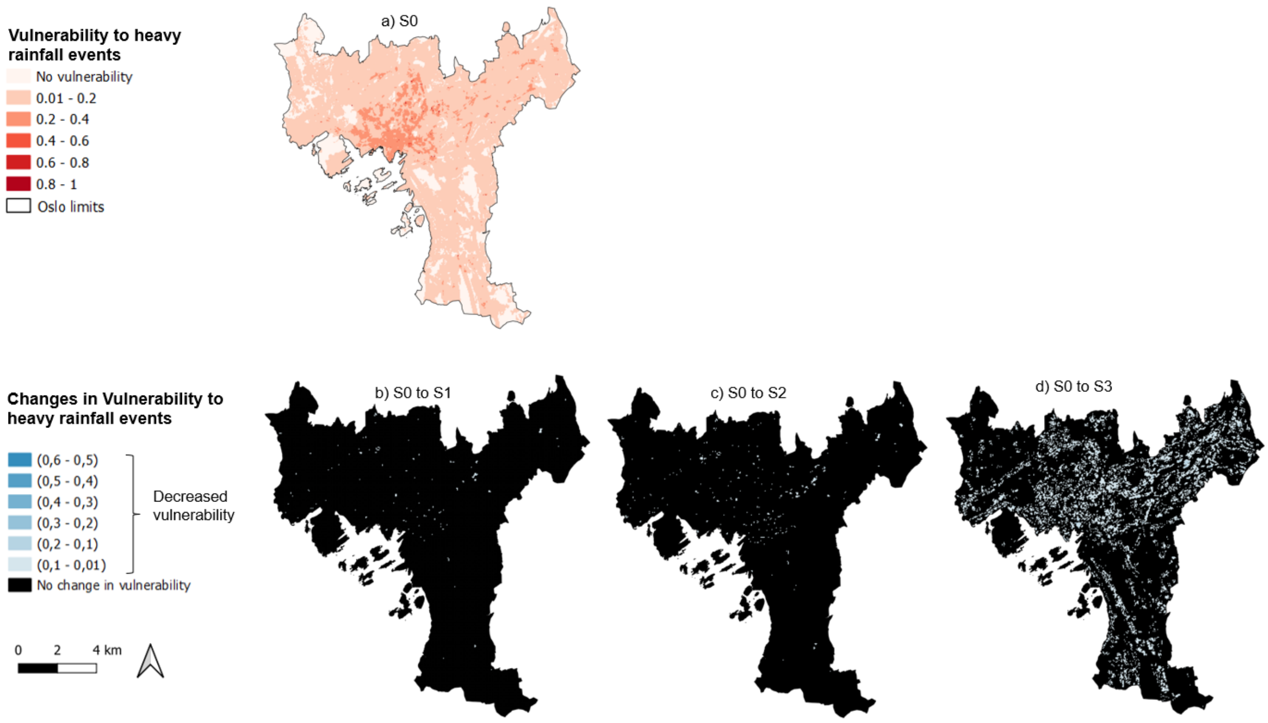


Figure 5. Spatial distribution of vulnerability to heavy rainfall events and changes across scenarios. Black areas represent no vulnerability changes between scenarios

IV.5.3. Vulnerability to heat

Vulnerability to heat is the third most significant vulnerability experienced in Oslo (see **Table 2**). It is mostly concentrated in the urban center where the highest outdoor night temperatures and indoor day temperatures are experienced (i.e., exposure), along with high population densities and low-income households. This results in pixel values of up to 0.68 on a 0-1 scale (see **Fig. 6a**).

This vulnerability consistently decreases with scenarios of expanding the green roof surface across the city. This is observed by the reductions of 3%, 5.3%, and 17.4% for S1, S2, and S3, respectively, compared to S0 (see **Table 2**). These changes are attributed solely to the capacity of green roofs to reduce inner temperatures, as they did not reduce either night or day outdoor temperatures (see **Table 1**). Moreover, day outdoor temperatures did not play a role in defining the vulnerability, nor its shifts, since it was always found to be below the exposure threshold.

Vulnerability shifts are found across Oslo in all scenarios (see **Fig. 6b, c, d**), with more significant decreases occurring in the city center, where sensitivities are highest (see **Appendix 2-C**, sections 3.12 – 3.15). Furthermore, in all scenarios, smaller and more widespread reductions in vulnerability are observed (e.g., west and north-west of Oslo). This is primarily due to the substantial reductions in daytime inner temperatures that green roofs offers, which, even when coinciding with low sensitivity areas, effectively mitigate the vulnerability.

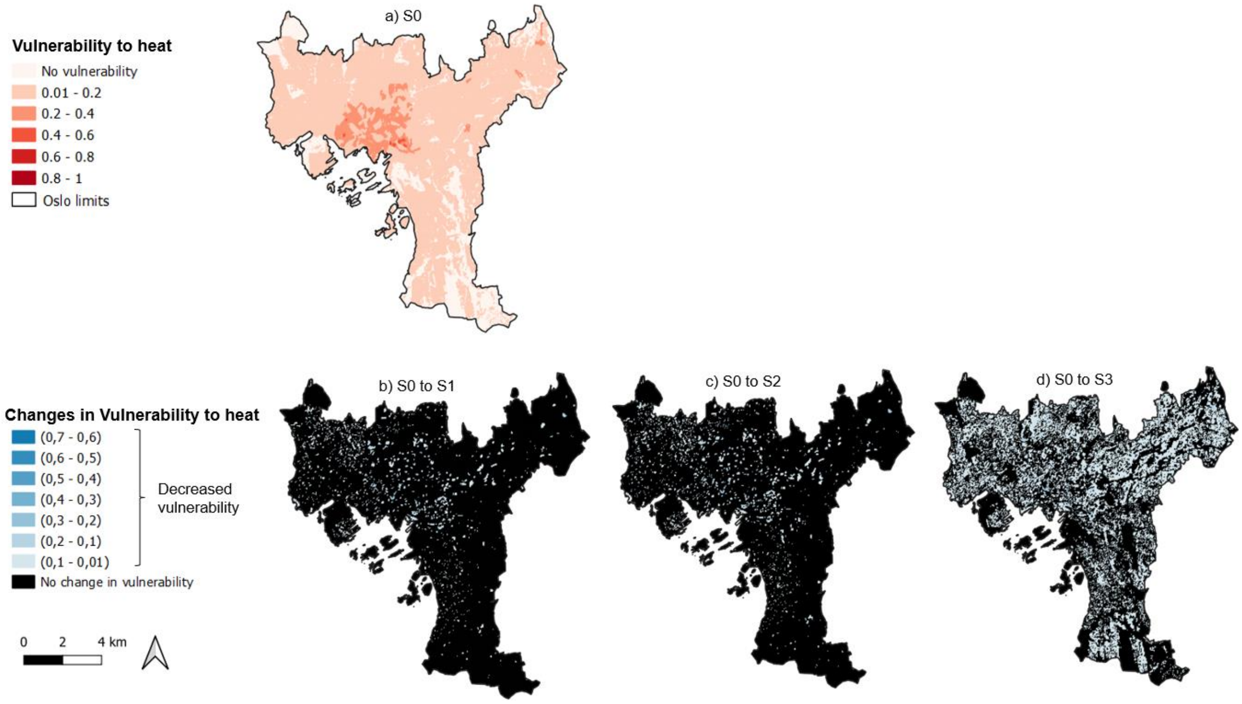


Figure 6. Spatial distribution of vulnerability to heat and changes across scenarios. Black areas represent no vulnerability changes between scenarios

IV.5.4. Vulnerability to air pollution

Vulnerability to air pollution is the fourth most prominent vulnerability in Oslo (see **Table 2**). The highest vulnerability values are found in the Oslo center, where the greatest PM10 concentrations are found, along with high population densities and low-income households. Together, these produce vulnerability values of up to 0.59 on a 0-1 scale (see **Fig. 7a**).

This vulnerability reduces by 0.1% in S2 and 1.3% in S3, while S1 does not provide any changes (see **Table 2**). This is related to the small capacity that green roofs have for capturing air pollutants, as observed in absolute changes in PM10 across the green roof scenarios (see **Table 1**). Nonetheless, S3 is able to reduce the vulnerability in a greater proportion than that found across the absolute values, as some of the PM10 reductions coincide with highly sensitive areas in the center of the city (see **Fig. 7d**).

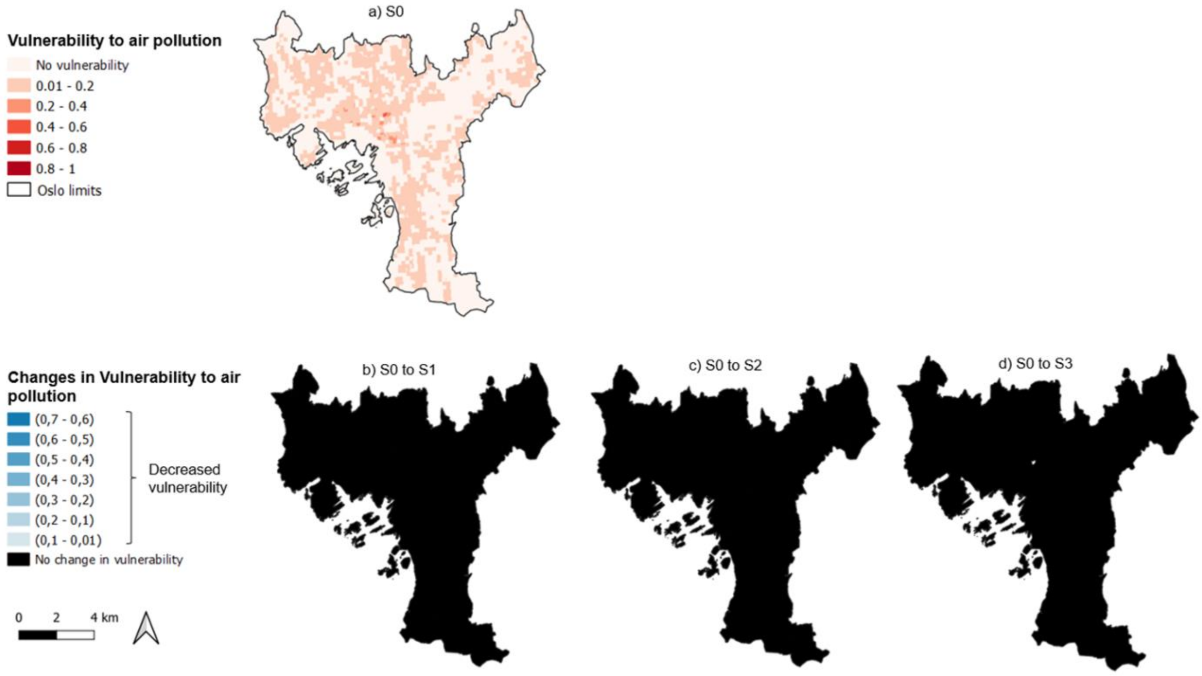


Figure 7. Spatial distribution of vulnerability to air pollution and changes across scenarios. Black areas represent no vulnerability changes between scenarios

IV.5.5. Vulnerability to lack of opportunities for interacting with natural environments

The *Vulnerability to lack of opportunities for interacting with natural environment* proved to be the least significant vulnerability at the local-scale (see **Table 2**). It is concentrated in the urban center, where exposure is highest due to the lowest share of green areas and high sensitivities resulting from dense populations and low-income households. Vulnerability values reach up to 0.4 on a scale of 0-1 (see **Fig. 8a**).

This vulnerability experiences the highest reduction across all vulnerabilities, as S1, S2 and S3 shift it by -2.8%, -5.8% and -65.1% respectively (see **Table 2**). These reductions are significantly more pronounced than those in the exposure values (see **Table 1**), which is explained by a great overlap between the reductions in exposure with highly sensitive areas.

These overlaps are concentrated in the city center (see **Appendix 2-C**, sections 5.7 and 5.12), where green roof expansions increase the amount of greenery, reducing the Gini coefficients, and, thus, providing a more equitable distribution. As a result, vulnerabilities decrease in all scenarios (see **Fig. 8b, c, d**).

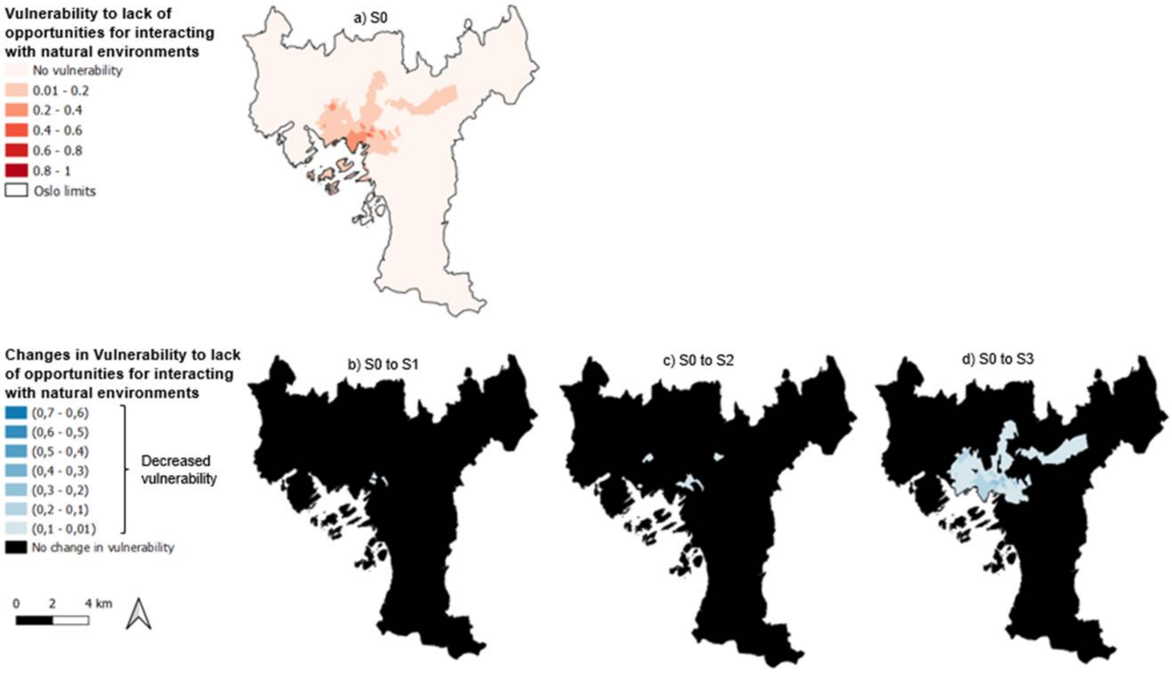


Figure 8. Spatial distribution of vulnerability to lack of opportunities for interacting with natural environments and changes across scenarios. Black areas represent no vulnerability changes between scenarios

IV.5.6. Broad-scale vulnerabilities

Vulnerability to climate change stands out as the most significant broad-scale vulnerability, concentrating the greatest overall impact under the assumptions of the study (i.e., the share of green roofs impacts within the total downscaled PB for Oslo city; see **Table 2**). Judging by S3, which maximizes green roof presence in the city, *Vulnerability to climate change* covers 8.7E-04 (i.e., 0.087%) of the total downscaled Climate Change PB value for Oslo city, even when considering the CO2 sequestration capacity of green roofs. In the case of *vulnerability to changes in biogeochemical flows*, the impact accounts for 6.0E-04 (i.e., 0.06%), while *vulnerability to novel entities* represents 2.0E-04 (i.e., 0.02%). Finally, *vulnerability to stratospheric ozone* depletion is the least pronounced among the broad-scale vulnerabilities at 7.2E-06 (i.e., 0.00072%).

Regarding the vulnerability shifts across scenarios, all vulnerabilities are increased by the green roof expansion, and exhibit identical and linear percentage changes, as observed in **Table 2**. This consistency arises from the initial calculation of impacts, which was based on 1m² of green roof and later scaled for each scenario. Compared to the baseline scenario S0, the Green Roof Strategy (S1) increases broad-scale vulnerabilities by around 120%, while the Ambitious scenario (S2) increases them by 300%, and S3 by 5,300%.

When it comes to the green roof processes and elements impacting each of the vulnerabilities, it was found that the production and disposal of root barriers, as well as the production of fertilizers, were the main drivers producing greenhouse gas emissions affecting the *Vulnerability to climate change* (see **Appendix 2-B**). In the case of *vulnerability to changes in biogeochemical flows*, the use of fertilizer was the main driver of terrestrial, marine, and freshwater eutrophication due to phosphorous and nitrous emissions. For *Vulnerability to novel entities*, impacts are mainly related to potential freshwater ecotoxicity damages associated with the production of pumice and gravel for the substrates, and by pollutants increasing human toxicity during the production of fertilizers. Lastly, impacts of *vulnerability to stratospheric ozone depletion* were mainly driven by the production of pumice and gravel for the substrates as these involved the emission of chlorofluorocarbon gases affecting the ozone layer.

IV.5.7. Most favorable scenario based on stakeholder and equal weighting schemes

Stakeholders ranked both local and broad-scale vulnerabilities from most to least relevant, in the following order: *vulnerability to heavy rainfall events, vulnerability to lack of habitats for pollinators, vulnerability to lack of opportunities for interacting with natural environments, vulnerability to heat, vulnerability to climate change, vulnerability to air pollution, vulnerability to novel entities, vulnerability to changes in biogeochemical flows* and *vulnerability to stratospheric ozone depletion*. Detailed weights are presented in **Table 1**.

Stakeholder weights and equal weights were employed for producing a new scenario: the most favorable scenario (S4) portraying the optimal area of green roof to maximize desired impacts on vulnerabilities while minimizing undesired impacts. When using stakeholder weights, the optimal area was found to be 706 hectares of green roofs, accounting for 68% of the potential green roofs that could be implemented in Oslo. Conversely, equal weights reduced the area of green roofs to 160 hectares, constituting only 15% of the potential green roofs.

Each of these alternatives has varying impacts on vulnerabilities. When employing stakeholder weights, reductions in local-scale vulnerabilities were akin to those seen with S3 which maximizes green roofs (see **Table 2**). For example, compared to S0, S3 reduced *vulnerability to heavy rainfall events* by 8.2%, while S4 reduced it by 7.1%, while *vulnerability to lack of habitats for pollinators* decreased by 4.6% with S3 and 4% with S4. In terms of broad-scale vulnerabilities, S4 exhibited significantly lower increases than S3 when compared to S0, with an increase of 3,700% for S4 and 5,300% for S3. Conversely, when applying equal weights in S4, reductions in local-scale vulnerabilities were less pronounced. Compared to S0, *vulnerability to heavy rainfall events* decreased by 1.7%, and *vulnerability to lack of habitats for* *pollinators* by 1.3%. Broad-scale vulnerabilities, on the other hand, increased less significantly, by 800% compared to S0.

Calculations for S4 under both weighting schemes are spatially portrayed in **Figure 8**, where Oslo areas are prioritized for the implementation of green roofs to simultaneously maximize desired impacts and minimize undesired impacts on vulnerabilities. Stakeholder weights allow for greater implementation of green roofs in Oslo (see **Fig. 8a**), as these give preference to the desired impacts on local-scale vulnerabilities before the undesired impacts on broad-scale vulnerabilities. High-priority areas are found in the city center, mainly in the coastal area, while low-priority areas expand across the rest of the urban space. A similar concentration is found under equal weights (see **Fig. 8b**), but much less widespread, as equal weights do not allow for such a great implementation of green roofs, as these expansions increase undesired impacts on broad-scale vulnerabilities.

Finally, the calculation of a most favorable scenario proved to be sensible to different weighting schemes, demonstrating that optimizing green roofs impacts can produce heterogeneous results depending on how impacts are prioritized.

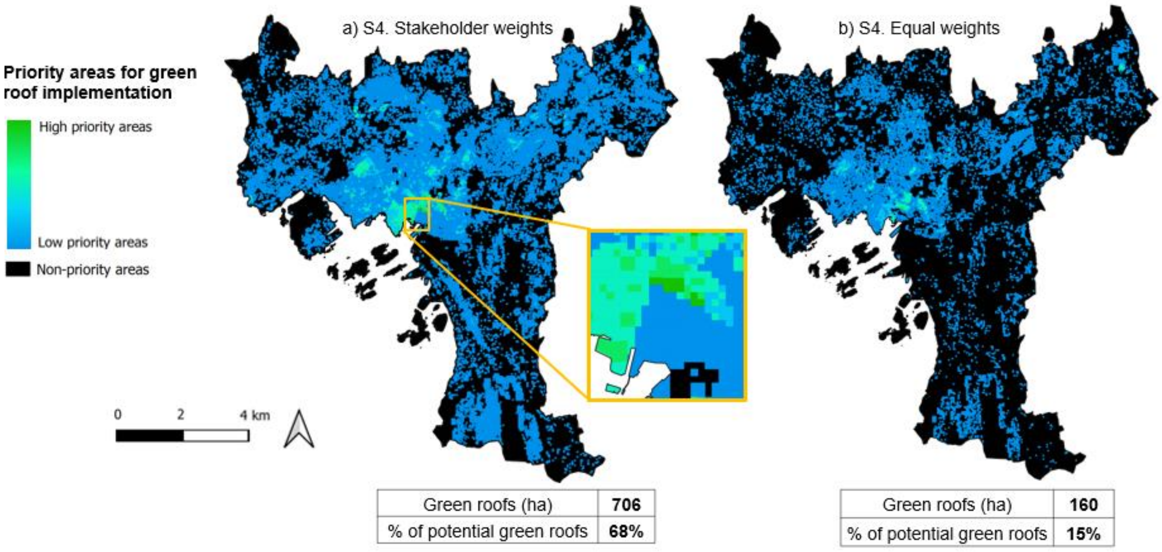


Figure 9. Priority areas for green roof implementation based on a most favorable scenario where desired impacts are maximized and undesired minimized. Two weighting schemes are considered, along with their respective green roof extensions and percentage occupation out of the total potential green roofs.

IV.6. Discussion

IV.6.1. Cross-scale understanding of green roofs impacts reveal both synergies and tradeoffs

The application of the NBS-vulnerability framework shows a wide range of NBS' impacts across different spatial scales, revealing both synergies and trade-offs among vulnerabilities.

To begin with, the assessment successfully illustrates how various green roof configurations in the Oslo Municipality impacted vulnerabilities across local and broader scales. As showcased by S4, my study highlights that when combining vulnerabilities across scales in a single MCDA assessment, the most favorable NBS scenario is not necessarily a maximization of green spaces. Under consideration of stakeholder preferences, an ideal green roof scenario, as well as spatial prioritization of urban areas where to implement green roofs, could be established.

Moreover, and at the local level, the expansion of green roofs managed to mitigate all vulnerabilities, albeit at varying degrees: Vulnerabilities related to limited opportunities for engaging with natural environments and to heat experienced the most significant reductions, while *vulnerability to air pollution* saw the smallest decrease. Consistent with previous findings, the impacts of NBS on spatial vulnerabilities are influenced by the specific NBS location (cf. Herreros-Cantis & McPhearson, 2021). An example of this can be observed in S2 and S3 with *vulnerability to heat* and *vulnerability to lack of opportunities for interacting with natural environments* as these were decreased in greatest proportion than the shift of its exposure indicators. In this sense, the city center portrayed a hotspot of local vulnerabilities, allowing for green roofs to tackle several of them simultaneously. This is a relevant finding for the strategic planning of NBS in general, but especially for green roofs, as rooftops often represent underutilized spaces. Consequently, implementing green roofs involves fewer changes in urban land use compared to other urban NBS (Vijayaraghavan, 2016).

Concerning broader-scale vulnerabilities, the expansion of green roofs resulted in their exacerbation across all scenarios. These results highlight how NBS impacts are able to transcend local boundaries, displaying cascading effects across different spatial scales (Little, 2010) causing unexpected and sometimes unintended negative impacts at global scales (Rödl & Arlati, 2022). This dynamic becomes relevant as it provides evidence that synergies and tradeoffs around urban NBS impacts cannot be judged solely by their immediate spatial effects, which could be described as a resulting product of urban land teleconnections (cf. Seto et al., 2012).

In the particular case of green roofs, their undesired impacts have been previously acknowledged (e.g., Dushkova & Haase, 2020; Firbank et al., 2007), including considerations of those happening beyond local environments (Shafique et al., 2019) which affect broad-scale vulnerabilities. However, it has also been noted that the environmental impacts of green roofs are smaller than those from conventional roofs (Shafique et al., 2019), which raises questions about the relative significance of undesired impacts of NBS compared to the impacts they mitigate. Regardless of these dynamics, addressing the negative impacts of NBS is crucial. For example, in the case of green roofs, recommendations include using construction byproducts, recycled materials, and natural elements like volcanic substrates to mitigate their environmental footprint (Gagliano & Cascone, 2024; Scolaro & Ghisi, 2022). All of these could aid in reducing green roofs' impacts on broad-scale vulnerabilities.

These considerations offer insights into the possible synergies and tradeoffs that influence NBS effectiveness in addressing sustainability, resilience, and equity challenges. For instance, green roofs produced undesired shifts in broad-scale vulnerabilities, associated with negative impacts on PBs, a valuable outcome when assessing green roof sustainability for being maintained over time while guaranteeing the provision of ecosystem services. Also, Oslo's resilience was enhanced by the green roof expansion (e.g., by reducing *Vulnerability to extreme rainwater events*) at the expense of increasing broad-scale vulnerabilities. This proves that cross-scale inequities can arise from NBS impacts, an aspect that has been acknowledged as a limitation of the sustainability efforts taking place in urban environments (Bozeman et al., 2022; Dsouza et al., 2023).

Results confirm that categorizing the impacts of urban NBS proves to be a complex task, and that a comprehensive understanding cannot be achieved through a sole perspective (Funtowicz et al., 1999). Based on this, the NBS-vulnerability framework can provide a way forward in better evaluating urban NBS, and therefore, aid in the policy decision-making of NBS in urban environments, as detailed in the next sections.

IV.6.2. Integrating cross-scale impacts of Nature-based solutions can support urban planning

Embracing integrative approaches for the assessment of NBS has been recommended for bolstering urban policymakers' ability to gauge their effectiveness in addressing urban challenges (European Commission, 2021). As of now, urban regeneration planning is often dominated by silo-thinking, which tends to treat social and ecological challenges as distinct and occasionally in conflict with one another (Dumitru et al., 2020). In this sense, the NBS-vulnerability framework can provide novel evaluative space due to its capacity for assessing a wide range of impacts across spatial scales, providing spatially explicit assessment and understanding tradeoffs and synergies associated with the NBS implementation.

Starting with the Oslo Case study, stakeholders shared their perspectives on the utility of this approach for aiding, not only in the planning of green roofs, but also for other NBS in the city. For example, drawing from areas in Oslo where green roofs prove most effective, the municipality might consider offering financial incentives to encourage private building owners to develop them. Also, performance-based indicators working as policy instruments to promote NBS in urban property development (Stange et al., 2022), such as Oslo's Blue-green Factor, could incorporate in its calculation the possible tradeoffs and synergies associated with the impacts of each type of NBS. These enhancements could improve NBS planning, ensuring their effectiveness is maximized through strategic location based on expected overall impacts

In a more general sense, the NBS-vulnerability framework allows to perform an accurate place-specific assessment of NBS. It builds upon urban agendas for selecting local-scale vulnerabilities, in which indicators are assessed via site-specific thresholds, while broad-scale vulnerabilities are based on and related to global objectives (i.e., planetary boundaries). Furthermore, it enables co-creative and participatory approaches to better depict urban conditions (i.e., co-creation of scenarios of green roofs). These qualities provide policymakers with a versatile tool for better understanding the dynamics of NBS within their local contexts, in comparison with standardized values and individual methods for assessing isolated NBS impacts, that fail to grasp the full extent of NBS impacts in complex urban environments (Pereira et al., 2023) and may result in uncertainty and disconnections between short-term and long-term impacts (Kabisch et al., 2016).

In this same line, the integrative method proposed by the NBS-vulnerability framework enables the creation of an optimal configuration of NBS by developing a most favorable scenario. This scenario considers the cross-scale impacts of NBS based on stakeholders' preferences and depicts conditions where synergies are maximized and tradeoffs are reduced. By doing so, it provides a clear and understandable overview of where and how NBS could be most effective, facilitating decision-making regarding its implementation (Sowińska-Świerkosz & García, 2021; Yang et al., 2023).

In summary, the assessment of NBS impacts across spatial scales can provide a new piece of information for urban decision-makers engaged in making sustainable environments

(United Nations, 2015). The cross-spatial examination of NBS has been described as an essential concern when examining NBS outcomes (Albert et al., 2020) and when planning for sustainable, resilient and equitable urban environments (Elmqvist et al., 2019). As urban policies continue to play important roles in shaping and changing the regional and global linkages of cities (Bai et al., 2010), integrative tools that relate urban developments with their global impacts have the potential to assist in the sustainable transition or urban spaces (Yigitcanlar & Teriman, 2015).

IV.6.3. An improved methodological approach with considerations for future applications

The proposed assessment has successfully employed the vulnerability approach as a common ground for interpreting and calculating different types of urban NBS impacts while considering their spatially explicit context. This is particularly valuable, given that spatial analysis is considered crucial when examining the outcomes of NBS interventions (Albert et al., 2020).

While similar approaches have been previously explored (e.g. Camacho-Caballero et al., 2024), the presented NBS-vulnerability framework introduces a novelty by considering NBS impacts extending beyond urban boundaries (i.e., broad-scale vulnerabilities). This was achieved by assessing the affectation of urban NBS on PB, an aspect that, to the best of my knowledge, has not been previously explored. Linking NBS impacts with PBs allows for a better understanding of the effectiveness of NBS, as it provides a broader picture of its overall effects, helping to avoid undesired ones (Pereira et al., 2023; Seddon et al., 2020). This approach has allowed a novel integrated assessment considering NBS trade-offs and synergies across spatial scales, able to produce spatially explicit outcomes depicting optimal NBS configurations where the desired NBS impacts are maximized while undesired minimized (i.e., most favorable scenario).

This integrative approach was achieved through the successful convergence of different types of NBS assessments and impacts within the vulnerability sphere. By doing so, the NBS-vulnerability framework has established a common ground for effectively calculating, interpreting, sharing and co-generating information about NBS impacts. This framework has the potential to enhance NBS research by facilitating transdisciplinarity through the integration of diverse knowledge sources (Albert et al., 2020).

Furthermore, unlike other MCDA approaches that carry on normalization relying solely on indicator values (e.g., Feng et al., 2023), the proposed framework recognizes that exposure needs to be defined and normalized by context-specific thresholds. Because of this, thresholds require careful consideration. PB, for instance, offer useful thresholds for broad-scale vulnerabilities but faces methodological and ethical limitations (see Biermann & Kim, 2020). In this case, even after downscaling PB to the urban level, the green roof assessment remains partial, as the threshold does not consider how other urban activities are affecting the PB (e.g., state of greenhouse emissions produced by urban activities as a whole), or which portion of the downscaled PB should be allocated to green roofs (e.g., a tolerable amount of greenhouse emissions associated with the lifecycle of green roofs). In this sense, a refinement of the PB threshold is encouraged to gain a more nuanced understanding of the state of broad-scale vulnerabilities, allowing a better assessment of NBS impacts. For doing so, following existing threshold definition endeavors can be helpful. For instance, The Nature Index (Certain et al., 2011) is a set of ecological indicators portraying the state of ecosystems in any given area relative to a reference state (i.e., threshold), defined via collaborative expert judgment and monitoring-based estimates. Further threshold estimations for better relating PB to NBS could follow a similar co-creative approach.

In this same line, it was observed that the assessment of NBS is sensitive to impact prioritizations. As shown in the green roof case, different weighting schemes provided heterogenous results, proving the complexity behind optimizing NBS. Because of this, I emphasize the need for stakeholder involvement as these provide a diversity of perspectives that can aid in the intricacy of assessing NBS (Nesshöver et al., 2017).

Finally, the assessment of NBS' impacts on broad-scale vulnerabilities should not be limited by their capacity to be related to PB values. A great example of this can be described in the capacity of urban NBS to support regional-scale ecological connectivity (Molné et al., 2023), whose assessment is based on geographical metrics (i.e., network structures) that are not relatable to the biosphere integrity variables depicted in the PB framework (i.e., genetic diversity). In this sense, it is recommended to expand the understanding of urban NBS impacts beyond urban limits by seeking and incorporating assessment approaches that complement the PB framework.

In short, while the proposed framework has broadened the evaluative space of NBS, there remains room for improvement in areas such as the co-creation of thresholds and the evaluation of broad-scale vulnerabilities not directly related to PB. Strengthening these aspects would bolster the overall capabilities of the framework for a better understanding of the impacts of urban NBS.

IV.7. Conclusion

The objective of this study is to develop a framework capable of integrating NBS' impacts experienced within and beyond urban limits. For doing so, NBS impacts were calculated based on spatially explicit land-use scenarios and assessed based on their capacity to alter socio-ecological vulnerabilities at both the local and broader scales.

This study novelly operationalized PB within an NBS assessment. While future applications of the framework could further benefit from refining thresholds for relating NBS impacts with the PBs, the findings of this study revealed important new insights for urban green space planning. Primarily, it showed that NBS interventions can lead to both desirable and undesirable shifts in vulnerabilities across spatial scales. For instance, while green roofs were effective in reducing local-scale vulnerabilities (e.g., to heatwave and extreme rainwater events), they also increased broad-scale vulnerabilities (e.g., to climate change) due to the environmental impacts associated with their construction, implementation, and removal. Consequently, a maximization of NBS has not shown to be the most desired scenario, and specific areas – particularly Oslo's inner-city areas – obtained stronger priority for NBS implementation.

An evaluation of NBS under a vulnerability framing allows for NBS planning beyond the consideration of net benefits – typical for classical ecosystem services-based assessments – and enables the consideration of social-ecological spatial inequalities from a differentiated needs perspective. This integrative capacity of maximizing desired NBS impacts while minimizing undesired ones, can provide policymakers with valuable foresight for enhancing NBS planning that contributes to creating more sustainable, resilient, and equitable environments.

IV. References

Abdullah, M. F., Siraj, S., & Hodgett, R. E. (2021). An Overview of Multi-Criteria Decision Analysis (MCDA) Application in Managing Water-Related Disaster Events: Analyzing 20 Years of Literature for Flood and Drought Events. *Water 2021, Vol. 13, Page 1358*, *13*(10), 1358. https://doi.org/10.3390/W13101358

Albert, C., Brillinger, M., Guerrero, P., Gottwald, S., Henze, J., Schmidt, S., Ott, E., & Schröter, B. (2020). Planning nature-based solutions: Principles, steps, and insights. *Ambio*. https://doi.org/10.1007/s13280-020-01365-1

Álvarez-Rogel, J., Barberá, G. G., Maxwell, B., Guerrero-Brotons, M., Díaz-García, C., Martínez-Sánchez, J. J., Sallent, A., Martínez-Ródenas, J., González-Alcaraz, M. N., Jiménez-Cárceles, F. J., Tercero, C., & Gómez, R. (2020). The case of Mar Menor eutrophication: State of the art and description of tested Nature-Based Solutions. *Ecological Engineering*, *158*, 106086. https://doi.org/10.1016/J.ECOLENG.2020.106086

Bai, X., McAllister, R. R. J., Beaty, R. M., & Taylor, B. (2010). Urban policy and governance in a global environment: complex systems, scale mismatches and public participation. *Current Opinion in Environmental Sustainability*, *2*(3), 129–135. https://doi.org/10.1016/J.COSUST.2010.05.008

Bellezoni, R. A., Meng, F., He, P., & Seto, K. C. (2021). Understanding and conceptualizing how urban green and blue infrastructure affects the food, water, and energy nexus: A synthesis of the literature. *Journal of Cleaner Production*, *289*, 125825. https://doi.org/10.1016/j.jclepro.2021.125825

Biermann, F., & Kim, R. E. (2020). The boundaries of the planetary boundary framework: A critical appraisal of approaches to define a "safe operating space" for humanity. *Annual Review of Environment and Resources*, *45*(Volume 45, 2020), 497–521. https://doi.org/10.1146/ANNUREV-ENVIRON-012320-080337/CITE/REFWORKS

Bozeman, J. F., Nobler, E., & Nock, D. (2022). A Path Toward Systemic Equity in Life Cycle Assessment and Decision-Making: Standardizing Sociodemographic Data Practices. *Environmental Engineering Science*, *39*(9), 759–769. https://doi.org/10.1089/EES.2021.0375/ASSET/IMAGES/EES.2021.0375_FIGURE1.JPG

Bush, J., & Doyon, A. (2019). Building urban resilience with nature-based solutions: How can urban planning contribute? *Cities*, *95*, 102483. https://doi.org/10.1016/J.CITIES.2019.102483

Camacho-Caballero, D., Langemeyer, J., Segura-Barrero, R., Ventura, S., Beltran, A. M., & Villalba, G. (2024). Assessing Nature-based solutions in the face of urban vulnerabilities: A multi-criteria decision approach. *Sustainable Cities and Society*, *103*, 105257. https://doi.org/10.1016/J.SCS.2024.105257

Canadell, J. G., & Raupach, M. R. (2008). Managing Forests for Climate Change Mitigation. *Science*, *320*(5882), 1456–1457. https://doi.org/10.1126/SCIENCE.1155458

Certain, G., Skarpaas, O., Bjerke, J. W., Framstad, E., Lindholm, M., Nilsen, J. E., Norderhaug, A., Oug, E., Pedersen, H. C., Schartau, A. K., van der Meeren, G. I., Aslaksen, I., Engen, S., Garnåsjordet, P. A., Kvaløy, P., Lillegård, M., Yoccoz, N. G., & Nybø, S. (2011). The Nature Index: A General Framework for Synthesizing Knowledge on the State of Biodiversity. *PLOS ONE*, *6*(4), e18930. https://doi.org/10.1371/JOURNAL.PONE.0018930

Chisholm, O., Sharry, P., & Phillips, L. (2022). Multi-Criteria Decision Analysis for Benefit-Risk Analysis by National Regulatory Authorities. *Frontiers in Medicine*, *8*, 820335. https://doi.org/10.3389/FMED.2021.820335/BIBTEX

Cohen-Shacham, E., Janzem, C., Walters, G., & Maginnis, S. (2016). Nature-based solutions to address global societal challenges. In *Nature-based solutions to address global societal challenges* (IUCN, 2016). IUCN International Union for Conservation of Nature. https://doi.org/10.2305/iucn.ch.2016.13.en

Cutter, S. L. (2016). Resilience to What? Resilience for Whom? *The Geographical Journal*, *182*(2), 110–113. https://doi.org/10.1111/GEOJ.12174

Dsouza, N., Devadason, A., Senerat, A. M., Watanatada, P., Rojas-Rueda, D., & Sebag, G. (2023). Sustainability and Equity in Urban Development (S&EUD): A Content Analysis of "Bright Spots" from the Accelerating City Equity (ACE) Project. *Sustainability (Switzerland)*, *15*(9), 7318. https://doi.org/10.3390/SU15097318/S1

Dumitru, A., Frantzeskaki, N., & Collier, M. (2020). Identifying principles for the design of robust impact evaluation frameworks for nature-based solutions in cities. *Environmental Science & Policy*, *112*, 107–116. https://doi.org/10.1016/J.ENVSCI.2020.05.024

Dushkova, D., & Haase, D. (2020). Not Simply Green: Nature-Based Solutions as a Concept and Practical Approach for Sustainability Studies and Planning Agendas in Cities. *Land 2020, Vol. 9, Page 19*, *9*(1), 19. https://doi.org/10.3390/LAND9010019

Elmqvist, T., Andersson, E., Frantzeskaki, N., McPhearson, T., Olsson, P., Gaffney, O., Takeuchi, K., & Folke, C. (2019). Sustainability and resilience for transformation in the urban century. *Nature Sustainability*, *2*(4), 267–273. https://doi.org/10.1038/s41893-019-0250-1

European Commission. (2021). *Evaluating the impact of nature-based solutions: A handbook for practitioners*.

Feng, D., Shi, X., & Renaud, F. G. (2023). Risk assessment for hurricane-induced pluvial flooding in urban areas using a GIS-based multi-criteria approach: A case study of Hurricane Harvey in Houston, USA. *Science of The Total Environment*, *904*, 166891. https://doi.org/10.1016/J.SCITOTENV.2023.166891

Firbank, L. G., Petit, S., Smart, S., Blain, A., & Fuller, R. J. (2007). Assessing the impacts of agricultural intensification on biodiversity: a British perspective. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *363*(1492), 777–787. https://doi.org/10.1098/RSTB.2007.2183

Funtowicz, S. O., Martinez-Alier, J., Munda, G., & Ravetz, J. R. (1999). Information tools for environmental policy under conditions of complexity. Environmental issues series No 9. *European Environment Agency. Copenhaguen*.

Füssel, H. M. (2010). How inequitable is the global distribution of responsibility, capability, and vulnerability to climate change: A comprehensive indicator-based assessment. *Global Environmental Change*, *20*(4), 597–611. https://doi.org/10.1016/J.GLOENVCHA.2010.07.009

Gagliano, A., & Cascone, S. (2024). Eco-friendly green roof solutions: Investigating volcanic ash as a viable alternative to traditional substrates. *Construction and Building Materials*, *411*, 134442. https://doi.org/10.1016/J.CONBUILDMAT.2023.134442

Gargari, C., Bibbiani, C., Fantozzi, F., & Campiotti, C. A. (2016). Environmental Impact of Green Roofing: The Contribute of a Green Roof to the Sustainable use of Natural Resources in a Life Cycle Approach. *Agriculture and Agricultural Science Procedia*, *8*, 646–656. https://doi.org/10.1016/J.AASPRO.2016.02.087

Giama, E., Papageorgiou, C., Theodoridou, I., & Papadopoulos, A. M. (2021). Life Cycle Analysis and Life Cycle Cost Analysis of green roofs in the Mediterranean climatic conditions. *Energy Sources, Part A: Recovery, Utilization, and Environmental Effects*. https://doi.org/10.1080/15567036.2021.1914782

Gilabert, J., Ventura, S., Segura, R., Martilli, A., Badia, A., Llasat, C., Corbera, J., & Villalba, G. (2021). Abating heat waves in a coastal Mediterranean city: What can cool roofs and vegetation contribute? *Urban Climate*, *37*, 100863. https://doi.org/10.1016/J.UCLIM.2021.100863

Girling, R. D., Lusebrink, I., Farthing, E., Newman, T. A., & Poppy, G. M. (2013). Diesel exhaust rapidly degrades floral odours used by honeybees. *Scientific Reports 2013 3:1*, *3*(1), 1–5. https://doi.org/10.1038/srep02779

Gulyas, B. Z., & Edmondson, J. L. (2021). Increasing City Resilience through Urban Agriculture: Challenges and Solutions in the Global North. *Sustainability 2021, Vol. 13, Page 1465*, *13*(3), 1465. https://doi.org/10.3390/SU13031465

Hanson, H. I., Wickenberg, B., & Alkan Olsson, J. (2020). Working on the boundaries—How do science use and interpret the nature-based solution concept? *Land Use Policy*, *90*, 104302. https://doi.org/10.1016/J.LANDUSEPOL.2019.104302

Hawes, J. K., Goldstein, B. P., Newell, J. P., Dorr, E., Caputo, S., Fox-Kämper, R., Grard, B., Ilieva, R. T., Fargue-Lelièvre, A., Poniży, L., Schoen, V., Specht, K., & Cohen, N. (2024). Comparing the carbon footprints of urban and conventional agriculture. *Nature Cities 2024 1:2*, *1*(2), 164–173. https://doi.org/10.1038/s44284-023-00023-3

Häyhä, T., Lucas, P. L., van Vuuren, D. P., Cornell, S. E., & Hoff, H. (2016). From Planetary Boundaries to national fair shares of the global safe operating space — How can the scales be bridged? *Global Environmental Change*, *40*, 60–72. https://doi.org/10.1016/J.GLOENVCHA.2016.06.008

Herreros-Cantis, P., & McPhearson, T. (2021). Mapping supply of and demand for ecosystem services to assess environmental justice in New York City. *Ecological Applications*, *31*(6), e02390. https://doi.org/10.1002/EAP.2390

Johannessen, B. G., Hanslin, H. M., & Muthanna, T. M. (2017). Green roof performance potential in cold and wet regions. *Ecological Engineering*, *106*, 436–447. https://doi.org/10.1016/J.ECOLENG.2017.06.011

Kabisch, N., Frantzeskaki, N., Pauleit, S., Naumann, S., Davis, M., Artmann, M., Haase, D., Knapp, S., Korn, H., Stadler, J., Zaunberger, K., & Bonn, A. (2016). Nature-based solutions to climate change mitigation and adaptation in urban areas: Perspectives on indicators, knowledge gaps, barriers, and opportunities for action. *Ecology and Society*, *21*(2). https://doi.org/10.5751/ES-08373-210239

Kallioniemi, E., Åström, J., Rusch, G. M., Dahle, S., Åström, S., & Gjershaug, J. O. (2017). Local resources, linear elements and mass-flowering crops determine bumblebee occurrences in moderately intensified farmlands. *Agriculture, Ecosystems & Environment*, *239*, 90–100. https://doi.org/10.1016/J.AGEE.2016.12.039

Keith, H., Vardon, M., Obst, C., Young, V., Houghton, R. A., & Mackey, B. (2021). Evaluating nature-based solutions for climate mitigation and conservation requires comprehensive carbon accounting. *Science of The Total Environment*, *769*, 144341. https://doi.org/10.1016/J.SCITOTENV.2020.144341

Kratschmer, S., Kriechbaum, M., & Pachinger, B. (2018). Buzzing on top: Linking wild bee diversity, abundance and traits with green roof qualities. *Urban Ecosystems*, *21*(3), 429–446. https://doi.org/10.1007/S11252-017-0726-6/FIGURES/6

Kruse, M., N, B. D., Zander, V., Megan, N., & Zofie, C. (2022). Making urban ecosystem mapping accessible to the public. *Papers: Regió Metropolitana de Barcelona: Territori, Estratègies, Planejament*, *0*(64). https://raco.cat/index.php/PapersIERMB/article/view/402529

Langemeyer, J., Madrid-Lopez, C., Mendoza Beltran, A., & Villalba Mendez, G. (2021). Urban agriculture — A necessary pathway towards urban resilience and global sustainability? *Landscape and Urban Planning*, *210*, 104055. https://doi.org/10.1016/J.LANDURBPLAN.2021.104055

Langemeyer, J., Wedgwood, D., McPhearson, T., Baró, F., Madsen, A. L., & Barton, D. N. (2020). Creating urban green infrastructure where it is needed – A spatial ecosystem service-based decision analysis of green roofs in Barcelona. *Science of The Total Environment*, *707*, 135487. https://doi.org/10.1016/J.SCITOTENV.2019.135487

Liberalesso, T., Oliveira Cruz, C., Matos Silva, C., & Manso, M. (2020). Green infrastructure and public policies: An international review of green roofs and green walls incentives. *Land Use Policy*, *96*, 104693. https://doi.org/10.1016/J.LANDUSEPOL.2020.104693

Little, R. G. (2010). Controlling Cascading Failure: Understanding the Vulnerabilities of Interconnected Infrastructures. *Http://Dx.Doi.Org/10.1080/106307302317379855*, *9*(1), 109–123. https://doi.org/10.1080/106307302317379855

McDougall, R., Kristiansen, P., & Rader, R. (2019). Small-scale urban agriculture results in high yields but requires judicious management of inputs to achieve sustainability. *Proceedings of the National Academy of Sciences of the United States of America*, *116*(1), 129–134. https://doi.org/10.1073/PNAS.1809707115

Molné, F., Donati, G. F. A., Bolliger, J., Fischer, M., Maurer, M., & Bach, P. M. (2023). Supporting the planning of urban blue-green infrastructure for biodiversity: A multi-scale prioritisation framework. *Journal of Environmental Management*, *342*, 118069. https://doi.org/10.1016/J.JENVMAN.2023.118069

Nesshöver, C., Assmuth, T., Irvine, K. N., Rusch, G. M., Waylen, K. A., Delbaere, B., Haase, D., Jones-Walters, L., Keune, H., Kovacs, E., Krauze, K., Külvik, M., Rey, F., van Dijk, J., Vistad, O. I., Wilkinson, M. E., & Wittmer, H. (2017). The science, policy and practice of nature-based solutions: An interdisciplinary perspective. In *Science of the Total Environment* (Vol. 579, pp. 1215–1227). Elsevier B.V. https://doi.org/10.1016/j.scitotenv.2016.11.106

Nieto, A. (2014). *European Red List of bees*. International Union for Conservation of Nature. https://doi.org/https://doi.org/10.2779/51181

Oslo Kommune. (2012). *Strategi for overvannshåndtering i oslo*.

Oslo Kommune. (2018). *Grøntregnskap: en måling av grønnstruktur i Oslos byggesone*.

Oslo Kommune. (2020). *Miljøhovedstaden 2019*. Oslo Kommune. https://www.oslo.kommune.no/miljo-og-klima/miljohovedstaden-2019/#gref

Oslo Kommune. (2021a). *Klimastrategi for Oslo mot 2030*.

Oslo Kommune. (2021b). *Strategi for grønne tak og fasader: Hovedresultater fra Plan- og bygningsetatens kartlegging av grønne tak* .

Oslo Kommune. (2021c). *Tiltaksutredning for bedre luftkvalitet i Oslo og Bærum 2020-2025*.

Oslo Kommune. (2022). *Strategi for grønne tak og fasader: 2030 grønne tak og fasader i 2030*. https://www.oslo.kommune.no/getfile.php/13452654-1654694941/Tjenester%20og%20tilbud/Plan%2C%20bygg%20og%20eiendom/Byggesaksveiledere%2C%20normer%20og%20skjemaer/Strategi%20for%20gr%C3%B8nne%20tak%20og%20fasader.pdf

Oslo Kommune. (2023a). *Handlingsplan for biologisk mangfold i Oslo 2023–2030*.

Oslo Kommune. (2023b). *Overvannsveileder: Retningslinjer og veiledning for overvannshåndtering i Oslo kommune*.

Panno, A., Carrus, G., Lafortezza, R., Mariani, L., & Sanesi, G. (2017). Nature-based solutions to promote human resilience and wellbeing in cities during increasingly hot summers. *Volume 159, Pages 249 - 256*, *159*, 249–256. https://doi.org/10.1016/j.envres.2017.08.016

Passaseo, A., Pétremand, G., Rochefort, S., & Castella, E. (2020). Pollinator emerging from extensive green roofs: wild bees (Hymenoptera, Antophila) and hoverflies (Diptera, Syrphidae) in Geneva (Switzerland). *Urban Ecosystems*, *23*(5), 1079–1086. https://doi.org/10.1007/S11252-020-00973-9/TABLES/4

Pereira, P., Yin, C., & Hua, T. (2023). Nature-based solutions, ecosystem services, disservices, and impacts on well-being in urban environments. *Current Opinion in Environmental Science & Health*, *33*, 100465. https://doi.org/10.1016/J.COESH.2023.100465

Randrup, T. B., Buijs, A., Konijnendijk, C. C., & Wild, T. (2020). Moving beyond the nature-based solutions discourse: introducing nature-based thinking. *Urban Ecosystems*, *23*(4), 919–926. https://doi.org/10.1007/S11252-020-00964-W/FIGURES/1

Raymond, C. M., Pam, B., Breil, M., Nita, M. R., Kabisch, N., de Bel, M., Enzi, V., Frantzeskaki, N., Geneletti, D., Cardinaletti, M., Lovinger, L., Basnou, C., Monteiro, A., Robrecht, H., Sgrigna, G., Munari, L., & Calfapietra, C. (2017). An Impact Evaluation Framework to Support Planning and Evaluation of Nature-based Solutions Projects. In *EKLIPSE Expert Working Group report* (Issue February).

Richardson, K., Steffen, W., Lucht, W., Bendtsen, J., Cornell, S. E., Donges, J. F., Drüke, M., Fetzer, I., Bala, G., von Bloh, W., Feulner, G., Fiedler, S., Gerten, D., Gleeson, T., Hofmann, M., Huiskamp, W., Kummu, M., Mohan, C., Nogués-Bravo, D., … Rockström, J. (2023). Earth beyond six of nine planetary boundaries. *Science Advances*, *9*(37). https://doi.org/10.1126/SCIADV.ADH2458/SUPPL_FILE/SCIADV.ADH2458_SM.PDF

Rockström, J., Steffen, W., Noone, K., Persson, A., Chapin, F. S. 3rd, Lambin, E. F., Lenton, T. M., Scheffer, M., Folke, C., Schellnhuber, H. J., Nykvist, B., de Wit, C. A., Hughes, T., van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P. K., Costanza, R., Svedin, U., … Foley, J. A. (2009). A safe operating space for humanity. *Nature*, *461*(7263), 472–475. https://doi.org/10.1038/461472a

Rödl, A., & Arlati, A. (2022). A general procedure to identify indicators for evaluation and monitoring of nature-based solution projects. *Ambio*, *51*(11), 2278–2293. https://doi.org/10.1007/S13280-022-01740-0/TABLES/3

Ryberg, M. W., Owsianiak, M., Richardson, K., & Hauschild, M. Z. (2018). Development of a life-cycle impact assessment methodology linked to the Planetary Boundaries framework. *Ecological Indicators*, *88*, 250–262. https://doi.org/10.1016/J.ECOLIND.2017.12.065

Sala, S., Crenna, E., Secchi, M., & Sanyé-Mengual, E. (2020). Environmental sustainability of European production and consumption assessed against planetary boundaries. *Journal of Environmental Management*, *269*, 110686. https://doi.org/10.1016/J.JENVMAN.2020.110686

Sandin, G., Peters, G. M., & Svanström, M. (2015). Using the planetary boundaries framework for setting impact-reduction targets in LCA contexts. *International Journal of Life Cycle Assessment*, *20*(12), 1684–1700. https://doi.org/10.1007/S11367-015-0984-6/TABLES/5

Santoro, S., Pluchinotta, I., Pagano, A., Pengal, P., Cokan, B., & Giordano, R. (2019). Assessing stakeholders' risk perception to promote Nature Based Solutions as flood protection strategies: The case of the Glinščica river (Slovenia). *Science of The Total Environment*, *655*, 188–201. https://doi.org/10.1016/J.SCITOTENV.2018.11.116

Sanye Mengual, E., & Sala, S. (2023). *Consumption Footprint and Domestic Footprint: Assessing the environmental impacts of EU consumption and production* (Issue KJ-NA-31-390-EN-N (online),KJ-NA-31-390-EN-C (print)). Publications Office of the European Union. https://doi.org/10.2760/218540 (online),10.2760/3878 (print)

Saarikoski, H., Mustajoki, J., Barton, D. N., Geneletti, D., Langemeyer, J., Gomez-Baggethun, E., Marttunen, M., Antunes, P., Keune, H., & Santos, R. (2016). Multi-Criteria Decision Analysis and Cost-Benefit Analysis: Comparing alternative frameworks for integrated valuation of ecosystem services. *Ecosystem Services*, *22*(October), 238–249. https://doi.org/10.1016/j.ecoser.2016.10.014

Scolaro, T. P., & Ghisi, E. (2022). Life cycle assessment of green roofs: A literature review of layers materials and purposes. *Science of The Total Environment*, *829*, 154650. https://doi.org/10.1016/J.SCITOTENV.2022.154650

Seddon, N., Chausson, A., Berry, P., Girardin, C. A. J., Smith, A., & Turner, B. (2020). Understanding the value and limits of nature-based solutions to climate change and other global challenges. In *Philosophical Transactions of the Royal Society B: Biological Sciences* (Vol. 375, Issue 1794). Royal Society Publishing. https://doi.org/10.1098/rstb.2019.0120

Seto, K. C., Reenberg, A., Boone, C. G., Fragkias, M., Haase, D., Langanke, T., Marcotullio, P., Munroe, D. K., Olah, B., & Simon, D. (2012). Urban land teleconnections and sustainability. *Proceedings of the National Academy of Sciences of the United States of America*, *109*(20), 7687–7692. https://doi.org/10.1073/pnas.1117622109

Shafique, M., Azam, A., Rafiq, M., Ateeq, M., & Luo, X. (2019). *An overview of life cycle assessment of green roofs*. https://doi.org/10.1016/j.jclepro.2019.119471

Shah, M. A. R., Renaud, F. G., Anderson, C. C., Wild, A., Domeneghetti, A., Polderman, A., Votsis, A., Pulvirenti, B., Basu, B., Thomson, C., Panga, D., Pouta, E., Toth, E., Pilla, F., Sahani, J., Ommer, J., El Zohbi, J., Munro, K., Stefanopoulou, M., … Zixuan, W. (2020). A review of hydro-meteorological hazard, vulnerability, and risk assessment frameworks and indicators in the context of nature-based solutions. *International Journal of Disaster Risk Reduction*, *50*, 101728. https://doi.org/10.1016/J.IJDRR.2020.101728

Sowińska-Świerkosz, B., & García, J. (2021). A new evaluation framework for nature-based solutions (NBS) projects based on the application of performance questions and indicators approach. *Science of the Total Environment*, *787*. https://doi.org/10.1016/j.scitotenv.2021.147615

Stange, E. E., Barton, D. N., Andersson, E., & Haase, D. (2022). Comparing the implicit valuation of ecosystem services from nature-based solutions in performance-based green area indicators across three European cities. *Landscape and Urban Planning*, *219*, 104310. https://doi.org/10.1016/J.LANDURBPLAN.2021.104310

Strand, N. P. (2023, June 5). *Befolkningsframskrivingen for Oslo 2023–2050*. Oslo Kommune. https://www.oslo.kommune.no/statistikk/befolkning/befolkningsframskrivingen-for-oslo-2023-2050

Taguchi, V. J., Weiss, P. T., Gulliver, J. S., Klein, M. R., Hozalski, R. M., Baker, L. A., Finlay, J. C., Keeler, B. L., & Nieber, J. L. (2020). It Is Not Easy Being Green: Recognizing Unintended

Consequences of Green Stormwater Infrastructure. *Water 2020, Vol. 12, Page 522*, *12*(2), 522. https://doi.org/10.3390/W12020522

Thokala, P., & Madhavan, G. (2018). Stakeholder involvement in Multi-Criteria Decision Analysis. *Cost Effectiveness and Resource Allocation*, *16*(1), 1–3. https://doi.org/10.1186/S12962-018-0120-0/METRICS

United Nations. (2015). *Goal 11: Make cities inclusive, safe, resilient and sustainable*. https://www.un.org/sustainabledevelopment/cities/

United Nations. (2022). *Resolution adopted by the United Nations Environment Assembly on 2 March 2022*.

URBAG. (2021). *Green roofs in Oslo: results from a co-creative and participatory approach for its evaluation*. https://urbag.eu/green-roofs-in-oslo-co-creative-approach/

van den Bosch, M., & Ode Sang. (2017). Urban natural environments as nature-based solutions for improved public health – A systematic review of reviews. *Environmental Research*, *158*, 373–384. https://doi.org/10.1016/J.ENVRES.2017.05.040

Venter, Z. S., Barton, D. N., Martinez-Izquierdo, L., Langemeyer, J., Baró, F., & McPhearson, T. (2021). Interactive spatial planning of urban green infrastructure – Retrofitting green roofs where ecosystem services are most needed in Oslo. *Ecosystem Services*, *50*(June), 101314. https://doi.org/10.1016/j.ecoser.2021.101314

Vijayaraghavan, K. (2016). Green roofs: A critical review on the role of components, benefits, limitations and trends. *Renewable and Sustainable Energy Reviews*, *57*, 740–752. https://doi.org/10.1016/J.RSER.2015.12.119

Wang, R., Goll, D., Balkanski, Y., Hauglustaine, D., Boucher, O., Ciais, P., Janssens, I., Penuelas, J., Guenet, B., Sardans, J., Bopp, L., Vuichard, N., Zhou, F., Li, B., Piao, S., Peng, S., Huang, Y., & Tao, S. (2017). Global forest carbon uptake due to nitrogen and phosphorus deposition from 1850 to 2100. *Global Change Biology*, *23*(11), 4854–4872. https://doi.org/10.1111/GCB.13766

Yang, S., Ruangpan, L., Torres, A. S., & Vojinovic, Z. (2023). Multi-objective Optimisation Framework for Assessment of Trade-Offs between Benefits and Co-benefits of Nature-based Solutions. *Water Resources Management*, *37*(6–7), 2325–2345. https://doi.org/10.1007/S11269-023-03470-8/TABLES/4

Yigitcanlar, T., & Teriman, S. (2015). Rethinking sustainable urban development: towards an integrated planning and development process. *International Journal of Environmental Science and Technology*, *12*(1), 341–352. https://doi.org/10.1007/S13762-013-0491-X/FIGURES/5

Zhang, Q., Qi, J., Cheng, B., Yu, C., Liang, S., Wiedmann, T. O., Liu, Y., & Zhong, Q. (2021). Planetary Boundaries for Forests and Their National Exceedance. *Environmental Science and Technology*, *55*(22), 15423–15434. https://doi.org/10.1021/ACS.EST.1C02513/SUPPL_FILE/ES1C02513_SI_002.XLSX

CHAPTER V. SCIENTIFIC CONTRIBUTIONS AND FUTURE RESEARCH

This PhD thesis has expanded the understanding and assessment of urban NBS impacts, by providing four main contributions:

1. Advancing the comprehension of how urban NBS impacts could be simultaneously affecting the urban challenges of Sustainability, Resilience and Equity (**theoretical**).
2. Developing a novel framework based on vulnerability assessment to simultaneously assess the desired and undesired impacts of urban NBS (**methodological**)
3. Demonstrating the complexity of assessing urban NBS, as its impacts can shift vulnerabilities across different spatial scales and create both synergies and tradeoffs (**empirical**).
4. Providing recommendations for the policy and practice involved in the strategic planning of urban NBS (**planning and policy**).

In the following subsections of this last concluding chapter, each contribution will be summarized by providing a brief discussion and the main conclusions of the research.

V.1. Nature-based solutions in the face of sustainability, resilience and equity

Previous studies have highlighted shortcomings in current methods for assessing NBS, suggesting a lack of clarity regarding their overall effectiveness (e.g., Dumitru et al., 2020; Rödl & Arlati, 2022). These deficiencies are particularly significant with NBS increasingly mainstreaming. This thesis thoroughly identifies specific limitations hindering the proper assessment of NBS by examining existing methodologies and frameworks in the literature (Chapter I). Among these limitations, it was found that existing approaches often struggle to integrate various types of NBS impacts, tend to overlook undesired impacts of NBS and fail to consider potential effects beyond the specific site of implementation.

These factors become relevant in complex urban environments, which are known for their intricate and sometimes unpredictable nature (McPhearson et al., 2016), where the challenges of sustainability, resilience, and equity are interconnected (Berbés-Blázquez et al., 2023; Elmqvist et al., 2019). While NBS are acknowledged for their potential to address urban challenges, there often exists an implicit assumption that they will simultaneously and positively impact resilience, equity, and sustainability (European Commission, 2019; United

Nations, 2022). However, this assumption remains unverified, as assessments of the NBS influence on these challenges typically focus on just one or two dimensions at a time (e.g., Langemeyer et al., 2021; Bush & Doyon, 2019; Meerow et al., 2019). In response to this oversight, this thesis explicitly explores the relationships and **potential interactions between NBS and multiple urban challenges.**

As detailed in Chapter II, NBS impacts can lead to unexpected consequences, simultaneously influencing urban challenges in both desirable and undesirable ways. While sustainability and resilience are often presented together in urban regeneration programs, this thesis has further described how NBS impacts can tackle both of these challenges. For instance, by providing energy reductions associated with temperature regulations (improving sustainability) while also reducing flood impacts during heavy rainfall periods (improving resilience). However, tensions may arise if NBS solely focus on enhancing resource efficiency for sustainability without considering their role in addressing unforeseen hazards during crises, which is crucial for enhancing resilience.

In terms of sustainability and equity, it was found that NBS influence not only the urban environments where they are implemented but also their surroundings and even distant locations (see Chapter IV). This means that while NBS aim to improve sustainability within urban areas, they can create unintended negative impacts beyond city limits, raising concerns about equitable impact distribution. For instance, although green roofs can bolster urban sustainability by improving rooftop durability, the production and use of fertilizers for their maintenance can exacerbate climate change and eutrophication risks both within and beyond urban limits.

Regarding resilience and equity, tensions around the provision of desired resilient NBS impacts can arise if these are distributed across areas with higher adaptative capacities for exposure hazards. Such is the case of the private development of flood-protected zones, leaving those with no capacity to access these developments with a lack of resilient capacities, and therefore, an inequitable distribution of NBS impacts.

The work presented here indicates that synergies between urban sustainability, resilience, and equity challenges could be achieved by carefully developing NBS that prioritize longevity, risk management, and environmental justice. Thereby, NBS interventions can efficiently serve vulnerable groups while maintaining adaptability. However, to achieve this multifunctionality across sustainability, resilience and justice objectives simultaneously, it is crucial to gain a more comprehensive understanding of the implications of NBS functioning in complex urban environments, to which this thesis has contributed. However, the analysis presented herein only provides the beginning of an integrated assessment towards a deeper insight into the tradeoffs and synergies inherent in NBS implementation. Future research will require shedding further light on the complexities involved in designing urban NBS to achieve sustainable, resilient, and equitable outcomes.

V.2. A vulnerability approach for assessing the impacts of Nature-based solutions

This thesis has made methodological advances by developing the NBS-vulnerability framework. This novel approach has successfully employed the vulnerability approach as a common ground for **integrating** and evaluating various types of urban NBS impacts, thereby expanding the use of vulnerability assessment beyond its typical application in risk and disaster management (Pan et al., 2021).

The NBS-vulnerability framework represents an important methodological advancement in understanding the complexities inherent in urban NBS endeavors. This is a valuable aspect considering that NBS can be positioned within the realm of post-normal science, as NBS produce unpredictable impacts, for which there is incomplete control. For instance, the proposed framework simultaneously **considers the potential impacts of NBS beyond their immediate surroundings** and relates them with the expected effects on the local environments where they are implemented, shedding light on possible urban teleconnections arising from NBS impacts (Seto et al., 2012). This was achieved by assessing the impacts of NBS on planetary boundaries, a novel approach that, to the best of my knowledge, has not been previously employed for assessing urban NBS. By doing so, the NBS-vulnerability framework provides a way forward for operationalizing the interpretation and calculation of the wide range of impacts that can arise from urban NBS implementation. Thereby, the NBS-vulnerability approach offers clearer evidence to support the implementation of NBS in urban environments, addressing the ambiguity surrounding the interpretation of NBS effectiveness (Melanidis & Hagerman, 2022).

Furthermore, the proposed methodology follows a **stepwise approach** to estimate shifts in urban vulnerabilities across scenarios representing different NBS interventions. These steps include the development of scenarios, selection of vulnerabilities, mapping/calculation of indicators, normalization of indicators, aggregation of indicators for single vulnerabilities, stakeholder weighting and development of a most favorable scenario. This methodological structure allows for a versatile application of the framework for assessing different types of NBS within diverse urban environments. This was successfully tested using two different case studies with different urban configurations in terms of geography, urban structure, green infrastructure and population density: urban and peri-urban agriculture in the Metropolitan Area of Barcelona and green roofs in the Municipality of Oslo.

By doing so, the proposed stepwise framework was able to depict different vulnerabilities experienced in each of these environments and assess their associated shifts under various land use configurations driven by NBS. By successfully applying the methodology to the two case studies, the NBS-vulnerability framework has shown its adaptability and robustness, indicating its potential for use in other urban areas.

Moreover, the framework dealt with two distinct types of NBS for each case study. In the Metropolitan Area of Barcelona, urban and peri-urban agriculture falls under Type 2 NBS, involving the enhancement or diversification of existing ecosystems. Conversely, green roofs in the Oslo municipality represent Type 3 NBS, entailing the creation of new ecosystems. These NBS have varying land use implications: the expansion of (peri)urban agriculture led to the reduction of other land uses in the urban area, competing with existing green spaces and built-up areas. In contrast, the implementation of green roofs did not involve land use competition, as they were installed in previously unused spaces of urban rooftops. Despite these diverse behaviors, the NBS-vulnerability framework effectively captured the impacts of each NBS on different vulnerabilities, demonstrating its versatility in assessing the effects of various NBS types.

In the application of the NBS-vulnerability framework, the significance of **stakeholder weighting** was found to be a crucial aspect in assessing NBS impacts, aligning with findings in MCDA applications (Chen et al., 2010). This becomes apparent in the overall results of the NBS-vulnerability framework, particularly in the development of most favorable scenarios for green roof implementation. There, different configurations emerge when comparing equal weights to stakeholder weights, highlighting the complexity of calculating and interpreting NBS impacts, which can vary based on participants' perceptions.

Another crucial yet relatively overlooked aspect in the NBS literature is the implementation of novel **normalization practices**. While normalization generally follows a min-max approach within the complex process of calculating and interpreting NBS impacts, explicitly incorporating thresholds appears to be a necessary additional step to enhance NBS assessments. This approach proved beneficial in both case studies, as thresholds enabled a contextualized understanding of NBS impacts. For example, the temperature regulation capacities of urban agriculture in the Metropolitan Area of Barcelona and green roofs in Oslo were assessed using distinct context-specific thresholds. These thresholds were tailored to define the social and meteorological conditions of heatwaves, enhancing the precision of impact evaluations.

V.3. Nature-based solutions impacts shift vulnerabilities across spatial scales, creating both synergies and tradeoffs

The studied NBS were able to **reduce local-scale vulnerabilities** in most of the cases. For instance, the expansion of both (peri)urban agriculture and green roofs was associated with a decrease in vulnerability to lack of recreational spaces and natural environments. However, other vulnerabilities, such as vulnerability to heat were only mitigated by increases in the green roof presence, even when (peri) urban agricultural areas have been associated with heatwave regulation in urban areas (Kueppers et al., 2007).

A greater contrast was found in the impacts of NBS on local biodiversity conditions: while the expansion of (peri)urban agriculture increases biodiversity vulnerability, green roofs contributed to reducing it. These contrasting behaviors, however, need further consideration in conjunction with their impacts on broad-scale vulnerabilities, as will be further described later in this section.

Furthermore, it was noted in both cases that **the impacts of NBS on local-scale vulnerabilities were directly influenced by the location and spatial distribution of NBS**, consistent with previous assessments of spatially explicit NBS impacts (Herreros-Cantis & McPhearson, 2021). For example, densely populated areas offered opportunities for NBS to mitigate local-scale vulnerabilities in both case studies, corroborating findings from previous studies that indicate urban areas with high human population density are at increased risk of hazards due to the greater concentration of people in small areas (Guan et al., 2022; Sera et al., 2019), and often require a higher amount of ecosystem services (Gómez-Baggethun & Barton, 2013).

In the case of broad-scale vulnerabilities, empirical results are only based on those observations from the green roof case study in Oslo. Even when these vulnerabilities experienced linear increases that followed the amount of green roofs planned for the city, certain vulnerabilities were more significantly impacted than others (e.g., *Vulnerability to climate* *change* was the most affected by the implementation of green roofs, while *Vulnerability to stratospheric ozone depletion* was the least affected).

When considering the impacts of NBS on both local and broad vulnerabilities, the green roof case study provides a clear illustration of **cross-scale tradeoffs**: while all local-scale vulnerabilities were reduced or at least remained unaffected by the expansion of green roofs, all broad-scale vulnerabilities were increased by this expansion. However, this may not always hold true for other NBS. Broad-scale vulnerabilities could also be reduced by the expansion of NBS, as exemplified by their utilization as carbon sinks, which mitigates *vulnerability to climate change* (Pereira et al., 2024). Furthermore, the urban agriculture case study demonstrated that NBS implementation could lead to an increase in local-scale vulnerabilities, as evidenced by *Vulnerability of loss of biodiversity*.

Moreover, the observed impacts of NBS on biodiversity conditions serve as a prime example of how an **insufficient assessment of NBS impacts could lead to a misinterpretation of their implications**. Merely evaluating the local-scale impacts of green roofs might suggest that they only provide desirable impacts on the *Vulnerability to lack of habitats for pollinators*. However, as revealed by an examination of broad-scale vulnerabilities, the expansion of green roofs also heightened V*ulnerability to changes in biogeochemical flows*, driven by the potential eutrophication produced during the green roofs' lifecycle, adversely affecting both terrestrial and marine biodiversity conditions (Firbank et al., 2007).

Therefore, **maximizing NBS deployment does not necessarily guarantee the most desirable outcomes**. As illustrated by the green roof case study, NBS can generate both positive and negative impacts on vulnerabilities. Recognizing and weighing these trade-offs offers a path forward for determining the optimal configuration of NBS deployment in urban environments. By strategically locating NBS, synergies can be harnessed to address the highest number of local-scale vulnerabilities while minimizing undesirable impacts on both local and broad-scale vulnerabilities.

V.4. Recommendations for policy and practice

These recommendations aim to address existing constraints in the strategic planning of urban NBS, in order to promote their benefits and reduce their unintended consequences:

- **Recognize NBS' synergies and tradeoffs**: while NBS has been promoted as a core instrument for addressing societal challenges, their actual impacts will differ between NBS and the urban environments in which they are implemented due to their context-sensitive and site-specific nature (Raparthi & Vedamuthu, 2022). As NBS implementation often follows a one-size-fits-all approach, urban planners must acknowledge that NBS interventions can generate both synergies and tradeoffs (Colléony & Shwartz, 2019) based on the unique conditions in which they are developed. Furthermore, tradeoffs in urban NBS extend beyond prioritizing benefits, as these can also entail negative or undesired effects. For instance, while the case of urban agriculture successfully reduced many vulnerabilities in the Metropolitan Area of Barcelona, it also led to an increase in the *vulnerability of loss of biodiversity*.

- **Consider NBS impacts happening beyond urban limits**: NBS play a role in urban teleconnections, with impacts extending beyond the boundaries of their implementation. As demonstrated in Chapter IV, the lifecycle of green roofs can produce undesired impacts in remote locations, offsetting the socio-ecological benefits within the urban environment. These cross-scale impacts are part of the broader urban dynamics that must be acknowledged and addressed to create more sustainable urban spaces.
- **Acknowledge different perspectives when interpreting NBS' impacts**: the desired and undesired impacts of NBS need to be considered from different perspectives as its interpretation is not always straightforward. For doing so, the sustainability, resilience and equity challenges provide a way forward to judge the overall effectiveness of NBS. In this sense, policymakers can make use of these premises to assess if NBS are able to simultaneously provide long-lasting results (i.e., sustainable), oriented towards risk management and risk reduction (i.e., resilient) while being environmentally just (i.e., equitable). Also, and in order to better understand and assess the NBS impacts and the environments in which these are being deployed, it is advised to engage with a wide range of stakeholders with different values and backgrounds that are able to aid in addressing the complexity of planning NBS by considering diverse urban perspectives (Nesshöver et al., 2017).
- **Pursue integrated assessments of NBS' impacts** to avoid assessing isolated NBS impacts and treating their social and ecological implications as opposing or conflicting factors (Dumitru et al., 2020). Instead, it is encouraged to pursue an integrated assessment of NBS impacts that simultaneously considers various NBS effects, enabling anticipation of their impacts across different spatial scales. By doing so, urban NBS implementation can facilitate the sustainable, resilient, and equitable transition of urban spaces (United Nations, 2015).

V.5. Limitations and caveats

The NBS-vulnerability framework relies on multi-criteria decision analysis, a method enabling the integration of diverse criteria and conflicting goals. However, this integration introduces **complexity in its application**, which may constrain the quality of results. Specifically, the aggregation of numerous criteria (in this case, vulnerabilities) can lead to information loss and challenges in result interpretation (Boggia et al., 2018; Gonzalez & Enríquez-De-Salamanca, 2018). A good example of this was observed during the homogenization of spatially explicit results, where indicators with varying spatial resolutions were transformed onto a common grid, sometimes resulting in the loss of detailed spatial data.

In this same vein, while the NBS-vulnerability framework draws upon the Planetary Boundaries methodology to assess impacts on broad-scale vulnerabilities, it encounters several limitations inherent in this approach (see Biermann & Kim, 2020). For instance, it has been highlighted that the **Planetary Boundaries methodology simplifies the complex interactions between Earth's systems** by isolating each of the different boundaries and disregarding that changes in one boundary may unexpectedly trigger abrupt changes in another. Also, that the definition of the exact thresholds beyond which irreversible environmental changes may occur entails high levels of uncertainty. In addition, the PB framework does not account for equity aspects involved in the fact that impacts of breaching planetary boundaries are not evenly distributed, with marginalized communities often bearing the brunt of environmental degradation.

Furthermore, the calculation of the indicators employed in this dissertation, although effective, usually has inherent limitations tied to **theoretical assumptions and modelling capabilities**. A good example can be found in the premise employed for the indicator *Indoor heatwave day temperature* in the green roof case study. Here, due to the lack of modelling capacity for accurately assessing this indicator, the inner temperature for each building available for the creation of green roofs was matched to the *Heatwave outdoor day temperatures* as a simplifying assumption. While such an approach has been used in the past (e.g., Marvuglia et al., 2020), it has been recognized that, depending on the urban morphologies, indoor temperatures can vary in comparison to outdoor temperatures (Franck et al., 2013).

Finally, it is worth noticing that the proposed framework relies on urban agendas for selecting the relevant vulnerabilities to assess. While this is useful for developing a tailormade NBS planning adapted to local necessities, it also involves the risk of **disregarding relevant vulnerabilities that are not being considered by local authorities**. Such an approach could entail recognitional injustices if urban agendas fail to contemplate the needs, identities, and everyday lives of marginalized groups (Anguelovski et al., 2020).

V.6. Recommendations for future research

The application of the framework proposed in this thesis dissertation could be **further tested** to validate its capacities and versatility. For instance, the NBS-vulnerability framework was applied in two case studies of similar urban scales (i.e., city level). Given that NBS implementation can occur at various local spatial scales—like local, neighborhood, city, and regional (Hutchins et al., 2021)—testing the framework across wider and smaller urban scales is encouraged.

Additionally, the application of the NBS-vulnerability framework has primarily occurred in data-rich contexts of the Global North. Future research should explore its **potential applicability in the Global South**, considering differences in urban morphologies and data availabilities in this region (Barron et al., 2017). This expansion not only broadens the capabilities of the framework but also taps into research and NBS development opportunities in the Global South, an area with comparatively less research conducted about NBS development (Kuller et al., 2022) and regarded as promising for adopting alternative infrastructures and technologies due to the ongoing development of urban infrastructures (Barron et al., 2017).

Furthermore, while environmental justice aspects were considered primarily from a distributional perspective (i.e., understanding the spatial distribution of environmental risks, amenities, and social disadvantages), **procedural and recognitional justice aspects were less emphasized**. These aspects, which focus on diverse social and cultural values and equitable engagement spaces, are vital for ensuring environmentally just cities and effective NBS (Langemeyer & Connolly, 2020). To effectively integrate these considerations into the framework, it is recommended to expand participatory approaches throughout the entire vulnerability process (Madruga De Brito et al., 2018). For example, by including a diverse set of stakeholders with different backgrounds and values in the selection of the vulnerabilities, rather than solely relying on urban agendas.

Finally, and in the case of broad-scale vulnerabilities, the use of **thresholds for the contextualization of Planetary boundaries could be further developed** in order to have a clearer view of the impacts of NBS. As discussed in Chapter IV, the contextualized Planetary boundaries provide a value that, even when relatable to the observed urban environment, does not consider how other urban activities are affecting it, or which portion of the downscaled boundary should be allocated to the NBS under study. Such a condition challenges the full understanding of NBS on broad-scale vulnerabilities. To enhance this, it is encouraged to further refine the thresholds used to assess NBS impacts. To do so, one approach is to engage in collaborative stakeholder participation to co-create specific thresholds that allow to determine how much of the downscaled Planetary boundaries should be allocated to the NBS under study.

V.7. Final thoughts

This dissertation started with the intention of answering the question of *how to better understand the impacts of Nature-based solutions implemented in urban environments, to enhance their benefits and reduce their unintended consequences?* Through my work, I was able to answer this question and propose a way forward to better assess the heterogenous ways in which urban NBS behave, and how these can shape the urban and non-urban spaces in both desirable and undesirable ways.

While doing this, I encountered that recognizing, anticipating and calculating NBS impacts entails the identification of interconnected and sometimes unpredictable behaviors, related to the inherently complex urban environments in which NBS are being implemented. This, in itself, has provided me with a challenging, yet rewarding experience of understanding NBS as both a single object of study and an integral part of larger urban systems. In these environments, NBS act as both initiators of change and responders to the system's dynamics. While navigating these, I could not help but recall the words from Alfred Korzybski, "A map is not the territory", to explain the challenge of conceiving a method able to fully represent the richness and complexity experienced in urban environments.

The theoretical, methodological and empirical results from this dissertation provide advances for the effective planning of urban NBS, by integrating a wide range of NBS assessments within the realm of vulnerability sphere, aiding in reducing the uncertainties around the unexpected and undesired NBS impacts. Yet, there are potential and exciting areas worth exploring for future research. These include the application of the presented framework at different spatial scales and in different locations outside of the Global North, the development of practices that further allow to comply with recognitional justices and the enhancement of the analysis of urban NBS impacts beyond urban limits.

In conclusion, this dissertation has been able to push the limits of the current assessment of NBS in complex urban environments. Within this line of research, there are still efforts to be carried out in order to keep advancing the understanding of urban NBS. Nevertheless, the findings of this work, especially the NBS-vulnerability framework, already could aid in the planning and implementation of urban NBS to foster more sustainable, equitable and resilient urban and global environments.

V. References

Anguelovski, I., Brand, A. L., Connolly, J. J. T., Corbera, E., Kotsila, P., Steil, J., Garcia-Lamarca, M., Triguero-Mas, M., Cole, H., Baró, F., Langemeyer, J., del Pulgar, C. P., Shokry, G., Sekulova, F., & Argüelles Ramos, L. (2020). Expanding the Boundaries of Justice in Urban Greening Scholarship: Toward an Emancipatory, Antisubordination, Intersectional, and Relational Approach. Annals of the American Association of Geographers*, 110(6), 1743–1769. https://doi.org/10.1080/24694452.2020.1740579*

Barron, N. J., Kuller, M., Yasmin, T., Castonguay, A. C., Copa, V., Duncan-Horner, E., Gimelli, F. M., Jamali, B., Nielsen, J. S., Ng, K., Novalia, W., Shen, P. F., Conn, R. J., Brown, R. R., & Deletic, A. (2017). Towards water sensitive cities in Asia: an interdisciplinary journey. *Water Science and Technology*, *76*(5), 1150–1157. https://doi.org/10.2166/WST.2017.287

Berbés-Blázquez, M., Cook, E. M., Grimm, N. B., Iwaniec, D. M., Mannetti, L. M., Muñoz-Erickson, T. A., & Wahl, D. (2023). Assessing resilience, equity, and sustainability of future visions across two urban scales. *Sustainability Science*, *18*(6), 2549–2566. https://doi.org/10.1007/S11625-023-01396-Z/FIGURES/9

Biermann, F., & Kim, R. E. (2020). The boundaries of the planetary boundary framework: A critical appraisal of approaches to define a "safe operating space" for humanity. *Annual Review of Environment and Resources*, *45*(Volume 45, 2020), 497–521. https://doi.org/10.1146/ANNUREV-ENVIRON-012320-080337/CITE/REFWORKS

Boggia, A., Massei, G., Pace, E., Rocchi, L., Paolotti, L., & Attard, M. (2018). Spatial multicriteria analysis for sustainability assessment: A new model for decision making. *Land Use Policy*, *71*, 281–292. https://doi.org/10.1016/J.LANDUSEPOL.2017.11.036

Bush, J., & Doyon, A. (2019). Building urban resilience with nature-based solutions: How can urban planning contribute? *Cities*, *95*, 102483. https://doi.org/10.1016/J.CITIES.2019.102483

Chen, Y., Yu, J., & Khan, S. (2010). Spatial sensitivity analysis of multi-criteria weights in GIS-based land suitability evaluation. Environmental Modelling & Software, 25(12), 1582–1591. https://doi.org/10.1016/J.ENVSOFT.2010.06.001

Colléony, A., & Shwartz, A. (2019). Beyond Assuming Co-Benefits in Nature-Based Solutions: A Human-Centered Approach to Optimize Social and Ecological Outcomes for Advancing Sustainable Urban Planning. *Sustainability 2019, Vol. 11, Page 4924*, *11*(18), 4924. https://doi.org/10.3390/SU11184924

Dumitru, A., Frantzeskaki, N., & Collier, M. (2020). Identifying principles for the design of robust impact evaluation frameworks for nature-based solutions in cities. *Environmental Science & Policy*, *112*, 107–116. https://doi.org/10.1016/J.ENVSCI.2020.05.024

Elmqvist, T., Andersson, E., Frantzeskaki, N., McPhearson, T., Olsson, P., Gaffney, O., Takeuchi, K., & Folke, C. (2019). Sustainability and resilience for transformation in the urban century. *Nature Sustainability*, *2*(4), 267–273. https://doi.org/10.1038/s41893-019-0250-1

European Commission. (2019). *The European green deal*.

Franck, U., Krüger, M., Schwarz, N., Grossmann, K., Röder, S., & Schlink, U. (2013). Heat stress in urban areas. Indoor and outdoor temperatures in different urban structure types and subjectively reported well-being during a heat wave in the city of Leipzig. *Meteorologische Zeitschrift (Berlin)*, *22*(2), 167–177. https://doi.org/10.1127/0941-2948/2013/0384

Gómez-Baggethun, E., & Barton, D. N. (2013). Classifying and valuing ecosystem services for urban planning. *Ecological Economics*, *86*, 235–245. https://doi.org/10.1016/J.ECOLECON.2012.08.019

Gonzalez, A., & Enríquez-De-Salamanca, Á. (2018). Spatial Multi-Criteria Analysis in Environmental Assessment: A Review and Reflection on Benefits and Limitations. Journal of Environmental Assessment Policy and Management 2018 20:03 https://doi.org/10.1142/S146433321840001X

Guan, W., Liu, Q., & Dong, C. (2022). Risk assessment method for industrial accident consequences and human vulnerability in urban areas. *Journal of Loss Prevention in the Process Industries*, *76*, 104745. https://doi.org/10.1016/J.JLP.2022.104745

Herreros-Cantis, P., & McPhearson, T. (2021). Mapping supply of and demand for ecosystem services to assess environmental justice in New York City. *Ecological Applications*, *31*(6), e02390. https://doi.org/10.1002/EAP.2390

Hutchins, M. G., Fletcher, D., Hagen-Zanker, A., Jia, H., Jones, L., Li, H., Loiselle, S., Miller, J., Reis, S., Seifert-D hnn, I., Wilde, V., Xu, C. Y., Yang, D., Yu, J., & Yu, S. (2021). Why scale is vital to plan optimal Nature-Based Solutions for resilient cities. *Environmental Research Letters*, *16*(4), 044008. https://doi.org/10.1088/1748-9326/ABD9F4

Kueppers, L. M., Snyder, M. A., & Sloan, L. C. (2007). Irrigation cooling effect: Regional climate forcing by land-use change. *Geophysical Research Letters*, *34*(3). https://doi.org/10.1029/2006GL028679

Kuller, M., Farrelly, M., Marthanty, D. R., Deletic, A., & Bach, P. M. (2022). Planning support systems for strategic implementation of nature-based solutions in the global south: Current role and future potential in Indonesia. *Cities*, *126*, 103693. https://doi.org/10.1016/J.CITIES.2022.103693

Langemeyer, J., & Connolly, J. J. T. (2020). Weaving notions of justice into urban ecosystem services research and practice. *Environmental Science and Policy*, *109*(April), 1–14. https://doi.org/10.1016/j.envsci.2020.03.021

Langemeyer, J., Madrid-Lopez, C., Mendoza Beltran, A., & Villalba Mendez, G. (2021). Urban agriculture — A necessary pathway towards urban resilience and global sustainability? *Landscape and Urban Planning*, *210*, 104055. https://doi.org/10.1016/J.LANDURBPLAN.2021.104055

Madruga De Brito, M., Evers, M., Delos, A., & Almoradie, S. (2018). Participatory flood vulnerability assessment: a multi-criteria approach. *Hydrol. Earth Syst. Sci*, *22*, 373–390. https://doi.org/10.5194/hess-22-373-2018

Marvuglia, A., Koppelaar, R., & Rugani, B. (2020). The effect of green roofs on the reduction of mortality due to heatwaves: Results from the application of a spatial microsimulation model to four European cities. *Ecological Modelling*, *438*, 109351. https://doi.org/10.1016/J.ECOLMODEL.2020.109351

McPhearson, T., Haase, D., Kabisch, N., & Gren, Å. (2016). Advancing understanding of the complex nature of urban systems. *Ecological Indicators*, *70*, 566–573. https://doi.org/10.1016/J.ECOLIND.2016.03.054

Meerow, S., Pajouhesh, P., & Miller, T. R. (2019). Social equity in urban resilience planning. *Local Environment*, *24*(9), 793–808. https://doi.org/10.1080/13549839.2019.1645103

Melanidis, M. S., & Hagerman, S. (2022). Competing narratives of nature-based solutions: Leveraging the power of nature or dangerous distraction? *Environmental Science & Policy*, *132*, 273–281. https://doi.org/10.1016/J.ENVSCI.2022.02.028

Nesshöver, C., Assmuth, T., Irvine, K. N., Rusch, G. M., Waylen, K. A., Delbaere, B., Haase, D., Jones-Walters, L., Keune, H., Kovacs, E., Krauze, K., Külvik, M., Rey, F., van Dijk, J., Vistad, O. I., Wilkinson, M. E., & Wittmer, H. (2017). The science, policy and practice of nature-based solutions: An interdisciplinary perspective. In *Science of the Total Environment* (Vol. 579, pp. 1215–1227). Elsevier B.V. https://doi.org/10.1016/j.scitotenv.2016.11.106

Pan, H., Page, J., Cong, C., Barthel, S., & Kalantari, Z. (2021). How ecosystems services drive urban growth: Integrating nature-based solutions. *Anthropocene*, *35*, 100297. https://doi.org/10.1016/J.ANCENE.2021.100297

Pereira, P., Wang, F., Inacio, M., Kalinauskas, M., Bogdzevič, K., Bogunovic, I., Zhao, W., & Barcelo, D. (2024). Nature-based solutions for carbon sequestration in urban environments. *Current Opinion in Environmental Science & Health*, *37*, 100536. https://doi.org/10.1016/J.COESH.2024.100536

Raparthi, K., & Vedamuthu, R. (2022). Assessing the Role of Nature-Based Solutions in Urban Resilience and Climate Change Adaptation. *Lecture Notes in Civil Engineering*, *191*, 13–21. https://doi.org/10.1007/978-981-16-5839-6_2

Rödl, A., & Arlati, A. (2022). A general procedure to identify indicators for evaluation and monitoring of nature-based solution projects. *Ambio*, *51*(11), 2278–2293. https://doi.org/10.1007/S13280-022-01740-0/TABLES/3

Sera, F., Armstrong, B., Tobias, A., Vicedo-Cabrera, A. M., Åström, C., Bell, M. L., Chen, B. Y., De Sousa Zanotti Stagliorio Coelho, M., Correa, P. M., Cruz, J. C., Dang, T. N., Hurtado-Diaz, M., Do Van, D., Forsberg, B., Guo, Y. L., Guo, Y., Hashizume, M., Honda, Y., Iñiguez, C., … Gasparrini, A. (2019). How urban characteristics affect vulnerability to heat and cold: a multi-country analysis. *International Journal of Epidemiology*, *48*(4), 1101–1112. https://doi.org/10.1093/IJE/DYZ008

Seto, K. C., Reenberg, A., Boone, C. G., Fragkias, M., Haase, D., Langanke, T., Marcotullio, P., Munroe, D. K., Olah, B., & Simon, D. (2012). Urban land teleconnections and sustainability. *Proceedings of the National Academy of Sciences of the United States of America*, *109*(20), 7687–7692. https://doi.org/10.1073/pnas.1117622109

Shukla, N., Das, A., & Mazumder, T. (2023). Assessment of urban form resilience: a review of literature in the context of the Global South. *Environment, Development and Sustainability*, 1–37. https://doi.org/10.1007/S10668-023-04058-3/FIGURES/7

United Nations. (2015). *Goal 11: Make cities inclusive, safe, resilient and sustainable*. https://www.un.org/sustainabledevelopment/cities/

APPENDICES

Appendix 1. Supplementary data for Chapter III

1-A. Supplementary scheme

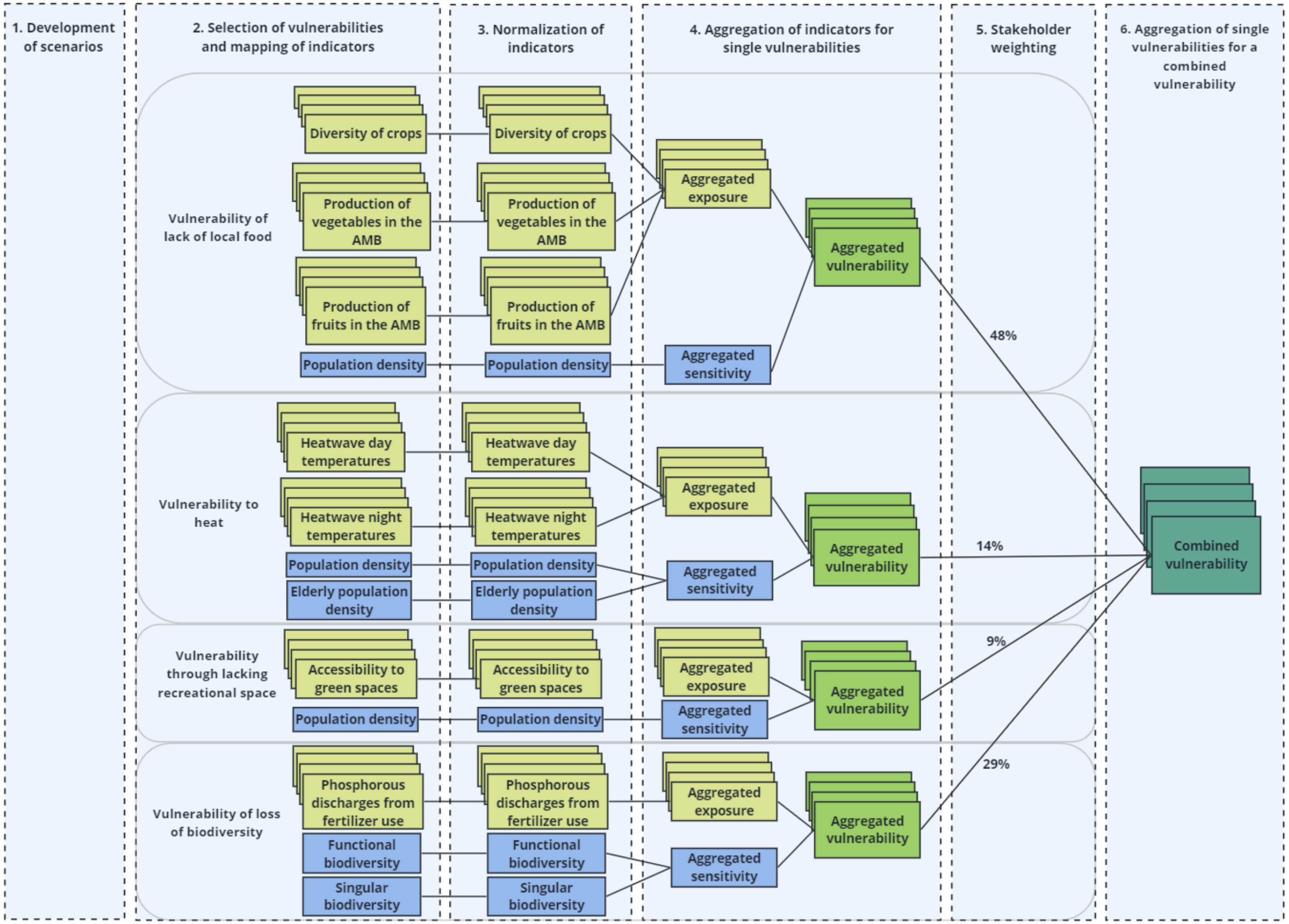


Figure A.1. Stepwise approach of NBS-vulnerability framework applied to the (peri)urban agriculture in the Metropolitan Area of Barcelona. Each box represents a single map for either indicators, normalized indicators, aggregated indicators, single vulnerabilities or combined vulnerability. Yellow boxes represent exposure indicators calculated for each scenario, while blue ones represent sensitivity calculated only for the reference scenario.

Figure A.2. Stakeholder workshop event: introduction

Figure A.3. Stakeholder workshop event: weighting exercise (group 1)

Figure A.3. Stakeholder workshop event: weighting exercise (group 2)

1-B. Methodological details

1. Vulnerability of lack of local food

1.1. Indicator Diversity of crops (exposure)

The Shannon index is a commonly used tool to evaluate the diversity of a landscape, which is determined by the quantity and proportion of different categories of land cover (Basnou et al., 2020). The formula for calculating the Shannon index is (Eq. B.1):

$$H = -\Sigma (Pi * ln (Pi)) \qquad \text{(Eq. B.1)}$$

where H is the Shannon index, Pi the proportion of the entire community made up of species i, and ln is the natural logarithm. The resulting value indicates the diversity of the crops in the field: the higher more diverse the crops are in terms of both the number of different species and their relative abundances. This can be employed to assess the overall health and productivity of the agricultural system. Crop diversity can describe vulnerabilities associated to food production. For example, high diversity can reduce the risk of crop failure due to pests, diseases, or adverse weather conditions and provide conditions such improved pollination, pest control, and soil conservation.

There is no threshold for this indicator, since the Shannon index is a relative measure of diversity within a sample. The context and scale of the study can influence the interpretation of diversity. Different regions, habitats, or agricultural systems may naturally have varying levels of diversity. Min max normalization was carried out using min-max indicator's sample values.

1.2. Indicator Production of vegetables/fruits in the AMB (exposure)

We included the food production of vegetables and fruits in the AMB as it represents a core element of the local food availability of the city.

The calculation of both indicators started by first estimating the food production for each scenario. First, geographically explicit crops were calculated by Mendoza Beltran et al. (2022) which had an associated productive yield per type of crop (kg/ha). Then, crops were geographically divided by a 50x50m grid, this way obtaining the amount of ha located within each pixel. Afterwards, each of these crops were divided by each product type, either being vegetables or fruits (see **Table B.1**). Then, total production of vegetables/fruits was calculated for each of the pixels, by multiplying the yield of fruits or vegetables by their respective extensions within the pixels, obtaining total *Production of vegetables/fruits* by pixel.

Parallel to this, food demand for the AMB was calculated. For this, I used the yearly demand estimation of fruits and vegetables per person for Catalonia based on Spanish yearly consumption of fruits and vegetables (Ministerio de Agricultura, 2022) and multiplied it by the total amount of residents of the AMB (INE, 2021) (see **Table B.1**). This calculation comes at hand for the indicator's thresholds. According to Inèdit (2022), by 2030 a third of the population of Barcelona should consume a majority of proximity products. This objective sets the reference for the threshold, which is achieving Barcelona's objective: the closer a scenario is to fulfilling this objective, the less exposed it is. In order to assign a specific value to the threshold, I first calculated the yearly fruits and vegetable demand needed for a third of the AMB (e.g., 1,066,156 hab. * Kg of fruits demanded per year per person) and then multiplied it by 51%, which represents the majority of the product demand. This final value provides the UA production needed to achieve threshold, that was named *Production Target: vegetables* and *Production Target: fruits* (see **Table B.1**).

In order to normalize these indicators, two different premises were considered: (1) the higher the food production within a pixel, the smaller the exposure (and vice versa), and (2) the closest the overall production of each scenario is to the threshold, the less exposed it is. For the first premise I normalized *Production of vegetables/fruits* of each pixel using min-max method, where single minimum and maximum value came from the pixels with the lowest and highest *Production of vegetables/fruits* values among of all scenarios. Values were normalized from 0-1, where 0 represents no exposure (high vegetable/fruits production) and 1 high exposure (no vegetable/fruits production).

For the second premise, first I calculated the *Production of vegetables/fruits missing to achieve threshold (tons)* (see **Table B.2**), an absolute value that portrays how many tons is each scenario missing for achieving the *Production Target: vegetables* and *Production Target: fruits* defined by the threshold. Then, I divided this value by the *Production target: vegetables/fruits (tons)*, to translate this absolute value into a proportion. I called this value *Production of vegetables/fruits missing to achieve threshold (% of threshold)*, which accounts for the second premise as it represents how close is each of the scenarios to the threshold (the lower the percentage, the less exposed the scenario is) (see **Fig. B.2**). Lastly, and in order to merge the two premises and obtain a final value, I multiplied the normalized *Production of vegetables/fruits* by the *Production of vegetables/fruits missing to achieve threshold (% of threshold)*. By doing this it was obtained a contextualized value for each of the pixels, in which their exposure is not only defined by the amount of vegetable/fruit production, but also by the overall production in the whole scenario in function to the defined threshold (see **Table B.3** for an illustrative example).

Table B.1. Per capita demand of fruits and vegetables for the Catalan region, Total food demand and threshold values

Vegetables	Kg per year per person	Fruits	Kg per year per person
POTATOES	32.72	AVOCADO	0.79
GARLIC	0.8	APRICOTS	0.95
ARTICHOKES	0	CHERRIES	1.36
CELERY	0	CUSTARD APPLE	0
AUBERGINES	2.26	PLUMS	1.36
BROCCOLI	0	STRAWBERRIES/STRAWBERRY	2.58
ZUCHINNI	4.29	KIWI	3.04
ONIONS	8.47	LEMONS	2.53
MUSHROOMS	1.36	TANGERINES	8.2
CABBAGE	1.91	MANGO	0
CAULIFLOWER	0	APPLES	12.52
ASPARAGUS	1.31	PEACHES	5.5
GREEN BEANS	3.21	MELON	8.99
LETTUCE/ENDIVIA	5.82	ORANGES	21.97
OTHER.VEGETABLES/GREEN.	13.41	NECTARINES	2.18
CUCUMBERS	2.56	OTHER FRESH FRUITS	5.28
PEPPERS	4.94	PEARS	5.84
LEEK	0	PINEAPPLE	2.04
TOMATOES	17.44	BANANAS	12.18
LEAFY VEGETABLES	1.54	GRAPEFRUIT	0
CARROTS	3.44	WATERMELON	8.66
Total	**105.48**	GRAPES	2.7
		Total	**108.67**

Vegetable demand of the AMB (Ton per year)	**337,342.42**	**Fruit demand of the AMB (Ton per year)**	**347,577.52**
Threshold	**Barcelona objective for 2030: a third of the population should consume a majority of proximity products**		
Production target: vegetables (Ton per year)	**57,348.21**	**Production target: fruits (Ton per year)**	**59,088.18**

Table B.2. Production of vegetables & fruits by scenario and production missing for achieving threshold

Threshold	Barcelona objective for 2030: a third of the population should consume a majority of proximity products		
Production target: vegetables (Ton per year)	57,348.21	Production target: fruits (Ton per year)	59,088.18

	S0	S1	S2	S3
Production of Vegetables (Ton per year)	39,148.37	34,369.43	49,013.85	64,983.83
Production of vegetables missing to achieve target (tons)	18,199.85	22,978.78	8,334.36	-
Production of vegetables missing to achieve target (% of threshold)	32%	40%	15%	0%

	S0	S1	S2	S3
Production of fruits (Ton per year)	9,283.88	7,766.93	12,137.51	23,104.19
Production of fruits missing to achieve target (tons)	49,804.30	51,321.24	46,950.66	35,983.99
Production of fruits missing to achieve target (% of threshold)	84%	87%	79%	61%

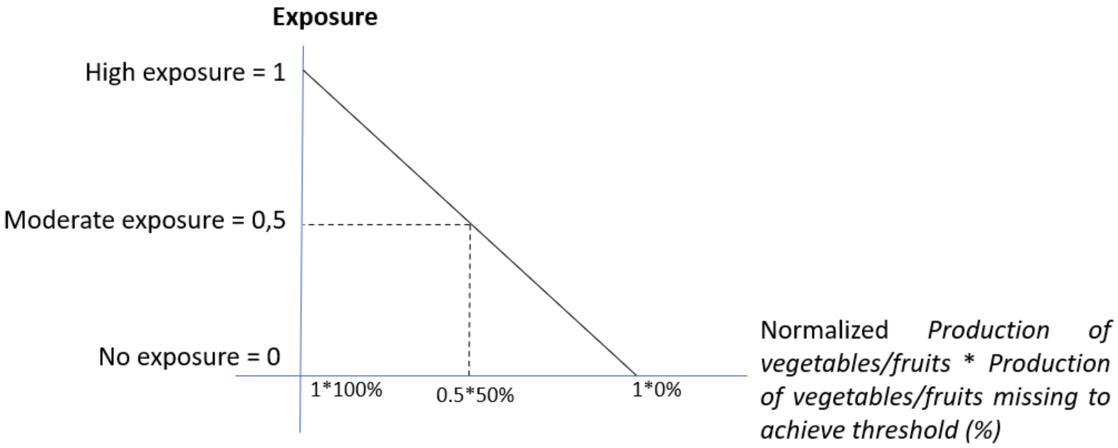


Figure B.1. Normalized exposure values from Production of vegetables/fruits in the AMB represented in a cartesian coordinate system. Horizontal axis portrays the compound normalization vegetables/fruits production and the percentage of production of vegetables/fruits to achieve threshold

Table B.3. Example of normalization process of indicator Production of vegetables/fruits in the AMB

1) Min-max normalizated values

Absolute values

Pixel Id	S0 (Kg of fruit per year)	S1 (Kg of fruit per year)	S2 (Kg of fruit per year)	S3 (Kg of fruit per year)
A	359.8	354.5	354.5	913.4
B	-	-	-	-
C	76.9	-	76.8	76.8
D	3,374.3	3,381.1	3,381.1	3,381.1
...Z				

Min (Kg of fruit per year)	-
Max (Kg of fruit per year)	3,381.07

Min-max normalized values

Pixel Id	S0	S1	S2	S3
A	0.9	0.9	0.9	0.7
B	1.0	1.0	1.0	1.0
C	1.0	1.0	1.0	1.0
D	0.0	-	-	-
...Z				

2) Production of vegetables missing to achieve threshold

	S0	S1	S2	S3
Production of fruits needed to achieve Threshold (Ton per year)	59,088.2	59,088.2	59,088.2	59,088.2
Production of fruits (Ton per year)	9,283.9	7,766.9	12,137.5	23,104.2
Production of fruits missing to achieve threshold (tons)	49,804.30	51,321.24	46,950.66	35,983.99
Production of fruits missing to achieve threshold (% of threshold)	84%	87%	79%	61%

3) Min-max normalized values * P treshold usage

Pixel Id	S0	S1	S2	S3
A	0.75	0.78	0.71	0.45
B	0.84	0.87	0.79	0.61
C	0.82	0.87	0.77	0.60
D	0.00	-	-	-
...Z				

1.3. Indicator: Population density (sensitivity)

Included as a way to track areas that are demanding higher quantities of food, and therefore, can be more sensible to changes in food supply. Moreover, it is also relevant in order to observe how food sources are distributed and if they coincide with areas with high food demand. In areas of high population density, urban agriculture spaces may be more important for providing fresh, healthy food to residents who may not have easy access to traditional food sources. In order to calculate population density, a map of total population by census tracts was obtained for the AMB (INE, 2021). The total population was then divided by the km^2 of census tracts, in order to obtain population per Km^2. Finally, the result was converted into a grid with a resolution of 50x50m. Each grid obtained its value based on the census tract with the biggest geographical overlap.

As a sensitivity indicator it does not count with a threshold value. Min max normalization was carried out using min-max indicator's sample values.

2. Vulnerability to heat

2.1. Indicator: heatwave day temperatures (exposure)

Understanding people's vulnerability to heat is closely related to daytime temperatures, as they are exposed to higher temperatures during the day, primarily due to increased levels of activity (Basagaña et al., 2011; Wang et al., 2012; Xu et al., 2013). During these periods, the risks of heatstroke, heat exhaustion and cardiovascular problems are increased. To address this matter, the average monthly temperatures (°C) during midday (13h-16h) were examined for the 2015 heatwave in the Metropolitan Area of Barcelona. These temperatures were obtained from 4 meteorological simulations conducted with the Weather Research and Forecasting model (WRF v4.3.3; Skamarock et al., 2021). The simulations encompass a period of extreme high temperatures and dry weather over the AMB, from 00 UTC on 20 June 2015 to 00 UTC on 25 July 2015 (see Segura et al. (2021) for a more detailed description of the event), with the first five days used for model spin-up. The WRF configuration consists of 4 two-way nested domains, where the innermost have a horizontal resolution of 333 m and covers entirely the AMB. The physical configuration of the model is similar to the described in Segura et al, 2021. Each WRF simulation used scenario-specific input data, such as land-use maps, urban fractions, building and street morphologies, and irrigation maps. Results were then mapped through GIS and converted into a grid of 50x50m in order to match sensitivity indicators. GIS operations were carried out using QGIS (version 3.28.0-Firenze).

The selected threshold for this indicator is a minimum value of 32 ºC, which represents the temperature from which exposure begins to increase (all areas with midday temperatures below 32ºC are not considered as exposed). Threshold value was defined based on Díaz et al. (2015) research, which defines the *maximum daily temperature above which a significant increase in heat-related mortality was observed Barcelona city.*

In order to relate the average monthly temperatures (°C) to the threshold value, a normalization process was carried out, employing the min max normalization process (see

Section 5, Eq. B.2). For doing this, the min value used was 32 ºC (threshold) and max value 39.37 °C, which is the highest temperature observed from the indicator's sample among all UA scenarios (see **Fig. B.2**). By doing this, the average monthly temperatures below 32 ºC is deemed insignificant, representing no exposure to heat, and therefore displaying a normalized value of 0 (see **Fig. B.2**). On the contrary, the average monthly temperatures exceeding 39.37 ºC represent high exposure and display a normalized value of 1.

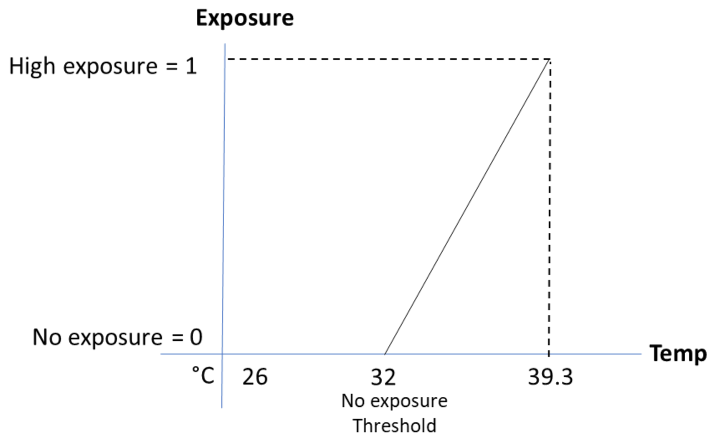


Figure B.2. Exposure values of Heatwave day temperatures represented in a cartesian coordinate system. Horizontal axis portrays °C absolute values, as well as the selected threshold. Vertical axis represents the level of exposure and the normalized values associated to the °C absolutes.

2.2. Indicator: heatwave night temperatures (exposure)

The urban heat island effect causes high temperatures at night during heatwave periods, which makes it difficult for people to cool down from the daytime heat (Smith et al., 2011). Such conditions can lead to health problems, particularly among vulnerable groups such as the elderly (Heaviside et al., 2016). In order to include these vulnerabilities in the assessments, I examined the average monthly temperatures (°C) during nighttime (21h-7h) for the 2015 heatwave in the Metropolitan Area of Barcelona. Similar to *Heatwave day temperatures*, temperatures were obtained from simulations for 00 UTC on 20 June 2015 to 00 UTC on 25 July 2015, the first five days discarded for spin-up. The temperatures were mapped following the same methods employed for *Heatwave day temperatures.*

The selected threshold for this indicator is a minimum value of 23ºC, which represents the temperature from which exposure begins to increase (all areas with night temperatures below 23ºC are not considered as exposed). The threshold value was defined based on (Royé, 2017) research, which defines as 23 ºC the night temperature where there is *an increased risk in mortality due to natural, respiratory, and cardiovascular causes in the Barcelona region.*

In order to relate the average monthly temperatures (°C) to the threshold value, a normalization process was carried out, employing the min max normalization process (see Section 5, Eq. B.2). For doing this, the min value used was 23 ºC (threshold) and max value 28.5 °C, which is the highest temperature observed from the indicator's sample among all UA scenarios (see Fig. **B.3**). By doing this, the average monthly temperatures below 23 ºC is deemed insignificant, representing no exposure to heat, and therefore displaying a normalized value of 0 (see **Fig. B.3**). On the contrary, the average monthly temperatures exceeding 28.5 ºC represent high exposure and display a normalized value of 1.

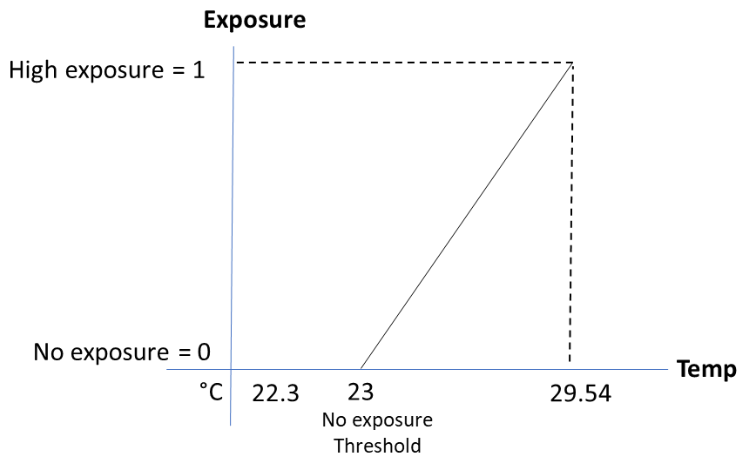


Figure B.3. Exposure values of Heatwave night temperatures represented in a cartesian coordinate system. Horizontal axis portrays °C absolute values from the indicator calculation, as well as the selected threshold. Vertical axis represents the level of exposure and the normalized values associated to the °C absolutes.

2.3. Indicator: Population density (sensitivity)

Generally speaking, residents in areas of high population density are at an increased risk of hazards due to the greater concentration of people in small areas (Guan et al., 2022; Sera et al., 2019). Population density maps were calculated following the same procedure described in section 1.3.

2.4. Indicator: Elderly population density (sensitivity)

Research indicates that elderly individuals are more prone to experiencing negative health impacts as a result of heat waves due to a combination of social and medical factors (i.e., reduced ability to regulate body temperature, mental disorders which impacts an individual's ability to recognize and respond to heat-related risks and living alone or having increased dependency (Hajat et al., 2010; Oudin Åström et al., 2011; Vandentorren et al., 2006). Elderly population density is then a proxy variable to include this sensitivity. The values were calculated focusing only on residents over 65 years old (INE, 2021) and following the same procedure as the one described for Population density.

As a sensitivity indicator it does not count with a threshold value. Min max normalization was carried out using min-max indicator's sample values.

3. Vulnerability of lacking recreational space

3.1. Indicator: accessibility to green spaces at less than 300m, less than 1000m and more than 1000m (sensitivity)

Recreational opportunities should consider the barriers that may prevent them from accessing and utilizing green spaces (Neuvonen et al., 2007). The availability of nearby recreational areas facilitates frequent participation in outdoor recreation, which in turn increases the likelihood of achieving health benefits (Humpel et al., 2002). Based on this, it was included the accessibility to green spaces as an exposure indicator, referring to the degree to which green spaces can be accessed by people and communities. For this assessment, accessibility is calculated employing GIS by mapping green areas and built-up areas polygons in the AMB using (CREAF, 2015). Green areas include forests, scrubs, agricultural lands, urban parks, meadows, grasslands, rivers, wetlands and beaches. Built-up areas include urbanized areas and communication roads. Next, green areas were intersected with pedestrian and biking roads to track the entry points to the facilities. Each of these coordinates serves as a starting point for calculating service areas based on pedestrian and cycling roads into built-up areas. Before continuing, both green areas and built-up areas were converted into a grid of 50x50m, which maintained the differentiation between the green and built-up areas. Continuing, a 5m buffer was created around each one of the service areas, which was then used to intersect it with the built-up pixels, this way obtaining 4 possible results for pixels: (1) a green space, (2) a build-up area with access to green spaces at 300m or less, (3) a build-up area with access to green spaces at 1000m or less and (4) a build-up area with no access to green spaces. This procedure carried on for each of the scenarios. Also, the variations between each scenario compared to the *S0* was calculated (e.g., *S1* vs *S0*) in order to observe the changes in accessibility. To do so, a series of values were assigned to each type of pixel: green spaces were represented by the value 8 (representing the highest level of accessibility), built-up areas with a 300m accessibility to green spaces had a value of 7, built-up areas with a 1000m accessibility to green spaces had a value of 6 and pixels with no accessibility to green spaces with value of 5. This allowed to subtract the values between pixels from each scenario to appreciate the change in accessibility, while being able to rank the transformation of pixels (e.g., pixels going from green spaces to built-up areas with no accessibility obtained a value of -3, representing the highest decline in accessibility).

The selected threshold for this indicator is both 300m and 1000m. In the case of 300m, it was selected as a minimum value threshold, which represents the point from which exposure begins to increase (all areas with up to 300m of accessibility to green spaces are not considered as exposed). The justification for selecting this value consist of the literature relating close access to green spaces (up to 300m) to reduced levels of stress (Stigsdotter et al., 2010; Vos et al., 2022) reduced risk of mortality associated with cardiovascular diseases (Bauwelinck et al., 2021) and blood pressure related issues (Grazuleviciene et al., 2014). In the case of the 1000m, it was selected as a maximum value threshold, which represents the point from which an area is considered as highly exposed (all areas located further away than 1000m from a green space have the highest exposure). This value was selected based on (1) that 1000m is the distance limit used within the accessibility to green areas and its relation to well-being (e.g., Bauwelinck et al., 2021; Grazuleviciene et al., 2014; Paquet et al., 2013; Reid et al., 2017) and (2) that residents living more than 1000m away from a green space experience lower amounts of well-being and higher morbidity (Maas et al., 2009; Stigsdotter et al., 2010).

As explained before, the calculation of the exposure indicator was made in order to assign one of the four defined categories related to accessibility to green spaces, meaning that each 50x50 pixel could either be: (1) a green space, (2) a build-up area with access to green spaces at 300m or less, (3) a build-up area with access to green spaces at 1000m or less and (4) a build-up area with no access to green spaces. The normalization process then was done by assigning specific values to each of the pixels based on its accessibility. Those pixels that had no access to green spaces had the highest exposure (normalized value of 1), those that were green spaces or a build-up area with access to green spaces at 300m or least exposure (normalized value of 0), and those build-up area with access to green spaces at 1000m or less were considered moderately exposed (normalized value of 0.5) (see **Fig. B.4**)

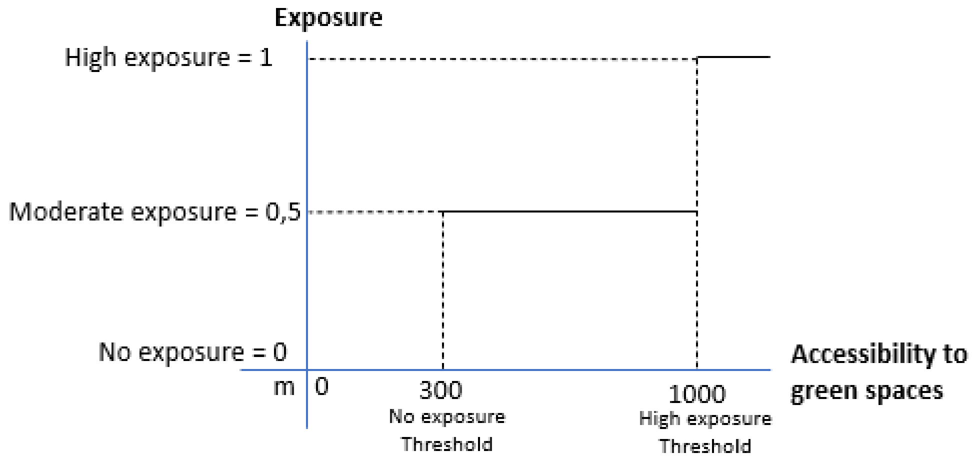


Figure B.4. Exposure values of accessibility to green spaces at less than 300m, less than 1000m and more than 1000m represented in a cartesian coordinate system. Horizontal axis portrays accessibility to green spaces in absolute values from the indicator calculation, as well as the selected threshold. Vertical axis represents the level of exposure.

3.2. Indicator: Population density (sensitivity)

Similar to *Vulnerability to heat*, areas with high population density pose an increased risk of hazards to their residents, as there is a greater number of people exposed to vulnerabilities. In the case of recreation, higher population densities demand higher amounts of green spaces, which could limit their capacity to cope with changes in its distribution (e.g., reduction of total amount of green areas in a highly dense neighborhood would put more people in risk of losing opportunities for recreational opportunities compared to a low-density neighborhood). Population density maps were calculated following the same procedure described in section 1.3.

4. Vulnerability of loss of biodiversity

4.1. Indicator: Phosphorous discharges from fertilizer use (exposure)

Fertilizers, particularly those containing nitrogen and phosphorus, can run off from urban agriculture sites and enter nearby waterways, contributing to eutrophication. Freshwater eutrophication can impact aquatic ecosystems by the depletion of oxygen in the water which can affect biodiversity in different ways: reduction of living organism in water, reduction of aquatic habitats, change in the species composition and reduction the food provision for animals. In order to assess for these impacts, phosphorous emissions (kg P eq per ha) from fertilizer are used as a proxy for potential eutrophication (Huang et al., 2017). Phosphorous emissions were derived for the regionalized life cycle assessment (LCA) for the AMB scenarios of UA. The full documentation of the assumptions and calculations made to estimate the emissions are described in Mendoza Beltran et al. (2022). Overall, emissions reflect conventional agricultural practices for nine different uses of land and include nutrient inputs from mineral fertilizers, animal manure and agricultural residues, all inventories estimated by means of the Socioecological Integrated Analysis (SIA) model (Marull et al., 2021; Padró et al., 2020). The kg P eq per ha calculated are spatially explicit for each of the AMB scenarios and crop type. Employing GIS, the crops mapped for each of the scenarios were intersected with the 50x50m grid. Afterwards, it was calculated the total Kg of *P discharge* in each of the pixels. The interpretation of this indicator assumes that the highest the *P discharge* per pixel, the greatest the exposure.

The selected threshold for this indicator is a maximum value that represents the highest exposure. The threshold is based on the European limits for selected planetary boundaries proposed by the European environmental agency (European Environmental Agency, 2020), specifically, on the median per capita loss of phosphorus from fertilizers and waste per year (0.11 kg P year). The limit is described as the amount of P beyond where there is a risk of potentially irreversible consequences associated to planetary boundaries. The per capita limit was then multiplied by the population of the AMB (INE, 2021), obtaining a threshold of 351,8 Tons P year (351,831.48 kg P year), which works as a maximum threshold from the AMB start being highly exposed. However, this threshold is not spatially explicit since it considers the whole AMB region as a single area. For normalizing the values calculated for the indicator some extra steps need to be carried on. First, two premises that need to be considered by the normalization process were defined: (1) the higher the P discharge in pixel, the higher the exposure, and (2) the closest the AMB (as a whole) is to the P limit (threshold) the higher the overall exposure of the scenario (i.e., UA scenarios with highest P discharges have higher exposure than those with lower P discharges), which will impact the single pixels. In order for these premises to be contemplated by final indicator value, first, I normalized the *P discharge* value by min-max method, using a single minimum and maximum value obtained from the pixels with the lowest and highest existing values among the sample of all scenarios. Values were normalized from 0-1, where 0 represents no exposure (no *P discharge*) and 1 high exposure (high *P discharge*). This first normalization accounts for the first premise. These normalized values were then multiplied by the *P threshold usage* of each scenario based on the scenario to which the pixel belongs. (see **Fig. B.5**). The *P threshold usage* comes from dividing the sum kg of P being discharged in each scenario by the indicator threshold of 351,8 Tons P year (351,831.48 kg P year). *P threshold usage* accounts for second premise, as it represents how close is each of the scenarios to the threshold (the closest it is, the highest the exposure). By multiplying the normalized *P discharge* value by the *P threshold usage,* it was obtained a contextualized value for each of the pixels, in which their exposure is not only defined by the amount of P discharges in the pixel location, but also by the overall P discharge of the whole region in function to the defined threshold (see **Table B.4** for an illustrative example).

Table B.4. Example of normalization process of indicator P discharges from fertilizer use

1) Min-max normalizated values

Absolute values

Pixel Id	S0 (Kg P per year)	S1 (Kg P per year)	S2 (Kg P per year)	S3 (Kg P per year)
A	-	-	-	-
B	-	-	0.0	0.0
C	10.7	5.2	10.7	10.7
D	13.0	3.0	13.0	13.8
...Z				

Min (Kg P per year)	-
Max (Kg P per year)	13.78

Min-max normalized values

Pixel Id	S0	S1	S2	S3
A	-	-	-	-
B	-	-	0.0	0.0
C	0.8	0.4	0.8	0.8
D	0.9	0.2	0.9	1.0
...Z				

2) P Threshold usage

	S0	S1	S2	S3
Sum (Kg P per year)	21,313.7	19,093.4	27,926.5	38,213.4
P threshold for AMB (Kg P per year)	351,831.48	351,831.48	351,831.48	351,831.48
P threshold usage	6%	5%	8%	11%

3) Min-max normalized values * P treshold usage

Pixel Id	S0	S1	S2	S3
A	-	-	-	-
B	-	-	0.00	0.00
C	0.05	0.02	0.06	0.08
D	0.06	0.01	0.07	0.11
...Z				

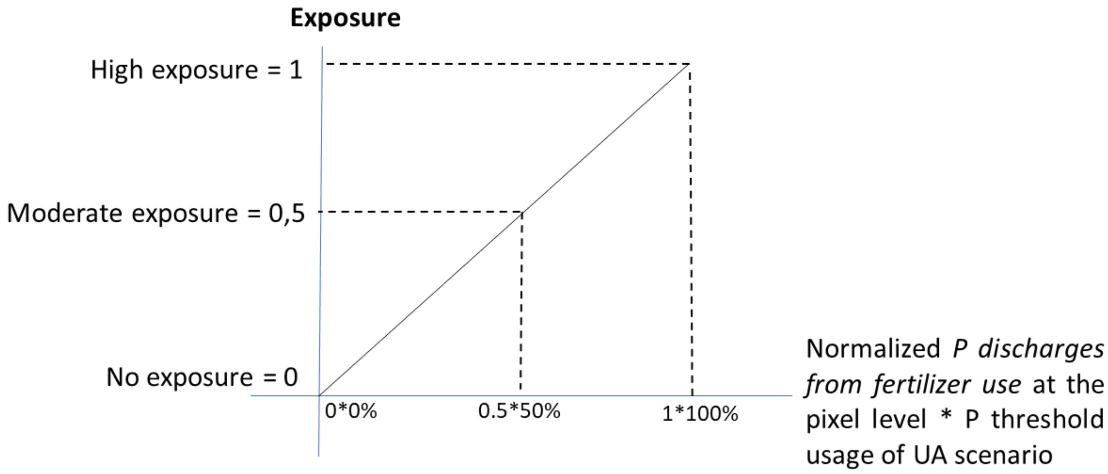


Figure B.5. Normalized exposure values from indicator Phosphorous (P) discharges from fertilizer use represented in a cartesian coordinate system. Horizontal axis portrays the compound normalization considering P discharges and P threshold for the whole Metropolitan Area of Barcelona.

4.2. Indicator: Functional diversity & singular biodiversity (sensitivity)

Biodiversity sensitivity is approached using two different but complementing indicators: singular biodiversity and functional diversity (Loreau et al., 2001). Singular focuses on a classical conservation perspective such as rareness, vulnerability and threat of species and habitats, as well as quality and threat of natural protected areas (Justus & Sarkar, 2002; Marull et al., 2007). Functional, focuses on ecosystems being supported by biodiversity (e.g., system stability, pollination and nutrient cycles) (Balvanera et al., 2006). Considering both indicators provide a more integrated vision on the vulnerability of biodiversity conditions, by looking not only the existence and diversity of species, but also the balance of ecological processes that allow biodiversity conditions to thrive. The full documentation of the assumptions and calculations made to estimate both indicators are described by Basnou et al. (2020). Overall, Singular biodiversity was quantified by combining distinct cartographic indicators that assess the conservation value of habitats and organisms. These indicators include areas with endangered flora, forests that are listed in the singular forests inventory of Catalonia, the intrinsic and chronological value of habitats, as well as various local conservation indices for birds, mammals, amphibians, and reptiles. Functional biodiversity was calculated by aggregating two variables: first, trough niche models for a set of species of birds and mammals, using specific field data in the province of Barcelona (six birds and seven mammal species, representing diverse ecological standards and distribution across the region), and second, by a species specialist index, which includes the majority of specialized species of plants, birds, mammals, amphibians, reptiles living in the province, and based on the premise of high-value functional ecosystems are dominated by specialist species (Basnou et al., 2020). Both indicators were converted into a grid with a resolution of 50x50m.

As sensitivity indicators, it does not count with a threshold values. Min max normalization was carried out using min-max indicator's sample values.

5. Normalization of indicators

Normalization formula employed in the third step of the NBS-vulnerability framework (Eq. B.2):

$$z_i = \frac{y(i) - \min(i)}{\max(i) - \min(i)} \qquad \text{(Eq. B.2)}$$

Where:
z: normalized value
i: selected indicator
y: absolute value
min: minimum absolute value or threshold
max: minimum absolute value or threshold

6. Aggregation of indicators for single vulnerabilities

Normalization formulas employed in the fourth step of the NBS-vulnerability framework.

6.1. Aggregation into a single exposure and a single sensitivity for each vulnerability

$$A_{e,s}(V) = \sum_{e,s\,|\mathbf{V}} i \times W_i \qquad \text{(Eq. B.3)}$$

Where
A: Aggregated indicators
e: exposure
s: sensitivity
V: selected vulnerability
i: selected indicator
W: associated weight to indicator

6.2. Aggregation of single exposure and single sensitivity for single vulnerabilities

$$V = A_{e|V} x A_{s|V} \qquad \text{(Eq. B.4)}$$

Where:
V: single vulnerability
A: Aggregated indicators
e: exposure
s: sensitivity

7. Stakeholder weighting

The workshop "Agricultural perspectives in the Metropolitan Area of Barcelona" was held on November 25th, 2022 at the premises of the Institute of Environmental Science and Technology in the Autonomous University of Barcelona (URBAG, 2022). It gathered 25 stakeholders to discuss the significance of each of the selected vulnerabilities which included academics, municipal officials, NGO representatives, and UA experts. Prior to the discussions and weighting exercise, attendees viewed a series of presentations on the current state of UA from different perspectives (e.g., decline in UA lands, state of food imports, farmers' conditions and metabolic impacts of UA). Followed by this, participants were divided into 4 groups with a mixed share of all backgrounds to rank relevant vulnerabilities that should be considered when planning for the future of UA in the AMB. Each group was given 100 pebbles (i.e., points), which needed to be distributed among the different vulnerabilities, based on the premise that the greatest the amount of 'pebbles' a vulnerability received, the most relevant was for the consideration when planning for the future of UA in the AMB. Participants agreed on the final distribution of the pebbles after discussing their perspectives on the relevance of each vulnerability. Values from the ranking were then converted to percentages.

8. Aggregation of single vulnerabilities for a combined vulnerability

Equation employed for the aggregation of single vulnerabilities via a weighted sum (Eq. B.5):

$$\mathrm{CV} = \sum V \times W_V \qquad \text{(Eq. B.5)}$$

Where
CV: combined vulnerability
V: single vulnerabilities
W: associated weight to vulnerability

References

Balvanera, P., Pfisterer, A. B., Buchmann, N., He, J. S., Nakashizuka, T., Raffaelli, D., & Schmid, B. (2006). Quantifying the evidence for biodiversity effects on ecosystem functioning and services. *Ecology Letters*, *9*(10), 1146–1156. https://doi.org/10.1111/J.1461-0248.2006.00963.X

Basagaña, X., Sartini, C., Barrera-Gómez, J., Dadvand, P., Cunillera, J., Ostro, B., Sunyer, J., & Medina-Ramón, M. (2011). Heat waves and cause-specific mortality at all ages. *Epidemiology*, *22*(6), 765–772. https://doi.org/10.1097/EDE.0B013E31823031C5

Basnou, C., Baró, F., Langemeyer, J., Castell, C., Dalmases, C., & Pino, J. (2020). Advancing the green infrastructure approach in the Province of Barcelona: integrating biodiversity, ecosystem functions and services into landscape planning. *Urban Forestry & Urban Greening*, *55*, 126797. https://doi.org/10.1016/J.UFUG.2020.126797

Bauwelinck, M., Casas, L., Nawrot, T. S., Nemery, B., Trabelsi, S., Thomas, I., Aerts, R., Lefebvre, W., Vanpoucke, C., Van Nieuwenhuyse, A., Deboosere, P., & Vandenheede, H. (2021). Residing in urban areas with higher green space is associated with lower mortality risk: A census-based cohort study with ten years of follow-up. *Environment International*, *148*, 106365. https://doi.org/10.1016/J.ENVINT.2020.106365

CREAF. (2015). *Land Cover Map of the Barcelona Metropolitan Area*. http://www.creaf.uab.es/mcsc/usa/index.htm. Accessed October 1, 2023

Díaz, J., Carmona, R., Mirón, I. J., Ortiz, C., León, I., & Linares, C. (2015). Geographical variation in relative risks associated with heat: Update of Spain's Heat Wave Prevention Plan. *Environment International*, *85*, 273–283. https://doi.org/10.1016/J.ENVINT.2015.09.022

European Environment Agency, Wugt Larsen, F., Lung, T. (2020*). Is Europe living within the limits of our planet?: an assessment of Europe's environmental footprints in relation to planetary boundaries*, Publications Office. https://data.europa.eu/doi/10.2800/890673

Grazuleviciene, R., Dedele, A., Danileviciute, A., Vencloviene, J., Grazulevicius, T., Andrusaityte, S., Uzdanaviciute, I., & Nieuwenhuijsen, M. J. (2014). The influence of proximity to city parks on blood pressure in early pregnancy. *International Journal of Environmental Research and Public Health*, *11*(3), 2958–2972. https://doi.org/10.3390/IJERPH110302958

Guan, W., Liu, Q., & Dong, C. (2022). Risk assessment method for industrial accident consequences and human vulnerability in urban areas. *Journal of Loss Prevention in the Process Industries*, *76*, 104745. https://doi.org/10.1016/J.JLP.2022.104745

Hajat, S., O'Connor, M., & Kosatsky, T. (2010). Health effects of hot weather: from awareness of risk factors to effective health protection. *The Lancet*, *375*(9717), 856–863. https://doi.org/10.1016/S0140-6736(09)61711-6

Heaviside, C., Vardoulakis, S., & Cai, X. M. (2016). Attribution of mortality to the urban heat island during heatwaves in the West Midlands, UK. *Environmental Health: A Global Access Science Source*, *15*(1), 49–59. https://doi.org/10.1186/S12940-016-0100-9/TABLES/3

Huang, J., Xu, C. chun, Ridoutt, B. G., Wang, X. chun, & Ren, P. an. (2017). Nitrogen and phosphorus losses and eutrophication potential associated with fertilizer application to cropland in China. *Journal of Cleaner Production*, *159*, 171–179. https://doi.org/10.1016/J.JCLEPRO.2017.05.008

Humpel, N., Owen, N., & Leslie, E. (2002). Environmental factors associated with adults' participation in physical activity. A review. *American Journal of Preventive Medicine*, *22*(3), 188–199. https://doi.org/10.1016/S0749-3797(01)00426-3

Inèdit. (2022). Petjada de carboni de l'alimentació a Barcelona. Ajuntament de Barcelona. https://ajuntament.barcelona.cat/premsa/wp-content/uploads/2021/11/Presentatio_PC_Alimentacio_RDP081121-final.pdf. Accessed October 1, 2023

INE. (2021). *Population by sex, municipalities and age (five-year groups)*. Instituto Nacional de Estadística. Retrieved from https://www.ine.es/jaxiT3/Tabla.htm?t=33716&L=0 . Accessed October 1, 2023

Justus, J., & Sarkar, S. (2002). The principle of complementarity in the design of reserve networks to conserve biodiversity: A preliminary history. *Journal of Biosciences*, *27*(4 SUPPL. 2), 421–435. https://doi.org/10.1007/BF02704970

Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Grime, J. P., Hector, A., Hooper, D. U., Huston, M. A., Raffaelli, D., Schmid, B., Tilman, D., & Wardle, D. A. (2001). Ecology: Biodiversity and ecosystem functioning: Current knowledge and future challenges. *Science*, *294*(5543), 804–808. https://doi.org/10.1126/SCIENCE.1064088

Maas, J., Verheij, R. A., De Vries, S., Spreeuwenberg, P., Schellevis, F. G., & Groenewegen, P. P. (2009). Morbidity is related to a green living environment. *Journal of Epidemiology & Community Health*, *63*(12), 967–973. https://doi.org/10.1136/JECH.2008.079038

Marull, J., Padró, R., Cirera, J., Giocoli, A., Pons, M., & Tello, E. (2021). A socioecological integrated analysis of the Barcelona metropolitan agricultural landscapes. *Ecosystem Services*, *51*, 101350. https://doi.org/10.1016/J.ECOSER.2021.101350

Marull, J., Pino, J., Mallarach, J. M., & Cordobilla, M. J. (2007). A Land Suitability Index for Strategic Environmental Assessment in metropolitan areas. *Landscape and Urban Planning*, *81*(3), 200–212. https://doi.org/10.1016/J.LANDURBPLAN.2006.11.005

Mendoza Beltran, A., Jepsen, K., Rufí-Salís, M., Ventura, S., Madrid Lopez, C., & Villalba, G. (2022). Mapping direct N2O emissions from peri-urban agriculture: The case of the Metropolitan Area of Barcelona. *Science of The Total Environment*, *822*, 153514. https://doi.org/10.1016/J.SCITOTENV.2022.153514

Ministerio de Agricultura (2022). *Household food consumption data series*. Ministerio de Agricultura, Pesca y Alimentación. https://www.mapa.gob.es/es/alimentacion/temas/consumo-tendencias/panel-de-consumo-alimentario/series-anuales/

Neuvonen, M., Sievänen, T., Tönnes, S., & Koskela, T. (2007). Access to green areas and the frequency of visits – A case study in Helsinki. *Urban Forestry & Urban Greening*, *6*(4), 235–247. https://doi.org/10.1016/J.UFUG.2007.05.003

Oudin Åström, D., Bertil, F., & Joacim, R. (2011). Heat wave impact on morbidity and mortality in the elderly population: A review of recent studies. *Maturitas*, *69*(2), 99–105. https://doi.org/10.1016/J.MATURITAS.2011.03.008

Padró, R., La Rota-Aguilera, M. J., Giocoli, A., Cirera, J., Coll, F., Pons, M., Pino, J., Pili, S., Serrano, T., Villalba, G., & Marull, J. (2020). Assessing the sustainability of contrasting land use scenarios through the Socioecological Integrated Analysis (SIA) of the metropolitan green infrastructure in Barcelona. *Landscape and Urban Planning*, *203*, 103905. https://doi.org/10.1016/J.LANDURBPLAN.2020.103905

Paquet, C., Orschulok, T. P., Coffee, N. T., Howard, N. J., Hugo, G., Taylor, A. W., Adams, R. J., & Daniel, M. (2013). Are accessibility and characteristics of public open spaces associated with a better cardiometabolic health? *Landscape and Urban Planning*, *118*, 70–78. https://doi.org/10.1016/J.LANDURBPLAN.2012.11.011

Reid, C. E., Clougherty, J. E., Shmool, J. L. C., & Kubzansky, L. D. (2017). Is All Urban Green Space the Same? A Comparison of the Health Benefits of Trees and Grass in New York City. *International Journal of Environmental Research and Public Health 2017, Vol. 14, Page 1411*, *14*(11), 1411. https://doi.org/10.3390/IJERPH14111411

Royé, D. (2017). The effects of hot nights on mortality in Barcelona, Spain. *International Journal of Biometeorology*, *61*(12), 2127–2140. https://doi.org/10.1007/S00484-017-1416-Z/METRICS

Segura, R., Badia, A., Ventura, S., Gilabert, J., Martilli, A., & Villalba, G. (2021). Sensitivity study of PBL schemes and soil initialization using the WRF-BEP-BEM model over a Mediterranean coastal city. *Urban Climate*, *39*, 100982. https://doi.org/10.1016/J.UCLIM.2021.100982

Sera, F., Armstrong, B., Tobias, A., Vicedo-Cabrera, A. M., Åström, C., Bell, M. L., Chen, B.-Y., de Sousa Zanotti Stagliorio Coelho, M., Matus Correa, P., Cruz, J. C., Dang, T. N., Hurtado-Diaz, M., Do Van, D., Forsberg, B., Guo, Y. L., Guo, Y., Hashizume, M., Honda, Y., Iñiguez, C., Jaakkola, J. Kan, H., Kim, H., Lavigne, E., Michelozzi, P., Ortega, N., Osorio, S., Pascal, M., Ragettli, M., Ryti, N., Saldiva, P., Schwartz, J., Scortichini, M., Seposo, X., Tong. S., Zanobetti, A., Gasparrini, A. (2019). How urban characteristics affect vulnerability to heat and cold: a multi-country analysis. *International Journal of Epidemiology*, *48*(4), 1101–1112. https://doi.org/10.1093/ije/dyz008

Smith, C. L., Webb, A., Levermore, G. J., Lindley, S. J., & Beswick, K. (2011). Fine-scale spatial temperature patterns across a UK conurbation. *Climatic Change*, *109*(3–4), 269–286. https://doi.org/10.1007/S10584-011-0021-0

Stigsdotter, U. K., Randrup, T. B., Ekholm, O., Schipperijn, J., Toftager, M., & Kamper-Jørgensen, F. (2010). Health promoting outdoor environments - Associations between green space, and health, health-related quality of life and stress based on a Danish national representative survey. *Scandinavian Journal of Public Health*, 38(4), 411–417. https://doi.org/10.1177/1403494810367468

URBAG. (2022). *Agricultural perspectives in the Metropolitan Area of Barcelona*. https://urbag.eu/wp-content/uploads/2023/03/UA-AMB_Report_01_EN.pdf. Accessed October 1, 2023

Vandentorren, S., Bretin, P., Zeghnoun, A., Mandereau-Bruno, L., Croisier, A., Cochet, C., Ribéron, J., Siberan, I., Declercq, B., & Ledrans, M. (2006). August 2003 heat wave in France: Risk factors for death of elderly people living at home. *European Journal of Public Health*, *16*(6), 583–591. https://doi.org/10.1093/EURPUB/CKL063

Vos, S., Bijnens, E. M., Renaers, E., Croons, H., Van Der Stukken, C., Martens, D. S., Plusquin, M., & Nawrot, T. S. (2022). Residential green space is associated with a buffering effect on stress responses during the COVID-19 pandemic in mothers of young children, a prospective study. *Environmental Research*, *208*, 112603. https://doi.org/10.1016/J.ENVRES.2021.112603

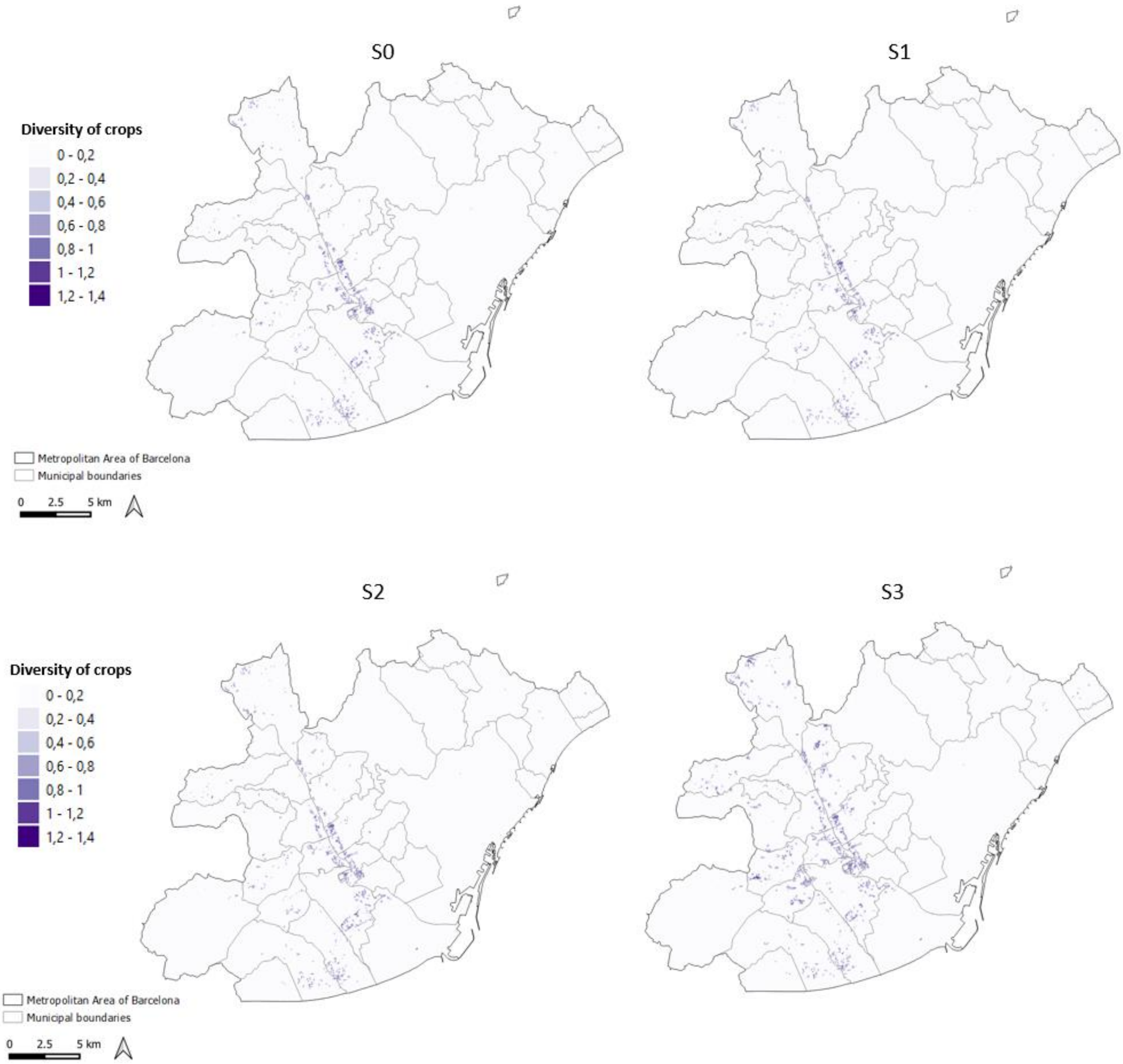
Wang, X. Y., Barnett, A. G., Yu, W., FitzGerald, G., Tippett, V., Aitken, P., Neville, G., McRae, D., Verrall, K., & Tong, S. (2012). The impact of heatwaves on mortality and emergency hospital admissions from non-external causes in Brisbane, Australia. *Occupational and Environmental Medicine*, *69*(3), 163–169. https://doi.org/10.1136/OEM.2010.062141

Xu, Y., Dadvand, P., Barrera-Gómez, J., Sartini, C., Marí-Dell'Olmo, M., Borrell, C., Medina-Ramón, M., Sunyer, J., & Basagaña, X. (2013). Differences on the effect of heat waves on mortality by sociodemographic and urban landscape characteristics. *J Epidemiol Community Health*, *67*(6), 519–525. https://doi.org/10.1136/JECH-2012-20189

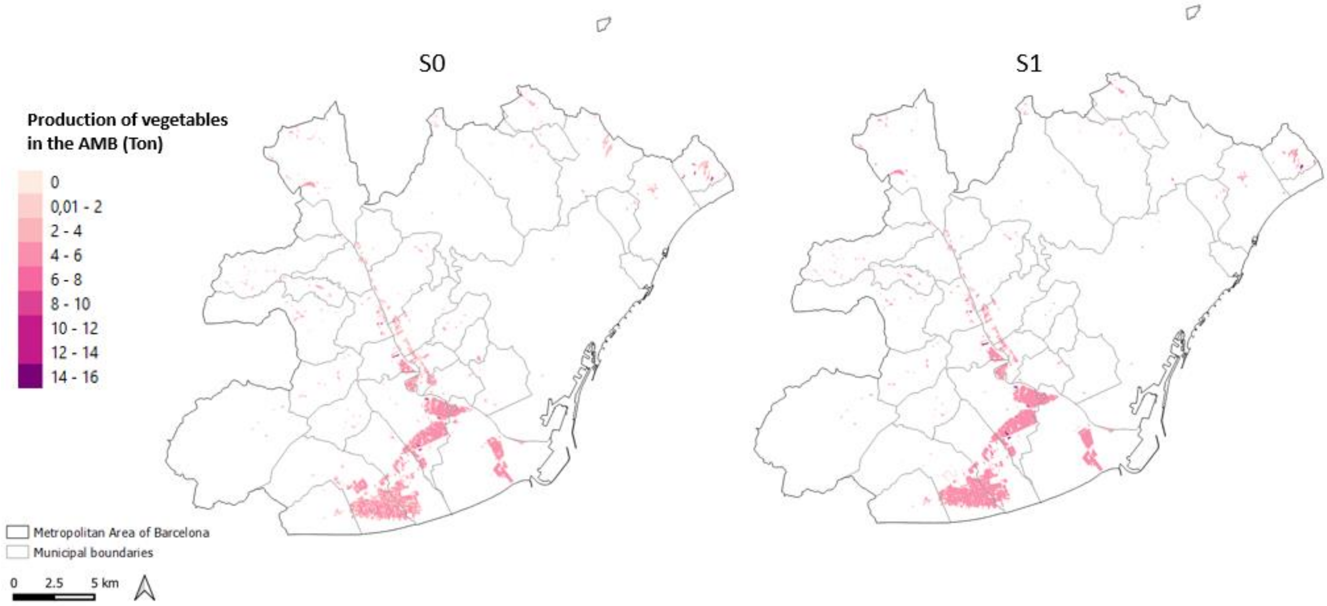
1-C. Supplementary maps

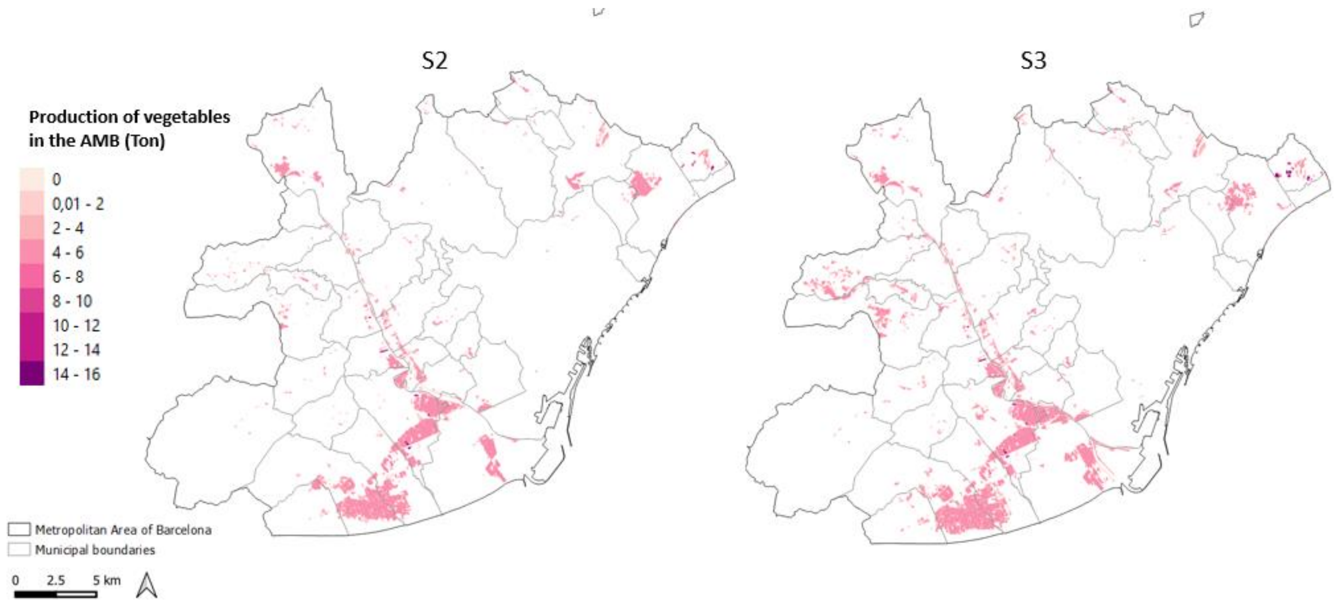
1. Vulnerability of lack of local food

1.1. Diversity of crops (exposure)

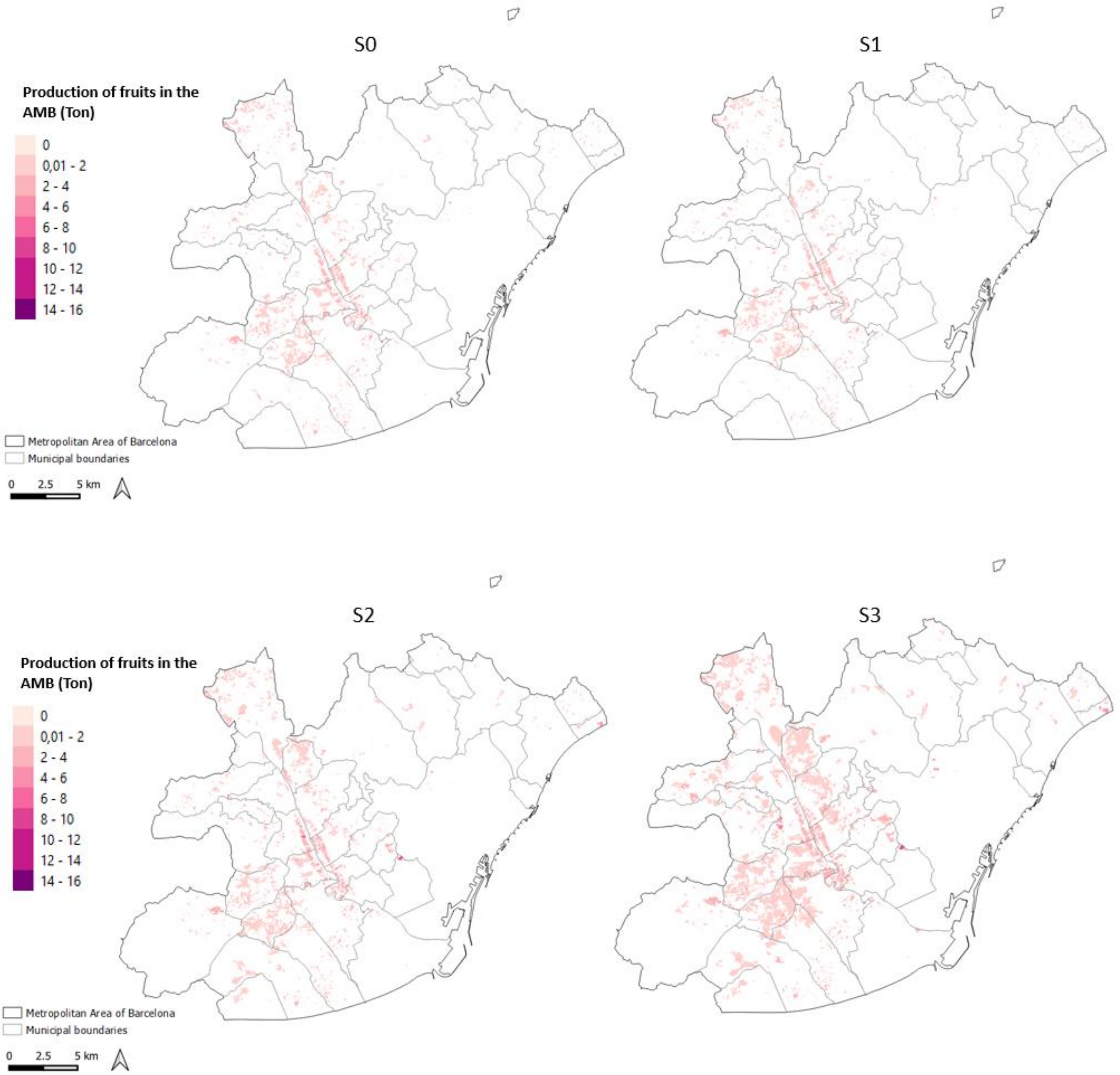


1.2. Production of vegetables in the AMB (exposure)

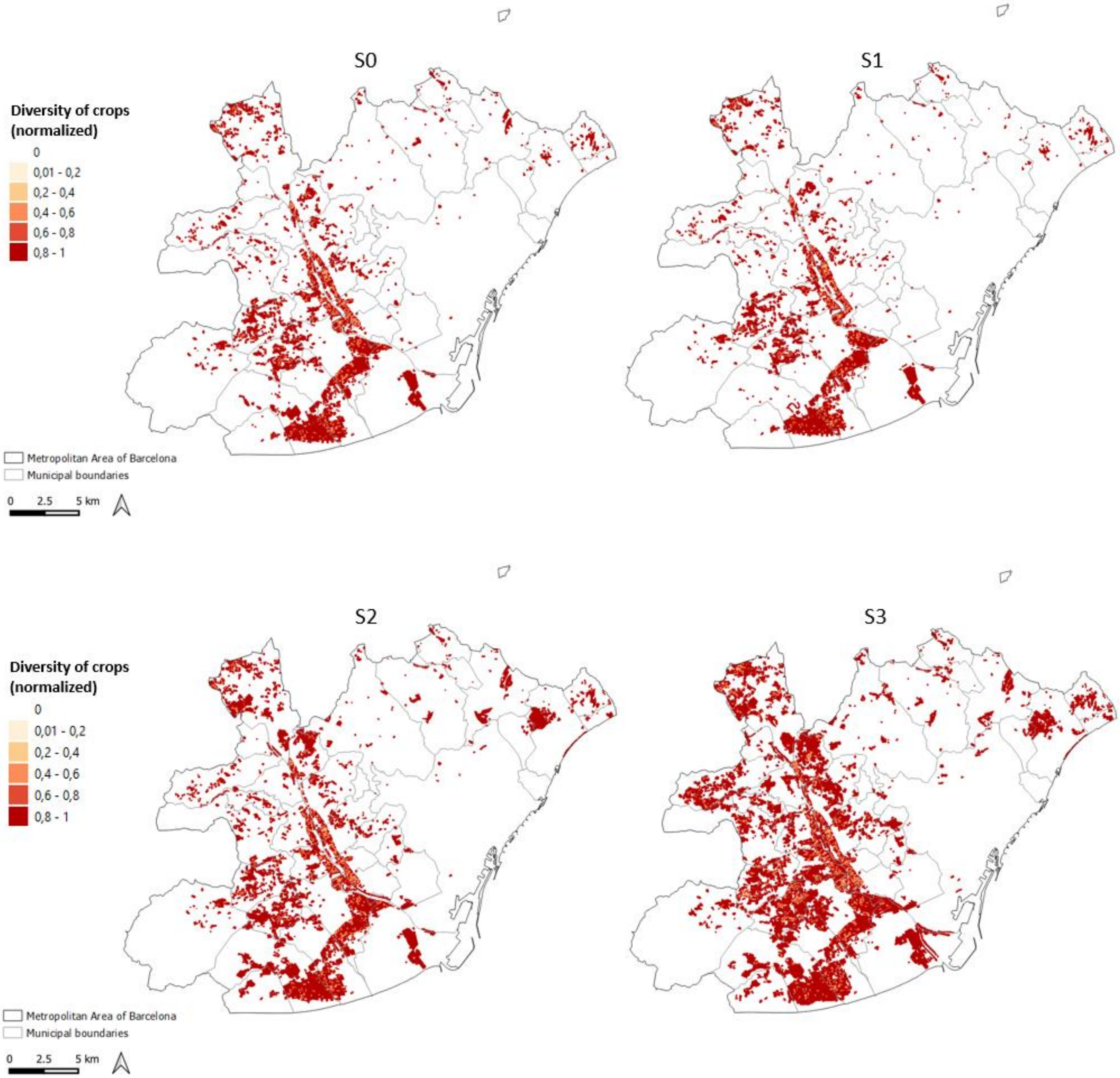




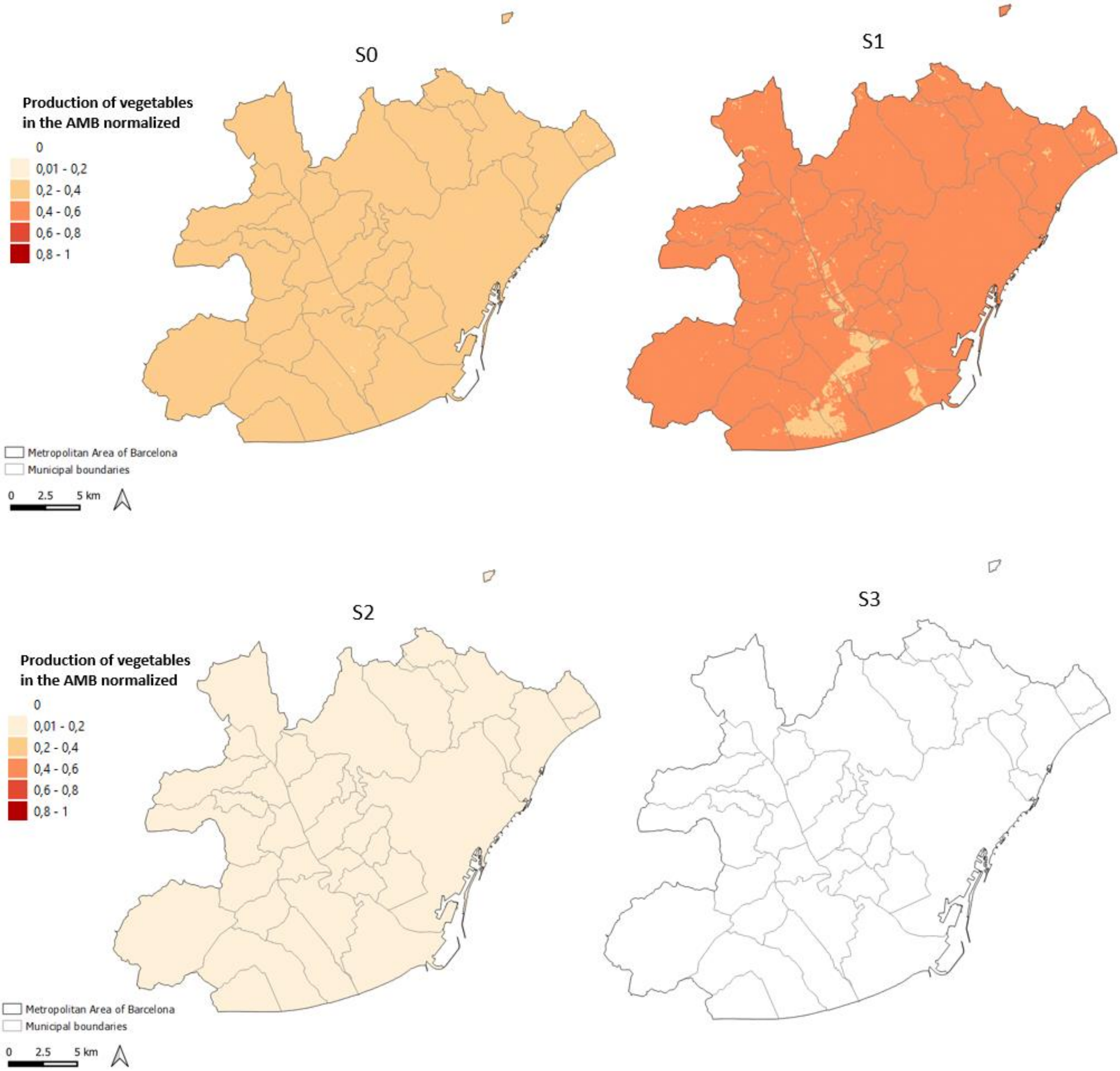
1.3. Production of fruits in the AMB (exposure)



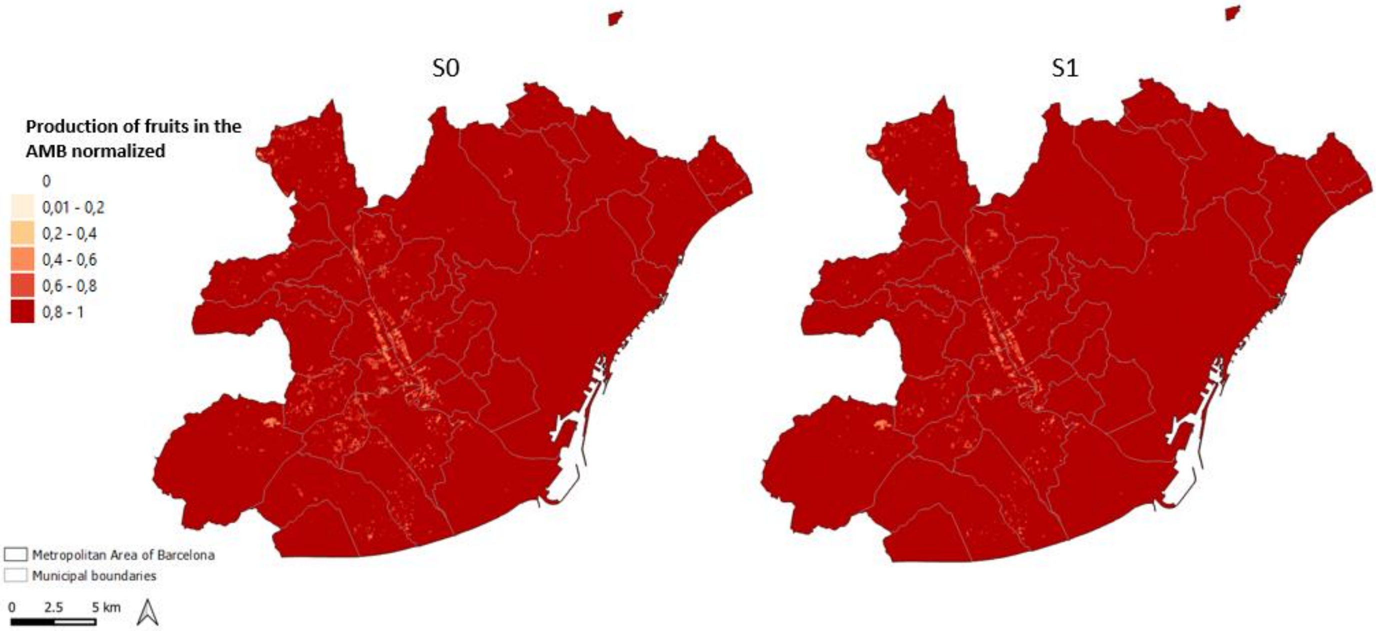
1.4. Diversity of crops normalized (exposure)

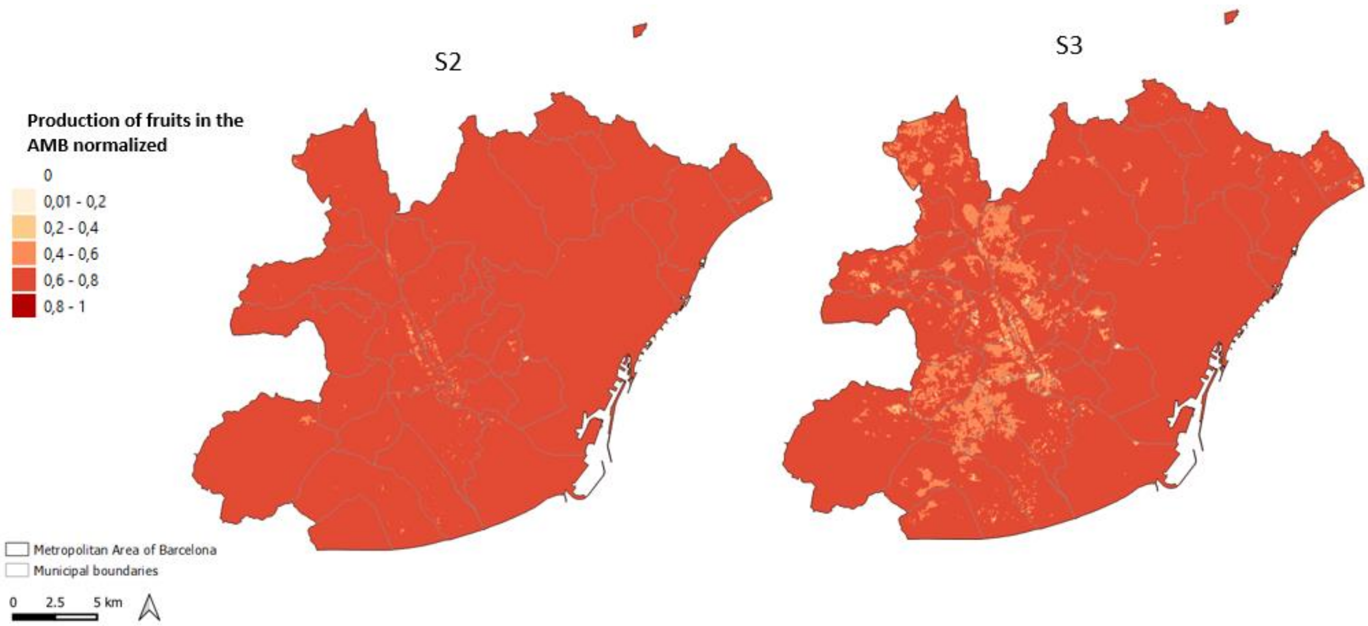


1.5. Production of vegetables in the AMB (normalized)

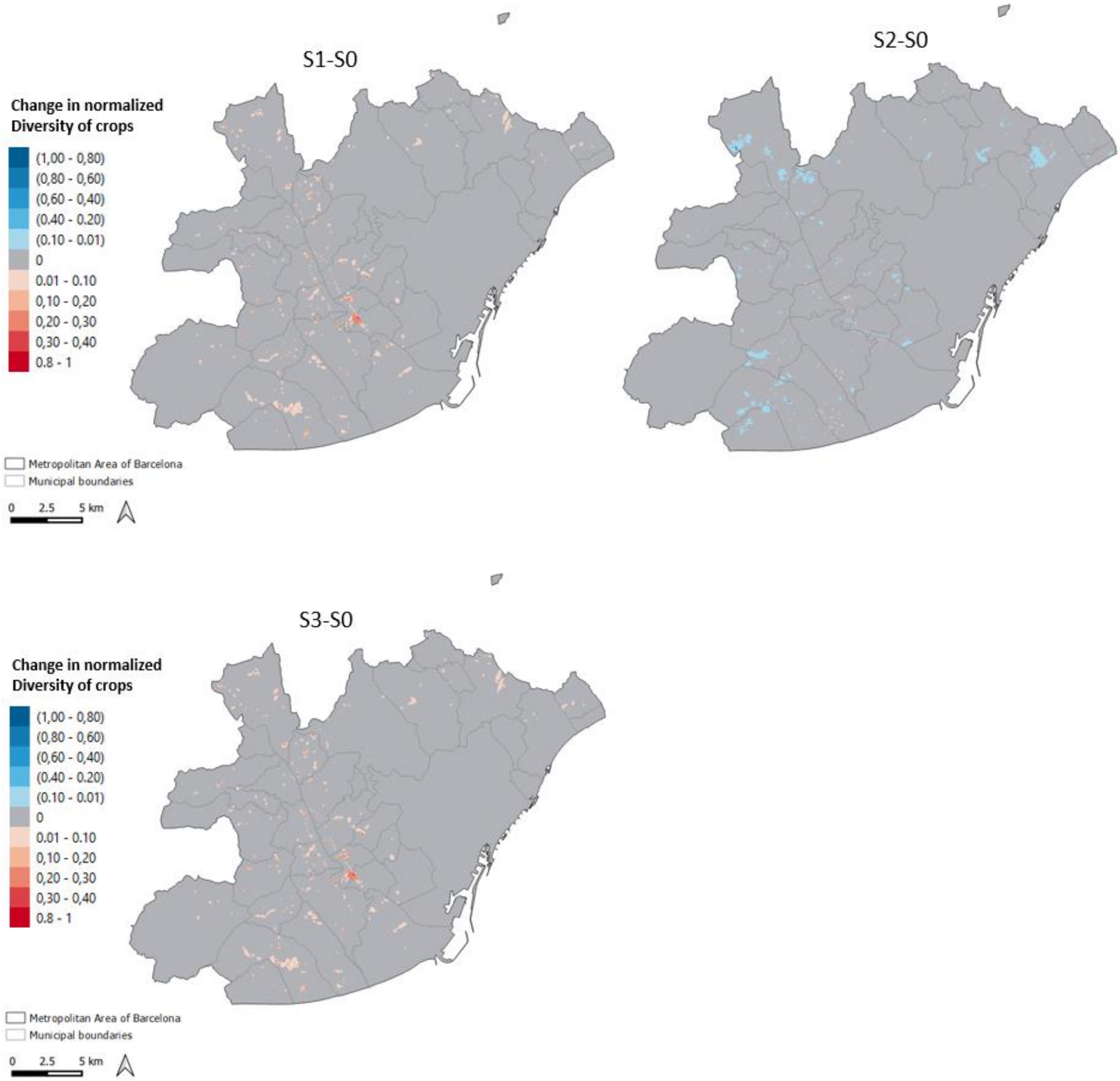


1.6. Production of fruits in the AMB (normalized)

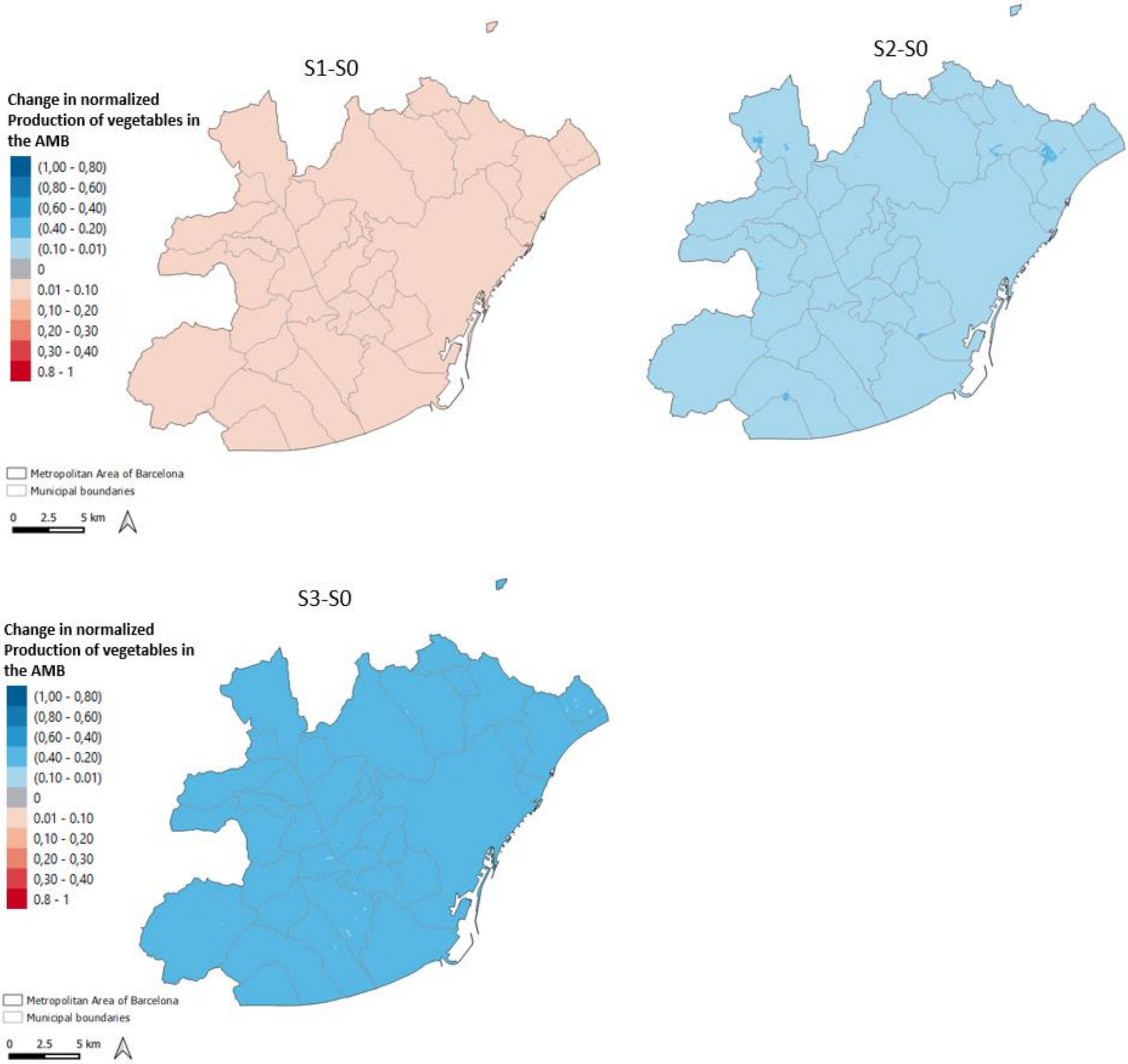




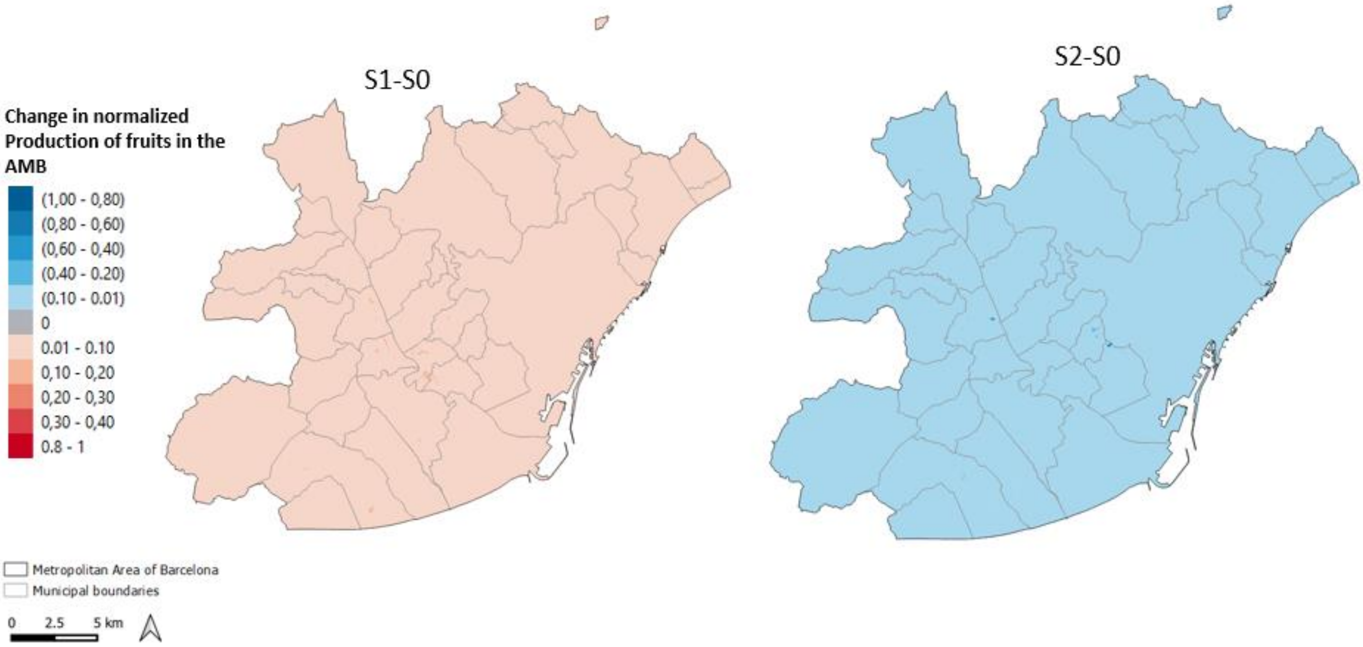
1.7. Change in normalized Diversity of crops (Exposure)

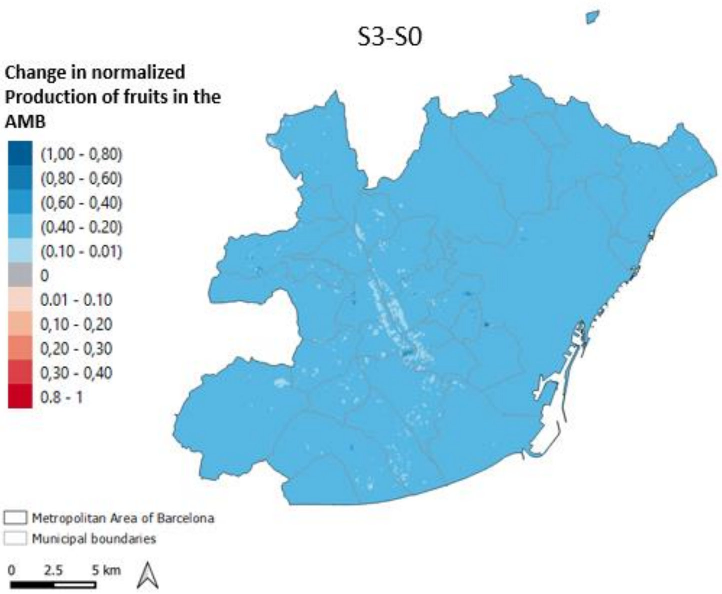


1.8. Change in normalized Production of vegetables in the AMB (Exposure)

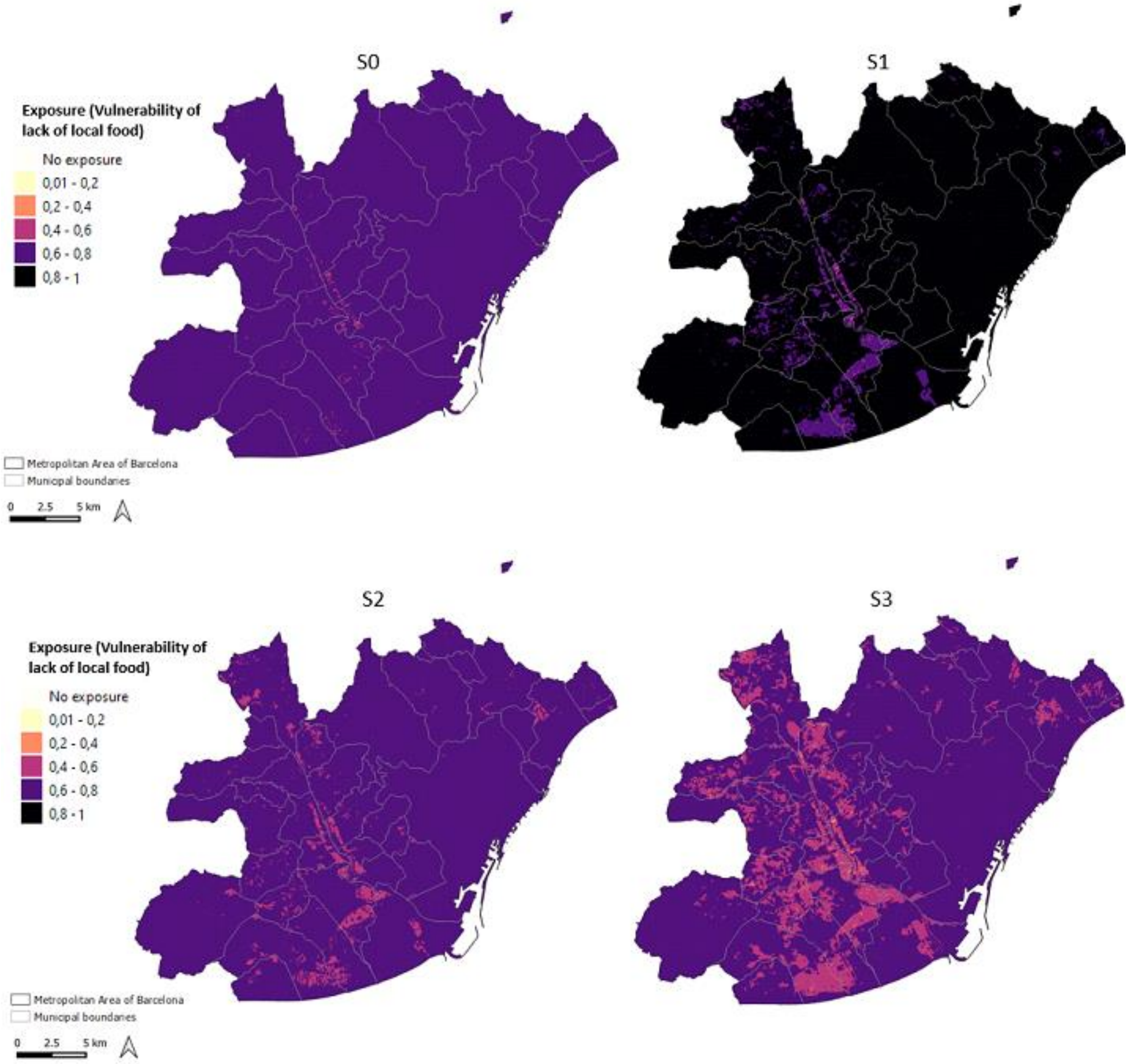


1.9. Change in normalized Production of fruits in the AMB (Exposure)

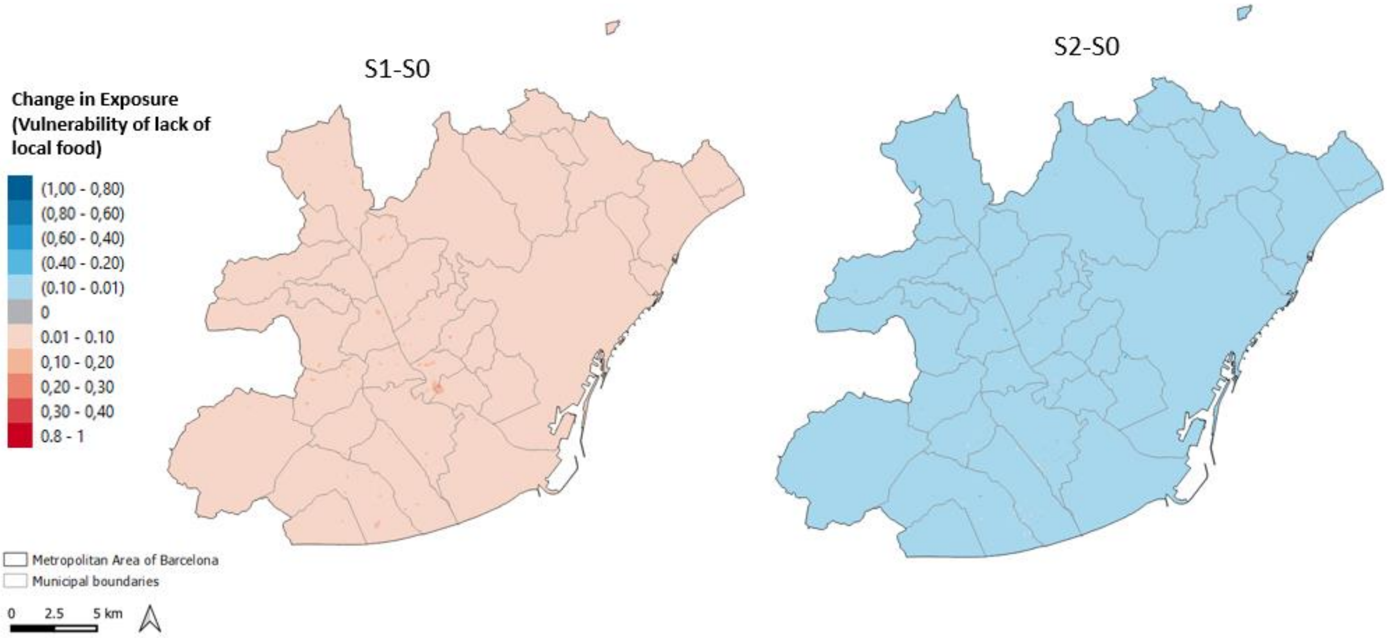


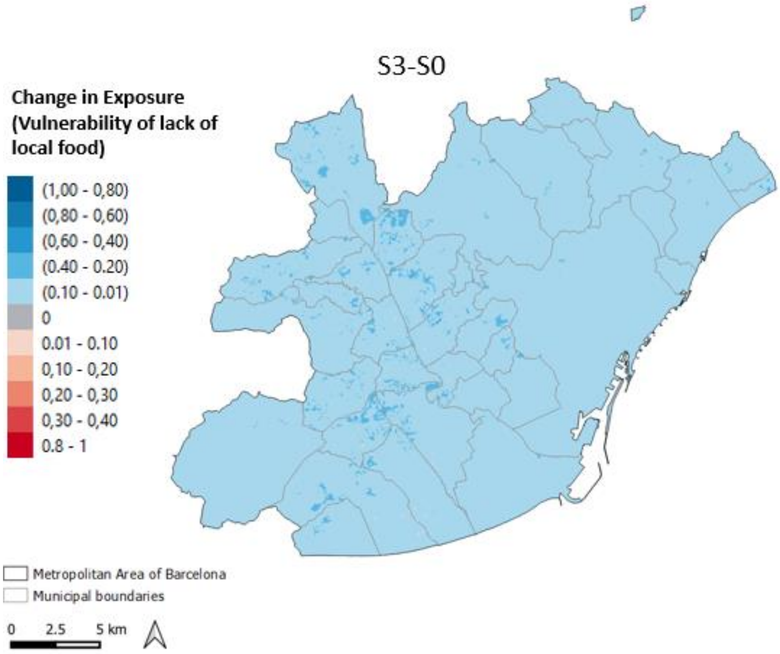


1.10. Exposure of Vulnerability of lack of local food

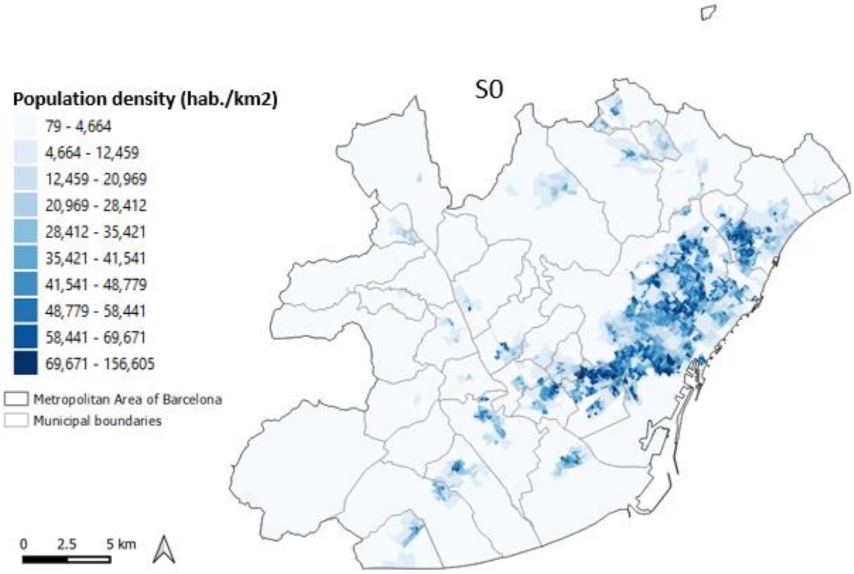


1.11. Changes in exposure of Vulnerability of lack of local food

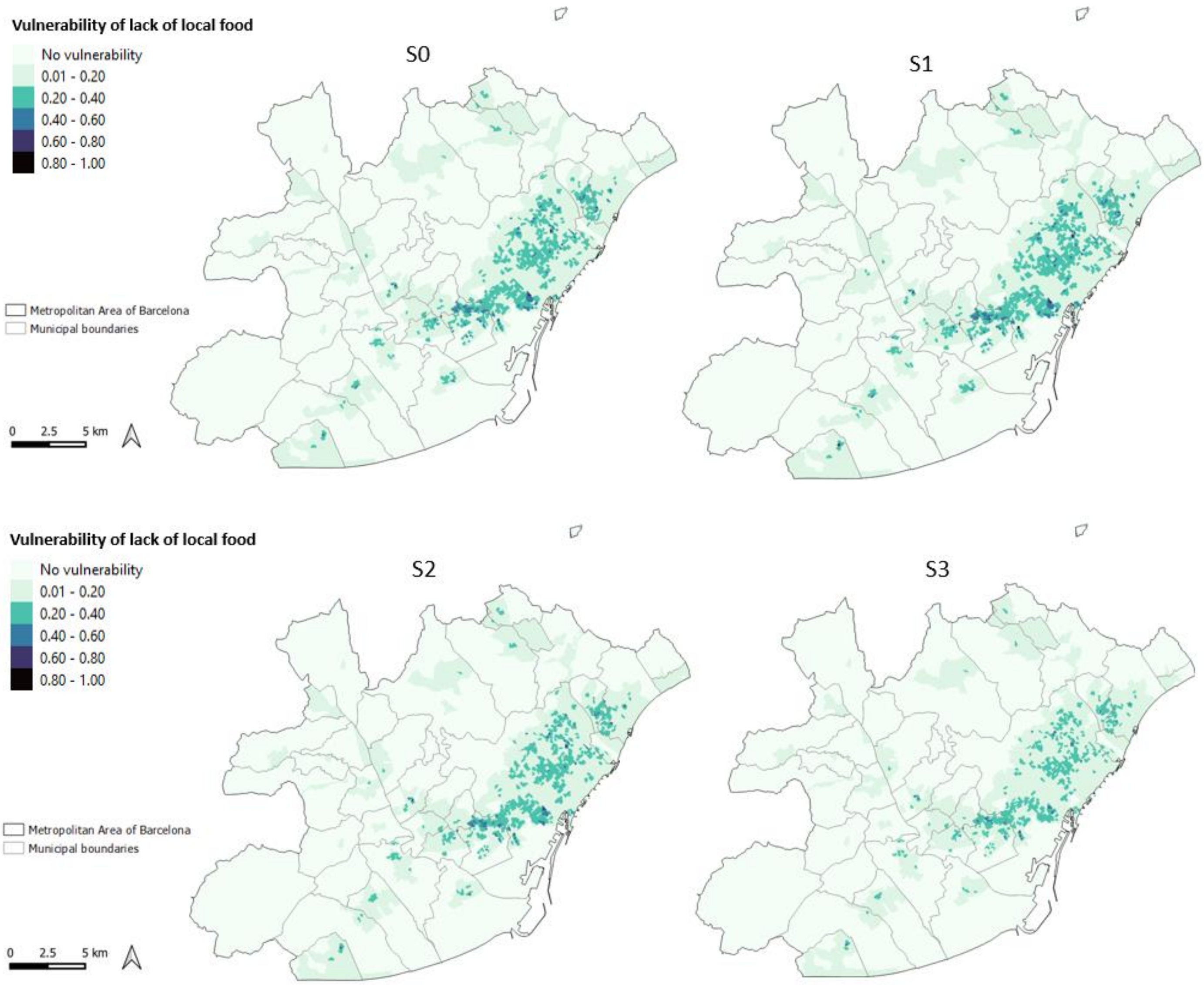




1.12. Population density

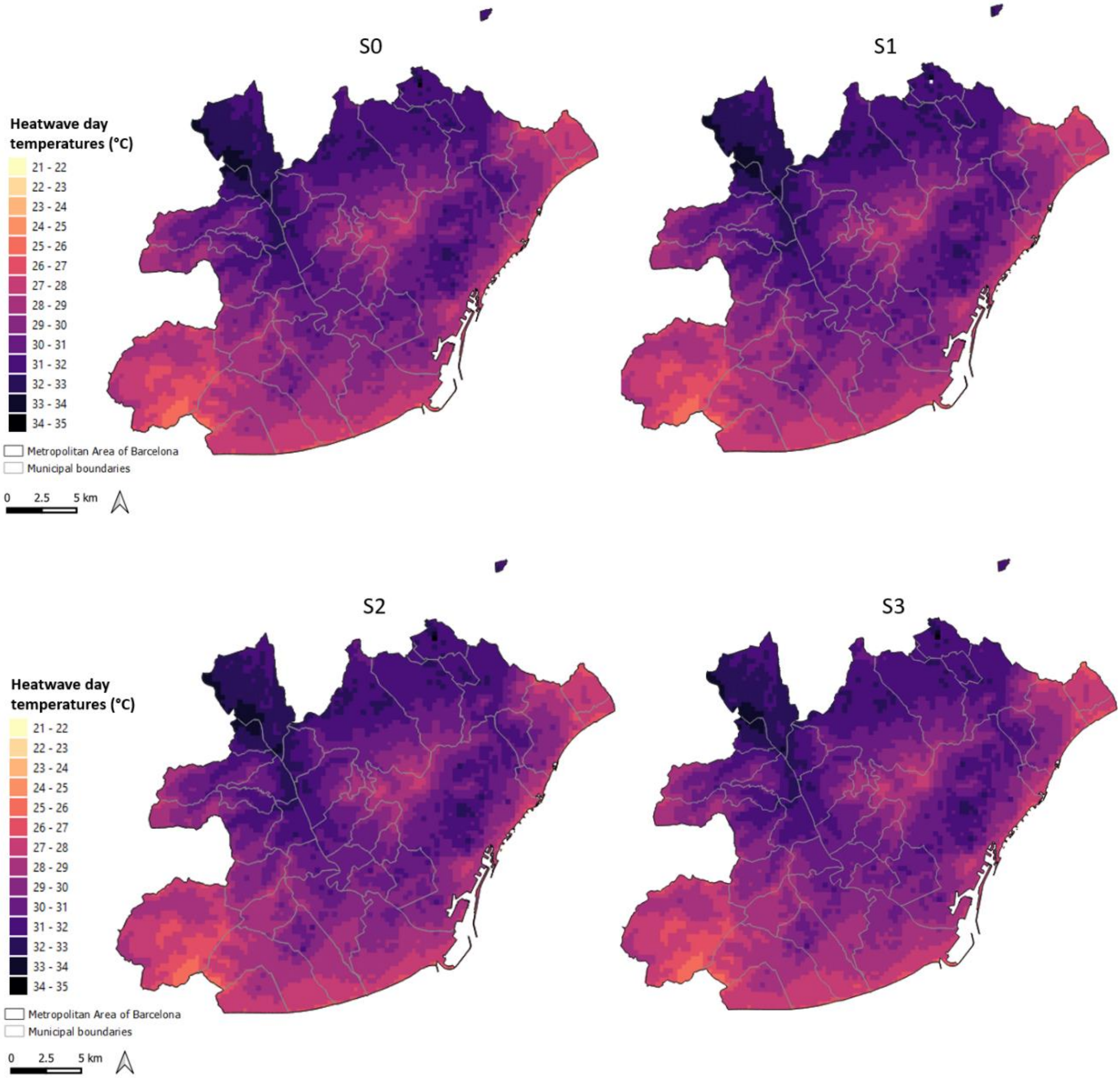


1.13. Vulnerability of lack of local food (single vulnerabilities)

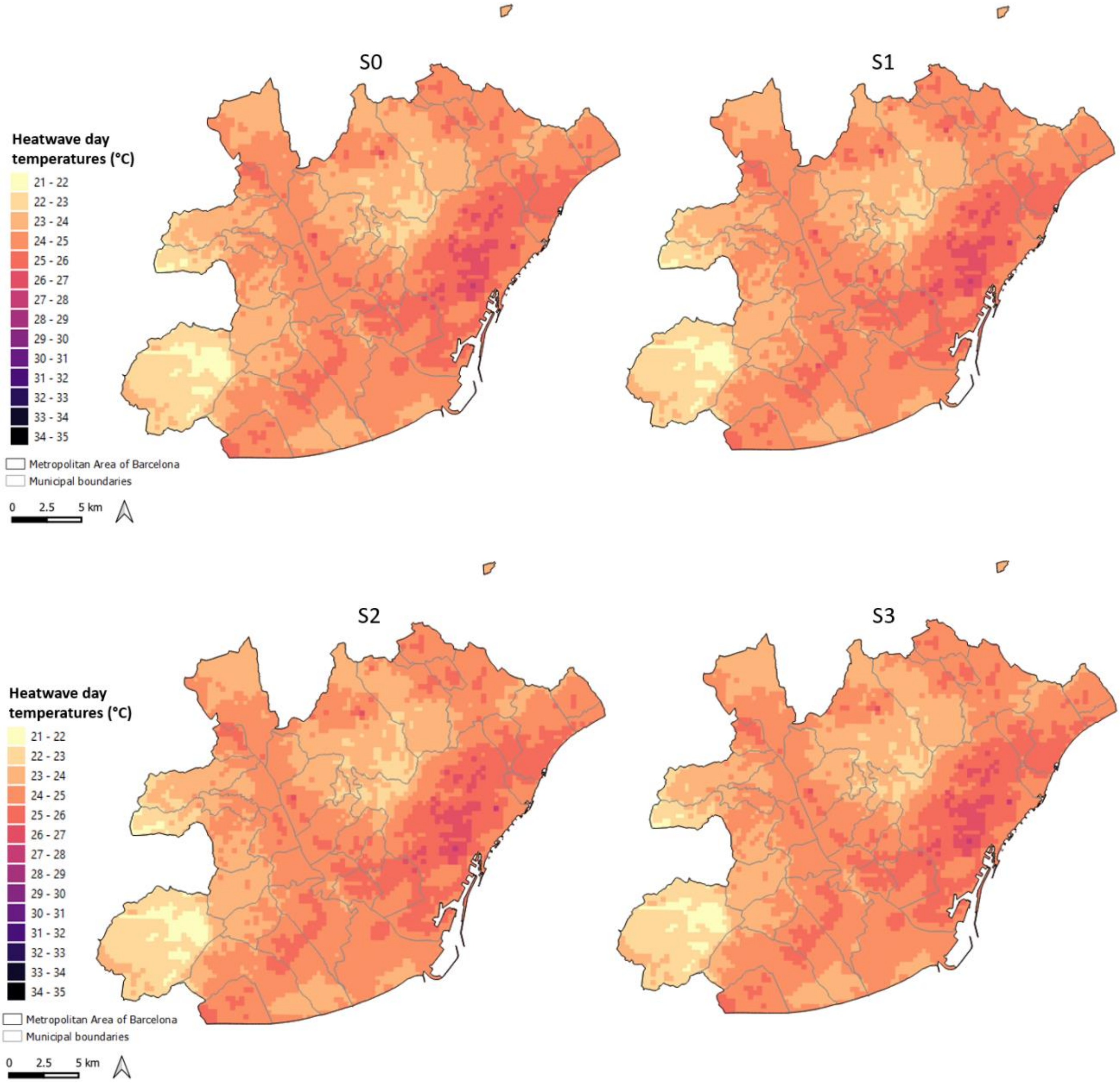


2. Vulnerability to heat

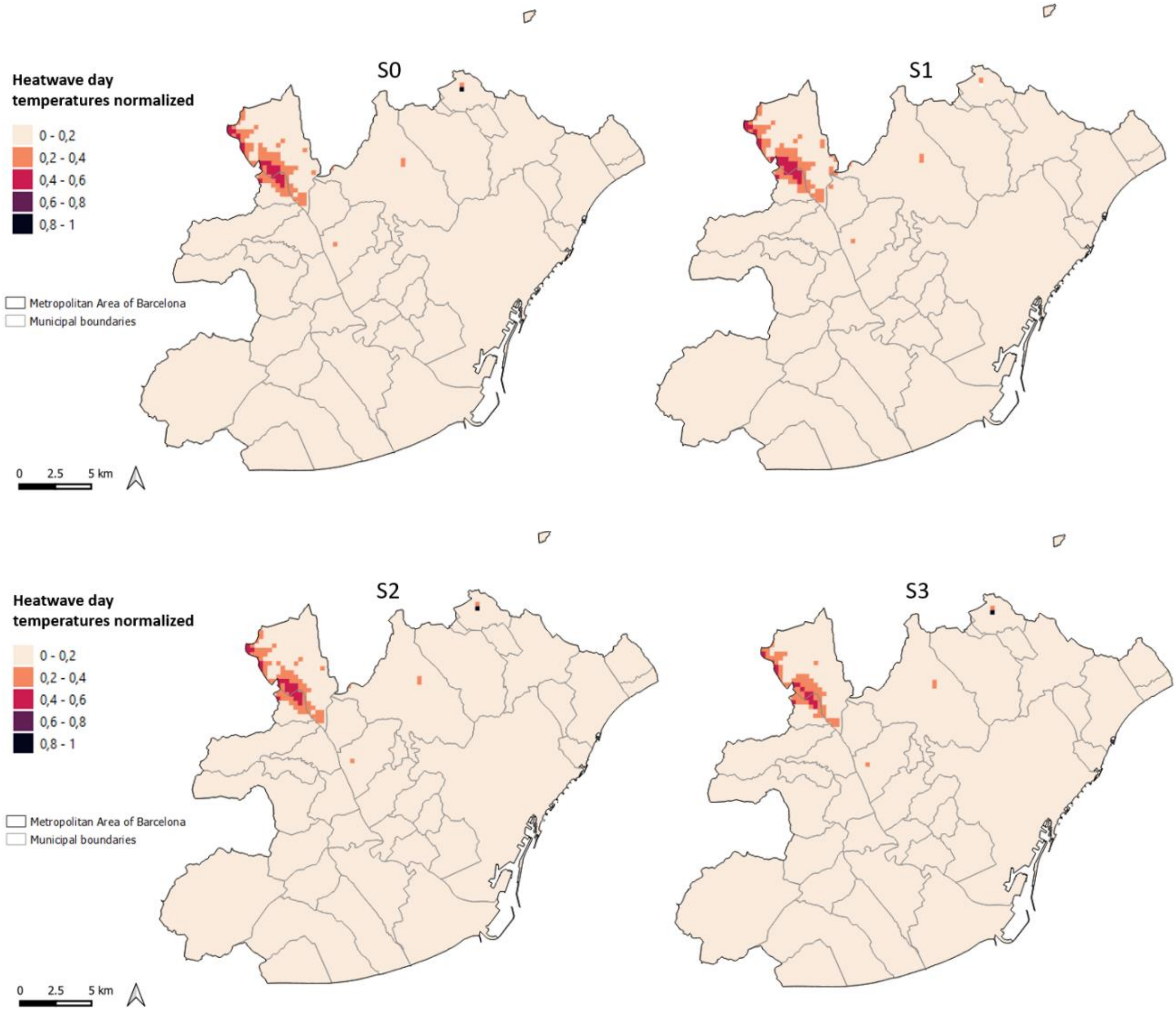
2.1. Heatwave day temperatures (Exposure)



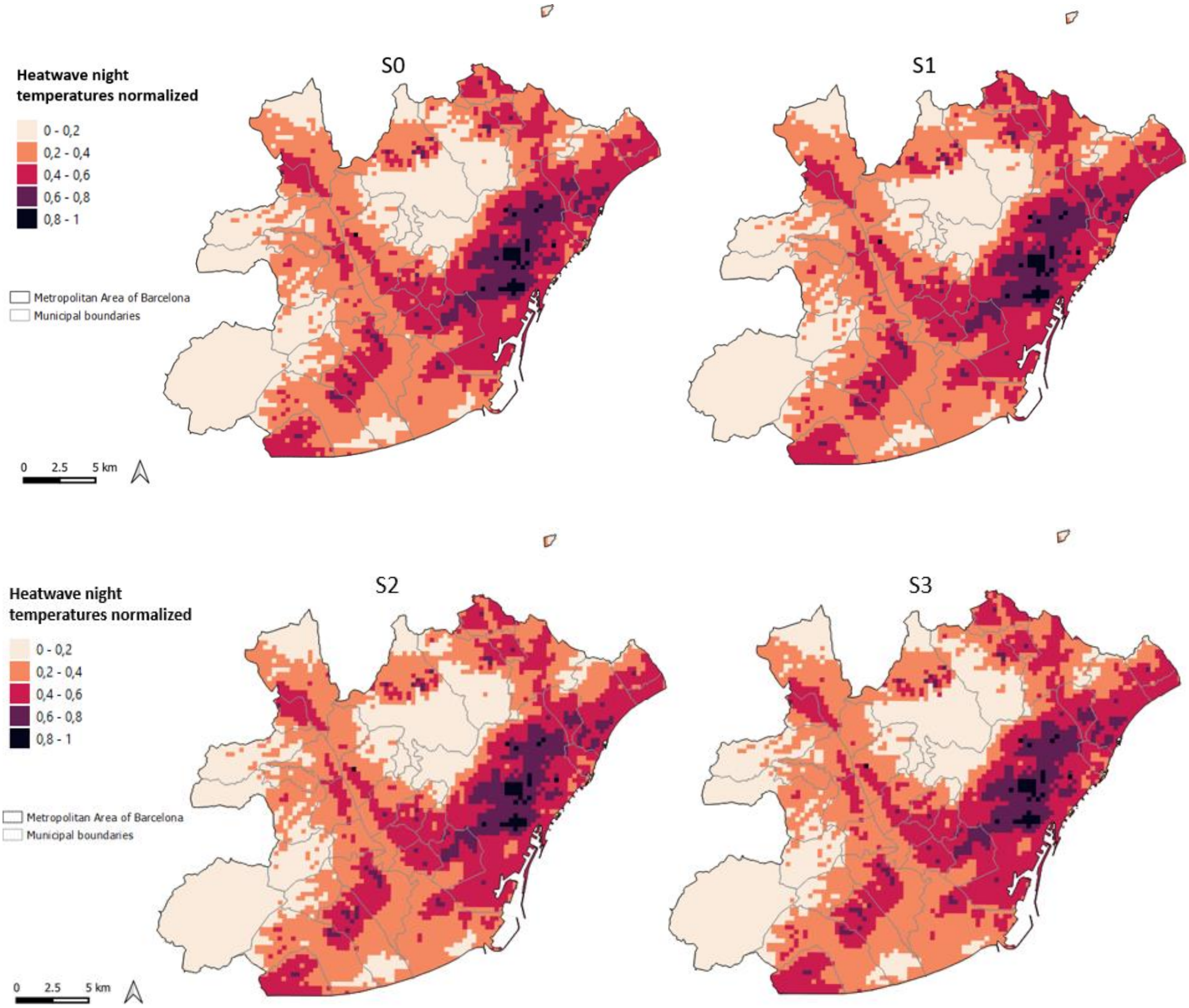
2.2. Heatwave night temperatures (Exposure)



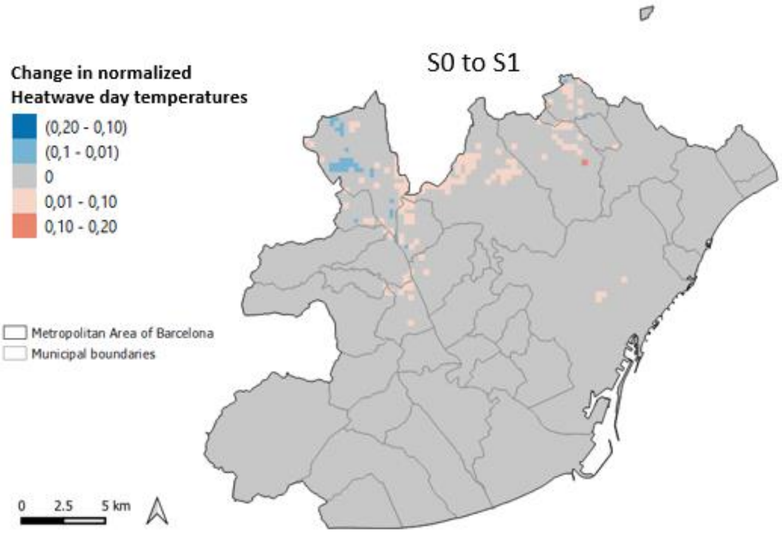
2.3. Heatwave day temperatures normalized (Exposure)

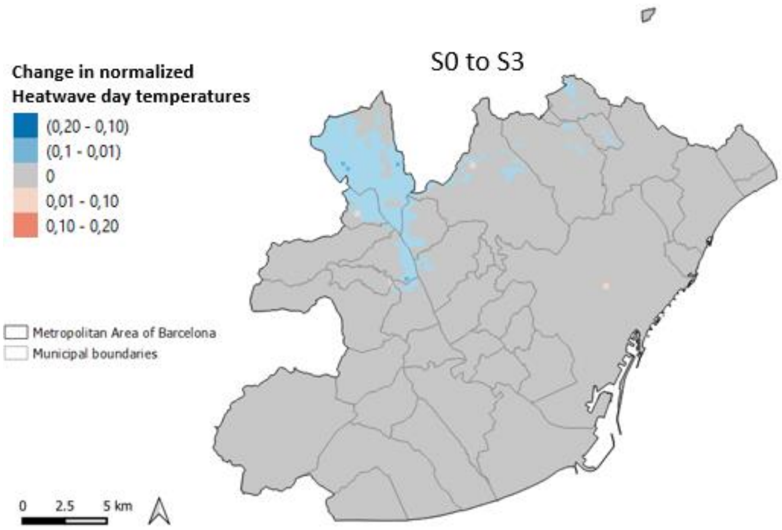


2.4. Heatwave night temperatures normalized (Exposure)

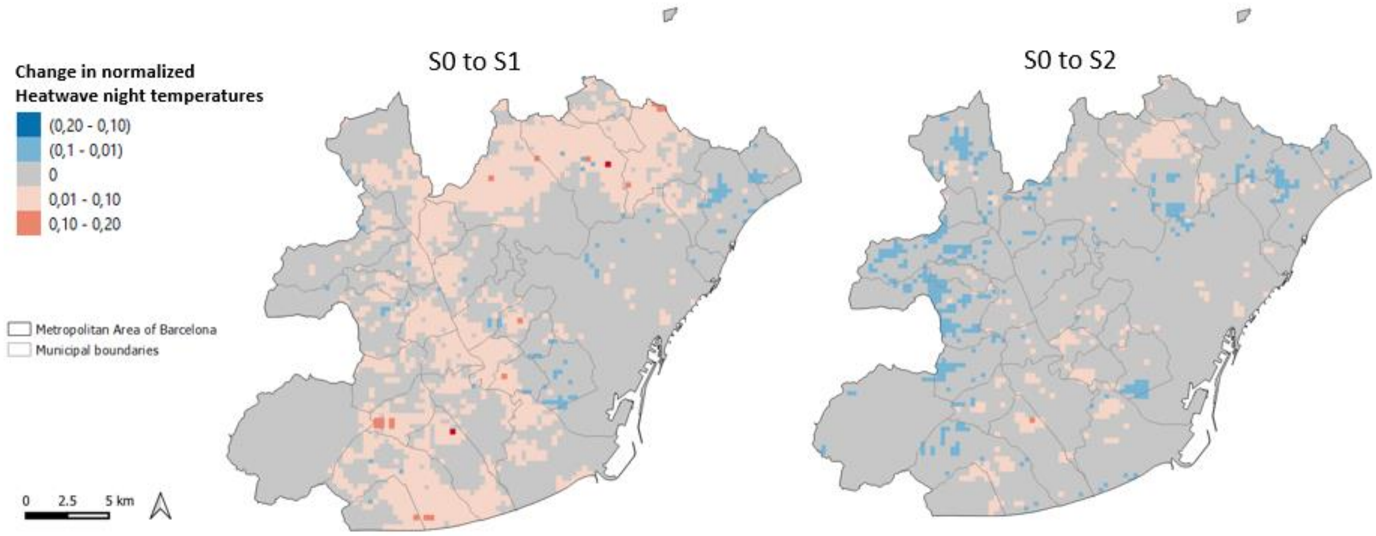


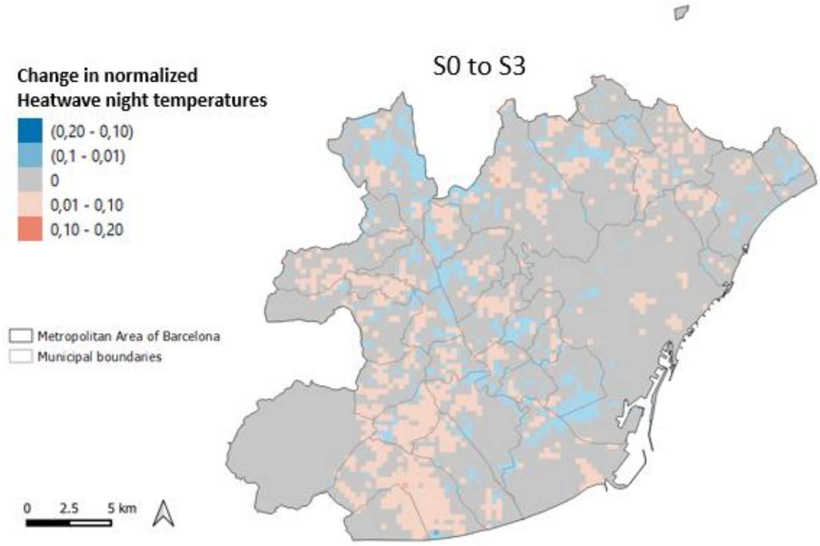
2.5. Change in normalized Heatwave day temperatures (Exposure)



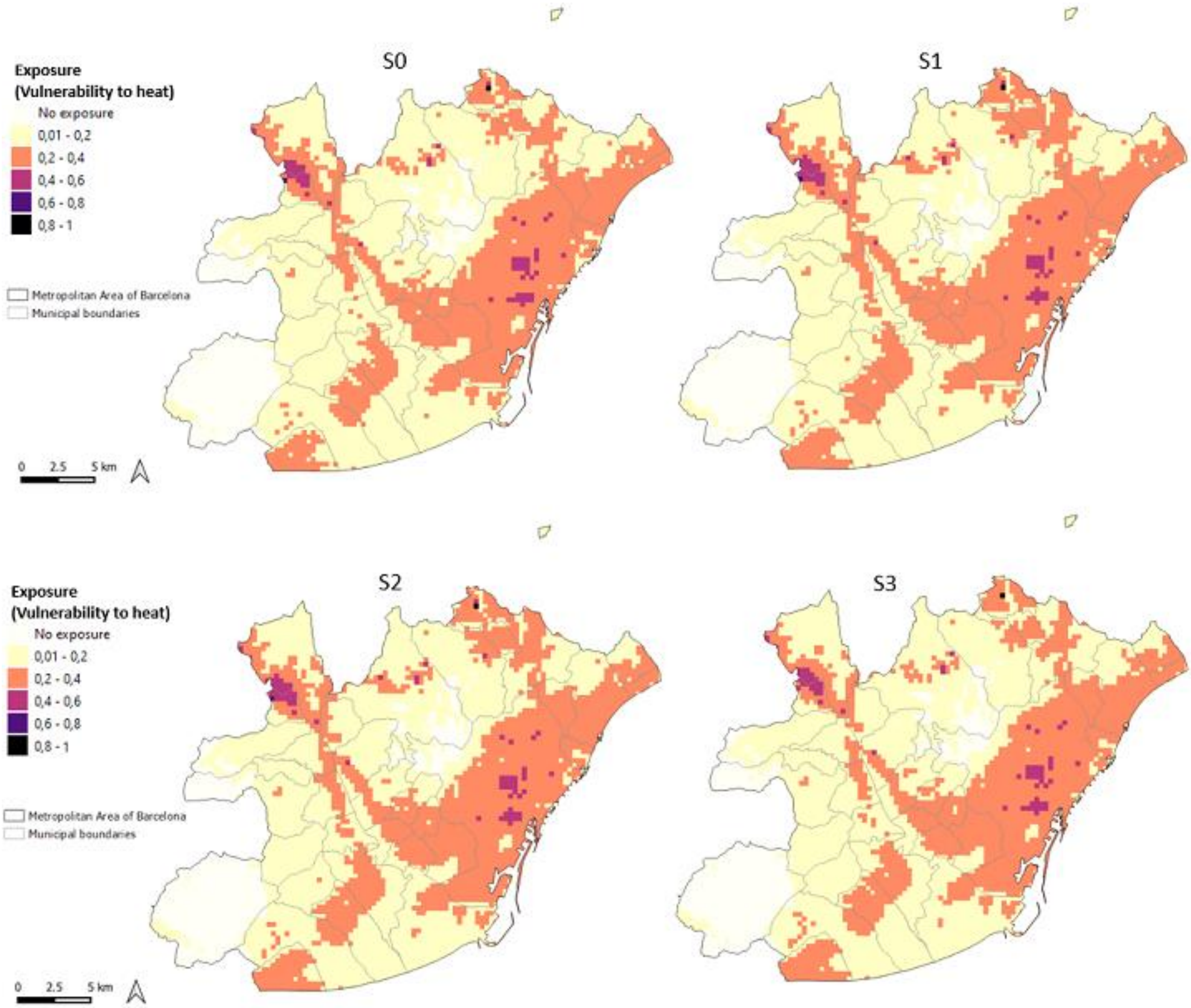


2.6. Change in normalized Heatwave night temperatures (Exposure)

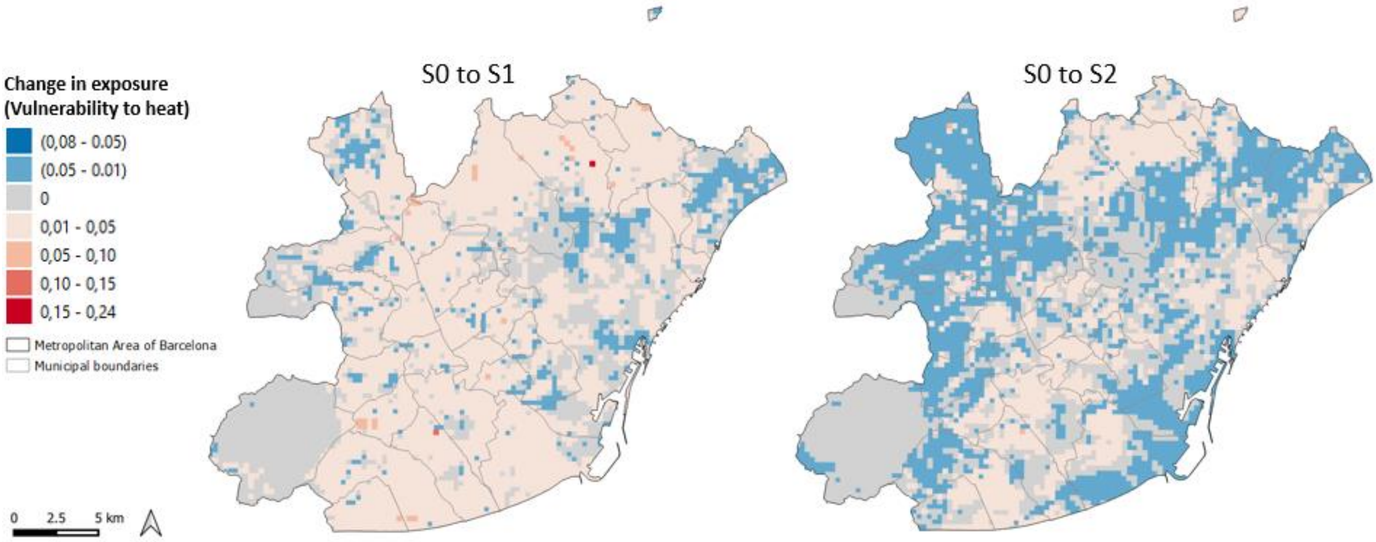


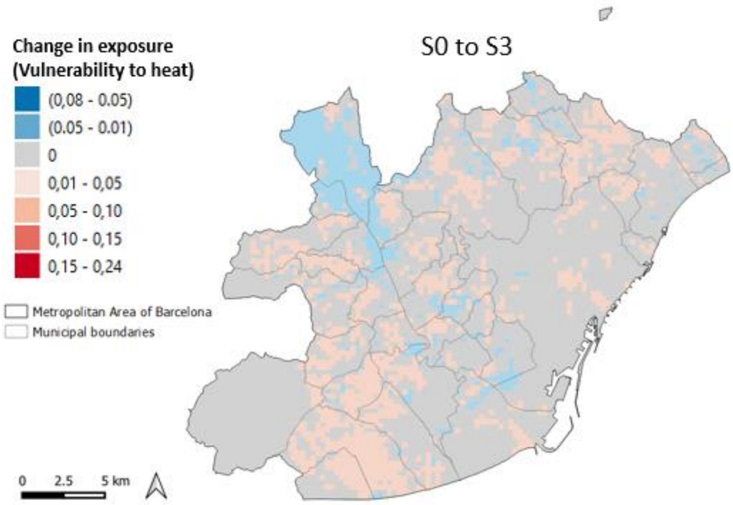


2.7. Exposure of Vulnerability to heat

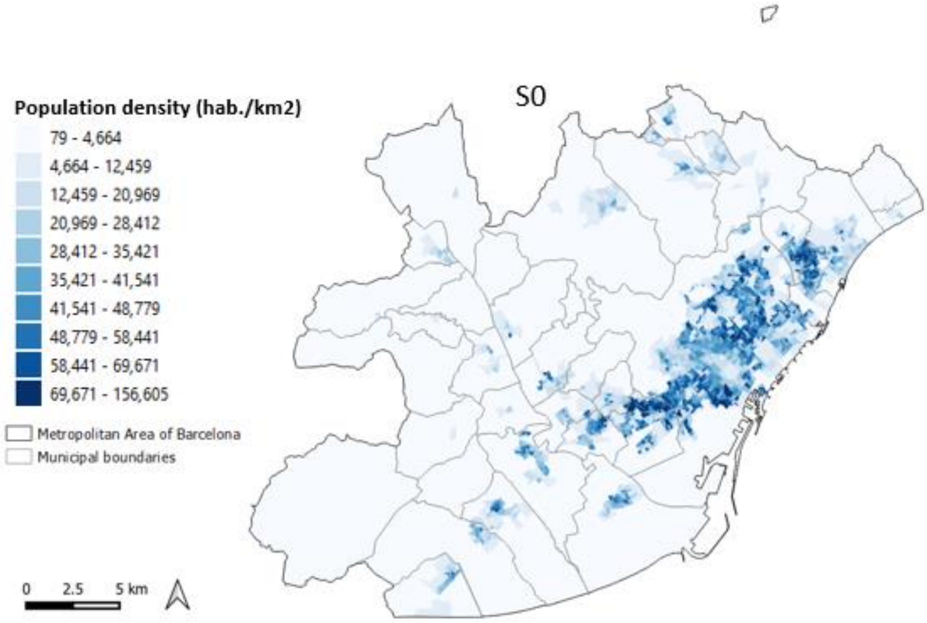


2.8. Change in exposure of Vulnerability to heat

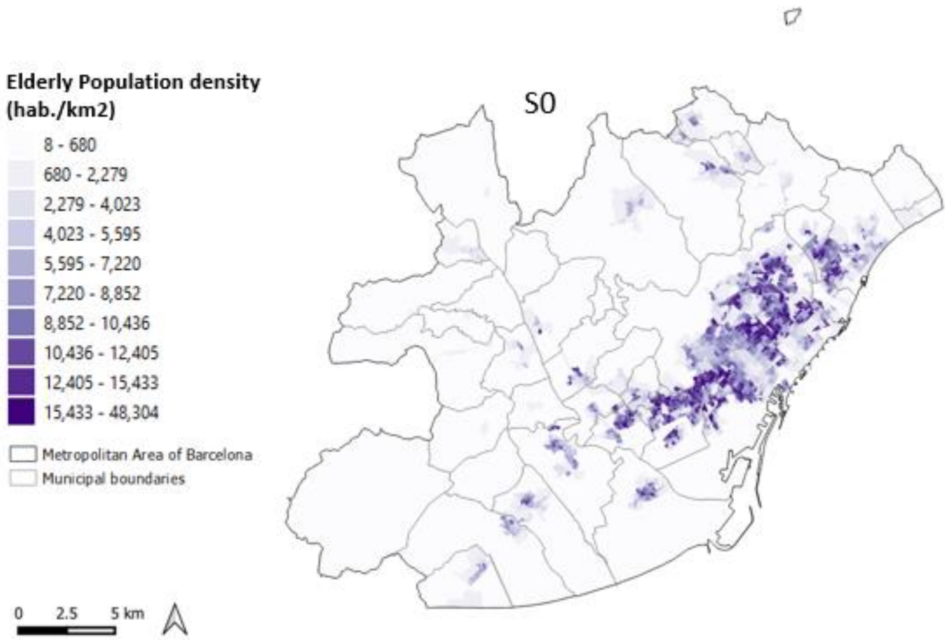




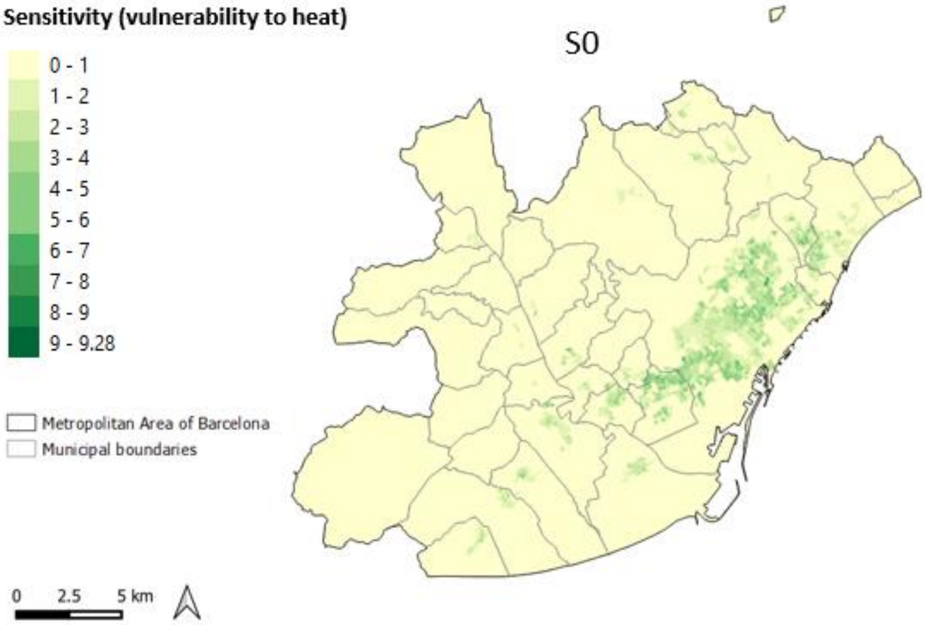
2.9. Population density (sensitivity)



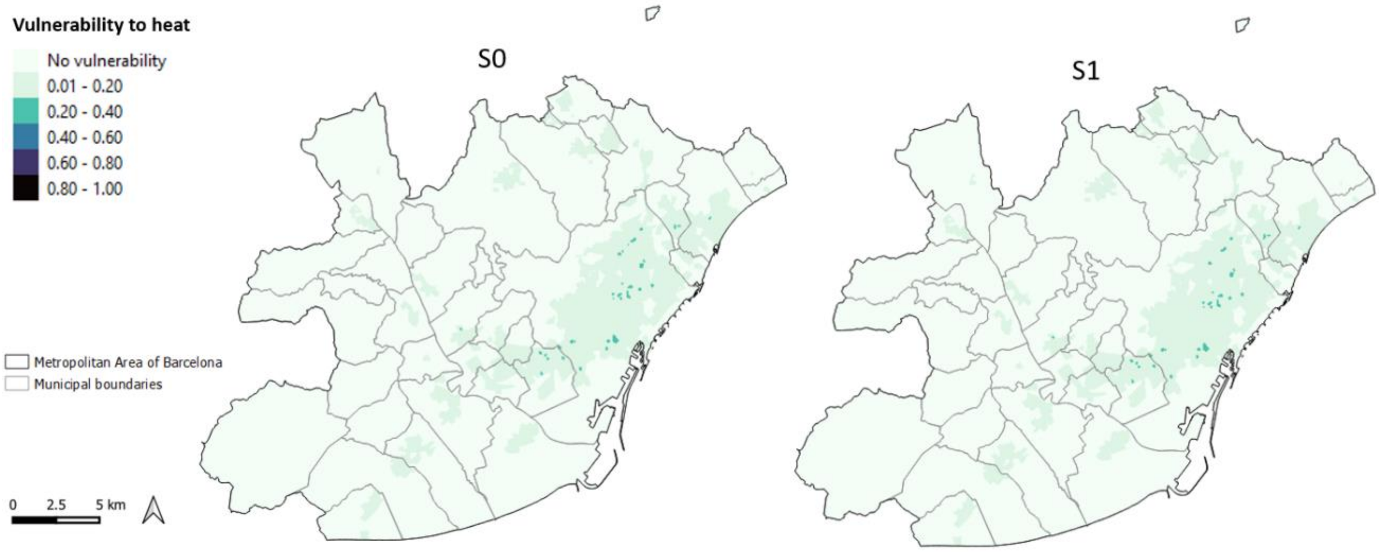
2.10. Elderly population density (sensitivity)

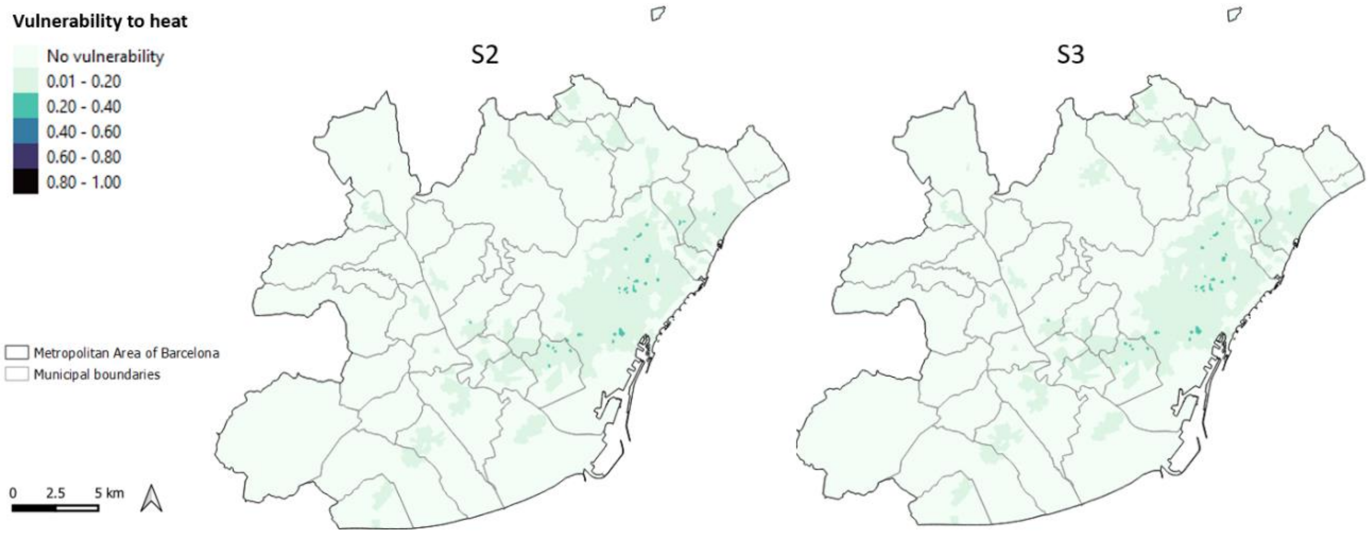


2.11. Sensitivity of Vulnerability to heat



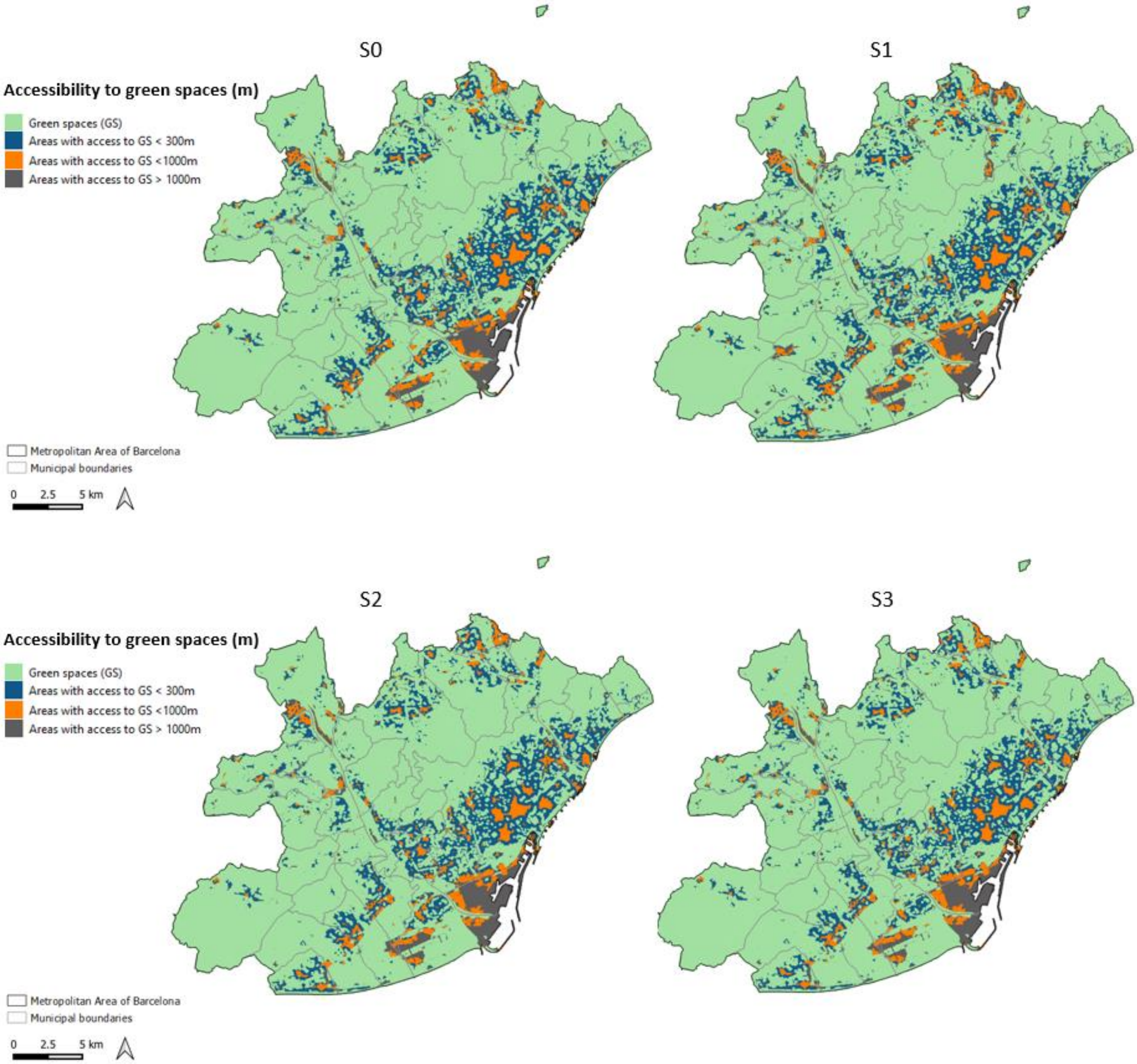
2.12. Vulnerability to heat (single vulnerability)



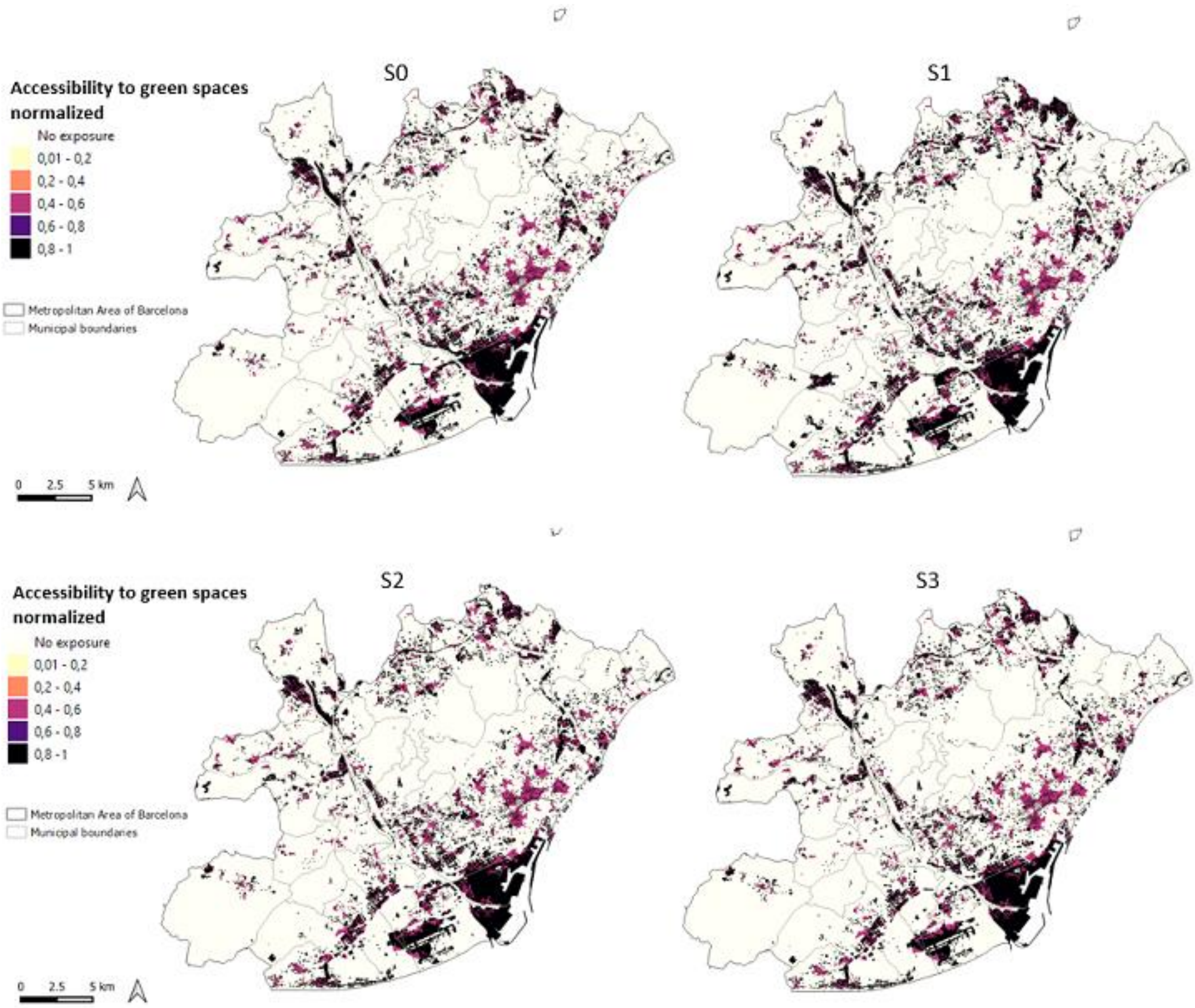


3. Vulnerability of lacking recreational space

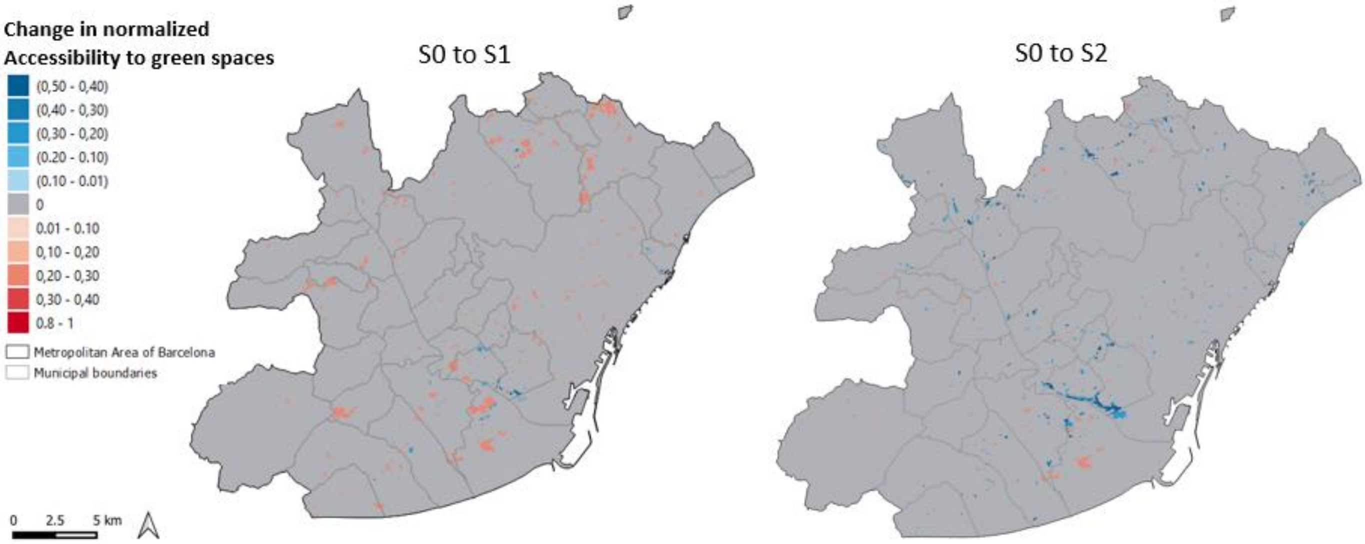
3.1. Accessibility to green spaces at less than 300m, less than 1000m and more than 1000m (exposure)

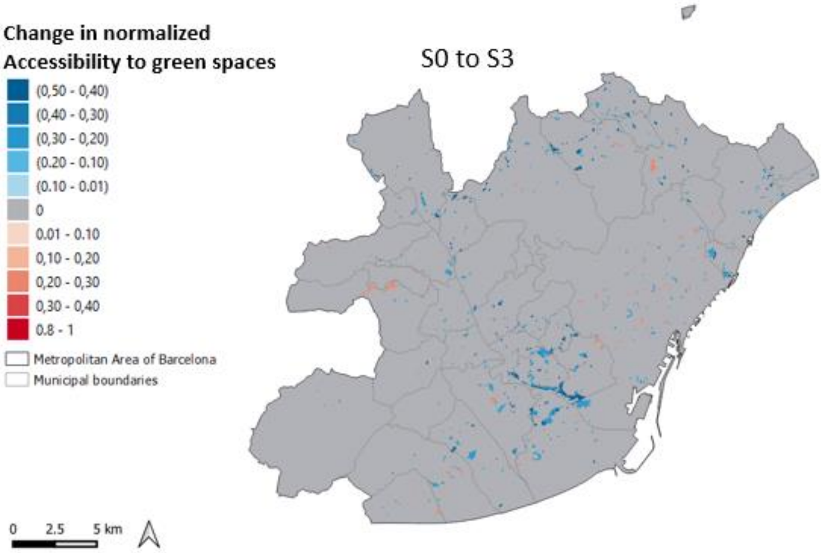


3.2. Accessibility to green spaces at less than 300m, less than 1000m and more than 1000m normalized (exposure)

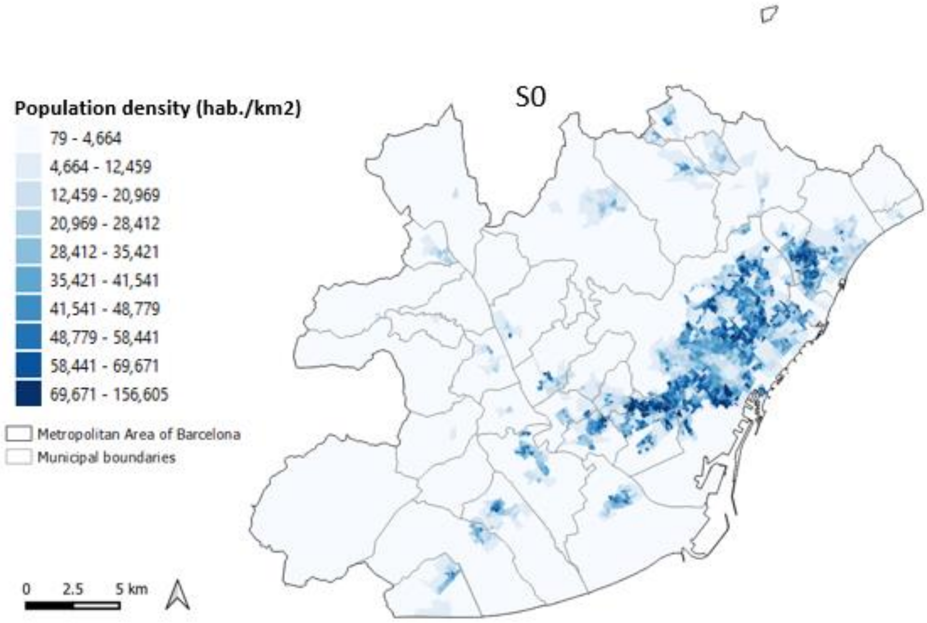


3.3. Change in normalized accessibility to green spaces at less than 300m, less than 1000m and more than 1000m normalized (exposure)

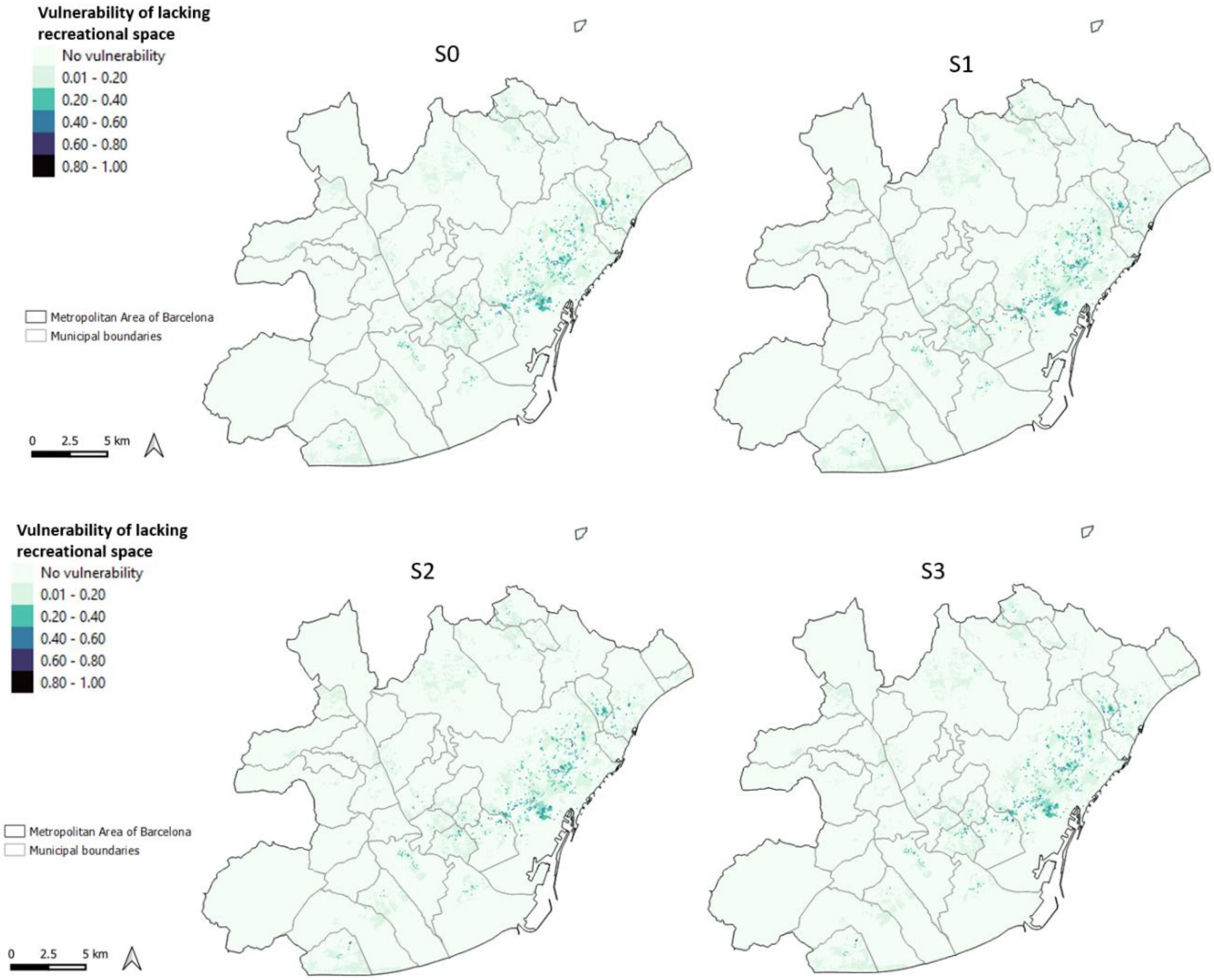




3.4. Population density

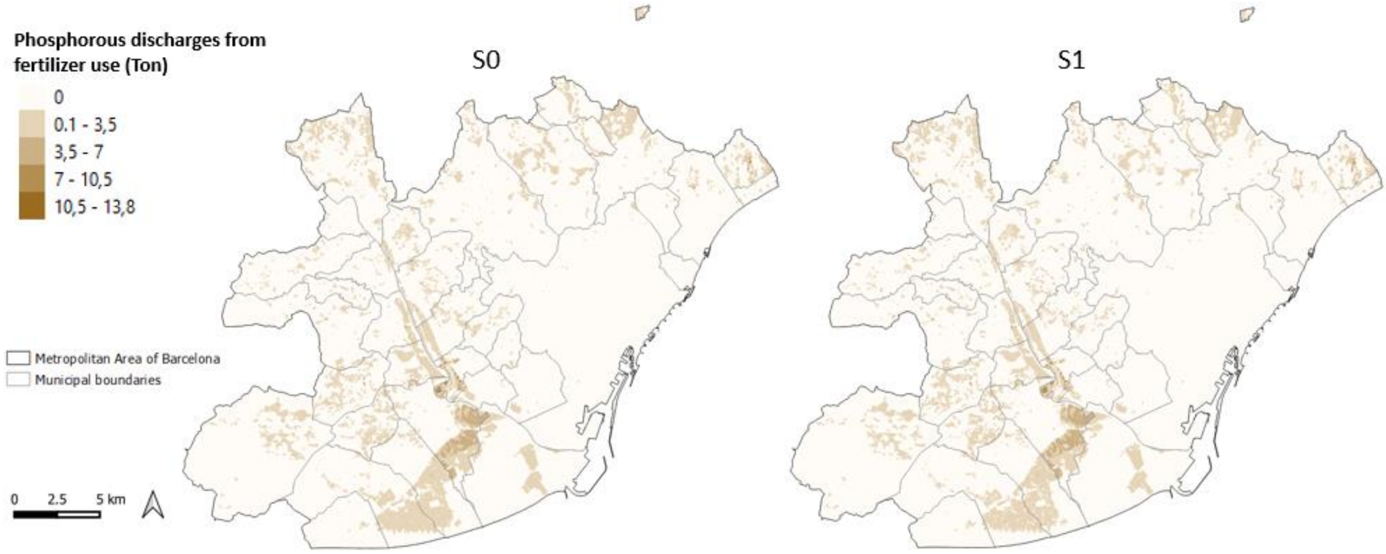


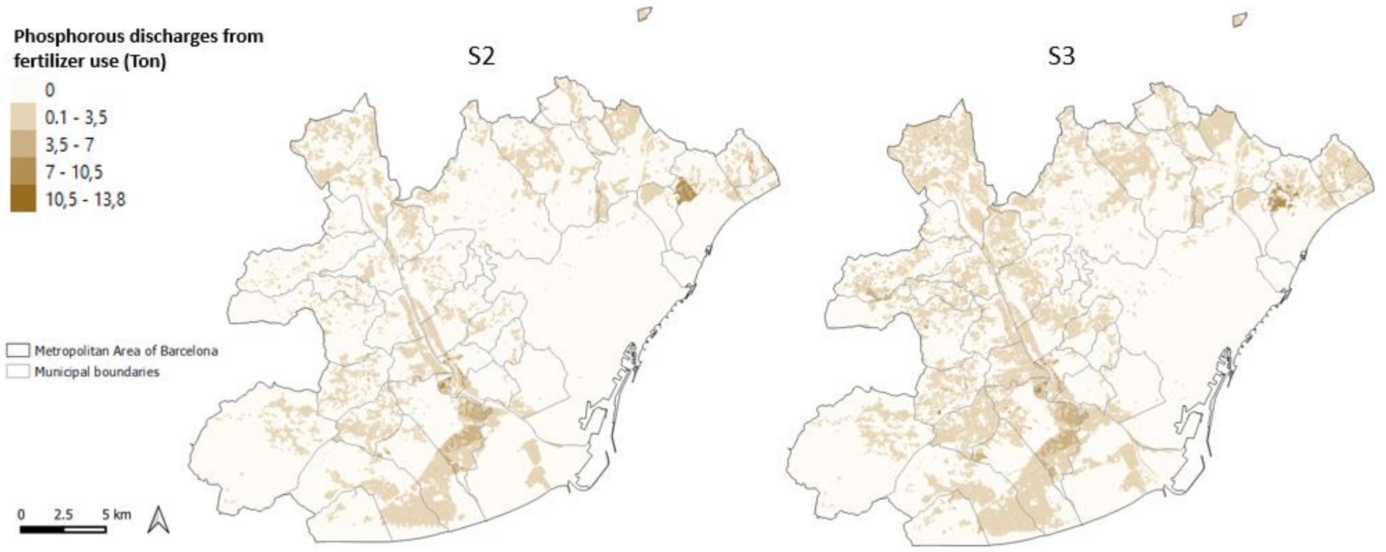
3.5. Vulnerability of lacking recreational spaces



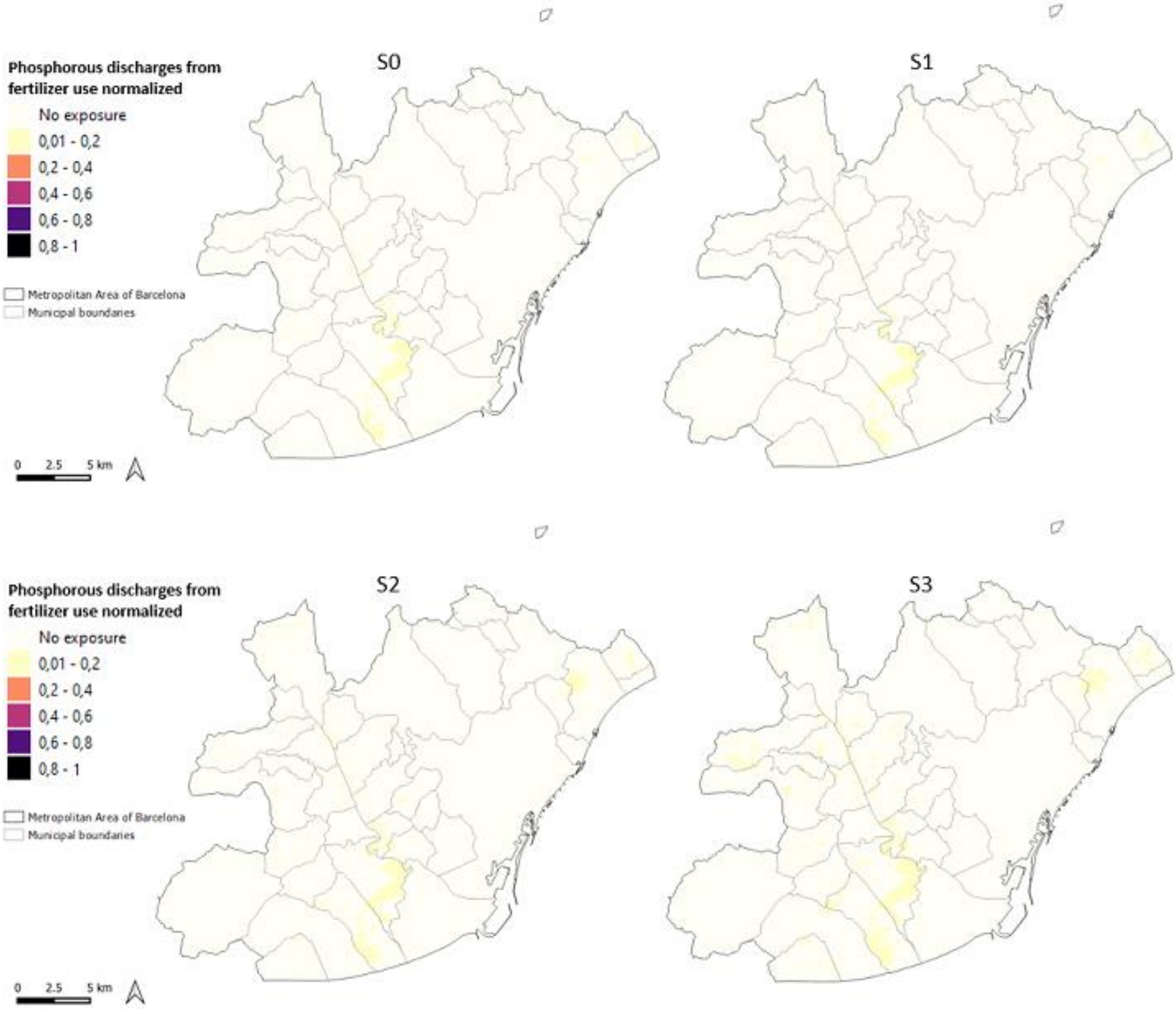
4. Vulnerability of loss of biodiversity

4.1. Phosphorous discharges from fertilizer use (exposure)

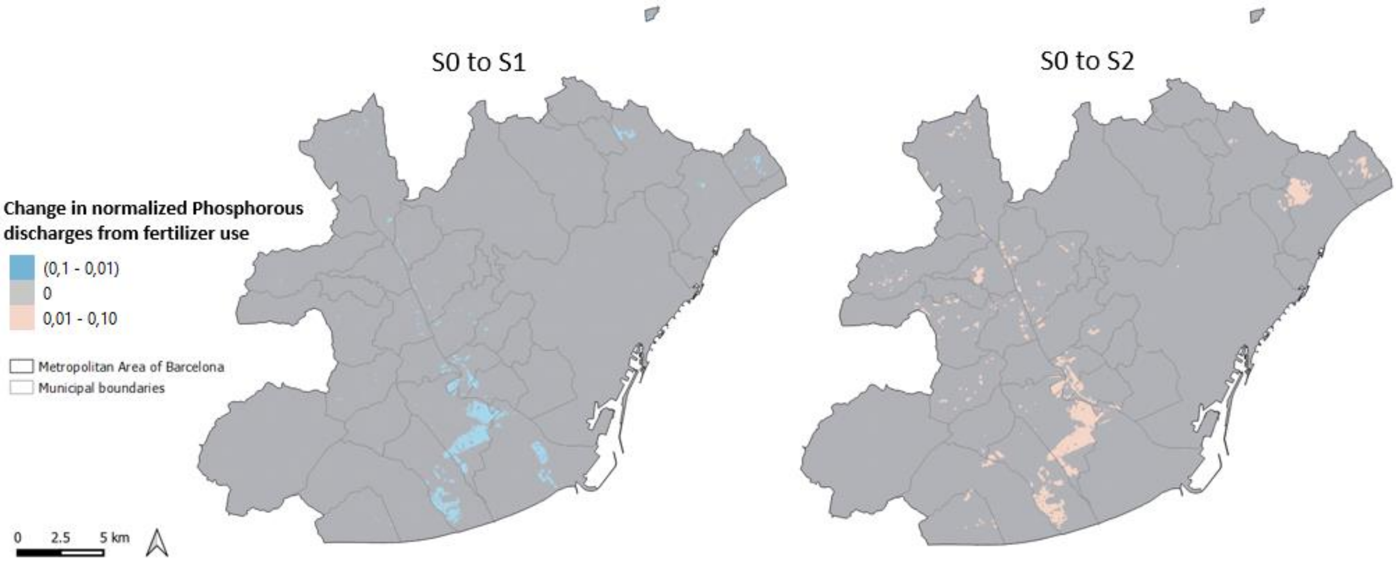


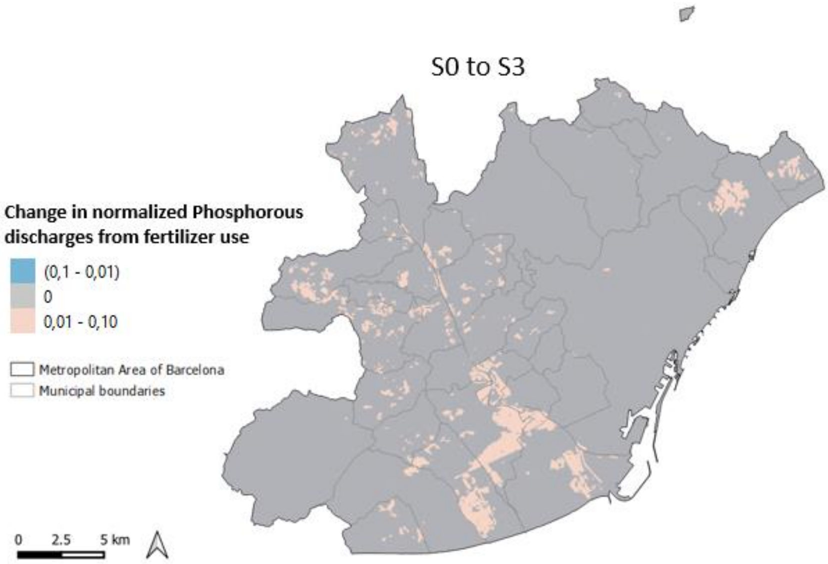


4.2. Phosphorous discharges from fertilizer use normalized (exposure)

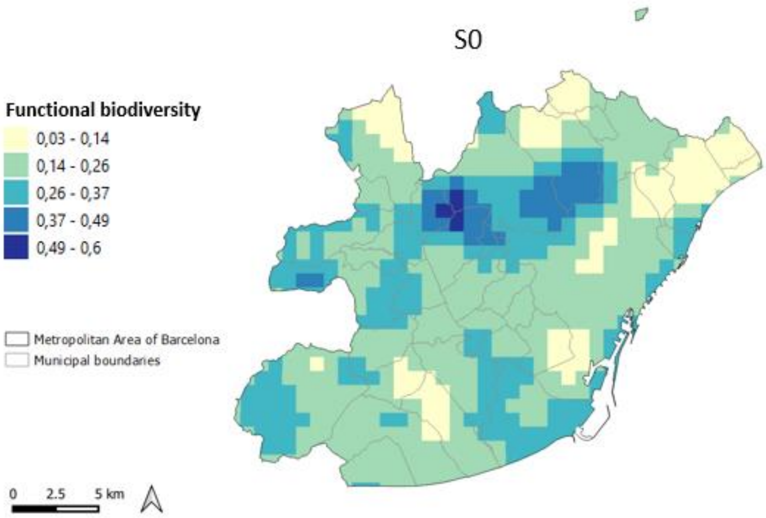


4.3. Change in phosphorous discharges from fertilizer use normalized (exposure)

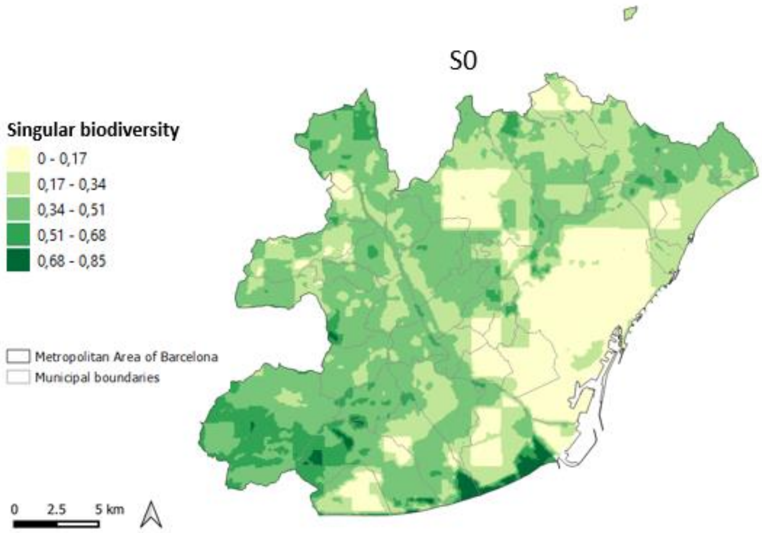




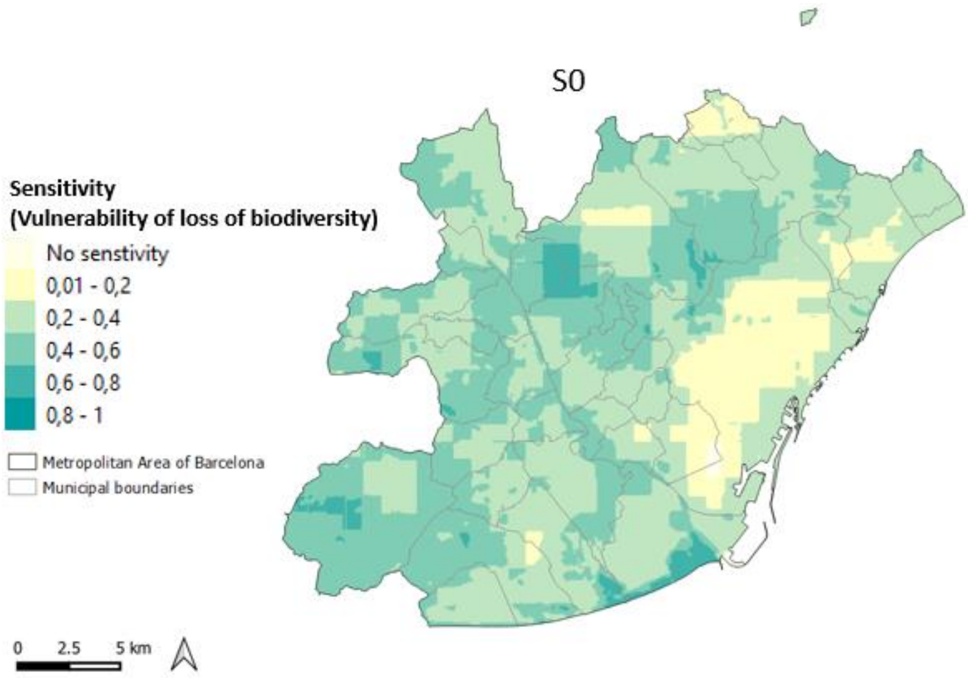
4.4. Functional diversity (sensitivity)



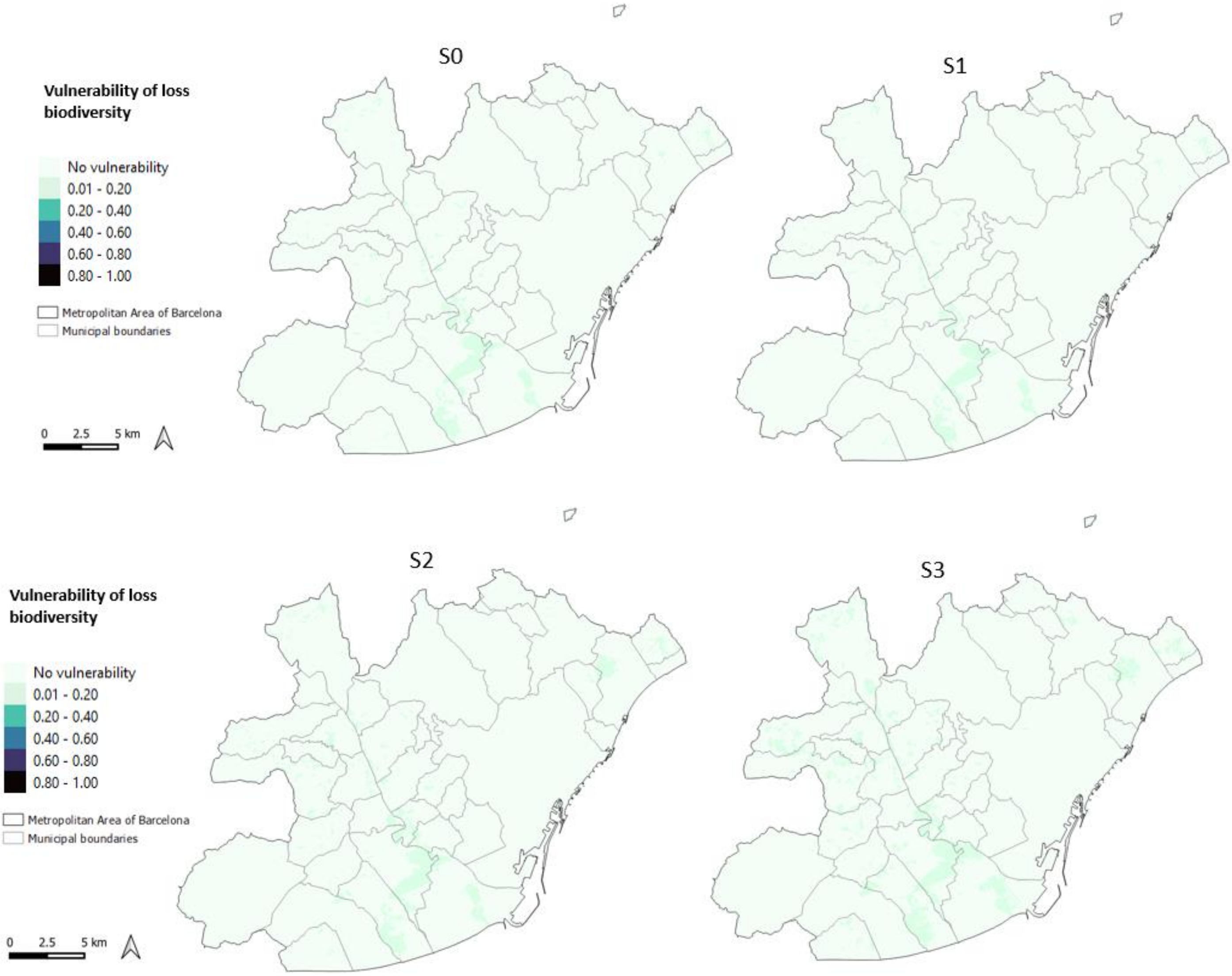
4.5. Singular biodiversity (sensitivity)



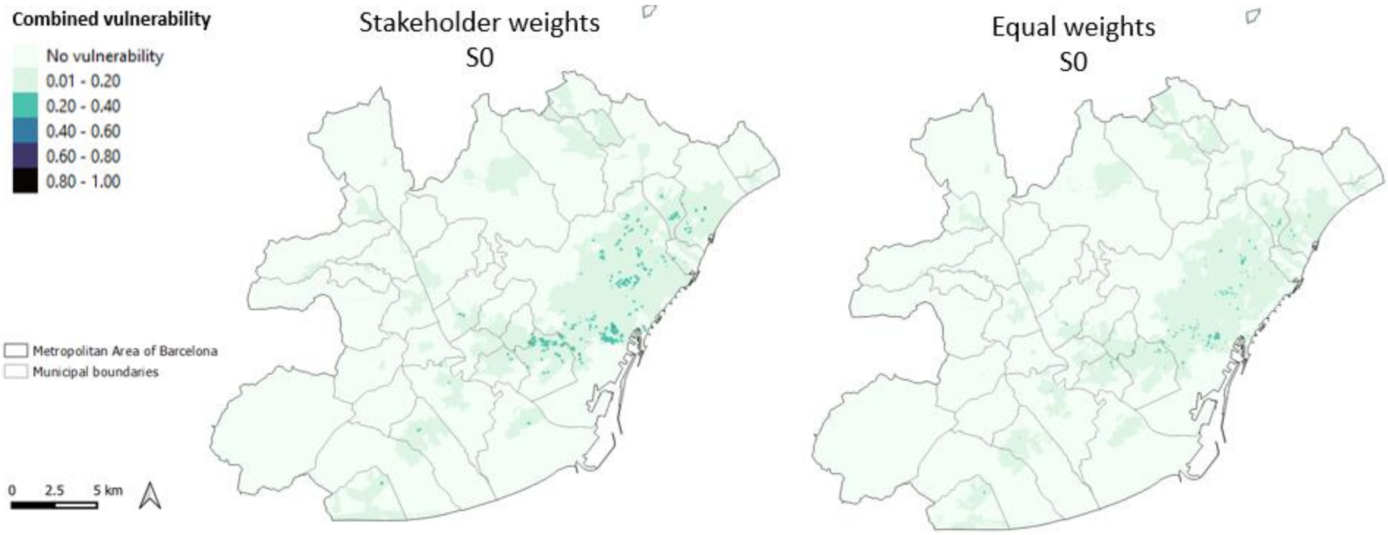
4.6. Sensitivity of Vulnerability of loss of biodiversity

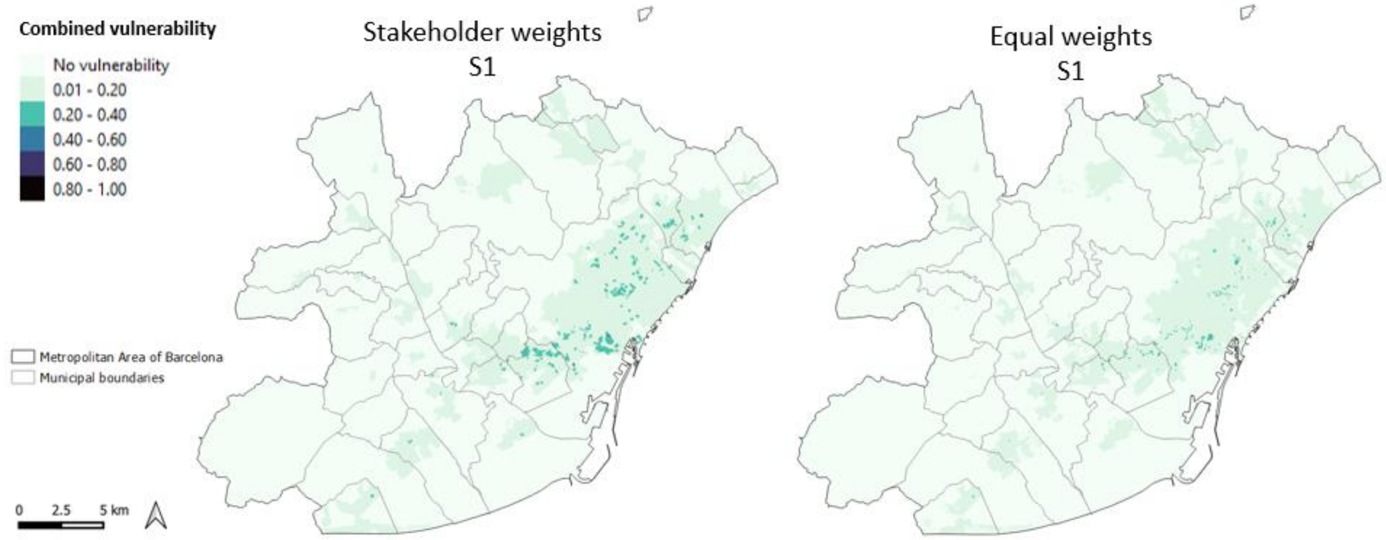


4.7. Vulnerability of loss of biodiversity

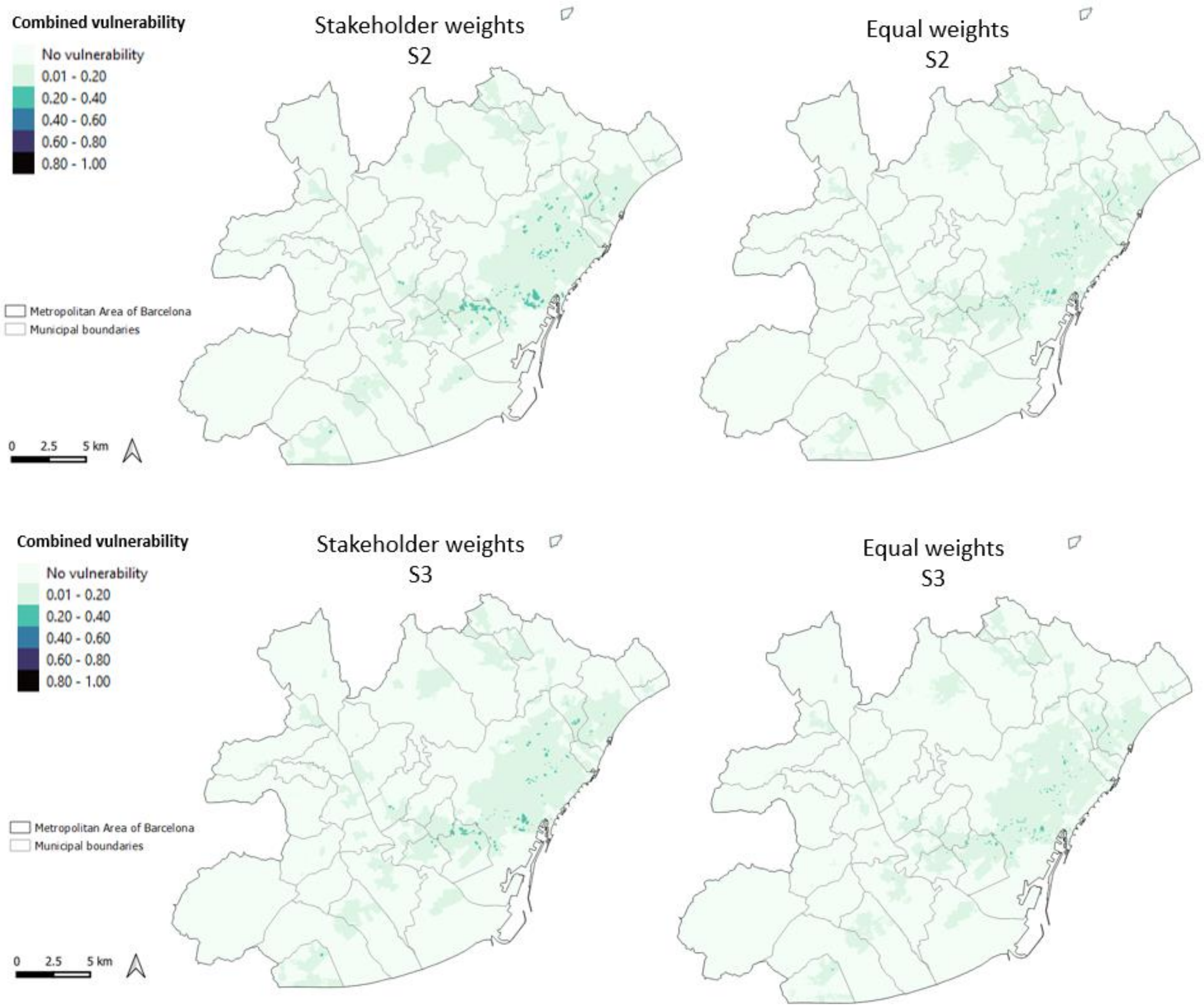


5. Combined vulnerabilities

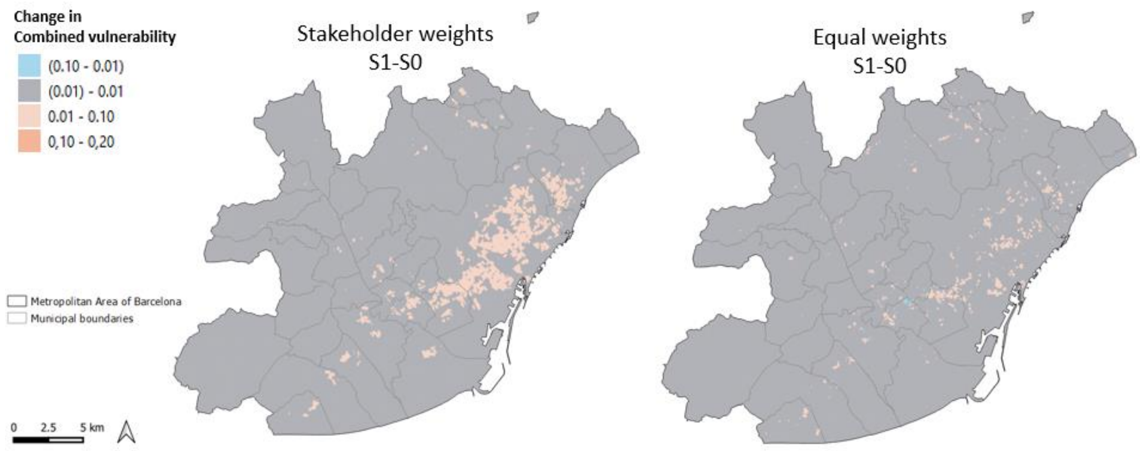


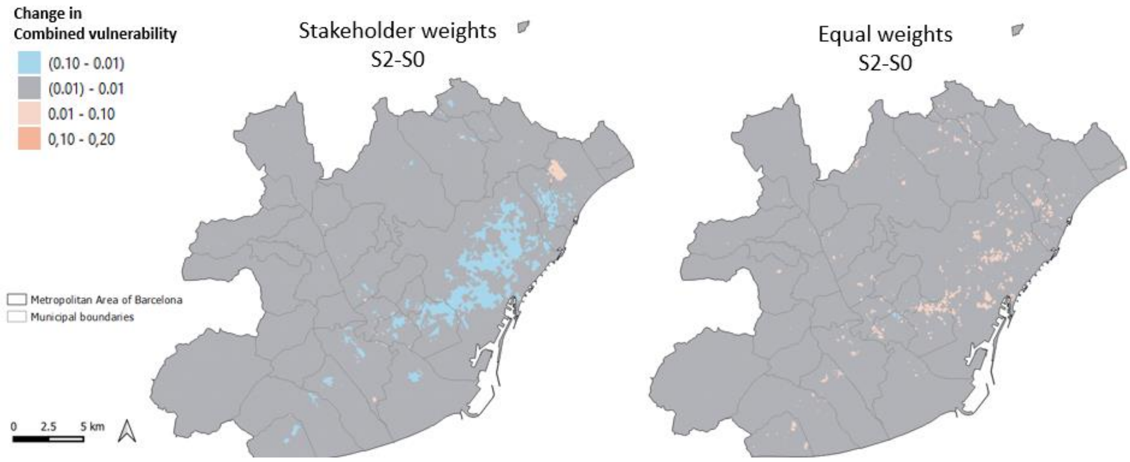


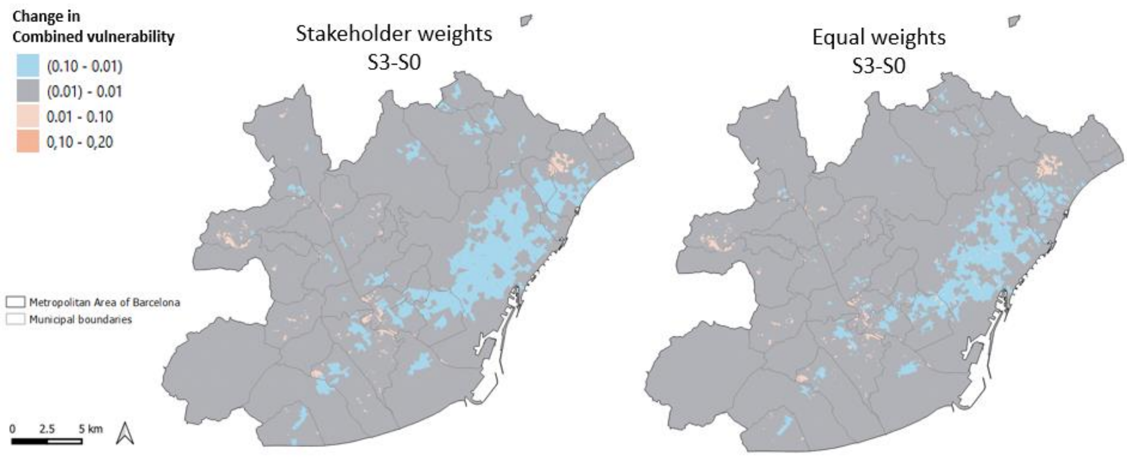
6. Changes in combined vulnerabilities



6.1. Change in Combined vulnerabilities







1-D. Supplementary table

Table 2. Sum of pixel values and its minimums and maximums for all scenarios, both for single vulnerabilities and combined vulnerability.

		S0. Current			S1. Trending			S2. Alternative			S3. Potential		
	Evaluation schemes	**Sum**	**Min**	**Max**	**Sum**	**Min**	**Max**	**Sum**	**Min**	**Max**	**Sum**	**Min**	**Max**
Combined vulnerabilities	Stakeholder weighting	3,525.39	-	0,423	3,635.23	-	0,435	3,313.89	-	0,399	2,998.52	-	0,363
	Equal weighting	2,421.68	-	0,397	2,482.74	-	0,402	2,314.91	-	0,386	2,154.39	-	0,371
	Vulnerability	**Sum**	**Min**	**Max**	**Sum**	**Min**	**Max**	**Sum**	**Min**	**Max**	**Sum**	**Min**	**Max**
Single vulnerabilities	Vulnerability of lack of local food	6,472.60	-	0,79	6,697.93	-	0,81	6,017.29	-	0,735	5,331.68	-	0,65
	Vulnerability to heat	2,230.00	-	0,31	2,251.41	-	0,31	2,235.02	-	0,311	2,239.19	-	0,31
	Vulnerability of lacking recreational space	945.05	-	0,74	950.14	-	0,74	942.01	-	0,744	925.61	-	0,74
	Vulnerability of loss of biodiversity	39.06	-	0,02	31.49	-	0,02	65.32	-	0,03	121.08	-	0,05

Appendix 2. Supplementary data for Chapter IV

2-A. Methodological details

1. Further considerations of the step-wise approach

1.1. Development of scenarios

NBS scenarios are developed to represent various land-use configurations specific to the urban environment under study. Within the MCDA methodology, scenarios — or "alternatives" — are useful for exploring potential future states of the environment in situations marked by uncertainty (Marttunen et al., 2017). The proposed framework starts by developing potential configurations of NBS in the form of land-use-change maps, to later contrast how the vulnerabilities shift when compared to a reference scenario. These maps require an appropriate resolution to accommodate vulnerabilities with different spatial patterns.

1.2. Planetary boundaries

Table A1. List of planetary boundaries and their current status

Earth-system process	Control variable	Boundary value in 2023	"Current" value (i.e. for the year provided in the source)	Boundary now exceeded beyond the 2023 values? (based on "current" value)	Preindustrial Holocene base value
1. Climate change	Atmospheric carbon dioxide concentration (ppm by volume)	350	417	yes	280
	Total anthropogenic radiative forcing at top-of-atmosphere (W/m2) since the start of the industrial revolution (~1750)	1	2.91	yes	0
2. Change in biosphere integrity	Genetic diversity: Extinction rate measured as E/MSY (extinctions per million species-years)	<10 E/MSY but with an aspirational goal of ca. 1 E/MSY (assumed background rate of extinction loss)	>100 E/MSY	yes	1 E/MSY
	Functional diversity: energy available to ecosystems (NPP) (% HANPP)	HANPP (in billion tonnes of C year−1) <10% of preindustrial Holocene NPP, i.e., >90% remaining for supporting biosphere function	30% HANPP	yes	1.9% (2σ variability of preindustrial Holocene century-mean NPP)
3. Modifications of Biogeochemical flows	Phosphate global: P flow from freshwater systems into the ocean; regional: P flow from fertilizers to erodible soils (Tg of P year−1)	Phosphate global: 11 Tg of P year−1; regional: 6.2 Tg of P year−1 mined and applied to erodible (agricultural) soils.	Global: 22 Tg of P year−1; regional: 17.5 Tg of P year−1	yes	0
	Nitrogen global: industrial and intentional fixation of N (Tg of N year−1)	62	190	yes	0

Table A1. List of planetary boundaries and their current status (continuation)

Earth-system process	Control variable	Boundary value in 2023	"Current" value (i.e. for the year provided in the source)	Boundary now exceeded beyond the 2023 values? (based on "current" value)	Preindustrial Holocene base value
4. Ocean acidification	Global mean saturation state of calcium carbonate in surface seawater (omega units)	2.75	2.8	no	3.44
5. Land use	Part of forests rested intact (percent)	75 from all forests including 85 from Boreal forest, 50 from Temperate forests and 85 from Tropical forests	Global: 60	yes	100
6. Freshwater change	Blue water: human induced disturbance of blue water flow	Upper limit (95th percentile) of global land area with deviations greater than during preindustrial, Blue water: 10.2%	18.20%	yes	9.40%
	Green water: human induced disturbance of water available to plants (% land area with deviations from preindustrial variability)	11.10%	15.80%	yes	9.80%
7. Ozone depletion	Stratospheric ozone concentration (Dobson units)	276	284.6	no	290
8. Atmospheric aerosols	Interhemispheric difference in AOD	0.1 (mean annual interhemispheric difference)	0.076	no	0.03
9. Novel entities	Percentage of synthetic chemicals released to the environment without adequate safety testing	0	Transgressed	yes	0

1.3. Normalization of indicators

Involves normalizing absolute values of exposure and sensitivity indicators to create a unified scale, enabling integration across different measurement units. Thresholds are included to determine the magnitude of the NBS impacts based on the selected indicators. Thresholds serve as cutoff values and are established based on scientific knowledge or urban objectives. Thresholds are context-specific, reflecting the urban environment where NBS are situated, enabling risk differentiation based on local conditions. For instance, the threshold for what is considered a heatwave can vary by region due to differing meteorological conditions (Kovats & Kristie, 2006). By the end of this stage, all indicator absolute values are transformed to a uniform scale ranging from 0 to 1.

1.3.1. Normalization of local-scale indicators

Involves normalizing absolute values of exposure and sensitivity indicators to create a unified scale, enabling integration across different measurement units. Thresholds are included to determine the magnitude of the NBS impacts based on the selected indicators. Thresholds serve as cutoff values and are established based on scientific knowledge or urban objectives. Thresholds are context-specific, reflecting the urban environment where NBS are situated, enabling risk differentiation based on local conditions. For instance, the threshold for what is considered a heatwave can vary by region due to differing meteorological conditions (Kovats & Kristie, 2006). By the end of this stage, all indicator absolute values are transformed to a uniform scale ranging from 0 to 1.

1.3.2. Aggregation of indicators for single local-scale indicators

The normalized exposure and sensitivity indicators of each vulnerability are aggregated to obtain single vulnerability maps, which identify urban areas experiencing exposure and sensitivity simultaneously. Aggregation is employed for representing multidimensional realities through single indexes (OECD & European Union, 2008). In this case, aggregation is necessary to sum the indicators per vulnerability, resulting in a single map.

2. Scenarios – methodological considerations

S0. *Reference* was mapped based on an aerial photo survey that the municipality of Oslo carried on to trace the state of green roofs in 2017 (Oslo Kommune, 2021a). Out of 950 green roofs, 928 were within the city limits of the case study. The survey does not differentiate between the typology of green roofs (e.g., extensive, intensive) so it is assumed that all are extensive.

S1. *Green roof strategy* considers the increase in the number of green roofs based on the estimations of the strategy for green roofs and façade (Oslo Kommune, 2022), which states that the city will count with a total of 2030 green roofs and facades by 2030. In this study, I am assuming that the 2030 infrastructures will be all extensive green roofs and that these will maintain the average size (m^2) observed in S0.

S2. *Ambitious*: represents a greater implementation of green roofs in the municipality compared to S1. This number can be backed up by the increasing demand for green roofs because of the necessity for stormwater management (URBAG, 2021). The scenario may seem ambitious in the present but may become less significant in the future as the market keeps growing.

S3. *Maximization*: focuses on the hypothetical case of creating green roofs in all available rooftops of the city. Rooftops were selected using the cadaster of the city and chosen based on the conditions described in the strategy for green roofs and facades (Oslo Kommune, 2022) of buildings with more than $10m^2$ of rooftop area and a slope no bigger than 30°. The number of green roofs reached 56,786 and a total of 10,039,747 m^2.

New green roofs for S1 and S2 were chosen in rooftops available in the city, with no preexisting green roofs. Available rooftops were selected using the cadaster of the city based on the conditions described in the strategy for green roofs and facades (Oslo Kommune, 2022) of buildings with more than $10m^2$ of rooftop area and slope no bigger than 30°.

The location and size selection for the new green roofs in scenarios S1 and S2 were based on maintaining the spatial distribution and average size observed in scenario S0. Initially, the number of existing green roofs in each Oslo district (Bydel) was determined, distinguishing between "small" (<250 m^2) and "big" (>=250 m^2) roofs as per municipal criteria (Oslo Kommune, 2022). Subsequently, the total area (m^2) for scenarios S1 and S2 was calculated by multiplying the number of green roofs chosen for each scenario by the average area of green roofs in S0 (refer to **Table A2**).

Following this, the total area for S1 and S2 was distributed across Oslo districts following the same distribution of green roofs m^2 by districts found in S0. A similar procedure was applied to allocate the quantity of big and small roofs in each district as observed in S0.

Finally, the specific amount of green roofs m^2 calculated for each district was randomly assigned to available rooftops within each district, with the condition of maintaining the proportion between small roofs and big roofs. This allocation process was carried out using the OpenSolver add-in for Excel. Specifically, in S1, existing green roofs from S0 were maintained while new green roofs were added, whereas in S2, existing green roofs from both S0 and S1 were preserved before introducing new green roofs. This means that S1 includes green roofs from S0, while S2 includes green roofs from S0 and S1. Results were then mapped using QGIS 3.28.0-Firenze.

Table A2. Detail of the number of green roofs, total extension, average size and percentage occupation out of the total potential green roofs for each of the scenarios

Scenario	Number	ha	m²	Average m²	% of potential green roofs
S0. Reference	928	18.99	189,927	205	2%
S1. Green roof strategy	2,030	41.55	415,525	205	4%
S2. Ambitious	3,550	72.93	729,333	205	7%
S3. Maximization	56,786	1,039.13	10,391,269	183	100%

3. Local-scale vulnerabilities and indicators – description and calculation

QGIS 3.28.0-Firenze was utilized to produce and manage all maps and indicators. Scenarios and indicators were transformed into a 50x50m grid, enabling (1) detailed examination of land-use changes across scenarios within the extension of the Oslo Municipality, (2) aggregation of various indicators, and (3) management of datasets within the data processing capabilities

3.1. Vulnerability of lack of habitats for pollinators

Oslo's location, with a short distance between the fjord and Marka and special ecological conditions, makes it the municipality in Norway with the highest record of biodiversity (Oslo Kommune, 2023a). Oslo is actively working on preserving biodiversity conditions (Oslo

Kommune, 2023a), considering that many biologically valuable habitats will be exposed to strong urban development pressure due to population growth in the city (Oslo Kommune, 2023a). In this sense, green roofs can improve urban conditions by increasing the presence and foraging of pollinators (Kratschmer et al., 2018; Passaseo et al., 2020).

To evaluate this vulnerability, I include the *Pollinator habitat suitability* as the exposure indicator, and *Precautionary zones for honeybee keeping* and *Areas with presence of red listed bee species* as sensitivity indicators.

3.1.1. Pollinator habitat suitability (exposure)

The calculation of this indicator begins with the ESTIMAP pollinator habitat developed by (Stange et al., 2017). In this framework, stakeholders evaluated various land cover and vegetation cover categories within the city of Oslo and assigned habitat suitability scores to the intersections of these categories.

The ESTIMAP pollinator habitat map, with a resolution of 10x10m, displays scores (on the scale of 0-10) based on the capacity the different land uses to provide habitat suitability to pollinators. To adapt the ESTIMAP layer to the presence of green roofs scenarios, first, there were found those sections of the ESTIMAP grid that intersected with the presence of green roofs in each scenario. For those sections, the scores were updated based on habitat suitability for land cover categories (see **Table A3**). The update was done by assuming that those 10x10 grid cells whose area was covered by at least 50% of green roofs were increasing their habitat suitability. For doing this the score was increased from 0.1 to 0.7, assuming that green roofs shifted the habitat suitability from "built" to "low vegetation" within the category of "medium built-up areas". This was done for all the grid cells experiencing the presence of green roofs for each of the scenarios.

The next step after the pollinator suitability values were updated was to normalize the values. Normalization was made based on the premise that urban vehicle presence negatively impacts pollinator foraging as the capacity of bees to sense floral odors can be compromised by emissions from vehicles (Girling et al., 2013). Also, as traffic speeds escalate, there is a corresponding reduction in pollination rates (Dargas et al., 2016) and the interaction with vehicles might result in elevated bee mortality (Kallioniemi et al., 2017). To capture these detrimental effects, it was followed the methodology of Stange et al. (2017), by reducing the pollinator suitability values based on their proximity to aboveground, high-traffic roads. To do so, first, it was employed a raster layer produced by Stange et al. (2017), portraying an exponential decay function, depicting how habitat suitability values reduce by 0.2 when located immediately adjacent to high-traffic roads, with the effect diminishing to zero at 200 m distances from road edges. Those cells with updated habitat values because of the implementation of green roofs, and being within 200m of high-traffic roads, were selected for reducing their habitat suitability value by the value obtained from the exponential decay function. For details more details on the methodology used for calculating the decay function around the aboveground, high-traffic roads, please refer to Stange et al. (2017).

Finally, to relate these values to the 50x50 grid cells, each 10x10 grid cell was associated with a corresponding 50x50 grid cell based on the greatest area overlap. Then, the value for each 50x50 grid cell was calculated by averaging the values of the 10x10 grid cells that overlapped with it.

Table A3. Habitat suitability scores for land cover categories produced by Stange et al. (2017)

	Pixel habitat suitability score based on Sentinel 2 satellite land cover classification				
Land cover category	Agricultural	Low vegetation	Tree	Built	Water
core FNF (forest with no floral resources)	0,1	0,4	0,1	0,1	0,05
core CO (conifer forest)	0,2	0,6	0,3	0,1	0,05
core OF (other forest)	0,3	0,5	0,4	0,1	0,05
core MFL (mixed forest low)	0,4	0,6	0,5	0,1	0,05
core MFH (mixed forest high)	0,3	0,8	0,6	0,1	0,05
core BLF (broad leaf forest)	0,3	0,7	0,6	0,1	0,05
core FYF (forest with floral resources)	0,4	0,7	0,7	0,1	0,05
edge FNF (forest with no floral resources)	0,5	0,4	0,3	0,1	0,05
edge CO (conifer forest)	0,6	0,9	0,8	0,2	0,05
edge OF (other forest)	0,5	0,7	0,6	0,2	0,05
edge MFL (mixed forest low)	0,6	1	0,9	0,2	0,05
edge MFH (mixed forest high)	0,6	1	0,9	0,2	0,05
edge BLF (broad leaf forest)	0,7	1	1	0,2	0,05
edge FYF (forest with floral resources)	0,7	1	1	0,2	0,05
agricultural land	0,3	0,6	0,6	0,1	0,05
medium built areas	0,7	0,7	0,6	0,1	0,05
densely built areas	0,35	0,45	0,25	0,05	0,05
mines	0,35	0,55	0,35	0,05	0,05
graveyard	0,5	0,9	0,7	0,1	0,05
industrial	0,4	0,6	0,4	0,05	0,05
Transportation-infrastructures	0,5	0,8	0,5	0,1	0,05
Sports-stadiums	0,1	0,1	0,4	0,1	0,05
alpine ski area	0,4	0,5	0	0,1	0,05
parks	0,5	0,6	0,8	0,1	0,05
golf course	0,4	0,5	0,7	0,1	0,05
pastures	0,2	0,3	0,6	0,2	0,05
semi-natural vegetation	0,7	1	0,8	0,2	0,05
open areas	0,7	1	0,8	0,2	0,05
bogs	0,4	0,4	0,3	0,1	0,05
freshwater	0,4	0,2	0,4	0,1	0,05
ocean	0,3	0,3	0,3	0,05	0,05

3.1.2. Precautionary zones for honeybee keeping (sensitivity) Precautionary zones for honeybee keeping (sensitivity)

Legislation in Norway (Oslo Kommune, 2023a) mandates that governmental entities, including the Oslo municipality, take measures to prevent the loss of both species and habitat types. Special attention should be given to rare and endangered species. To proactively prevent potential adverse impacts that elevated honeybee concentrations may pose to biodiversity of national and international significance, the Oslo Urban Environmental Agency has suggested the creation of eight "precautionary zones" within the municipality. These zones contain biologically important flowering meadows and are designed to "safeguard" butterfly and bee species from being close to honeybee concentrations. For this assessment, these areas are being included to account for urban areas that are sensitive to the well-being of wild pollinators, and therefore, would be sensitive to changes in the number of habitats for pollinators. Precautionary zones were assigned a value of 1 (high sensitivity), while all other areas were given a value of 0 (no sensitivity).

3.1.3. Areas with presence of red listed bee species (sensitivity)

To account for the actual presence of sensitive pollinators within the municipality of Oslo, six red listed bee families were selected (Andrenidae, Apidae, Colletidae, Halictidae, Megachilidae, Melittidae) according to European Red List of Bee species (Nieto, 2014). Then, it was mapped the locations where these species have been found or observed based on the Norwegian Biodiversity Information Centre within the municipality of Oslo since 2007 (Artsdatabanken, 2020). Each of the coordinates was then used to create a 250m buffer representing the foraging distance of wild bees (Venter et al., 2021). Such areas were highly sensitive and associated with a value of 1, while areas of the city not falling within these buffer zones were assigned a value of 0. These buffers were subsequently correlated to a 50x50 grid, where grid cells covered by more than 50% of buffer areas were considered highly sensitive (value of 1), while the rest were considered not sensitive (value of 0).

3.2. Vulnerability to heavy rainfall events

Green roofs have been described as a valuable option for stormwater management within urban environments as they are able to retain rainwater for enough time to delay peak water discharges (Shafique et al., 2018). Norway anticipates an increase in the periods of heavy local precipitation, which will add pressure to the stormwater management systems across their cities (Norwegian Directorate for Civil Protection, 2019). Oslo municipality is preparing for such climatic conditions, that, coupled with a growing population and the development of more densely populated urban areas, are expected to heighten its *vulnerability to heavy rainfall events* (Oslo Kommune, 2023b). In this sense, green roofs offer solutions for reducing urban runoff due to their capacity to retain during rain periods (Oslo Kommune, 2012).

To assess the exposure to heavy rainfall events, the indicator selected was the runoff coefficients observed for an average annual rainfall of 800 mm and a storm event with a one in

10-year recurrence. The sensitivity indicators selected were areas with the presence of critical infrastructure, population density, elderly population density and low-income households.

3.2.1. Runoff coefficient (exposure)

In order to calculate the runoff coefficient for the whole Oslo municipality, first, it was employed the model developed by Sælthun et al. (2021) for calculating the urban runoff values under an annual rainfall of 800 mm and a storm event with a one in 10-year recurrence and no presence of green roofs. This step produced a map with the cumulative runoff, expressed in liters per second (l/s), across various land cover areas, each characterized by heterogeneous sizes.

The next step in the proposed approach involved subdividing these diverse land cover regions into a 50x50 grid. For every grid cell, the total runoff (l/s) was computed by aggregating the runoff linked to the square meters of each type of land cover within that specific cell. To accomplish this, it was determined the runoff (l/s) corresponding to each land cover within a grid cell based on its square meters. For example, consider a land cover segment spanning 2,772 m^2 with an associated runoff of 128.81 (l/s). This segment was partitioned by the 50x50 grid into two portions: the first comprising 1,334 square meters and the second comprising 1,438 m^2. The original runoff of 128.81 l/s was then proportionally distributed among these two sections, resulting in 61.99 l/s for the 1,334 square meter section and 66.82 l/s for the 1,438 square meter section. This allocation procedure was repeated for each subdivided polygon.

Following this, the individual runoffs within each grid were aggregated to yield the total runoff experienced within that grid. This approach allowed to standardize the city's runoff into a homogeneous and comparable spatial unit (50x50m grid cells). By employing this methodology, I transformed the initial heterogeneous spatial units, characterized by varying accumulated runoff values based on size, into a unified and comparable framework. This map will serve as a base for calculating the changes in the runoff conditions for each green roofs' scenario.

The subsequent step of the assessment involved the computation of expected runoff values for rooftops with and without green roofs, factoring in an average annual rainfall of 800 mm and a storm event with a recurrence interval of one in 10 years. This calculation employed the Excel runoff tool developed by Sælthun et al. (2021). For the analysis, it was determined the runoff values for a standard rooftop configuration (1 m^2, 1 meter in length) both with and without green roofs. This was conducted across slopes ranging from 0.01º to 30º, which represents the maximum slope considered feasible for the installation of green roofs. For this assessment, it was made the assumption that all new green roofs presented in scenarios S1, S2 and S3 are installed on rooftops without any prior green roofs' infrastructure, and these green roof installations contribute to a reduction in the existing runoff from the rooftops. **Table A4** displays the calculated runoff values of this step.

These values form the basis for computing runoff conditions in each scenario. To adapt them to the spatially explicit green roof scenarios, a necessary step involved aligning each scenario with the 50x50m grid. This alignment was achieved by subdividing the polygons representing green roofs within each scenario using the 50x50m grid. As a result, each grid cell contained the respective square meterage of green roofs for each scenario, along with the associated slope.

For each grid cell, the reduction in runoff was then calculated based on the square meterage and slope of the green roofs within that cell. For example, if a grid cell contained two distinct sections of green roofs —a 25 m^2 section with a 1º slope and a 10 m^2 section with a 15º slope— the anticipated runoff reduction would be -1.25 l/s. This reduction was computed as the sum of the reductions for each section: (5 m^2* -0.037472) + (10 m^2 * -0.032001). Detailed values for the calculated runoff reductions are presented in **Table A4**. This procedure was replicated for each scenario, generating a distinct map per scenario that illustrates the updated runoff values after the implementation of green roofs.

For normalizing the indicator, a threshold value of the maximum discharge quantity to active stormwater lines is defined in the guidelines for stormwater management in the City of Oslo (Oslo Kommune, 2023b). Here, the maximum quantity manageable by active storm water lines (*Maksimal påslippsmengde til aktiv overvannsledning*) is defined at 3.5 l/s per hectare. Since the proposed spatial unit is of 50x50 m^2 (25% of a hectare), the value was scaled to 0.875 l/s per 50x50 m^2.

To relate the runoff values to the threshold value, a normalization process was carried out, employing the min-max normalization process (see Eq. A.3). For doing this, the min value used was 0.875 l/s (threshold) and max value 243.5 l/s, which is the highest runoff observed from the indicator's sample among all scenarios (see **Fig. A1**). By doing this, the runoff values below 0.875 l/s are deemed insignificant, representing no exposure to heavy rainfall events, and therefore displaying a normalized value of 0 (see **Fig. A1**). On the contrary, the average runoff values exceeding 243.5 l/s represent high exposure and display a normalized value of 1.

Table A4. Runoff calculated values for rooftops with and without green roofs, considering an average annual rainfall of 800 mm and a storm event with a recurrence interval of one in 10 years

Area (m^2)	**Slope (º)**	**Length (m)**	**Runoff for rooftop with no green roof (l/s)**	**Runoff for rooftop with no green roof (l/s)**	**Difference in runoff (l/s)**
1	1	1	0,046207	0,008735	-0,037472
1	2	1	0,046207	0,007305	-0,038902
1	3	1	0,046207	0,008351	-0,037856
1	4	1	0,046207	0,009183	-0,037024
1	5	1	0,046207	0,009885	-0,036322
1	6	1	0,046207	0,010498	-0,035709
1	7	1	0,046207	0,011046	-0,035161
1	8	1	0,046207	0,011544	-0,034663
1	9	1	0,046207	0,012002	-0,034205
1	10	1	0,046207	0,012427	-0,033780
1	11	1	0,046207	0,012824	-0,033383
1	12	1	0,046207	0,013197	-0,033009
1	13	1	0,046207	0,013551	-0,032656
1	14	1	0,046207	0,013886	-0,032321
1	15	1	0,046207	0,014206	-0,032001
1	16	1	0,046207	0,014512	-0,031695
1	17	1	0,046207	0,014806	-0,031401
1	18	1	0,046207	0,015088	-0,031119
1	19	1	0,046207	0,015359	-0,030848
1	20	1	0,046207	0,015622	-0,030585
1	21	1	0,046207	0,015875	-0,030332
1	22	1	0,046207	0,016121	-0,030086
1	23	1	0,046207	0,016359	-0,029848
1	24	1	0,046207	0,016591	-0,029616
1	25	1	0,046207	0,016816	-0,029391
1	26	1	0,046207	0,017035	-0,029172
1	27	1	0,046207	0,017248	-0,028959
1	28	1	0,046207	0,017457	-0,028750
1	29	1	0,046207	0,017660	-0,028547
1	30	1	0,046207	0,017859	-0,028348

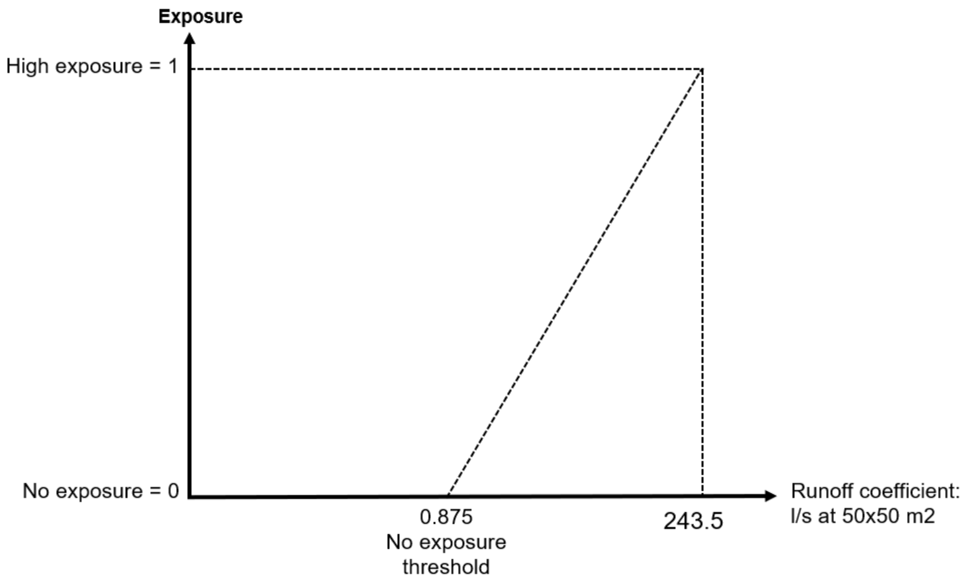


Figure A1. Exposure values of runoff coefficient represented in a cartesian coordinate system. Horizontal axis portrays runoff coefficient absolute values, as well as the selected threshold. Vertical axis represents the level of exposure and the normalized values associated with the indicator absolute values.

3.2.2. Areas with presence of critical infrastructure (sensitivity)

As described in the Analyses of Crisis Scenarios that may affect Norwegian society (Norwegian Directorate for Civil Protection, 2019), extreme weather poses a possible threat that may exacerbate in the upcoming years due to climate change, and critical infrastructure has been described as a key aspect to consider when these types of events happen.

To assess these, the process began by reviewing the list of critical infrastructures outlined by the Norwegian Directorate for Civil Protection (Norwegian Directorate for Civil Protection, 2016) for society's essential functions. From this list, the following were identified: motorways, freeways, major roads, regular railway tracks, railway stations, ferry terminals, helipads, schools, kindergartens, clinics, hospitals, fire stations, police stations, pharmacies, nursing homes, supermarkets, and ATMs. These categories were then geolocated using OpenStreetMap (OpenStreetMap contributors, 2020). After mapping the coordinates (or vector lines in the case of roads and railways), a buffer of 200m was calculated around each infrastructure to evaluate its surrounding area. Buffer areas were assigned a value of 1 (highly sensitive), while the remaining areas were assigned a value of 0. These buffers were subsequently correlated to a 50x50 grid, where grid cells covered by more than 50% of buffer areas were considered highly sensitive, while the rest were considered not sensitive.

3.2.3. Population density (sensitivity)

Generally speaking, residents in areas of high population density are at an increased risk of hazards due to the greater concentration of people in small areas (Guan et al., 2022; Sera et al., 2019). Also, urban areas with high human population density have high demand per unit area for ecosystem services (Gómez-Baggethun & Barton, 2013). Therefore, the inclusion of population density as an indicator can be useful for portraying sensitive areas to the presence of vulnerability hazards.

To calculate population density, a map of the total population by census tracts was obtained for the Oslo Municipality (Statistikkbanken Oslo commune, 2023). The total population was then divided by the km^2 of census tracts, to obtain population per Km^2. Finally, the result was converted into a grid with a resolution of 50x50m. Each grid obtained its value based on the census tract with the biggest geographical overlap. As a sensitivity indicator, it does not count with a threshold value. Min-max normalization was carried out using minimum and maximum indicator's sample values.

3.2.4. Elderly population density (sensitivity)

Research indicates that the elderly population is particularly vulnerable due to their reliance on assistance from others (Hossain & Meng, 2020; Morrow, 1999). Moreover, during extreme rain events, individuals with limited mobility, such as the elderly, may encounter challenges in protecting their belongings from flood damage once warnings are issued or when floods are in their initial stages (Kaźmierczak & Cavan, 2011). Elderly population density is then a proxy variable to include this sensitivity. The values were calculated focusing only on residents over 67 years old (Statistikkbanken Oslo commune, 2023) and following the same procedure as the one described for Population density (section 3.2.3)

3.2.5. Low-income households (sensitivity)

Low-income populations often receive insufficient attention and support during recovery efforts, mainly because they may not be readily visible within communities (Cutter, 2016). Furthermore, individuals such as those with lower incomes, caregivers, single parents, or retirees may struggle to access the necessary resources, energy, and mental resilience to reconstruct damaged infrastructure following extreme events (Clark, 1998; Tapsell et al., 2002). The prevalence of low-income households can serve as an indicator of this vulnerability.

This indicator is portrayed at the subdistrict level, employing data depicting the households whose income after tax per unit of consumption is less than 60 per cent of the median income for Oslo (Statistikkbanken Oslo commune, 2023b).

3.3. Vulnerability to heat

Summer temperatures in Oslo are expected to rise 5.6 °C in a scenario of moderate climate change, assuming alignment with the goals of the Paris Agreement (Bastin et al., 2019). The municipality of Oslo is aware of its exposure to heat risks and intends to become a climate-resilient city (Oslo Kommune, 2020). Similar to other urban green infrastructures, green roofs offer temperature regulation during heatwaves, both within and outside of buildings (Jaffal et al., 2012; Liu et al., 2021).

For assessing *Vulnerability to heat*, three exposure indicators were selected: outdoor heatwave day and night temperatures, and indoor heatwave day temperatures. Sensitivity indicators include population density, elderly population density and low-income households.

3.3.1. Outdoor heatwave day temperatures (exposure)

Understanding people's vulnerability to heat is closely related to daytime temperatures, as they are exposed to higher temperatures during the day, primarily due to increased levels of activity

(Basagaña et al., 2011; Wang et al., 2012; Xu et al., 2013). During these periods, the risks of heatstroke, heat exhaustion and cardiovascular problems are increased. To address this matter, I examined the average monthly temperatures (°C) during midday (13h-16h) for the 2018 heatwave in the Municipality of Oslo, under each of the green roof's scenarios proposed in this research. These temperatures were obtained from 4 meteorological simulations conducted with the Weather Research and Forecasting model (WRF v4.3.3; Skamarock et al., 2021). The simulations encompass a period of extreme high temperatures and dry weather over Oslo, from 00 UTC on July 21st to 00 UTC on July 28th, 2018 (see Segura et al. (2021) for a more detailed description of the event), with the first five days used for model spin-up. The WRF configuration consists of 4 two-way nested domains, where the innermost have a horizontal resolution of 333m and cover entirely the Municipality of Oslo. The physical configuration of the model is similar to the one described in Segura et al, 2021. Each WRF simulation used scenario-specific input data, such as land-use maps, urban fractions, building and street morphologies, and irrigation maps. Results were then mapped through GIS and converted into a grid of 50x50m in order to match sensitivity indicators. GIS operations were carried out using QGIS (version 3.28.0-Firenze).

The designated threshold for this indicator is set at a minimum of 30 ºC. This threshold is determined by the heatwave plan for England (UK Health Security Agency, 2022) and European Environment Agency reference temperatures (European Environment Agency, 2023), in the absence of Norwegian guidelines. It defines days as high-risk when surface air temperatures surpass 30 °C, prompting the implementation of targeted actions for high-risk groups. The min-max normalization process for this indicator uses as min value 30 ºC (threshold) and max value 29.7 °C, which is the highest temperature observed from the indicator's sample among all green roof scenarios (see **Fig. A2**). Since maximum temperature is below threshold, the exposure of this indicator is 0.

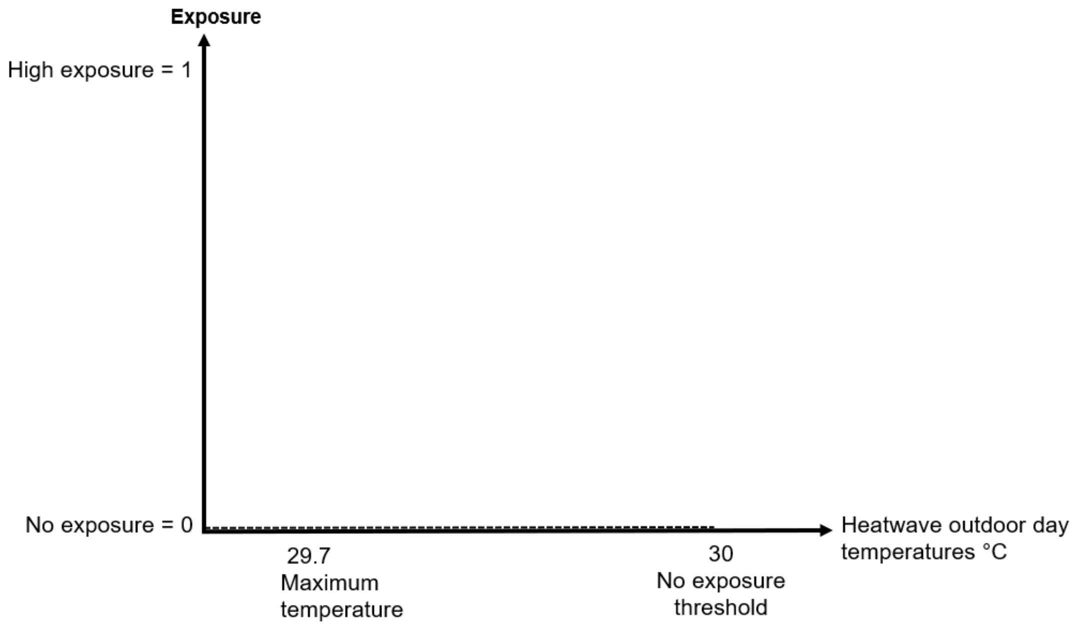


Figure A2. Exposure values of Heatwave outdoor day temperatures represented in a cartesian coordinate system. Horizontal axis portrays temperature absolute values runoff coefficient absolute values, as well as the selected threshold. Vertical axis represents the level of exposure and the normalized values associated with the indicator absolute values.

3.3.2. Outdoor heatwave night temperatures (exposure)

The urban heat island effect causes high temperatures at night during heatwave periods, which makes it difficult for people to cool down from the daytime heat (Smith et al., 2011). Such conditions can lead to health problems, particularly among vulnerable groups such as the elderly (Heaviside et al., 2016). In order to include these vulnerabilities in the assessments, it was examined the average monthly temperatures (°C) during nighttime (21h-7h) for the 2018 heatwave in the Municipality of Oslo. Similar to *Heatwave day temperatures*, temperatures were obtained from simulations for 00 UTC on July 21st to 00 UTC on July 28th 2018, the first five days discarded for spin-up. The temperatures were mapped following the same methods employed for *Heatwave day temperatures*.

The selected threshold for this indicator is a minimum value of 20ºC, which represents the temperature from which exposure begins to increase (all areas with night temperatures below 20ºC are not considered as exposed). This threshold is determined by European Environment Agency reference temperatures (European Environment Agency, 2023), in the absence of Norwegian guidelines. It defines the increased heat risk at night when surface air temperatures do not drop below 20ºC.

The min-max normalization process for this indicator employes a min value of 20ºC (threshold) and a max value 22.9 °C, which is the highest temperature observed from the indicator's sample among all green roof scenarios (see **Fig. A3**).

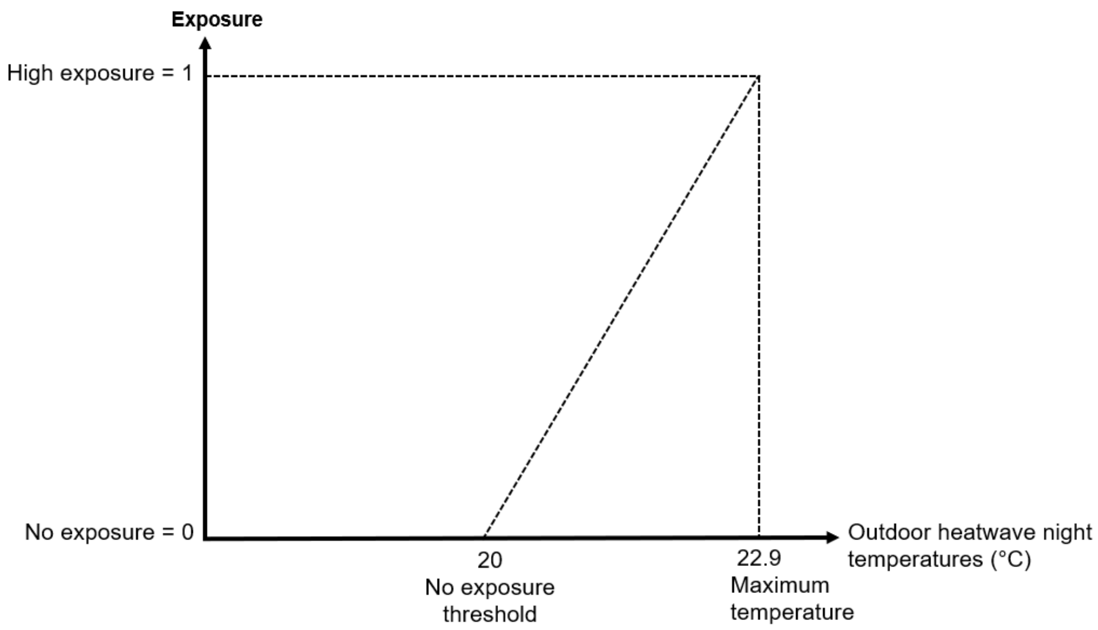


Figure A3. Exposure values of Outdoor heatwave night temperatures represented in a cartesian coordinate system. Horizontal axis portrays temperature absolute values, as well as the selected threshold. Vertical axis represents the level of exposure and the normalized values associated with the indicator absolute values.

3.3.3. Indoor heatwave day temperatures (exposure)

Elevated indoor temperatures have a significant impact on the well-being, satisfaction, and efficiency of occupants, with extreme instances potentially resulting in health issues and even fatalities (Taylor et al., 2023). To assess green roofs impacts on these temperatures, first, the inner temperature for each building available for the creation of green roofs was matched to the *Heatwave outdoor day temperatures* as a simplifying assumption (Marvuglia et al., 2020). When green roofs were present on buildings, it was hypothesized that indoor temperatures were decreased by 1.4 °C, based on calculation made by (Marvuglia et al., 2020) for Stockholm, a city with geographical similarities to Oslo. The selected threshold for this indicator is a minimum value of 26 ºC, which represents the temperature from which exposure begins to increase. This threshold was chosen because it is commonly used in overheating evaluation methods for residential buildings in temperate climates in Europe (Tian et al., 2020). The min-max normalization process for this indicator employes a min value of 26ºC (threshold) and a max value 29.7 °C, which is the highest temperature observed from the indicator's sample among all green roof scenarios (see **Fig. A4**).

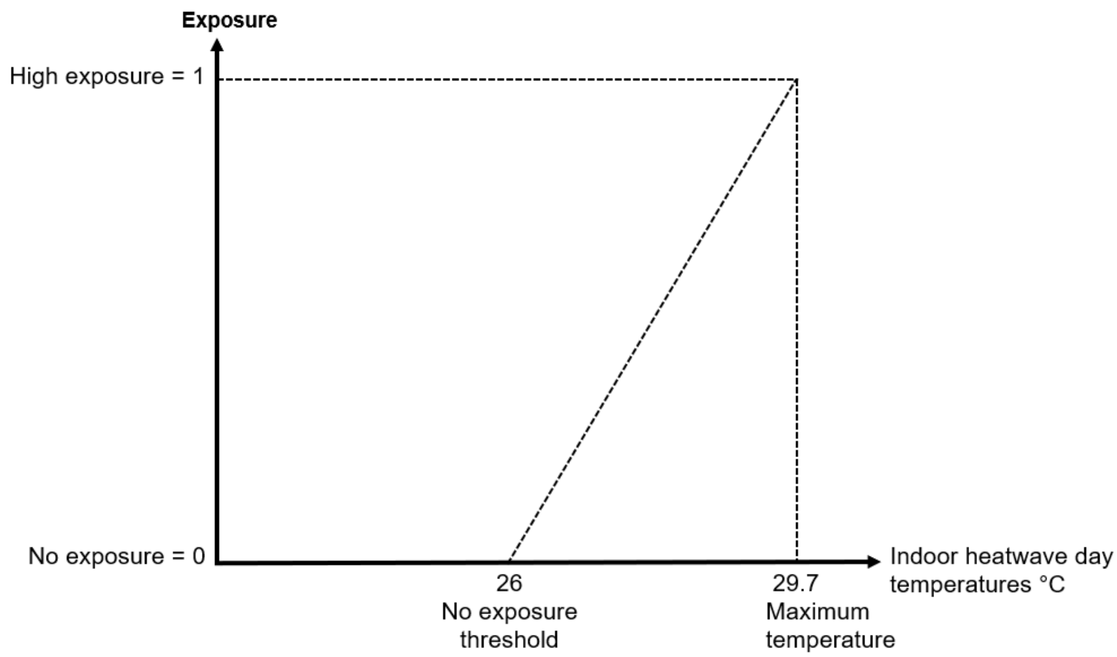


Figure A4. Exposure values of Indoor heatwave day temperatures represented in a cartesian coordinate system. Horizontal axis portrays temperature absolute values, as well as the selected threshold. Vertical axis represents the level of exposure and the normalized values associated with the indicator absolute values.

3.3.4. Elderly population density (sensitivity)

Research indicates that elderly individuals are more prone to experiencing negative health impacts as a result of heat waves due to a combination of social and medical factors (i.e., reduced ability to regulate body temperature, mental disorders which impact an individual's ability to recognize and respond to heat-related risks and living alone or having increased dependency) (Hajat et al., 2010; Oudin Åström et al., 2011). Elderly population density is then a proxy variable to include this sensitivity. The values were calculated following the same procedure as the one described in section 3.2.4.

3.3.5. Population density (sensitivity)

Generally speaking, residents in areas of high population density are at an increased risk of hazards due to the greater concentration of people in small areas (Guan et al., 2022; Sera et al., 2019). Population density maps were calculated following the same procedure described in section 3.2.3.

3.3.6. Low-income households (sensitivity)

Lower-income neighborhoods tend to experience a disproportionately higher levels of exposure to urban heat (Chakraborty et al., 2019), considering that wealthier households often have more options available to seek thermal comfort compared to lower-income households (Berger et al., 2022). The presence of low-income households can serve as an indicator of this sensitivity. This was assessed based on the same approach presented in section 3.2.5.

3.4. Vulnerability to air pollution

Air pollution in Oslo has steadily decreased over the last decades due to local measures, but more efforts are required since exceedances from pollutants associated with road traffic and domestic heating still occur (Oslo Kommune, 2021b). The use of vegetation as a passive filter of urban air has been previously investigated, including extensive green roofs (Gourdji, 2018; Speak et al., 2012), showcasing positive outcomes.

For this vulnerability, the selected exposure indicators were the presence of particulate matter smaller than 10 µm (PM10), along with population density, children population density, and low-income households as sensitivity indicators.

3.4.1. Particulate matter 10 (PM_{10}) (exposure)

PM10 is listed as one of the priority air pollutants to be reduced within the Municipality of Oslo due to its impact on health (Oslo Kommune, 2021b). To calculate this vulnerability, first, it was mapped the PM10 produced by woodburning and construction activities during the whole year of 2019, expressed in grams/year (Miljødirektoratet, 2023). Presented in grids of 250x250m, this data was transformed into the 50x50 grid employed for the vulnerability assessment. For doing so the 250x250m grid was first split using the 50x50m grid. Then, the split 250x250m polygons area was used to compute the original values of PM10 (g/year). For instance, if a 250x250m grid cell had PM10 value of 150,000 g/year, and was equally split in 5 sections of 50x50m, then each 50x50 grid cell would have a PM10 value of 30,000 g/year.

Next, each of the calculated PM10 values at the 50x50 grid is reduced by the amount of green roofs (m^2) present in each of the grids, based on the assumption that each m^2 of green roof can capture 0.42 g PM10/year, according to estimations proposed by speak et al. (2012) for the species S. album (sedum). This last calculation was made for each one of the green roof scenarios.

To assign a threshold for assessing these accumulated yearly PM10 values, first it was assessed the current state of air pollution in Oslo according to PM10 concentrations values. Judging by 2019, the average PM10 µg/m3 was 8.5% above the recommendation by Oslo Municipality (Statistikkbanken Oslo Kommune, 2020). Based on this, it is assumed that there is a need to reduce PM10 pollution by 8.5%. This percentage was then applied to the total accumulated yearly PM10 (grams/year) obtained from the spatially explicit calculation, in order to obtain an objective PM10 accumulated yearly value. Next, this value was divided by the number of grid cells in the 50x50 grid, to have an objective value for each grid cell (see **Table A5**). By doing this, it was established a threshold where, below it, there is no exposure to air pollution (see **Fig. A5**).

Table A5. Calculation of threshold value for exposure indicator PM_{10}

Description	Value
Total 2019 PM_{10} (grams)	441,755,599
Reduction based on concentration PM_{10} objective (%)	8.5%
Total after PM_{10} reduction (grams)	404,044,755
Num of pixels	2789
Threshold (g)	144,870.83

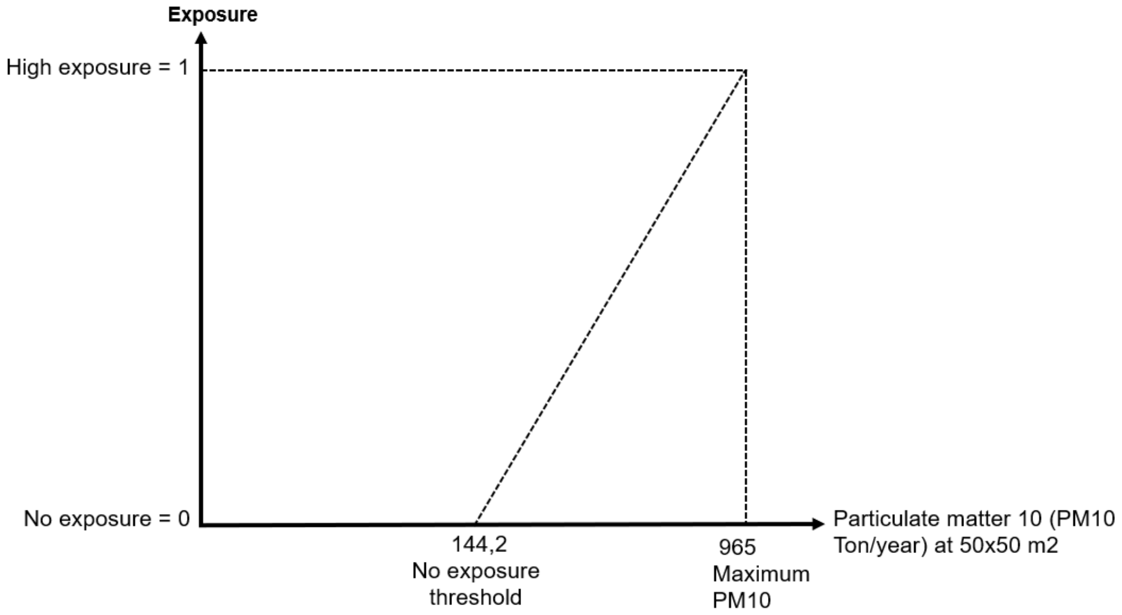


Figure A5. Exposure values of Particulate matter 10 (PM_{10}) represented in a cartesian coordinate system. Horizontal axis portrays PM10 presence at the pixel level, as well as the selected threshold. Vertical axis represents the level of exposure and the normalized values associated with the indicator absolute values.

3.4.2. Population density (sensitivity)

Similar to *Vulnerability to heat*, areas with high population density pose an increased risk of hazards to their residents, as there is a greater number of people exposed to vulnerabilities. Furthermore, population density has already been employed for a better understanding of vulnerability to air pollution (Demoury et al., 2024). Population density maps were calculated following the same procedure described in section 3.2.3.

3.4.3. Children population density (sensitivity)

Children are commonly recognized as a vulnerable group in environmental health risk assessments, facing negative health effects as a result of exposure to various atmospheric pollutants (Vanos, 2015). The values for this vulnerability were calculated focusing only on residents of 15 years old or less (Statistikkbanken Oslo commune, 2023) and following the same procedure as the one described for Population density (section 3.2.3).

3.4.4. Low-income households (sensitivity)

Low economic status has been linked to a disproportionate risk of suffering the adverse cardiovascular effects of exposure to ambient air pollution (Tibuakuu et al., 2018). For example, low income can restrict the ability to reduce exposure, such as through housing choices and quality, as well as to manage illnesses, including accessing healthcare, preventive measures, and social services (Makri & Stilianakis, 2008). The presence of low-income households can serve as an indicator for better understanding this vulnerability and was mapped using the same approach presented in section 3.2.5.

3.5. Vulnerability to a lack opportunities for interactions with natural environments

Oslo features one of the world's greatest availabilities of green space (Huang et al., 2021). However, in the last decades, it has experienced increasing population densities (Næss, 2022) and it is estimated that between 55-76% of Oslo's population currently resides in areas that fall short of meeting the WHO targets for exposure to green space (Barboza et al., 2021). The city of Oslo intends to preserve and further develop the presence of urban green areas in the city (Oslo kommune, 2015), aiming to create inclusive environments that offer *tranquillity, recreational opportunities, joy, playfulness, proximity, and meaningful experiences* for all residents (Oslo Kommune, 2017). For this matter, and given the competition for space in compact cities, green roofs can contribute to enriching urban environments by enhancing residents' contact with nature through expanded visible green areas (Gagnon et al., 2018), and by heightened multisensory experiences related to the fauna present on green roofs (e.g., bird sighting/hearing) (Mesimäki et al., 2017). In this sense, it is relevant to consider that the city of Oslo considers that all types of green roofs are a valuable tool for completing existing urban green infrastructure, but not for substituting it (Oslo Kommune, 2022).

3.5.1. Share of green areas (exposure)

To assess the availability of green spaces fostering interactions with natural environments, the proportion of green areas within the city of Oslo was measured. This indicator, recommended by the EU handbook for the evaluation of NBS impacts, proves valuable for green space management. For the green roofs case study, it was employed the Normalized Difference Vegetation Index (NDVI) at a 20m resolution, based Landsat8 images spanning from May 1st to November 30th, 2015. NDVI values below 0.2 were interpreted as water, artificial land covers, bare soil, and dead vegetation (ICGC - Institut Cartogràfic i Geològic de Catalunya, 2021), while values above 0.2 represented various vegetation covers. Following this, the entire municipality of Oslo was mapped at 20m pixels, distinguishing green areas from non-green areas.

The subsequent step involved augmenting this layer with the presence of green roofs. I selectively included green roofs where the majority of their area fell within pixels denoting non-green areas. This selection aimed to identify green roofs capable of genuinely increasing the city's green coverage. These chosen green roofs were then considered as new green areas, contributing to the expansion of green spaces within the city. This process was repeated for each of the green roof scenarios.

To account for green areas for each of the subdistricts in the city, it was considered the m^2 of green falling within a 250m buffer around subdistrict limits (Venter et al., 2021). Finally, for each subdistrict, the accounted m^2 of green was divided by the total area of the subdistrict, considering the 250m buffer surrounding it. By doing so, it was obtained a percentage of green cover for each of the subdistricts of the city of Oslo. In order to normalize the data, it was employed the reference values depicted by (Konijnendijk, 2023), which suggest that every neighborhood should have no less than 30% of the area covered by vegetation. It is worth noting that this framework primarily suggests that the 30% cover should be composed of tree canopy, however, it also expands this threshold to include other green areas (such as green roofs) in cases where trees are less present.

In order to normalize the data, it was employed the reference values depicted by (Konijnendijk, 2023), which suggest that every neighborhood should have no less than a 30% of area covered by vegetation. It is essential to note that while the framework primarily emphasizes achieving this 30% cover through tree canopy, it recognizes the need for flexibility in different contexts. The application of this threshold results in an exposure enhanced in those areas close to 0% of green cover, and an absence of exposure beyond the 30% of green cover (see **Fig. A6**)

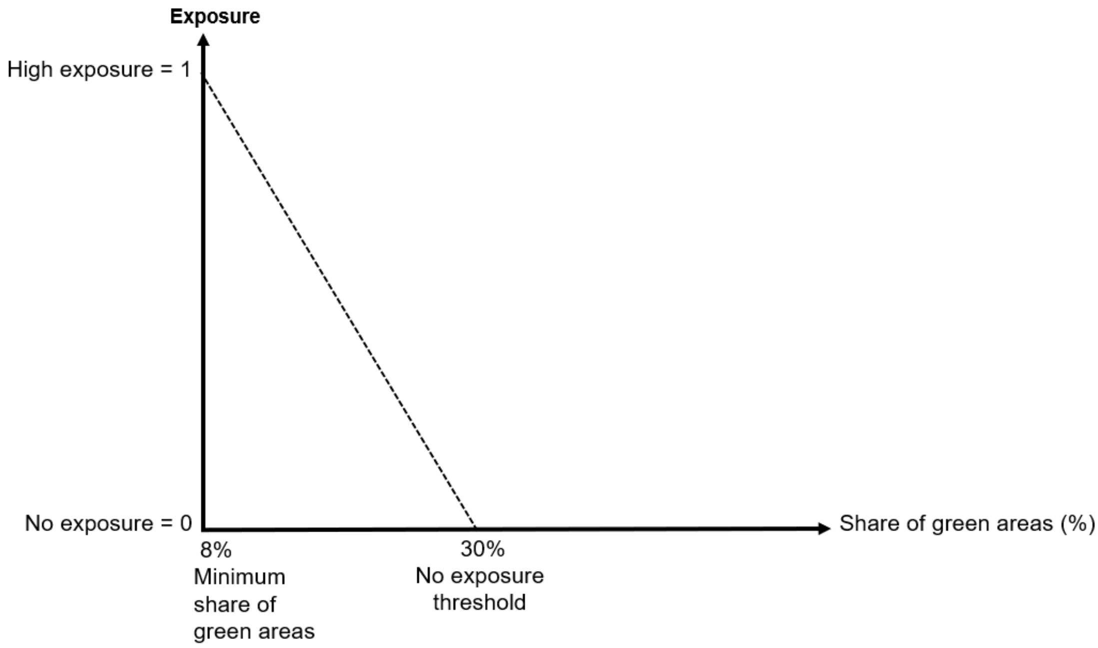


Figure A6. Exposure values share of green areas represented in a cartesian coordinate system. Horizontal axis portrays percentage of green areas by subdistrict, as well as the selected threshold. Vertical axis represents the level of exposure and the normalized values associated to the indicator values.

3.5.2. Green Gini coefficient (exposure)

In order to account for the spatial equity of the green cover across the city of Oslo, it was employed the Gini coefficient approach. So far, the Gini coefficient has been utilized in the field of economics as a reliable indicator for assessing income inequality among residents. However, some authors have employed the Gini approach for accounting for the distribution of green within cities (see Wang et al., 2019; Xu et al., 2018). In such cases, the Gini coefficient was applied as an indicator of green space equity and its relationship with socioeconomic variables (e.g., population density). However, for this assessment, the Gini coefficient is calculated considering only the unequal distribution of green spaces without considering socio-economic variables, as these are being accounted for in the sensitivity indicators. The calculation of the Green Gini coefficient is done at the district level (n=97) based on the *Share of green areas* at the subdistrict level (n=574). In other words, the Green Gini coefficient is calculated for each district, based on how the green areas are distributed among its subdistricts according to the Share of green areas. Gini coefficients were calculated following the same approach as (Chen et al., 2022), where:

$$Gini = A/(A+B) \qquad \text{(Eq. A1)}$$

Where *A* represents the region of inequality, specifically the area situated between the Lorenz curve and the equality line, while *B* corresponds to the area beneath the Lorenz curve (see

Fig. A7). The Gini coefficient varies from 0 (indicating complete equality) to 1 (indicating complete inequality). The computation for the area beneath the Lorenz curve, denoted as *B* in the diagram, was performed as follows:

$$B = \frac{\sum_{s=1}^{n}(G_s - G_{s-1}) * (M_s - M_{s-1})}{2} \quad \text{(Eq. A2)}$$

Here, *s* represents the count of subdistricts of any certain district, arranged in ascending order based on its *Share of green areas*. The parameter *s* takes values from 0 to *n*, where *n* represents the cumulative number of subdistricts within each district. *Gs* represents the cumulative *Share of green areas* considering a 250m buffer around the subdistrict limits. *Ms* denotes the cumulative share of each subdistrict's area considering its 250m buffer, and relative to its corresponding district. The area between the line of equality and the coordinate axis is established as 0.5. Consequently, the calculation for *A* was conducted as follows:

$$A = 0.5 - B \quad \text{(Eq. A3)}$$

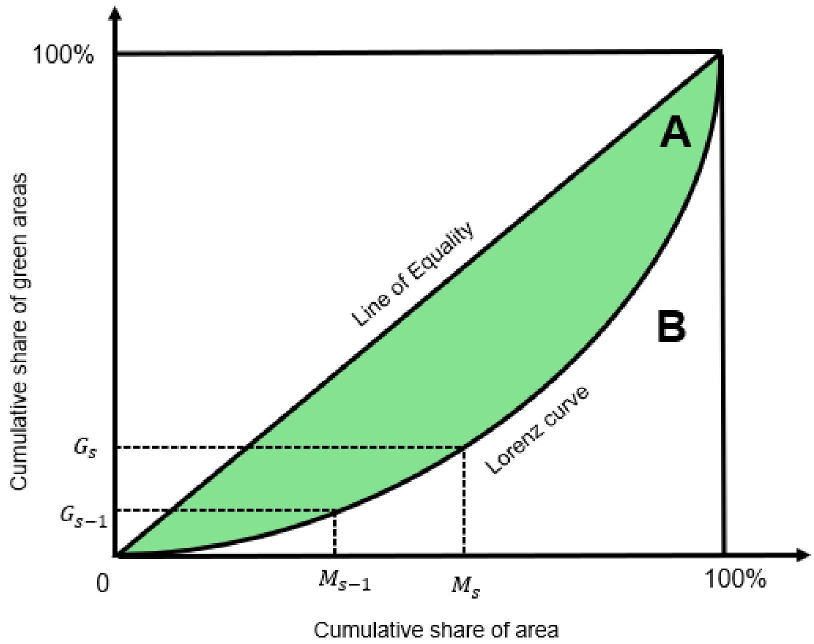


Figure A7. Graphical representation of the Green Gini coefficient. The Gini coefficient is determined by the proportion of the region situated between the Line of Equality and the Lorenz curve (indicated as A in the diagram) to the entire area beneath the line of equality (indicated as A and B in the diagram). Its values range from 0, signifying perfect equality, to 1, indicating total inequality.

For the Gini coefficient, a nuanced approach to normalization was employed. Unlike other indicators in this study that utilize predetermined thresholds, the normalization of the Gini coefficient was contextualized within its own distribution due to the lack of a clear threshold value for the city of Oslo. The maximum Gini coefficient within Oslo served as the max value (i.e., 0.4719) employed for normalization. By using this, the minimum (i.e., 0.0078) value was transformed into an exposure value of 0 and the highest to a value of 1 (see **Fig A8**).

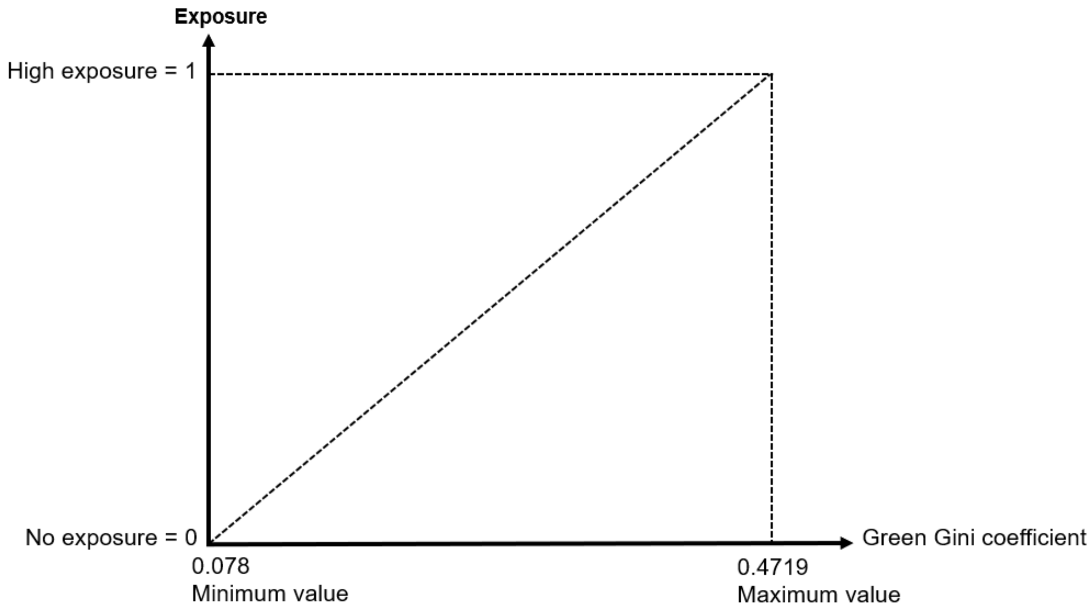


Figure A8. Exposure values share of Green Gini coefficient in a cartesian coordinate system. Horizontal axis Green Gini Coefficient, as well as for the minimum and maximum values employed for normalization. Vertical axis represents the level of exposure and the normalized values associated with the indicator values.

3.5.3. Children population density (sensitivity)

Urban green areas have been positively correlated to children's wellbeing in the past. Exposure and interaction with urban green and blue spaces have been linked to enhanced physical and mental well-being in children (Kabisch et al., 2017). This includes alleviating stress (Akpinar, 2016) and ameliorating symptoms of attention-deficit/hyperactivity disorder (ADHD) (Faber Taylor & Kuo, 2011). This indicator was calculated following the same methodology depicted in section 3.4.3.

3.5.4. Low-income households (sensitivity)

Low-income areas have been correlated with a variety of urban environmental injustices, such as lack of green spaces (Wolch et al., 2014). Various factors contribute to the uneven distribution of green spaces within urban areas, such as park design approaches, historical patterns of land development, changing perceptions of leisure and recreational activities, as well as legacies of social class and racial inequality (Byrne, 2012). In this sense, the presence of low-income households can serve as an indicator of this sensitivity. This was assessed based on the same approach presented in section 3.2.5.

4. Broad-scale vulnerabilities and indicators – descriptions and calculations

4.1. Life-cycle assessment of extensive green roof

We are utilizing life cycle assessment (LCA) to evaluate the environmental impact of green roofs. LCA is a widely recognized and commonly employed methodology for assessing the environmental impacts associated with various products, processes, and services. It is defined by the ISO 14040 standards and consists of four stages: goal and scope, life cycle inventory, life cycle impact assessment, and interpretation. LCA provides valuable insights into identifying opportunities to reduce environmental impact and informs decision-making regarding sustainable practices. For this study, it employed Simapro 9.3 software, and used the Ecoinvent 3.8 database and the Environmental Footprint 3.1 method.

The objective of the LCA is to evaluate the environmental impacts associated with the entire life cycle of green roofs. To accomplish this, a comprehensive review of pertinent literature was carried out, including existing standards (Standard Norge, 2015). Based on these findings, it was made the decision to focus the green roofs analysis on two distinct components: a structural part, comprising the necessary layers for roof membrane protection, and a substrate part, consisting of the growing media essential for sustaining vegetation.

The functional unit selected for the LCA is the production, installation, use and dismantling of 1 m2 of extensive green roof capable of retaining 5 mm of water for any precipitation event, considering its use over a period of 1 year.

The system boundaries encompass all stages of the life cycle, including (1) the extraction of raw materials and their manufacturing into components ready for use, (2) the installation process, which involves machinery for placing the different components on the roof, (3) ongoing maintenance, and (4) the end-of-life management of each component. The end-of-life stage includes machinery used for deconstruction and the different waste treatment processes for all materials. Transportation between these stages is also taken into account (see **Fig. A9**).

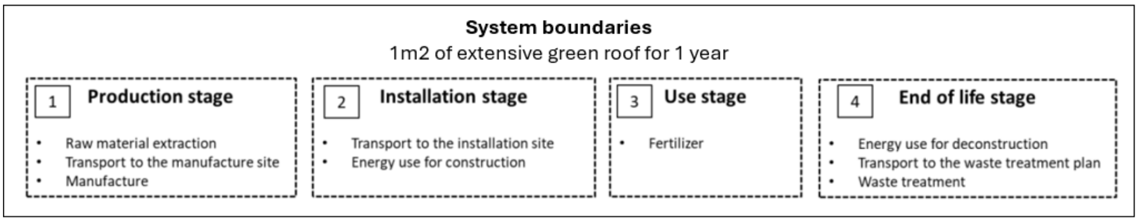


Figure A9. System boundaries of the green roofs LCA

Now it will be described the green roofs' layers and substrate, as well as their life-cycle stages.

4.1.1. Layer description

The three structural parts designed in this study contain the layers required to protect the roof membrane and support the growing medium. The green roof configuration complies with the overarching specifications delineated in various reports and standards regarding the construction of an extensive green roof system and incorporates all of the requisite layers (Standard Norge, 2015; Noreng et al., 2012). This includes a root barrier to prevent plant root penetration, a water retention layer comprising fibers to supply water and manage runoff, and a filter layer of non-woven polypropylene fabric to prevent clogging and preserve the growing medium (see **Table A6**). This configuration was defined based on a study conducted by Braskerud (2022) that has demonstrated that removing the drainage layer from an extensive green roof system leads to enhanced water retention capacity and improved peak flow attenuation. These findings are consistent with other scholars in the field (Braskerud & Paus, 2022). Moreover, the relevant normative standard highlights that a water retention layer can fulfil the functions of both a drainage layer and a protective layer (Standard Norge, 2015).

4.1.2. Substrate description

The substrate layer plays a vital role in supporting the growth and survival of vegetation in extensive green roofs. It ensures a suitable environment for the development of vegetation by offering a growing medium that can store and give water and nutrients to the plants (Standard Norge, 2015). Additionally, the substrate layer acts as an anchor for the plants, preventing them from being dislodged by wind or water runoff (Nagase & Dunnett, 2011). The depth and weight of the substrate layer are important factors that must be considered to ensure the structural integrity of the roof. According to the Landscape Development and Landscaping Research Society (FLL, 2018) the depth of a substrate layer for extensive green roofs can vary from 2.5 cm to 20cm. The composition of substrate used in extensive green roofs is usually made up of mineral components and a maximum of 20% of organic matter on a volume basis (Standard Norge, 2015), with natural, artificial or recycled materials being used. Each component has its own advantages and disadvantages in terms of weight, water retention capacity, and porosity (Ampim et al., 2010). The optimal substrate should have high aeration, drainage, and nutrient retention and be sturdy, permanent, and lightweight (Friedrich, 2008).

The substrate is made of 70% pumice, 20 % gravel and 10% compost (see **Table A6**). This composition originates from research conducted by Ji et al. (2018) where they examined the relationship between substrate structure and vegetation in the Norwegian climate. Their analysis focused on investigating the connection between four different substrates - a fine substrate, a coarse substrate, a mixed substrate, and a layered substrate - and various Sedum plant species. The findings indicated that the fine substrate, consisting of a mixture of pumice, gravel, and compost, showed notable advantages. It demonstrated higher shoot biomass, a reduced proportion of roots, and a higher shoot biomass per unit root length. These outcomes are consistent with the fine substrate's ability to retain water, allowing it to sustain moisture levels over an extended duration between weekly watering sessions. Considering these results, the fine substrate composition was included for the study.

Table A6. Composition of layers and substrates

Layer	Element	Weight (kg/m²)	Height (mm)
Root Barrier	Polyethylene (LDPE)	0.8	0.4
Water retention	Recycled textile fiber	1.28	10
Filter Layer	Textile - nonwoven polypropylene	0.2	1.9
Substrate	Pumice	31.97	70
	Gravel	21.42	20
	Compost	2.5	10

4.1.3. Installation stage

The installation stage includes both the transportation and installation processes. In this study, it was approximated a transportation distance of around 500 km by truck from a manufacturing facility in Sweden to the subsequent utilization phase in Oslo. This estimation is based on a green roof supplier situated in Stockholm, Sweden, and was calculated using Google Earth software to measure the distance between the two locations. During the installation process, a tower crane is employed to lift and position all the necessary materials onto the roof. An energy consumption of 0.0039 kWh/kg was assumed for the lifting operation.

4.1.4. Use stage

The use stage involves their maintenance, which is essential for sustaining their intended functionality. It is recommended to conduct annual maintenance activities biannually, specifically during the spring and late summer or early autumn (Noreng et al., 2012). To prevent excessive weed growth on the roof Nagase & Dunnett (2011) suggest applying slow-acting fertilizers at a rate of 15-20 g/m2. Therefore, an application rate of 15 g/m2 per year is assumed in this study.

4.1.5. End of life stage

The end-of-life stage covers the deconstruction, transportation, and waste treatment processes. In this assessment, there were considered multiple factors related to the end-of-life scenario for the materials used in the green roof's assembly. This includes the energy required for their removal from the roof, the transportation distance to waste treatment facilities, and the specific waste treatment methods employed. It was assumed a transportation distance of 100 km by truck to the designated waste treatment plant. Different waste treatment scenarios were considered for each component. The substrate is assumed to be landfilled, the textile-based material is slated for incineration, and the plastic-based materials are subjected to a combination of 50% recycling and 50% incineration.

For details of the life cycle inventory, please see **Appendix C.**

4.1.6. Life Cycle Impact Assessment

The impact categories selected for the LCA are presented in **Table A7.** The calculated impacts are presented in **Appendix C.**

Table A7. Life cycle impacts and descriptions

Impact category	Unit	Description
Climate change	kg CO2 eq	Contribution to climate change by the emissions of greenhouse gases, primarily carbon dioxide (CO2), methane (CH4), and nitrous oxide (N2O).
Ozone depletion	kg CFC11 eq	Potential depletion of the stratospheric ozone layer. It is typically associated with the emission of substances like chlorofluorocarbons (CFCs) that have the potential to break down ozone molecules in the atmosphere.
Human toxicity, cancer	CTUh	Accounts for the adverse health effects on human beings caused by the intake of toxic substances through inhalation of air, food/water ingestion, and penetration through the skin insofar as they are related to cancer.
Human toxicity, non-cancer	CTUh	Impact category that accounts for the adverse health effects on human beings caused by the intake of toxic substances through inhalation of air, food/water ingestion, penetration through the skin insofar as they are related to non-cancer effects that are not caused by particulate matter/respiratory inorganics or ionizing radiation.
Photochemical ozone formation	kg NMVOC eq	Photochemical oxidation evaluates the potential for substances to contribute to the formation of ground-level ozone and other secondary pollutants through chemical reactions in the presence of sunlight. It accounts for Volatile organic compounds (VOCs) and nitrogen oxides (NOx)
Ecotoxicity, freshwater	CTUe	Freshwater aquatic ecotoxicity evaluates the potential toxic effects of substances on freshwater ecosystems. It considers the impact on aquatic life, such as fish and other organisms living in freshwater bodies. Considers pesticides (Agricultural pesticides, herbicides, and insecticides can be toxic to aquatic life in freshwater ecosystems when they run off into rivers and lakes), heavy metals (elements like lead, mercury, and copper can accumulate in freshwater ecosystems and harm aquatic organisms) and organic chemicals (Polycyclic aromatic hydrocarbons)
Ionising radiation	kBq U-235 eq	Detrimental effects on human health and ecosystems associated with the release of radionuclides.
Acidification	mol H+ eq	Acidification assesses the potential for emissions to lead to acid rain or other forms of environmental acidification. It quantifies the impact of emissions on soil and water acidity, which can harm ecosystems and aquatic life.
Eutrophication, marine	kg N eq	Increased algae and plant growth, which can disrupt aquatic ecosystems, deplete oxygen levels, and harm aquatic life due to algal blooms. Accounts for nitrogen (N) emissions
Eutrophication, freshwater	kg P eq	Increased algae and plant growth, which can disrupt aquatic ecosystems, deplete oxygen levels, and harm aquatic life due to algal blooms. Accounts for phosphorus (P) emissions
Eutrophication, terrestrial	mol N eq	Measure of increased nutrient presence in the terrestrial ecosystem resulting from the release of nitrogen-containing compounds.

4.2. Vulnerability to climate change

Vulnerability to climate change is defined by the Planetary Boundary "Climate Change", described as a notable deviation from the established patterns of natural variability witnessed throughout the Holocene era. This period, characterized by the emergence of agriculture and subsequent human civilizations, serves as a reference point for stability. A more detailed understanding can be achieved by considering specific climate-related benchmarks. These may encompass swift sea level escalation (approximately 1 meter or more per century), disturbances in regional climates caused by droughts, floods, and other extreme phenomena, as well as alarmingly high rates of biodiversity depletion, which directly impact the ecosystem services they sustain (Rockström et al., 2009).

Climate change Planetary Boundary is measured by atmospheric CO2 concentration, which works as the proxy for radiative forcing due to changes in all greenhouse gas concentrations, on the basis that the current cooling effect of aerosols approximately counteracts the warming effect of non-CO2 greenhouse gases. The planetary boundary for atmospheric CO2 concentration is set at 350 ppm (Rockström et al., 2009), which implies that with a current concentration of 417 ppm, the boundary is already surpassed (Richardson et al., 2023).

A second threshold for climate change is established by alterations in the Earth's surface energy balance, quantified by shifts in radiative forcing measured in watts per square meter (W m^-2) (Rockström et al., 2009). This threshold, however, has not been included in the *Vulnerability to Climate change* due to limitations on assessing how green roofs implementation could be affecting it.

Within the green roofs assessment, this vulnerability is assessed by the LCA impact category *Climate Change.*

4.3. Vulnerability to stratospheric ozone depletion

Vulnerability to stratospheric ozone depletion is defined by the Planetary Boundary "Stratospheric Ozone Depletion", which focuses on the maintenance of the ozone layer in the Earth's stratosphere. The ozone layer, located in the stratosphere, filters (UV) radiation from the sun. In the past, the combination of heightened levels of anthropogenic ozone-depleting substances, such as chlorofluorocarbons, along with the presence of polar stratospheric clouds has led to the ozone effectively disappearing in the lower stratosphere (Rockström et al., 2009). The depletion of the stratospheric ozone layer has adverse effects on marine organisms and presents health hazards to human populations (Rockström et al., 2009)

The Stratospheric Ozone Depletion Planetary Boundary is set by considering the anthropogenic release of new substances, such as gaseous halocarbon compounds from industrial and other human activities, into the atmosphere. To maintain a safe operating space, the boundary is established at 276 Dobson Units (DU), permitting an increase of less than 5% from the preindustrial level of 290 DU, evaluated by latitude (Richardson et al., 2023). The current global estimate is 284 DU is within the safe operating space.

Within the green roofs assessment, this vulnerability is assessed by the LCA impact category *Ozone Depletion*.

4.4. Vulnerability to novel entities

Vulnerability to novel entities is defined by the Planetary Boundary "Novel Entities", described as new anthropogenic introductions to the Earth system. These encompass a range of synthetic chemicals and substances such as microplastics, endocrine disruptors, and organic pollutants. Additionally, they include anthropogenically mobilized radioactive materials, including nuclear waste, as well as human modifications of evolution through genetically modified organisms and other direct interventions in evolutionary processes (Richardson et al., 2023). Hundreds of thousands of synthetic chemicals are currently manufactured and released into the environment. For many of these substances, the potentially significant and long-lasting effects on Earth system processes resulting from their introduction, especially concerning the integrity of functional biospheres, are poorly understood. Additionally, their usage lacks adequate regulation (Richardson et al., 2023).

The overall impacts of novel entities within the Earth system remain largely unstudied. The planetary boundaries framework primarily focuses on the stability and resilience of the Earth system, rather than human or ecosystem health. Therefore, it presents a scientific challenge to determine the extent to which the Earth system can tolerate the loading of novel entities before irreversibly transitioning into a potentially less habitable state (Richardson et al., 2023). For this class of novel entities, the only genuinely safe operating space that can guarantee the preservation of Holocene-like conditions is one in which these entities are absent unless their potential impacts on the Earth system have been thoroughly evaluated (Richardson et al., 2023).

Within the green roofs assessment, this vulnerability is assessed by the LCA impact categories *Human toxicity (cancer), Human toxicity (non-cancer)*, *Photochemical ozone formation*, *Ecotoxicity (freshwater)* and *Ionizing radiation.*

4.5. Vulnerability to changes in biogeochemical flows

Vulnerability to changes in biogeochemical flows is defined by the Planetary Boundary "Biogeochemical flows: P and N cycles", which reflects on anthropogenic perturbation of global element cycles (Richardson et al., 2023). As of now, the PB framework acknowledges nitrogen (N) and phosphorus (P) due to their crucial roles as fundamental building blocks of life, with their global cycles significantly altered by agricultural and industrial activities. For both N and P, the anthropogenic release of reactive forms into land and oceans is of significant concern (Richardson et al., 2023). Human activities, primarily through the manufacture of fertilizer for food production and the cultivation of leguminous crops, convert approximately 120 million tonnes of N2 from the atmosphere into reactive forms each year (Rockström et al., 2009). Altered nutrient flows and element ratios have profound effects on ecosystem composition and long-term Earth system dynamics. Some of the changes observed today will only manifest on evolutionary timescales, while others are already impacting climate and biosphere integrity. For instance, Eutrophication resulting from human-induced influxes of N and P can push aquatic and marine systems beyond critical thresholds. This can trigger abrupt, non-linear changes, transitioning from, for instance, a clear-water oligotrophic state to a turbid-water eutrophic state (Rockström et al., 2009).

Biogeochemical flows N cycles Planetary Boundary has been set by restricting the human fixation of N2 from the atmosphere to 35 million tons of nitrogen per year (Rockström et al., 2009). In the case of P flows, it was defined that the global limit for mining and application to soils was 100 teragrams per year (Richardson et al., 2023).

Within the green roofs assessment, this vulnerability is assessed by the LCA impact categories *Acidification*, *Eutrophication* (*marine*), *Eutrophication* (*freshwater*), *Eutrophication* (*terrestrial*).

4.6. Normalization of local-scale indicators

Normalization formula employed in the third step of the NBS-vulnerability framework (Eq. A4):

$$z_i = \frac{y(i) - \min(i)}{\max(i) - \min(i)} \qquad \text{(Eq. A4)}$$

Where:

z: normalized value
i: selected indicator
y: absolute value
min: minimum absolute value or threshold
max: minimum absolute value or threshold

4.7. Normalization of broad-scale indicators

In order to relate the LCA impacts to PB it was followed the methodology proposed by Sala et al. (2020), where PB were directly related to LCA impact categories. The metrics that specifically relate each of the impacts to PB are specified in **Table A8**.

Each of the green roofs LCA impacts was divided by the PB downscaled to Oslo for each of the scenarios, obtaining a ratio over the PB downscaled to Oslo (see **Table A9**). Finally, these ratios were related to the Broad-scale vulnerabilities (based on PB). In those cases that a Broad-scale vulnerability included more than one LCA impact category (e.g., *vulnerability to changes in biogeochemical flows*), impacts were averaged. Final results are presented in **Table 2** from Chapter IV.

Table A8. Planetary boundaries (adapted to the environmental footprint metrics of each LCA impact category

Impact category	Unit	PB	PB per capita	PB downscaled to Oslo
Climate change	kg CO2 eq	6.8E+12	9.9E+02	7.0E+08
Ozone depletion	kg CFC-11 eq	5.4E+08	7.8E-02	5.5E+04
Eutrophication, marine	kg N eq	2.0E+11	2.9E+01	2.1E+07
Eutrophication, freshwater	kg P eq	5.8E+09	8.4E-01	6.0E+05
Eutrophication, terrestrial	molc N eq	6.1E+12	8.9E+02	6.3E+08
Acidification	molc H+ eq	1.0E+12	1.5E+02	1.0E+08
Photochemical ozone formation, human health	kg NMVOC eq	4.1E+11	5.9E+01	4.2E+07
Human toxicity, cancer	CTUh	9.6E+05	1.4E-04	9.9E+01
Human toxicity, non-cancer	CTUh	4.1E+06	5.9E-04	4.2E+02
Ecotoxicity, freshwater	CTUe	1.3E+14	1.9E+04	1.4E+10
Ionizing radiation, human health	kBq 235U eq	5.3E+14	7.6E+04	5.4E+10
Land use	kg soil loss	8.7E+15	1.3E+06	9.0E+11
Water use	m3 world eq	1.8E+14	2.6E+04	1.9E+10
Particulate matter	Disease incidence	5.2E+05	7.5E-05	5.3E+01
Resource use, fossils	MJ	2.2E+14	3.3E+04	2.3E+10
Resource use, mineral and metals	kg Sb eq	2.2E+08	3.2E-02	2.3E+04

Table A9. Calculated green roofs LCA impacts for each and its ratio over the downscaled PB to Oslo for each scenario

			Total impacts				Ratio over downscaled PB to Oslo			
Impact category	**Unit**	**Impact by m2 of green roof**	**S0**	**S1**	**S2**	**S3**	**S0**	**S1**	**S2**	**S3**
Climate change	kg CO2 eq	5.86E-02	1.11E+04	2.44E+04	4.27E+04	6.09E+05	1.59E-05	3.49E-05	6.12E-05	8.72E-04
Ozone depletion	kg CFC11 eq	3.85E-08	7.31E-03	1.60E-02	2.81E-02	4.00E-01	1.32E-07	2.89E-07	5.08E-07	7.23E-06
Human toxicity, cancer	CTUh	3.69E-10	7.01E-05	1.53E-04	2.69E-04	3.83E-03	7.10E-07	1.55E-06	2.73E-06	3.89E-05
Human toxicity, non-cancer	CTUh	9.86E-09	1.87E-03	4.10E-03	7.19E-03	1.02E-01	4.46E-06	9.75E-06	1.71E-05	2.44E-04
Photochemical ozone formation	kg NMVOC eq	2.15E-03	4.08E+02	8.93E+02	1.57E+03	2.23E+04	9.79E-06	2.14E-05	3.76E-05	5.36E-04
Ecotoxicity, freshwater	CTUe	1.94E-01	3.68E+04	8.05E+04	1.41E+05	2.01E+06	2.73E-06	5.98E-06	1.05E-05	1.49E-04
Ionizing radiation	kBq U-235 eq	4.70E-02	8.92E+03	1.95E+04	3.42E+04	4.88E+05	1.65E-07	3.61E-07	6.34E-07	9.03E-06
Acidification	mol H+ eq	2.80E-03	5.32E+02	1.16E+03	2.04E+03	2.91E+04	5.18E-06	1.13E-05	1.99E-05	2.84E-04
Eutrophication, marine	kg N eq	6.60E-04	1.25E+02	2.74E+02	4.81E+02	6.85E+03	6.09E-06	1.33E-05	2.34E-05	3.33E-04
Eutrophication, freshwater	kg P eq	9.23E-05	1.75E+01	3.83E+01	6.73E+01	9.59E+02	2.94E-05	6.44E-05	1.13E-04	1.61E-03
Eutrophication, terrestrial	mol N eq	9.89E-03	1.88E+03	4.11E+03	7.22E+03	1.03E+05	2.99E-06	6.55E-06	1.15E-05	1.64E-04

4.8. Aggregation of indicators for single local-scale vulnerabilities

For aggregating the indicators for obtaining single local-scale vulnerabilities, first, it was carried out and aggregation into a single exposure and a single sensitivity for each vulnerability (Eq. A5):

$$A_{e,s}(V) = \sum_{e,s\,|\mathbf{V}} i \times W_i \qquad \text{(Eq. A5)}$$

Where
A: Aggregated indicators
e: exposure
s: sensitivity
V: selected vulnerability
i: selected indicator
W: associated weight to indicator

Then, it was followed by an aggregation of single exposure and single sensitivity for single vulnerabilities (Eq. A.6):

5. $$V = A_{e|V} x A_{s|V} \qquad \text{(Eq. A6)}$$

Where:
V: single vulnerability
A: Aggregated indicators
e: exposure
s: sensitivity

5.1. Aggregation of indicators for single broad-scale vulnerabilities

For obtaining single broad-scale vulnerabilities, normalized exposure indicators were aggregated following Eq. A5.

5.2. Stakeholder weighting

The workshop "Green roofs in Oslo by 2030: co-creating a common understanding of impacts and relevance for the city" was held on January 29th, 2024, as an online session via Zoom. It gathered 13 stakeholders to discuss the significance of each of the selected vulnerabilities which included academics, municipal officials, NGO representatives, and green roofs experts. Prior to the discussions and weighting exercise, attendees viewed two presentations: first, a presentation on the advances of the Green roof strategy and facades provided by Oslo Municipality representatives, and a second one provided by the researchers carrying on the study and showing the calculated impacts of each of the green roof's scenarios across all vulnerabilities. Following this, participants were divided into 2 groups with a mixed share of all backgrounds to rank relevant vulnerabilities that should be considered when assessing the impacts of implementing green roofs in Oslo. Each group was given 100 pebbles (i.e., points), which needed to be distributed among the different vulnerabilities, based on the premise that the greatest the amount of 'pebbles' a vulnerability received, the most relevant was for the consideration when assessing the impacts of implementing green roofs in Oslo. Participants agreed on the final distribution of the pebbles after discussing their perspectives on the relevance of each vulnerability. Values from the ranking were then converted to percentages.

5.3. Development of most favorable scenario

To determine the quantity of green roofs and their spatial allocation to maximize desired impacts on vulnerabilities while minimizing undesired impacts, first, the impacts on vulnerabilities from S3 (maximization) were considered for each grid pixel. Specifically, the differences in single vulnerabilities (both local and broad-scale) in comparison with S0, bearing in mind that S3 encompass all the possible impacts that green roofs could be generating in both local and broad-scale vulnerabilities.

It is important to take into account that, for all vulnerabilities (local and broad-scale), increases in vulnerabilities are understood as undesired impacts and are portrayed by positive numbers when calculating the differences between the scenarios, while decreases in vulnerabilities are perceived as desired impacts, and represented by negative numbers when calculating the differences between the scenarios (see Fig. **A.10**). Furthermore, different combinations of vulnerabilities shifts can happen across spatial scales resulting in both tradeoffs and synergies (**See Table A10**).

Figure A10. Types of impacts on vulnerabilities, and mathematical values that represent them within the NBS-vulnerability framework

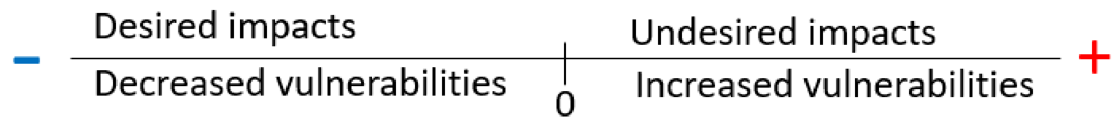


Table A10. Different combinations of vulnerabilities shifts happening across spatial scales and the result of their interactions. The +/- signs represent the way in which vulnerabilities shifts are mathematically represented in the NBS-vulnerability framework

Possible impacts on Local scale vulnerabilities	**Possible impacts on Broad scale vulnerabilities**	**Interaction results in**
Desired (-)	Undesired (+)	Tradeoff
Desired (-)	Desired (-)	Synergy
Undesired (+)	Desired (-)	Tradeoff
Undesired (+)	Undesired (+)	Synergy

After listing all shifts in all single vulnerabilities between S0 and S3 for each pixel, differences were normalized using min-max normalization (see Eq. A7). This is done because differences between vulnerabilities have very different ranges, which makes them unrelatable among each other. By normalizing them, all differences between vulnerabilities become algebraically relatable, while maintaining the proportion of the original difference values. At the end of this process, it is obtained the Normalized vulnerability difference for each single vulnerability between S3 and S0. This approach allows for the values to be relatable among each other, as all of them share a common scale, where -1 represents the greatest decrease in vulnerabilities, 0 represents no change in vulnerabilities and 1 the greatest increase in vulnerability.

$$U = \frac{y(i) - \min(i)}{\max(i) - \min(i)} \qquad \text{(Eq. A7)}$$

Where:
U: Normalized vulnerability difference
i: selected single vulnerability
y: difference in single vulnerability when S3 is compared to S0
min: minimum difference in single vulnerability when S3 is compared to S0
max: maximum difference in single vulnerability when S3 is compared to S0

Following this, the *Normalized vulnerability differences* are merged to obtain a *Combined shift of overall vulnerability*, which includes all shifts in all vulnerabilities. This calculation is made for each pixel. The calculation is done by multiplying each of the *Normalized vulnerability differences* by its corresponding weight (i.e., stakeholder weight or equally distributed weight) and algebraically summing all normalized vulnerability differences, as shown in (Eq. A8). The *Combined shift of overall vulnerability* behaves in the same way that single vulnerabilities (see **Fig. A10**).

$$\Delta\, C\; shift = \sum V \times W_V \qquad \text{(Eq. A8)}$$

Where:
Δ C shift: Combined shift of overall vulnerability
V: single vulnerabilities
W: associated weight to vulnerability

Continuing, for developing the most favorable scenario it is considered that pixels with a negative *Combined shift of overall vulnerability* values represent those with opportunity for the implementation of green roofs, as these offer the opportunity to maximize desired impacts on vulnerabilities while minimizing undesired impacts. Pixels with a *Combined shift of overall vulnerability* equal to 0 are disregarded, as these represent no change in vulnerabilities, as well as those with positive values that represent the greatest undesired impacts on vulnerabilities compared to the desired impacts. Pixels are then sorted based on the *Combined shift of overall vulnerability* from smallest (i.e., greatest negative values) to largest (i.e., greatest positive values). Pixels with the smallest values are those with the highest priority, while those with values of 0 or positive values are considered non-priority.

Finally, it was calculated the amount of green roofs by summing the green roofs m2 that were present in each of the selected pixels, based on the values from S3.

References

Akpinar, A. (2016). How is high school greenness related to students' restoration and health? *Urban Forestry & Urban Greening*, *16*, 1–8. https://doi.org/10.1016/J.UFUG.2016.01.007

Ampim, P. A. Y., Sloan, J. J., Cabrera, R. I., Harp, D. A., & Jaber, F. H. (2010). Green Roof Growing Substrates: Types, Ingredients, Composition and Properties. *Journal of Environmental Horticulture*, *28*(4), 244–252. https://doi.org/10.24266/0738-2898-28.4.244

Artsdatabanken (2020). Data from: BioFokus, GBIF-noder untenfor Norge, Naturhistorik Museum – UiO, Norsk entomologisk forening, Norsk institutt for naturforkning, Tromsø museum – Universitetsmuseet, Universitetsmuseet i Bergen. Downloaded through the Species Map service. https://artsdatabanken.no/

Barboza, E. P., Cirach, M., Khomenko, S., Iungman, T., Mueller, N., Barrera-Gómez, J., Rojas-Rueda, D., Kondo, M., & Nieuwenhuijsen, M. (2021). Green space and mortality in European cities: a health impact assessment study. *The Lancet Planetary Health*, *5*(10), e718–e730. https://doi.org/10.1016/S2542-5196(21)00229-1

Basagaña, X., Sartini, C., Barrera-Gómez, J., Dadvand, P., Cunillera, J., Ostro, B., Sunyer, J., & Medina-Ramón, M. (2011). Heat waves and cause-specific mortality at all ages. *Epidemiology*, *22*(6), 765–772. https://doi.org/10.1097/EDE.0B013E31823031C5

Bastin, J. F., Clark, E., Elliott, T., Hart, S., van den Hoogen, J., Hordijki, I., Ma, H., Majumder, S., Manoli, G., Maschler, J., Mo, L., Routh, D., Yu, K., Zohner, C. M., & Crowther, T. W. (2019). Understanding climate change from a global analysis of city analogues. *PLOS ONE*, *14*(7), e0217592. https://doi.org/10.1371/JOURNAL.PONE.0217592

Berger, T., Chundeli, F. A., Pandey, R. U., Jain, M., Tarafdar, A. K., & Ramamurthy, A. (2022). Low-income residents' strategies to cope with urban heat. *Land Use Policy*, *119*, 106192. https://doi.org/10.1016/J.LANDUSEPOL.2022.106192

Braskerud, B. C., & Paus, K. H. (2022). Retention of snowmelt and rain from extensive green roofs during snow-covered periods. *Blue-Green Systems*, *4*(2), 184–196. https://doi.org/10.2166/BGS.2022.011

Byrne, J. (2012). When green is White: The cultural politics of race, nature and social exclusion in a Los Angeles urban national park. *Geoforum*, *43*(3), 595–611. https://doi.org/10.1016/J.GEOFORUM.2011.10.002

Chakraborty, T., Hsu, A., Manya, D., & Sheriff, G. (2019). Disproportionately higher exposure to urban heat in lower-income neighborhoods: a multi-city perspective. *Environmental Research Letters*, *14*(10), 105003. https://doi.org/10.1088/1748-9326/AB3B99

Chen, Y., Ge, Y., Yang, G., Wu, Z., Du, Y., Mao, F., Liu, S., Xu, R., Qu, Z., Xu, B., & Chang, J. (2022). Inequalities of urban green space area and ecosystem services along urban center-edge gradients. *Landscape and Urban Planning*, *217*, 104266. https://doi.org/10.1016/J.LANDURBPLAN.2021.104266

Clark, M. J. (1998). Flood Insurance as a Management Strategy for UK Coastal Resilience. *The Geographical Journal*, *164*(3), 333. https://doi.org/10.2307/3060621

Cutter, S. L. (2016). Resilience to What? Resilience for Whom? *The Geographical Journal*, *182*(2), 110–113. https://doi.org/10.1111/GEOJ.12174

Dargas, J. H. F., Chaves, S. R., & Fischer, E. (2016). Pollination of lark daisy on roadsides declines as traffic speed increases along an Amazonian highway. *Plant Biology*, *18*(3), 542–544. https://doi.org/10.1111/PLB.12437

Demoury, C., Aerts, R., Berete, F., Lefebvre, W., Pauwels, A., Vanpoucke, C., Van der Heyden, J., & De Clercq, E. M. (2024). Impact of short-term exposure to air pollution on natural mortality and vulnerable populations: a multi-city case-crossover analysis in Belgium. *Environmental Health: A Global Access Science Source*, *23*(1), 1–10. https://doi.org/10.1186/S12940-024-01050-W/TABLES/2

European Environment Agency. (2023). *Heat and cold — extreme heat.* https://www.eea.europa.eu/publications/europes-changing-climate-hazards-1/heat-and-cold/heat-and-cold-extreme-heat

Faber Taylor, A., & Kuo, F. E. M. (2011). Could Exposure to Everyday Green Spaces Help Treat ADHD? Evidence from Children's Play Settings. *Applied Psychology: Health and Well-Being*, *3*(3), 281–303. https://doi.org/10.1111/J.1758-0854.2011.01052.X

FLL. (2018). *Green roof guidelines - Guidelines for the planning, construction and maintenance of green roofs*.

Friedrich, C. (2008). Selecting the Proper Components for a Green Roof Growing Media. *Low Impact Development: New and Continuing Applications - Proceedings of the 2nd National Low Impact Development Conference 2007*, *331*, 240–251. https://doi.org/10.1061/41007(331)22

Gagnon, C., Dagenais, D., & Côté, V. (2018). A Place to Be or, at Least, a Space to See: a qualitative inquiry on the experience and appreciation of extensive green roofs. *DRS Biennial Conference Series*, *1*. https://doi.org/10.21606/DRS.2018.358

Girling, R. D., Lusebrink, I., Farthing, E., Newman, T. A., & Poppy, G. M. (2013). Diesel exhaust rapidly degrades floral odours used by honeybees. *Scientific Reports 2013 3:1*, *3*(1), 1–5. https://doi.org/10.1038/srep02779

Gómez-Baggethun, E., & Barton, D. N. (2013). Classifying and valuing ecosystem services for urban planning. *Ecological Economics*, *86*, 235–245. https://doi.org/10.1016/J.ECOLECON.2012.08.019

Gourdji, S. (2018). Review of plants to mitigate particulate matter, ozone as well as nitrogen dioxide air pollutants and applicable recommendations for green roofs in Montreal, Quebec. *Environmental Pollution*, *241*, 378–387. https://doi.org/10.1016/J.ENVPOL.2018.05.053

Guan, W., Liu, Q., & Dong, C. (2022). Risk assessment method for industrial accident consequences and human vulnerability in urban areas. *Journal of Loss Prevention in the Process Industries*, *76*, 104745. https://doi.org/10.1016/J.JLP.2022.104745

Hajat, S., O'Connor, M., & Kosatsky, T. (2010). Health effects of hot weather: from awareness of risk factors to effective health protection. *The Lancet*, *375*(9717), 856–863. https://doi.org/10.1016/S0140-6736(09)61711-6

Heaviside, C., Vardoulakis, S., & Cai, X. M. (2016). Attribution of mortality to the urban heat island during heatwaves in the West Midlands, UK. *Environmental Health: A Global Access Science Source*, *15*(1), 49–59. https://doi.org/10.1186/S12940-016-0100-9/TABLES/3

Hossain, M. K., & Meng, Q. (2020). A thematic mapping method to assess and analyze potential urban hazards and risks caused by flooding. *Computers, Environment and Urban Systems*, *79*, 101417. https://doi.org/10.1016/J.COMPENVURBSYS.2019.101417

Huang, C., Yang, J., Clinton, N., Yu, L., Huang, H., Dronova, I., & Jin, J. (2021). Mapping the maximum extents of urban green spaces in 1039 cities using dense satellite images. *Environmental Research Letters*, *16*(6), 064072. https://doi.org/10.1088/1748-9326/AC03DC

ICGC - Institut Cartogràfic i Geològic de Catalunya. (2021). *NDVI Índex de vegetació de diferència normalitzada*. https://www.icgc.cat/ca/Geoinformacio-i-mapes/Dades-i-productes/Imatge/NDVI

Jaffal, I., Ouldboukhitine, S. E., & Belarbi, R. (2012). A comprehensive study of the impact of green roofs on building energy performance. *Renewable Energy*, *43*, 157–164. https://doi.org/10.1016/J.RENENE.2011.12.004

Ji, P., Sæbø, A., Stovin, V., & Hanslin, H. M. (2018). Sedum root foraging in layered green roof substrates. *Plant and Soil*, *430*(1–2), 263–276. https://doi.org/10.1007/S11104-018-3729-Z/FIGURES/5

Kabisch, N., van den Bosch, M., & Lafortezza, R. (2017). The health benefits of nature-based solutions to urbanization challenges for children and the elderly – A systematic review. *Environmental Research*, *159*, 362–373. https://doi.org/10.1016/J.ENVRES.2017.08.004

Kallioniemi, E., Åström, J., Rusch, G. M., Dahle, S., Åström, S., & Gjershaug, J. O. (2017). Local resources, linear elements and mass-flowering crops determine bumblebee occurrences in moderately intensified farmlands. *Agriculture, Ecosystems & Environment*, *239*, 90–100. https://doi.org/10.1016/J.AGEE.2016.12.039

Kaźmierczak, A., & Cavan, G. (2011). Surface water flooding risk to urban communities: Analysis of vulnerability, hazard and exposure. *Landscape and Urban Planning*, *103*(2), 185–197. https://doi.org/10.1016/J.LANDURBPLAN.2011.07.008

Konijnendijk, C. C. (2023). Evidence-based guidelines for greener, healthier, more resilient neighbourhoods: Introducing the 3–30–300 rule. *Journal of Forestry Research*, *34*(3), 821–830. https://doi.org/10.1007/S11676-022-01523-Z/FIGURES/1

Kovats, R. S., & Kristie, L. E. (2006). Heatwaves and public health in Europe. *European Journal of Public Health*, *16*(6), 592–599. https://doi.org/10.1093/EURPUB/CKL049

Kratschmer, S., Kriechbaum, M., & Pachinger, B. (2018). Buzzing on top: Linking wild bee diversity, abundance and traits with green roof qualities. *Urban Ecosystems*, *21*(3), 429–446. https://doi.org/10.1007/S11252-017-0726-6/FIGURES/6

Liu, H., Kong, F., Yin, H., Middel, A., Zheng, X., Huang, J., Xu, H., Wang, D., & Wen, Z. (2021). Impacts of green roofs on water, temperature, and air quality: A bibliometric review. *Building and Environment*, *196*, 107794. https://doi.org/10.1016/J.BUILDENV.2021.107794

Makri, A., & Stilianakis, N. I. (2008). Vulnerability to air pollution health effects. *International Journal of Hygiene and Environmental Health*, *211*(3–4), 326–336. https://doi.org/10.1016/J.IJHEH.2007.06.005

Marttunen, M., Lienert, J., & Belton, V. (2017). Structuring problems for Multi-Criteria Decision Analysis in practice: A literature review of method combinations. *European Journal of Operational Research*, *263*(1), 1–17. https://doi.org/10.1016/J.EJOR.2017.04.041

Marvuglia, A., Koppelaar, R., & Rugani, B. (2020). The effect of green roofs on the reduction of mortality due to heatwaves: Results from the application of a spatial microsimulation model to four European cities. *Ecological Modelling*, *438*, 109351. https://doi.org/10.1016/J.ECOLMODEL.2020.109351

Mesimäki, M., Hauru, K., Kotze, D. J., & Lehvävirta, S. (2017). Neo-spaces for urban livability? Urbanites' versatile mental images of green roofs in the Helsinki metropolitan area, Finland. *Land Use Policy*, *61*, 587–600. https://doi.org/10.1016/J.LANDUSEPOL.2016.11.021

Miljødirektoratet. (2023). *Lokal luftforurensning: Utslippssystem og database*. Lokal Luftforurensning: Utslippssystem Og Database. utslippssystem.miljodirektoratet.no

Morrow, B. H. (1999). Identifying and Mapping Community Vulnerability. *Disasters*, *23*(1), 1–18. https://doi.org/10.1111/1467-7717.00102

Næss, P. (2022). Compact urban development in Norway: Spatial changes, underlying policies and travel impacts. *Advances in Transport Policy and Planning*, *9*, 95–133. https://doi.org/10.1016/BS.ATPP.2021.02.003

Nagase, A., & Dunnett, N. (2011). The relationship between percentage of organic matter in substrate and plant growth in extensive green roofs. *Landscape and Urban Planning*, *103*(2), 230–236. https://doi.org/10.1016/J.LANDURBPLAN.2011.07.012

Nieto, A. (2014). *European Red List of bees*. International Union for Conservation of Nature. https://doi.org/https://doi.org/10.2779/51181

Noreng, K., Kvalvik, M., Busklein, J. O., Ødegård, I. M., Clewing, C. S., & French, H. K. (2012). Grønne tak. Resultater fra et kunnskapsinnhentingsprosjekt.

Norwegian Directorate for Civil Protection. (2016). *Samfunnets kritiske funksjoner*

Norwegian Directorate for Civil Protection. (2019). *Analyses of Crisis Scenarios*.

OECD, & European Union. (2008). *Handbook on Constructing Composite Indicators: Methodology and User Guide*. OECD. https://doi.org/10.1787/9789264043466-en

OpenStreetMap Contributors. 2020. Map of Oslo, OpenStreetMap, https://www.openstreetmap.org. Accessed on October 23, 2023

Oslo Kommune. (2012). *Strategi for overvannshåndtering i oslo*.

Oslo kommune. (2015). *Oslo 2030: Smart, trygg og grønn*.

Oslo Kommune. (2017). *Handlingsprogram for økt byliv i Oslo sentrum*.

Oslo Kommune. (2020). *Climate change vulnerability analysis for Oslo*.

Oslo Kommune. (2021a). *Strategi for grønne tak og fasader: Hovedresultater fra Plan- og bygningsetatens kartlegging av grønne tak*

Oslo Kommune. (2021b). *Tiltaksutredning for bedre luftkvalitet i Oslo og Bærum 2020-2025*.

Oslo Kommune. (2022). *Strategi for grønne tak og fasader: 2030 grønne tak og fasader i 2030*.

Oslo Kommune. (2023a). *Handlingsplan for biologisk mangfold i Oslo 2023–2030*.

Oslo Kommune. (2023b). *Overvannsveileder: Retningslinjer og veiledning for overvannshåndtering i Oslo kommune*.

Oudin Åström, D., Bertil, F., & Joacim, R. (2011). Heat wave impact on morbidity and mortality in the elderly population: A review of recent studies. *Maturitas*, *69*(2), 99–105. https://doi.org/10.1016/j.maturitas.2011.03.008

Passaseo, A., Pétremand, G., Rochefort, S., & Castella, E. (2020). Pollinator emerging from extensive green roofs: wild bees (Hymenoptera, Antophila) and hoverflies (Diptera, Syrphidae) in Geneva

(Switzerland). *Urban Ecosystems*, *23*(5), 1079–1086. https://doi.org/10.1007/S11252-020-00973-9/TABLES/4

Richardson, K., Steffen, W., Lucht, W., Bendtsen, J., Cornell, S. E., Donges, J. F., Drüke, M., Fetzer, I., Bala, G., von Bloh, W., Feulner, G., Fiedler, S., Gerten, D., Gleeson, T., Hofmann, M., Huiskamp, W., Kummu, M., Mohan, C., Nogués-Bravo, D., … Rockström, J. (2023). Earth beyond six of nine planetary boundaries. *Science Advances*, *9*(37). https://doi.org/10.1126/SCIADV.ADH2458/SUPPL_FILE/SCIADV.ADH2458_SM.PDF

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F. S., Lambin, E. F., Lenton, T. M., Scheffer, M., Folke, C., Schellnhuber, H. J., Nykvist, B., De Wit, C. A., Hughes, T., Van Der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P. K., Costanza, R., Svedin, U., … Foley, J. A. (2009). A safe operating space for humanity. *Nature 2009 461:7263*, *461*(7263), 472–475. https://doi.org/10.1038/461472a

Sælthun, N. R., Barton, D. N., & Venter, Z. S. (2021). *REO: estimering av overflateavrenning fra urbane felt. Beregningsgrunnlag for et arealdifferensiert overvannsgebyr (revidert utgave).* https://www.mendeley.com/reference-manager/reader-v2/706bcb4d-75c5-3072-873c-c1c86674f1ff/3b63f83a-b93e-f96d-2af6-7fcd08f66b4b

Sala, S., Crenna, E., Secchi, M., & Sanyé-Mengual, E. (2020). Environmental sustainability of European production and consumption assessed against planetary boundaries. *Journal of Environmental Management*, *269*, 110686. https://doi.org/10.1016/J.JENVMAN.2020.110686

Segura, R., Badia, A., Ventura, S., Gilabert, J., Martilli, A., & Villalba, G. (2021). Sensitivity study of PBL schemes and soil initialization using the WRF-BEP-BEM model over a Mediterranean coastal city. *Urban Climate*, *39*, 100982. https://doi.org/10.1016/J.UCLIM.2021.100982

Sera, F., Armstrong, B., Tobias, A., Vicedo-Cabrera, A. M., Åström, C., Bell, M. L., Chen, B. Y., De Sousa Zanotti Stagliorio Coelho, M., Correa, P. M., Cruz, J. C., Dang, T. N., Hurtado-Diaz, M., Do Van, D., Forsberg, B., Guo, Y. L., Guo, Y., Hashizume, M., Honda, Y., Iñiguez, C., … Gasparrini, A. (2019). How urban characteristics affect vulnerability to heat and cold: a multi-country analysis. *International Journal of Epidemiology*, *48*(4), 1101–1112. https://doi.org/10.1093/IJE/DYZ008

Shafique, M., Kim, R., & Kyung-Ho, K. (2018). Green Roof for Stormwater Management in a Highly Urbanized Area: The Case of Seoul, Korea. *Sustainability 2018, Vol. 10, Page 584*, *10*(3), 584. https://doi.org/10.3390/SU10030584

Skamarock, W. C., Klemp, J. B., Dudhia, J., Gill, D. O., Liu, Z., Berner, J., Wang, W., Powers, J. G., Duda, M. G., Barker, D. M., & Huang, X.-Y. (2021). *A Description of the Advanced Research WRF Model Version 4.3*. https://doi.org/10.5065/1DFH-6P97

Smith, C. L., Webb, A., Levermore, G. J., Lindley, S. J., & Beswick, K. (2011). Fine-scale spatial temperature patterns across a UK conurbation. *Climatic Change*, *109*(3–4), 269–286. https://doi.org/10.1007/S10584-011-0021-0

Speak, A. F., Rothwell, J. J., Lindley, S. J., & Smith, C. L. (2012). Urban particulate pollution reduction by four species of green roof vegetation in a UK city. *Atmospheric Environment*, *61*, 283–293. https://doi.org/10.1016/J.ATMOSENV.2012.07.043

Statistikkbanken Oslo kommune. (2020). Gjennomsnittlig konsentrasjon av PM10 (svevestøv) ved de ulike målestasjonene i Oslo. Retrieved from https://statistikkbanken.oslo.kommune.no/webview/. Accessed on May 12, 2023

Statistikkbanken Oslo kommune. (2023). Oslo kommunes befolkningsstatistikk 2020. Retrieved from https://statistikkbanken.oslo.kommune.no/webview/. Accessed on May 12, 2023

Statistikkbanken Oslo kommune. (2023b). Husholdninger med lavinntekt etter husholdningstyper, med og uten korrigering for formue. Retrieved from https://statistikkbanken.oslo.kommune.no/statbank/pxweb/no/db1/db1__Inntekt/OK-INN007.px//. Accessed on May 12, 2023

Standard Norge. (2015). *Green roofs - Planning, design, execution, maintenance and operation. NS 3840:2015.*

Stange, E. E., Zulian, G., Rusch, G. M., Barton, D. N., & Nowell, M. (2017). Ecosystem services mapping for municipal policy: ESTIMAP and zoning for urban beekeeping. *One Ecosystem*, *2*. https://doi.org/10.3897/oneeco.2.e14014

Strand, N. P. (2023, June 5). *Befolkningsframskrivingen for Oslo 2023–2050*. Oslo Kommune. https://www.oslo.kommune.no/statistikk/befolkning/befolkningsframskrivingen-for-oslo-2023-2050

Tapsell, S. M., Penning-Rowsell, E. C., Tunstall, S. M., & Wilson, T. L. (2002). Vulnerability to flooding: health and social dimensions. *Philosophical Transactions of the Royal Society of London. Series A: Mathematical, Physical and Engineering Sciences*, *360*(1796), 1511–1525. https://doi.org/10.1098/RSTA.2002.1013

Taylor, J., McLeod, R., Petrou, G., Hopfe, C., Mavrogianni, A., Castaño-Rosa, R., Pelsmakers, S., & Lomas, K. (2023). Ten questions concerning residential overheating in Central and Northern Europe. *Building and Environment*, *234*, 110154. https://doi.org/https://doi.org/10.1016/j.buildenv.2023.110154

Tian, Z., Zhang, S., Deng, J., & Hrynyszyn, B. D. (2020). Evaluation on Overheating Risk of a Typical Norwegian Residential Building under Future Extreme Weather Conditions. *Energies 2020, Vol. 13, Page 658*, *13*(3), 658. https://doi.org/10.3390/EN13030658

Tibuakuu, M., Michos, E. D., Navas-Acien, A., & Jones, M. R. (2018). Air Pollution and Cardiovascular Disease: a Focus on Vulnerable Populations Worldwide. *Current Epidemiology Reports 2018 5:4*, *5*(4), 370–378. https://doi.org/10.1007/S40471-018-0166-8

UK Health Security Agency. (2022). *Heatwave plan for England*. https://beta.southglos.gov.uk/static/501386ef062f383923501c6a1cfc0082/heatwave-plan-for-England-2022-5-August-2022.pdf

URBAG. (2021). *Green roofs in Oslo: results from a co-creative and participatory approach for its evaluation*. https://urbag.eu/green-roofs-in-oslo-co-creative-approach/

Vanos, J. K. (2015). Children's health and vulnerability in outdoor microclimates: A comprehensive review. *Environment International*, *76*, 1–15. https://doi.org/10.1016/J.ENVINT.2014.11.016

Venter, Z. S., Barton, D. N., Martinez-Izquierdo, L., Langemeyer, J., Baró, F., & McPhearson, T. (2021). Interactive spatial planning of urban green infrastructure – Retrofitting green roofs where ecosystem services are most needed in Oslo. *Ecosystem Services*, *50*(June), 101314. https://doi.org/10.1016/j.ecoser.2021.101314

Wang, J., Xu, C., Pauleit, S., Kindler, A., & Banzhaf, E. (2019). Spatial patterns of urban green infrastructure for equity: A novel exploration. *Journal of Cleaner Production*, *238*, 117858. https://doi.org/10.1016/J.JCLEPRO.2019.117858

Wang, X. Y., Barnett, A. G., Yu, W., FitzGerald, G., Tippett, V., Aitken, P., Neville, G., McRae, D., Verrall, K., & Tong, S. (2012). The impact of heatwaves on mortality and emergency hospital admissions from non-external causes in Brisbane, Australia. *Occupational and Environmental Medicine*, *69*(3), 163–169. https://doi.org/10.1136/OEM.2010.062141

Wolch, J. R., Byrne, J., & Newell, J. P. (2014). Urban green space, public health, and environmental justice: The challenge of making cities 'just green enough.' *Landscape and Urban Planning*, *125*, 234–244. https://doi.org/10.1016/J.LANDURBPLAN.2014.01.017

Xu, C., Haase, D., Pribadi, D. O., & Pauleit, S. (2018). Spatial variation of green space equity and its relation with urban dynamics: A case study in the region of Munich. *Ecological Indicators*, *93*, 512–523. https://doi.org/10.1016/J.ECOLIND.2018.05.024

Xu, Y., Dadvand, P., Barrera-Gómez, J., Sartini, C., Marí-Dell'Olmo, M., Borrell, C., Medina-Ramón, M., Sunyer, J., & Basagaña, X. (2013). Differences on the effect of heat waves on mortality by sociodemographic and urban landscape characteristics. *J Epidemiol Community Health*, *67*(6), 519–525. https://doi.org/10.1136/JECH-2012-201899

2-B. LCA impacts and inventory

1. Lifecycle assessment impacts of green roof layers (absolute values)

		FILTER LAYER					WATER RETENTION					Root barrier				
Impact category	**Unit**	**Production**	**Installation**	**Use**	**End of life**	**Total**	**Production**	**Installation**	**Use**	**End of life**	**Total**	**Production**	**Installation**	**Use**	**End of life**	**Total**
Climate change	kg CO2 eq	1,73E-02	7,05E-05	0,00E+00	3,79E-03	2,12E-02	1,51E-02	4,51E-04	0,00E+00	2,42E-02	3,98E-02	5,81E-02	2,82E-04	0,00E+00	3,03E-02	8,87E-02
Ozone depletion	kg CFC11 eq	2,71E-10	1,36E-11	0,00E+00	3,14E-11	3,16E-10	7,06E-10	8,68E-11	0,00E+00	2,01E-10	9,94E-10	6,05E-10	5,43E-11	0,00E+00	7,91E-12	6,67E-10
Human toxicity, cancer	CTUh	1,12E-11	3,72E-14	0,00E+00	6,99E-13	1,20E-11	1,36E-11	2,38E-13	0,00E+00	4,47E-12	1,83E-11	3,44E-11	1,49E-13	0,00E+00	2,12E-12	3,66E-11
Human toxicity, non-cancer	CTUh	2,69E-10	1,20E-12	0,00E+00	9,11E-12	2,79E-10	3,41E-10	7,66E-12	0,00E+00	5,83E-11	4,07E-10	8,00E-10	4,79E-12	0,00E+00	9,24E-11	8,98E-10
Photochemical ozone formation	kg NMVOC eq	6,20E-05	6,58E-07	0,00E+00	1,67E-05	7,93E-05	5,86E-05	4,21E-06	0,00E+00	1,07E-04	1,69E-04	2,49E-04	2,63E-06	0,00E+00	5,06E-06	2,56E-04
Ecotoxicity, freshwater	CTUe	4,83E-03	4,88E-05	0,00E+00	5,69E-05	4,94E-03	1,02E-02	3,12E-04	0,00E+00	3,64E-04	1,09E-02	1,43E-02	1,95E-04	0,00E+00	4,61E-05	1,46E-02
Ionising radiation	kBq U-235 eq	1,54E-03	8,28E-06	0,00E+00	1,04E-05	1,55E-03	3,99E-03	5,30E-05	0,00E+00	6,65E-05	4,11E-03	5,47E-03	3,31E-05	0,00E+00	1,21E-05	5,52E-03
Acidification	mol H+ eq	7,80E-05	5,23E-07	0,00E+00	1,36E-05	9,22E-05	7,13E-05	3,35E-06	0,00E+00	8,71E-05	1,62E-04	2,49E-04	2,09E-06	0,00E+00	3,99E-06	2,55E-04
Eutrophication, marine	kg N eq	1,60E-05	2,18E-07	0,00E+00	1,01E-05	2,63E-05	1,77E-05	1,40E-06	0,00E+00	6,45E-05	8,36E-05	5,07E-05	8,74E-07	0,00E+00	2,00E-06	5,35E-05
Eutrophication, freshwater	kg P eq	4,54E-06	9,69E-09	0,00E+00	8,31E-08	4,64E-06	6,12E-06	6,20E-08	0,00E+00	5,32E-07	6,71E-06	1,55E-05	3,88E-08	0,00E+00	5,13E-08	1,56E-05
Eutrophication, terrestrial	mol N eq	1,67E-04	2,38E-06	0,00E+00	6,82E-05	2,37E-04	1,82E-04	1,53E-05	0,00E+00	4,36E-04	6,34E-04	5,22E-04	9,53E-06	0,00E+00	2,04E-05	5,51E-04

2. Lifecycle assessment impacts of green roof substrate (Absolute values)

		PUMICE					GRAVEL					COMPOST				
Impact category	**Unit**	**Production**	**Installation**	**Use**	**End of life**	**Total**	**Production**	**Installation**	**Use**	**End of life**	**Total**	**Production**	**Installation**	**Use**	**End of life**	**Total**
Climate change	kg CO2 eq	7,55E-02	1,13E-02	0,00E+00	5,42E-03	9,22E-02	4,82E-02	7,55E-03	0,00E+00	3,64E-03	5,94E-02	2,59E-02	1,43E-03	0,00E+00	2,91E-02	5,64E-02
Ozone depletion	kg CFC11 eq	1,34E-08	2,17E-09	0,00E+00	1,51E-10	1,58E-08	9,21E-09	1,45E-09	0,00E+00	1,01E-10	1,08E-08	1,24E-09	2,75E-10	0,00E+00	7,53E-11	1,59E-09
Human toxicity, cancer	CTUh	4,29E-11	5,94E-12	0,00E+00	4,66E-12	5,35E-11	2,56E-11	3,98E-12	0,00E+00	3,12E-12	3,27E-11	4,23E-12	7,53E-13	0,00E+00	1,59E-11	2,09E-11
Human toxicity, non-cancer	CTUh	1,20E-09	1,91E-10	0,00E+00	7,25E-11	1,47E-09	7,92E-10	1,28E-10	0,00E+00	4,86E-11	9,69E-10	1,71E-10	2,42E-11	0,00E+00	5,63E-10	7,59E-10
Photochemical ozone formation	kg NMVOC eq	3,67E-04	1,05E-04	0,00E+00	5,39E-05	5,26E-04	2,24E-04	7,05E-05	0,00E+00	3,61E-05	3,30E-04	6,13E-05	1,33E-05	0,00E+00	1,99E-05	9,45E-05
Ecotoxicity, freshwater	CTUe	6,13E-02	7,79E-03	0,00E+00	3,49E-03	7,26E-02	3,34E-02	5,22E-03	0,00E+00	2,34E-03	4,09E-02	4,53E-03	9,88E-04	0,00E+00	2,41E-04	5,75E-03
Ionising radiation	kBq U-235 eq	7,22E-03	1,32E-03	0,00E+00	4,31E-04	8,98E-03	8,30E-03	8,87E-04	0,00E+00	2,89E-04	9,47E-03	1,15E-03	1,68E-04	0,00E+00	9,08E-05	1,41E-03
Acidification	mol H+ eq	3,18E-04	8,35E-05	0,00E+00	3,98E-05	4,41E-04	2,01E-04	5,60E-05	0,00E+00	2,67E-05	2,83E-04	2,10E-04	1,06E-05	0,00E+00	1,65E-05	2,38E-04
Eutrophication, marine	kg N eq	1,07E-04	3,49E-05	0,00E+00	1,46E-05	1,57E-04	6,67E-05	2,34E-05	0,00E+00	9,76E-06	9,98E-05	2,44E-05	4,42E-06	0,00E+00	8,94E-06	3,78E-05
Eutrophication, freshwater	kg P eq	7,92E-06	1,55E-06	0,00E+00	9,35E-07	1,04E-05	5,66E-06	1,04E-06	0,00E+00	6,27E-07	7,33E-06	7,69E-07	1,96E-07	0,00E+00	2,20E-06	3,17E-06
Eutrophication, terrestrial	mol N eq	1,17E-03	3,81E-04	0,00E+00	1,55E-04	1,71E-03	7,38E-04	2,55E-04	0,00E+00	1,04E-04	1,10E-03	9,43E-04	4,83E-05	0,00E+00	7,59E-05	1,07E-03

Lifecycle assessment impacts of green roof substrate (Absolute values) – continuation

FERTILIZER

Impact category	Unit	Production	Installation	Use	End of life	Total
Climate change	kg CO2 eq	7,55E-02	1,13E-02	0,00E+00	5,42E-03	9,22E-02
Ozone depletion	kg CFC11 eq	1,34E-08	2,17E-09	0,00E+00	1,51E-10	1,58E-08
Human toxicity, cancer	CTUh	4,29E-11	5,94E-12	0,00E+00	4,66E-12	5,35E-11
Human toxicity, non-cancer	CTUh	1,20E-09	1,91E-10	0,00E+00	7,25E-11	1,47E-09
Photochemical ozone formation	kg NMVOC eq	3,67E-04	1,05E-04	0,00E+00	5,39E-05	5,26E-04
Ecotoxicity, freshwater	CTUe	6,13E-02	7,79E-03	0,00E+00	3,49E-03	7,26E-02
Ionising radiation	kBq U-235 eq	7,22E-03	1,32E-03	0,00E+00	4,31E-04	8,98E-03
Acidification	mol H+ eq	3,18E-04	8,35E-05	0,00E+00	3,98E-05	4,41E-04
Eutrophication, marine	kg N eq	1,07E-04	3,49E-05	0,00E+00	1,46E-05	1,57E-04
Eutrophication, freshwater	kg P eq	7,92E-06	1,55E-06	0,00E+00	9,35E-07	1,04E-05
Eutrophication, terrestrial	mol N eq	1,17E-03	3,81E-04	0,00E+00	1,55E-04	1,71E-03

3. Lifecycle assessment impacts of green roof layers & substrate (Absolute values)

		Layers					Substrate					Total				
Impact category	Unit	Production	Installation	Use	End of life	Total	Production	Installation	Use	End of life	Total	Production	Installation	Use	End of life	Total
Climate change	kg CO2 eq	9,06E-02	8,04E-04	0,00E+00	5,83E-02	1,50E-01	1,50E-01	3,60E-02	2,02E-01	3,81E-02	4,26E-01	2,40E-01	3,68E-02	2,02E-01	9,64E-02	5,76E-01
Ozone depletion	kg CFC11 eq	1,58E-09	1,55E-10	0,00E+00	2,41E-10	1,98E-09	2,39E-08	7,02E-09	5,30E-09	3,28E-10	3,65E-08	2,55E-08	7,17E-09	5,30E-09	5,69E-10	3,85E-08
Human toxicity, cancer	CTUh	5,92E-11	4,24E-13	0,00E+00	7,29E-12	6,69E-11	7,27E-11	1,79E-11	1,88E-10	2,37E-11	3,02E-10	1,32E-10	1,83E-11	1,88E-10	3,10E-11	3,69E-10
Human toxicity, non-cancer	CTUh	1,41E-09	1,36E-11	0,00E+00	1,60E-10	1,58E-09	2,17E-09	5,81E-10	4,84E-09	6,85E-10	8,28E-09	3,58E-09	5,95E-10	4,84E-09	8,44E-10	9,86E-09
Photochemical ozone formation	kg NMVOC eq	3,69E-04	7,50E-06	0,00E+00	1,28E-04	5,05E-04	6,52E-04	2,62E-04	6,21E-04	1,10E-04	1,64E-03	1,02E-03	2,69E-04	6,21E-04	2,38E-04	2,15E-03
Ecotoxicity, freshwater	CTUe	2,94E-02	5,56E-04	0,00E+00	4,67E-04	3,04E-02	9,92E-02	2,52E-02	3,29E-02	6,07E-03	1,63E-01	1,29E-01	2,57E-02	3,29E-02	6,53E-03	1,94E-01
Ionising radiation	kBq U-235 eq	1,10E-02	9,44E-05	0,00E+00	8,89E-05	1,12E-02	1,67E-02	3,86E-03	1,44E-02	8,10E-04	3,58E-02	2,77E-02	3,96E-03	1,44E-02	8,99E-04	4,70E-02
Acidification	mol H+ eq	3,99E-04	5,96E-06	0,00E+00	1,05E-04	5,09E-04	7,29E-04	2,12E-04	1,27E-03	8,30E-05	2,29E-03	1,13E-03	2,18E-04	1,27E-03	1,88E-04	2,80E-03
Eutrophication, marine	kg N eq	8,43E-05	2,49E-06	0,00E+00	7,66E-05	1,63E-04	1,99E-04	8,41E-05	1,80E-04	3,33E-05	4,96E-04	2,83E-04	8,66E-05	1,80E-04	1,10E-04	6,60E-04
Eutrophication, freshwater	kg P eq	2,61E-05	1,11E-07	0,00E+00	6,67E-07	2,69E-05	1,43E-05	4,18E-06	4,31E-05	3,76E-06	6,54E-05	4,05E-05	4,29E-06	4,31E-05	4,43E-06	9,23E-05
Eutrophication, terrestrial	mol N eq	8,70E-04	2,72E-05	0,00E+00	5,25E-04	1,42E-03	2,85E-03	9,18E-04	4,37E-03	3,35E-04	8,47E-03	3,72E-03	9,46E-04	4,37E-03	8,60E-04	9,89E-03

4. Lifecycle inventory

1 m² of extensive green roof							Construction	Operation	Deconstruction	Total years 40	
Process	Layer	Element	Material / processes	Lifetime (year)	Aux value	Aux Unit	C, Per m2·y	O, Per m2·y	D, Per m2·y	Per lifetime	Unit
Layers	Filter Layer	Non woven polypropylene	Non woven polypoprylene	40	0,2	kg/m²	0,2			0,2	kg
			Process : extrusion							0,2	kg
			Transport		500	km				0,1	tkm
			Machinery for construction / deconstruction (1/2): tower crane		0,0039	kWh/kg	0,00078		0,00078	0,00156	kWh
			End of life: transport lorry to recycling		100	km				0,02	tkm
			Incineration		100	%				0,2	kg
	Water retention	Felt	Recycled textile fiber	40	1,28	kg/m²	1,28			1,28	kg
			Transport		500	km				0,64	tkm
			Machinery for construction / deconstruction (1/2): tower crane		0,0039	kWh/kg	0,004992		0,004992	0,009984	kWh
			End of life: transport lorry to waste treatment plan		100	km				0,128	tkm
			Incineration							1,28	kg
	Root Barrier	Polyethylene (LPDE)	Polyethylene, low density - LDPE	40	0,8	kg/m²	0,8			0,8	kg
			Process : extrusion							0,8	kg
			Transport		500	km				0,4	tkm
			Machinery for construction / deconstruction (1/2): tower crane		0,023	kWh/kg	0,00312		0,00312	0,00624	kWh
			End of life: transport lorry to waste treatment plan		100	km				0,08	tkm
			Incineration		50	%				0,4	kg
			Recycling		50	%				0,4	kg

Lifecycle inventory (continuation)

				Simapro input	Comment
Process	**Layer**	**Element**	**Source**	**Ecoinvent**	**Note & assumption**
Layers	**Filter Layer**	Non woven polypropylene	Chenani et al., 2015	Textile, nonwoven polypropylene {GLO}| market for textile, nonwoven polypropylene | Cut-off, S	
			Chenani et al., 2015	Extrusion, plastic film {RER}| extrusion, plastic film | Cut-off, S	
			Own calculation	Transport, lorry >32t, EURO5/RER S	0.2kg/1000 = 0.0002 t --> 0.0002t*500km (distance from the supplier in Sweden to the instalation site) = 0.1 tkm
			Own calculation	Electricity, low voltage {Europe without Switzerland}| market group for | Cut-off, S	Tower crane -> 0.0039kWh/kg * 0.2kg = 0.00078 kWh * 2 (construction/deconstruction) = 0.00156 kWh
			Own calculation	Transport, lorry >16t, fleet average/RER S	0.2kg/1000 = 0.0002 t --> 0.0002t*100km (distance from the site to the waste treatment plan) = 0.02tkm
			Chenani et al., 2015	Waste textile, soiled {RoW}| treatment of, municipal incineration | Cut-off, S	
	Water retention	Felt	Braskerud, Bent C. (2014).	FELT-WATER RETENTION	
			Own calculation	Transport, lorry >32t, EURO5/RER S	1,28kg/1000 = 0,00128 t --> 0,00128t*500km (distance from the supplier in Sweden to the instalation site) = 0,64 tkm
			Own calculation	Electricity, low voltage {Europe without Switzerland}| market group for | Cut-off, S	Tower crane -> 0.0039kWh/kg * 1.28kg = 0.004992 kWh * 2 (construction/deconstruction) = 0.009984kWh
			Own calculation	Transport, lorry >16t, fleet average/RER S	1,28kg/1000 = 0,00128 t --> 0,0028t*100km (distance from the site to the waste treatment plan) = 0,128 tkm
			Own calculation	Waste textile, soiled {RoW}| treatment of, municipal incineration | Cut-off, S	
	Root Barrier	Polyethylene (LPDE)	Chenani et al., 2015	Polyethylene, low density, granulate {GLO}| market for | Cut-off, S	
			Chenani et al., 2015	Extrusion, plastic film {RER}| extrusion, plastic film | Cut-off, S	
			Own calculation	Transport, lorry >32t, EURO5/RER S	0.8kg/1000 = 0,0008 t --> 0,0008t*500km (distance from the supplier in Sweden to the instalation site) = 0,4 tkm
			Own calculation	Electricity, low voltage {Europe without Switzerland}| market group for | Cut-off, S	Tower crane -> 0.0039kWh/kg * 0.8kg = 0.00312 kWh* 2 (construction/deconstruction) = 0.00624 kWh
			Own calculation	Transport, lorry >16t, fleet average/RER S	1,6kg/1000 = 0,0016 t --> 0,0016t*100km (distance from the site to the waste treatment plan) = 0,16 tkm
			Chenani et al., 2015	Waste polyethylene {RoW}| treatment of waste polypropylene, municipal incineration | Cut-off, S	
			Chenani et al., 2015	Recycling mixed plastics/RER S	

Lifecycle inventory (continuation)

1 m² of extensive green roof							Installation	Operation	Deconstruction	Total years 40	
Process	**Layer**	**Element**	**Material / processes**	**Lifetime (year)**	**Aux value**	**Aux Unit**	**I, Per m2·y**	**O, Per m2·y**	**D, Per m2·y**	**Per lifetime**	**Unit**
SUBSTRATE	**Substrate**	Pumice	**Pumice**	40	450	kg/m^3	31,96			31,96	kg
			Transport		500	km				15,98	tkm
			Machinery for construction / deconstruction (1/2): tower crane		0,0039	kWh/kg	0,1246		0,1246	0,2492	kWh
			End of life: transport lorry to recycling		100	km				3,16	tkm
			Landfill		100	%				31,96	kg
		Gravel	**Crushed gravel**	40	1400	kg/m^3	21,42			21,42	kg
			Transport		500	km				10,71	tkm
			Machinery for construction / deconstruction (1/2): tower crane		0,0039	kWh/kg	0,083538		0,083538	0,167076	kWh
			End of life: transport lorry to recycling		100	km				2,142	tkm
			Landfill		100	%				21,42	kg
		Compost	**Compost**	40	500	kg/m^3	4,05			4,05	kg
			Transport		500	km				1,125	km
			Machinery for construction / deconstruction (1/2): tower crane		0,0039	kWh/kg	0,015795		0,015795	0,03159	kWh
			End of life: transport lorry waste treatment plan		100	km				0,225	tkm
			Landfill		100	%				2,25	kg
		Slow acting Fertilizer $15g/m^2$	**Fertilizer**	1	7,5	kg/m^2		0,15		7,5	kg
			Transport		500	km				3,75	tkm

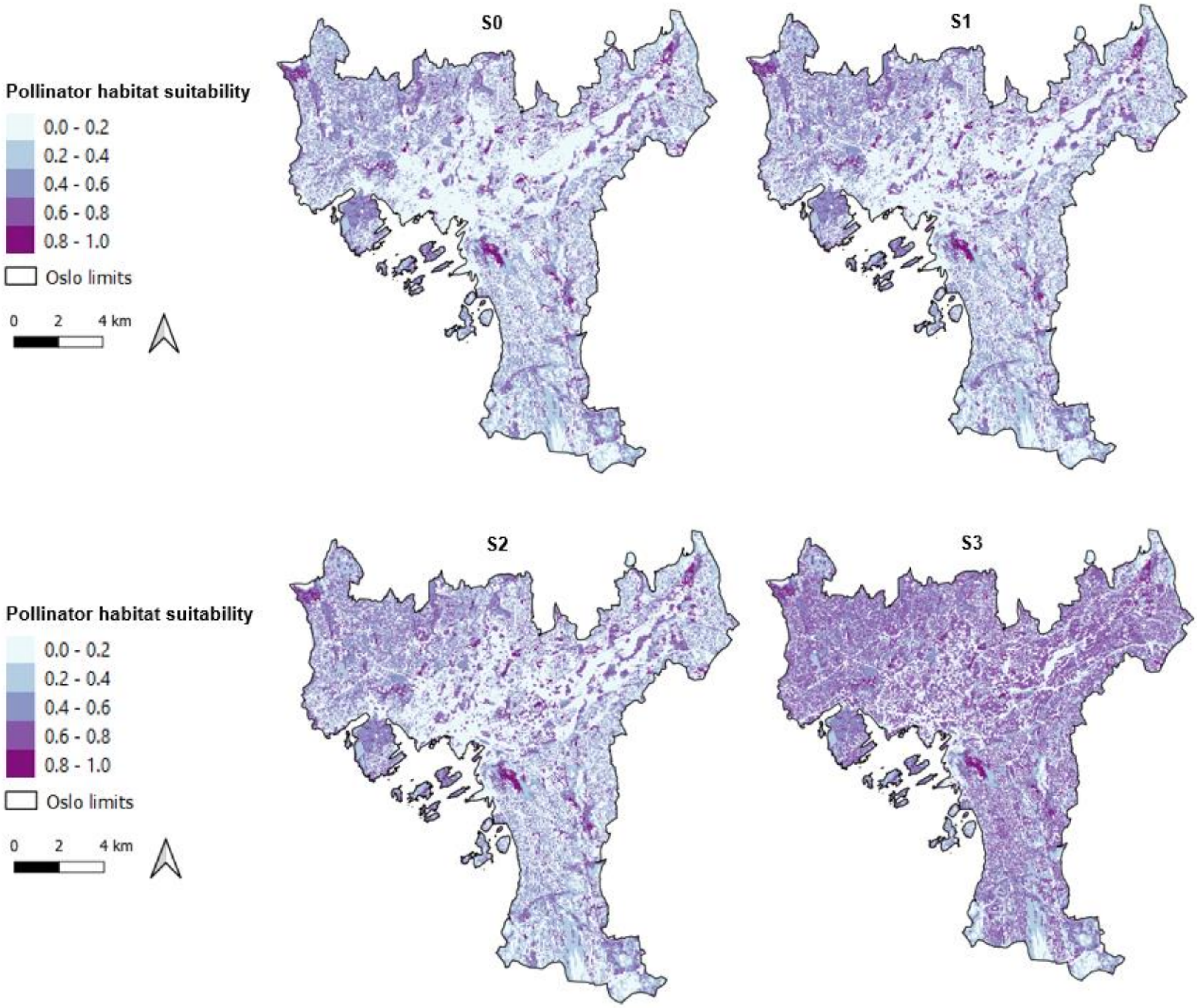
Lifecycle inventory (continuation)

				Simapro input	Comment
Process	**Layer**	**Element**	**Source**	**Ecoinvent**	**Note & assumption**
SUBSTRATE	**Substrate**	Pumice	Hanslin et al., 2018	Pumice {GLO}| market for | Cut-off, U	
			Own calculation	Transport, lorry >32t, EURO5/RER S	31.96kg/1000 = 0.03196 t --> 0.00225t*500km (distance from the supplier in Sweden to the instalation site) = 1.125 tkm
			Own calculation	Electricity, low voltage {Europe without Switzerland}| market group for | Cut-off, S	Tower crane -> 0.0039kWh/kg * 2.25kg = 0.008775 kWh * 2 (construction/deconstruction) = 0.01755 kWh
			Own calculation	Transport, lorry >16t, fleet average/RER S	2.25kg/1000 = 0.00225 t --> 0.00225t*100km (distance from the site to the waste treatment plan) = 0.225tkm
			Chenani et al	Inert waste, for final disposal {RoW}| treatment of inert waste, inert material landfill | Cut-off, S	
		Gravel	Hanslin et al., 2018	Gravel, crushed, at mine/CH U	
			Own calculation	Transport, lorry >32t, EURO5/RER S	2.25kg/1000 = 0.00225 t --> 0.00225t*500km (distance from the supplier in Sweden to the instalation site) = 1.125 tkm
			Own calculation	Electricity, low voltage {Europe without Switzerland}| market group for | Cut-off, S	Tower crane -> 0.0039kWh/kg * 2.25kg = 0.008775 kWh * 2 (construction/deconstruction) = 0.01755 kWh
			Own calculation	Transport, lorry >16t, fleet average/RER S	2.25kg/1000 = 0.00225 t --> 0.00225t*100km (distance from the site to the waste treatment plan) = 0.225tkm
			Chenani et al., 2015	Inert waste, for final disposal {RoW}| treatment of inert waste, inert material landfill | Cut-off, S	
		Compost	Hanslin et al., 2018	Compost, at plant/CH U	
			Own calculation	Transport, lorry >32t, EURO5/RER S	2.25kg/1000 = 0.00225 t --> 0.00225t*500km (distance from the supplier in Sweden to the instalation site) = 1.125 tkm
			Own calculation	Electricity, low voltage {Europe without Switzerland}| market group for | Cut-off, S	Tower crane -> 0.0039kWh/kg * 2.25kg = 0.008775 kWh * 2 (construction/deconstruction) = 0.01755 kWh
			Own calculation	Transport, lorry >16t, fleet average/RER S	2.25kg/1000 = 0.00225 t --> 0.00225t*100km (distance from the site to the waste treatment plan) = 0.225tkm
			Chenani et al., 2015	Municipal solid waste (waste scenario) {Europe without Switzerland}| Treatment of municipal solid waste, landfill | Cut-off, S	
		Slow acting Fertilizer 15g/m^2	Sintef 2012	NPK (15-15-15) fertiliser {RER}| market for NPK (15-15-15) fertiliser | Cut-off, S	15 g/m2 per year over 40 year = 7.5kg
			Own calculation	Transport, lorry >32t, EURO5/RER S	7.5kg/1000 = 0.0075 t --> 0.0075t*500km (distance from the supplier in Sweden to the instalation site) = 3.75 tkm

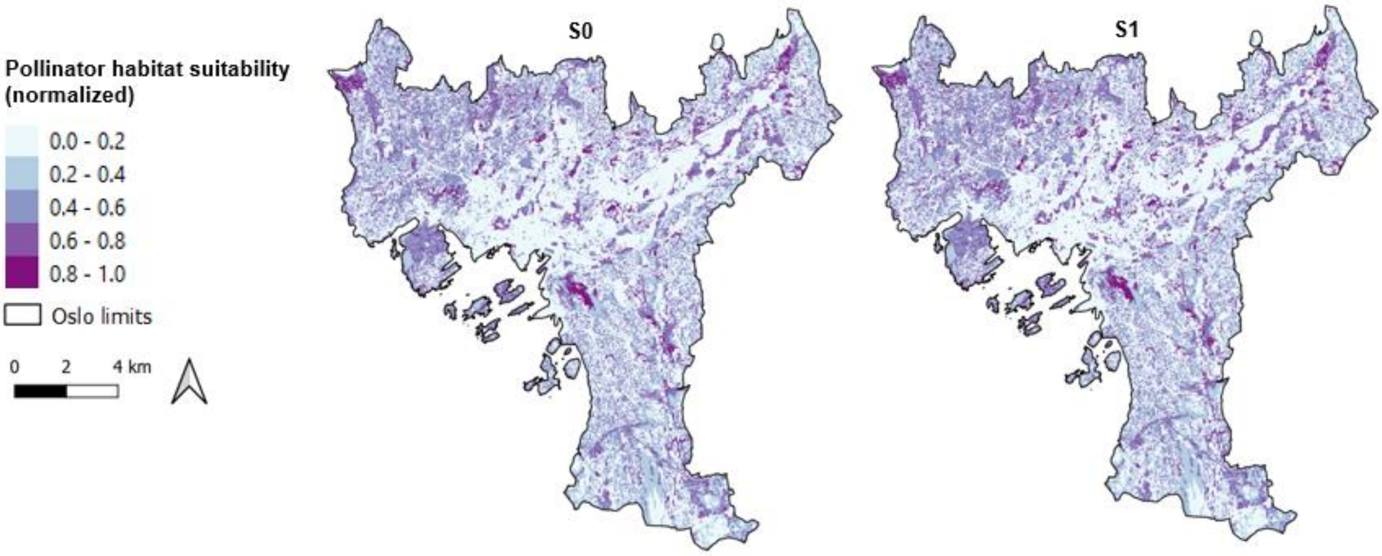
2-C. Supplementary maps

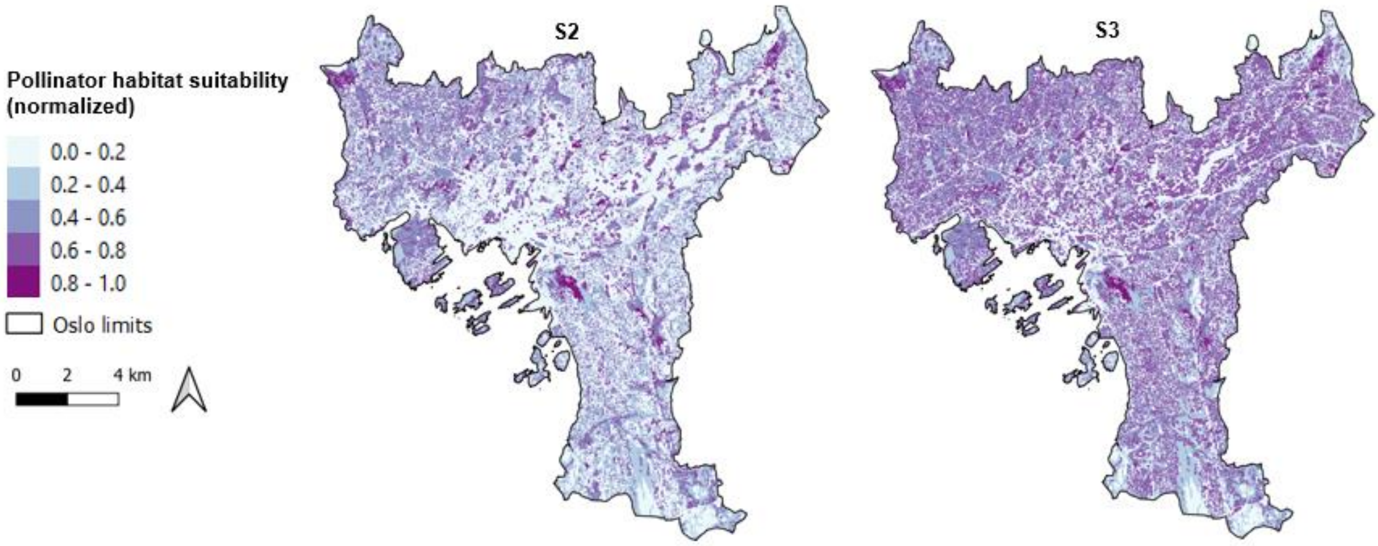
1. Vulnerability to lack of habitats for pollinators

1.1. Pollinator habitat suitability (exposure)

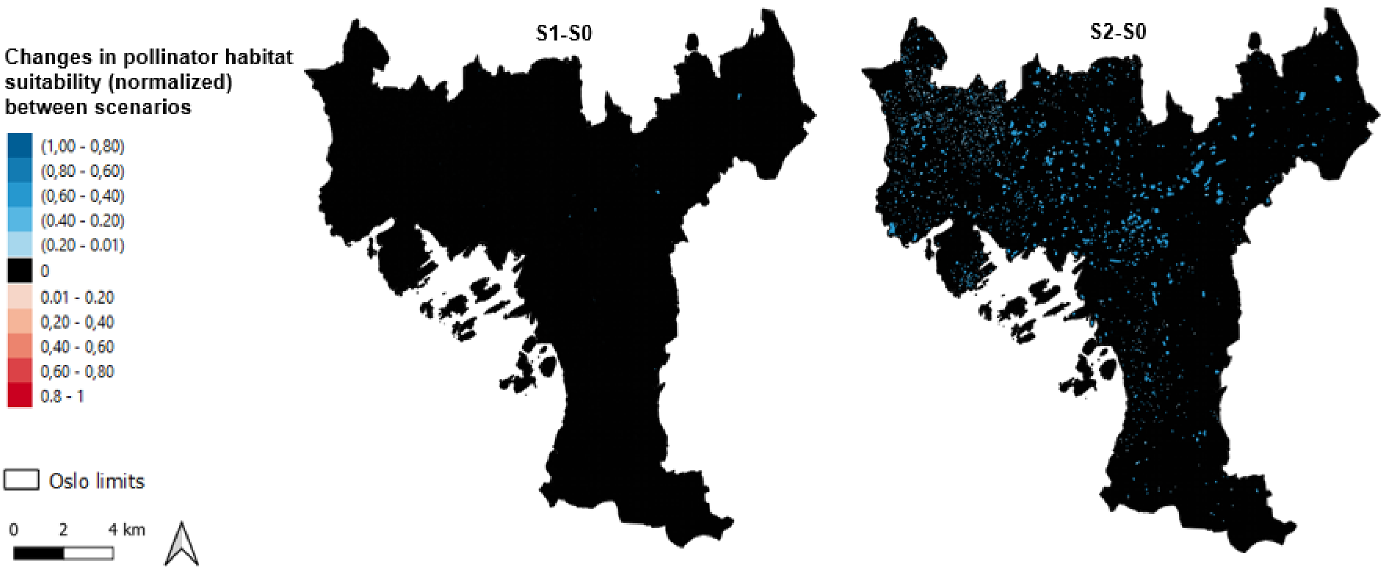


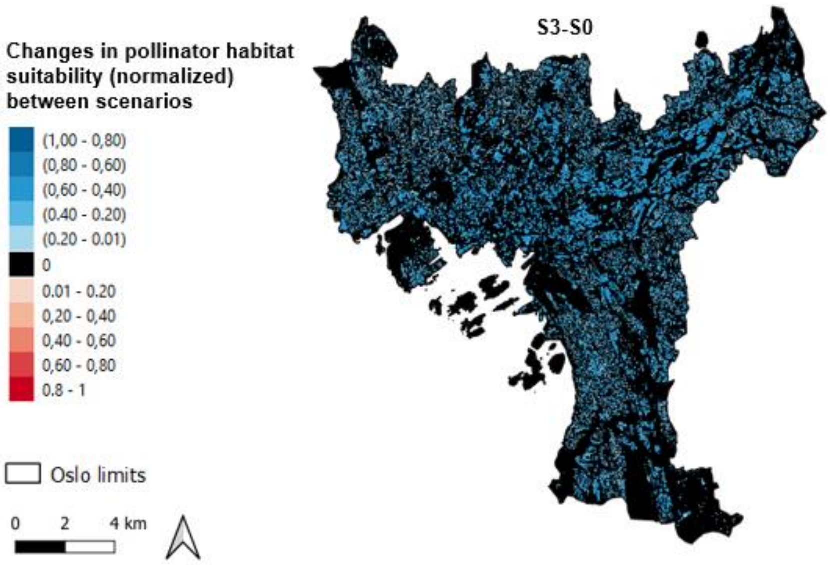
1.2. Pollinator habitat suitability (exposure)



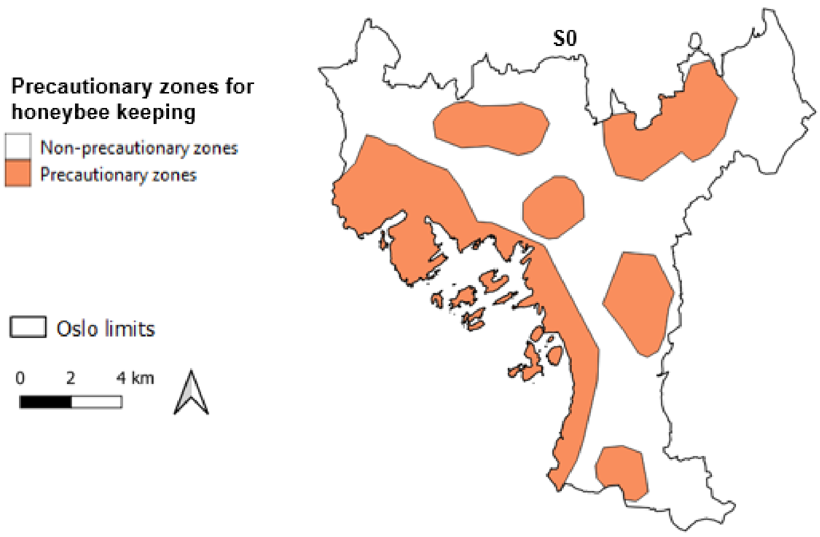


1.3. Changes in Pollinator habitat suitability (exposure)

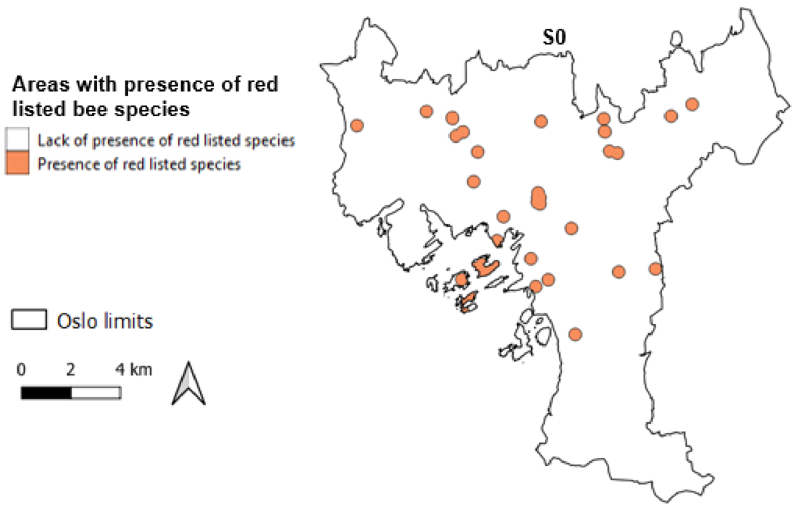




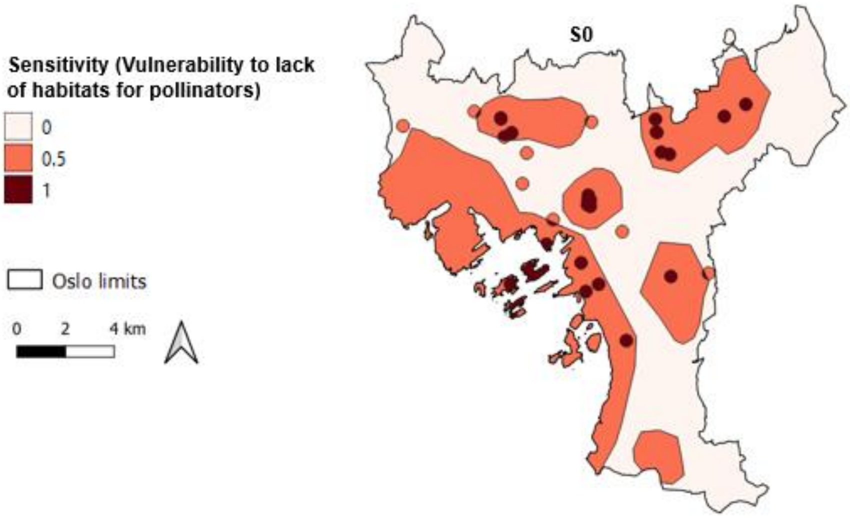
1.4. Precautionary zones for honeybee keeping



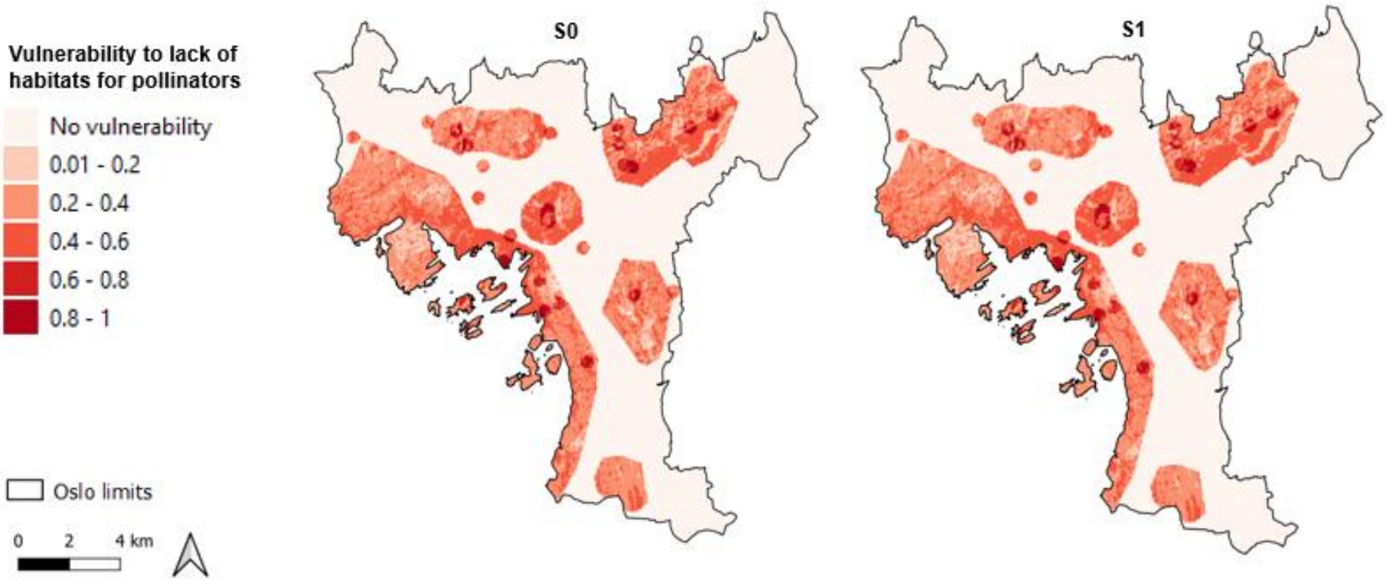
1.5. Areas with presence of red listed bee species

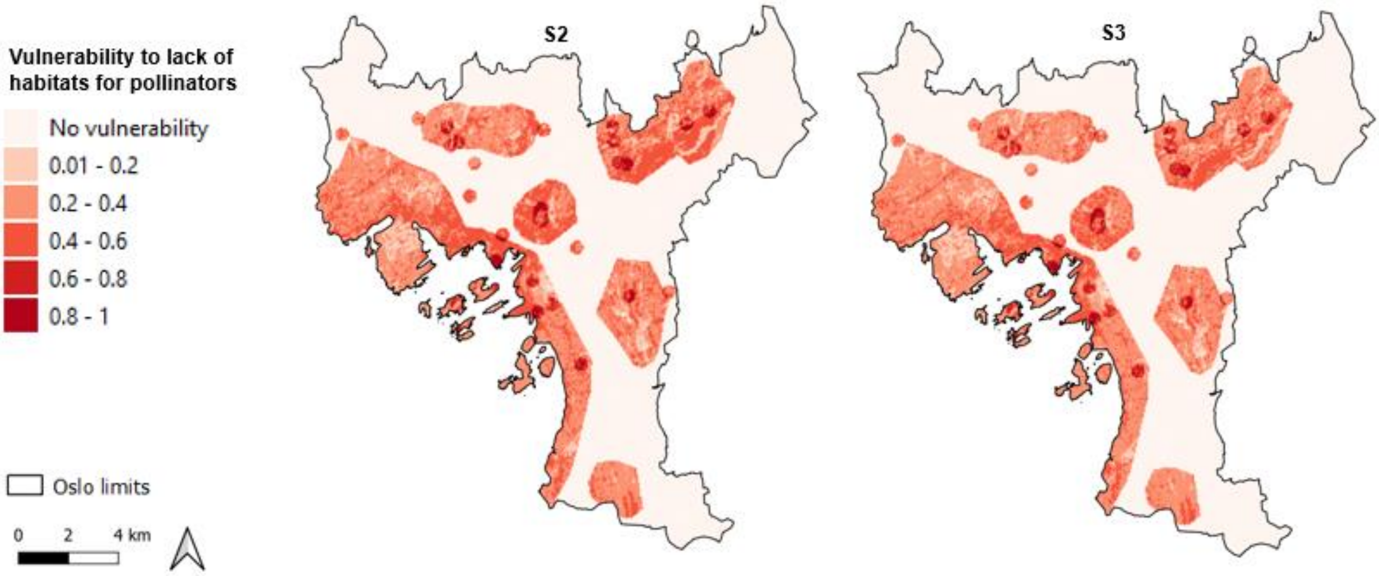


1.6. Sensitivity (Vulnerability to lack of habitats for pollinators)



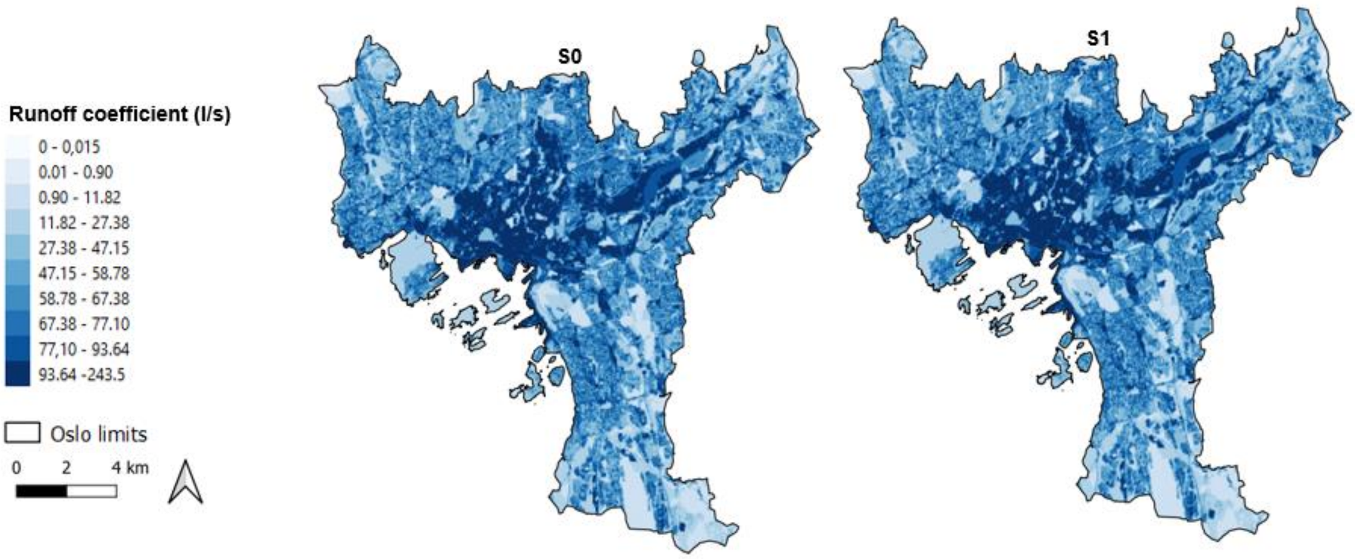
1.7. Vulnerability to lack of habitat for pollinators

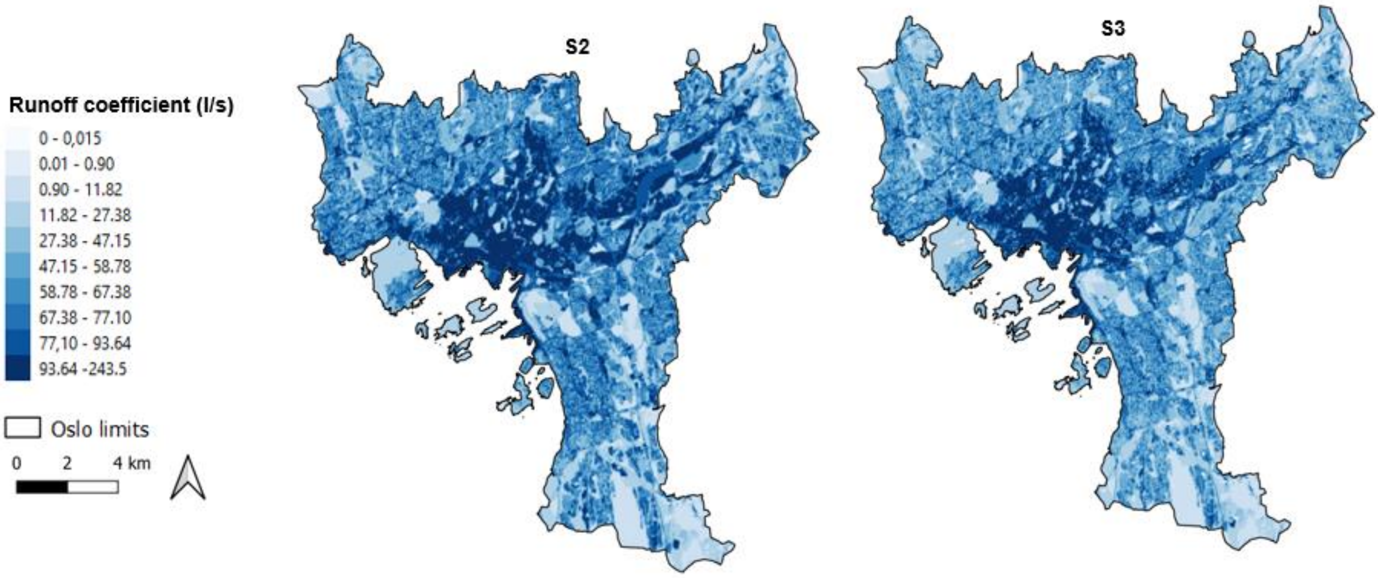




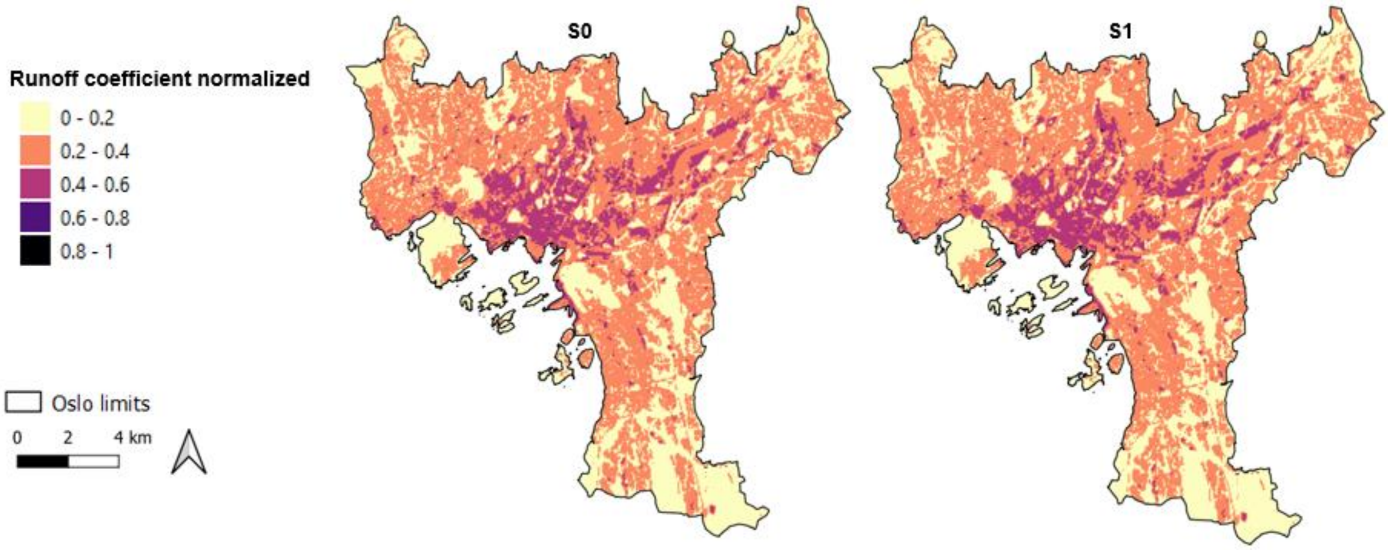
2. Vulnerability to heavy rainfall events

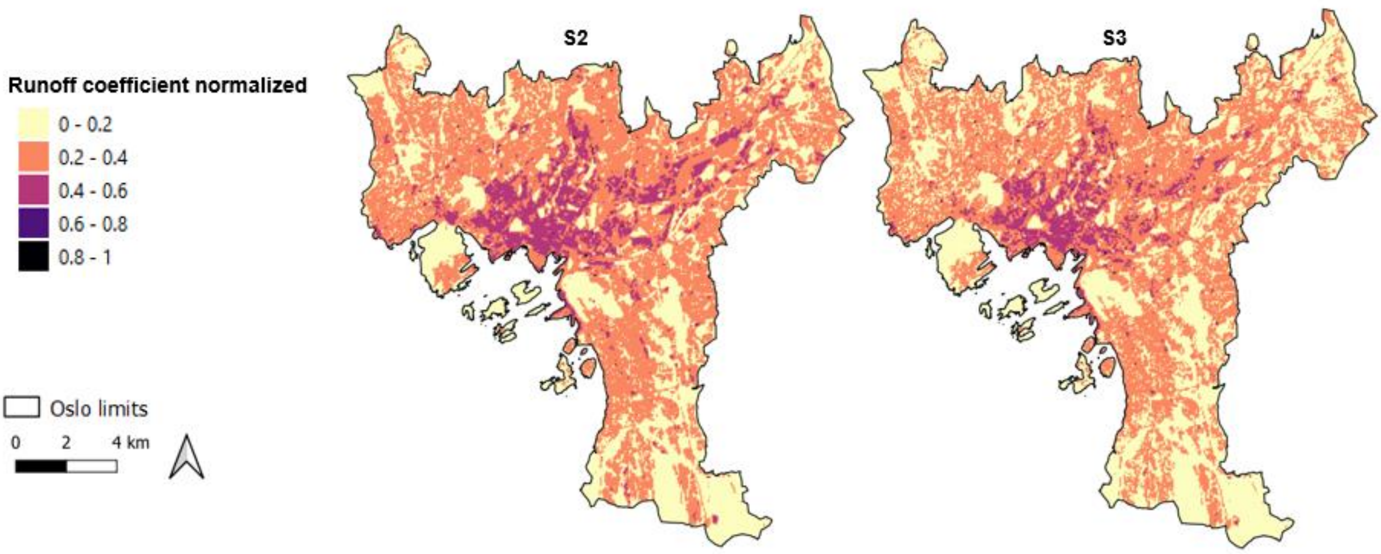
2.1. Runoff coefficients (exposure)



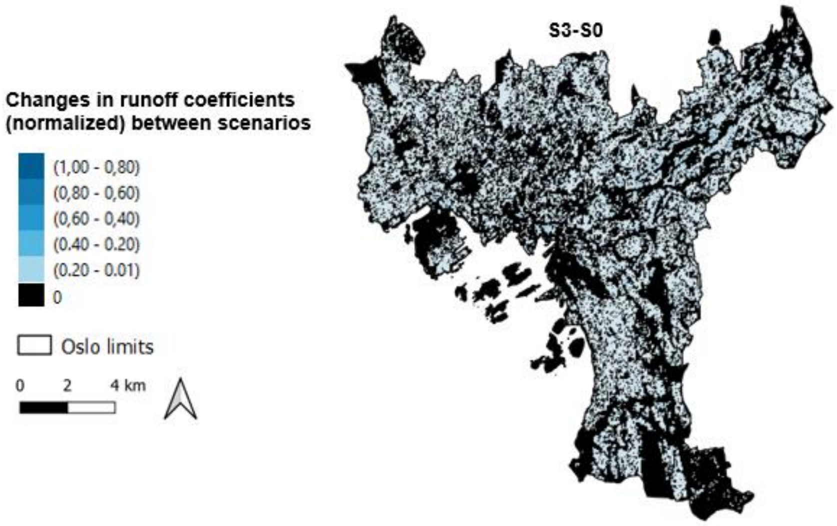


2.2. Runoff coefficients normalized (exposure)

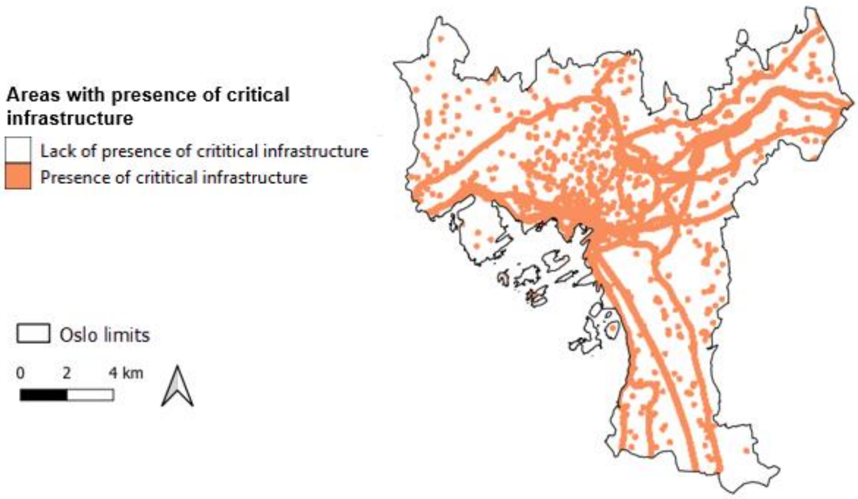




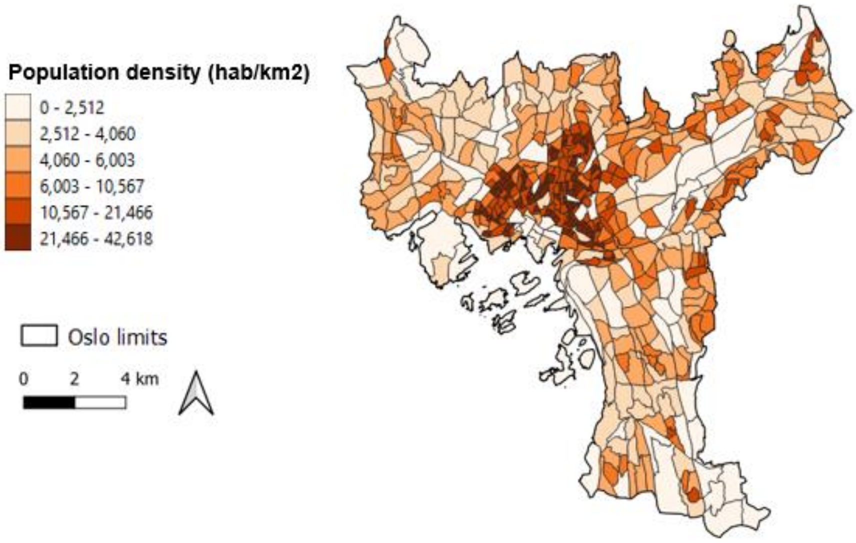
2.3. Changes in runoff coefficients normalized (exposure)



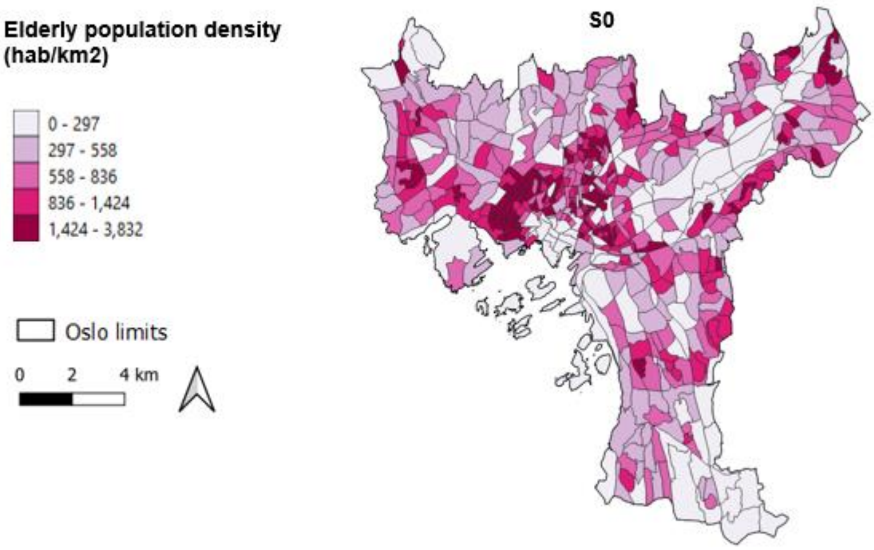
2.4. Areas with presence of critical infrastructure (sensitivity)



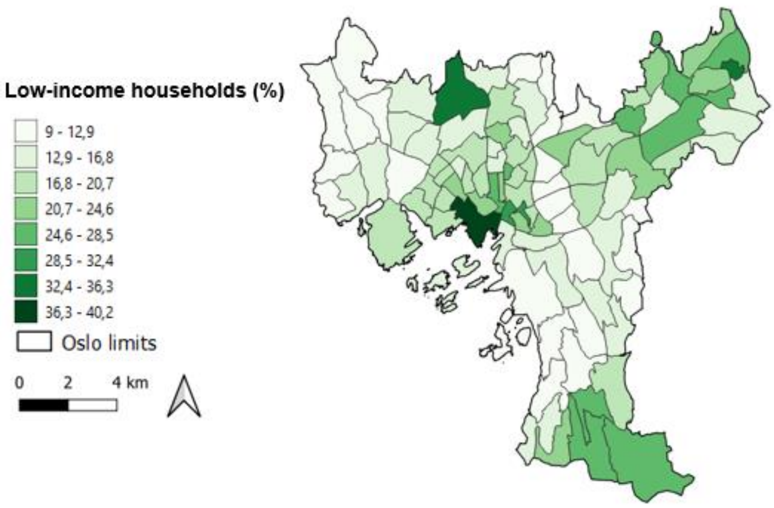
2.5. Population density (sensitivity)



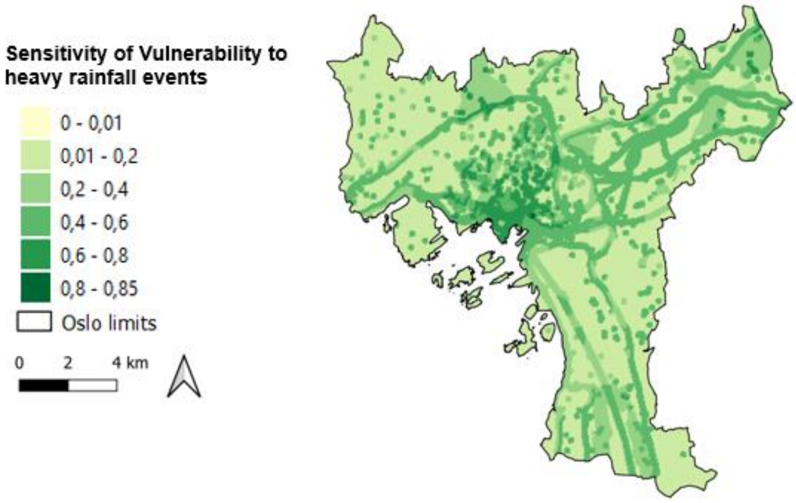
2.6. Elderly population density (sensitivity)



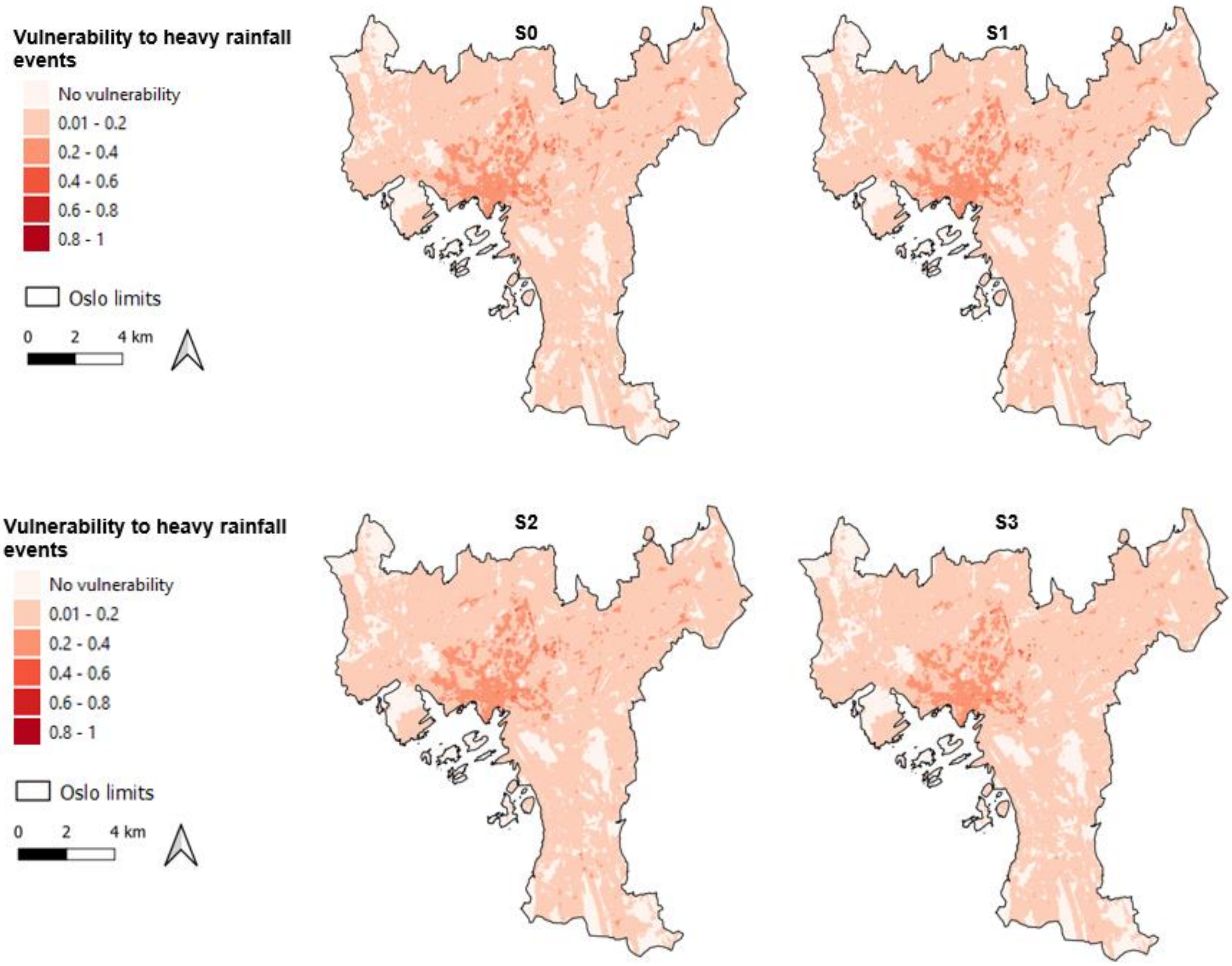
2.7. Low-income households (sensitivity)



2.8. Sensitivity (Vulnerability to heavy rainfall events)

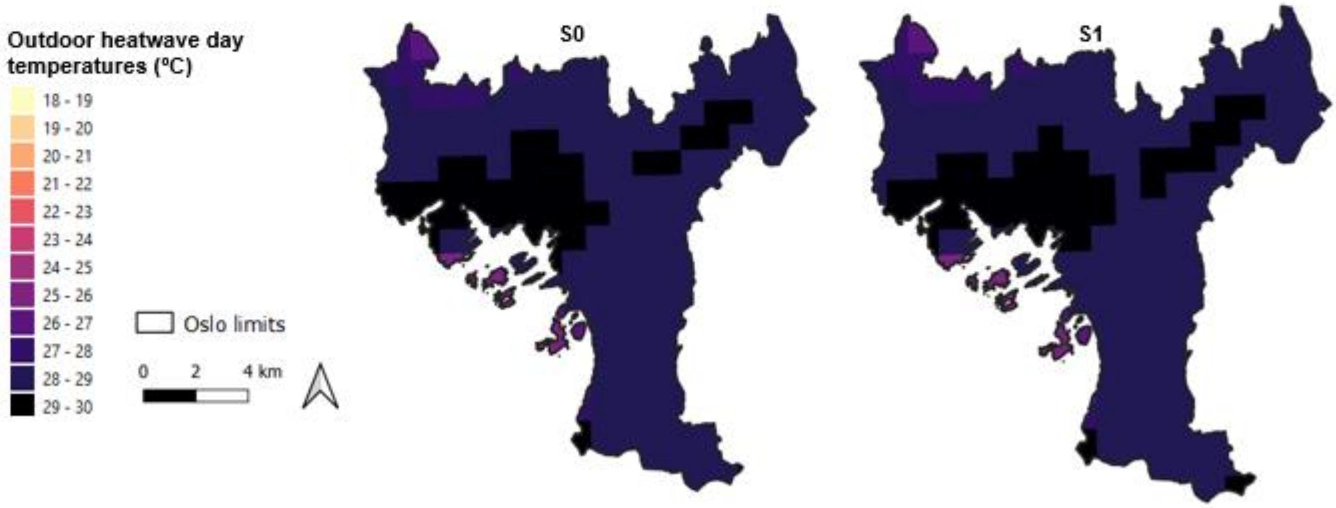


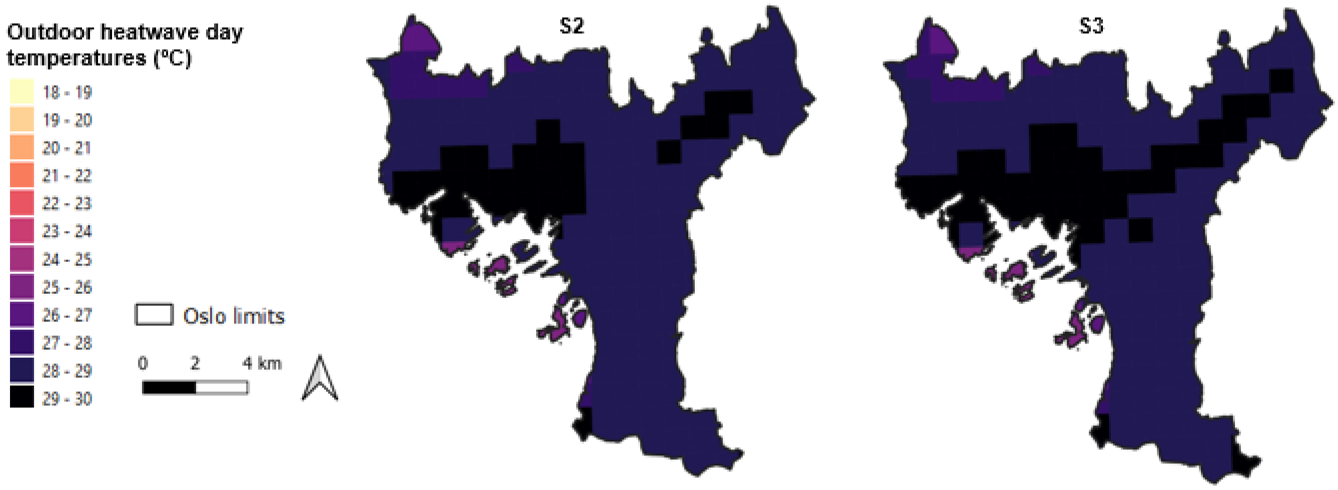
2.9. Vulnerability to heavy rainfall events



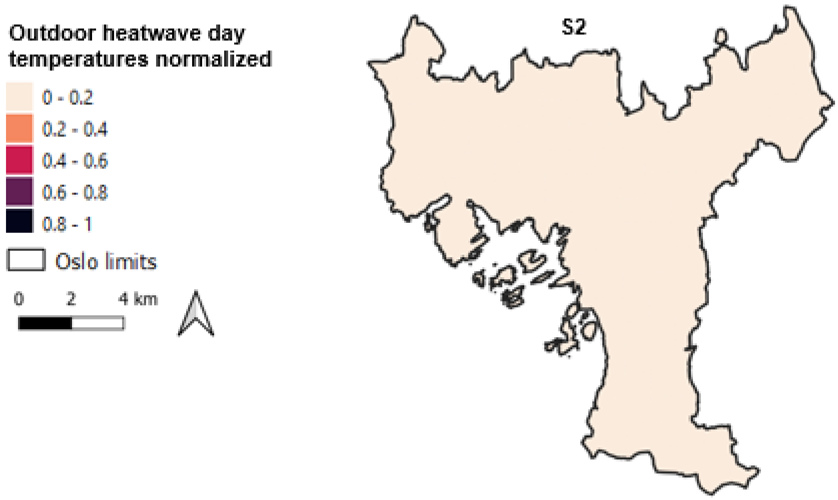
3. Vulnerability to heat

3.1. Outdoor heatwave day temperatures



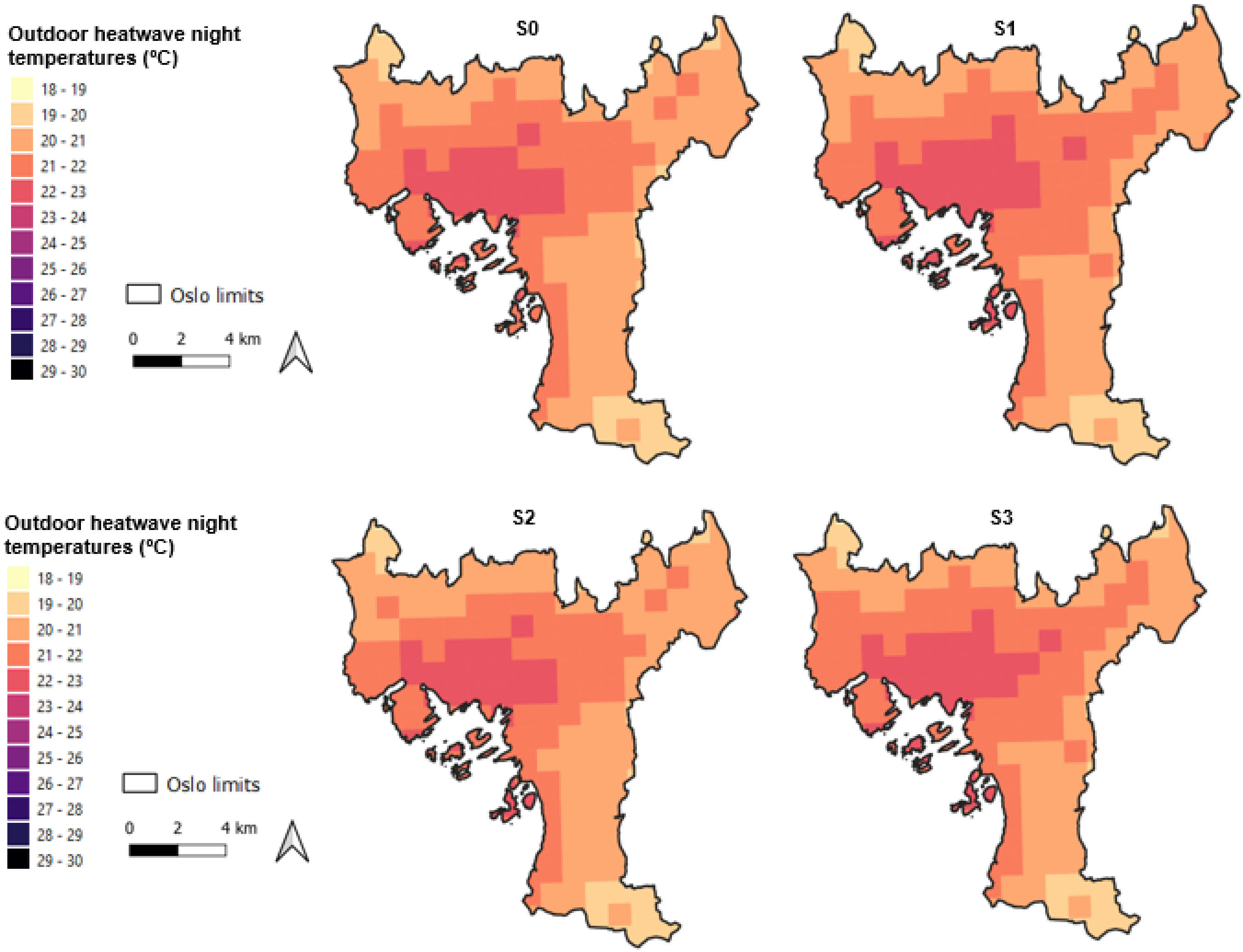


3.2. Outdoor heatwave day temperatures (normalized)

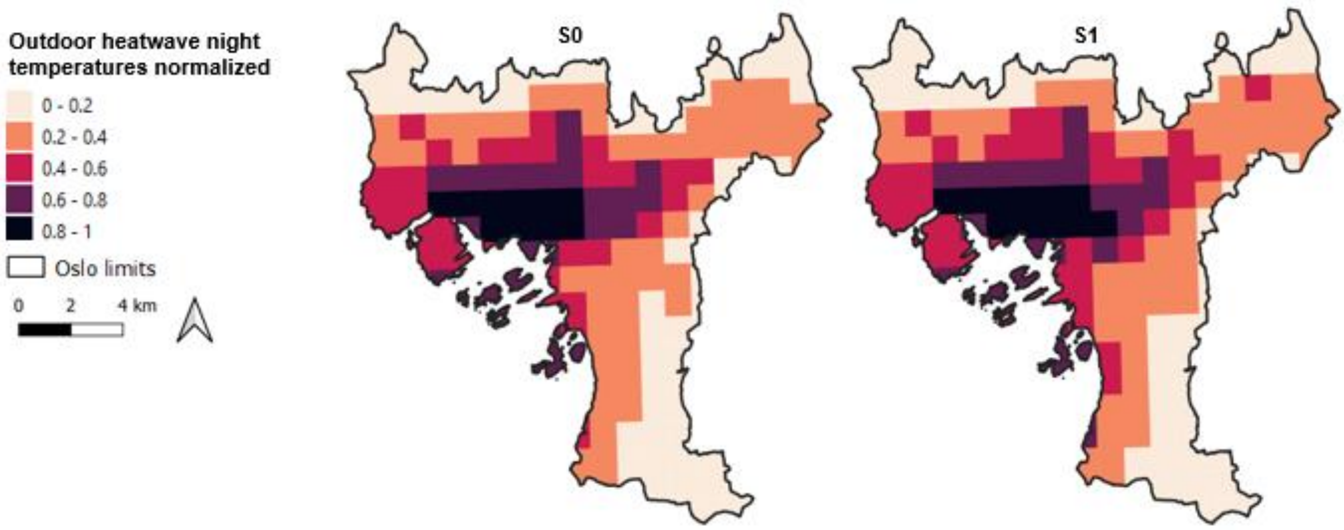


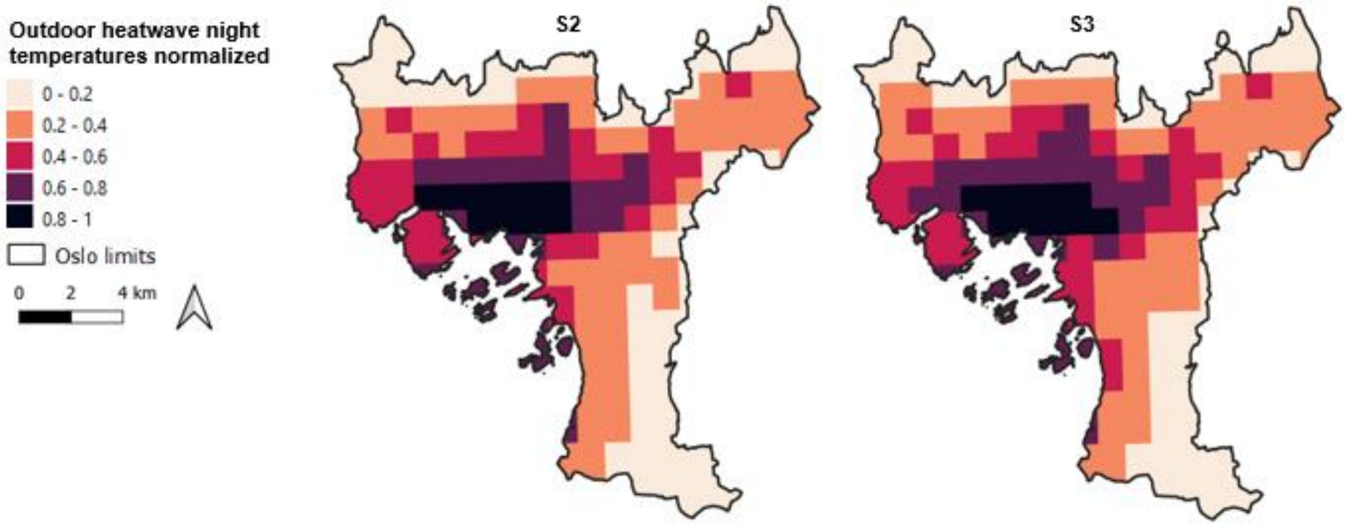
3.3. Changes in Outdoor heatwave day temperatures (normalized)

3.4. Outdoor heatwave night temperatures



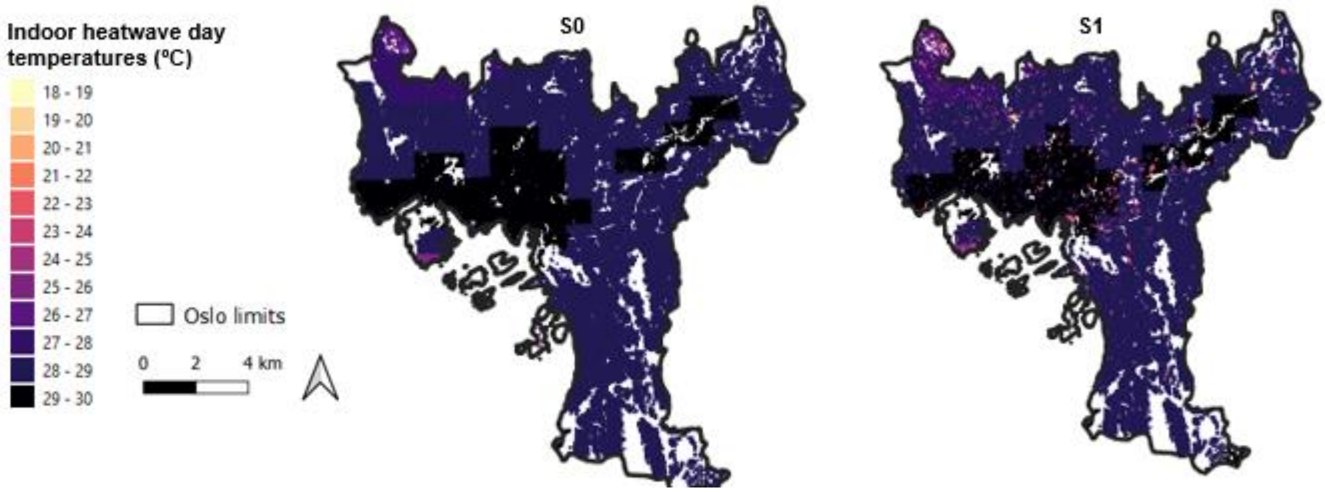
3.5. Outdoor heatwave night temperatures (normalized)

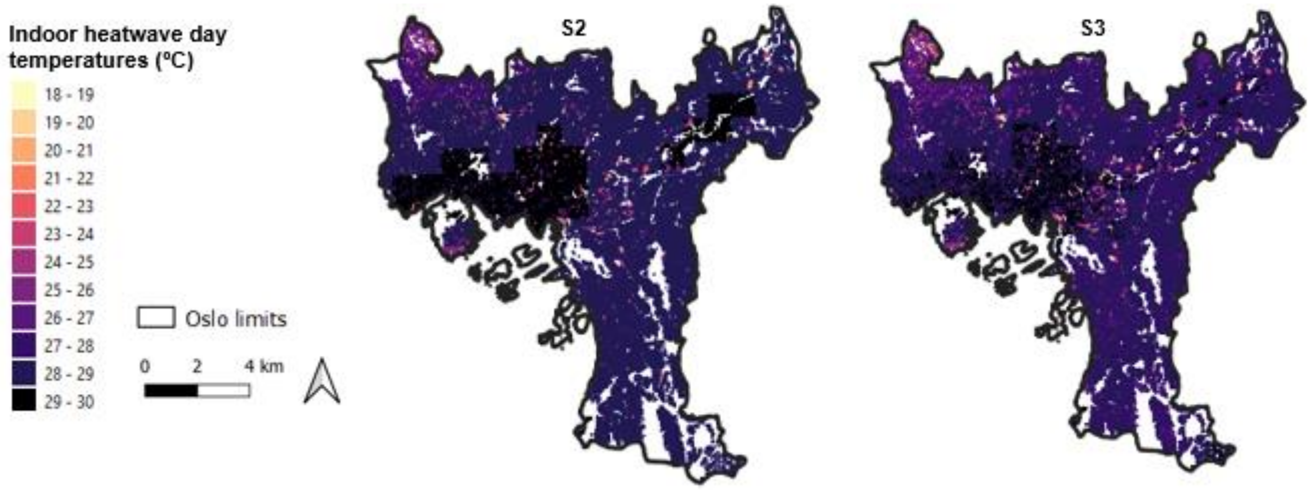




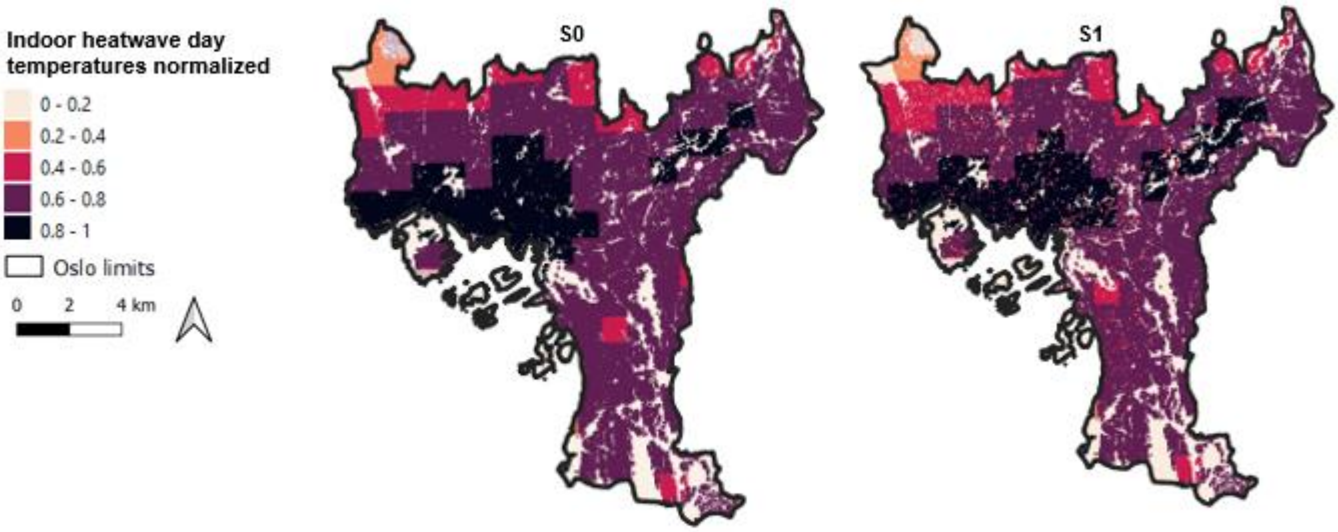
3.6. Changes in Outdoor heatwave night temperatures (normalized)

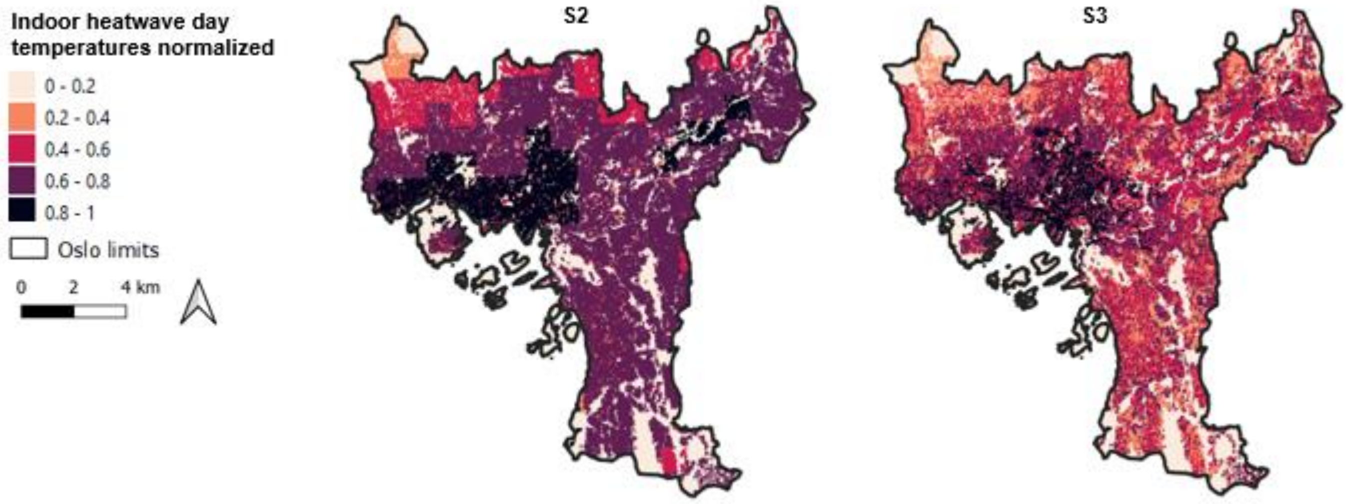
3.7. Indoor heatwave day temperatures



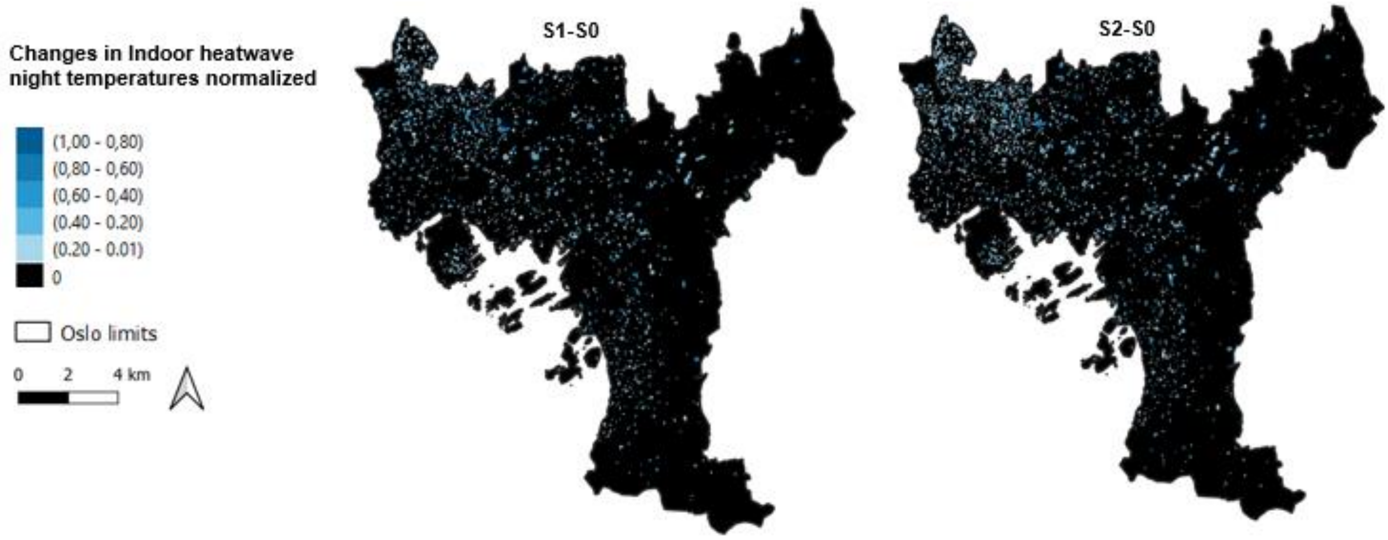


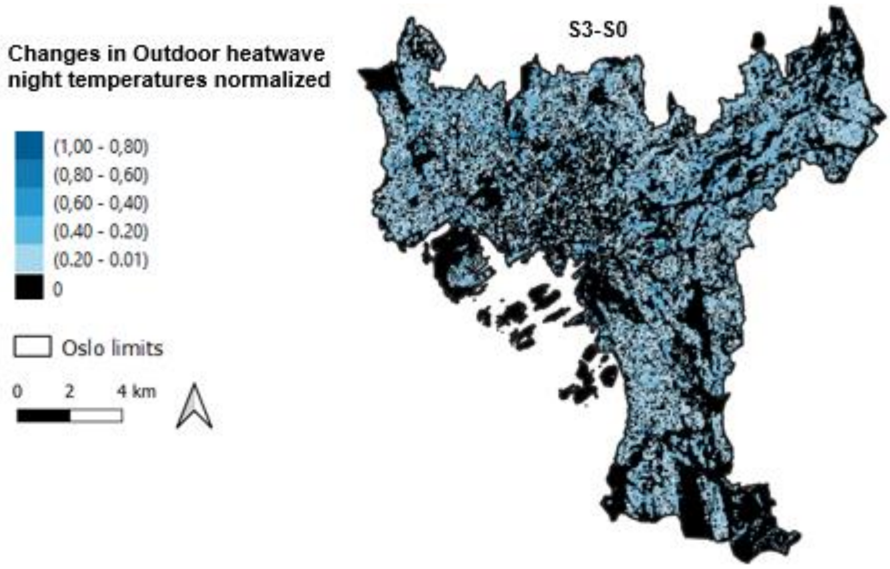
3.8. Indoor heatwave day temperatures (normalized)



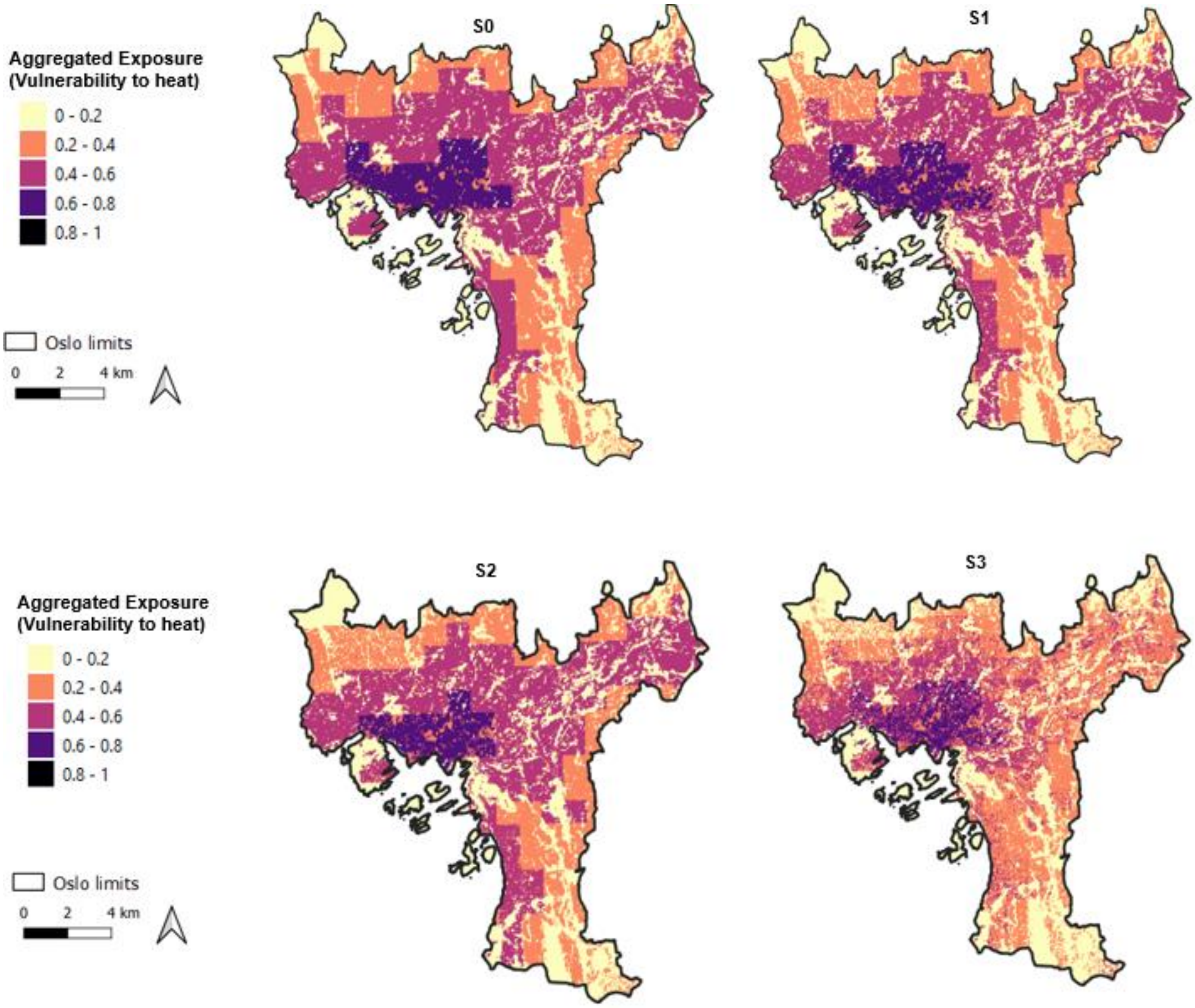


3.9. Changes in Indoor heatwave day temperatures (normalized)

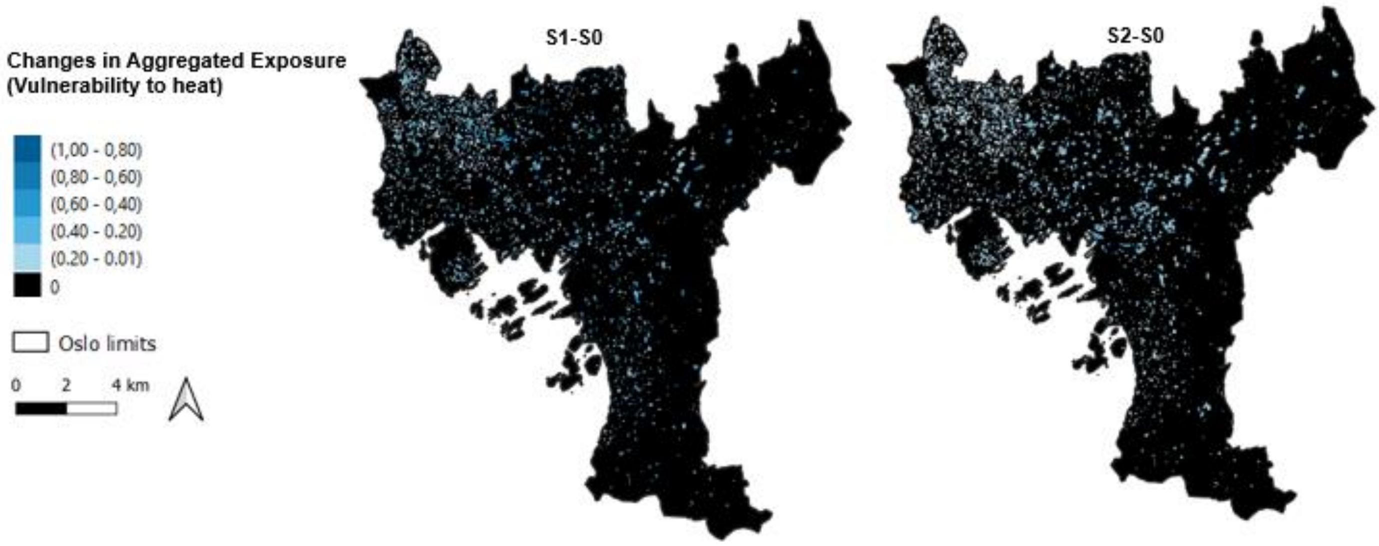


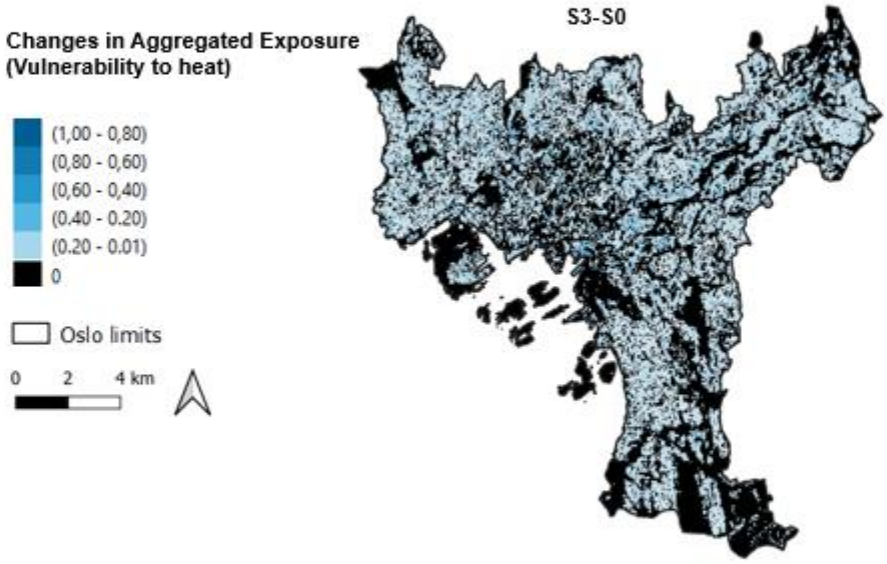


3.10. Aggregated Exposure (Vulnerability to heat)

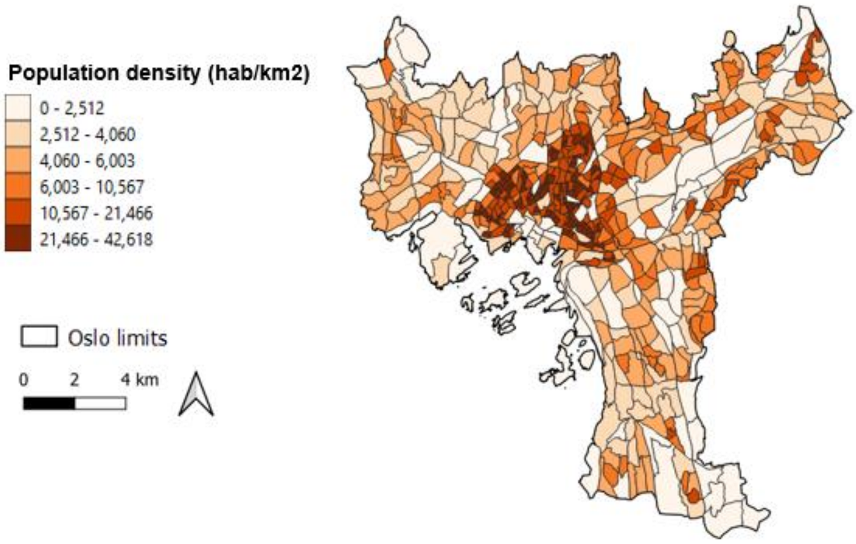


3.11. Changes in Aggregated Exposure (Vulnerability to heat)

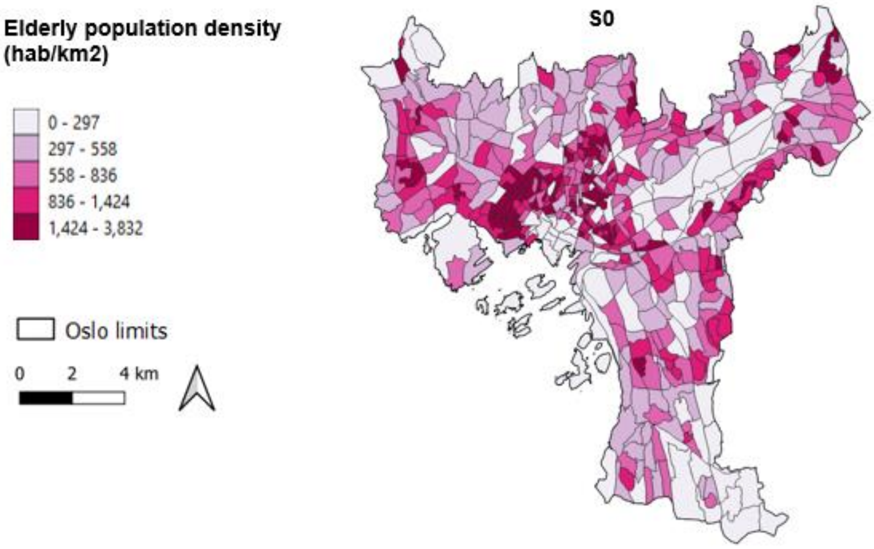




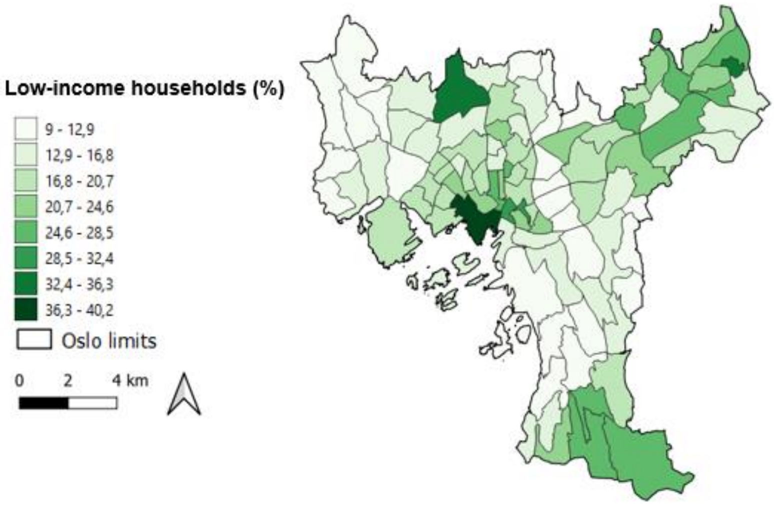
3.12. Population density



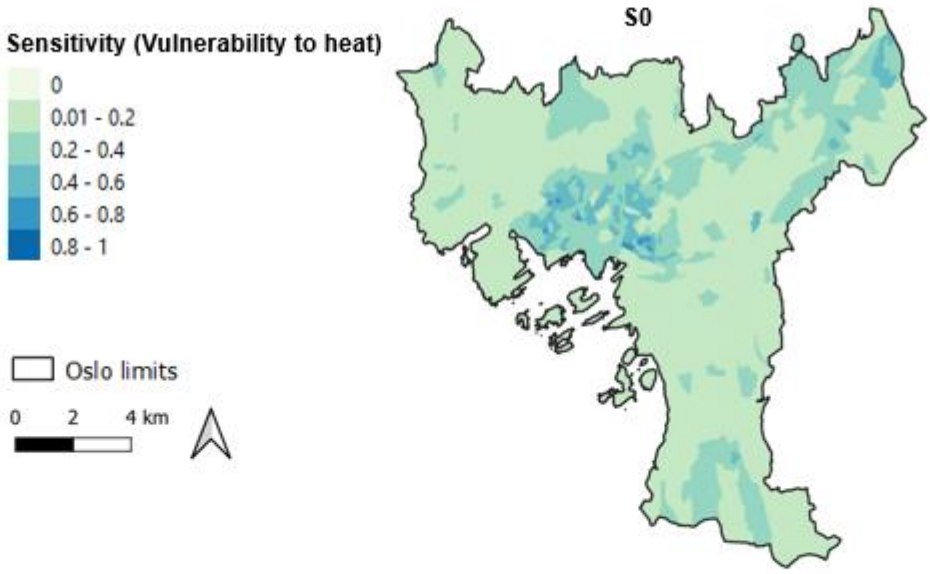
3.13. Elderly population density (75yo<)



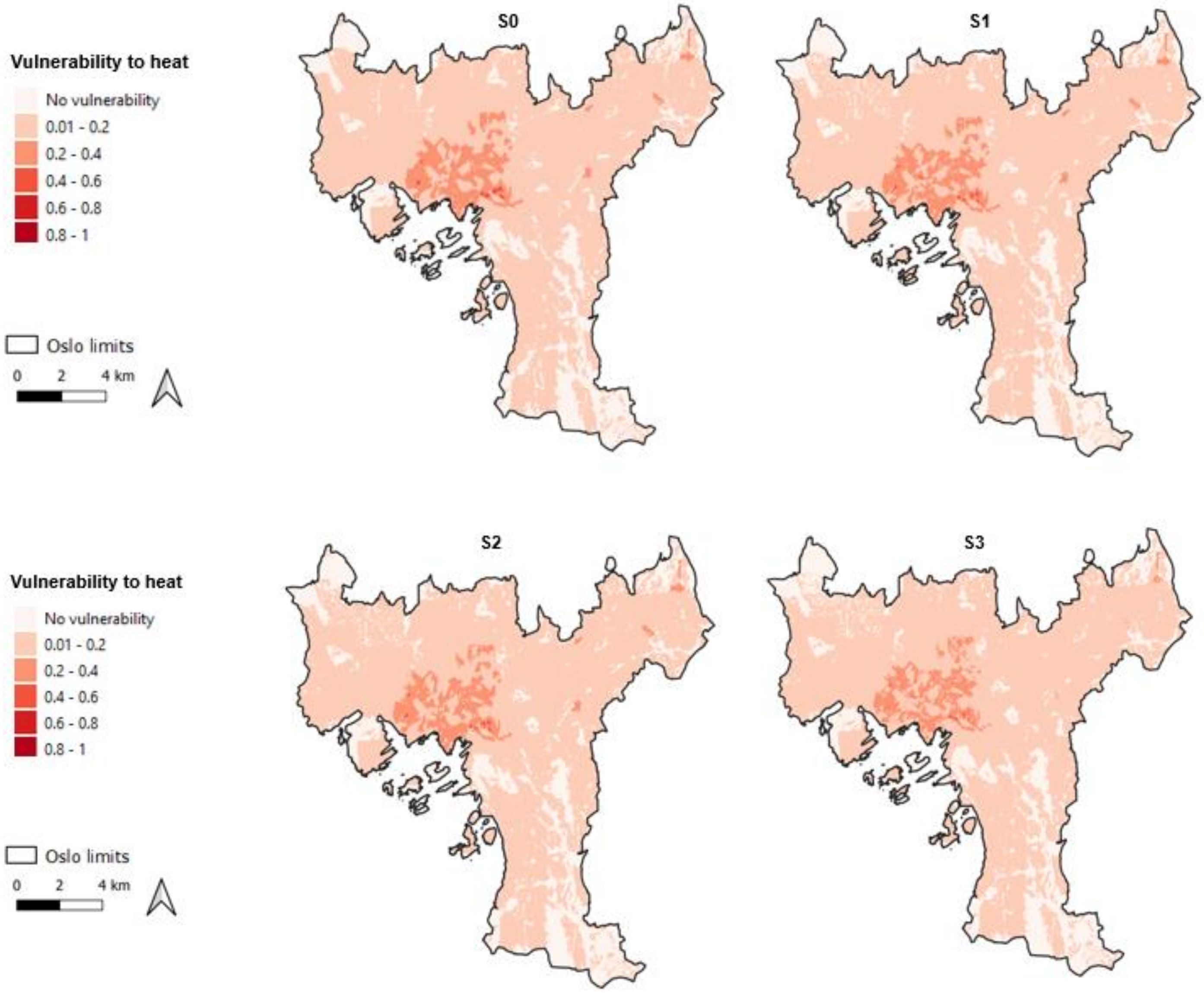
3.14. Low-income households



3.15. Sensitivity (Vulnerability to heat)

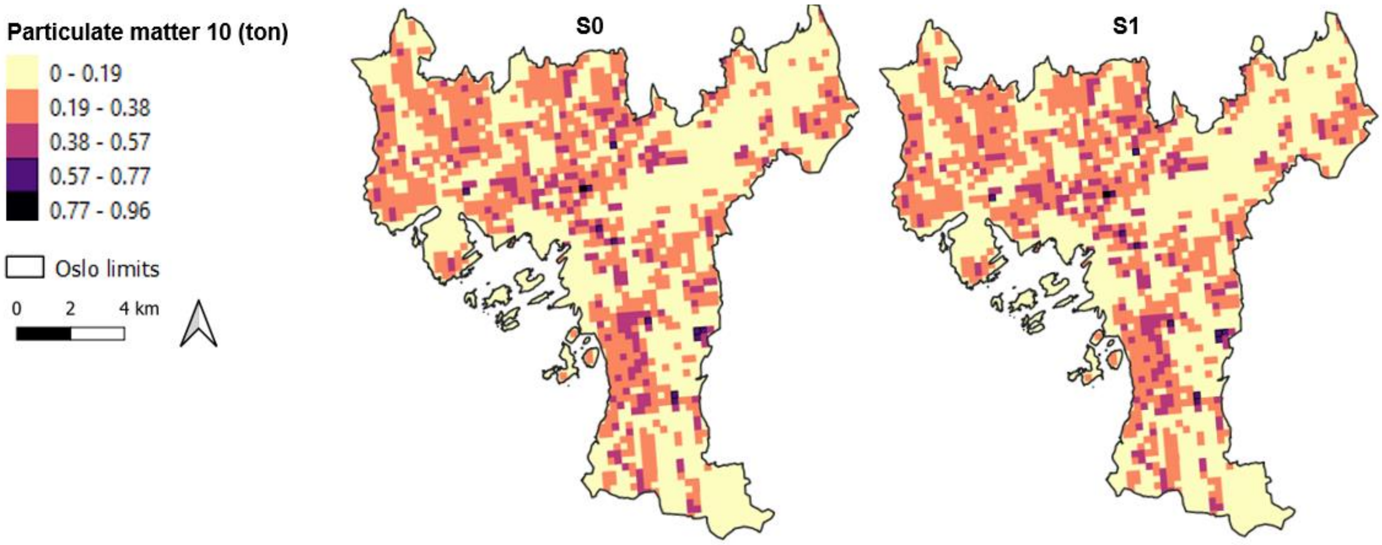


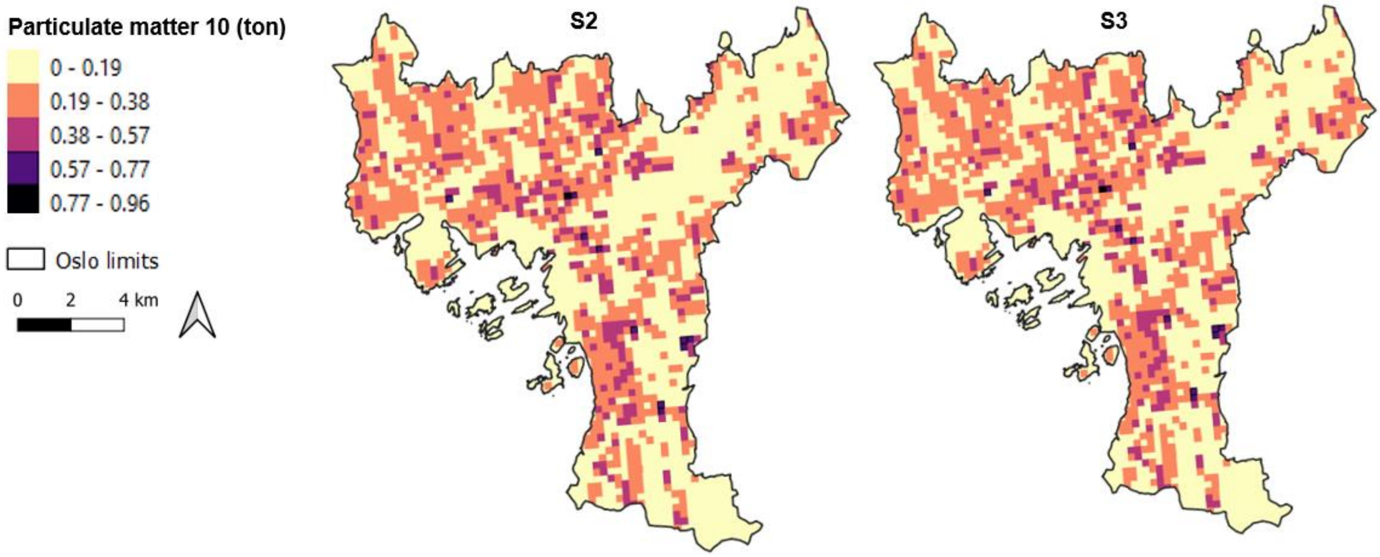
3.16. Vulnerability to heat



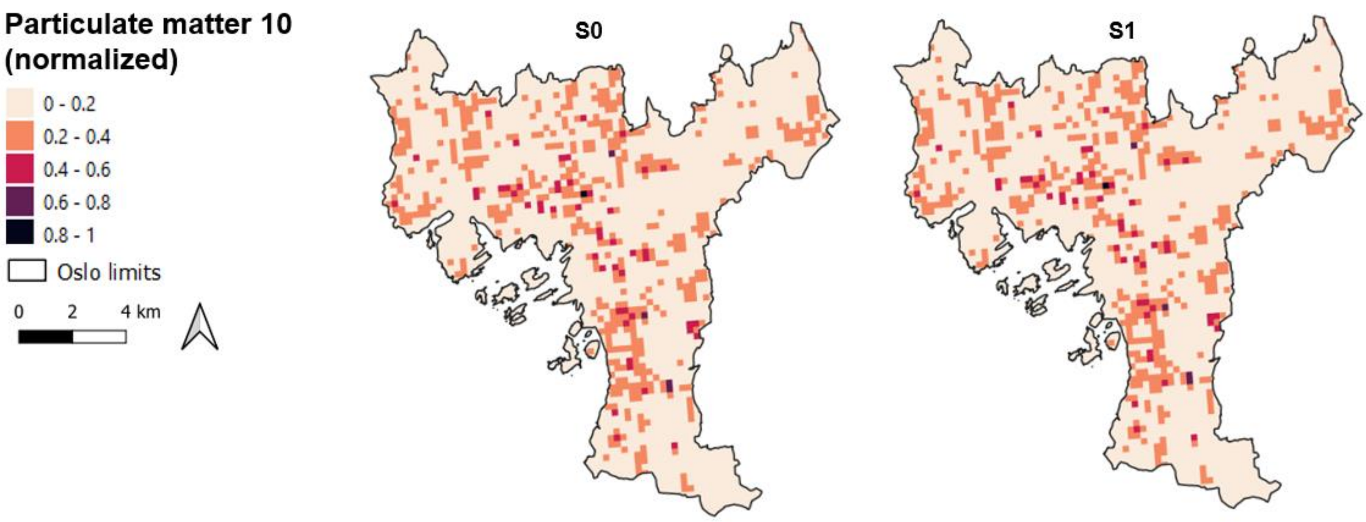
4. Vulnerability to air pollution

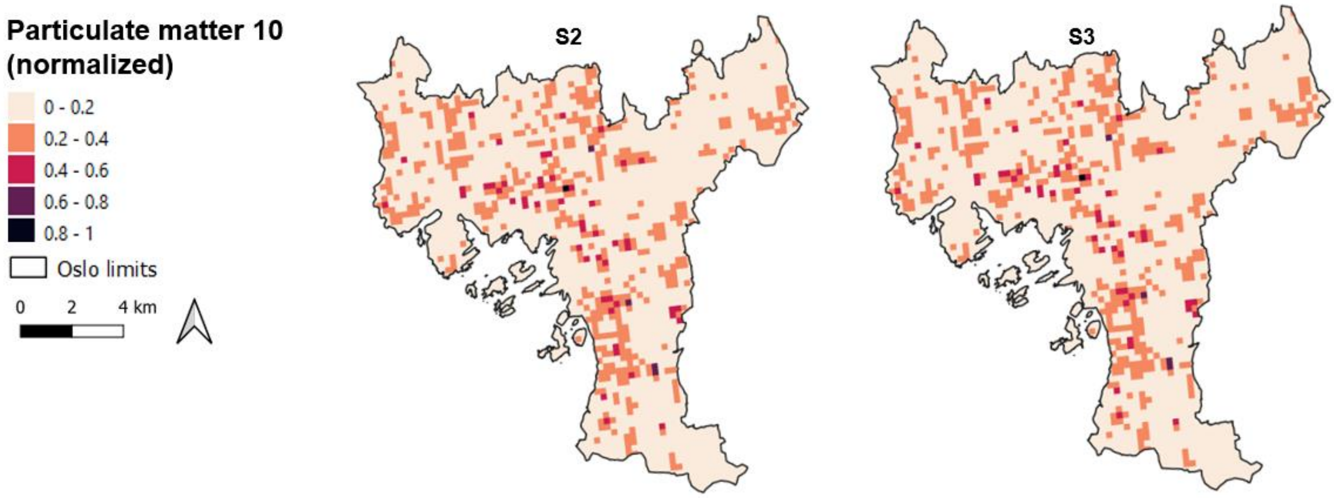
4.1. Particulate matter 10 (PM_{10})



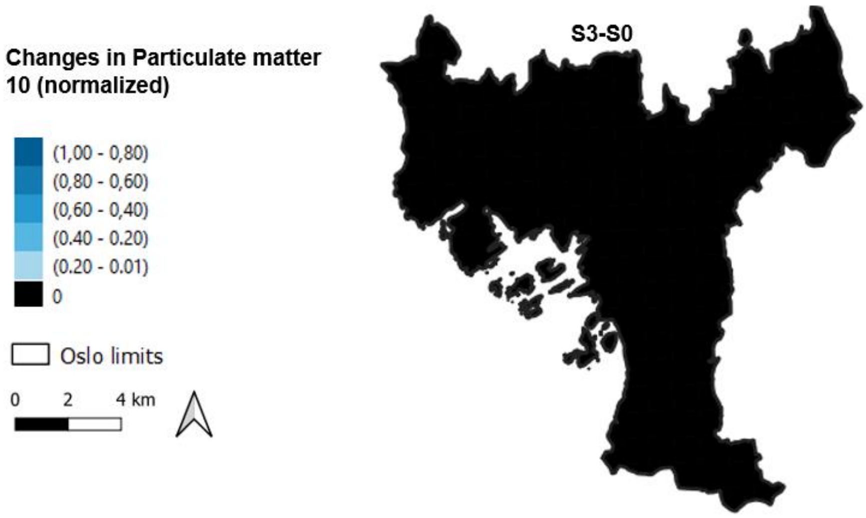


4.2. Particulate matter 10 (PM_{10}) (normalized)

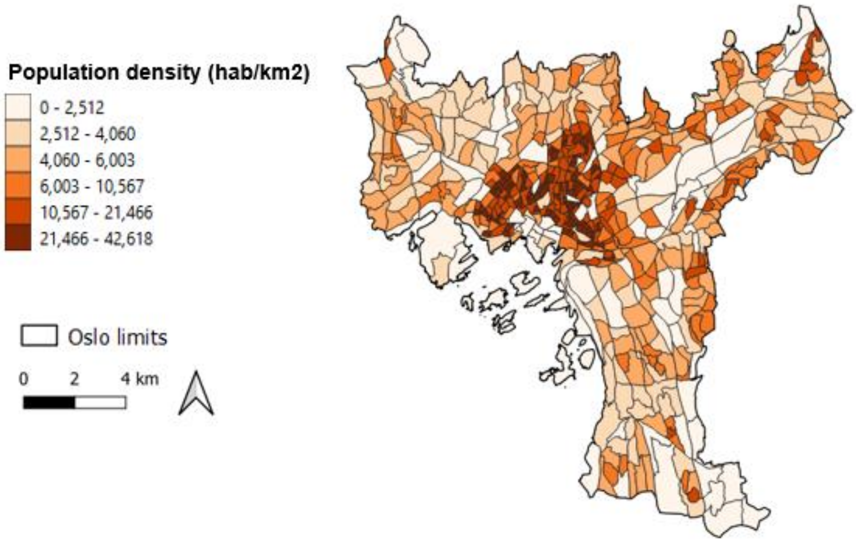




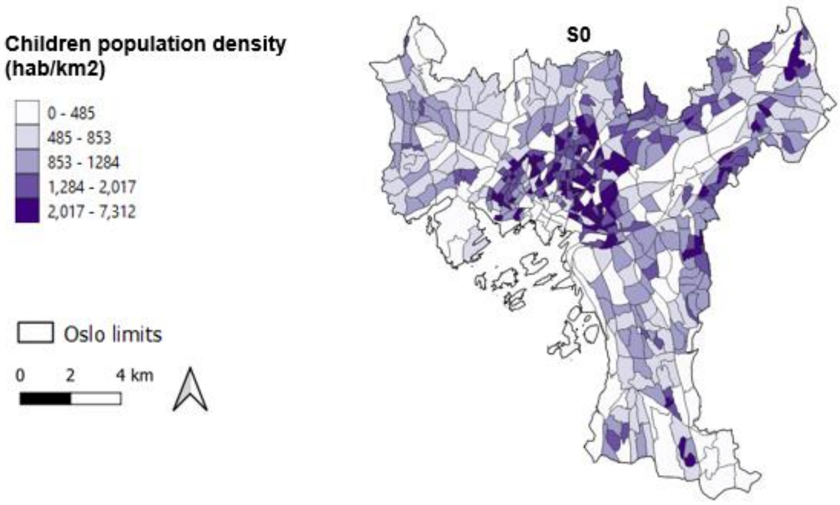
4.3. Changes in Particulate matter 10 (PM_{10}) (normalized)



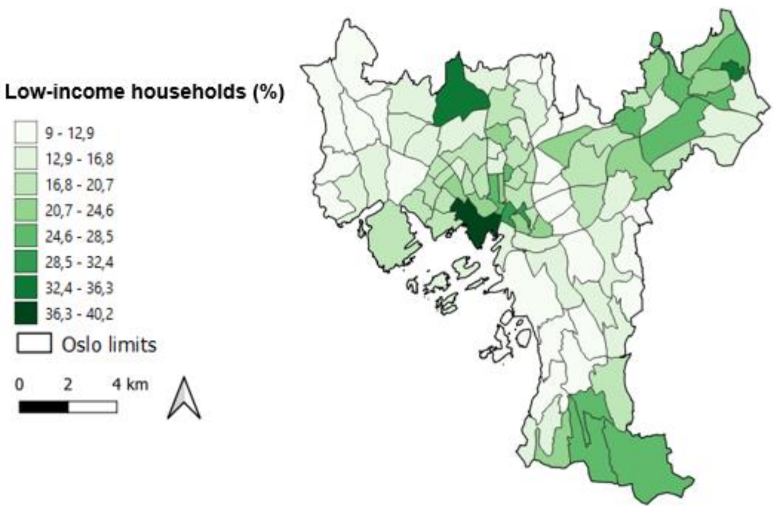
4.4. Population density



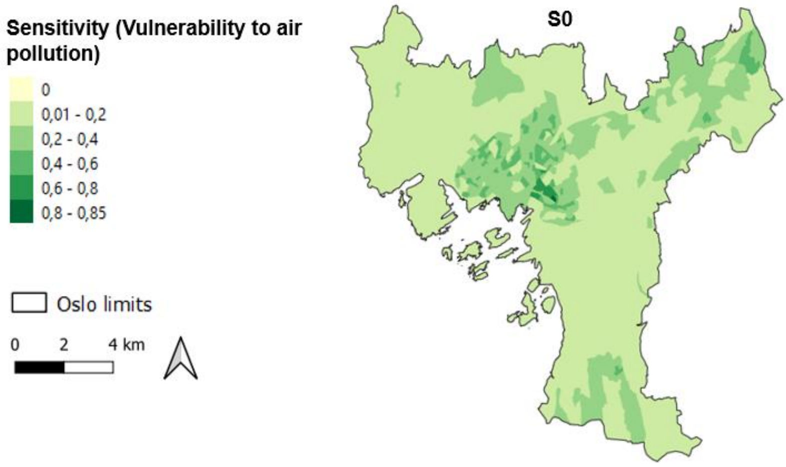
4.5. Children population density



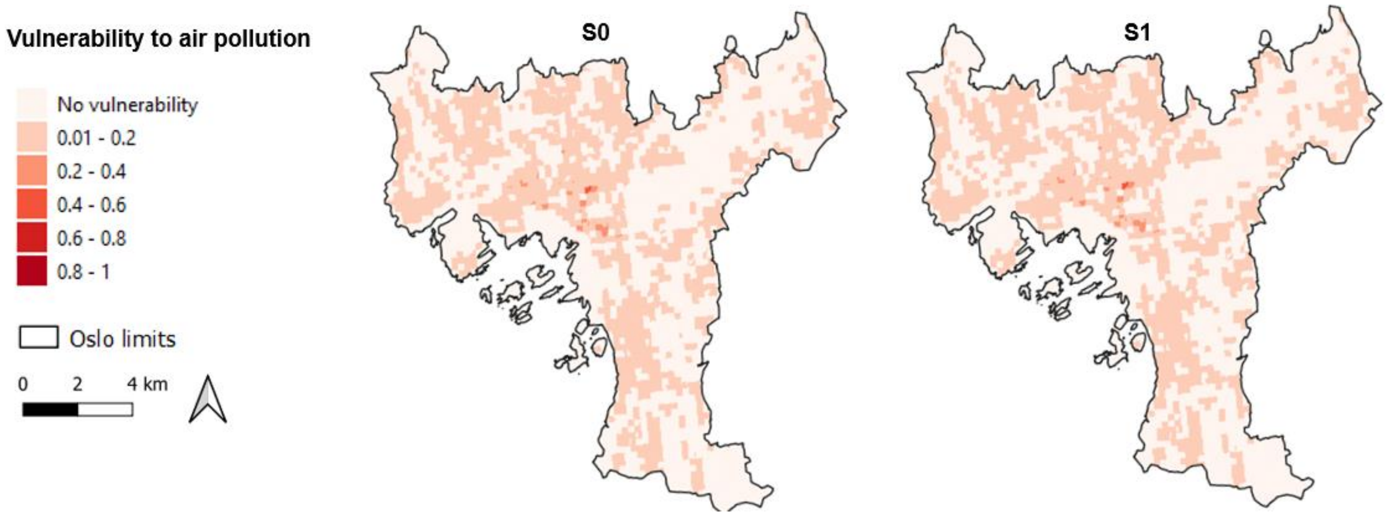
4.6. Low-income households

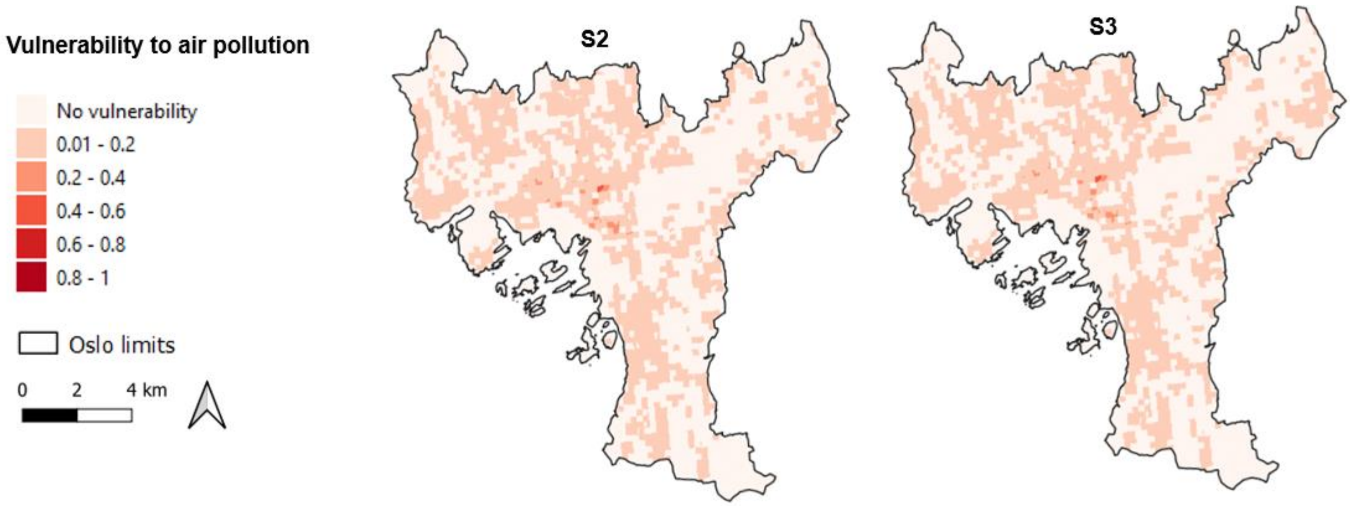


4.7. Sensitivity (Vulnerability to air pollution)



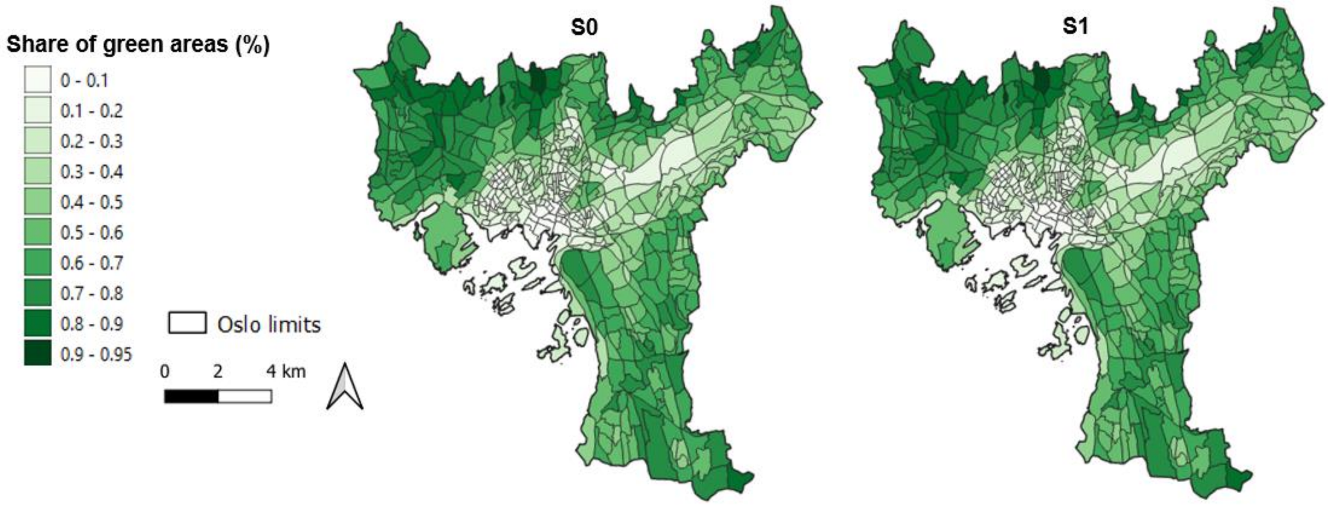
4.8. Vulnerability to air pollution

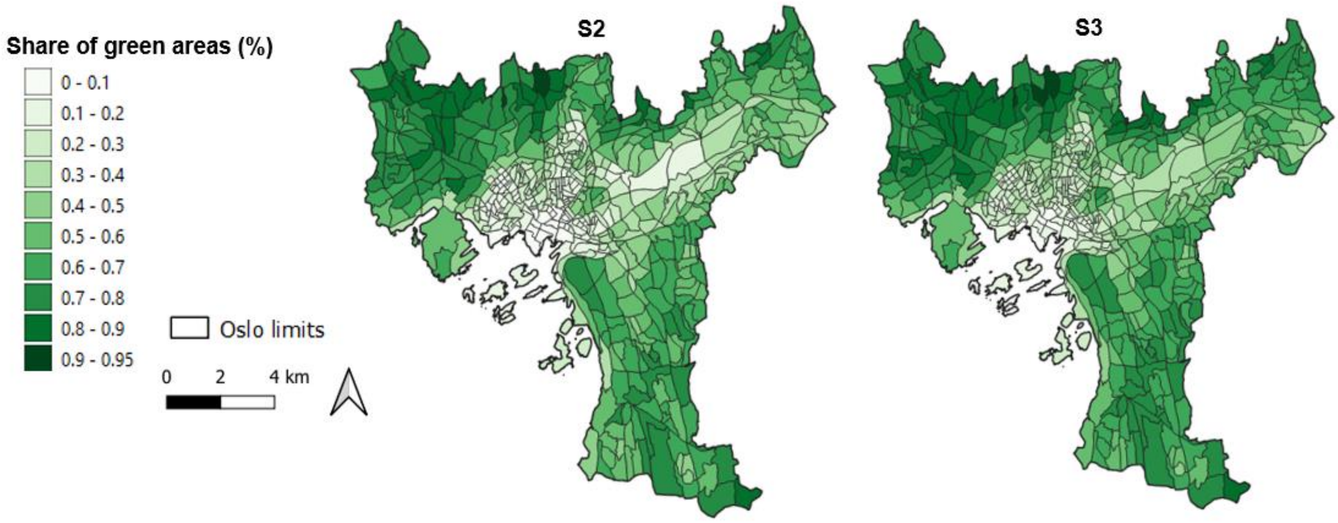




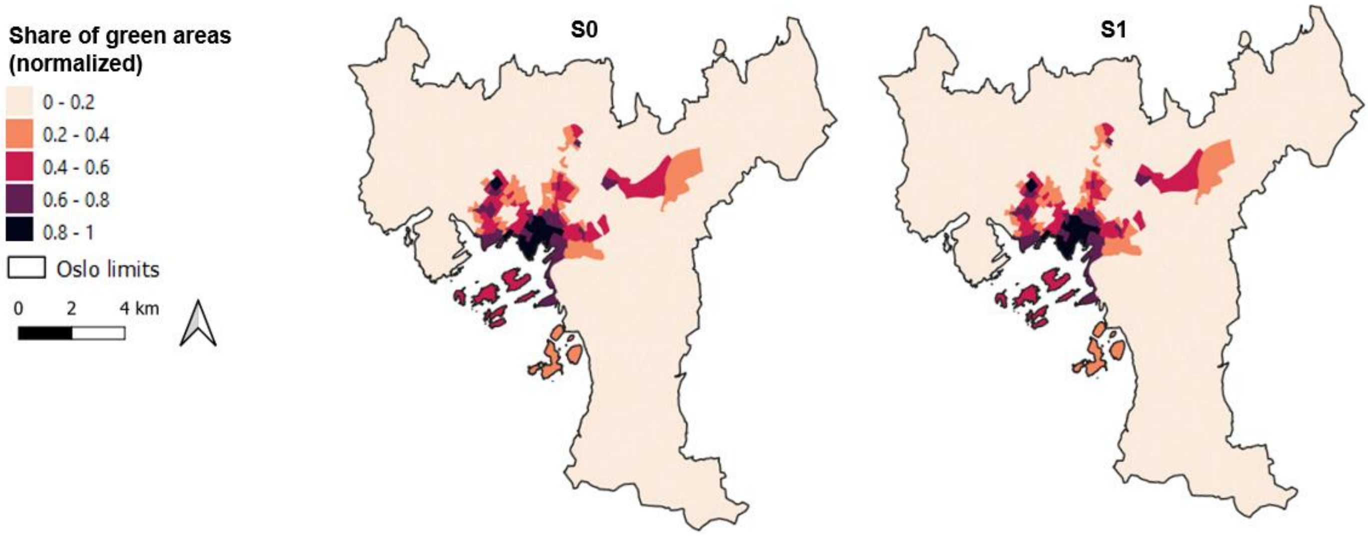
5. Vulnerability to lack of opportunities for interacting with natural environments

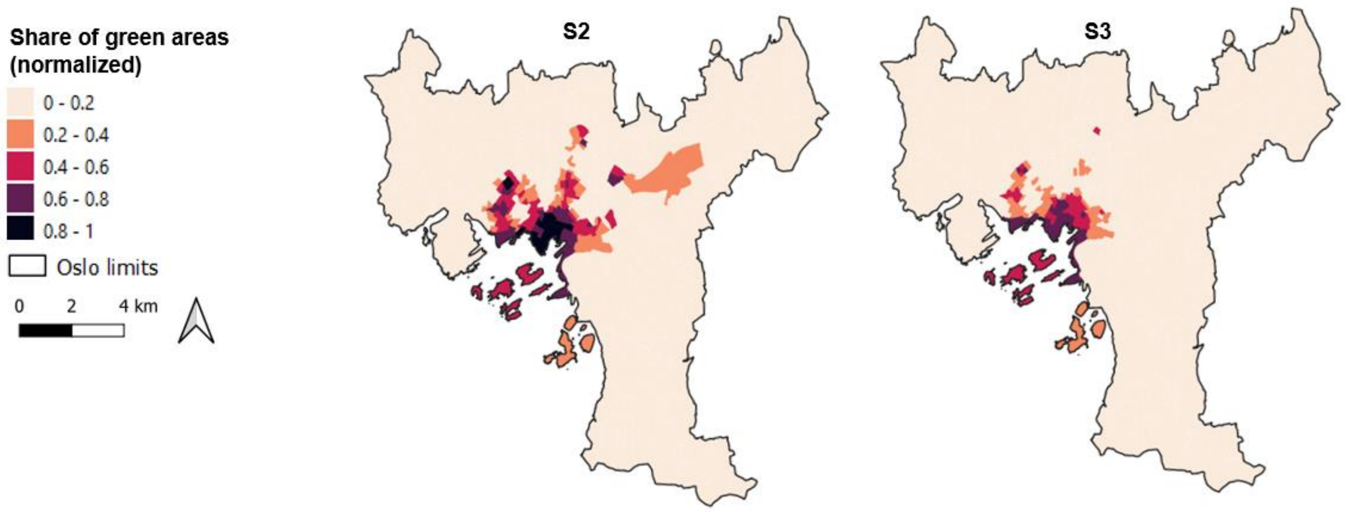
5.1. Share of green areas



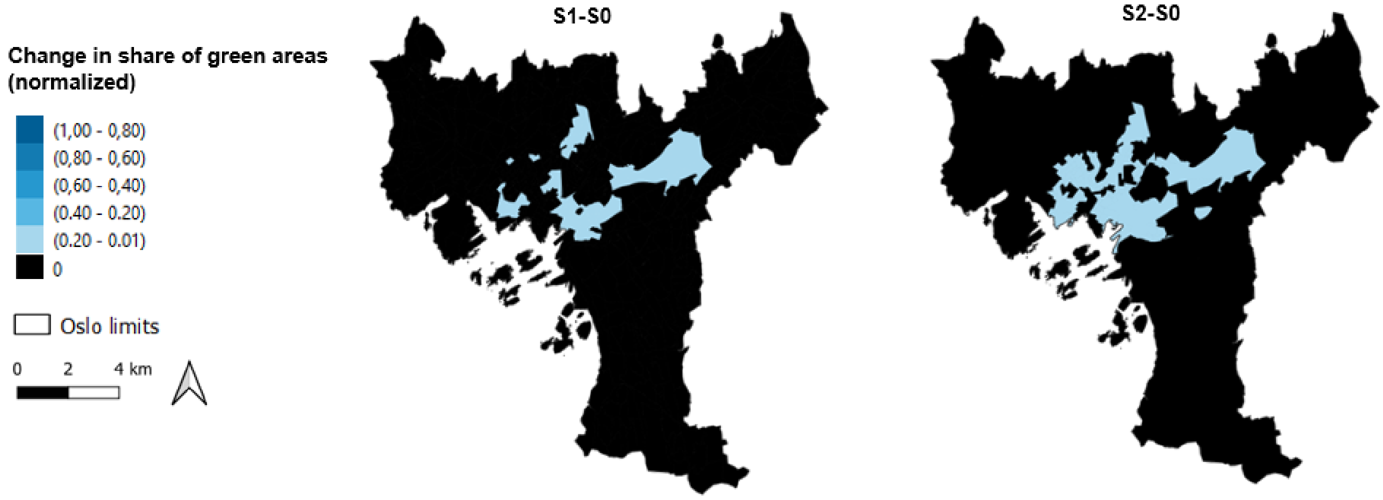


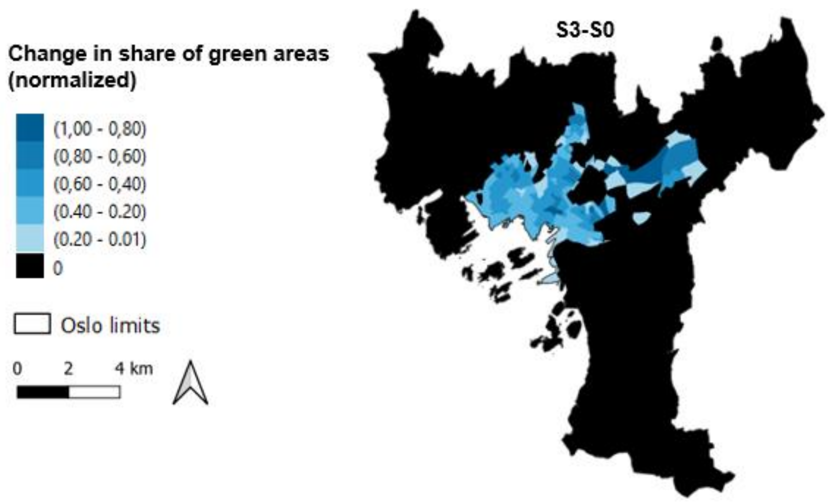
5.2. Share of green areas (normalized)



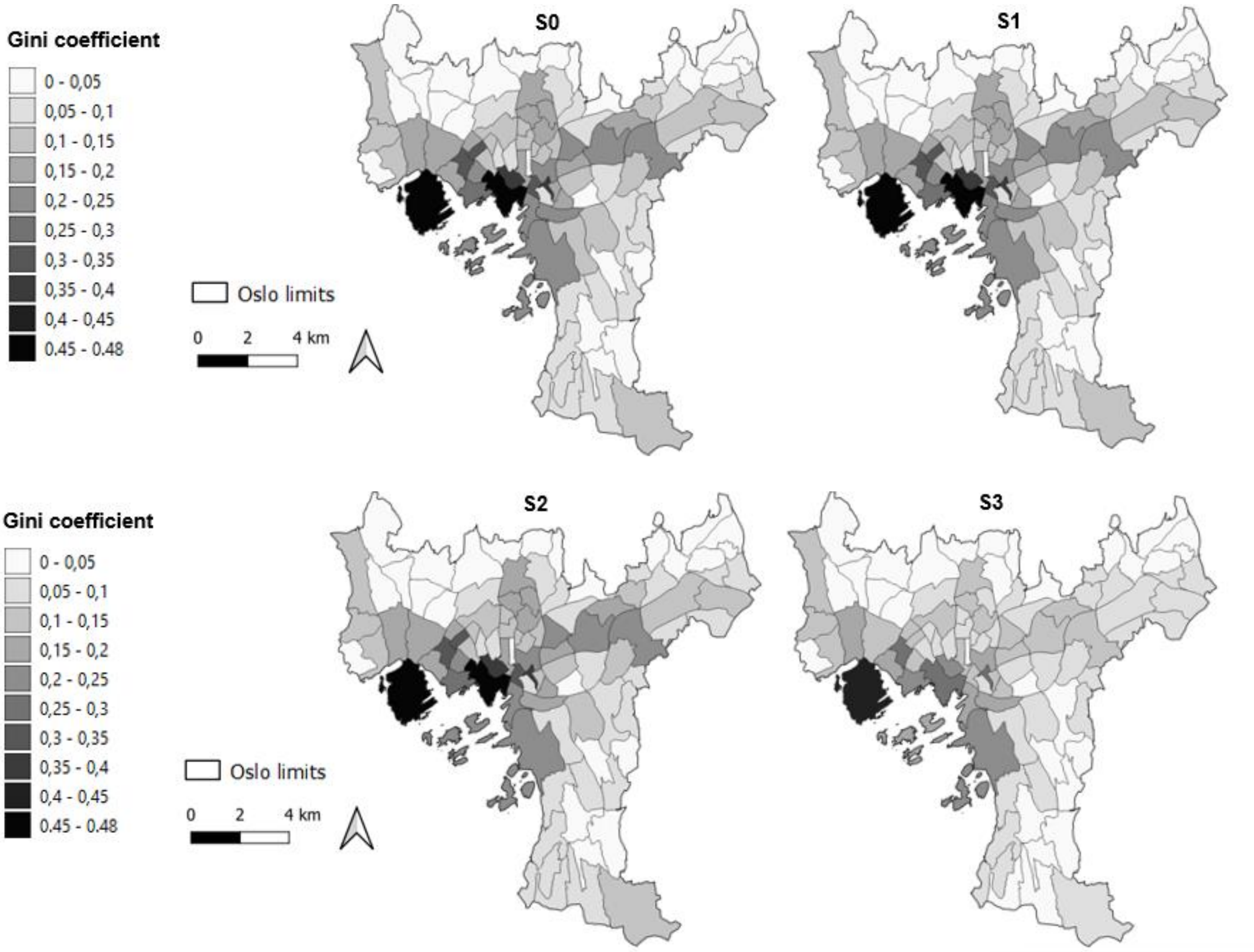


5.3. Change in share of green areas (normalized)

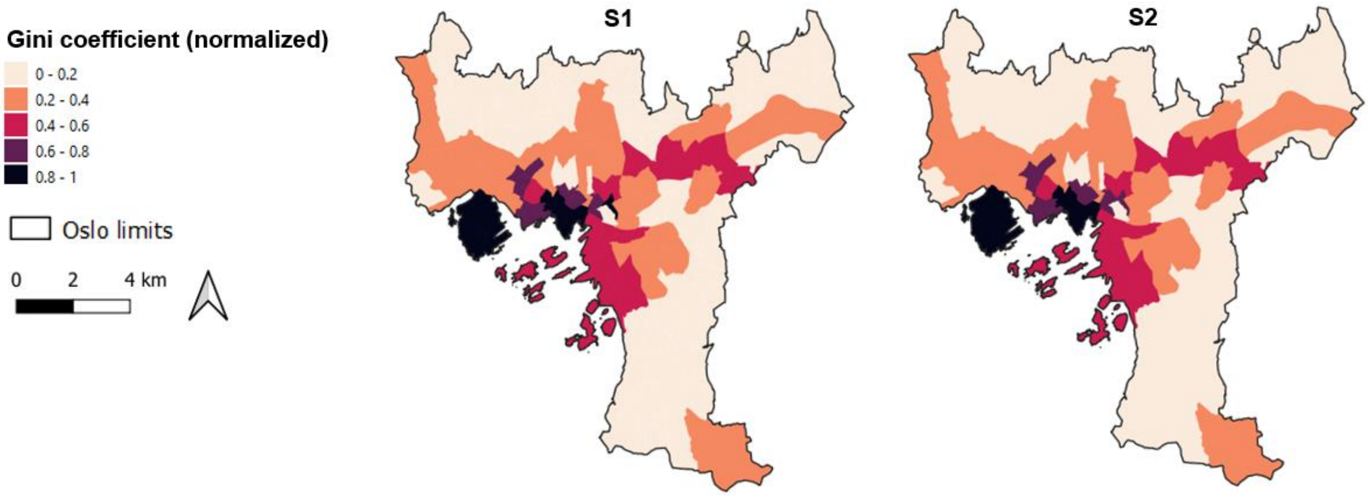


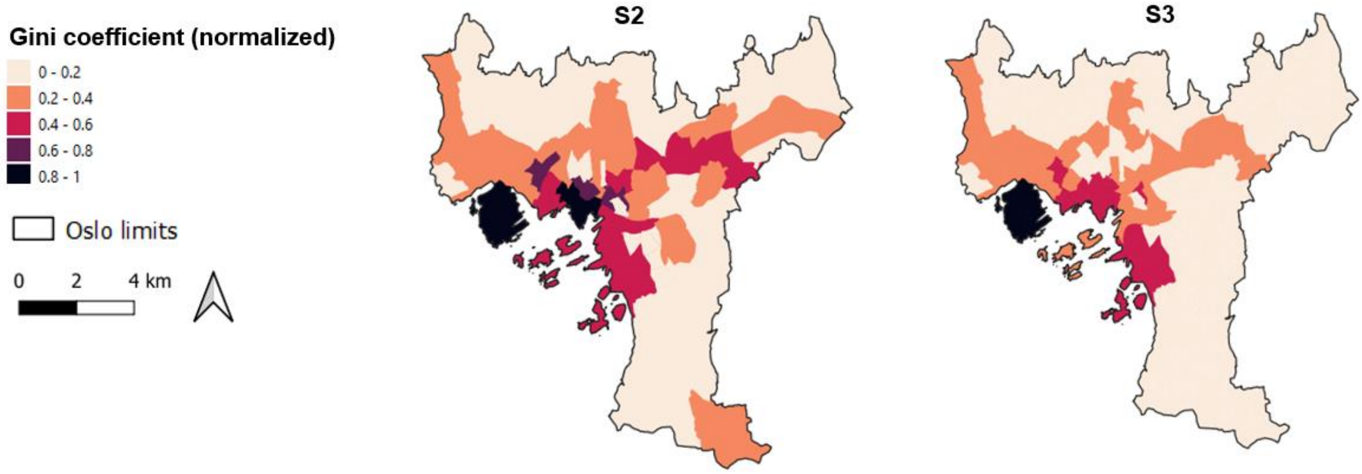


5.4. Green Gini coefficient

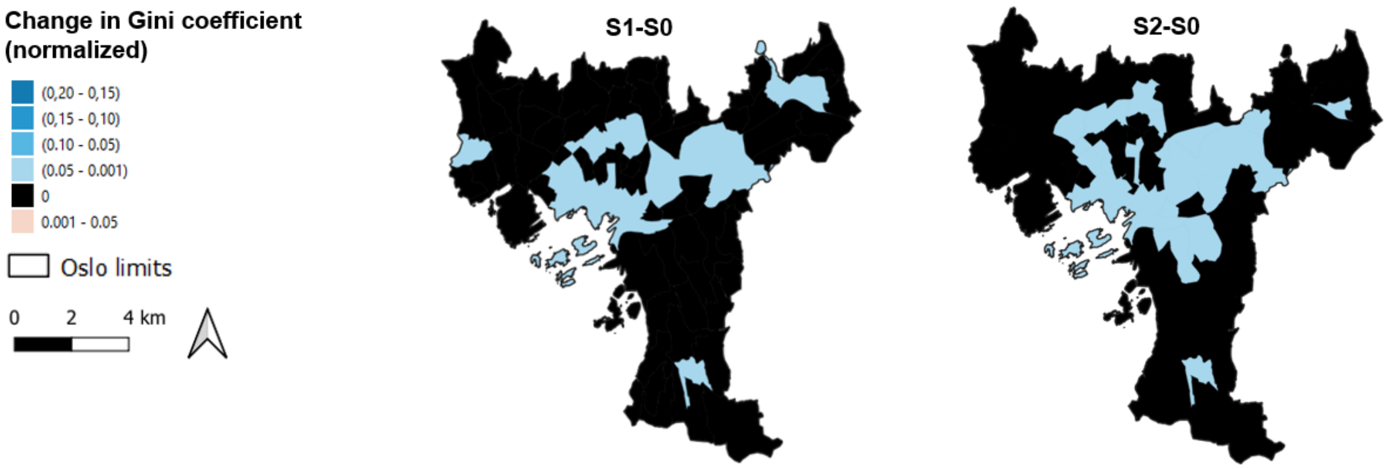


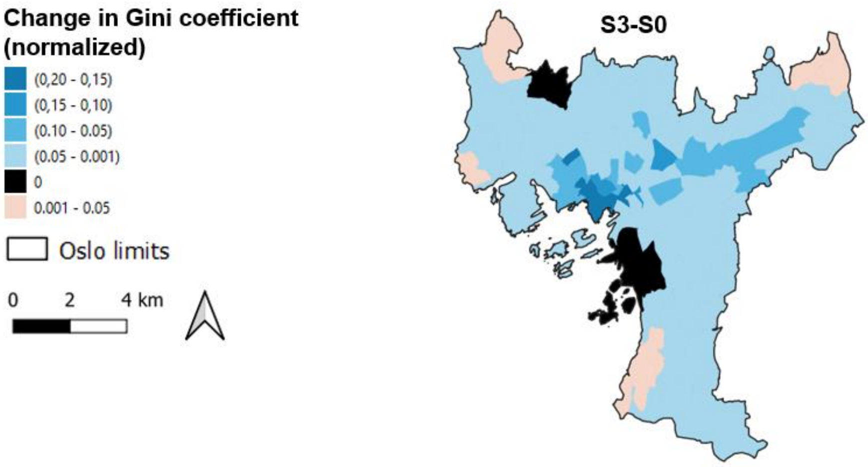
5.5. Green Gini coefficient (normalized)



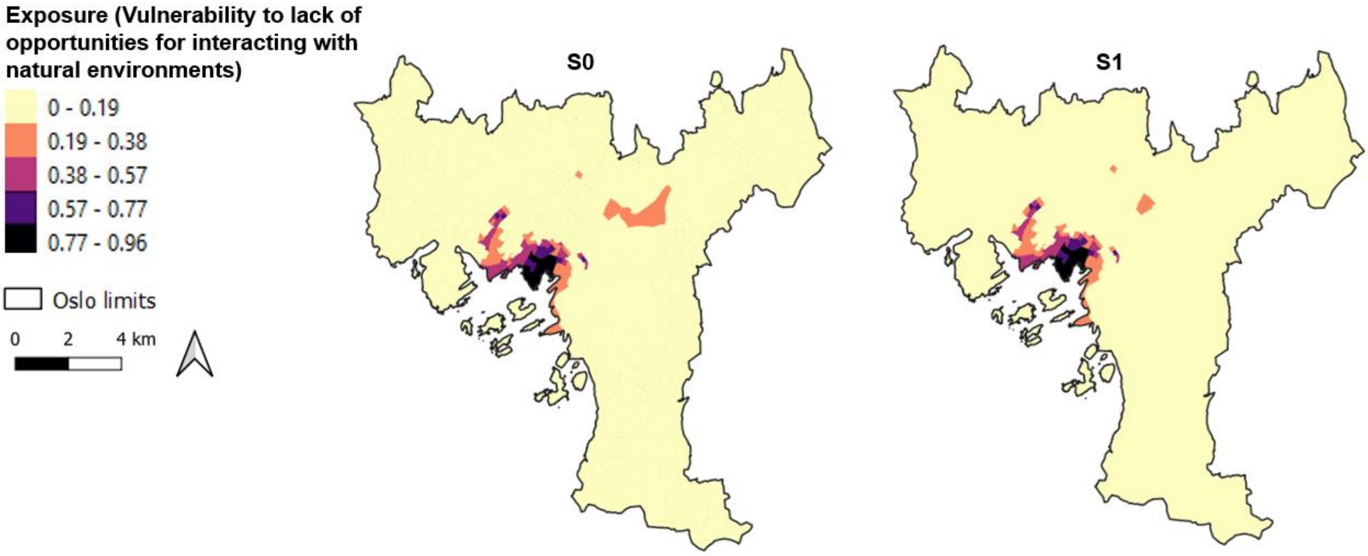


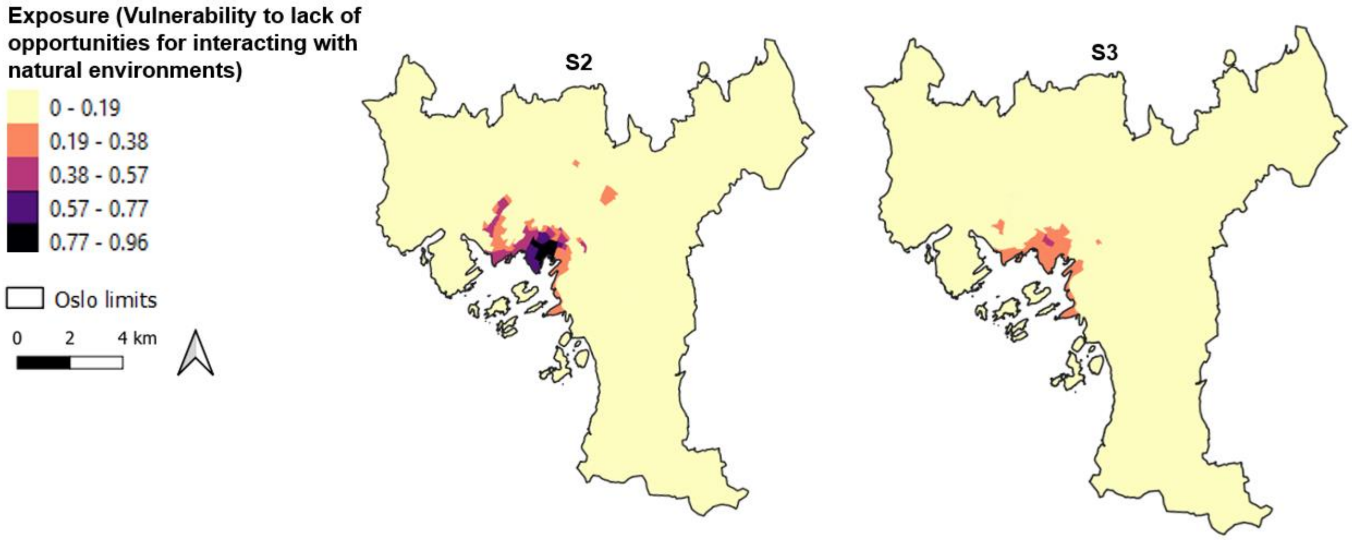
5.6. Change in Gini coeficient (normalized)



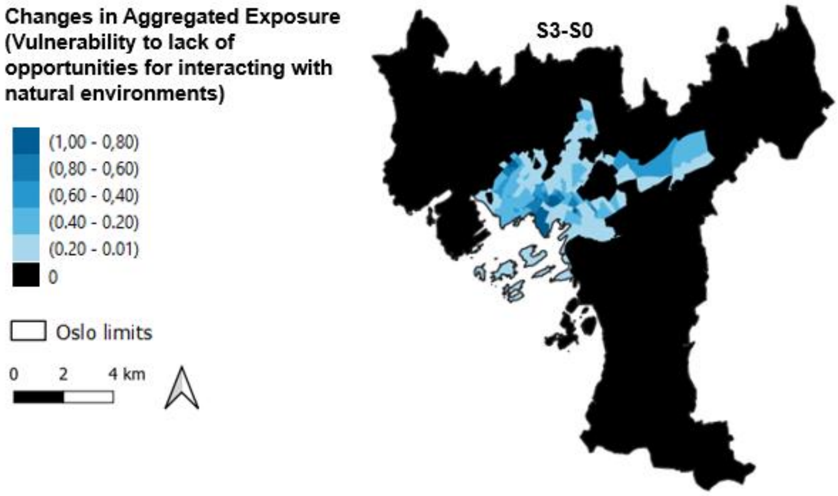


5.7. Exposure (Vulnerability to lack of opportunities for interacting with natural environaments)

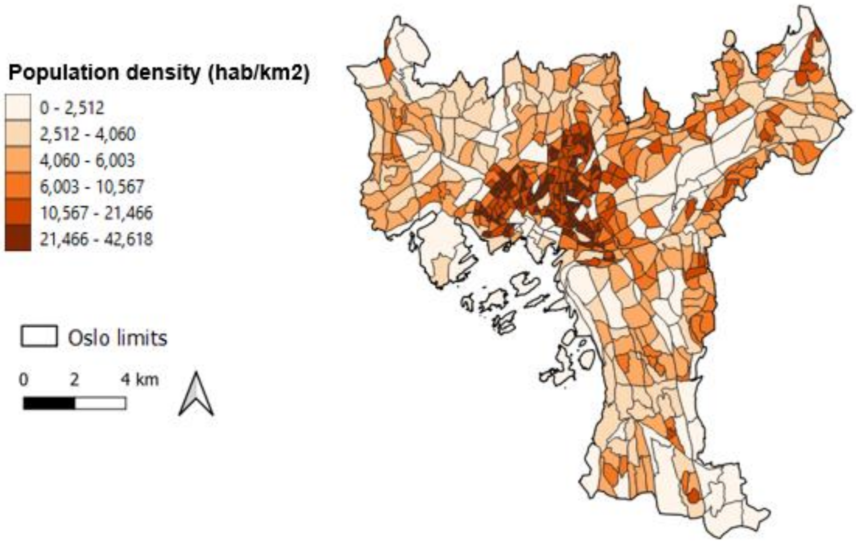




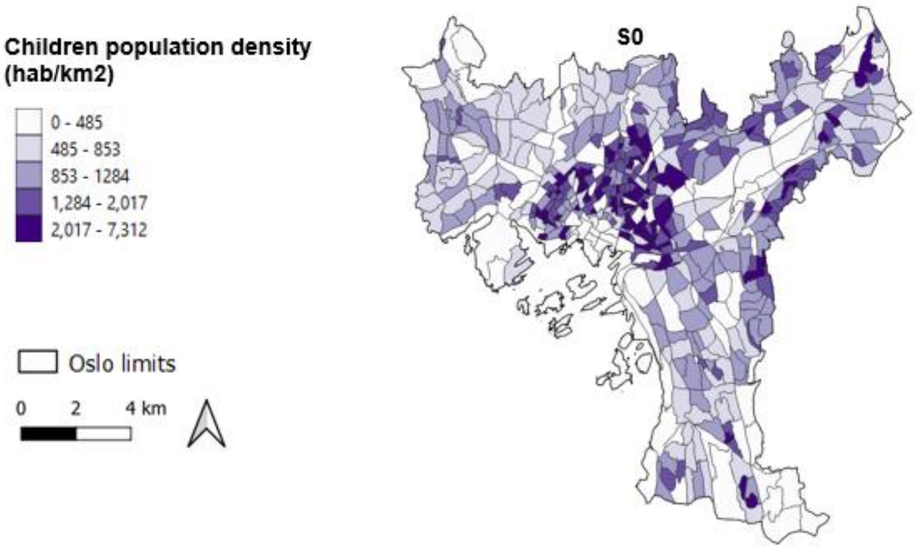
5.8. Changes in aggregated Exposure (Vulnerability to lack of opportunities for interacting with natural environaments)



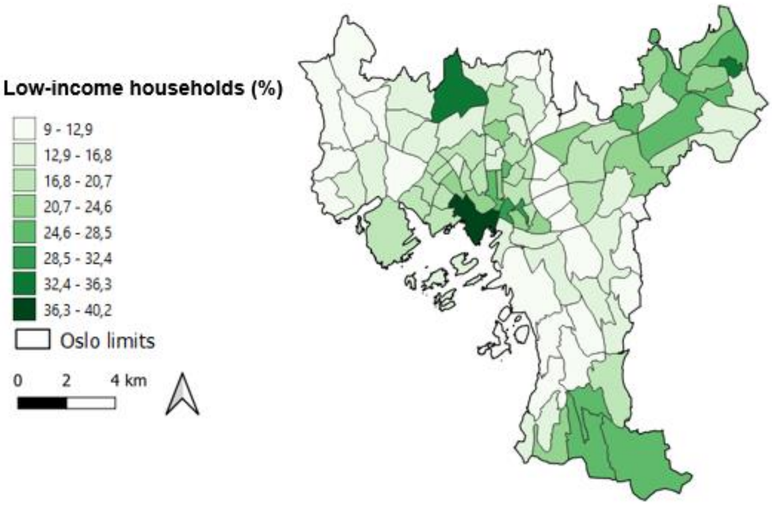
5.9. Population density



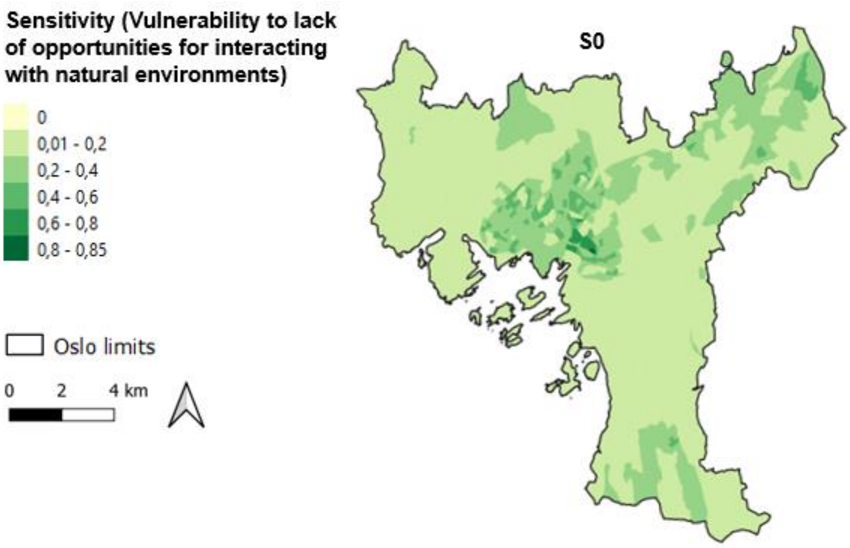
5.10. Children population density



5.11. Low-income households



5.12. Aggregated sensitivity (Vulnerability to lack of opportunities for interacting with natural environaments)



5.13. Vulnerability to lack of opportunities for interacting with natural environments