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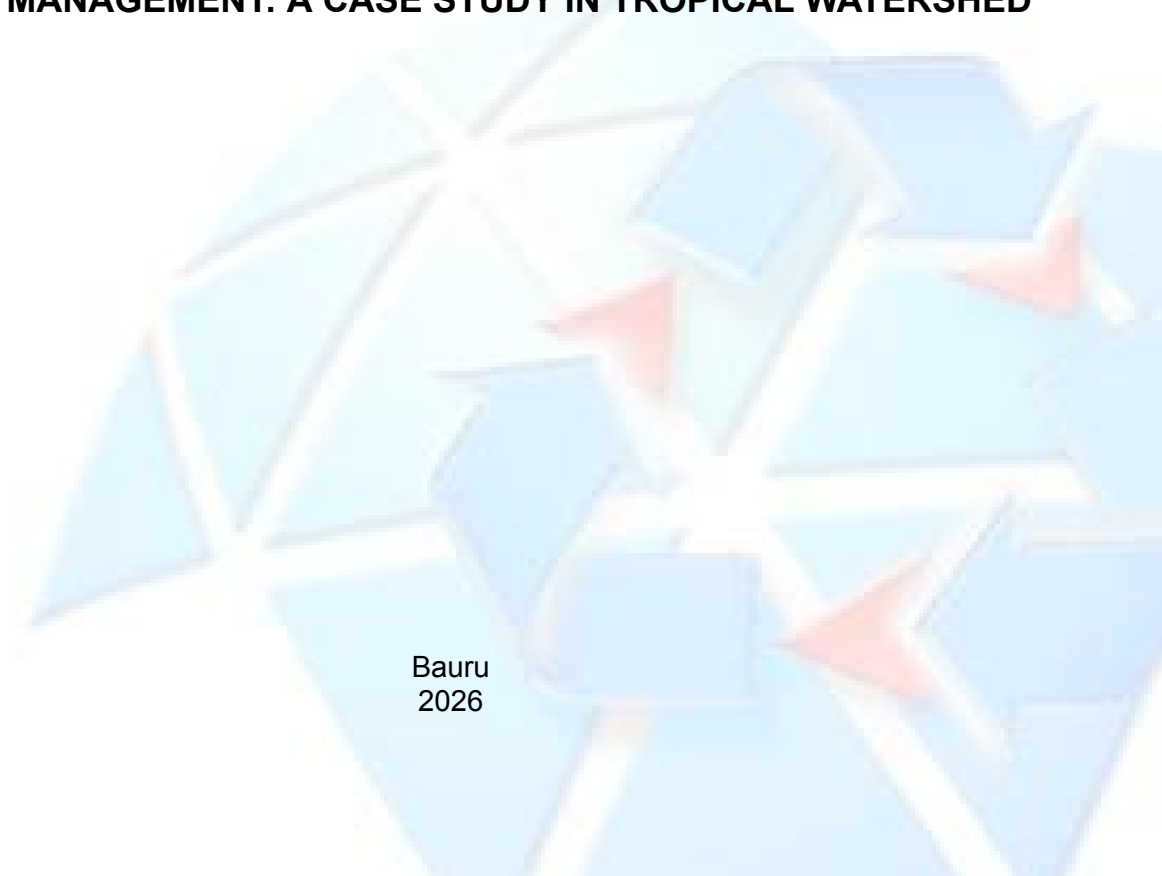
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**INTEGRATING NATURE-BASED SOLUTIONS (NBS) INTO URBAN
WATER MANAGEMENT: A CASE STUDY IN TROPICAL WATERSHED**

Bauru
2026



ANA LAURA PASCUCCI DE OLIVEIRA

**INTEGRATING NATURE-BASED SOLUTIONS (NBS) INTO URBAN
WATER MANAGEMENT: A CASE STUDY IN TROPICAL WATERSHED**

Master's thesis presented to São Paulo State University (UNESP), School of Engineering, Bauru, to obtain the degree of Master in Civil and Environmental Engineering.

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Advisor: Prof^a. Dr^a. Grazielle Ruas Lagoas da Silva

Co-advisor: Prof^a. Dr^a. Simone Andrea Furegatti



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"Our task is to create problems, awaken powerful responses to devastating events, calm troubled waters and rebuild tranquil places".

- *Donna Haraway*

ABSTRACT

Rapid urbanization has significantly altered the natural hydrological cycle, exacerbating problems such as flooding, watercourse degradation, and reduced infiltration. This research examines the application of Nature-Based Solutions (NbS) - specifically rain gardens, constructed wetlands, and bioswales - as sustainable strategies for urban surface water management in the Água Comprida watershed, Bauru/SP, Brazil. The study employs an integrated approach combining geoenvironmental analysis, automated spatial data processing, and hydrological modeling using the Storm Water Management Model (SWMM) engine. Due to the high volume of geospatial data, customized pre-processing routines were developed to automate data extraction and parameterization for 330 drainage units. The research also addressed significant spatial and constructive constraints inherent to consolidated urban environments, which limited the feasibility of certain NbS typologies in densely built-up sectors. Despite these restrictions, the hydrological performance analysis demonstrated that NbS effectiveness is highly scale-dependent; while centralized units like constructed wetlands ensure downstream volumetric stability, decentralized units such as rain gardens and bioswales are decisive for local hydrological recovery. An integrated scenario revealed a hydraulic synergy effect, where the serial arrangement of units optimized total runoff retention beyond the sum of individual contributions. These results highlight that NbS performance is highly dynamic and sensitive to local site-specific conditions and rainfall intensity, suggesting that effectiveness should be evaluated beyond average capacity. Sensitivity analysis identified imperviousness and rain gardens' berm height as the primary governing parameters for the system's hydrological response. Although performance declines under extreme hydrological stress, the findings confirm the technical feasibility and systemic benefits of nature-based interventions. This study provides important contributions for formulating public policies, offering both technical parameters and more robust guidelines to support the planning and prioritization of areas for NbS implementation in consolidated urban contexts.

Keywords: Nature Based Solutions, Sustainable Development, Environmental Management, Urban Hydrology, Hydrological Model.

RESUMO

A rápida urbanização alterou significativamente o ciclo hidrológico natural, exacerbando problemas como inundações, degradação de cursos d'água e redução da infiltração. Esta pesquisa examina a aplicação de Soluções baseadas na Natureza (SbN) — especificamente jardins de chuva, *wetlands* construídas e *bioswales* — como estratégias sustentáveis para o manejo de águas superficiais urbanas na bacia do Córrego Água Comprida, Bauru/SP, Brasil. O estudo utiliza uma abordagem integrada que combina análise geoambiental, processamento automatizado de dados espaciais e modelagem hidrológica através do software *Storm Water Management Model* (SWMM). Devido ao elevado volume de dados geoespaciais, foram desenvolvidas rotinas customizadas de pré-processamento para automatizar a extração de dados e a parametrização de 330 unidades de drenagem. A pesquisa também abordou restrições espaciais e construtivas significativas inerentes a ambientes urbanos consolidados, as quais limitaram a viabilidade de certas tipologias de SbN em setores densamente edificadas. Apesar dessas restrições, a análise de desempenho hidrológico demonstrou que a eficácia das SbN é altamente dependente da escala; enquanto unidades centralizadas, como as *wetlands* construídas, garantem a estabilidade volumétrica a jusante, unidades descentralizadas, como jardins de chuva e *bioswales*, são decisivas para a recuperação hidrológica local. Um cenário integrado revelou um efeito de sinergia hidráulica, onde a disposição das unidades em série otimizou a retenção total do escoamento além da soma das contribuições individuais. Estes resultados destacam que o desempenho das SbN é altamente dinâmico e sensível às condições locais específicas e à intensidade da precipitação, sugerindo que a eficácia deve ser avaliada além da capacidade média. A análise de sensibilidade identificou a impermeabilidade e a altura da berma dos jardins de chuva como os principais parâmetros governantes para a resposta hidrológica do sistema. Embora o desempenho decline sob estresse hidrológico extremo, os achados confirmam a viabilidade técnica e os benefícios sistêmicos das intervenções baseadas na natureza. Este estudo fornece contribuições importantes para a formulação de políticas públicas, oferecendo parâmetros técnicos e diretrizes mais robustas para

apoiar o planejamento e a priorização de áreas para a implementação de SbN em contextos urbanos consolidados.

Palavras-chave: Soluções Baseadas na Natureza, Desenvolvimento Sustentável, Gestão Ambiental, Hidrologia Urbana, Modelo Hidrológico.

LIST OF FIGURES

Figure 1-1: Impacts of watershed urbanization on the water cycle.....	3
Figure 1-2: Summary of Eggermont's NbS classification.....	9
Figure 1-3: Comparison of the capital and operating costs of structural solutions and NbS over time. CAPEX: Capital Expenditures, OPEX: Operational Expenditures.....	11
Figure 1-4: Typical configuration of a rain garden.....	15
Figure 1-5: Constructed wetlands types and description.....	21
Figure 1-6: Typical configuration of a Free Water Surface Constructed Wetlands.....	24
Figure 1-7: Typical configuration of a bioswales.....	29
Figure 2-1: Location map.....	51
Figure 2-2: Basin area, erosion features and flood area.....	52
Figure 2-3: Photographic record of a stretch near the outlet of the Água Comprida stream, emphasis on the sandbank and solid waste.....	55
Figure 2-4: Delineation of changes in the banks of two stretches (A and B) near the outlet of the stream between 2004 and 2022.....	55
Figure 3-1: Workflow diagram.....	65
Figure 3-2: Illustrative result of the practical application of the “Outlets” and “Drainage Areas” scripts.....	69
Figure 3-3: Illustrative result of the practical application of the “Equivalent Slope” script.....	69
Figure 4-1: Location map of the study area.....	83
Figure 4-2: Flowchart of the preprocessing routine.....	88
Figure 4-3: Potential implementation sites for Nature-based Solutions (rain gardens, bioswales, and constructed wetlands) identified in the study area.....	91
Figure 4-4: Outlets (red points), drainage areas (orange polygons), flow paths (light blue lines), and Água Comprida Stream (central dark blue line).....	96
Figure 4-5: Example of a longitudinal profile of the flow path.....	97

Figure 4-6. Spatial representation of flow routing between river basins, generated from the D8 model. The arrows indicate the direction of surface runoff, connecting the source and destination areas.....	98
Figure 4-7: Map of reclassified land cover in drainage areas.....	99
Figure 5-1: Location Map of the Study Area.....	113
Figure 5-2: Number of drainage areas with total elimination of peak flow across study scenarios and rainfall events.....	135
Figure 5-3: Spatial variability of the surface runoff coefficient under the baseline and integrated scenarios for the design rainfall.....	138
Figure 5-4: Spatial variability of the surface runoff coefficient under the baseline and integrated scenarios for the real event.....	139
Figure 5-5: Comparison of runoff removal efficiency (infiltration and storage) by NBS scenario for Design Rainfall and Real Event.....	140
Figure 5-6: Unit performance in terms of volume removed by NBS type.....	143
Figure 5-7: Average Unitary Performance of Removed Volume per Unit in each Scenario.....	144
Figure 5-8: Average unitary performance of infiltrated volume by NBS type.....	146
Figure 5-9: Average unitary performance of infiltrated volume per unit in each scenario.....	147
Figure 5-10: Average unitary performance of storage volume by NBS type.....	149
Figure 5-11: Average unitary performance of stored volume per unit in each scenario.	150
Figure 5-12: Average unitary peak flow reduction performance per NBS type.....	151
Figure 5-13: Average unitary peak flow reduction performance per unit in each scenario.....	153
Figure 5-14: Average unitary peak delay performance per NBS typology.....	154
Figure 5-15: Average unitary peak flow delay performance per unit in each scenario... ..	155
Figure 8-S1. Screenshot of the preprocessing routine repository interface on GitHub. Available at https://github.com/anapascucci/preprocessingroutine	173
Figure 8-S2. Slope map of the Água Comprida stream watershed.....	174

LIST OF TABLES

Table 2-1: Analysis of non-compliant samples at the point near the headwaters.....	57
Table 2-2: Analysis of non-compliant samples at the points near the outlet.....	57
Table 3-1: Packages and functions common to the scripts.....	66
Table 3-2: Input data, description and file format.....	66
Table 4-1: Key Implementation criteria adopted.....	85
Table 4-2: Input data, description and file format.....	87
Table 5-1: Horton infiltration model parameters adopted.....	116
Table 5-2: Construction criteria for rain gardens.....	122
Table 5-3: Construction criteria for wetlands.....	122
Table 5-4: Construction criteria for bioswales.....	123
Table 5-5: Input parameters in SWMM.....	125
Table 5-6: Classification intervals for the surface runoff coefficient.....	126
Table 5-7: Parameters selected for sensitivity analysis, including baseline values and variation intervals.....	129
Table 5-8: Summary of Water Balance Components and Continuity Errors for Simulated Scenarios.....	131
Table 5-9: Summary of hydrological performance parameters at the outfall: average and maximum flows, total volume, volume reduction and time to peak for the baseline and intervention scenarios.....	133
Table 5-10: Distribution of drainage areas by runoff coefficient classification under the Design Rainfall scenario.....	136
Table 5-11: Distribution of drainage areas by runoff coefficient classification under the Monitored Real Event.....	137
Table 5-12: Normality Test Results (Shapiro-Wilk) for Raw Data and Model Residuals	141
Table 5-13. Post-hoc comparative analysis of Intrinsic Typological Efficiency for removed volume across different NBS techniques.....	143

Table 5-14. Post-hoc comparative analysis of Strategic Scenario Efficiency for removed volume across implementation scenarios, grouped by rainfall event type.....	145
Table 5-15. Post-hoc comparative analysis of Intrinsic Typological Efficiency for infiltrated volume across different NBS techniques.....	146
Table 5-16. Post-hoc comparative analysis of Strategic Scenario Efficiency for infiltrated volume across implementation scenarios, grouped by rainfall event type.....	147
Table 5-17. Post-hoc comparative analysis of Intrinsic Typological Efficiency for storage volume across different NBS techniques.....	149
Table 5-18. Post-hoc comparative analysis of Strategic Scenario Efficiency for storage volume across implementation scenarios, grouped by rainfall event type..	150
Table 5-19. Post-hoc comparative analysis of Intrinsic Typological Efficiency for peak flow reduction across different NBS techniques.....	152
Table 5-20. Post-hoc comparative analysis of Strategic Scenario Efficiency for peak flow reduction across implementation scenarios, grouped by rainfall event type.....	153
Table 5.21. Post-hoc comparative analysis of Intrinsic Typological Efficiency for peak flow delay across different NBS techniques.....	154
Table 5-22. Post-hoc comparative analysis of Strategic Scenario Efficiency for peak flow delay across implementation scenarios, grouped by rainfall event type.....	155
Table 5-23: Qualitative assessment of parameter influence on runoff and storage variables.....	156
Table 5-24: Summary of the sensitivity analysis results.....	157
Table 8-S1. Characterization of drainage areas: total drainage area values, type of NbS located, the maximum and minimum limits recommended for each type of solution, and the area available for implementation in each drainage area. The available area values that do not meet the minimum required criteria have been highlighted in red.....	174
Table 8-S2. Distribution of drainage areas by type of land use. For each drainage area, the total values in square meters (m ²) and the respective percentage in relation to the total area of the drainage area are shown.....	187

Table 9-1: Data, sources and access links to obtain the information used in the study..
..... 225

LIST OF ACRONYMS AND ABBREVIATIONS

NbS	Nature-based Solutions
CAPEX	Capital Expenditure
OPEX	Operational Expenditure
IISD	International Institute for Sustainable Development
ES	Ecosystem Service
LID	Low Impact Development
BMPs	Best Management Practices
WSUD	Water Sensitive Urban Design
SuDS	Sustainable Drainage Systems
GI	Green Infrastructure
BGI	Blue-Green Infrastructure
EbA	Ecosystem-based Adaptation
Eco-DRR	Ecosystem-based Disaster Risk Reduction
SAVi	Sustainable Asset Valuation
SDGs	Sustainable Development Goals
O ₂	Oxygen
Fe	Iron
Ca	Calcium
Al	Aluminum
P	Phosphorus
SWMM	Storm Water Management Model
DTM	Digital Terrain Model
IDF	Intensity-Duration-Frequency
DAE	Bauru Water and Wastewater Department
EPA	U.S. Environmental Protection Agency
CW	Constructed Wetlands
FWS CW	Surface Flow Constructed Wetlands
SSF CW	Subsurface Flow Constructed Wetlands
IGC	São Paulo Geographic and Cartographic Institute
INPE	National Institute for Space Research
IPMET	Meteorological Research Institute
CETESB	Companhia Ambiental do Estado de São Paulo

CONTENTS

Chapter 1: Introduction.....	17
1.1 Urbanization and its effects on the hydrological cycle.....	17
1.2 Scope of this Dissertation.....	20
1.2.1 General Objectives.....	20
1.2.2 Specific Objectives.....	20
1.3 Organization of this Dissertation.....	21
1.4 Water resources management.....	22
1.5 Nature-based Solutions.....	24
1.5.1 Rain gardens.....	29
1.5.2 Constructed Wetlands.....	36
1.5.3 Bioswales.....	44
1.6 Data Acquisition.....	48
1.7 Hydrological Model in SWMM.....	50
1.8 References.....	51
Chapter 2: Geoenvironmental Analysis of a Tropical Urban Stream Basin: Diagnosis and Prognosis.....	64
2.1 Introduction.....	65
2.2 Materials and Methods.....	66
2.3 Results.....	68
2.3.1 Erosion Processes.....	68
2.3.2 Siltation and banks changes.....	70
2.3.3 Hydrological risk.....	71
2.3.4 Water Quality.....	72
2.4 Discussion.....	75
2.5 Concluding Remarks.....	76
2.6 References.....	77
Chapter 3: Extracting, Structuring and Organizing Spatial, Physical and Hydrological Data for Modeling: an Approach Using	

	Programming Environment.....	79
3.1	Introduction.....	80
3.2.1	General structure of the computer routine.....	81
3.2.2	Input Data.....	82
3.3.1	Outlets.....	83
3.3.2	Drainage Areas.....	84
3.3.3	Flow Path.....	85
3.3.4	Equivalent Slope.....	87
3.3.5	Flow Routing.....	88
3.3.6	Land Cover.....	89
3.4	Discussion.....	90
3.5	Concluding Remarks.....	91
3.6	Data Availability Statement.....	91
3.7	References.....	91
	Chapter 4: Planning Nature-Based Solutions in Urban Watersheds: Site Selection Criteria and Automated Hydrological Data Pre-Processing.....	93
4.1	Introduction.....	94
4.2	Material and Methods.....	97
4.3	Results.....	106
4.4	Discussion.....	115
4.5	Conclusions.....	119
4.6	References.....	122
	Chapter 5: Integrated Management of Nature-Based Solutions: Planning and Hydrological Performance in a Tropical Urban Watershed.....	127
5.1	Introduction.....	128
5.2	Materials and methods.....	129
5.2.1	Study Area.....	129
5.2.2	Input Data, Parameters, and Hydrological Model Configuration.....	130

5.2.3	Rainfall Data Processing and Event Definition.....	135
5.2.4	Construction Criteria for NbS.....	138
5.2.5	Input parameters of NBS into the model.....	140
5.2.6	Performance Analysis.....	142
5.2.7	Model Calibration and Validation.....	144
5.3	Results.....	146
5.3.1	Integrated Hydrological Response at the Watershed Scale.....	147
5.3.2	Spatial Variation of Impact Across Drainage Areas.....	151
5.3.3	Comparative Performance of NBS Typologies and Scenarios.....	158
5.3.4	Sensitivity analyses.....	174
5.4	Discussions.....	176
5.4.1	Study Limitations and Future Perspectives.....	177
5.5	Conclusions.....	178
5.6	References.....	180
	Chapter 6: Conclusions and Recommendations.....	187
	Chapter 7: Academic background of the author.....	189
7.1	Academic Profile.....	189
7.2	PAADES Program.....	189
7.3	Research Projects.....	189
7.4	Conference Presentations.....	190
7.5	Publications.....	190
7.6	Additional Activities.....	190
	Chapter 8: Appendix.....	192
8.1	Supplementary Material relating to Chapter 3.....	192
8.2	Supplementary Material relating to Chapter 4.....	193
	Chapter 9: Annexes.....	244

Chapter 1:

Introduction¹

1.1 Urbanization and its effects on the hydrological cycle

Historically, human activities lead to changes in land use and occupation patterns and water use, triggering a series of processes that alter the environment and the local hydrological system. These changes, aimed at expanding agricultural and economic development, often cause negative impacts and result in the degradation of the environment. In this sense, urbanization represents one of the anthropogenic actions that generate the greatest environmental impacts, linked to unplanned changes in the original characteristics of land use and the lack of proper water resource management (MIGUEZ; VERÓL & REZENDE, 2016; MAGALHÃES JÚNIOR & LOPES, 2022; TUNDISI, 2003).

As a consequence of the urbanization process, ecosystems are degraded, resulting in reduced capacity for carbon sequestration, water retention, maintenance of biodiversity, and preservation of healthy watercourses. In addition, these processes affect social well-being and the sustainability of agricultural productivity. The situation is aggravated by climate change, which intensifies these impacts by modifying rainfall regimes and increasing the frequency and severity of hydrological disasters, hindering the establishment of resilient and more habitable territories (SHIAO et al., 2020; BUSH & DOYON, 2019).

A watershed in its natural state has its behavior defined by the hydrological cycle, generally with a balanced configuration. Water moves through the cycle by changing state, generating flows between the atmosphere, the biosphere and the lithosphere. In general, vegetation, while promoting evapotranspiration, increases water infiltration and protects against runoff on slopes and river banks. Meanwhile, in the lithosphere, infiltration processes favor the recharge of aquifers, guaranteeing a base flow for watercourses. Thus, managing the quality and quantity of water resources is closely related to the conservation conditions of its watershed (MIGUEZ; VERÓL & REZENDE, 2016; FERRAZ & LIMA, 2022).

¹This chapter was written exclusively for this dissertation and has not been submitted for publication.

In urban centers, uncontrolled occupation combined with urban development pressures, planning failures, vegetation removal, climate change, and human activities modifies the urban water cycle. The impacts of urbanization on the natural hydrological cycle begin with land clearing and continue through infrastructure implementation. The initial removal of vegetation eliminates its natural hydrological functions: intercepting rainfall, slowing surface flow, and facilitating evapotranspiration. Subsequent earthworks flatten the terrain, removing natural depressions that serve as temporary water storage for infiltration and evaporation, thereby accelerating runoff.

The removal of surface soil horizons and subsoil compaction reduces or eliminates groundwater recharge and moisture retention capacity, impairing evapotranspiration (TORONTO AND REGION CONSERVATION, 2006). Impermeable surfaces, a hallmark of urbanization, further restrict water percolation into soils, exacerbating runoff volume and speed. Consequently, watershed sealing produces earlier and higher flood peaks compared to natural conditions, increasing risks of urban flooding, waterlogging, siltation, and erosion (SHIKANGALAH et al., 2016).

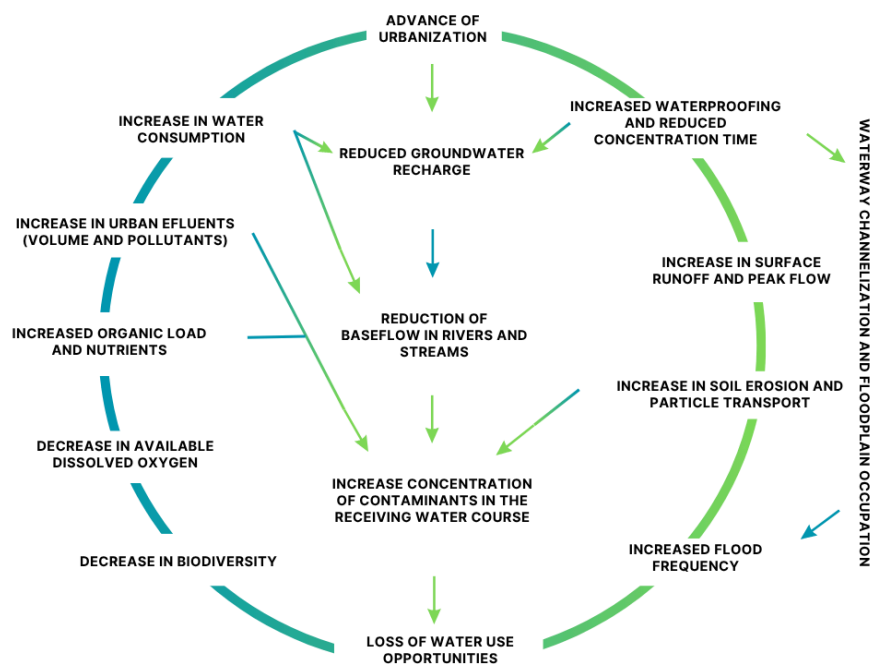
To manage excess runoff, artificial drainage systems are introduced to quickly discharge stormwater and prevent floods. However, the artificial drainage systems alter runoff patterns, reducing infiltration opportunities and generating rapid hydrological responses, often exacerbating the watershed degradation. Thus, water that previously percolated through the soil and recharged aquifers becomes surface runoff, carrying waste, suspended soil particles, nutrients, organic matter, pesticides, and other surface-derived chemical compounds. These constituents generate diffuse pollution that degrades both surface and groundwater quality (PORTO et al., 2009; TUCCI, 2005; VON SPERLING, 2016).

Among the environmental impacts caused by watershed modifications are changes in the flow regime of the rivers, resulting in siltation, accumulation of urban waste in the channels and plains, and imbalance of the aquatic biota due to diffuse pollution and sediment loads, which lead to degradation of the aquatic habitat. These changes degrade aquatic habitats while simultaneously inducing hydrological

extremes - including floods and droughts from diminished base flow - and modifying channel geometry through bank erosion, ultimately compromising water quality.

Thus, one can assert that the two main problems of water conservation, quantity and quality, culminate in environmental damage, hydrological disasters and loss of opportunities to use water, as illustrated in Figure 1-1, due to compromised availability in volume and/or adequate conditions (FERRAZ & LIMA, 2022). Furthermore, as urbanization increases, the sustainable development of water resources increasingly depends on the successful management of urban growth, especially in low-income countries, where the pace of urbanization tends to be faster (UNITED NATIONS, 2018).

Figure 1-1: Impacts of watershed urbanization on the water cycle.



Source: Adapted from Chocat *et al.*, 2007.

Due to the interaction between the hydrological cycle and the effects of urbanization, water resource management is a growing challenge, both from the point of view of water management and surface water degradation. Ineffective management compromises water security and triggers a series of negative environmental, social and economic impacts.

1.2 Scope of this Dissertation

This dissertation analyzes the application of Nature-Based Solutions (NbS) for water management in the Água Comprida urban watershed (Bauru/SP), with a focus on bioswales, wetlands and rain gardens.

1.2.1 General Objectives

The research project presented here aims to analyze three types of Nature-Based Solutions (NBS), namely: bioswales, wetlands, and rain gardens, in terms hydrological effectiveness within an urban watershed, analyzing their impact across macroscopic, sub-catchment, and unit-scale levels to support sustainable water resource management.

1.2.2 Specific Objectives

- Conduct a physical and environmental characterization of the urban sub-watershed of Água Comprida Stream, located in the city of Bauru/Brazil, determining the land use and occupation of the land and assessing its modifications over time (2004, 2010, 2013–2024).
- Describe the design features, socio-environmental benefits, ecosystem services, and physicochemical treatment processes of three types of Nature-Based Solutions (NBS), bioswales, wetlands, and rain gardens, for surface water treatment.
- Locate the Nature-Based Solutions according to the characteristics of the Água Comprida watershed and design aspects. Quantifying how many SbN can be applied to the watershed (bioswales, wetlands, and rain gardens).
- Extract information for hydrological modeling through a preprocessing routine of the contributing areas of each NBS. And create an open source and accessible data preprocessing routine.
- Simulate the performance of the NBS in the study area using hydrological modeling and evaluate the water balance in Água Comprida watershed and the individual response of the drainage areas.
- Apply a mixed-effects model to evaluate the unit-scale effectiveness of NBS, comparing performance differences across typologies (rain gardens, wetlands,

and bioswales) as well as implementation scenarios (Integrated, Rain Garden-only, Wetland-only, and Bioswale-only), verifying if intrinsic characteristics and spatial arrangements lead to statistically coherent results under varying operational conditions.

- Determine the overall impact of NBS on surface water management in the study area, providing technical evidence to support the integration of nature-based strategies into regional urban planning.

1.3 Organization of this Dissertation

The dissertation is organized into five chapters. Chapter 1 (Introduction) provides a general context of the impacts of urbanization on the hydrological cycle and aspects of water resource management, including the problem statement and an overview of Brazilian legislation regarding the use of sustainable techniques. It also presents the characterization of Nature-Based Solutions (NBS), detailing their structures, physicochemical water treatment processes, associated ecosystem benefits, and research objectives. Chapter 2 consists of a geo-environmental analysis, describing the characteristics and processes occurring in the study watershed. Chapter 3 addresses the routine developed for preprocessing information for hydrological modeling. Chapter 4 contains a description of the design criteria surveyed and the results of applying the routine in the study area. Chapter 5 covers the configuration of the hydrological model, including the adopted parameters, definition of simulated scenarios, and final results. Chapter 6 presents the final considerations of the work, highlighting the study's limitations and recommendations for future research. Chapter 7 refers to the author's academic background, including coursework, participation in research projects, accepted publications, conference presentations, and other relevant activities. Chapter 8 corresponds to the appendices, containing supplementary materials prepared by the author, while Chapter 9 compiles the annexes, featuring documents and the access of information used in the dissertation.

1.4 Water resources management

For the development of actions and measures, analyses must begin within the context of the territorial unit for water resources management: the river watershed, which represents the drainage area of a watercourse (LEAL, 1998). The adoption of river basins as the fundamental territorial units for water resources management is justified by their capacity to integrate spatial dynamics through hydrological connections, encompassing both natural processes and human interventions. Within the territory bounded by the topographic divides and the watershed outlet, water interacts in an interdependent, continuous, and dynamic process with both physical and biotic systems (DOUROJEANNI et al., 2002; MAGALHÃES JÚNIOR & LOPES, 2022).

Urban water management requires a holistic approach, integrating the components of the hydrological cycle; changes caused by land use and occupation; the quantitative and qualitative properties of surface water resources; hydrogeological and climatic conditions; institutional and organizational bodies; public policies; economic and private sectors; as well as social, economic, and environmental conservation aspects (SILVA et al., 2019). A frequently overlooked factor in management practices is the need for aquifer recharge and the existence of unconfined aquifers, which are hydraulically interconnected with perennial surface water bodies. The higher the capacity of these aquifers to transmit water, the greater the river-aquifer dependency. Neglecting the local hydrogeology compromises the effectiveness of policies and strategies for conserving and sustainably using water resources (OLIVEIRA, CARDOSO & NETO, 2007). Therefore, it is necessary to find a balance between the physical environment, natural resources, and the socioeconomic context within the limits of the watersheds, not only in land use management but also in the implementation of more adaptable and resilient urban infrastructures that strengthen the surface water–aquifer interface (MIGUEZ, VERÓL & REZENDE, 2016).

When the characteristics of water and watersheds are unknown or disregarded, water crises, financial losses, environmental damages, and tragedies arising from hydrogeological disasters occur (PIROLI, 2022). To mitigate such

damages and promote efficient water management, the concept of Water Security emerges, which is associated with the following indicators: sufficient supply of potable water in adequate quantity and quality to meet human needs, support productive activities and multiple uses; maintenance of minimum flow to preserve ecosystems and water bodies; and promotion of resilience to extreme hydrological events (UN-WATER, 2013).

In a global context marked by the urgent need to enhance socio-environmental resilience in the face of climate change, urbanization, and their effects on water resources, intervention measures are necessary. In Brazil, due to the demand for incorporating water resources management measures, the “Resilient Green Cities Program” is established by Decree No. 12041/2024, which aims to: I) enhance ecosystem services, improve green areas and water resources in an integrated manner with other systems; II) standardize parameters for sustainable and resilient urban-environmental planning and management; III) provide technical support for diagnostics, planning, governance, management, and projects focusing on greenhouse gas emission mitigation and climate change adaptation in urban areas; and IV) promote scientific research and the use of sustainable and nature-based solutions to increase urban resilience and improve water resources management (BRAZIL, 2024). In line with the “Resilient Green Cities Program,” Law No. 14904, dated June 27, 2024, which establishes public policies to develop climate change adaptation plans. Among the strategic actions, one of the priorities is establishing water security with a view to resilient socioeconomic development aligned with the reduction of social inequalities (BRAZIL, 2024).

Water resources management aims to ensure sustainable and equitable use of water by promoting conservation, protection, and restoration. In this context, sustainable management policies and practices align with strategies focused on environmental restoration, recovery, and rehabilitation. Nature-Based Solutions (NbS) are increasingly recognized as effective and economically viable approaches to addressing socio-environmental challenges and strengthening water security, contributing to resilience and sustainable management in urban areas (MASHIYI et al., 2023).

1.5 Nature-based Solutions

The term “Nature-based Solutions” broadly designates approaches based on ecosystems and natural processes to address contemporary challenges, optimizing sustainable and resilient management (COHEN-SHACHAM et al., 2016). Due to their diverse political origins and the different aspects of natural processes or functions, nine related terms can be found in the scientific literature, including: Low Impact Development (LIDs), Best Management Practices (BMPs), Water Sensitive Urban Design (WSUD), Sustainable Drainage Systems (SuDS), Green Infrastructure (GI), Blue-Green Infrastructure (BGI), Ecosystem-based Adaptation (EbA), and Ecosystem-based Disaster Risk Reduction (Eco-DRR). Although these terms share a core idea embedded in the overarching concept of NbS, their definitions differ based on historical perspectives, the contextual knowledge deemed relevant at the time of research, and their approaches to addressing social, economic, and environmental challenges (RUANGPAN et al., 2020).

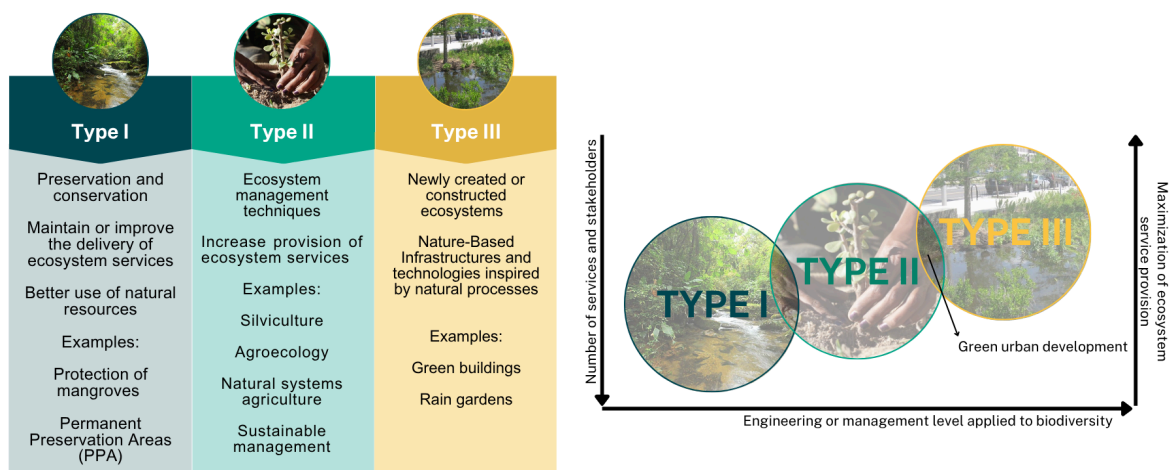
Terms such as SuDS, LIDs, BMPs, and WSUD are specifically related to stormwater management (FLETCHER et al., 2014). Similarly, GI and BGI prioritize technology-based infrastructure, incorporating natural alternatives especially aimed at urban planning and stormwater management (NESSHÖVER et al., 2017). In contrast, EbA focuses on analyzing long-term changes in the context of biodiversity conservation, ecosystem services, and climate change, while Eco-DRR is directed toward disaster risk reduction, using ecosystems as a mitigation mechanism (FAIVRE et al., 2018; UNEP, n.d.). As a broad term, the concept of NbS allows for a wide range of interpretations.

According to the United Nations Environment Programme (2022), the definition of NbS involves effective and adaptive actions to protect, conserve, restore, sustainably use, and manage ecosystems, addressing social, economic, and environmental challenges, while promoting human well-being, ecosystem services (ES), resilience, and benefits for biodiversity.

In the literature, NbS can be defined in various ways, depending on the approach and stakeholder perspectives. One framework, proposed by Eggermont (2015), classifies NbS into three complementary types, based on aspects such as:

the level of applied engineering or biodiversity management, the number of services to be delivered and stakeholders involved, and the degree of environmental transformation—i.e., the extent to which ecosystem service delivery is maximized. Type I refers to preservation or conservation, with minimal intervention in ecosystems, aimed at maintaining or enhancing the delivery of multiple services from these preserved ecosystems. Type II corresponds to the implementation of sustainable and multifunctional ecosystem and landscape management approaches, which include agricultural planning techniques (such as natural systems agriculture, agroecology, and forestry) with the goal of enhancing the delivery of ES compared to what would be achieved through more conventional interventions. Type III is based on newly created or constructed ecosystems through Nature-Based Infrastructure (NBI), including green and blue infrastructure and techniques that replicate natural processes without direct involvement of ecosystems, to manage environments in a more intrusive manner. The goals of Type III NbS include providing environmental benefits and restoring highly degraded or polluted areas. Figure 1-2 presents a summary of the NbS classification by Eggermont (2015).

Figure 1-2: Summary of Eggermont's NbS classification.



Source: Adapted from Eggermont et al., 2015.

According to the definition by the International Union for Conservation of Nature (IUCN, 2016), Nature-based Solutions specifically refer to direct actions involving the restoration or protection of natural or modified ecosystems. Unlike Eggermont (2015), this definition excludes technologies that do not directly interact with living ecosystems, such as biomimicry and artificial air purification systems. The

IUCN classifies NbS into five approaches: i) Restorative; ii) Goal-oriented; iii) Infrastructure-based; iv) Management-based and v) Protective. Restorative approaches include ecological restoration and forest landscape restoration, while goal-oriented approaches focus on specific objectives such as climate adaptation and disaster risk reduction. Infrastructure-based approaches involve using ecosystems – such as natural and green infrastructure – to perform traditional functions. Management-based approaches include integrated coastal zone management and water resource management. Protective approaches involve the conservation of areas, such as reserves and other land protection measures aimed at preservation.

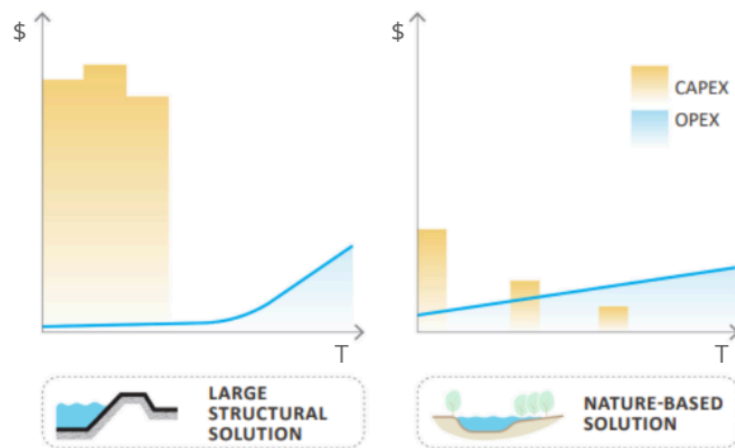
The concept of NbS defined by the European Commission (2016a) emphasizes that solutions must deliver environmental, social, and economic benefits simultaneously, while supporting the development of a resilient environment through systemic, adaptable, and resource-efficient interventions. The Commission's definition is primarily oriented toward urban NbS and does not impose restrictions on the elements included in the concept, but it incorporates cost-effectiveness as a key dimension of analysis.

In terms of cost-effectiveness analysis, although the cost components are similar to those of grey infrastructure projects, there are some important distinctions between the cost profiles of Nature-based Solutions and grey infrastructure alternatives that should be considered. In general, the capital expenditures (CAPEX) of NbS – which include design, planning, and construction costs – are lower compared to those of conventional infrastructures.

Cost assessment should also include operational expenditures (OPEX), which cover monitoring, maintenance, and operation costs. In the case of NbS, OPEX often requires considerable ongoing investment due to the need for regular maintenance of the techniques, potentially including the remediation of biological components of the system (WISHART, 2021; VAN ZANTEN et al., 2023). While grey infrastructure can be effective in achieving specific objectives, such as water treatment or retention, the construction and maintenance costs can be high, and over time, the invested value tends to depreciate (BRILL ET AL., 2021). Other cost components of NbS may also

include opportunity costs, related to alternative land uses; transaction costs, which encompass expenses for developing implementation strategies; and costs associated with negative externalities, such as mosquito breeding, pollen-related allergies, and others. Figure 1-3 shows the CAPEX and OPEX trends of structural solutions and nature-based solutions over time.

Figure 1-3: Comparison of the capital and operating costs of structural solutions and NbS over time. CAPEX: Capital Expenditures, OPEX: Operational Expenditures.



Source: Van Zanten et al., 2023.

Comparative studies between green and grey (conventional or built) infrastructure conducted by the International Institute for Sustainable Development (IISD) have shown favorable results for the use of NbS. The assessment involved applying the Sustainable Asset Valuation (SAVi) methodology – developed by IISD to quantify the full costs of risks and externalities over the life cycle of projects – in various case studies. In most cases, green infrastructure showed costs approximately 50% lower than those of built infrastructure alternatives, while delivering similar or even superior results.

The cost savings are primarily due to lower capital investments, especially in contexts where natural ecosystems are already present and do not require intensive restoration and reconstruction processes (BASSI et al., 2021). In addition to their economic advantages, NbS also provides ecological, social, cultural, and economic benefits (TSATSOU, FRANTZESKAKI & MALAMIS, 2023). These benefits are known as ecosystem services and contribute to ecosystem preservation and the resilience

of urban areas in the face of extreme weather events and gradual changes to which cities are exposed (JENNINGS & BAMKOLE, 2019).

Examples of ecosystem services typically provided by NbS include, beyond the control of water quality and quantity: carbon sequestration; climate regulation and prevention of heat islands; air purification; pollination support; preservation of native flora and fauna; biodiversity enhancement; promotion of public health; provision of food and supplies; recreational, spiritual, and religious functions; aesthetic enhancement of environments; tourism stimulation; educational and cultural purposes; soil formation; oxygen production; encouragement of environmental management; nutrient cycling; and primary production, among others (MANCUSO et al., 2021).

The benefits associated with ecosystem services add 28% in value, stemming from job creation for construction and maintenance, the opportunity to stimulate growth in other economic sectors, and the promotion of climate resilience, which actively prevents losses and costs associated with the impacts of extreme weather events such as floods and heatwaves (BASSI et al., 2021). Due to their ability to provide multiple benefits, the positive effects of Nature-based Solutions (NbS) tend to accumulate over time, with the potential to be up to five times more cost-effective than conventional engineering solutions. Thus, investments in NbS can increase in value as more ecosystem services are delivered (MATSLER, 2019; NARAYAN et al., 2016).

Being scalable, NbS projects also offer opportunities to improve and adapt the approach as conditions change and more information becomes available, which is particularly advantageous in data-scarce environments (WISHART, 2021). Compared to conventional adaptation measures – which are often associated with high costs, inflexibility, and conflicts of interest – NbS are potentially more resilient, cost-effective, and advantageous in terms of the benefits linked to the techniques employed (KABISCH et al., 2016; TEMMERMAN et al., 2013; BRINK et al., 2016).

Regarding performance in water management specifically, evidence indicates that Nature-based Solutions represent an opportunity to optimize this management by contributing to improvements in the spatial, temporal, and quantitative distribution

of available water. By reducing surface runoff and performing treatment functions, NbS have the potential to simultaneously address issues related to water quality, quantity, and hydrogeological risk management (WWDR, 2018). Thus, NbS are systems that, when implemented in urban areas, can present a favorable cost-benefit ratio due to their water management capabilities and the ecosystem services they provide, which restore balance in the built environment and contribute to the socioeconomic and environmental well-being of cities, as well as enhancing environmental resilience.

Examples of Nature-based Solutions include linear parks, green roofs, infiltration trenches, porous pavements, bioswales, rain gardens, wetlands, and bioswales. Among these, rain gardens, wetlands, and bioswales will be evaluated in this study, focusing on the analysis of their urban water management capacity for runoff control, groundwater recharge, and pollutant load reduction, as well as information related to their implementation, operation, and maintenance.

The implementation of Nature-based Solutions (NbS) aligns with several of the 17 Sustainable Development Goals (SDGs) established by the United Nations (UN) to eradicate global challenges. Among the goals related to NbS, two are associated with water resources (6 – Clean Water and Sanitation; 14 – Life Below Water) and another is connected to the development of inclusive, safe, resilient, and sustainable cities and communities (11 – Sustainable Cities and Communities).

Thus, carrying out projects involving NbS is highly relevant, especially for municipalities that need to improve water management, in terms of mitigating erosive processes, reducing water contamination, and minimizing hydrological risks, such as floods and droughts, such as the study area in Bauru, located in the state of São Paulo, in the southeastern region of Brazil.

1.5.1 Rain gardens

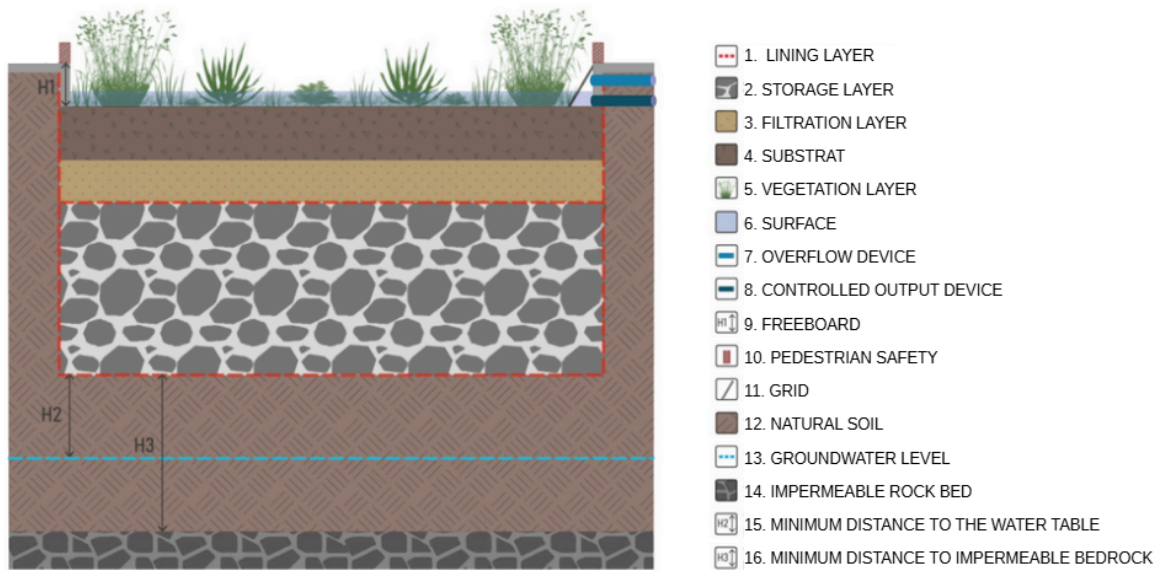
Rain gardens are shallow depressions designed for the bioretention of stormwater, intended for the collection, storage, reduction of runoff volume, and treatment of surface runoff. These structures primarily aim to retain and infiltrate

rainwater (MELO, 2011). Additionally, rain gardens have the capacity to treat water through physical, chemical, and biological processes.

These systems are considered a stormwater and land use management strategy, aiming to reproduce natural hydrological conditions using low-cost techniques with simplified maintenance (LI & ZHAO, 2008). In this way, they promote infiltration, groundwater recharge, base flow restoration, the reduction of erosive processes and hydrogeological disasters, and improvement in water quality through the retention, removal, and reduction of contaminants transported by surface runoff. Consequently, they contribute to the protection of receiving water bodies in urban areas by mitigating the impacts caused by soil impermeabilization in cities (SOLUÇÕES PARA CIDADES, 2021). Overall, rain gardens, as a source control and impact prevention alternative, contribute to the urban-environmental balance of cities (MUTHANNA et al., 2008; LI & ZHAO, 2008).

Typically, the structure of rain gardens includes layers of porous soils and granular materials, native vegetation cover, and some hyperaccumulator plants capable of performing phytoremediation (UACDC, 2010). The rain garden can be built directly over the natural soil, taking advantage of its drainage and infiltration characteristics. However, if there is a risk of soil erosion or if the site is subject to heavy vehicle or pedestrian traffic, it may be necessary to build impermeable side walls. In some cases, structures such as footings may be required to ensure stability and prevent soil displacement. Figure 1-4 illustrates the layers that make up the structure of a rain garden.

Figure 1-4: Typical configuration of a rain garden.



Source: Adapted from Prefeitura Municipal de Belo Horizonte, 2022.

The typical vertically arranged layers of rain gardens are the surface/vegetation layer (represented in Figure 1-4 by numbers 5 and 6), the substrate (4), the filtration layer (3), the lining layer (1), the storage layer (2), and the natural soil (12). The surface layer contains the vegetation, usually grasses, ground covers, or shrubs, typically native species due to their greater adaptation to local climatic conditions, requiring little or no irrigation or fertilization (ISHIMATSU et al., 2017). The layer beneath the surface is called substrate, which provides support for the vegetation cover and supplies the necessary nutrients for plant development. Next is a layer composed mainly of washed coarse sand, designed to promote infiltration and the redistribution of water in the soil, referred to as the filtration layer. The subsequent lining layer should consist of a geotextile fabric intended to retain fine particles carried during the infiltration process. Following this is the storage layer, formed by crushed stone, gravel, or porous materials, which may be followed by another lining layer and contain perforated drainage pipes connected to an overflow device. Finally, there is the natural soil layer, where the infiltrated water can be used for groundwater recharge, storage, or both in the case of a combined system (DUNNETT & CLAYDEN, 2007).

The structures of rain gardens include an overflow control system, known as an overflow device (7) (URBAN GREENUP, 2018). According to Saatkamp (2019), the layers can be adjusted according to storage needs, and determining these layers requires collecting information such as rainfall intensity, effective precipitation, runoff volume, infiltration volume, and storage volume. With these data, the dimensions of the layers can be calculated. In addition, it is common for rain gardens to have drainage pipes connected to the stormwater system, allowing the discharge of peak flows when the capacity of the porous soil is exceeded. In such cases, rainwater begins to accumulate on the surface of the planting soil, preventing system overload. This accumulated water can be drained by infiltration into the subsoil (infiltration design), through a bottom drain (filter design), or through a combination of both methods (XU et al., 2022).

Xu et al. (2022) evaluated the effect of using subsurface drainage in rain gardens on runoff flow and pollutant retention by comparing two structures operating under similar conditions, with and without subsurface drainage, monitored during 28 storm events. The results indicated that the structure with a drainage pipe, referred to as a drainage rain garden, is capable of providing rapid drainage and sustaining runoff discharge in rain gardens with moderately permeable soils. However, the pollutant removal processes were compromised because the short flow path within the rain garden reduced pollutant removal rates.

The structure without drainage, known as an infiltration rain garden, promotes the retention of runoff and pollutants through the infiltration process, enabling more efficient pollutant removal compared to merely retaining the flow. This is especially true when the initial stormwater runoff is captured in an empty or dry infiltration garden. The structure functions optimally during small rainfall events and particularly in dry periods, when surface runoff carries diffuse loads. During periods of torrential rain, when the absorption capacity for which the structure was designed may be exceeded, the excess flow is diverted from the area and directed straight to the conventional drainage system (MELO et al., 2014; WOODS-BALLARD, 2007).

General recommendations or criteria for the location and sizing of rain gardens are necessary as a starting point, but the design must consider local

variables such as soil type and its infiltration capacity, rainfall intensity, return period, and design storm duration, surface runoff coefficient, contributing area, and overflow discharge (WORLD BANK, 2021). These factors are essential for determining the storage and drainage capacity. In addition, the priority nutrients for removal can also influence the characteristics needed to optimize treatment capacity (SHARMA & MALAVIYA, 2021; RUANGPAN et al., 2020).

In a rain garden, two aspects are important regarding its hydrological performance: controlling the quantity of runoff water and improving the quality of the retained water through the removal of pollutants and the association of nutrients carried by surface runoff, as well as promoting infiltration and groundwater recharge provided by this system (DUNNETT & CLAYDEN, 2007; DIETZ & CLAUSEN, 2005; ARAVENA & DUSSAILLANT, 2009). Its retention function, usually designed in terms of capture, is essential for retaining, delaying, and minimizing or preventing the impacts caused by surface runoff. At this stage, water is held on the surface and subsequently infiltrates or evaporates. Alongside the capture and infiltration processes, the removal and retention of contaminants carried by surface runoff occurs, such as chemicals, oils, greases, debris, soil particles, bacteria, and others (DAVIS et al., 2009; TROWSDALE & SIMCOCK, 2011).

In general, contaminant removal in rain gardens is achieved through sorption (adsorption and absorption), filtration, volatilization, sedimentation, ion exchange, and degradation, which can be abiotic (such as photochemical degradation) or occur through the biological activities of plants and microorganisms (PRINCE GEORGE'S COUNTY, 2007; AL-AMERI et al., 2018). Thus, rain gardens are capable of removing and retaining sediments, heavy metals, pathogens, nutrients, hydrocarbons, and other contaminants in both suspended and dissolved forms (SHARMA & MALAVIYA, 2021). As stormwater enters the rain garden, finer solids are trapped in the surface layers, and suspended solids are retained at the bottom of the media (the deeper part of the filtration layer). These trapped particulate pollutants are converted into usually harmless byproducts through chemical processes or other plant-based mechanisms (LI & DAVIS, 2009).

The efficiency of removal and retention mechanisms is affected by seasonal variations, garden specifications, and stormwater characteristics (HUNT et al., 2008). In the case of metals, substrate composition, climatic conditions, and retention time are factors that influence the removal rate. Low saturated hydraulic conductivity values in sand-based rain gardens reduce water infiltration between rain events but increase the efficiency of metal removal, especially copper. Metal removal is also favored by warmer temperatures due to higher plant efficiency (BLECKEN et al., 2009; KHAN et al., 2012; SHARMA & MALAVIYA, 2021).

The forms of phosphorus removal are influenced by climate, the presence or absence of O_2 , variations in medium composition, inflow concentration, and phosphorus forms. When phosphorus inflow concentrations are high, phosphorus is not adsorbed, and organic matter is converted into bacterial orthophosphate. As phosphorus accumulates in the adsorbents and eventual saturation occurs, the adsorption capacity decreases, and the filter media material is exhausted. On the other hand, removal is favored by media with high concentrations of metals such as Fe, Ca, and Al, and by the presence of higher temperatures, which increase reaction efficiency. Low saturated permeability coefficient and high water retention time promote complexation and reduce phosphorus export. Rain gardens with an anoxic layer can increase total P removal by favoring the precipitation of phosphorus as insoluble compounds (metal phosphates), enhancing phosphorus adsorption, and creating conditions more favorable for biogeochemical processes that remove phosphorus from solution (HSIEH et al., 2007a; DENICH, BRADFORD, & DRAKE, 2013; ZHANG et al., 2008).

The basic structure of a rain garden is not very successful in removing nitrate, as nitrates are not retained or adsorbed by most soil types. The removal processes occurring in rain garden substrates convert nitrates into nitrogen gas and release it into the atmosphere. However, a rain garden containing soil with carbon additives, ideal pH, and moisture conditions in deeper soil layers – due to microbial requirements for moist conditions to drive the reaction – can increase nitrate removal to a significant level. Extending the contact time through the use of less permeable soils can also improve NO_3-N removal efficiency (STANDER & BORST, 2008).

Rain gardens, in addition to performing essential functions in stormwater management and treatment, provide socio-environmental and ecosystem benefits that contribute to improving quality of life in urban areas. The planting of native species in these systems encourages the attraction of pollinators and pest-controlling animals, directly contributing to increased biodiversity and habitat conservation in urban environments (KIM & SONG, 2019). Bioretention areas also help regulate heat by reducing surface and air temperatures through plant evapotranspiration. Reducing the urban heat island effect is particularly important in urban settings, where high temperatures can affect population well-being. Another significant environmental benefit associated with rain gardens is carbon storage and sequestration, which depends on design criteria, materials, and plant species selected. Carbon sequestration contributes to climate change mitigation and promotes environmental sustainability in urban areas (EUROPEAN COMMISSION, 2016).

The ability to reduce flood risk and promote water security is also a characteristic of these systems. By slowing down and infiltrating stormwater, they help prevent the worsening of floods and inundation in downstream watershed areas. Moreover, the flexible design of rain gardens allows for a significant reduction in the size and cost of downstream drainage infrastructure, making these systems more efficient and cost-effective (OICS/CGE, 2024). From an economic perspective, rain gardens can play a significant role in stimulating local economies and creating jobs. The implementation of these systems can increase nearby property values and generate opportunities for economic development, such as the creation of green jobs (KIM & SONG, 2019).

In addition to environmental and economic benefits, rain gardens can provide educational and recreational contributions. As they offer opportunities for interaction with nature, these systems can be used to engage the public in their planning and maintenance, raising awareness of environmental issues and strengthening community cohesion (KASPRZYK et al., 2022). Finally, the implementation of rain gardens also brings social benefits, improving quality of life and aesthetic comfort in urban areas. Spaces with this structure can serve as areas for recreation and social interaction, creating opportunities for informal gatherings. Social interaction is strengthened through community involvement in the planning and maintenance of

these spaces, which can also improve public safety and promote the spontaneous use of urban areas.

Rain gardens provide provisioning services by contributing to water and vegetation availability in urban areas. In terms of supporting services, they promote biodiversity by attracting pollinators and natural pest controllers. As for regulating services, they assist with water infiltration, reducing floods, mitigating heat islands, and promoting carbon sequestration, thereby contributing to climate change mitigation. Furthermore, they offer cultural services by providing spaces for environmental education, leisure, and social interaction, strengthening community ties and improving quality of life in cities (BRAZIL, 2021). These benefits make rain gardens an efficient and multifunctional solution for cities, contributing to environmental improvement and quality of life for urban populations.

1.5.2 Constructed Wetlands

Constructed wetlands (CW), also referred to in the literature by various terms such as root zones, macrophyte-planted filters, cultivated beds or filters, artificial marshes, and macrophyte biofilters, among others, are systems based on natural wetland areas such as swamps, mangroves, floodplains, and marshes (SEZERINO et al., 2015). They consist of permanently saturated areas designed to replicate and enhance the contaminant assimilation and conversion processes of natural ecosystems for the treatment of different types of contaminated water.

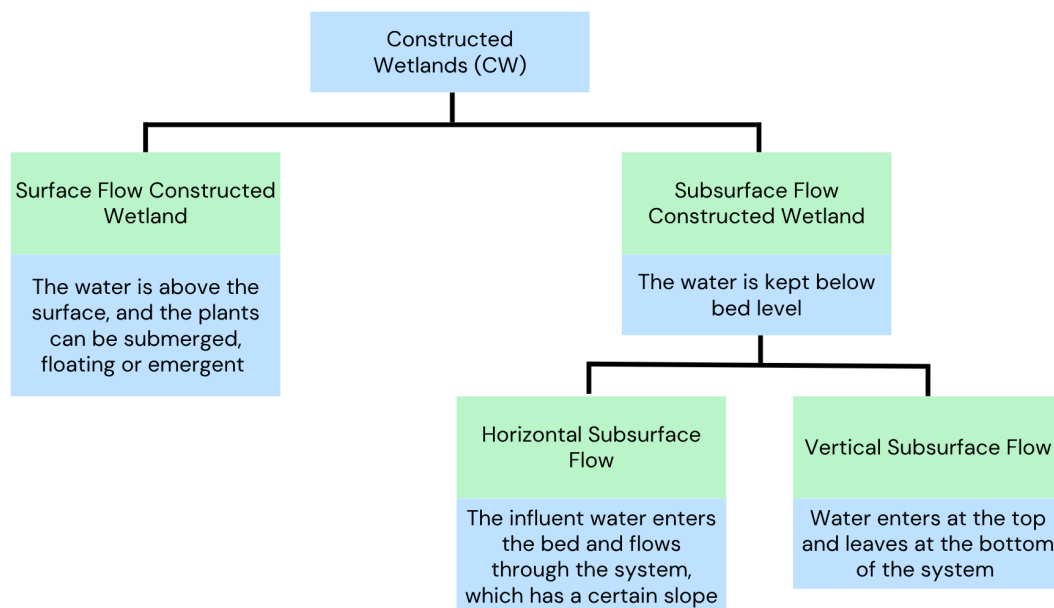
Due to their flexibility, CW systems are capable of efficiently treating various types of influents, including landfill leachate, sludge, raw, primary, secondary, and tertiary sewage, industrial effluents, stormwater, and water bodies under recovery (KENNEN & KIRKWOOD, 2015; DOTRO et al., 2017; SEZERINO et al., 2018). The main functions of constructed wetlands include improving water quality and controlling floods through the storage of stormwater and surface runoff (CT DEEP, 2004; HASSAN, 2021).

Although they play an important role in runoff control, stormwater wetlands generally have low infiltration rates due to high groundwater levels or subsoils that are compacted during construction to prevent the wetland from drying out. Therefore, CWs are not classified as low impact development (LID) practices, which emphasize

the use of infiltration-based stormwater practices, such as rain gardens and bioswales. Still, depending on the site and climatic conditions, stormwater wetlands can considerably reduce runoff volume at a location (MOORE & HUNT, 2001).

In general, constructed wetland designs vary according to aspects such as the type of effluent, design criteria, the types of macrophytes used, and the flow direction of the wastewater within the system, all of which directly influence the system's efficiency (VYMAZAL, 2011). Regarding hydraulic flow type, constructed wetlands can be classified as surface flow wetlands, where a free water layer is present, and subsurface flow wetlands, where the influent percolates through the filtering medium without being visible at the surface. Additionally, surface flow constructed wetlands (FWS CW) can be classified according to the types of macrophytes, and subsurface flow constructed wetlands (SSF CW) can be classified according to water flow direction into horizontal and vertical subsurface flow CW, as shown in Figure 1-5.

Figure 1-5: Constructed wetlands types and description.



Source: Adapted from Kadlec & Wallace, 2008.

Given the different flow categories of wetland systems, this section focuses on free water surface constructed wetlands (FWS CWs), as they are the primary choice for infiltration and treatment of surface runoff (KADLEC & WALLACE, 2009). Free water surface constructed wetlands consist of a shallow, open flooded area with a

visible water layer, wastewater flow, and a high density of floating, submerged, and/or emergent macrophytes (KADLEC & WALLACE, 2008). The structure of FWS CWs requires a larger area and handles lower pollutant loads compared to subsurface flow constructed wetlands. They are widely used for urban stormwater management due to their low cost per unit area, their capacity to receive variable flow rates, and their ability to accommodate changing water levels (DOTRO et al., 2017; RUSSELL, 2021).

FWS CWs show great diversity in their structural characteristics. The lining may or may not be present, the depth may remain constant or vary throughout the structure, and vegetation cover may be partial or complete. The size of the wetlands can also vary significantly, from small areas of just a few square meters to large expanses covering several square kilometers. Additionally, different types of vegetation can develop, including emergent, submerged, or floating species. Floating macrophyte systems are characterized by plants whose aerial parts remain above the water surface, while their roots extend throughout the water column without anchorage to any solid substrate. One of the main concerns in this type of system is the potential proliferation of mosquitoes, as the plants may provide shelter for eggs and larvae. Emergent macrophyte systems, on the other hand, make use of plants adapted to wet conditions, with roots anchored in the substrate and stems and leaves that are partially submerged. Lastly, submerged macrophyte systems consist of vegetation with all vegetative parts remaining underwater. This configuration can be further divided into two subtypes based on plant mobility: fixed vegetation, which is rooted to the substrate or a support medium at the bottom of the wetland; and free-floating vegetation, which is not anchored and drifts freely within the water column (ZANELLA, 2008).

Although designs may vary, FWS CWs (Free Water Surface Constructed Wetlands) are typically composed of three main zones, each with a specific role in the removal of sediments and pollutants. The first is the inlet zone, which functions as a sedimentation basin for large and medium-sized particles. There are also deep pools, which are designed to retain water throughout the dry season in order to maintain water within the wetland system, prevent stagnation, and enhance denitrification. The deep pools may be strategically located near the wetland inlet,

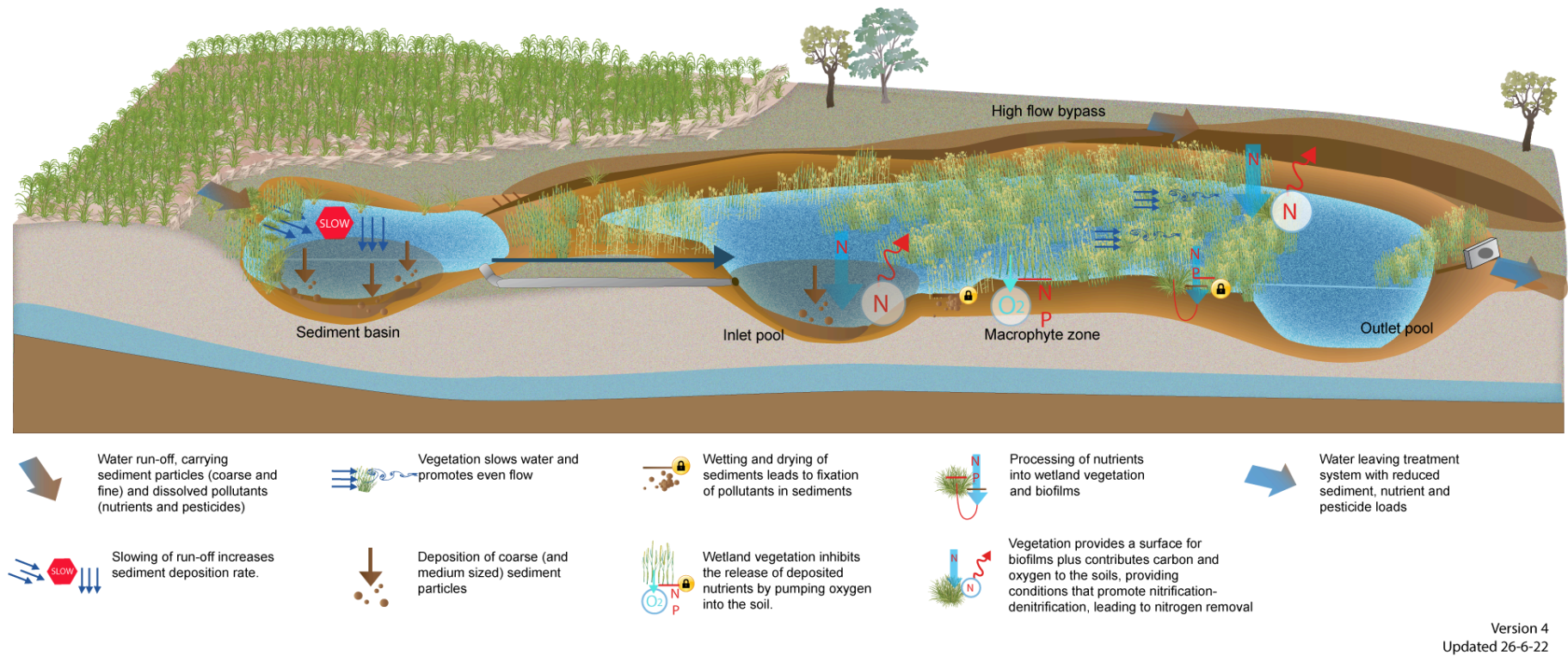
outlet, or both. The macrophyte zone, which remains permanently inundated, comprises a shallow area with vegetation for the removal of fine particulate matter and soluble pollutants.

To ensure system efficiency, it is essential that the deep pools and open water areas are lined with either natural or synthetic liners to control excessive infiltration into the soil. The macrophyte zone, in turn, may not require lining, depending on the groundwater table level. In addition, topsoil should be added over the liner in the areas where it is required, creating a suitable environment for plant growth, as plants play a fundamental role in pollutant removal (NCDER, 2009).

Constructed wetlands also include high-flow bypass channels, which divert excess water around the macrophyte zone to prevent erosion, damage to the vegetation, or pollutant resuspension. The structure also consists of an outlet, which regulates water levels and flow to achieve the required detention time, and transitional areas between the deep and shallow parts of the wetland, known as semi-wet zones (GEORGIA ENVIRONMENTAL PROTECTION DIVISION, 2011). The outlet may be designed with a raised elevation to allow temporary water storage that is gradually reduced to the normal level (GIZ, 2023; DEPARTMENT OF ENVIRONMENT, SCIENCE AND INNOVATION OF QUEENSLAND, 2022).

The construction of FWS CWs should be integrated into urban drainage infrastructure to intercept runoff and promote its treatment before discharge into receiving water bodies, contributing to the improvement of surface water quality (GIZ, 2023). Figure 1-6 illustrates a representation of an FWS CW, showing its structures and the processes involved.

Figure 1-6: Typical configuration of a Free Water Surface Constructed Wetlands.



Source: Department of Environment, Science and Innovation of Queensland, 2022.

The operation of constructed wetlands (CWs) is designed to replicate the physical, chemical, and biological processes that occur in natural wetlands. In free water surface flow systems, the main treatment processes include sedimentation, filtration, reduction, adsorption, precipitation, and volatilization (Kadlec and Wallace, 2008). FWS CWs typically exhibit high efficiency in reducing biochemical and chemical oxygen demand, as well as in removing suspended solids through filtration and sedimentation processes, traces of metals, and some complex organics (Clary et al., 2020). When detention time is adequate or in large wetland areas, these systems can be more efficient at removing nitrogen and phosphorus (EPA, 2020).

Regarding suspended solids, larger and denser particles tend to settle mainly in the inlet zone. Smaller and less dense particles can remain suspended for longer periods and are retained only after entering the vegetated area of the system. The presence of vegetation favors sedimentation by reducing water column movement and minimizing resuspension of accumulated sediments (VYMAZAL, 2008).

The efficiency of nitrogen removal varies depending on factors such as influent concentration, chemical form of nitrogen, water temperature, season, availability of organic carbon, dissolved oxygen concentration, vegetation type, and presence of microorganisms (KADLEC & WALLACE, 2008; BALDERAS-GUZMÁN, 2013). Nitrogen removal occurs through nutrient uptake by macrophytes, oxygen transport to the rhizosphere, and microbial activity. Ammonia nitrogen is removed via nitrification in aerobic zones of the wetlands and denitrification in anaerobic zones, such as the wetland bottom where plant matter accumulates.

Phosphorus removal in FWS CW systems occurs through various mechanisms, including adsorption, absorption, complexation, and precipitation with Al, Fe, and Ca ions. However, precipitation is relatively slow and limited due to reduced interaction between the water column and soil, which restricts the availability of these ions for reaction (KADLEC & KNIGHT, 1996; VYMAZAL et al., 1998; VYMAZAL, 2011). Substantial amounts of phosphorus can be stored in the organic substrate of FWS CWs, and plant assimilation may be significant under conditions of low influent loading rates (REDDY and DEBUSK, 1985).

Heavy metal removal processes in wetlands may involve several mechanisms such as macrophyte absorption and precipitation, with sedimentation being the primary removal process. In more productive wetlands, macrophytes may also play a relevant role in this process (LUNG & LIGHT, 1996; WONG et al., 2000).

The efficiency of CW treatment processes is related to several factors, including the hydraulic conductivity of the supporting substrate, the type and quantity of microorganisms present, oxygen supply to these microorganisms, chemical characteristics of the substrate, and the climatic and geographic conditions of the region (SAEED & SUN, 2012; ZHANG et al., 2015). In the case of FWS CWs, solar radiation can accelerate degradation rates, making these systems especially effective in warmer climates with high solar incidence, which favors plant growth (KYAMBADDE et al., 2004). Another important factor is the type of vegetation used, which can increase the efficiency of removing certain contaminants.

The roots of floating macrophytes provide support for microorganisms and assist in the adsorption and filtration of solids present in the water, as well as reducing sunlight penetration and gas exchange, limiting algae proliferation and creating near-anaerobic conditions in the water column (IWA, 2000). In the case of emergent macrophytes, the larger water surface area in contact with the atmosphere provides greater oxygenation, increased biofilm biomass, and higher uptake of soluble pollutants. Oxygen transfer enables the development of aerobic microorganisms that enhance the decomposition of organic substances. Moreover, oxygen presence in sediments also facilitates the immobilization of some pollutants, such as heavy metals, which can bind to sediment particles in an oxygenated environment, making them less soluble and more easily captured and retained (TOWNSVILLE CITY COUNCIL, 2010).

Submerged macrophytes are effective at ammonia removal due to their high photosynthetic rate, which transfers significant amounts of oxygen to the water, facilitating nitrification. Simultaneously, with the consumption of CO₂ during photosynthesis, pH levels rise, and ammonia is converted to its volatile, non-ionized form, facilitating its removal to the atmosphere. However, submerged macrophytes

depend on sunlight penetration, which can be limited by turbidity or shading, making this configuration more sensitive compared to others (IWA, 2000).

Constructed wetlands have great potential to provide various ecosystem services, significantly contributing to stormwater management and environmental benefits in urban areas. CWs have an ecology similar to natural wetlands and can, in some cases, achieve biodiversity levels and population sizes comparable to natural ecosystems (BALDERAS-GUZMÁN, 2013). Besides playing a vital role in protecting aquatic environments, wetlands add significant aesthetic value to the landscape. By transforming urban areas into landscapes integrating water bodies, floating, emergent and submerged vegetation, and rich associated biodiversity, constructed wetlands promote the integration of urban infrastructure with natural ecosystems. These urban natural ecosystems are vital for city sustainability and public health, contributing to improved air quality, minimizing urban heat island effects, and efficient stormwater management (BOLUND & HUNHAMMAR, 1999).

Regulating services provided by CWs include improving water quality, retention and storage of stormwater, flood and erosion control, local climate regulation, increased groundwater recharge, and reduction of urban heat island effects. Wetlands also enhance air quality through filtration and humidity, and in carbon sequestration. Vegetation in these areas removes CO₂ from the atmosphere and stores it both in plant tissues and soil. Additionally, the typical saturated soil conditions create an anaerobic environment that favors slow decomposition of organic matter, promoting carbon accumulation in the soil.

Provisioning services provided by CWs include supplying raw materials from pruning and harvesting for composting and ornamental uses. Cultural services include opportunities for wildlife observation, studies of physical-chemical and biological processes, valuable resources for educational programs, and providing aesthetic and well-being benefits to urban residents. Regarding supporting services, wetlands contribute to nutrient cycling and provide habitat services, offering shelter for a wide diversity of plant and animal species such as fish, birds, amphibians, and aquatic invertebrates.

Despite their great potential, stormwater wetlands can accumulate contaminants from the urban landscape, raising concerns about the habitat value they provide for wildlife. Sparling et al. (2004) emphasize that the habitat benefits offered by wetlands likely outweigh the negative impacts of contaminant accumulation on wildlife. Therefore, stormwater wetlands not only play a vital role in urban water management but also provide a wide range of ecosystem services that benefit both the environment and urban populations. However, it is essential to monitor and explore the long-term effects of contaminant exposure to ensure these ecosystems continue to sustainably provide their valuable services.

1.5.3 Bioswales

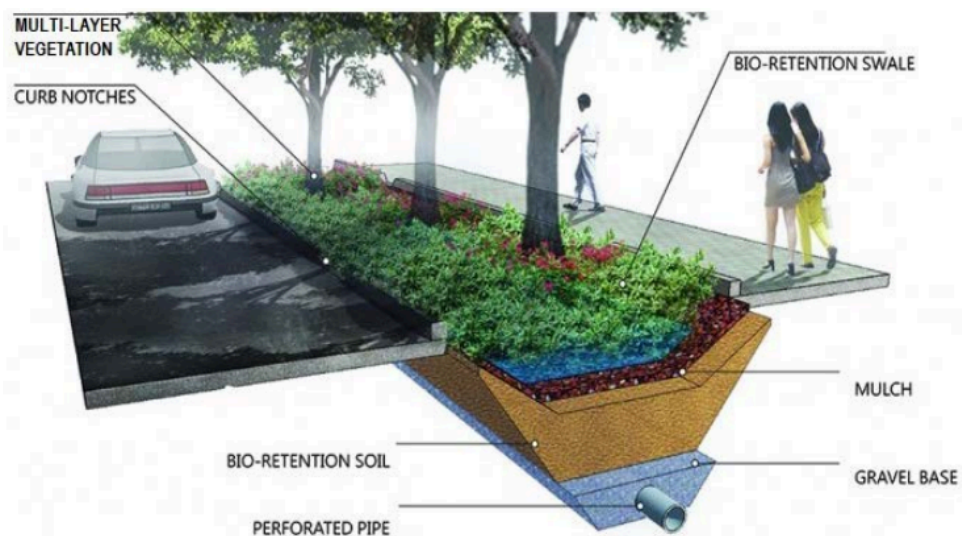
Bioswales, also called vegetated swales or biochannels, are bioretention systems designed for stormwater management. The system consists of shallow linear depressions with raised sides, forming a vegetated trapezoidal channel open for water flow, functioning as a conveyance system (PENNSYLVANIA DEPARTMENT OF ENVIRONMENTAL PROTECTION, 2006). The structure's purpose is to extend the function of conveyance systems by integrating attributes that reduce the velocity of surface runoff, promote infiltration with consequent groundwater recharge, improve water quality, and direct excess water to an area where it can be absorbed or managed sustainably (UNIVERSITY OF FLORIDA, 2008). Bioswales can be connected in series or integrated with existing drainage elements and also with ecological drainage devices such as rain gardens. Although rain gardens and bioswales are both considered bioretention systems, rain gardens are more effective at handling small, uniform stormwater discharges, while bioswales are more effective in scenarios where rainfall peaks occur early (ISHIMATSU et al., 2017; LU et al., 2024). Regarding treatment, bioswales typically exhibit lower efficiency compared to bioretention cells such as rain gardens due to shorter retention and storage times (EKKA et al., 2021).

Generally, bioswales employ a combination of grasses and other plants to stabilize soil, reduce erosion, slow flow, absorb runoff, and retain contaminants (KONDRATENKO et al., 2024). Thus, by improving water quality through pollutant

retention and removal, bioswales show great potential in sustainable stormwater management and maintaining hydrological conditions in urban areas.

Bioswales are designed with gently sloped sides and a wide, flat bottom, usually composed of layers of filtering media, soil, and vegetation. The top layer of the bioswale consists of amended soil, typically composed of local soil, sand, and organic compost (usually in proportions of 60%, 30%, and 10%, respectively), covered by vegetation composed of grasses and other plants (SÃO PAULO, 2024). The layer beneath the surface is composed of aggregates such as gravel, slag, or expanded clay, contained within geotextile to prevent clogging and allow water infiltration. At the interface between the two layers, there may also be a layer of sand. Figure 1-7 illustrates a representation of the structure of a bioswale.

Figure 1-7: Typical configuration of a bioswales.



Source: Echols & Pennypacker (2015).

Some bioswale structures contain an underground drainage pipe at the bottom of the aggregate layer, which may not be necessary in the case of naturally well-drained soils (UNIVERSITY OF FLORIDA, 2008). Also, to prevent overflow along the edges of bioswales, overflow devices connected directly to the infiltration pipe are incorporated. Bioswales should be sized so that the use of overflow devices is necessary only in exceptional situations (URBAN GREEN BLUE GRIDS, n.d.).

Surface runoff from adjacent areas is diffusely directed to the bioswales through curb cuts and flows along the swale section. As water flows into the bioswales, vegetation slows the water down, allowing heavy pollutants such as sediments and debris to settle. Various plant species present in bioswales absorb nutrients like nitrogen and phosphorus, preventing these contaminants from reaching downstream water bodies. Additionally, the soil within bioswales harbors microorganisms that play a crucial role in contaminant decomposition (THE WATERSHED PROJECT, n.d.).

The main pollutant removal mechanisms in bioswales include filtration; particle settling and sedimentation; infiltration and chemical precipitation; microbial degradation; and uptake by vegetation (VASCONCELLOS, 2011; GODYŃ et al., 2024). Rainfall characteristics, preceding dry periods, influent load, and watershed imperviousness significantly influence stormwater treatment (GONG et al., 2019). Studies show that swales are more effective at pollutant removal when influent concentrations are high and pollutants are bound to coarser particles that can be removed by filtration and sedimentation. However, they may act as pollutant sources when influent concentrations are low due to the resuspension of materials previously deposited on the swale bed (EKKA et al., 2021).

Regarding sediment removal, bioswales are more effective at removing larger, medium, and coarse sediment particle sizes, as well as sediment-bound pollutants (UNIVERSITY OF FLORIDA, 2008). However, retention of smaller particles is limited in bioswales due to shorter hydraulic retention times, which depend on channel length and flow velocity (BACKSTROM et al., 2006; DELETIC AND FLETCHER, 2006).

Concerning nitrogen removal, the efficiency of vegetated swales varies widely in the available studies. Some research highlights significant reductions in both total and dissolved nitrogen forms such as ammonia and nitrate, while others indicate that these structures can release nitrogen due to the decomposition of accumulated organic matter (GODYŃ et al., 2024; CLARY et al., 2020; LUCKE, 2014). For effective nitrogen treatment, the presence of anaerobic conditions that favor nitrification and denitrification processes is essential. Structural and compositional

modifications, such as implementing internal water storage zones and using vegetation cover and organic matter with a high carbon-to-nitrogen ratio, can optimize nitrogen removal (UNIVERSITY OF FLORIDA, 2008).

Research on phosphorus (P) removal in bioswales shows mixed results, with evidence of both reduction and net export of phosphorus. The sediment-associated fraction of phosphorus is generally captured by filtration and sedimentation processes in swales, whereas dissolved phosphorus is not removed. The use of engineered media with low phosphorus content and high chemical adsorption capacity can reduce dissolved phosphorus concentrations. Furthermore, vegetated swales should be designed with inlet protection, dense vegetation, and retention structures such as check dams to reduce particle resuspension. It is important to avoid the use of virgin manure or chemical fertilizers, and planting media should be tested for phosphorus content if relevant to the project (EKKA et al., 2021; UNIVERSITY OF FLORIDA, 2008).

Bioswales are effective at treating heavy metals in runoff, with metals such as zinc, copper, nickel, and cadmium typically present in dissolved forms, while lead and chromium are more frequently particle-bound. Removal of these metals in bioswales occurs through filtration, sedimentation, and adsorption, with variable efficiency. However, bioswales do not show capacity to reduce fecal indicator bacteria concentrations in their effluents. In some cases, increased concentrations of fecal coliforms have been observed, suggesting these systems may export fecal indicator bacteria either by scouring previously deposited bacteria, bacteria naturalized in the soil, or new sources such as animal feces (CLARY et al., 2020).

Besides hydrological benefits in terms of quantity and quality of water and hydrological cycle regulation, bioswales also provide various environmental and socioeconomic benefits. When strategically designed, these green areas can create ecological linkages that play a significant role in promoting biodiversity and urban quality of life.

Regarding biodiversity, besides maintaining native flora, these solutions create habitats for fauna, especially small mammals, amphibians, and insects. The vegetation in bioswales improves air quality, reduces the urban heat island effect,

and enhances evaporation, lowering local temperatures and promoting quality of life in urban centers (NATURAL RESOURCES CONSERVATION SERVICE, 2020).

From a socioeconomic perspective, bioswales offer a low-cost solution for stormwater treatment, reducing the load on drainage systems, which minimizes expenses on conventional drainage infrastructure, avoids costly expansions of existing systems, and generates green jobs (NACTO, 2004). The presence of vegetation cover adds significant landscape potential to the urban environment, improving the visual identity and aesthetics of the space. Using more diverse vegetation can also increase the positive impact on urban biodiversity and confer a more attractive and functional aspect to urban areas (URBAN GREEN BLUE GRIDS, n.d.).

Different stakeholders and sectors involved in bioswale implementation recognize their contribution to ecosystem services provision, including: i) Regulatory services, such as improved air and water quality, pollination, and local climate regulation; ii) Supporting services, involving biodiversity and habitat for fauna and flora; iii) Cultural services, associated with aesthetic value, recreation and tourism, science and education, social and spiritual, cultural and historical values (MILLER & MONTALTO, 2019).

1.6 Data Acquisition

This study utilized spatial data from public and institutional sources, as well as datasets defined by the author based on research objectives. The Digital Terrain Model (DTM), in raster format with a 0.5-meter spatial resolution, was obtained from the Bauru Municipal Government (Prefeitura Municipal de Bauru) through a request via its public information access portal (<https://www2.bauru.sp.gov.br/acessoainformacao/>). This raster dataset was critical for watershed delineation, outlet identification, and topographic and surface runoff analysis. The existing drainage network, provided in vector format by the same source, represented watercourses. Both datasets were used for hydrological preprocessing.

Land cover data were obtained through the processing (concatenation, geometric adjustment and *pansharpening*) of multispectral images acquired by the WPM sensor onboard the CBERS-04A satellite, resulting in imagery with a spatial resolution of 2 meters, suitable for urban studies such as land use mapping and urban expansion analyses (INPE,2019). For this study, images acquired in August 2025 and available in the National Institute for Space Research catalog (<https://www.dgi.inpe.br/catalogo/explore>) were used in combination with a Geographic Object-Based Image Analysis (GEOBIA) approach and the Random Forest machine learning algorithm, which is based on the construction of multiple decision trees to perform image classification. Image fusion was carried out using QGIS version 3.34.10 and the Orfeo Toolbox plugin.

In addition to the land cover mapping, Nature-Based Solution (NBS) implementation zones were delineated by the author based on visual interpretation of satellite imagery available in Google Earth plugin in QGIS, combined with technical and spatial criteria defined in the research scope. These datasets were developed in a Geographic Information System (GIS) environment and structured as polygonal vector data, representing proposed interventions for hydrological simulation.

The rainfall data used in the simulations were obtained from the Meteorological Research Institute (IPMET), which provides historical data at 5-minute intervals, through a formal request to the institution. The historical precipitation dataset covers the period from 2001 to 2025; however, data gaps were identified for September 2002. Consequently, only records from 2003 onward were considered in the analyses, both for the derivation of the IDF curves, the assessment of rainfall patterns, and the selection of a representative rainfall event.

For data processing, hydrological years were adopted, defined as the period from October of one year to September of the following calendar year (reference year). This approach was used to preserve the statistical independence of the annual maximum series, ensuring that critical seasonal events are fully captured within a single cycle. Consequently, each annual maximum can be regarded as an independent realization of a random variable (NAGHETTINI, 2007).

These datasets supported the hydrological modeling framework, allowing the definition of scenarios that more accurately represent real-world conditions for SWMM simulations.

1.7 Hydrological Model in SWMM

In this research, the hydrological and hydraulic model Storm Water Management Model (SWMM) was used, a software developed by the United States Environmental Protection Agency (EPA), version 5.2 (64-bit). SWMM is a dynamic rainfall-runoff model capable of simulating both the quantity and quality of surface and subsurface runoff, widely employed in urban drainage management.

SWMM can be applied to both single storm event simulations and long-term continuous series, being particularly useful for urbanized areas where soil impermeability significantly affects the hydrological cycle. As described by Rossman (2010), the model represents a variety of relevant hydrological processes, including time-varying precipitation, evaporation of accumulated water, rainfall interception by depressions, infiltration into unsaturated soils, percolation into deeper layers, flow exchanges between aquifers and the drainage system, as well as surface runoff modeling based on nonlinear reservoirs.

The spatial structure of SWMM is based on subdividing the study area into homogeneous drainage areas, each composed of permeable and impermeable sub-areas. This approach allows for a more accurate representation of urban landscape heterogeneity and runoff dynamics, which is transferred between sub-areas, sub-watersheds, and drainage system entry points.

The software also includes functionalities for simulating sustainable stormwater management solutions through the LID Controls module (Low Impact Development). These practices include systems such as rain gardens, green roofs, and permeable pavements, as well as techniques like infiltration trenches and swales, and the use of rain barrels (Low Impact Development Controls Module). It is also possible to insert storage units in the hydraulic section of the software. These solutions aim to mitigate environmental impacts and improve stormwater management efficiency in urban areas (ROSSMAN & HUBER, 2016).

Regarding the present study, this research first sought to investigate the feasibility of integrating different NBS typologies into densely urbanized watersheds to identify which solutions are most easily implemented within the local context. Building on this, the study addressed the following question: how does the implementation of these solutions - Rain Gardens, Bioswales, or Wetlands - alter the water balance and the macroscopic hydrological response of the watershed? It was hypothesized that flood volumes and peak discharges would be reduced differently depending on the chosen scenario. Additionally, the study questioned the extent to which unit-scale effectiveness differs among these typologies under varying operational conditions, using a mixed-effects model to test whether the intrinsic characteristics of each NBS result in statistically coherent performances. Finally, the research examined the actual contribution of increased infiltration and storage at the drainage area level, under the premise that recalculated runoff coefficients would determine whether NBS can significantly contribute to regional water resource management and support sustainable urban planning.

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Chapter 2:

Geoenvironmental Analysis of a Tropical Urban Stream Basin: Diagnosis and Prognosis²

Abstract. Water security and the prevention of hydrogeological disasters are vital for promoting environmental sustainability, the well-being of communities and infrastructure integrity in urban basins. Water security is related to ensuring the quantity and quality of water for human use, the maintenance of ecosystems and resilience to extreme hydrological events. In this context, the prevention of hydrogeological disasters is essential to avoid phenomena such as erosion, landslides, among others, which can cause social, economic and environmental damage. To mitigate the risks of natural disasters, enhance water resource management, and promote resilient and sustainable urban planning, conducting geoenvironmental analyses of urban basins is valuable for understanding local dynamics. The study area is an urban basin in Brazil (Água Comprida Stream, Bauru), where erosive processes and water contamination are observed. Additionally, the area downstream of the watershed has been classified as a region of high hydrological risk. The physical (geomorphology, slope and pedology), environmental (preserved green areas and water ecosystems) and anthropic (land use and urban infrastructure) characteristics were analyzed and related to processes identified in the watershed. The analysis was based on maps from Bauru City Hall, satellite and aerial images, water quality data and studies conducted by other researchers in the watershed. After identifying the processes, relationships were established with the predisposing and conditioning characteristics of the watershed. In general, it is observed that the processes (erosion, water contamination and hydrological risks) are associated with a lack of urban planning, which require the implementation of compensatory, mitigating and preventive measures. Among the main measures are Nature-based Solutions and integration between science and public policy (science-based landscaping planning).

² Article presented at the "Mediterranean Geosciences Union 4th Annual Meeting" and currently being edited for publication as a book chapter.

Keywords: Water Security, Hydrogeologic Sinisters, Urban Basin, Geoenvironmental Analysis.

2.1 Introduction

Actions to guarantee water security and prevent hydrogeological disasters are fundamental in promoting environmental sustainability, the well-being of communities, and the conservation of infrastructure in urban basins. The concept of water security involves ensuring the availability and quality of water for human use and economic activities, as well as preserving ecosystems and increasing resilience to extreme hydrological events (UNITED NATIONS, 2013). To increase resilience, the prevention of hydrogeological disasters is essential to avoid phenomena such as erosion and landslides, among others, which can cause social, economic and environmental damage. Urban areas are complex systems of interconnected services, with numerous factors that can contribute to disasters related to inadequate hydrological and geotechnical management (UNISDR, 2012). As global temperatures rise and precipitation patterns change due to climate change, cities are increasingly vulnerable to extreme hydrological events such as flooding and droughts. This vulnerability underscores the need to prioritize the integration of sustainable urban planning with resilient water management a priority. In this context, a geo-environmental analysis of an urban basin is proposed to promote resilient and sustainable urban planning that mitigates risks and improves water resource management.

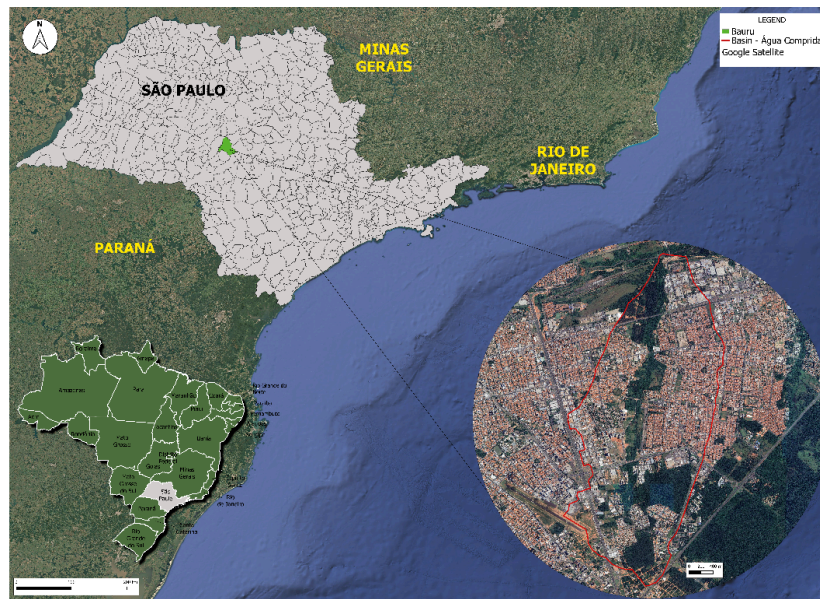
The motivation for considering the river basin as a spatial unit of analysis involves the possibility of integrating different planning, management, study and environmental activity systems into a multidisciplinary approach. From this perspective, river courses are analyzed not only in terms of their hydrological dimension, related to the behavior of water flows in their physical and limnological processes, but also in terms of their role in the environment. Changes in climate and land use alter landforms, geomorphic processes and hydrological dynamics, increasing the vulnerability of the basin. To avoid damage and disasters caused by these alterations, it is necessary to understand the mechanics of the processes, the factors influencing them and identifying sensitive sites for which conservation,

recovery and restoration actions should be a priority (CLAUDINO-SALES, LIMA & DINIZ, 2020). Disregarding the characteristics of the basin exacerbates degradation processes, increasing the risk of hydrogeological and environmental disasters. Thus, knowledge of geo-environmental elements is essential to reverse the socio-environmental degradation of water resources and to promote sustainable territorial and urban expansion (FERNANDES PEREIRA et al., 2021).

This study aims to analyze the dynamics between physical characteristics (geomorphology, slope, and pedology), environmental characteristics (preserved green areas, preservation of water ecosystems), anthropogenic characteristics (history of land use and urban infrastructure), and local processes. Insights from this analysis provide a basis that can be used to develop public policies, management plans, and actions. This approach aligns with global challenges addressed by the 17 Sustainable Development Goals (SDGs) of the United Nations, such as SDG 6 – Clean Water and Sanitation, SDG 11 – Sustainable Cities and Communities, and SDG 15 – Life on Land, highlighting the importance of sustainable management of water resources, urban planning, and the preservation of terrestrial ecosystems.

2.2 Materials and Methods

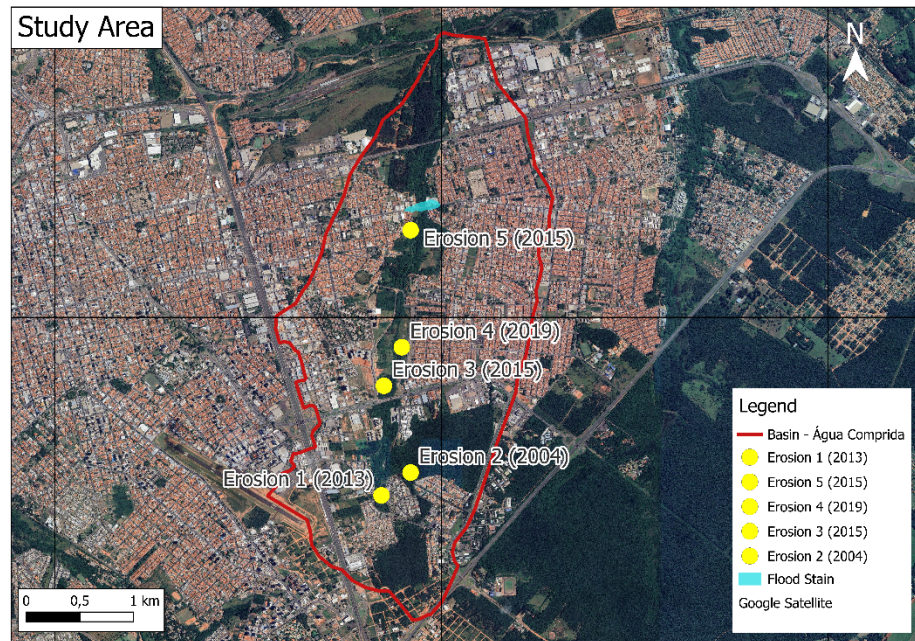
The area proposed for the geoenvironmental analysis was the Água Comprida stream basin, located in the municipality of Bauru, in the central-western part of São Paulo state, Brazil (Figure 2-1). In terms of hydrography, the Água Comprida stream, which is approximately 4.92 km long, is a tributary of the Bauru River. The stream's sub-basin has an area of 9.35 km² and an altitude variation of around 134 meters, with elevations ranging from 623m upstream of the basin to 489m at the outlet. The basin was chosen due to existing studies addressing erosion processes in the region (MOTA & FUREGATTI, 2021), susceptibility to groundwater contamination (OLIVEIRA et al., 2022), and the classification of the area downstream as a region of high hydrological risk (PEDRINI, 2018).

Figure 2-1: Location map

Source: Authors (2024).

The processes were identified using multiple data sources: Google Earth satellite images (2004 to 2024), aerial images of São Paulo Geographic and Cartographic Institute (Instituto Geográfico e Cartográfico de São Paulo (IGC)) from 1962, 1972, and 1979, and physicochemical water parameters at two points in the Água Comprida stream (2016 to 2023), provided by the Bauru Water and Wastewater Department (Departamento de Água e Esgoto de Bauru (DAE - Bauru)), and the aforementioned studies. To develop the geoenvironmental analysis, relationships were established with the basin's characteristics shown on the maps of the Bauru City Hall and other authors who have studied the area. Figure 2-2 highlights the locations of the erosion features identified and the flood patch delineated by the City Hall's flood map. Satellite and aerial images also made it possible to analyze local characteristics and the history of land use. After the processes occurring in the basin had been identified, relationships with local characteristics were established.

Figure 2-2: Basin area, erosion features and flood area.



Source: Authors (2024).

2.3 Results

2.3.1 Erosion Processes

According to the Semi-detailed pedological map (PREFEITURA MUNICIPAL DE BAURU, n.d.), the study area's soil composition includes sandy and slightly clayey soils with a porous structure and high collapsibility. The relief is characterized by the formation of low and medium-intensity rills, and river plains close to the banks of the watercourse (OLIVEIRA et al., 2022). From the aerial and satellite images, urban occupation has increased on the middle and lower slopes of the basin since 1962, there was an expansion of occupation at the outlet of the Água Comprida stream and, in 1972, there was land subdivision in the region of the stream's headwater. In 1979, there was a densification of urban occupation on the middle slope with the suppression of green areas on the basin from 2004 onwards.

The land use map shows a predominance of residential areas, along with commercial and industrial areas, and few green areas, these include a relevant Area of Ecological Interest (named "Floresta Urbana Água Comprida" at the stream's headwater and the Conservation Unit (named "Horto Florestal") at the outlet of the stream (PREFEITURA MUNICIPAL DE BAURU, n.d.). With urbanization advancing,

surface sealing has become predominant, leading to a substantial reduction in the soil's infiltration capacity. Satellite images analysis reveals a continuous expansion of urban areas, which has aggravated the occurrence of floods due to the concentration of runoff. In addition, there are empty plots distributed throughout the basin, with exposed soil and few or no vegetation cover, which are potential areas for leaching and erosion. As a result, erosion processes have intensified, especially in hillside areas, leading to the silting up of water bodies and the degradation of local aquatic ecosystems.

Based on satellite images and data from previous studies, the erosion processes at the site were located and detailed (Figure 2-2). Erosion 1, visualized by satellite image in 2013, was associated with the lack of maintenance of the facilities for dissipating energy from the water outlets of the nearby residential condominiums and the sloping topography of the land, which favors the formation of water flow channels without adequate energy dissipation in the pipes (CAMPOS & RODRIGUES, 2014).

Similarly to Erosion 1, Erosion 2 developed alongside the establishment of condominiums in the area, which discharged rainwater without proper energy dissipation. Since 2003, a small erosion process was present, widening into a gully after the housing developments were built, dragging up a lot of soil and aggravating the silting up of the stream. Even with planimetric alterations and the implementation of containment measures at the site, during a period of rain, slopes were destabilized and collapsed, as well as altering the bed of the stream. In 2004, containment works were carried out, as well as the damming of a water mine, which may have contributed to the appearance and expansion of the gully (CORGHI, 2008).

Aerial images showed that the appearance of Erosion 3 appeared with the start of the paving of a nearby Avenue (Figure 1). The presence of a structure, possibly a drainage system, can be observed, designed to channel rainwater to the riverbed. In 2019, after a heavy rainfall event, Erosion 4 caused a rupture in public infrastructure. According to a local newspaper, studies requested by Bauru City Hall indicated that the rainwater drainage system in the region was malfunctioning due to structural issues, contributing to frequent flooding during rainy periods. Evidence of

groundwater rising at the foot of the slope downstream from the damaged structure suggested it may have contributed to the destabilization of the building (G1, 2024).

In Erosion 5, the concentration of flow from a street lacking a drainage network may have directly influenced the erosion process. As such, urban occupation combined with the sandy soil substrates, the shape of the terrain and the absence of vegetation cover, have led to the emergence of erosion processes in the area. There is a real need to implement compensatory measures in the basin and restore existing features.

2.3.2 Siltation and banks changes

Satellite images also reveal changes along the banks of the Água Comprida stream, caused by the accumulation of sediment (siltation), waste and organic matter (Figure 2-3). The soil on the bank is classified as Melanic Gleisil, consisting of varying sands, silt, clay, and unconsolidated gravel (PREFEITURA MUNICIPAL DE BAURU, n.d.). The continuous deposition of sediment on the banks of the Água Comprida stream has resulted in the formation of sand banks and sedimentary islands along its course. These geomorphological changes have directly impacted on the natural course of the stream, forcing it to deflect and alter its route over time. This is evidenced by the analysis of satellite images, which show the gradual displacement of the stream bed due to the accumulation of sediment (Figure 2-4). The geomorphological changes not only impact the stream's morphology they can also disturb aquatic habitats and reduce water quality due to suspended solids and associated pollutants carried downstream.

Measures to reduce the transportation of sediment and urban waste from the upper basin areas to the stream's banks are essential. Moreover, ongoing monitoring through satellite imagery and field surveys should be conducted to assess the effectiveness of these interventions and to track further changes in the stream's morphology. Integrating geomorphological studies with hydrological and ecological assessments enables the development of a comprehensive management plan that addresses both the causes and consequences of siltation and bank alterations.

Figure 2-3: Photographic record of a stretch near the outlet of the Água Comprida stream, emphasis on the sandbank and solid waste



Source: Author (2024).

Figure 2-4: Delineation of changes in the banks of two stretches (A and B) near the outlet of the stream between 2004 and 2022



Source: Authors (2024).

2.3.3 Hydrological risk

At the site of the flood area the stream is channelled for crossing, restricting the natural flow of water and increasing the potential for flooding during periods of heavy rain. The area is characterized by low topographic elevation and proximity to the stream's floodplain, which exacerbates its vulnerability to hydrological risks. The convergence of rainwater from nearby streets and urban surfaces into this area can overwhelm the capacity to manage stormwater runoff.

The insufficient drainage infrastructure, combined with the urbanization of impermeable surfaces, contributes to the occurrence of floods. This scenario is especially problematic during peak rainfall events, when the stream's capacity to

accommodate increased water flow is limited, and can result in water overflow onto adjacent streets and properties.

In addition, the lack of green areas capable of absorbing excess rainwater has been identified as a possible contributing factor to the flooding of the site. Consequently, the risk of damage to public and private infrastructure requires the implementation of mitigation measures. Potential measures include the expansion of drainage networks, incorporation of Nature-based Solutions, and reinforcement of floodplain management policies. These measures could significantly help to reduce hydrological risks in this area.

2.3.4 Water Quality

The parameters used to assess the water quality of the Água Comprida stream, sampled and supplied by the DAE – Bauru, were dissolved oxygen (DO), hydrogen potential (pH), chemical oxygen demand (COD), biochemical oxygen demand (BOD), turbidity, conductivity, total dissolved solids (TDS), phosphate (PO_4), ammonia (NH_3), nitrate (NO_3) and nitrite (NO_2). The limit values for the parameters were established for Class II watercourses, as outlined by Brazil's environmental regulations in National Council of the Environment (Conselho Nacional do Meio Ambiente (CONAMA)) Resolution No. 357 (BRASIL, 2005), except for conductivity, for which the value set by the Environmental Sanitation Technology Company (Companhia Ambiental do Estado de São Paulo (CETESB)) of 100 $\mu\text{S}/\text{cm}$ was adopted (CETESB,2014) .

The sampling points were located near the head of the canal, where untreated sewage is discharged, and a point located at the other end, downstream. The percentages of the parameters that frequently failed to meet the limits were, for the upstream and downstream sampling points, respectively: phosphate (100% of irregular samples at both points), conductivity (98.73% and 100%), BOD (82.35% and 80.88%) and ammonia (41.18% and 68.33%). For the fifth parameter with the most changes, for the upstream's point, it was nitrate (39.13%) and downstream it was COD (33.77%). The samples were not outside the limit for TDS at both points and, in the case of the upstream's point, for nitrite. The limit values, total number of samples, number and percentage of non-compliant samples for each parameter are

shown in Table 1 and Table 2 for both points: near the headwaters and near the outlet, respectively.

Table 2-1: Analysis of non-compliant samples at the point near the headwaters.

Parameters	Limit Values	Near the head of the stream		
		Total of Samples	Non-compliant samples	Percentage of non-compliant samples
DO [mg/L]	≥ 5 mg/L	70	6	8,57
pH	pH 6 a pH 9	79	1	1,27
COD [mg/L]	≤ 50 mg/L	79	8	10,13
BOD [mg/L]	≤ 5 mg/L	68	56	82,35
Turbidity [N.T.U.]	≤ 100 N.T.U.	79	3	3,80
Conductivity [μs/cm]	≤ 100 μs/cm	79	78	98,73
TDS [mg/L]	≤ 500 mg/L	79	0	0,00
PO ₄ [mg/L]	≤ 0,1 mg/L	49	49	100,0
NH ₃ [mg/L]	pH ≤ 7,5: 3,7 mg/L 7,5 < pH ≤ 8,0: 2 mg/L	68	28	41,18
NO ₃ [mg/L]	≤ 10 mg/L	46	18	39,13
NO ₂ [mg/L]	≤ 1 mg/L	61	0	0,00

Source: Author (2024).

Table 2-2: Analysis of non-compliant samples at the points near the outlet.

Parameters	Limit Values	At the downstream, near the outfall		
		Total of Samples	Non-compliant samples	Percentage of non-compliant samples
DO [mg/L]	≥ 5 mg/L	80	25	31,25
pH	pH 6 a pH 9	88	5	5,68
COD [mg/L]	≤ 50 mg/L	77	26	33,77
BOD [mg/L]	≤ 5 mg/L	68	55	80,88
Turbidity [N.T.U.]	≤ 100 N.T.U.	76	3	3,95
Conductivity [μs/cm]	≤ 100 μs/cm	77	77	100,0
TDS [mg/L]	≤ 500 mg/L	77	0	0,00
PO ₄ [mg/L]	≤ 0,1 mg/L	47	47	100,0
NH ₃ [mg/L]	pH ≤ 7,5: 3,7 mg/L 7,5 < pH ≤ 8,0: 2 mg/L	60	41	68,33

NO ₃ [mg/L]	≤ 10 mg/L	44	8	18,18
NO ₂ [mg/L]	≤ 1 mg/L	58	2	3,45

Source: Author (2024).

A comparison between the upstream and downstream sites shows that although the general contamination trends persist along the watercourse, the downstream site tends to have slightly higher non-compliance for ammonia and COD, while the upstream site has more frequent problems with nitrate. This spatial variation may be associated with different sources of pollution, due to sewage discharge, or dispersed pollution throughout the basin.

The degradation of water quality, as demonstrated by the frequent exceedances of critical parameters, poses serious risks not only to the aquatic ecosystem, but also to the potential uses of the Água Comprida stream. The stream, classified as Class II by CONAMA Resolution 357, should ideally support activities such as recreational fishing, irrigation and the supply of drinking water after treatment. However, recurrent exceedances of some parameters such as BOD, phosphate and ammonia suggest that the water may not be suitable for these uses without substantial treatment, and the ecological integrity of the stream is under significant threat. Efforts are needed to reduce nutrient loads and organic pollutants in the stream, through treatment infrastructure, to restore water quality.

Another critical factor is associated with the quality of groundwater in the municipality's basins, which sometimes compromises the use of wells for supply in certain locations, due to contamination of the Bauru Aquifer (Aquífero Bauru ((AB)), a shallow aquifer of the free type, which extends throughout the study area. Previous studies show that the main contaminant in the AB is nitrate, but some sites also contain heavy metals, petroleum hydrocarbons, chlorinated solvents and, potentially, emerging contaminants (BARRETO et al., 2023).

The study basin presents moderate to high natural vulnerability to contamination of the Bauru Aquifer System (BAS) in the vicinity of the stream and moderate to low in the rest of the basin (ROSENBERGER et al., 2013). Although nitrate concentrations in the study basin do not exceed CONAMA potable water limits, the continuous generation and persistence of nitrate can lead to an increase in the concentration of the contaminant and the expansion of plumes in aquifers. Groundwater contamination restricts the use of these resources for human

consumption due to their inadequate quality. This highlights the importance of including measures that consider the interface between surface water and groundwater, to avoid contamination of the free aquifer, to provide for the conscious and sustainable use of these underground springs as a source of supply and, with this, to favor the municipality's water security (MINISTÉRIO DO DESENVOLVIMENTO REGIONAL, 2022).

2.4 Discussion

Urban expansion over the years has played a crucial role in intensifying phenomena in the Água Comprida stream basin, such as erosion processes, associated with growing urban occupation and the suppression of green areas. The presence of sandy and slightly clayey soils with low compaction and particle cohesion facilitated the emergence of erosion. Analysis of satellite images and historical maps reveals how changes in land use have contributed to the formation and worsening of erosive features over time. The combination of geomorphological and pedological characteristics in the river plains contributes to a series of fragilities, such as groundwater contamination, the occurrence of periodic floods, bank and slope erosion, silting and sliding of banks, which can cause damage to urban infrastructure.

The flood spot also highlights the area's predisposition due to its proximity to the floodplain and the convergence of rainwater. This scenario emphasizes the need for improvements in drainage infrastructure and urban planning to mitigate the impacts of flooding and ensure the safety and functionality of the streets during periods of heavy rain.

Regarding water quality, several parameters – including phosphate, conductivity, biochemical oxygen demand, ammonia, nitrate, and chemical oxygen demand – do not meet the standards and may be linked to anthropogenic sources of contamination, such as the discharge of industrial and domestic effluents, as well as runoff. The possible anthropogenic origin of the contaminants, particularly sewage, is corroborated by the existence of discharge points in the stream.

Although the Sanitary Sewerage System map indicates the existence of a treatment plant near the site, the lack of completion and activation of this treatment plant means that discharges continue untreated, negatively impacting the quality of

the water (OLIVEIRA et al., 2022). With the growing expansion of the basin and the occupation of the area, the inadequate discharge of effluent into the stream indicates an urgent need for improvements to the sanitation infrastructure such as the installation of a Wastewater treatment plant.

2.5 Concluding Remarks

The geoenvironmental analysis of the Água Comprida stream basin, Bauru/Brazil, revealed that a series of environmental and hydrological challenges could be associated with the physical and environmental factors, along with the lack of planned urban occupation and infrastructure. Cases like this demonstrate the need for integration and commitment on the part of the agents involved in setting up urban development, starting with a responsible study of the environmental constraints, both on a micro and macro scale. Based on the analysis, recommendations can be made for planning land and water use, along with the implementation of compensatory, mitigating, and preventive measures suited to the ecosystem's fragility along the stream. Examples of measures to be adopted include Nature-based Solutions, such as rain gardens, wetlands, riverbank filtration, green roofs, reforestation and other techniques that can help increase the resilience of areas. These actions aim to ensure water security, prevent hydrogeological disasters, and promote the well-being of communities in the basin.

While this study provides insights about the geomorphological and hydrological processes affecting the Água Comprida stream basin, future research should focus on the solutions implementation, long-term monitoring of implemented mitigation measures and their effectiveness in reducing erosion and improving water quality. To propose the implementation of technical solutions to mitigate these processes and promote the resilience of the basin, further studies should be conducted in the area, including research on reforestation, soil physical and chemical parameters, hydrological modeling, and additional physical-chemical water analyses, as well as other relevant studies. An integrated basin management approach, which combines land-use planning, infrastructure development, and Nature-based Solutions, is essential to create resilient urban environments capable of withstanding the increasing pressures from urbanization and climate change.

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Chapter 3:

***Extracting, Structuring and Organizing Spatial, Physical and Hydrological Data for Modeling: an Approach Using Programming Environment*³**

Abstract. Preparing spatial data for hydrological modeling is a complex task, especially when it involves multiple contributing areas and urban contexts, where land use intensifies spatial variability. This study presents an automated routine developed in R to extract, structure, and organize spatial, physical, and hydrological information needed for modeling in software such as Storm Water Management Model (SWMM). The routine helps optimize data pre-processing; nevertheless, critical evaluation of input data remains an essential step to ensure the quality and consistency of the analyses. Thus, the proposed approach offers an accessible, replicable, and adaptable tool for different datasets and contexts, designed to support the pre-processing of hydrological models, with the ability to inform detailed hydrological analyses and intervention planning.

Resumo: A preparação de dados espaciais para modelagem hidrológica é uma etapa complexa, especialmente quando envolve múltiplas áreas de contribuição e em contextos urbanos, nos quais a ocupação do solo intensifica a variabilidade espacial. Este estudo apresenta uma rotina automatizada desenvolvida em R para extrair, estruturar e organizar informações espaciais, físicas e hidrológicas necessárias à modelagem em softwares como o SWMM. A rotina contribui para otimizar o pré-processamento de dados; ainda assim, a avaliação crítica dos insumos permanece uma etapa essencial, garantindo a qualidade e a consistência das análises. Portanto, a abordagem proposta apresenta uma ferramenta acessível, replicável e adaptável a diferentes bases de dados e contextos, destinada ao pré-processamento de dados para modelagem hidrológica, com capacidade de subsidiar análises hidrológicas detalhadas e o planejamento de intervenções.

Keywords: Hydrological Model, Data pre-processing, R programming.

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3.1 Introduction

With the advance of urban occupation, the management of water resources has become an increasing challenge, both in terms of directing water flows and maintaining the quality of water courses. The history of occupation and the indiscriminate transformation of space highlight the impacts on natural resources and the difficulties in integrating environmental and urban aspects in territorial planning processes (PEIXOTO, 2006). As a consequence of this disarticulation, water security is compromised, resulting in environmental, social, and economic impacts in occupied areas (BAO & FANG, 2008). Thus, the balance between urban development and the conservation of water resources constitutes a fundamental basis for effective management of water resources as well as a strategy to minimize the negative impacts of urbanization and achieve sustainable urban development (PEIXOTO, 2006; CHEN et al., 2024). In this context, the basin is consolidated as the planning and management unit, being a well-defined physical unit representative of hydrological processes (AHER et al., 2014; VILAÇA et al., 2009).

In order to support urban water management through the adoption of public policies that integrate public spaces with sustainable stormwater management measures – including runoff forecasting, structures sizing, and flood mitigation – simulation models are currently used to represent hydrological processes in urban basins (TUCCI, 2007). Computational numerical models such as the Storm Water Management Model (SWMM) allow simulation of the hydrological and hydraulic behavior of urban basins, serving as a convenient tool for drainage system planning and for the implementation of Nature-based Solutions (NbS). These solutions demonstrate effectiveness in reducing surface runoff and contribute to increasing biodiversity, strengthening climate resilience, recharging aquifers, improving water quality, and enhancing urban spaces (WOODS BALLARD et al., 2015; EL BAIDA, CHOURAK & BOUSHABA, 2025; TSATSOU, FRANTZESKAKI & MALAMIS, 2023).

Adequate representation of hydrological dynamics and the effects of NbS on the basin in models such as SWMM demands detailed characterization of basins or drainage areas, with spatial and physical data such as land cover, slope, etc (NIEMI et al., 2019). Distributed models that incorporate spatial variability provide a more realistic representation of urban hydrological processes (NGUYEN et al., 2016).

Despite advances in hydrological modeling, the use of models with detailed input data often presents challenges, especially due to the volume and complexity of the information, particularly when working with multiple areas. In such cases, the process of data acquisition and organization requires greater care to avoid errors and ensure model quality.

Due to the complexity of preprocessing input data for hydrological models, implementing automated computational routines for this process constitutes an important strategy to reduce preparation time, minimize errors, and ensure data traceability. In this regard, the present study proposes a routine within the RStudio environment, using the R language, with the aim to facilitate the extraction, structuring, and organization of spatial and physical data necessary for hydrological modeling in software. The proposed routine seeks to optimize the workflow and provide an accessible, replicable, and efficient tool for researchers and professionals in water resources management.

3.2 Material and Methods

3.2.1 General structure of the computer routine

The routine consists of a set of modular scripts, each dedicated to a specific task within the hydrological modeling preprocessing workflow. Each script follows a standardized structure: loading packages, setting the working directory, importing input data with defined formats and coordinate reference systems (CRS), performing targeted operations (e.g., attribute extraction, topological analyses), and finally visualizing and exporting the results. These scripts are designed to be used as a complete routine or individually, as long as the necessary inputs, which also include the outputs of the previous processes, are available. Figure 3-1 presents the workflow diagram outlining the overall logic.

Figure 3-1: Workflow diagram.



Source: Authors (2025).

Table 3-1 presents the packages and functions commonly used across the scripts, grouped by processing steps such as setting the working directory, data import, CRS definition, result visualization, and export. The following subsections describe each script individually, according to its specific role within the modular preprocessing routine.

Table 3-1: Packages and functions common to the scripts.

Packages	Functions	Description
<i>Base R</i>	<i>setwd(); dirname()</i>	Defines the working directory.
<i>sf</i>	<i>st_read(); st_crs();</i>	Import data; transform CRS; convert to sf;
	<i>st_transform(); st_as_sf();</i>	export GIS formats; remove geometry;
	<i>st_write(); st_drop_geometry();</i>	ensure topologically correct geometries;
	<i>st_make_valid (); st_is_empty()</i>	check empty geometries.
<i>writexl</i>	<i>write_xlsx()</i>	Export dataframes to Excel files (.xlsx).
<i>raster</i>	<i>raster(); writeRaster()</i>	Import and export raster data.
<i>ggplot2</i>	<i>ggplot(); geom_sf();</i>	Create graphic; plot spatial data;
	<i>theme_minimal(); aes(); labs()</i>	minimalist theme; aesthetic mapping; add titles and subtitles.

Source: Author (2025).

3.2.2 Input Data

The input data required to run the routine includes spatial and physical information about the basin. Table 2 provides a detailed summary of all input data required for the routine, specifying the type of information, a brief description of each item, and the respective formats accepted for processing. In addition, some of the outputs generated by the processing of the modular steps are also inputs for subsequent steps in the routine.

Table 3-2: Input data, description and file format.

Input Data File	Description	File Format
Digital Terrain Model (DTM)	Terrain surface elevation	Raster (.tif)
Stream Network	Lines representing the watercourses.	Vector - Line (.shp)
Land Cover	Classification of land cover.	Raster (.tif)

Areas of Hydrological Interest	Areas for sub-basin delimitation, as NbS areas.	Vector - Polygon (.shp)
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Source: Authors (2025).

3.3 Development and Application of the Routine

3.3.1 Outlets

The identification of preferential outlet points within hydrological areas of interest is structured through an RStudio script using the packages *sf*, *raster*, *dplyr*, *ggplot2*, and *writexl*. The main objective of the code is to extract, within each area, the points with the lowest elevation, which represent outlets, contributing to the delineation of drainage areas.

The spatial data processing begins with the application of the *st_buffer()* function to create buffers around the geometries corresponding to the areas of interest. This step is necessary to ensure that narrow features are properly considered as areas in subsequent analyses. After standardizing the geometries, a column identifying the type of area of interest is added to each simple feature object. Subsequently, the layers of areas of interest are merged into a single simple feature using the *bind_rows()* function from the *dplyr* package.

To optimize computational performance, the spatial resolution of the Digital Terrain Model (DTM) is adjusted using the *aggregate()* function from the *raster* package, which aggregates pixels by increasing their size and reducing data granularity according to the aggregation factor configured. This step aims to reduce processing complexity while maintaining an adequate representation of the terrain for analysis. Next, a list is created to store the points corresponding to the lowest elevations within each polygon of interest, with the number of points to be extracted predefined. The core processing involves an iterative loop controlled by the *nrow()* function, which iterates through each feature of the area of interest individually. For each polygon, the *mask()* function from the *raster* package is applied to clip the DTM, isolating the elevation values within that spatial extent.

Non-null elevation values are extracted using the *values()* and *st_zm()* functions, from the packages *raster* and *sf*, respectively, to check the sufficiency of the data for analysis. When data is available, the indices corresponding to the n

lowest elevation points are identified via the *order()* function from *base R*. These indices are converted into geographic coordinates using the *values()* and *xyFromCell()* functions (*raster* package), transforming raster cells into XY points in the DTM's spatial reference system. To ensure uniqueness of the extracted points, the *unique()* function from *base R* is applied to remove duplicate coordinates, giving priority points with the lowest elevations and thus guaranteeing an adequate spatial representation of the identified outlets. Each lower elevation point is converted into a spatial object with associated attributes (area ID, area of interest classification, point sequence number, and elevation value) and stored in a list. At the end of the loop, all extracted points are combined into a single simple feature and exported as shapefile and summary table for quantitative analysis.

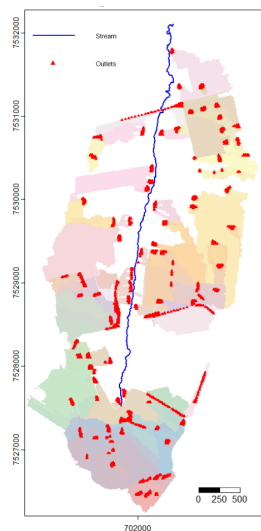
3.3.2 Drainage Areas

The delineation of drainage areas associated with outlet points is performed using the *terra*, *sf*, *dplyr*, and *whitebox* packages, with the latter providing specialized tools for spatial hydrological analysis. Initially, the vector stream network is rasterized using the *rasterize()* function to generate a mask in which the pixels of watercourses are assigned a value of 1. These pixels are subsequently set as NA within the modified Digital Terrain Model (DTM) to ensure that surface runoff adhered to the fluvial topology and to prevent the encroachment of contributing areas onto existing stream channels.

Hydrological preprocessing begins with the application of *wbt_fill_depressions()* to correct artificial depressions present in the DTM. The flow direction is computed using *wbt_d8_pointer()*, which assigns to each raster cell the neighboring cell that receives flow. Flow accumulation is then calculated using *wbt_d8_flow_accumulation()*, quantifying the volume of flow accumulated per pixel and delineating natural channels. The outlet points are refined by employing *wbt_snap_pour_points()*, which adjusts their positions to adjacent cells that exhibit greater flow accumulation within a predetermined threshold distance, thus ensuring accurate spatial representation of hydrological outflows. Subsequently, drainage areas are delineated using *wbt_drainage_areas()*, which identifies raster cells that contribute flow to each outlet, thus defining discrete catchment polygons.

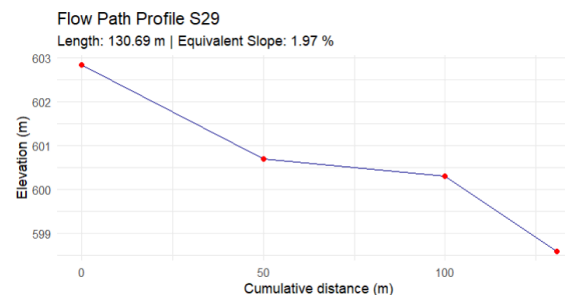
These raster-based drainage areas are converted to vector polygons using *as.polygons()* and subsequently transformed into simple feature objects via *st_as_sf()*. A spatial join is conducted with *st_join()* to associate each polygon with its corresponding outlet point, allowing unique identifiers to be assigned and facilitating further analyses. Finally, polygons related to the same outlet are aggregated and consolidated using *group_by()* and *summarise()* functions from the *dplyr* package, optimizing the structure of the dataset. The resulting output comprises a set of validated vector polygons representing the delineated drainage areas for each hydrological outlet (Figure 3-2).

Figure 3-2: Illustrative result of the practical application of the “Outlets” and “Drainage Areas” scripts.



Source: Authors (2025).

Figure 3-3: Illustrative result of the practical application of the “Equivalent Slope” script.



Source: Authors (2025).

3.3.3 Flow Path

The developed script employs the packages *whitebox*, *terra*, *sf*, *igraph*, *dplyr*, *purrr*, *ggplot2*, and *writexl* to extract and analyze the preferred flow paths within the delineated drainage areas. The main objective of the code is to identify the longest preferential flow paths within each drainage area polygon. The processing begins with reading the spatial data: the DEM (*raster*) and the drainage areas in shapefile format. The function *process_drainage_area()* is designed to execute the processing

and vectorization steps for each drainage area. The function starts by selecting a drainage area based on the Name field and uses its geometry to crop the DEM using the *crop()* and *mask()* functions from the *terra* package. The clipped DEM is temporarily exported as a .tif file to enable its use by the whitebox tools. The model is smoothed with the function *wbt_gaussian_filter()*, which applies a Gaussian filter to reduce noise. The depressions are filled using the *wbt_fill_depressions()* function to ensure flow continuity.

From the filled DEM, flow direction (*wbt_d8_pointer()*) and flow accumulation (*wbt_d8_flow_accumulation()*) files are generated, which represent the flow orientation in each raster cell and the relative water contribution, respectively. Based on this data, the raster drainage network is extracted through the *wbt_extract_streams()* function, where a minimum contribution threshold can be set to define channels, controlling the sensitivity of identification according to the minimum number of contributing cells at each point. The rasterized drainage network is then converted into linear vector features with the *wbt_raster_streams_to_vector()* function. This vector network is read again using *st_read()* and, if necessary, the correct spatial projection is assigned.

Next, to identify the longest path within the network, a graph structure is built using the *igraph* package. Each network segment is converted into a graph edge using the functions *map()*, *paste()*, and *tibble()*, with vertices defined by the start and end coordinates of the lines. From the graph generated by *graph_from_data_frame()*, a search is performed for the longest path among all possible node pairs, using the *shortest_paths()* function with negative weights with the following commands.

```

125 g <- graph_from_data_frame(edges, directed = TRUE)
126 sources <- V(g)[degree(g, mode = "in") == 0]
127 sinks <- V(g)[degree(g, mode = "out") == 0]
128 max_length <- -Inf
129 longest_path <- NULL
130 for (source in sources) {
131   for (sink in sinks) {
132     path <- shortest_paths(
133       g, from = source, to = sink,
134       weights = -E(g)$length,
135       output = "vpath"
136     )$vpath[[1]]
137     if (length(path) > 0) {
138       path_length <- sum(E(g, path = path)$length)
139       if (path_length > max_length) {
140         max_length <- path_length
141         longest_path <- path
142       }
143     }
144   }
145 }
146 if (is.null(longest_path))
147   stop("Could not identify the main flow path.")

```

The segments comprising the main flow path are extracted from the original network using *semi_join()* and *mutate()* to identify matching vertex pairs (from-to). The final result is saved as a simple feature, named according to the corresponding drainage area, for export. After running the *process_drainage_area()* function, results are returned as a list. This function is applied iteratively to all drainage areas using *purrr*'s *map()*. The consolidated results are combined into a data frame with *map_dfr()* and exported as a spreadsheet and shapefile.

3.3.4 Equivalent Slope

The script aims to calculate the equivalent slope along the longitudinal profiles of the flow paths. A Digital Terrain Model (DTM) and a vector layer of the flow paths are used as input data. Processing is carried out using the packages *terra*, *sf*, *ggplot2*, *dplyr*, *purrr*, and *writexl*.

Initially, the script performs data import and verification – a step that ensures the correct association between each geometry and its respective identifier. It then defines the function *sample_continuous_points()*, which is responsible for generating elevation sampling points along each flow path, with a regular spacing defined by the user, as well as including the residual point corresponding to the final segment

leftover after this regular division. Then, the function *process_flow_path()* uses the sampled points to extract the corresponding elevations for each flow path from the DTM. Based on this information, the slope of each segment (D_i) between consecutive point pairs is calculated using the *base R diff()* function. With these data, the equivalent slope of the main channels is calculated according to equation (1) (DAEE, 2005).

$$I_{eq} = \left(\frac{\sum L_i}{\sum \left(\frac{L_i}{\sqrt{D_i}} \right)} \right)^2 \quad \text{where } D_i = \frac{\Delta H_i}{L_i} \quad (\text{Eq.1})$$

Deq = equivalent slope (m/m); L_i = horizontal length of each segment i (m); D_i = slope of each segment i (m/m); ΔH_i = elevation difference between the ends of each segment i (m).

With the functions defined, the script iterates over all the channels present in the shapefile, applying the processing function to each one. The results are stored in a list that consolidates information on total length, the equivalent slope, and the sampled points with their respective elevations. In the final step, the script organizes and exports the data in a summary table containing, for each channel, its name, total length in meters, and equivalent slope as a percentage. Additionally, longitudinal profile graphs are generated, showing the variation in elevation along each flow path, with distance and elevation indications (Figure 3-3).

3.3.5 Flow Routing

To characterize the hydrological connections between drainage areas through surface runoff, the “Flow Routing” script uses the *sf*, *dplyr*, *whitebox*, *ggplot2*, and *writexl* packages. The main goal of the code is to identify the flow paths between different drainage basins, linking the start and end points of the flow paths to their respective drainage areas, thus enabling the generation of flow direction lines and a table indicating the source and destination of the flow.

Processing begins with the loading of the files: DTM, flow path vector, and drainage areas. The subsequent steps—smoothing of the digital terrain model (*wbt_gaussian_filter()*), filling depressions (*wbt_fill_depressions()*), and determining

flow directions (*wbt_d8_pointer()*) – are carried out in a similar manner to the “Flow Path” script.

The *get_extreme_points()* function identifies the origin and destination points of the flow paths (highest and lowest elevation, respectively) by extracting the coordinates and elevation values from the DTM. From the identified end point, the *find_destination_drainage_area()* function simulates the water path cell-by-cell on the flow direction raster, moving in the direction indicated by the D8 code. The D8 model codes the water movement toward one of the eight neighboring cells using discrete values: 1 for east, 2 for southeast, 4 for south, 8 for southwest, 16 for west, 32 for northwest, 64 for north, and 128 for northeast.

Subsequently, the script goes through a sequence of conditionals to identify and update the position of the current cell, progressing until the path reaches a cell that intersects the drainage area receiving the flow. The main processing routine applies the described functions to each flow path and associates the resulting information with the corresponding drainage areas. The flow paths between drainage areas are organized into a data frame, converted into a spatial object with linestring geometry, and exported.

3.3.6 Land Cover

The land cover analysis script for each drainage basin employs the *terra*, *sf*, *dplyr*, and *writexl* packages. The input data include a land cover raster and a shapefile of the drainage areas. Merging the drainage areas using *st_union()* from the *sf* package creates a single continuous polygon covering the entire basin, which is then used to crop the raster with the *crop()* and *mask()* functions.

Next, the raster is reclassified using the *classify()* function, based on a matrix (*reclass_matrix*) that groups different land use classes into broader categories. In the current configuration, the categories include: Urban Area (1) - intensive anthropogenic uses such as cities, agriculture, and mining; Barren or Non-Vegetated Areas (2) - undefined or naturally non-vegetated areas, such as rivers, dunes, and rocky outcrops; and Green Area (3) - native vegetation formations or reforested areas. These categories can be adjusted as necessary, depending on the analysis objectives or different land use and land cover classifications. The reclassified raster

is then converted to vector format using *as.polygons()*, with the value column renamed to "Classes" using *rename()*.

The intersection between the land cover polygons and the drainage areas is performed with *st_intersection()*, and for each polygon intersected, the area in square meters is calculated using *st_area()*. A new column (LAND_USE_TYPE) is created using *case_when()* to translate the reclassified values into meaningful labels. The *group_by()* and *summarise()* functions from the *dplyr* package are used to calculate the total area of each land use class per drainage basin. Then the percentage of each class within its respective basin is calculated and stored in the percent column.

As a verification step, the percentage totals per basin are summed to ensure they approximately equal 100%, helping to identify possible data inconsistencies or overlaps. Finally, the results are exported as a table. The reclassified raster can also be saved to disk using *writeRaster()*. The land cover categories are visualized on a map using *plot()*, with a color palette that differentiates the three categories.

3.4 Discussion

The routine integrated tools for spatial analysis, hydrological processing, and data manipulation within the R environment, with the aim of reducing the time required to extract and organize spatial, physical, and hydrological information from areas of interest. The automation of these steps represents a significant advancement, especially in hydrological studies involving multiple sub-basins or large territorial extensions.

The routine's modular structure allows easy parameter customization and adaptation to different geographic contexts, data sources, and analytical objectives, according to the user's data, goals, and scale of work. This flexibility considerably expands the routine's potential applications, making it suitable for both individual case studies and comparative analyses across different scenarios or regions.

However, it is important to highlight that the quality of the results generated depends directly on the quality of the input data. The spatial resolution of the Digital Terrain Model (DTM) and the quality and detail of the land use and land cover classification are critical factors that can significantly influence the accuracy of the

derived variables. Therefore, although the routine provides a systematic and replicable process for data preprocessing, it does not replace the critical evaluation of the inputs used. Users should check the suitability of the data for the requirements of the hydrological model or the intended analysis.

3.5 Concluding Remarks

The automated routine developed in this study represents a practical advancement in supporting hydrological modeling, especially in contexts that require processing large volumes of spatial data. Its implementation in R, using widely adopted packages such as *sf*, *terra*, and *ggplot2*, ensures accessibility and ease of adaptation to different projects. By enabling the systematic extraction of variables such as elevation, slope, and land use, the tool helps optimize the workflow of researchers and technicians, promoting greater efficiency, reproducibility, and reliability in the preparation of input data. Future perspectives could include the incorporation of machine learning methods to enhance land use classification and the prediction of hydrological variables, as well as the development of graphical user interfaces to make the routine more accessible to users with varying levels of programming expertise. Additionally, integration with real-time data, such as hydrometeorological sensors and updated satellite imagery, could increase the routine's applicability in dynamic environmental monitoring and management.

3.6 Data Availability Statement

All codes generated and used during the study are available in a repository online: <https://github.com/anapascucci/preprocessingroutine.git>.

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Chapter 4:

Planning Nature-Based Solutions in Urban Watersheds: Site Selection Criteria and Automated Hydrological Data Pre-Processing⁴

Keywords: Nature-Based Solutions, Urban Watershed management, Stormwater management, Site Selection Criteria, Hydrological modeling, Urban Resilience

Highlights

- **Environmental Impact:** The manuscript examines the integration of Nature-Based Solutions (NbS) into urban environments to enhance ecosystem services and strengthen urban resilience. It identifies key implementation constraints, including regulatory gaps and spatial limitations, and emphasizes the need for regionally adapted technical guidelines. The findings also highlight the importance of incorporating NbS considerations into urban planning processes at early stages to ensure their effective and scalable application.
- **Economic Implications:** The study presents an integrated approach that combines site-selection criteria with an automated hydrological data preprocessing routine to support the planning and evaluation of NbS in urban watersheds. By relying on open-access tools and spatial datasets, the methodology enables the processing of large volumes of information and their transformation into actionable knowledge for urban planning and public policy development.
- **Social Relevance:** The research contributes to addressing societal challenges by supporting the development of public policies aimed at improving urban resilience to climate-related risks. The proposed framework provides valuable information for water resource managers, urban planners, and local governments, facilitating the design of mitigation and adaptation strategies that promote more sustainable and resilient urban environments.

Abstract. Urban water resource management is increasingly challenged by rapid urbanization, climate variability, and growing pressures on drainage infrastructure. In this context, Nature-Based Solutions (NbS) have emerged as promising strategies to

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enhance stormwater management while providing additional environmental and social benefits. However, their effective implementation in urban environments requires clear technical criteria for site selection and analytical tools capable of supporting hydrological assessment at appropriate spatial scales. This study proposes an integrated framework that combines technical criteria for NbS site selection with an automated hydrological data pre-processing routine. The framework was applied to the Água Comprida stream watershed in Bauru, São Paulo, Brazil, focusing on three NbS types: rain gardens, constructed wetlands, and bioswales. Site identification was guided by criteria reported in the literature, including terrain slope, proximity to groundwater levels and building foundations, and minimum spatial requirements. Spatial datasets and thematic maps were integrated using geospatial analysis to identify suitable implementation areas. In parallel, an automated routine developed in R processed topographic, land-use, and hydrological data to structure model-ready inputs, including drainage area delineation, flow path identification, slope calculation, and land-cover characterization. Sensitivity analyses were conducted to evaluate key parameters and ensure robustness in the automated procedures.

The results demonstrate that the proposed framework supports systematic NbS planning at the watershed scale by improving the organization and reliability of hydrological input data while enabling spatial identification of suitable intervention areas. In addition, the study highlights regulatory and spatial constraints that may limit NbS implementation in consolidated urban areas, emphasizing the importance of adapting urban planning instruments and technical guidelines. The proposed methodology offers a replicable approach that can support urban water management strategies and the integration of NbS into planning processes in other urban watersheds.

4.1 Introduction

Urban areas constitute complex socio-ecological systems increasingly shaped by population growth and unplanned spatial expansion, processes that frequently compromise both urban resilience and the ecological integrity of supporting environments. Urbanization alters the natural functioning of the hydrological cycle by modifying key processes such as surface runoff generation,

aquifer recharge, infiltration, interception, and evaporation. These alterations have direct consequences for water availability, drainage performance, and water quality within urban catchments (SAJUKUMAR and REMYA, 2015; TONG et al, 2012). As impervious surfaces expand and natural drainage pathways are disrupted, runoff volumes and peak flows tend to increase, often exceeding the capacity of conventional stormwater infrastructure. In many cases, traditional drainage systems designed to rapidly convey water away from urban areas can intensify hydrological problems by accelerating runoff transport and facilitating the mobilization of pollutants into receiving water bodies (ALVES et al., 2012).

In response to these challenges, Nature-based Solutions (NbS) have emerged as promising strategies for improving urban water management while enhancing ecosystem services and urban resilience. NbS integrates ecological processes with engineering practices to regulate stormwater dynamics, mitigate runoff impacts, and improve water quality in urban environments. Beyond their hydrological functions, these interventions also contribute to broader governance objectives by connecting environmental management with social, economic, and planning dimensions through institutional arrangements and community engagement. Consequently, NbS are increasingly recognized not only as technical measures for stormwater control but also as instruments capable of promoting more integrated and sustainable urban development.

A wide range of measures fall under the NbS framework, including linear parks, green roofs, permeable pavements, reforestation, stormwater ponds, rain gardens, bioswales, and constructed wetlands (TSATSOU, FRANTZESKAKI and MALAMIS, 2023; ORAL et al., 2021). Constructed wetlands are engineered saturated or unsaturated systems designed to reproduce key ecological functions of natural wetlands by retaining and treating urban runoff while supporting biodiversity, even within densely built environments (MOORE and HUNT, 2011). Rain gardens and bioswales represent bioretention practices intended to manage stormwater through infiltration and pollutant removal, although they differ in configuration and hydrological behavior. Bioswales typically consist of shallow vegetated channels that convey runoff while promoting infiltration, performing particularly well when peak flows occur shortly after precipitation events. Rain gardens, in contrast, are vegetated depressions designed to capture runoff from impervious surfaces and facilitate

temporary storage, evaporation, and infiltration, thereby enhancing pollutant removal and attenuating peak discharge (LU et al., 2024; EKKA et al., 2021; SHARMA and MALAVIYA, 2021).

Despite the increasing recognition of their benefits, the implementation of NbS in urban environments remains limited by several practical and methodological constraints. One major challenge is the scarcity of locally adapted technical criteria to guide the design and siting of these interventions. International guidelines offer useful starting points, yet they are often dispersed across multiple sources, strongly influenced by local environmental conditions, and rarely provide specific recommendations for different climatic regimes (KÕIV-VAINIK et al., 2022). As a result, practitioners and planners frequently encounter difficulties when attempting to translate general design principles into context-specific applications, particularly in tropical urban environments.

In addition to the lack of consolidated design criteria, relatively few studies have addressed the methodological challenges associated with identifying suitable NbS locations and preparing hydrological datasets to support watershed-scale analyses. Existing research has largely concentrated on evaluating the hydrological performance of individual NbS installations or small groups of interventions, typically at the neighborhood or site scale (FOOMANI and MALEKMOHAMMADI, 2020; SU et al.; 2023). However, the effectiveness of NbS in improving urban water management often depends on their distributed implementation across an entire watershed, where cumulative hydrological effects can influence runoff generation, flow routing, and stormwater retention across interconnected drainage areas. Fragmented land availability in highly urbanized environments further complicates this process, as spatial constraints frequently limit the feasibility of isolated interventions and require more strategic planning approaches.

Consequently, evaluations of NbS potential should extend beyond existing land-use patterns and regulatory frameworks to consider the institutional and spatial adjustments necessary to enable broader implementation in urban water management systems. A watershed-scale perspective is particularly important in this context. While individual interventions can generate localized benefits, coordinated implementation across multiple locations within a watershed is more likely to produce measurable hydrological improvements and address systemic drainage challenges.

Nevertheless, comparatively few hydrological studies have examined NbS planning and assessment at this broader spatial scale, especially in tropical urban watersheds (CASTELO and FERREIRA, 2023; PRADO et al., 2024). This scale mismatch highlights the need for methodological approaches capable of supporting integrated watershed-level planning and hydrological analysis.

Addressing these challenges requires approaches that simultaneously consider NbS selection criteria, spatial planning constraints, and watershed-scale hydrological processes. In this context, computational tools capable of organizing and processing large spatial datasets can play a key role in supporting systematic NbS planning and evaluation. Automated data-processing routines can improve the efficiency and transparency of hydrological analyses by structuring model inputs, delineating drainage areas, identifying flow paths, and integrating topographic and land-use information required for watershed-scale modeling.

Accordingly, this study develops and evaluates a methodological framework that combines technical siting criteria for Nature-based Solutions with automated hydrological data preprocessing to support watershed-scale planning and hydrological analysis in urban environments under tropical conditions. Specifically, the study consolidates and operationalizes technical criteria reported in the literature to enable the spatial identification of suitable locations for rain gardens, constructed wetlands, and bioswales within a consolidated urban watershed. It also introduces an automated routine for hydrological data preprocessing that integrates topographic, land-use, and drainage information to generate structured inputs required for watershed-scale hydrological modeling. Finally, the study demonstrates how the combined application of spatial siting criteria and automated data processing can support systematic NbS planning while identifying spatial and regulatory constraints that influence implementation in highly urbanized watersheds.

4.2 Material and Methods

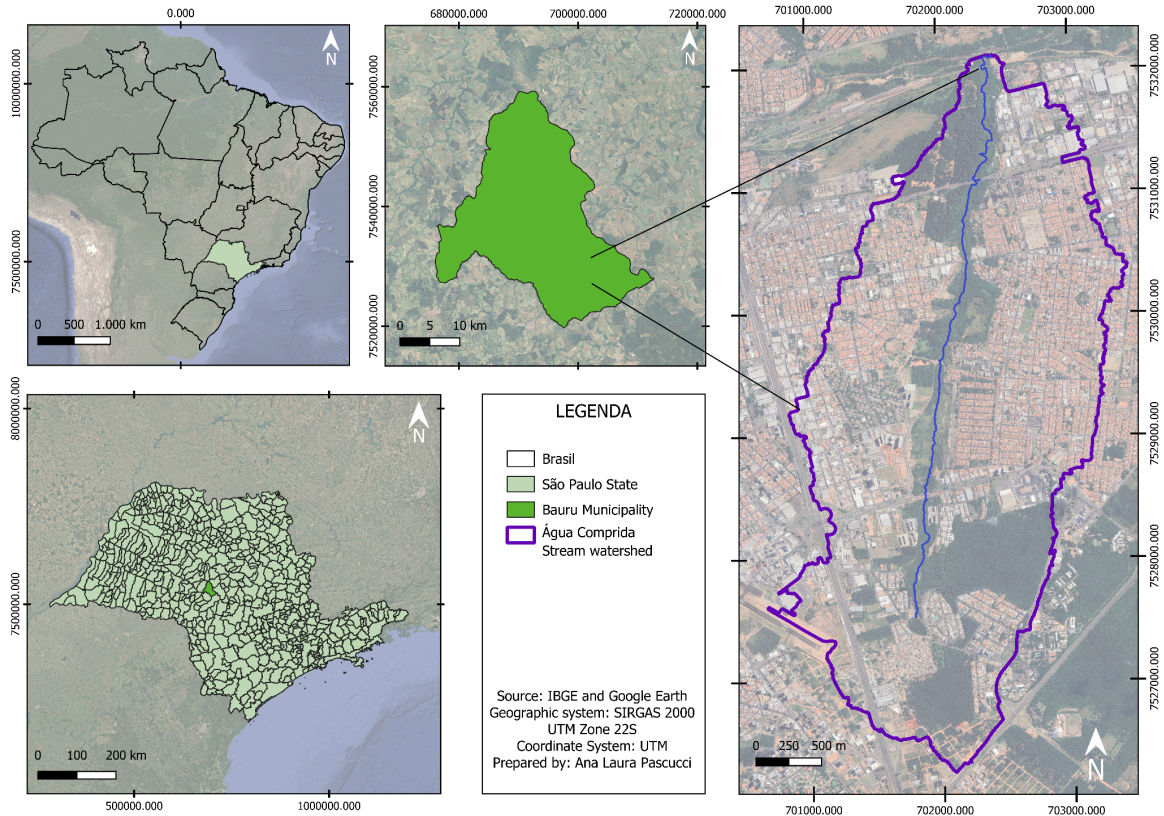
This methodological framework was designed to combine literature-based implementation criteria with geospatial modeling and local environmental data. The approach prioritizes transparency and reproducibility, seeking to demonstrate a replicable procedure for pre-processing in tropical urban watersheds.

4.2.1 Study Area Description

The study area is located in the municipality of Bauru, in the west-central region of São Paulo State, Brazil. The municipality territory is drained by two principal river systems: the Batalha River, which predominantly drains rural areas, and the Bauru River, which flows through the urban perimeter. Within the urbanized portion of the municipality, the drainage network is subdivided into nine sub-watersheds. Among these, the Água Comprida watershed is particularly relevant due to its high degree of urban occupation and associated hydrological pressures (Figure 4-1).

According to the Köppen climate classification, the region presents a high-altitude tropical climate (Cwa) (KÖPPEN, 1948), characterized by warm, wet summers and dry winters. Average annual temperature is approximately 19.8 °C, and mean monthly precipitation is about 121.0 mm, with rainfall concentrated mainly in January and February during the summer season.

The soils within the Água Comprida watershed are predominantly classified as Dark Red Latosols (Lea8), corresponding to Ferralsols according to the World Reference Base for Soil Resources (WRB), the international soil classification system supported by FAO. These soils typically exhibit deep profiles and high weathering levels but are also associated with significant susceptibility to erosion under conditions of intense rainfall and urban soil disturbance (MUNICÍPIO DE BAURU, 2016; PREFEITURA MUNICIPAL DE BAURU, 2015; MOTA and FUREGATTI, 2021; SANTOS et al., 2018).

Figure 4-1: Location map of the study area.

Source: Authors (2025).

4.2.2 Nature-Based Solutions (NbS) Selected

The NbS selected for application in the Água Comprida watershed, i) Rain Gardens, ii) Surface Flow Wetlands, and iii) Bioswales, were chosen for their complementary roles in addressing hydrological processes that are strongly altered in densely urbanized catchments. Urbanization typically increases impervious surfaces, accelerates runoff generation, and elevates peak flows while reducing infiltration and groundwater recharge. In this context, NbS that act at different stages of the runoff pathway can contribute to restoring part of the watershed's natural hydrological functions. These practices provide key ecosystem services related to regulation and provision, particularly flood mitigation, surface water quality improvement, and aquifer recharge (ALVES et al., 2024).

Constructed wetlands are suitable for retaining and treating larger runoff volumes, particularly in downstream areas where flows from multiple drainage areas converge. Rain gardens promote infiltration and pollutant removal and can be integrated into fragmented and highly impervious urban fabrics by occupying residual

spaces, thereby contributing to both hydrological regulation and landscape restoration. Bioswales function as vegetated conveyance channels that facilitate runoff routing while attenuating peak flows that typically occur shortly after rainfall events. When implemented in combination across the watershed, these practices perform complementary hydrological functions, detention, infiltration, and conveyance, supporting a distributed approach to stormwater management rather than relying on isolated interventions.

The selection was also guided by their representativeness among small-scale and medium-scale stormwater NbS commonly applied in urban catchments and by the availability of consolidated design criteria in the literature. Together, these solutions encompass the principal hydrological functions required in the study area while maintaining technical feasibility and social acceptability. Although additional measures, such as green roofs or permeable pavements, could complement this set of practices in other urban contexts, the selected NbS represent a combination capable of addressing the main runoff processes affecting the watershed. Nevertheless, their performance ultimately depends on local implementation conditions and long-term maintenance practices.

4.2.3 NbS - Criteria and Site Selection

Once the NbS types were defined, the next step involved identifying spatial and physical conditions that determine their suitability for implementation within the watershed. The design and siting of NbS must account for site-specific characteristics, including precipitation regimes, soil properties, and topographic conditions. In contexts where local technical standards or design guidelines are unavailable, parameters reported in widely cited literature and technical manuals can provide a reference for preliminary assessments. These criteria typically include minimum distances from building foundations and groundwater tables, maximum slope limits, and recommended dimensions necessary to ensure adequate functionality of each NbS type.

For this study, implementation criteria were identified through a comprehensive review of technical manuals. When discrepancies occurred among sources, priority was given to the values most frequently reported. Criteria cited by only a single reference were retained but explicitly noted as originating from a unique

source. The key implementation criteria adopted for the analyzed NbS are summarized in Table 4-1.

Table 4-1: Key Implementation criteria adopted.

Criteria	Type of restriction	Rain Garden	Wetlands	Bioswales
Slope [%]	Maximum	12 ^{1,2}	15 ^{3,4}	5 ^{5,6,7}
Drainage Area, Da [ac]	Maximum	5 ⁴ (~ 20234,3m ²)	N/A	10 ^{5,8} (~ 40468,6m ²)
	Minimum	N/A	5 ^{3,4,9} (~ 20234,3m ²)	N/A
Distance from foundations [m]	Minimum	3,0 ^{10,11}	6,0 ^{4,12}	3,0 ^{8,10}
Distance from water table [m]	Minimum	1,2 ^{10,2}	N/A*	1,5 ⁸
Width [m]	Minimum	1,5 ¹³	N/A	0,60 ^{13,2}
Length [m]	Minimum	3,0 ¹³	N/A	2,40 ^{13,2}
NbS Area	Minimum	5% Da ^{1,4}	1% Da ^{4,12,6}	1% Da ^{3,7}

*Water table interception within surface flow wetland structures is common and may help maintain permanent inundation. Da: Drainage Area [m²]. N/A: not applicable. Da: Drainage Area [m²].^{(1)Hinman, 2013; 2Prince George's County Department of Environmental Resources, 2010; 3Knox County Tennessee Stormwater Management Manual, 2008; 4EPA, 2021; 5University of Florida, 2008; 6GIZ, 2023; 7Iwaszuk et al., 2019; 8Nacto, 2017; 9Minnesota Pollution Control Agency, 2022; 10Secretaria de Desenvolvimento Urbano e Habitação do Estado de São Paulo, 2024; 11Portland City, 2020; 12Connecticut, 2004 ; 13Eskin et al., 2021). Source: Author (2025).}

While the international criteria in Table 4-1 provide an important starting point and ensure basic functionality, their direct application in the Água Comprida watershed requires consideration of the local tropical urban context. In tropical environments, parameters such as drainage area thresholds and NbS structural dimensions, including length and width, may require adjustments. However, the literature does not provide consensus values for these adaptations. Subsequent hydrological modeling can therefore help evaluate whether the adopted criteria are adequate or whether modifications are necessary to optimize infiltration and retention performance under local conditions.

Guided by these international criteria as a foundational framework, the identification of potential NbS sites faced an additional challenge: the absence of detailed cadastral mapping. This limitation was addressed through visual interpretation of satellite imagery, slope maps (Figure S1), and municipal planning guidelines (PREFEITURA MUNICIPAL DE BAURU, 2022). The lack of precise

groundwater data also required a conservative approach, leading to the adoption of precautionary offsets for some NbS from watercourse banks.

The delineation of potential NbS locations was therefore carried out through manual digitization. This approach enabled careful interpretation of complex urban configurations and allowed the identification of feasible locations that automated procedures would likely overlook given the current data limitations. At the same time, the method introduces constraints related to scalability and potential operator subjectivity. Manual delineation was adopted primarily due to incomplete or inconsistent spatial datasets, such as building footprints and cadastral boundaries, that would otherwise support automated site-selection procedures. Despite these limitations, the resulting dataset provides an initial spatial basis that may support the development of automated approaches in future research.

4.2.4 Computational Routine

The preprocessing routine was developed in *RStudio* (POSIT TEAM, 2024) using geospatial and hydrological packages to extract, process, and organize large datasets required for hydrological modeling with software such as the Storm Water Management Model (SWMM) (US EPA, 2021). The routine operates from a minimum set of input data - Digital Terrain Model (DTM), stream network, land cover, and NbS areas (Table 2) - from which several intermediate products are generated. These include outlets, drainage areas, flow paths, equivalent slopes, flow routing structures, and land cover attributes, which serve as inputs for subsequent hydrological simulations.

In conventional geoprocessing environments (e.g., QGIS and similar GIS platforms), the generation of these intermediate datasets often requires multiple manual processing steps, making the procedure time-consuming and susceptible to operator variability. The implementation of a scripted routine in the R programming environment enables the automation of these steps, facilitating the handling of large datasets and improving consistency in data preparation. Although certain stages – such as the delineation of NbS areas, still require manual input, automated preprocessing reduces the potential for operator bias commonly associated with manual GIS workflows.

The workflow is structured into six integrated modules: i) Outlets, ii) Drainage Areas, iii) Flow Path, iv) Equivalent Slope, v) Flow Routing and vi) Land Cover. A

flowchart detailing the routine and its corresponding outputs is presented in Figure 4-2. Table 4-2 summarizes the required input datasets, their sources, and file formats. Outputs generated by individual modules are subsequently used as inputs for downstream steps; for example, watershed boundaries produced in the Drainage Areas module and flow paths generated in the Flow Path module support later analyses within the workflow.

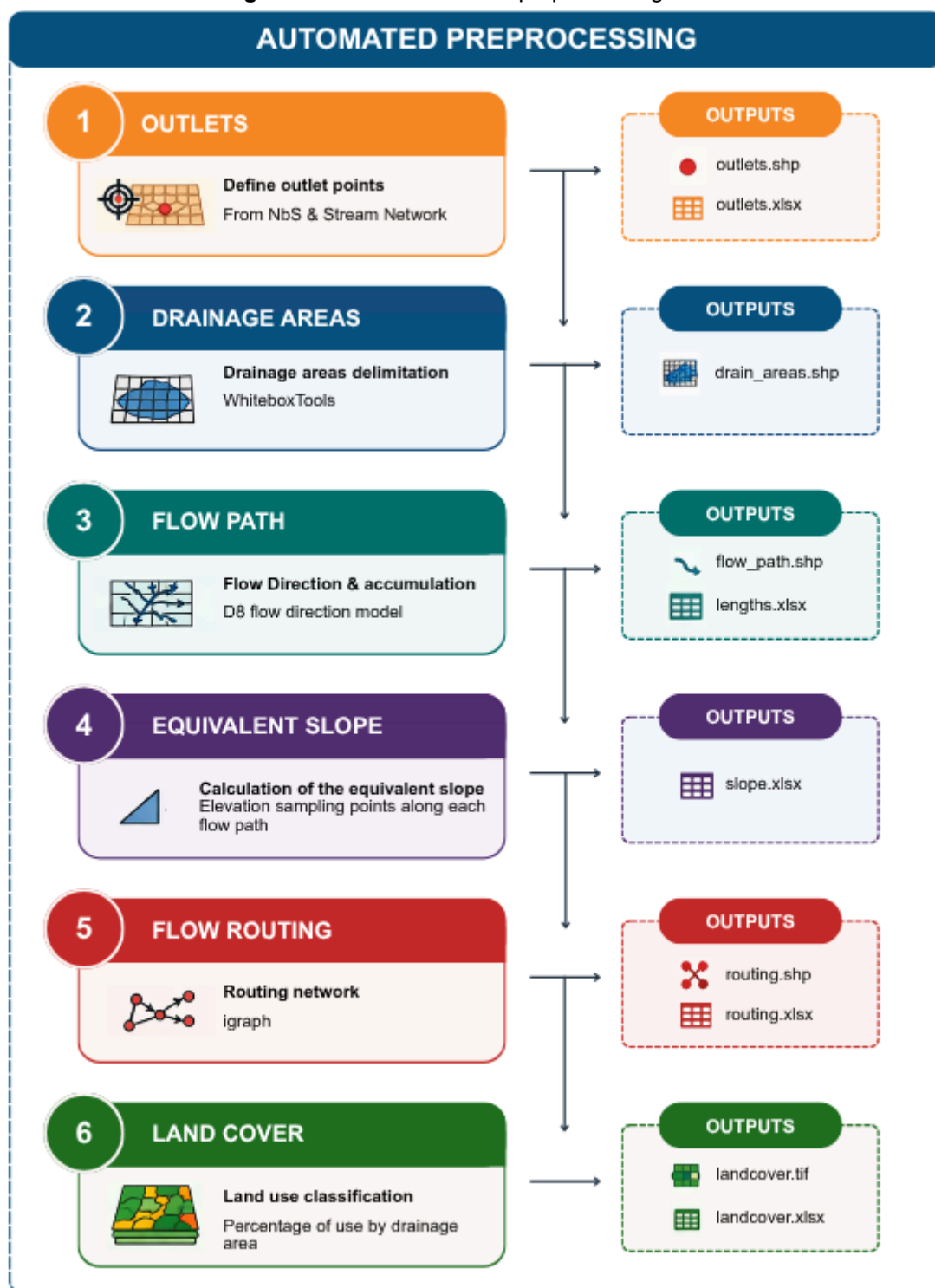
The routine was implemented using the following R packages: *Base R*, *sf*, *raster*, *dplyr*, *whitebox*, *terra*, *igraph*, *purrr*, *writexl*, and *ggplot2*. All scripts are publicly available at the project repository (<https://github.com/anapascucci/preprocessingroutine.git>), while detailed descriptions of the code structure and algorithmic logic were previously established and detailed in Oliveira, Furegatti, and Ruas (2025).

Table 4-2: Input data, description and file format.

Input Data File	Source	File Format	Resolution/Scale
Digital Terrain Model (DTM)	Bauru City Hall ¹	Raster (.tif)	0,5 meters
Stream Network	Bauru City Hall ¹	Vector - Line (.shp)	1:80000
Land Cover	CBERS4A ²	Raster (.tif)	2 meters
NbS Areas	User-defined (digitized in QGIS)	Vector - Polygon (.shp)	1:200 - 1:2000

¹Website for requisition <<https://www2.bauru.sp.gov.br/acessoainformacao/>>. ²Website for acquisition <<https://www.dgi.inpe.br/catalogo/explore>>. Source: Authors (2025).

Figure 4-2: Flowchart of the preprocessing routine.



Source: Authors (2025).

The NbS areas were manually digitized in QGIS based on visual interpretation and overlay with the base layers. The delineation process was performed at approximate visualization scales ranging from 1:200 to 1:2000, ensuring compatibility with the level of spatial detail required for urban-scale analysis. Because the

delineation was conducted manually, these scales represent only the range used during cartographic editing and should not be interpreted as a fixed mapping scale.

Land cover classification was obtained by processing CBERS-04A imagery acquired in August 2025 and processed using QGIS 3.34.10 and Orfeo Toolbox (OTB) software (QGIS DEVELOPMENT TEAM, 2024). Image preprocessing included band stacking, geometric adjustment (superimpose), pansharpening fusion to a spatial resolution of 2 m, and clipping to the study area. The classification followed a Geographic Object-Based Image Analysis (GEOBIA) approach using the Random Forest algorithm available in OTB. In this procedure, the VectorClassifier module assigned final land cover classes to the segmented vector dataset based on a model generated during the TrainVectorClassifier stage, which contained representative samples of each land cover class.

Reference labeling for validation points was established through visual interpretation of high-resolution imagery from Google Satellite dated August 2025, ensuring temporal consistency with the CBERS-04A image used in the classification. This image-based validation approach was adopted due to the logistical complexity and costs associated with conducting field surveys to georeference 1627 validation points distributed across the urban watershed. In this context, the method provided a practical, reproducible, and cost-effective alternative while minimizing potential biases related to landscape changes between image acquisition and field data collection.

Accuracy assessment was performed using AcATaMa, which generated the confusion matrix together with user's accuracy, producer's accuracy, and overall accuracy. At this stage, a summary table containing the percentage values for these accuracy metrics was also produced. Based on the confusion matrix, the Kappa index was calculated according to Equation 1 (CONGALTON, 1991; STEHMAN, 1996; COHEN, 1960).

$$k = \frac{P_o - P_e}{1 - P_e} \quad (1)$$

The computational approach adopted in this study presents methodological limitations that should be acknowledged. Although the Digital Terrain Model used in the analysis has a high spatial resolution (0.5 m), the reliability of hydrological modeling in urban environments is often constrained by recurrent challenges, such

as the limited availability of observational data for parameter calibration and the lack of detailed information regarding the capacity of existing drainage infrastructure (WANG et al., 2018). In addition, the manual digitization of areas designated for Nature-based Solutions, while necessary given current data availability, imposes operational constraints on the scalability of the approach.

To evaluate the influence of topographic data processing and parameterization, a sensitivity analysis was conducted on two key routines within the R workflow: drainage area delineation and flow path extraction. The analysis examined the effect of systematic variations in their principal parameters.

For drainage area delineation, the sensitivity analysis focused on watersheds containing Nature-based Solutions (272 NbS), as these locations already had outlet points defined during the site selection stage. The analysis tested seven values for the *snap distance* parameter, which controls the search radius used to connect outlet points to the flow network: 0, 5, 10, 25, 50, 100, and 200 pixels. The evaluation compared geometric properties of the resulting watershed boundaries, including drainage area and the number of delineated catchments.

For flow path extraction, the *threshold* parameter – defining the minimum flow accumulation required for stream initiation – was tested using eight values: 2000, 1500, 1000, 800, 500, 250, 100, and 50. These tests were applied to all 330 drainage areas (S1 to S330). Sensitivity was assessed by calculating absolute and percentage variations in flow path length between consecutive thresholds, complemented by the coefficient of variation for each watershed across the threshold series to evaluate the internal consistency of the results.

4.3 Results

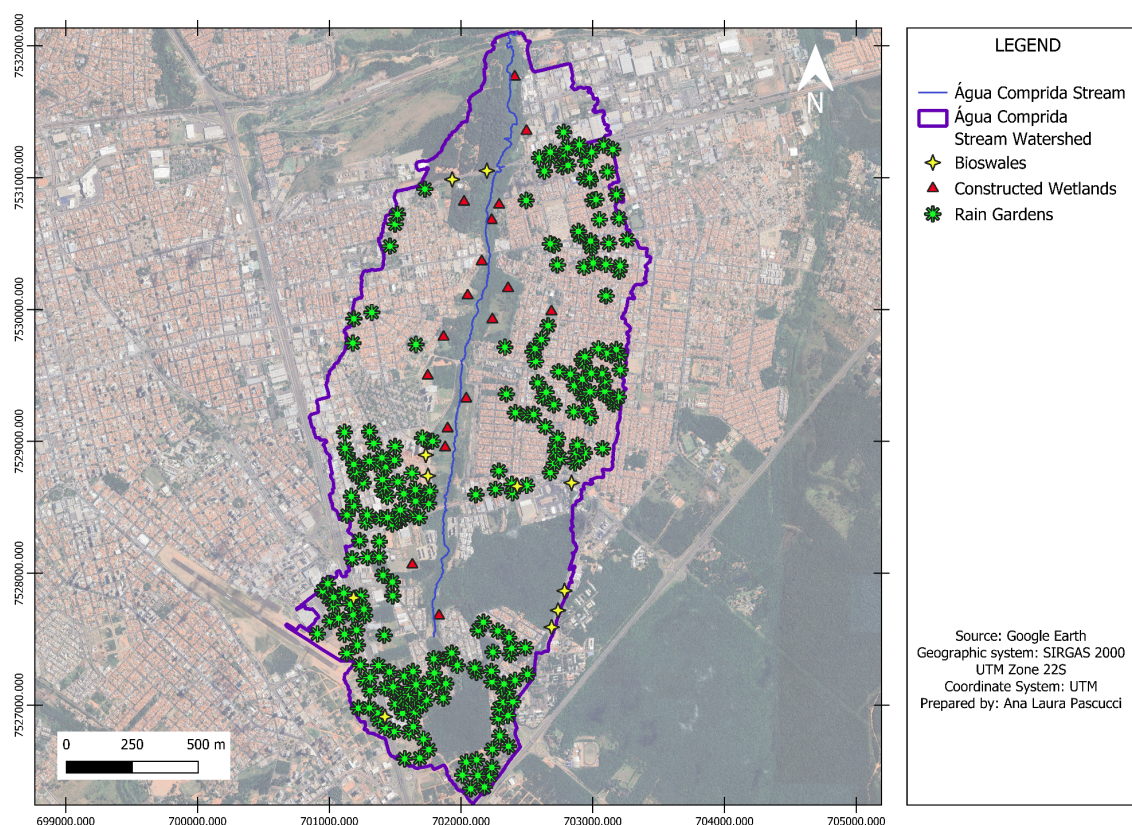
4.3.1 NbS Site Selection in the Água Comprida Stream Watershed

The identification of sites for the implementation of three Nature-based Solutions (NbS): (i) rain gardens, (ii) surface flow wetlands, and (iii) bioswales; within the Água Comprida stream watershed was conducted based on the technical criteria presented in Table 4-1, considering both physical characteristics and the existing urban configuration.

The initial screening focused on urban spaces with potential suitability for NbS implementation, including urban voids, sparsely vegetated areas, public squares, median strips, and undeveloped parcels that could accommodate multifunctional uses as public or green spaces. In highly urbanized areas where open land is scarce, potential locations were also identified within residential developments, gated communities, and commercial properties, where smaller-scale interventions could be integrated into the built environment.

In total, 244 sites were identified as suitable for rain gardens, 17 for surface flow wetlands, and 11 for bioswales. Figure 4-3 illustrates the spatial distribution of these potential implementation sites across the watershed, highlighting how local physical constraints and patterns of urban occupation influenced the allocation of each NbS type.

Figure 4-3: Potential implementation sites for Nature-based Solutions (rain gardens, bioswales, and constructed wetlands) identified in the study area.



Source: Authors (2025).

The spatial analysis revealed distinct distribution patterns for each type of NbS. In sectors characterized by high urban density and limited availability of open space, some rain gardens were designed with larger footprints and implemented in a more concentrated configuration. Although smaller and more distributed units are

generally preferable from a technical and operational standpoint, spatial constraints within the built environment required localized adaptations in certain parts of the watershed.

Municipal urbanization guidelines in Bauru currently establish a minimum standard for public sidewalks consisting of a total width of 3.0 m, composed of a central 1.5 m concrete strip flanked by two grassy strips of 0.75 m each (PREFEITURA MUNICIPAL DE BAURU, 2022). This configuration illustrates how conventional urban design standards may unintentionally limit the implementation of Nature-based Solutions. The predominance of hard surfaces and narrow vegetated strips makes the installation of rain gardens unfeasible along most public sidewalks within the sub-watershed. This result highlights the need to revise urban regulations so that the spatial requirements of green infrastructure are explicitly considered in street design, allowing future streetscapes to incorporate stormwater management functions and contribute to watershed-scale hydrological regulation.

For bioswales, the maximum slope criterion of 5% emerged as the main limiting factor for implementation and the most restrictive technical constraint in the site selection process. Installing bioswales in areas exceeding this slope could compromise hydraulic performance and increase erosion risks, an especially relevant concern in the Bauru region, where soils are highly susceptible to erosion (SANTOS et al., 2018). Although engineering adaptations such as energy dissipaters or stabilization structures could enable installation on steeper slopes, these alternatives were not considered in this study to maintain focus on the direct applicability and cost-effectiveness of bioswales as a core green infrastructure strategy. As a result, because only a limited number of locations within the watershed exhibit slopes equal to or lower than 5%, the application of bioswales was restricted to a small number of sites with suitable topographic conditions.

Surface flow wetlands were primarily located near the Água Comprida stream, where sufficient contributing drainage areas and available physical space were present and where interaction with the groundwater table was possible. The effectiveness of this NbS is closely linked to the maintenance of water levels over time, which depends on local hydrological conditions as well as land-use compatibility and groundwater dynamics. In regions with tropical climates and seasonal drought periods, such as Bauru, evaluating the balance between water

inflows and losses due to evapotranspiration becomes essential to ensure the long-term feasibility of wetland systems.

After defining the potential implementation areas for each NbS type, the hydrological information required for modeling was obtained using automated preprocessing routines developed in RStudio. This step allowed the determination of the effective drainage area associated with each device, generating the necessary inputs for subsequent hydrological simulations and future assessments of NbS performance within the watershed.

Overall, the results indicate that rain gardens present the highest implementation potential within the consolidated urban fabric of the watershed, as they can be integrated into smaller and fragmented spaces throughout the built environment. In contrast, bioswales and surface flow wetlands are more strongly constrained by topographic and hydrological conditions, which limits their spatial distribution. These findings highlight the importance of combining multiple NbS types to address the diverse physical and spatial conditions present across the watershed. Once the potential implementation sites were defined, the automated preprocessing routine was applied to generate the hydrological parameters associated with each NbS location, enabling the extraction of drainage areas, flow paths, and other inputs required for subsequent hydrological modeling.

4.3.2 Hydrological Pre-processing Outputs

The R-based routine integrated topographic, land-use, and hydrological dataset to generate structured outputs for hydrological modeling platforms such as the Storm Water Management Model (SWMM) (US EPA, 2021). The workflow sequentially processes spatial inputs to derive hydrological descriptors required for watershed-scale modeling, including outlet locations, drainage areas, flow paths, equivalent slopes, and flow routing relationships.

The workflow begins with the Outlets module, which automatically identifies the lowest elevation points within each area designated for NbS implementation. In this study, the routine was configured to detect the 20 lowest elevation points within each geometric unit. These outlets served as reference locations for delineating contributing areas in the subsequent Drainage Areas module, enabling the identification of the catchments associated with each NbS location.

A sensitivity analysis was conducted to evaluate the influence of the snap distance parameter on watershed delineation. The results revealed a three-phase behavior. The first phase (0 - 50 pixels) represents a stability stage, in which the maximum number of 272 watersheds is preserved while the mean drainage area increases progressively from 0,41 ha to 2,25 ha. The second phase (75 - 100 pixels) corresponds to a transition stage, characterized by the initial merging of adjacent watersheds (269 - 268 units) and a corresponding increase in mean drainage area (2,36 - 2,50 ha). The third phase (200 pixels) represents a consolidation stage, where a more pronounced reduction in watershed count occurs (259 units) and the mean drainage area increases to 2.84 ha. Statistical analysis indicated a strong positive correlation between snap distance and drainage area ($r = 0,79$) and a negative correlation with the number of delineated watersheds ($r = -0,92$). Based on these results, a value of 50 pixels was selected, as it maximizes drainage area representation while preserving the spatial independence of NbS catchments.

Post-processing analysis revealed that some NbS sites, particularly certain rain gardens, did not meet the minimum recommended drainage area requirements. This limitation results from the combination of relatively large contributing areas associated with the identified outlets and the restricted physical space available for installing additional infrastructure capable of redistributing or subdividing runoff contributions. The spatial relationship between the proposed NbS sites and their respective drainage areas is detailed in Supplementary Material Table S1. Given these constraints, solutions were evaluated according to the effectively available implementation areas, ensuring compatibility with existing land-use conditions while allowing the preprocessing workflow to proceed.

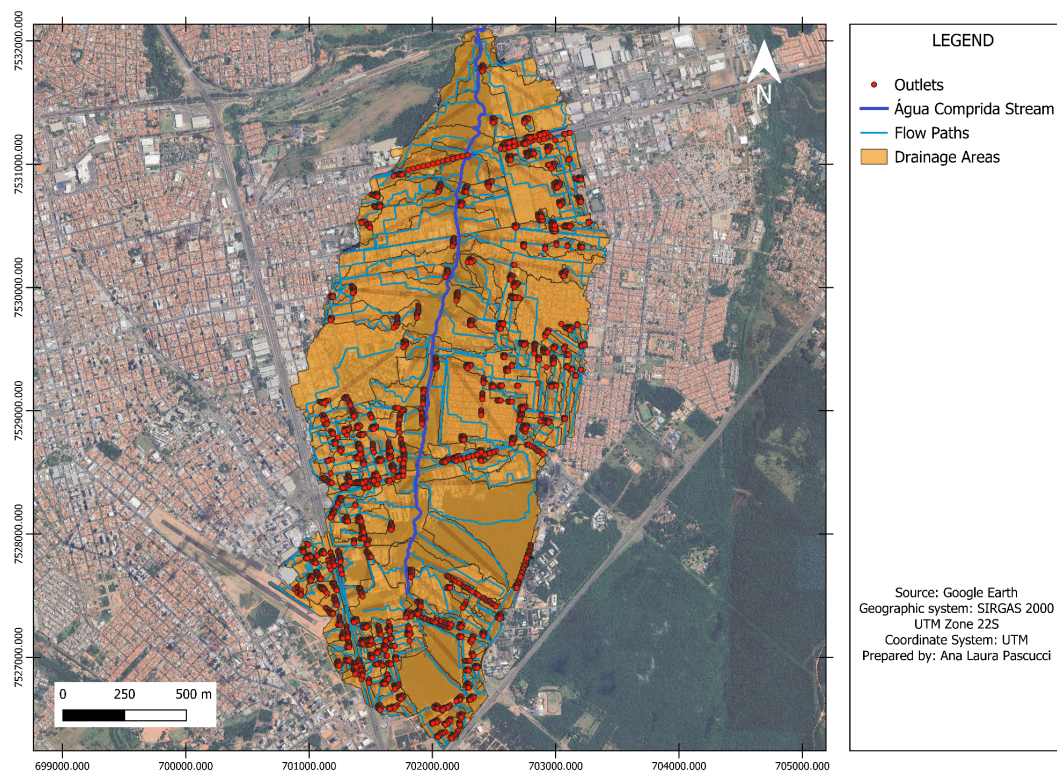
Following drainage area delineation, the Flow Path module was applied to identify preferential runoff pathways within each contributing area. The drainage network was extracted using a minimum contributing threshold of 800 cells, which served as a filtering criterion to represent the hierarchical structure of the local drainage system. Within each catchment, the principal thalweg was defined as the longest channel segment derived from the extracted drainage network. This approach ensures the identification of representative and reproducible flow paths, since longer pathways typically integrate runoff across larger drainage extents.

Figure 4-4 illustrates the outlets, drainage areas, and flow paths delineated by the routine, together with the Água Comprida stream.

To evaluate the robustness of the selected threshold, a sensitivity analysis was performed considering drainage basins of different sizes (small: $<10000 \text{ m}^2$; medium: $10000 - 30000 \text{ m}^2$; and large: $>30000 \text{ m}^2$). The results revealed similar sensitivity patterns across all basin sizes, with comparable absolute variations in thalweg length. This consistency indicates that the observed sensitivity primarily reflects characteristics of the raster-based flow accumulation method rather than differences in watershed scale.

The analysis identified 800 cells as a stabilization threshold for drainage network extraction. The mean absolute variation in thalweg length decreased substantially to 2,84 m for the T1000 - T800 interval and remained similarly low (2,72 m) for the T800 - T500 interval, indicating the emergence of a stable regime of variation. In contrast, higher variation observed for the T500 - T250 interval (3,34 m) suggests that thresholds below 500 cells may introduce instability by capturing small-scale topographic noise rather than consistent drainage pathways. These results support the selection of 800 cells as a robust reference value for representing the drainage network structure while minimizing sensitivity to parameter selection.

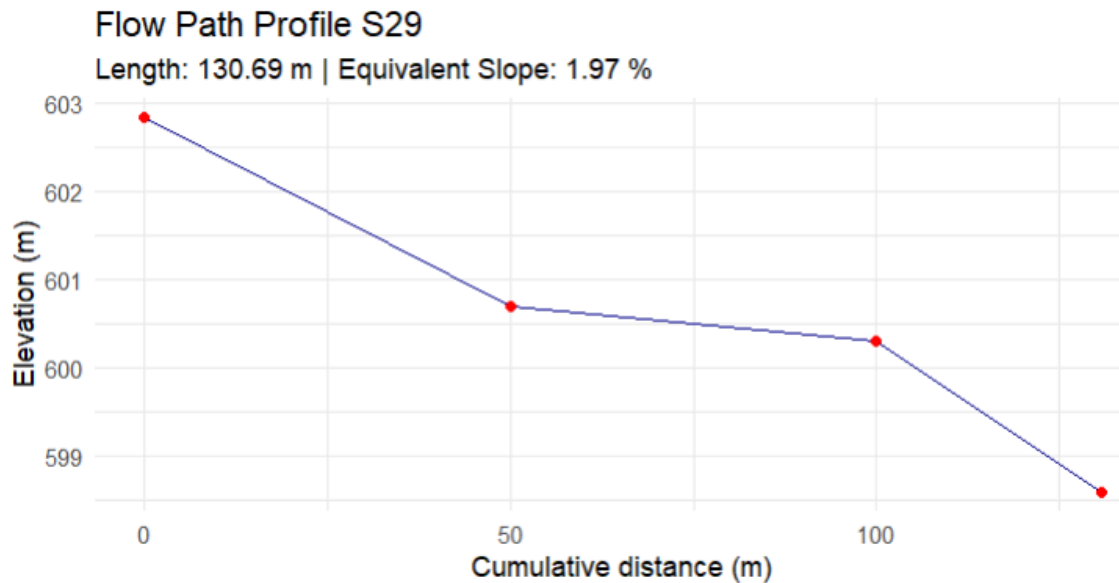
Figure 4-4: Outlets (red points), drainage areas (orange polygons), flow paths (light blue lines), and Água Comprida Stream (central dark blue line).



Source: Authors (2025).

After identifying the flow paths, the Equivalent Slope module calculated equivalent slopes and generated longitudinal profiles for each watershed flow path (Figure 4). The algorithm sampled elevation points at 50 m intervals along each path, extracting elevation values directly from the Digital Terrain Model (DTM) to determine local slope gradients. These elevation data were then applied to the equivalent slope equation, producing a structured dataset containing the calculated slope values for each watershed.

Figure 4-5: Example of a longitudinal profile of the flow path.

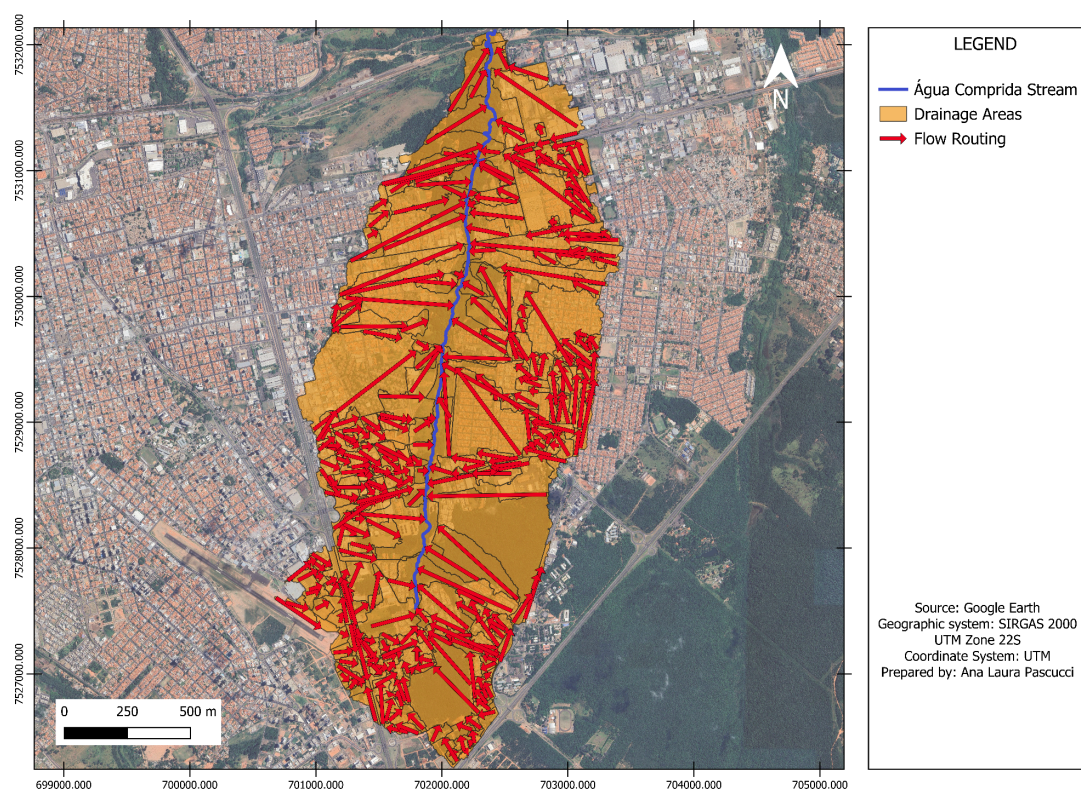


Source: Authors (2025).

Finally, the Flow Routing module identified runoff connections between drainage areas, establishing the hydrological relationships between upstream and downstream units. Using the Digital Terrain Model, delineated drainage areas, and extracted flow paths as inputs, the script performed hydrological preprocessing to determine the origin and outlet points of each path. Runoff movement was simulated cell by cell using the D8 flow direction model. As an output, the module generated a structured table linking each flow path to its corresponding source and destination watershed. Figure 4-6 illustrates the resulting flow routing network, providing a spatial representation of runoff connectivity and flow dynamics across the watershed.

Together, these modules organize the spatial and hydrological information required for watershed-scale analysis, creating a structured dataset that can be directly used as input for subsequent hydrological simulations.

Figure 4-6. Spatial representation of flow routing between river basins, generated from the D8 model. The arrows indicate the direction of surface runoff, connecting the source and destination areas.



Source: Authors (2025).

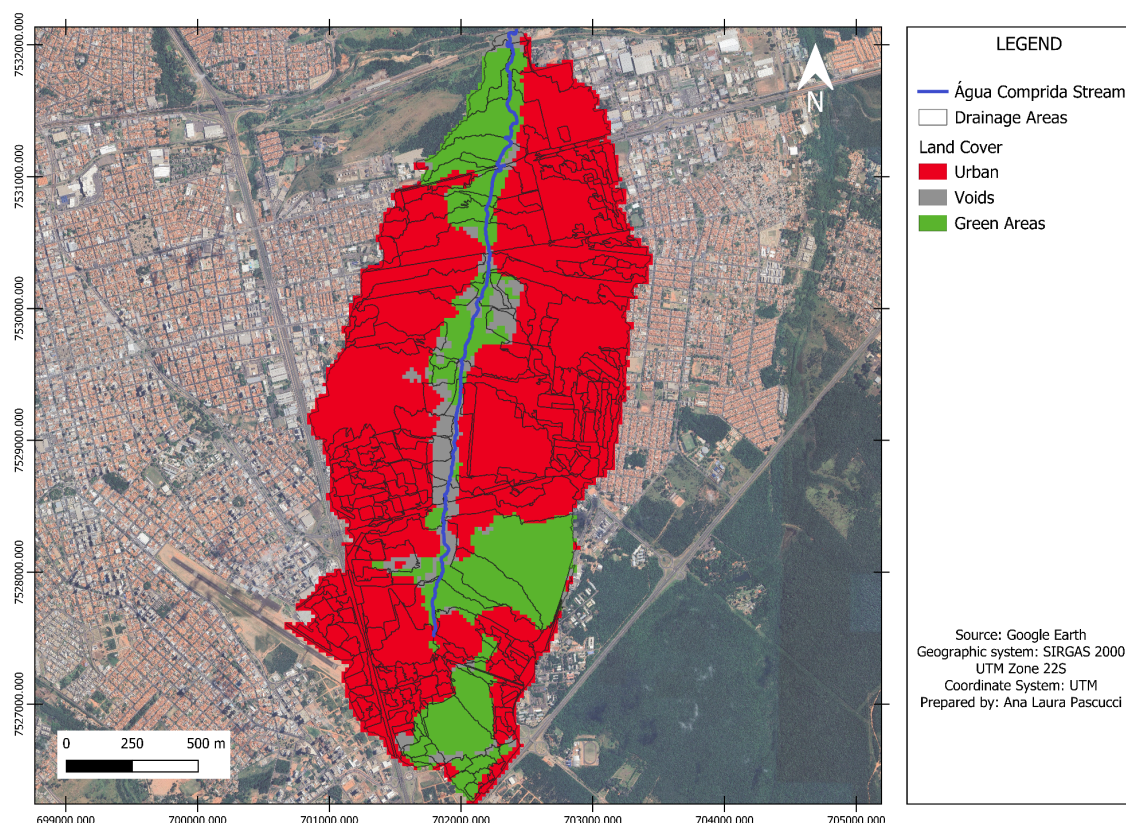
The land cover classification results, processed using the AcATaMa plugin, demonstrated high classification performance, with an Overall Accuracy of 94.31% and a Kappa index of 0.90, indicating an “almost perfect” agreement between the thematic map and the reference dataset (LANDIS and KOCH, 1977). The confusion matrix showed producer’s and user’s accuracies consistently above 80% across all classes, with the *urban* and *vegetation* classes exceeding 94%.

Following validation, the land cover dataset was processed by the Land Cover module to evaluate land cover patterns within the delineated drainage areas. For analytical purposes, the original classes were reclassified into three consolidated categories: urban areas, urban voids - defined as areas with exposed soil or sparse ground vegetation, typically degraded or underutilized - and green areas. The algorithm overlays the land cover raster with the drainage area polygons, calculating both the absolute area and the proportional coverage of each class within each watershed.

As illustrated in Figure 4-7, the spatial distribution of these categories reveals clear contrasts in land occupation across the study area. Highly urbanized areas

account for approximately 62,0% of the total watershed area, while vegetated areas represent about 26,7%, and urban voids comprise roughly 11,2%. The quantitative results are systematically presented in the Supplementary Material (Table S2), whereas Figure 4-7 highlights the spatial fragmentation of urban voids and the presence of green corridors within the watershed.

Figure 4-7: Map of reclassified land cover in drainage areas.



Source: Authors (2025).

4.4 Discussion

The effectiveness of NbS in urban water management has been quantitatively demonstrated in a wide range of climatic, scales and urban contexts. Recent studies consistently report substantial reductions in runoff volume and peak discharge following the implementation of green infrastructure. For example, El Baida et al. (2025), using the Storm Water Management Model (SWMM), reported reductions of 44.7% in peak pluvial flows and 28.95% in fluvial flows in an urban watershed in Morocco [36]. Similarly, Mabrouk et al. (2023) observed a 73.7% reduction in total runoff volume through the combined implementation of Low Impact Development (LID) practices in Alexandria, Egypt, while Rahman et al. (2025) identified reductions of 18.56% in surface runoff associated with green infrastructure implementation in

Seoul, South Korea. Comparable outcomes were reported by Xu and Liu (2018), who observed reductions of approximately 20–25% in runoff within a wetland park in China [37–39]. Collectively, these studies confirm the capacity of NbS to mitigate urban flooding while highlighting the importance of developing planning and site selection methodologies capable of supporting their implementation at the watershed scale.

In this context, the present study proposes an automated workflow that integrates spatial analysis and hydrological preprocessing to support NbS planning in urban watersheds.

4.4.1 Reproducibility and Efficiency of the Automated Workflow

The R-based routine developed in this research enables the systematic delineation of drainage areas, identification of preferential flow paths, and characterization of land cover conditions for each contributing catchment. By integrating topographic, hydrological, and land-use datasets, the approach substantially reduces the time required for preprocessing operations while minimizing errors associated with manual data manipulation. Compared with traditional procedures for watershed delineation and morphometric analysis, which frequently rely on manual interpretation of topographic information, the automated workflow offers a more efficient and reproducible framework for preparing hydrological modeling inputs at watershed scales.

The sensitivity analyses conducted for the drainage area delineation and flow path extraction routines were essential for validating the efficiency of the proposed workflow. The drainage delineation procedure proved particularly sensitive to the snap distance parameter, which controls the spatial tolerance for connecting outlet points to the drainage network. Increasing the snap distance resulted in larger contributing areas but also promoted the merging of adjacent catchments, potentially obscuring the distributed structure of the NbS network. The selected value of 50 pixels represented an appropriate compromise, maximizing the captured drainage area while preserving the independence of individual intervention sites. Similarly, the sensitivity analysis of the threshold parameter used for flow path extraction revealed that drainage basins of different sizes exhibited remarkably similar responses to parameter variation. This behavior indicates that the observed sensitivity is an

intrinsic characteristic of the raster-based flow accumulation method rather than a scale-dependent phenomenon. The identification of 800 cells as a stabilization threshold ensured the extraction of representative thalwegs while avoiding excessive sensitivity to small-scale topographic irregularities.

4.4.2 Spatial Constraints

Beyond the methodological contribution, the results also provide insights into the feasibility of different NbS strategies within the Água Comprida stream sub-watershed. Rain gardens emerged as the solution with the greatest potential for implementation, primarily because their installation requires relatively modest spatial areas and can be integrated into highly urbanized environments. This finding is consistent with previous research that identifies rain gardens as particularly adaptable solutions for consolidated urban landscapes due to their flexible design requirements and relatively low construction complexity (ALVES et al., 2024; ÁVILA, AMARIS and BUELVAS, 2016). Constructed wetlands represented the second most viable NbS option, followed by bioswales. However, both alternatives typically require more restrictive environmental conditions, including suitable slope gradients, available surface area, and favorable groundwater conditions, which can limit their applicability in dense urban contexts (MELL, 2010; PANKHURST, 2012). Nevertheless, when implemented at key hydrological locations, these solutions can complement distributed rain garden systems and enhance overall watershed-scale stormwater management.

Despite the potential identified for NbS implementation, several constraints emerged during the site selection process. One important limitation relates to the absence of consolidated technical guidelines for NbS design and implementation in the study region. The establishment of technical parameters required consultation of multiple international sources, which revealed inconsistencies between recommended criteria and gaps in guidance for certain design variables. This regulatory fragmentation underscores the need for regionally adapted technical manuals capable of addressing local environmental and urban conditions while ensuring equitable access to the benefits associated with NbS implementation (COOK et al., 2025; MCPHEARSON et al., 2025). The challenge is particularly relevant in tropical regions, where the development of locally adapted NbS design

standards remains limited despite the growing international body of knowledge on the subject.

Spatial constraints associated with existing urban density also influenced the feasibility of NbS implementation. In several locations, rain gardens did not meet minimum recommended area requirements because relatively large contributing drainage areas coincided with limited available surface space for complementary interventions capable of redistributing runoff. This limitation may compromise the effectiveness of a fully distributed NbS strategy and lead to a more concentrated configuration of interventions within the watershed. Furthermore, the manual procedure adopted for site identification, although necessary given current data availability, introduces a degree of subjectivity and limits the scalability of the methodology. Similar challenges related to land availability and terrain conditions have been reported in previous studies addressing NbS site selection in urban environments (MCPHEARSON et al., 2025; SARABI et al., 2019). Future applications would therefore benefit from the development of automated spatial algorithms capable of identifying suitable NbS locations based on standardized technical criteria.

In addition to technical and spatial constraints, the successful integration of NbS into urban stormwater management systems depends strongly on the existence of supportive institutional and regulatory frameworks. Several international experiences illustrate how policy instruments can accelerate the widespread adoption of green infrastructure solutions. In Toronto, Canada, the implementation of the Green Roofs Bylaw established mandatory requirements for vegetated roofs in new developments, accompanied by municipal financial incentives, resulting in measurable reductions in surface runoff [46]. Similarly, the Sponge City Program implemented in Wuhan, China introduced planning guidelines requiring urban developments to incorporate strategies for infiltration, retention, and natural water purification (PENG and REILLY, 2021). Monitoring results from pilot areas indicate increases in stormwater infiltration rates of up to 70%, reductions in runoff volumes of approximately 60%, and improvements in urban water quality. These experiences demonstrate that the large-scale consolidation of NbS depends not only on the availability of suitable technologies but also on the institutionalization of nature-based approaches within urban planning and environmental governance frameworks

[9;48;49]. (CASTELO, AMADO and FERREIRA, 2023; DUSHKOVA and HAASE, 2020; MOSISA et al, 2025).

4.4.3 Local Regulatory Challenges

The spatial limitations identified in this study also highlight the need to incorporate NbS considerations directly into urban planning instruments. Although the methodology proposed here allows the identification of potential implementation sites within the constraints imposed by existing regulations, the results indicate that certain NbS solutions remain restricted by current land-use norms and infrastructure configurations. For instance, the limited width of sidewalks and road medians in many areas constrains the installation of rain gardens and bioswales. These findings provide practical evidence that urban planning codes may need to evolve to explicitly accommodate green infrastructure elements, thereby transforming NbS from isolated interventions into integral components of urban water management systems (VENUTI, 2025).

Overall, the workflow developed in this study demonstrates significant potential for supporting watershed-scale NbS planning. The methodology is replicable and transferable to other urban watersheds, provided that appropriate input datasets and parameter values are adopted according to the spatial resolution, study scale, and research objectives. In the subsequent hydrological modeling phase, which will employ the preprocessing outputs generated in this research, additional sensitivity analyses will be conducted to evaluate how variations in key model parameters influence simulation outcomes, including runoff volumes and flow accumulation patterns. Future research may further enhance the proposed approach by developing automated procedures for NbS site selection and applying the methodology to watersheds with different climatic, geomorphological, and urban characteristics.

4.5 Conclusions

The results show that the integration of automated hydrological preprocessing with spatial analysis substantially improves the identification and evaluation of Nature based Solutions at the watershed scale. The proposed workflow produced consistent and structured inputs for hydrological modeling while enabling the analysis of multiple distributed interventions across an urban catchment. Parameter sensitivity analysis

demonstrated that appropriate calibration is essential to preserve drainage network integrity and avoid distortions in catchment representation. The spatial analysis further indicated that rain gardens present the greatest implementation potential in dense urban areas, whereas wetlands and bioswales are more restricted by topography, space availability, and local environmental conditions. These findings highlight the importance of combining distributed small scale interventions with more localized systems in order to effectively address runoff processes across the watershed.

The results also reveal that spatial constraints and existing urban configurations limit the direct application of standard design criteria, particularly where large contributing areas coincide with restricted installation space. This emphasizes the need for flexible and context sensitive planning approaches capable of accommodating diverse urban and environmental conditions. In addition, the spatial distribution of suitable sites exposed regulatory constraints that restrict the integration of Nature based Solutions into current urban planning practices. The framework developed in this study provides spatially explicit evidence to support planning decisions and facilitate the incorporation of Nature based Solutions into urban water management strategies, contributing to more effective and scalable implementation at the watershed scale.

4.6 References

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Chapter 5:

*Integrated Management of Nature-Based Solutions: Planning and Hydrological Performance in a Tropical Urban Watershed*⁵

Abstract. Nature-Based Solutions (NbS) are sustainable alternatives to traditional drainage systems, aiming to mitigate urban runoff, enhance infiltration, and improve water quality. While their benefits are well recognized at the local scale, there is still limited understanding of their cumulative hydrological effects when applied across broader urban catchments. This study aims to assess the hydrological performance of three types of NbS – rain gardens, constructed wetlands, and bioswales – implemented in a distributed manner across the Água Comprida Stream sub-watershed in Bauru, São Paulo, Brazil. A hydrological model was developed incorporating detailed spatial and physical parameters derived from remote sensing and GIS-based pre-processing. The sub-watershed, covering 9.35 km², was divided into 330 drainage units, of which 271 were assigned to NbS based on spatial availability and technical feasibility. Two rainfall events were simulated: a real event and a 10-year design storm. The model includes Horton infiltration parameters, Manning-based surface runoff calculations, and dynamic routing between subcatchments. Due to the lack of observed hydrological data, a sensitivity analysis was conducted, identifying imperviousness and rain gardens' berm height as the primary governing parameters for the system's hydrological response. Results indicate that NbS effectiveness is scale-dependent, where downstream volumetric efficiency may conceal significant local-level improvements in hydrological functions. The integrated scenario demonstrated a hydraulic synergy effect, optimizing total retention through the serial arrangement of units. Despite the inherent uncertainties and the dynamic performance under hydrological stress, this study underscores the need for strategic spatial distribution to provide more robust guidelines to support planning and prioritization of areas for NbS implementation in consolidated urban contexts.

⁵The contents of this chapter are intended for publication. Selected strategic and specific papers will be submitted to two conferences: 1. The 16th National Conference on Urban Water (ENAU) and the 6th Symposium on Urban River Revitalization (SRRU)(<https://www.abrhidro.org.br/xvienau/>); 2. The 5th International Congress on Environmental Engineering (<https://www.5ciea.com.br/>).

Keywords: Nature-based Solutions, Urban Watershed, Storm Water Management Model, Urban Planning, Hydrological Modelling.

5.1 Introduction

The Nature-based Solutions (NbS) framework emerged as a multifunctional and sustainable approach to urban water management, offering a viable alternative to conventional grey infrastructure. Conventional stormwater drainage systems, designed for rapid runoff discharge, fundamentally modify hydrological regimes and degrade water quality (ROY et al., 2008). As sustainable alternatives to conventional infrastructure, Nature-based Solutions (NbS) contribute to a more resilient urban water cycle and enhance water resource management by promoting natural stormwater infiltration, storage, and treatment while simultaneously delivering environmental, social, and economic benefits (EEA, 2012; PACETTI et al., 2022; BECEIRO et al., 2020; AHIABLAME et al., 2012).

Despite the growing recognition of the hydrological and environmental effectiveness of Nature-based Solutions (NbS), their implementation at broader territorial scales remains underexplored. According to Roy et al. (2008), this scenario is influenced by institutional and operational challenges, including the lack of consolidated technical standards and guidelines in the local context and uncertainties regarding the performance of these solutions at the watershed scale. Most hydrological performance studies of NbS focus on localized interventions, such as streets, plots, parcels, or neighborhoods, limiting the understanding of the cumulative effects of NbS in watersheds (SU et al., 2023; CASTELO, AMADO & FERREIRA, 2023; ZHANG et al., 2021; KABISCH et al., 2016). While local-scale modeling is valid for project parameterization and understanding individual solution performance, analyses at larger watershed scales enable the evaluation of an integrated NbS system (LISENBEE et al., 2021).

In this context, studies such as Loperfido et al. (2014) demonstrate that the hydrological effectiveness of NbS is significantly enhanced when implemented in a distributed and integrated manner at the watershed scale. Comparing different spatial arrangements of practices like green roofs, rain gardens, permeable pavements, bioretention, riparian barriers, cisterns, and bioswales, the authors observed that distributed configurations led to substantial improvements in hydrological regimes.

Reinforcing this perspective, the research by Tansar, Duan & Mark (2022) found that a uniformly distributed implementation strategy across urban watersheds performed better than strategies concentrated in specific sectors in terms of reducing surface runoff, peak flow, and flood volume.

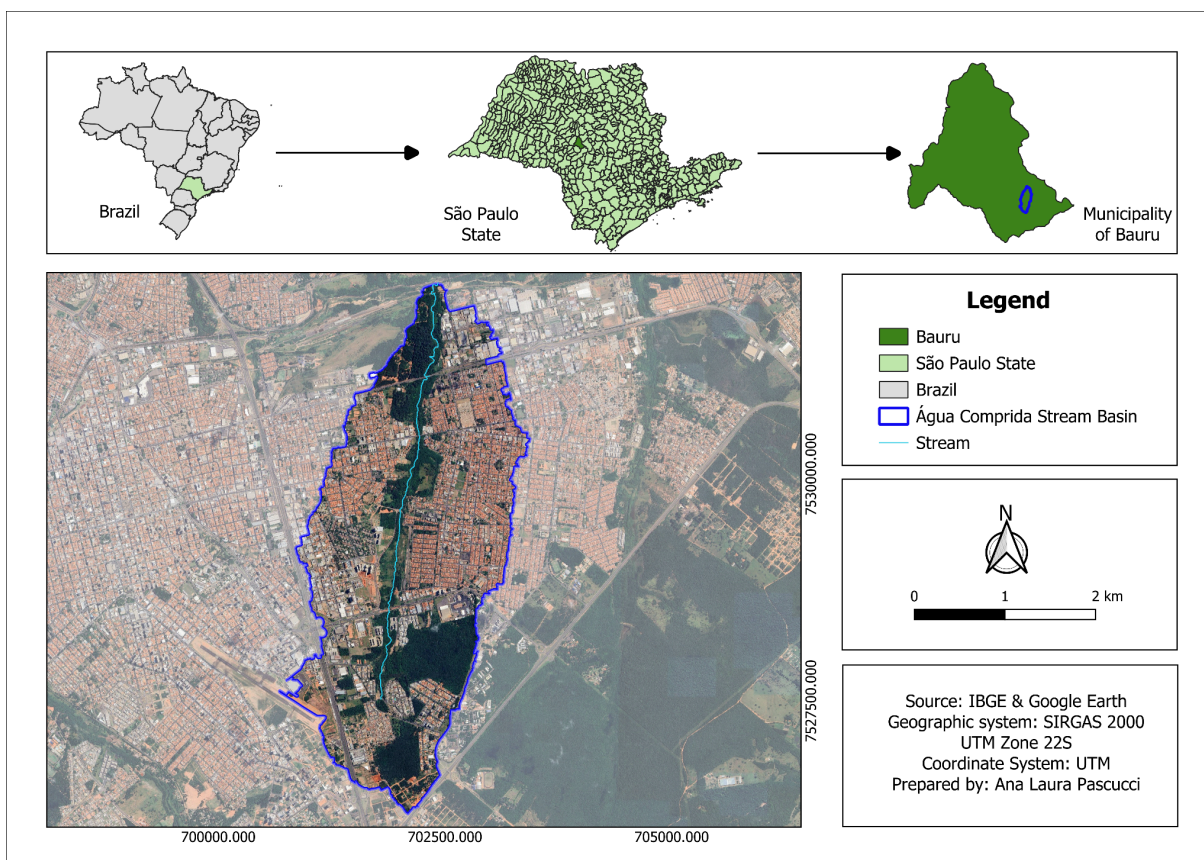
Despite promising evidence, the scarcity of large-scale assessments still hinders decision-making and the formulation of guidelines tailored to local particularities. Additionally, challenges such as limited implementation in consolidated urban areas restrict the large-scale adoption of NbS (ROY et al., 2008; DEMUZERE et al., 2014). In this scenario, hydrological modeling emerges as a fundamental tool to support planning decisions, allowing the quantification of hydrological impacts of existing or proposed interventions and the prioritization of strategic areas for NbS implementation in watersheds (BRAUMAN et al., 2021).

Given these gaps, this study proposes hydrological modeling at the sub-watershed scale to evaluate the performance of three types of NbS, rain gardens, constructed wetlands, and bioswales, in terms of water infiltration and storage. This study evaluates the impact of NbS on the watershed's water balance and hydrological response, assessing both individual and integrated scenarios. By adopting a multiscale approach, analyzing performance from the local drainage unit to the watershed outlet, the study identifies hydraulic synergies and functional specializations among different typologies. This systemic perspective aims to advance the understanding of how strategic NbS distribution influences hydrological recovery, providing a technical basis for incorporating nature-based interventions into urban water management policies and practices.

5.2 Materials and methods

5.2.1 Study Area

The study area is the Água Comprida Stream sub-watershed, located within the urban perimeter of the municipality of Bauru, in the central-western region of São Paulo State, Brazil (Figure 1). The Água Comprida Stream is approximately 4.92 km long, with a catchment area of around 9,35 km², and is a tributary of the Bauru River, one of the main rivers in the municipality.

Figure 5-1: Location Map of the Study Area.

Source: Authors (2025).

The region's climate is classified as tropical highland (Cwa) according to Köppen (1948), with rainfall concentrated in the summer. The area features Dark Red Latosol soils (corresponding to Ferralsols in the World Reference Base for Soil Resources – WRB), which are highly susceptible to erosion (SANTOS et al., 2018; BAURU, 2015). These factors, combined with intense urbanization, increase the watershed's susceptibility to degradation processes linked to surface runoff, both due to higher flow volume and velocity and the risk of water quality deterioration, commonly observed in heavily impermeable areas. This scenario reinforces the selection of the area as a case study for evaluating sustainable urban drainage solutions.

5.2.2 Input Data, Parameters, and Hydrological Model Configuration

Prior to entering input data into the hydrological model, areas for NbS implementation were selected based on construction criteria and spatial availability within the sub-watershed, followed by preprocessing of spatial data. The preprocessing aimed to extract, structure, and organize information on: outlets, drainage areas, flow path, equivalent slope, flow routing and land cover (urban areas,

green areas and areas with exposed soil or sparse ground vegetation, typically degraded or underutilized, referred as urban voids) using a custom script developed in RStudio (Oliveira, Furegatti & Ruas, 2025), as described in Chapter 3 and Chapter 4. The input data consisted of the Digital Terrain Model (DTM), delineated watercourses, land-use raster data, and polygons representing the designated NbS implementation areas.

The Água Comprida Stream sub-watershed was subdivided into drainage areas to assess the hydrological response of smaller, more specific zones within the overall watershed. This delineation allows for more detailed modeling of the impacts of installed NbS on runoff volumes, water retention, and infiltration capacity within their respective contributing areas. Drainage areas without NbS were generated in QGIS using a threshold of 2000000 cells to create larger drainage zones in regions without NbS, which were then integrated with the other areas using QGIS geoprocessing tools (clip and dissolve functions) (QGIS DEVELOPMENT TEAM, 2024). The sub-watershed was divided into 330 drainage areas, of which 244 were associated with rain gardens, 17 with constructed wetlands, 11 with bioswales, and 59 without NbS. This subdivision enables the generation of a distributed hydrological model and a more precise evaluation of the local hydrological response to NbS interventions.

For hydrological modeling in the Storm Water Management Model (SWMM) version 5.2 (US EPA, 2021), five distinct scenarios were considered to evaluate the impact of NbS on water management (Table 5-1). Baseline Scenario represents the current conditions of the sub-watershed, without any NbS interventions. Integrated Scenario incorporates three types of NbS combined – rain gardens (RG), constructed wetlands (CW), and bioswales (BW) – installed in locations previously identified as physically feasible based on the adopted criteria. The definition of these scenarios allows for a comparison between current conditions and those resulting from NbS implementation, facilitating an analysis of their effects on urban water management. Three additional scenarios (RG Scenario, CW Scenario and BW Scenario) were also simulated, each considering the implementation of only one NbS type within the study area (rain gardens only, constructed wetlands only, and bioswales only). These scenarios were designed to analyze, in general terms, the influence of the integrated scenario and, more specifically, its performance in terms of the efficiency per unit of NbS installed.

Table 5-1: Scenarios and boundary conditions for hydrological modeling.

Scenario	NbS Representation	Infiltration Mechanism	Storage Mechanism	Impervious Routing to Permeable Areas
Baseline	None	Horton (Native Soil)	Depression Storage (Dstore-perv and Dstore-imperv)	0% (Routed Direct to Outlet)
RG	Bioretention Cell	LID Percolation (Soil Conductivity+ Seepage Rate + Horton)	Berm height, Soil and Storage thickness, Void ratio	100% (Routed to Rain Gardens)
CW	Storage Unit	Storage Seepage Loss + Horton	Max. depth; Surface area; Seepage rate	100% (Routed to Wetlands)
BW	Bioretention Cell	LID Percolation (Soil Conductivity+ Seepage Rate + Horton)	Berm height, Soil and Storage thickness, Void ratio	100% (Routed to Bioswales)
Integrated	Combined	Combined mechanisms	Combined mechanisms	100% (Routed to NbS)

Source: Authors (2025).

For rain gardens and bioswales, the “Bioretention Cell” option available in the SWMM LID Control module was used, while constructed wetlands were modeled using storage units to integrate their specific conditions into the software, following the methodology adopted by Sauer et al. (2024). Although bioswales are commonly represented in SWMM as vegetative swales, this structure does not explicitly simulate infiltration or subsurface storage. Therefore, bioswales were represented as bioretention cells in order to assess infiltration and water storage processes, based on their physical construction characteristics.

The Horton infiltration model was adopted for this study, which accounts for the exponential decrease in infiltration rate from an initial to a final value over time (Rossman, 2016). The model parameters were defined based on experimental data obtained by Pertussatti (2011), with specific equations for Brazilian Dark Red Latosols (classified as Ferralsols in the World Reference Base for Soil Resources – WRB), the predominant soil type in the watershed, both with and without vegetation cover (SANTOS et al., 2018). Table 5-2 presents the Horton infiltration model parameters used for Dark Red Latosols, distinguishing between vegetated and non-vegetated areas. The drying time, which corresponds to the time required for the soil to recover its infiltration capacity between rainfall events, was set based on the

SWMM manual recommendation, suggesting values between 2 and 7 days (ROSSMAN, 2016). The infiltration values with vegetation cover were assigned to drainage areas where vegetation predominated over exposed surfaces. Conversely, in areas with a higher proportion of bare soil or non-vegetated cover, the parameters corresponding to conditions without vegetation cover were adopted. In urban areas, infiltration was disregarded, following the assumption of complete impermeability.

Table 5-2: Horton infiltration model parameters adopted.

	Max infiltration rate (mm/h)	Min infiltration rate (mm/h)	Decay constant (h ⁻¹)	Drying time (days)
With Vegetation Cover	55,68	41,24	18,3	5
No Vegetation Cover	53,68	27,75	15,42	5

Source: Authors (2025).

In addition to infiltration parameters, the SWMM hydrological model requires the definition of specific variables related to surface runoff, which influence both the volume and velocity of flow. To estimate surface runoff, the method based on Manning's equation was adopted, which considers the resistance offered by the surface to flow through roughness coefficients. Among these coefficients, the values for permeable and impermeable areas stand out, called *N-perv* and *N-imperv* respectively, which represent roughness, i.e., the resistance to flow on these different surfaces. This study used *N-perv* values of 0.09, corresponding to dense grass, and *N-imperv* values of 0.04, referring to dense residential land use (YEN, 2001). The initial water storage in surface depressions, before the onset of surface runoff, represented by *Dstore-perv* and *Dstore-imperv* for permeable and impermeable surfaces respectively. For this study, values of 2.5mm were adopted for *Dstore-perv* and 0.0625mm for *Dstore-imperv* (Rossman, 2016). Additionally, the *Width* parameter was defined, representing the equivalent width of the main thalweg within each sub-watershed, directly influencing the velocity and concentration time of surface runoff. This parameter was calculated as the ratio between the drainage area and the length of the main thalweg of each contribution unit, following the SWMM manual's recommendation (Rossman, 2016).

Furthermore, the model requires the definition of the *Subarea Routing* parameter, which determines the routing of runoff generated in impermeable areas.

This parameter was configured for all subcatchments as Outlet, so that surface runoff is directed directly to the sub-watershed outlet point and then routed to the corresponding destination sub-watershed. This connection is defined in the SWMM *Outlet* parameter, where the names of receiving sub-watersheds are specified for each contribution unit. The *PctZero* and *PctRouted* parameters complement this configuration: the first represents the percentage of impermeable area without initial storage, adopted as zero in all scenarios, simulating smooth surfaces, while the second indicates the fraction of impermeable runoff that is routed to permeable areas. In Scenario B, without NbS, the adopted *PctRouted* value was zero, representing no connection with permeable areas or NbS. In Scenarios with NbS interventions (Integrated, Scenario CW, Scenario BW, Scenario RG), a value of 100% was adopted for *PctRouted*, simulating the complete routing of runoff to infiltration structures.

The Dynamic Wave routing method was adopted, which allows simulation of storage, overflow and detention time in reservoirs (Rossman, 2010). This method solves the complete Saint Venant equations, being more suitable for representing hydraulic structures and their interactions with surface flow.

Since detailed geometric data for the Água Comprida Stream channel were not available, its hydraulic behavior was not explicitly represented in the model. Given that the primary focus of this study is the evaluation of NbS performance in the sub-watersheds of the stream, a simplified representation of the receiving watercourse was adopted. To represent the Água Comprida Stream in SWMM, the Ponded Area parameter was assigned to the junction nodes. This approach was used to avoid mass losses caused by node flooding, ensuring the conservation of the overall water balance within the model. By allowing excess water to temporarily remain ponded at the channel nodes, the simulated outflow volumes remain conservative and representative.

The preprocessed data, along with the infiltration and runoff parameters, were added to SWMM using the *Generate SWMM INP* plugin as inputs for the hydrological model. The model was configured with flow units in CMS (m^3/s), and all data were provided in compatible metric system units.

5.2.3 Rainfall Data Processing and Event Definition

For rainfall event simulations, two distinct scenarios were considered: a design rainfall calculation and real high-intensity rainfall event. The design rainfall was defined based on the Intensity-Duration-Frequency (IDF) equation, developed from data provided by the Meteorological Research Institute (IPMET) for the municipality of Bauru, which provides historical data at 5-minute intervals. The historical series used cover the hydrological years from 2004 to 2025, corresponding to 22 years of sub-daily precipitation data.

According to the World Meteorological Organization (WMO), in order to quantify rainfall and select intense events, the historical series must have a minimum period of 30 consecutive years of observations (WMO, 2017). However, such long data series are not always available, and it is common in meteorological studies to use data series with observation times of less than 30 years. Furthermore, even when they do exist, rain gauges sometimes present rainfall data with many gaps (CASTILLO E SENTIS, 1996, GONÇALVES, 2011). Due to the difficulty in obtaining high-resolution pluviographic data, most studies in Brazil are conducted using historical rainfall series shorter than those recommended in the literature. Consequently, this limitation indicates that the use of Intensity–Duration–Frequency (IDF) equations derived from short time series should be associated with conservative return periods to ensure the reliability of the estimates (SILVA et al., 2002; OLIVEIRA et al., 2008; SILVA et al., 2012; TORRES et al., 2025).

To define the IDF equation, events were initially separated considering a dry period of 6 hours, and for each event, the highest accumulated precipitation was extracted for durations of 5, 10, 15, 20, 25, 30, 60, 120, 360, 720, and 1440 minutes. With the accumulated precipitation of the events with the highest precipitation, the intensities were calculated and the Annual Maximum Series (AMS) were constructed for each duration. Next, the intensities of the AMS were adjusted by the Generalized Extreme Value (GEV) distribution to estimate the rainfall intensities for return periods of 2, 5, 10, 25, 50, 100, 200, 500, and 1000 years. The GEV encompasses the three asymptotic forms of extreme value distribution: Gumbel (Type I), Fréchet (Type II), and Weibull (Type III), used to represent the distribution of extreme values in different fields, mainly in studies of maximum annual daily precipitation (JENKINSON, 1955;

NAGHETTINI and PINTO, 2007). The Probability Density Function is given by Equation 2.

$$FDP = f(x) = \frac{1}{\sigma} \cdot \left[1 - \xi \cdot \left(\frac{x - \mu}{\sigma} \right) \right]^{\left(\frac{1+\xi}{\xi} \right)} \cdot \exp \left\{ - \left[1 - \xi \cdot \left(\frac{x - \mu}{\sigma} \right) \right]^{\frac{1}{\xi}} \right\} \quad (\text{Eq. 2})$$

The GEV distribution is defined by three parameters: shape (ξ), scale (σ) e position (μ). The the sign and magnitude of ξ controls the asymptotic behavior of extreme values. When $\xi < 0$, the GEV corresponds to a Type III (Weibull) distribution and is defined for values of $x < \mu - \sigma/\xi$. For $\xi > 0$, the distribution follows a Type II (Fréchet) form and is defined for values of $x > \mu - \sigma/\xi$. In the limiting case $\xi = 0$, the GEV reduces to the Gumbel (Type I) distribution. For each rainfall duration, the shape, scale, and location parameters were estimated using the *fevd* function from the *extRemes* package of the R software.

To assess the adequacy of the GEV distribution to the maximum precipitation data for each analyzed duration, graphical procedures and a statistical goodness-of-fit test were employed. The graphical analysis was conducted using probability–probability (PP) plots and quantile–quantile (QQ) plots. The PP plot indicated good overall agreement between empirical and theoretical probabilities, with only minor deviations across the probability range, suggesting satisfactory model performance in representing the overall distribution. For higher quantiles, corresponding to the rarest events in the sample, the QQ plot showed a progressive deviation of the points from the reference line, particularly for the series associated with longer durations. This behavior indicates greater uncertainty in the representation of the upper tail of the distribution and is consistent with the presence of isolated extreme events and the limited sample size.

Additionally, the Anderson–Darling (AD) test was applied at a 5% significance level, using the critical values of the Gumbel distribution as reference (BESKOW et al., 2015). The AD test results indicated acceptance of the GEV fit for 9 out of the 11 analyzed durations, with the exception of two intermediate and long durations (120 and 360 minutes, respectively), for which more pronounced deviations in the upper tail were identified.

Overall, the graphical and statistical results indicated that the GEV distribution was able to adequately represent the behavior of maximum precipitation extremes for the different analyzed durations. The cases of rejection were interpreted as a

reflection of the high natural variability of the series and the uncertainties associated with extrapolation to high return periods, without compromising its applicability in the development of IDF curves.

Finally, the parameters of the IDF equation were estimated by fitting a linear regression model to the logarithmically transformed variables. The original nonlinear equation was linearized by applying natural logarithms, resulting in Equation 3.

$$I = \frac{a \times TR^b}{(t_d + c)^d} = \frac{1028,132 \times TR^{0,1683}}{(t_d + 6,987)^{0,759}} \quad (\text{Eq. 3})$$

Where: I is the rainfall intensity (mm/h); TR is the return period (years); t_d is the precipitation duration (minutes); a , b , c , d are location-specific coefficients calibrated for Bauru municipality (COLLISCHONN and DORNELLES, 2013).

The quality of the IDF equation fitted by logarithmic regression was evaluated using statistical metrics calculated on the original scale of precipitation intensities. The high coefficient of determination ($R^2 = 0,97$) indicates that the model explains most of the variability in the estimated intensities, demonstrating strong agreement between observed and modeled values. The mean relative error (9,9%) and the mean absolute error (11,4 mm h⁻¹) indicate discrepancies compatible with the uncertainties inherent to the empirical fitting of IDF relationships based on extreme precipitation data. Similar magnitudes of MALE, RMSE (18,5 mm/h) and NRMSE (14,5%) have been reported in previous IDF studies conducted with sub-daily rainfall data in Brazil (GARRIDO, 2024; SABINO et al., 2020).

According to Willmott and Matsuura (2005), RMSE should be interpreted relative to the variability of the observed data. An analysis stratified by return period shows that RMSE values remain relatively low and stable for return periods up to 100 years (approximately 8,6 - 11,9 mm/h), increasing progressively for higher return periods. Residual analysis did not reveal systematic bias, with positive and negative deviations distributed across different durations and return periods. Smaller and more stable residuals were observed for medium and long durations (> 60-100 min), whereas larger deviations were concentrated at short durations and high return periods. These patterns are consistent with the greater variability of convective rainfall and with the inherent uncertainties in estimating short-duration extreme precipitation (CHOW et al., 1988). Overall, the statistical indicators and residual behavior indicate that the logarithmic regression provided a robust and hydrologically

consistent representation of the intensity-duration-frequency relationship, supporting its application for design and planning purposes.

Rainfall simulations based on a design storm with a 10-year return period and a duration of 120 minutes were adopted to evaluate NbS performance during prolonged rainfall events (PREFEITURA MUNICIPAL DE CURITIBA, 2017; ALVES, 2020). The temporal distribution of rainfall was defined for the generation of the design hyetograph using the Alternating Block Method, with an advanced peak factor of 0,25.

A rainfall event recorded on January 18, 2017, exhibited an intensity of 70,866 mm/h for a duration of 120 minutes, which resulted in the highest estimated return period among the analyzed durations. An estimated return period of approximately 280 years was obtained and adopted as a reference value in the analysis. However, this estimate is subject to uncertainty and should not be interpreted as an exact return period, given the limitations associated with GEV fitting and IDF extrapolation for long return periods. This observed high-intensity event was used to the hydrological response of the watershed under critical rainfall conditions.

For both the design storm and the observed event, the precipitation data were entered through the Time Series item in the SWMM's project browser. The simulation period was defined to ensure near-zero flow conditions at the outfall, corresponding to an advanced recession stage of the hydrograph. A duration of 20 h was adopted for the design rainfall, while 30 h was used for the observed real event.

5.2.4 Construction Criteria for NbS

The construction criteria, including aspects related to layers, depths, vegetation, and other parameters, were compiled through a systematic review of specialized technical sources, such as manuals and technical publications. In cases of discrepancies between references, the most frequently cited values in the literature were prioritized, while unique criteria mentioned by only one source were retained with proper citation. Due to the distinct structural characteristics of each NbS type, the design criteria were systematically organized in dedicated tables: Table 5-3 details rain garden specifications, Table 5-4 covers constructed wetlands, and Table 5-5 presents bioswale parameters.

Table 5-3: Construction criteria for rain gardens.

Criterion	Type of restriction	Rain Garden
Filtration layer [m]	Minimum	0,30 ^{1,2}
Storage layer [m]	Maximum	0,30 ²
Water depth [m]	Minimum	0,15 ^{3,4}
	Maximum	0,30 ^{1,5}
Vegetation density [%]	Minimum	90 ⁶
Rain garden infiltration rate [mm/h]	Minimum	12,7 ^{2,7}

¹Prince George's County Department of Environmental Resources, 2007; ²Eskin, 2021; ³NACTO, 2017; ⁴GIZ, 2023; ⁵Pennsylvania Stormwater Best Management Practices Manual, 2006; ⁶UNI-GROUP USA, 2014; ⁷CIRIA, 2013. Source: Authors (2025).

Table 5-4: Construction criteria for wetlands.

Criterion	Type of restriction	Wetlands
Length-to-width ratio	Minimum	3:1 ^{1,2}
Depth of deep pools [m]	Minimum	1,20 ^{2,3}
Depth of High Marsh Zone [m]	Minimum	0,15 ⁴
Depth of Deep Water Zone and micropools [m]	Minimum	0,45 ^{4,6}
	Maximum	1,80 ^{4,6}
Depth of Low Marsh Zone [m]	Minimum	0,15 ^{4,6}
	Maximum	0,45 ^{4,6}
Infiltration rate [mm/h]	Maximum	0,254 ⁵
Freeboard [m]	Minimum	0,30 ⁴
Temporary flooding zone [m]	Recomended	0,38 ¹
Vegetation density [plants/m ²]	Minimum	2,70 ¹
Topsoil layer thickness [m]	Minimum	0,10 ⁵
Storage capacity [m ³]	Minimum	102,8 ⁵

¹North Carolina Department of Environmental Quality, 2018; ²Department of Environment, Science and Innovation of Queensland, 2022; ³Connecticut Department of Environmental Protection, 2004; ⁴Knox County Stormwater Management, 2008; ⁵North Carolina Department of Enviromental Quality, 2009; ⁶Minnesota Pollution Control Agency, 2022. Source: Authors (2025).

Table 5-5: Construction criteria for bioswales.

Criterion	Type of restriction	Bioswales
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Infiltration rate [mm/h]	Minimum	12,7 ¹
Longitudinal slope [%]	Maximum	4 ²
Width [m]	Minimum	0,60 ^{3,4}
Length [m]	Minimum	30 ¹
Aggregate layer [m]	Minimum	0,30 ^{2,5}
Permeable soil layer [m]	Minimum	0,60 ^{2,5}
Side slope	Maximum	3:1 ^{5,6}
Vegetation density [%]	Minimum	80 ⁵
Depth [m]	Maximum	0,60 ^{7,8}
Freeboard [m]	Minimum	0,15 ¹
Water Depth [m]	Maximum	0,30 ^{1,5,7}

¹Pennsylvania Stormwater Best Management Practices Manual, 2006; ²Secretaria de Desenvolvimento Urbano e Habitação do Estado de São Paulo, 2024; ³Prince George's County Department of Environmental Resources, 2007; ⁴Eskin, 2021; ⁵Sand Diego, 2014 ⁶Maryland, 2009; ⁷NACTO, 2017; ⁸CIRIA, 2013. Source: Authors (2025).

5.2.5 Input parameters of NBS into the model

The input parameters of the Nature-Based Solutions (NbS) were defined to represent their physical structure and hydrological functioning within the SWMM modeling framework. The contributing areas of each NBS were explicitly delineated and individually assigned in the model, ensuring that rain gardens, bioswales, and constructed wetlands were represented according to their specific spatial distribution.

In SWMM, the Seepage Rate parameter represents the permanent infiltration rate from the storage layer of a Low Impact Development (LID) unit to the underlying native soil, expressed in mm/h, accounting for water losses through vertical percolation. In this study, an intermediate seepage rate value was adopted based on field measurements obtained from the SACRE project, which evaluated hydraulic conductivity at different depths and locations across the study watershed. Measurements conducted in soil trenches indicated average hydraulic conductivity values ranging from 7 to 10 mm/h in the predominantly sandy clay loam soils. Based on these observations, a seepage rate of 10 mm/h was adopted to represent the mean percolation capacity of the soils underlying the rain gardens and bioswales.

The structural parameters of the LID units (rain gardens and bioswales) were defined to reflect typical design configurations reported in the literature and technical guidelines. In SWMM, LID units are implemented within the Subcatchment object

through the LID Control module, which allows the representation of decentralized stormwater management structures and their interaction with runoff generated in the drainage area. In contrast, constructed wetlands were modeled as storage units, consisting of a subcatchment draining into a storage unit followed by a weir connected to a junction in the stream. The maximum depth of the storage unit was defined as the physical limit of the water column, adopting an intermediate depth representative of the deep water zone. A non-zero initial depth was assigned to characterize the wetland as a permanent system, while the surcharge depth represented a temporary flood storage zone activated under high inflow conditions. Seepage from the wetland bottom was modeled using a low hydraulic conductivity value, reflecting reduced percolation rates typically associated with consolidated sediments in permanent wetlands.

Overflow from the wetland was controlled by the weir structure, with the crest height defined at the upper limit of the normal operating water level. The weir height was configured as the sum of the crest height and the surcharge depth, ensuring that overflow occurs only after the temporary flood storage zone is fully occupied. Table 5-6 presents the input parameters adopted for wetlands in SWMM.

Table 5-6: Input parameters in SWMM.

Surface Parameters	Rain Gardens	Bioswales	Wetlands
Berm Height [mm]	300	150	N/A
Vegetation Volume Fraction [m]	0,10 ¹	0,10 ¹	N/A
Surface Roughness	0,20 ¹	0,20 ¹	N/A
Surface Slope [%]	1,00	4,00	N/A
Soil Parameters	Rain Gardens	Bioswales	Wetlands
Thickness [mm]	300	600	N/A
Porosity	0,40	0,40 ¹	N/A
Conductivity [mm/h]	50,8 ²	25,4	N/A
Storage Parameters	Rain Gardens	Bioswales	Wetlands
Thickness [mm]	300	300	N/A
Void Ratio	0,40	0,40	N/A
Seepage Rate [mm/h]	10,00	10,00	N/A
Maximum Depth [m]	N/A	N/A	1,50
Initial Depth [m]	N/A	N/A	0,45
Surcharge Depth [m]	N/A	N/A	0,01
CrestHt - WEIR [m]	N/A	N/A	1,10
Conductivity [mm/h]	N/A	N/A	0,254

¹Rossman, 2010; ²NOAA, 2012. Source: Authors (2025). N/A: not applicable.

5.2.6 Performance Analysis

At this stage, the data extracted from the Summary Results and Status Report were organized into three spatial and operational levels of analysis: the macroscopic behavior of the study watershed, the individual response of the drainage areas, and the specific efficiency of the NBS.

The evaluation of the impact of Nature-Based Solutions (NBS) at the macroscopic watershed scale was based on the analysis of the water balance and the hydrological response at the outfall, considering the precipitation scenarios. The analysis procedure involved the model's water balance and performance indicators at the outfall. For each analyzed scenario (Baseline Scenario, BW Scenario, RG Scenario, CW Scenario, and Integrated Scenario), total precipitation volumes were extracted, representing the total water input to the system, along with infiltration and storage volumes.

Additionally, the continuity errors calculated by the software for both surface runoff and flow routing were monitored. These errors represent the percentage difference between the system's input and output balance. To ensure numerical stability and mass conservation, the criterion established by Rossman (2015) was adopted, which states that errors exceeding 10% invalidate the simulation's consistency. Thus, the validation of the scenarios followed this technical rigor, ensuring that the results complied with the accepted limits in hydrological literature.

Regarding the effectiveness of the interventions in flood mitigation, the hydrological response was monitored at a macroscopic scale at the watershed's outfall. The comparison between the Baseline Scenario and the intervention scenarios was based on performance indicators reflecting surface flow dynamics: average and maximum flow rates were used to evaluate the capacity for peak flow reduction, while the total runoff volume allowed for measuring the relative performance of each scenario in volumetric control. Finally, the time to peak was analyzed to identify the potential for flood wave delay provided by green infrastructure solutions within the drainage system.

To assess the spatial impact of SBNs and the individual retention capacity of the 330 drainage areas, the Runoff Coefficient (C) was used as a performance indicator. This dimensionless parameter represents the fraction of inflow water that effectively converts to direct surface runoff. The calculation was performed by SWMM

for each sub-watershed, considering direct precipitation and the contribution of external flows from upstream watershed (Runon), according to Equation 1.

$$C = \frac{\text{Outflow}}{\text{Precipitation} + \text{Total Runon}} \quad (\text{Eq.1})$$

Where: Outflow is the total water depth of surface runoff leaving the drainage area (mm); Precipitation corresponds to the total water depth precipitated directly over the drainage area (mm); and Total Runon is the total water depth received from adjacent drainage areas (mm).

The classification intervals for the runoff coefficient were defined based on the typical urban land-use ranges recommended by Tucci (2000) and technical drainage guidelines (BRASIL, 2021), segmenting the drainage areas into areas of Low, Medium, and High Imperviousness (Table 5-7).

Table 5-7: Classification intervals for the surface runoff coefficient.

Classification	Coefficient Range
Low Impermeabilization	$C \leq 0,3$
Medium Impermeabilization	$0,3 < C \leq 0,7$
High Impermeabilization	$C > 0,7$

Source: Authors (2026).

In addition, to further analyze the impacts on the 330 drainage areas, the number of drainage areas that showed a complete elimination of peak flow in each studied scenario was recorded. Furthermore, an analysis of the relative efficiency of runoff removal was conducted, comparing the reduction in surface runoff between the proposed scenarios and the Baseline Scenario. This analysis made it possible to visualize the distribution of SBN performance across drainage areas, highlighting the marked variability in hydrological response under integrated scenarios, evidenced by the amplitude of the boxes, as well as the median removal efficiency values, which serve as a central point of comparison among SBN types under different rainfall conditions.

The evaluation of the intrinsic unit efficiency of NBS and implementation scenarios was conducted using Linear Mixed-Effects Models (LMM). This approach was adopted to address the inherent dependency between observations from the same territorial unit (drainage area) and to isolate the efficacy of NBS types from uncontrolled local variables, such as topography and land use. The statistical significance of fixed effects was determined using Satterthwaite's approximation for

degrees of freedom, which refines the estimation and ensures greater precision for p-values in mixed models.

The model structure included as fixed effects the NBS type (RG, BW, CW), the implementation scenario, the rainfall type, and the interaction between scenario and rainfall. Hydraulic capacity was included as a control covariate to normalize the data, while the identification of drainage areas was defined as a random effect to account for intrinsic spatial variability. Given the inherent skewness of hydrological data and the presence of heteroscedasticity, a bilateral logarithmic transformation (log-log) was applied to both the dependent variable and the covariate. This strategy aimed to linearize the hydrological response, mitigate the leverage effect of hydraulic capacity outliers, and enable the interpretation of results in terms of elasticity.

The validity of the model's inferences was confirmed through a rigorous diagnostic. Residual normality was assessed using the Shapiro-Wilk test alongside visual inspection of Quantile-Quantile (Q-Q plots), prioritizing visual adherence in cases of large sample sizes where hypothesis tests tend to be overly conservative. Homoscedasticity was verified using Levene's test and the analysis of Residuals vs. Fitted values plots, ensuring that the applied log-log transformation successfully stabilized error variance.

For statistical inference and to ensure the robustness of t-tests regarding fixed effects, Satterthwaite's approximation was used for the calculation of degrees of freedom. In cases of statistical significance ($p < 0.05$), Tukey's multiple comparison test was applied as a post-hoc procedure. This analysis was conducted using Estimated Marginal Means (EMM), an approach that allowed the intrinsic effect of each factor to be isolated by marginalizing hydraulic capacity at its mean level. Consequently, the post-hoc analysis enabled: (i) the isolation of the intrinsic effectiveness of the NBS; (ii) a comparison of individual versus integrated scenarios to verify potential synergistic efficiency gains; and (iii) an assessment of the resilience of the arrangements (Individual vs. Integrated Scenarios) during the transition from design storms to real-world events.

5.2.7 Model Calibration and Validation

Due to the absence of experimental data for calibration, a sensitivity analysis was performed to evaluate the influence of key parameters of the hydrological model

on the simulated hydrological responses. Sensitivity analysis is widely used in hydrological modeling to estimate the rate of change in model outputs as a function of variations in input parameters, allowing the identification of those that most strongly influence model behavior (JAMES, 2005).

The selection of parameters considered three main criteria: (i) parameters associated with greater uncertainty regarding their actual values under local conditions, such as soil characteristics and surface properties; (ii) parameters known from the literature to significantly affect hydrological outputs of interest, particularly peak flow and total volume; and (iii) parameters that may be measured or refined through future field investigations. Based on these criteria, parameters related to watershed physical characteristics, infiltration processes, and surface roughness were selected for analysis. The sensitivity analysis was performed by individually varying each selected parameter while maintaining the remaining variables constant at their baseline values. This procedure allowed for the isolated evaluation of how each parameter influences the model's hydrological responses (SALTELLI *et al.*, 2000).

For most parameters, a variation range of $\pm 30\%$ relative to the baseline value was adopted, following recommendations commonly reported in hydrological modeling studies (LENHS, 2012; SILVEIRA *et al.*, 2022; LI *et al.*, 2025). This interval allows the assessment of model sensitivity to typical uncertainties associated with parameter estimation while maintaining values within the operational limits recommended by the EPA SWMM user manual. For the Manning roughness coefficients of permeable and impermeable areas (n_{perv} and n_{imperv}), the minimum and maximum values established in the SWMM documentation were considered (LENHS, 2012).

Model sensitivity was evaluated through the relative percentage variation of the simulated results, calculated by comparing the outputs obtained after parameter perturbation with those generated using the baseline configuration. This procedure allows the normalization of model responses and facilitates the comparison of parameters with different physical magnitudes (AKDOĞAN; GÜVEN, 2016).

In total, 24 simulations were performed for the sensitivity analysis. Each of the twelve selected parameters was individually adjusted within predefined ranges, while

all other parameters remained fixed at their baseline values. Table 5-8 presents the base values and the specific variations applied to each parameter.

Table 5-8: Parameters selected for sensitivity analysis, including baseline values and variation intervals.

Parameters	Base Value	Variation
% Imperv	Specific to each drainage area	± 30%
Width	Specific to each drainage area	± 30%
Minimum Infiltration Rate	27,75 mm/h	± 30%
Maximum Infiltration Rate	53,68 mm/h	± 30%
Berm Height - Rain Gardens	300 mm	± 30%
Berm Height - Bioswales	150 mm	± 30%
Soil Thickness - Rain Gardens	300 mm	± 30%
Soil Thickness - Bioswales	600 mm	± 30%
Area - Wetlands	Specific to each wetland	± 30%
Initial Depth - Wetlands	0,45 m	± 30%
Manning's roughness (n-imperv)	0,04	0,011 0,024
Manning's roughness (n-perv)	0,09	0,008 0,05

Source: Authors (2026).

The results of this analysis support the identification of the parameters that most influence the model responses, contributing to the evaluation of model robustness and guiding future calibration efforts based on field measurements. The variation in the sensitivity analysis was classified according to the magnitude of the relative change: High for changes greater than 10%, Medium for values between 5% and 10%, Low for changes less than 5%, and “None” in cases where no noticeable change was observed.

5.3 Results

Hydrological modeling enabled the extraction of a robust dataset that underpins the comparative analysis between baseline scenarios and the implementation of Nature-Based Solutions (NBS). The results presented below first detail the integrated hydrological response at the watershed scale (Section 5.3.1), quantifying changes in peak flows, total volumes, and the overall efficiency of the solutions at the main outlet. Subsequently, the spatial variation of the impact across

the 330 drainage areas is examined (Section 5.3.2), focusing on the calculation of runoff coefficients and a comparison of the average increase in local infiltration and storage rates. Finally, the performance of NBS typologies and scenarios (Section 5.3.3) is evaluated through a statistical analysis using mixed-effects models, isolating the intrinsic characteristics of Rain Gardens, Bioswales, and Constructed Wetlands to determine their unit effectiveness. This structure allows for an assessment of the cumulative systemic benefits of the interventions to support sustainable urban planning in the region.

5.3.1 Integrated Hydrological Response at the Watershed Scale

At the watershed scale, the implementation of NBS resulted in a substantial modification of the hydrological response for both the design storm and the real rainfall event, demonstrating the effectiveness of the intervention in attenuating floods across the study area. Table 5-9 presents the total precipitation, infiltration, and storage volumes in the watershed, and the continuity errors obtained in the SWMM status report.

Table 5-9: Summary of Water Balance Components and Continuity Errors for Simulated Scenarios.

Event Type	Indicator	Baseline Scenario	RG Scenario	BW Scenario	CW Scenario	Integrated Scenario
Design Rainfall	Precipitation Volume [10 ³ m ³]	697,05	697,05	697,05	697,05	697,05
	Infiltration [10 ³ m ³]	232,37	288,19	235,69	229,49	285,70
	Storage [10 ³ m ³]	3,56	73,69	7,66	49,82	120,48
	Outflow [10 ³ m ³]	470,65	344,74	463,25	417,76	290,92
	Continuity Error Runoff	-0,006	-0,008	-0,005	-0,006	-0,008
	Continuity Error Flow Routing	0,002	0,002	0,002	0,002	0,002
Real Event	Precipitation Volume [10 ³ m ³]	1512,31	1512,31	1512,31	1512,31	1512,31
	Infiltration [10 ³ m ³]	427,04	537,76	432,75	427,98	543,14

Storage [10 ³ m ³]	6,1	88,34	9,87	75,87	161,76
Outflow [10 ³ m ³]	1078,85	886,25	1069,55	1008,31	807,45
Continuity Error Runoff	0,008	-0,004	0,008	0,008	-0,004
Continuity Error Flow Routing	0,004	0,002	0,002	0,003	0,002

Source: Authors (2026).

Based on the results presented in the Status Report, it is evident that the implementation of NBS has substantially increased the watershed's resilience to rainfall events. In the design rainfall event, the Integrated Scenario demonstrated a robust performance, increasing the stored volume from 3,560 m³ to 120,480 m³ and the infiltration volume from 232,370 m³ to 285,700 m³, resulting in a reduction of approximately 38,2% in the outflow volume compared to the Baseline Scenario.

In the Real Event, it was observed that, due to a significantly higher precipitation volume, the reduction in outflow volume from the Baseline Scenario to the Integrated Scenario was 25,15%. This variation indicates that, although the watershed becomes more efficient with NBS, its buffering capacity is inversely proportional to the magnitude of the event. In extreme events, while the absolute volume removed from the system is higher than in the design rainfall, the performance in relative terms (percentage of total volume) decreases. This occurs because the watershed's storage and infiltration mechanisms reach their capacity limits and become saturated under high-intensity rainfall, although they still provide a better hydrological response than the Baseline Scenario.

However, the scenarios involving individual NBS types revealed distinct contributions to the overall water balance. The RG Scenario demonstrated the highest isolated efficiency in promoting infiltration, raising this component to 288,190·10³ m³ in the design rainfall and reaching the significant value of 537,76·10³ m³ in the real event. This performance identifies the widespread implementation of rain gardens as the most effective strategy for watershed-wide infiltration, primarily due to the high spatial distribution of the 244 units across the area.

Beyond infiltration, the RG Scenario also exhibited the highest total stored volume for both events. This dual performance stems from the technique's versatility,

as the 244 units were allocated in densely urbanized areas with few topographic constraints, resulting in a large cumulative storage footprint. In contrast, the CW Scenario focused almost exclusively on storage despite having significantly fewer units (only 17). While its impact on infiltration was marginal, even showing a slight reduction compared to the Baseline Scenario in the design rainfall due to the replacement of natural permeable areas with storage structures, its accumulation capacity reached approximately $75,87 \cdot 10^3 \text{ m}^3$ during the real event. Notably, these 17 wetland units provided a storage volume nearly equivalent to the entire network of 244 rain gardens, playing a key role in hydraulic attenuation. This discrepancy in unit count is justified by implementation constraints, as wetlands require larger catchment areas and greater land availability than rain gardens.

Conversely, the BW Scenario yielded the most modest results, with volume variations closely mirroring the Baseline. The limited performance of the 11 bioswale units is attributed to strict design constraints, such as specific gradients and minimum lengths of 30 meters, which complicate their integration across diverse sections of the watershed. Thus, the poor performance observed for this type suggests that, for this watershed and this specific arrangement of bioswales, the bioswales were not sufficient to significantly alter the overall water balance.

This disparity underscores that the watershed's resilience is optimized not by isolated techniques, but by the complementarity of functions. Specifically, while rain gardens maximize diffuse infiltration, wetlands provide the critical volume needed to manage excess water during larger-scale events. Thus, although a higher density of rain gardens could theoretically achieve similar total removal, the integrated approach combines distinct physical processes, source infiltration and centralized detention, thereby increasing the system's reliability through functional diversity.

The continuity errors obtained for both runoff generation and flow routing were very small across all simulated scenarios. Runoff continuity errors, associated with the difference between total precipitation and the sum of the water balance, ranged between $-0,008\%$ and $0,008\%$, while flow routing errors, associated with the difference between the water that entered the nodes/ducts and the sum of what was stored or discharged at the outfall, varied from $0,002\%$ to $0,004\%$. These values are well below the commonly accepted, indicating a consistent mass balance and numerical stability in the SWMM simulations.

Regarding the hydrological performance at the outfall, Table 5-10 summarizes the hydrological response at the outlet of the Água Comprida Watershed for all simulated scenarios under both rainfall events, including average and maximum flows and total runoff volumes.

Table 5-10: Summary of hydrological performance parameters at the outfall: average and maximum flows, total volume, volume reduction and time to peak for the baseline and intervention scenarios.

Event Type	Indicator	Baseline Scenario	RG Scenario	BW Scenario	CW Scenario	Integrated Scenario
Design Rainfall	Average Flow [m ³ /s]	6,579	4,819	6,476	5,840	4,067
	Maximum Flow [m ³ /s]	114,243	86,562	111,266	89,915	58,903
	Total Volume [10 ³ m ³]	470,657	344,738	463,253	417,764	290,922
	Time to peak	00:56:45	00:54:20	00:56:15	01:03:55	00:57:30
Real Event	Average Flow [m ³ /s]	23,339	8,229	9,931	9,363	7,498
	Maximum Flow [m ³ /s]	183,388	135,586	180,063	171,420	121,155
	Total Volume [10 ³ m ³]	1078,852	886,255	1069,546	1008,303	807,455
	Time to peak	02:14:35	02:14:10	02:14:30	02:16:00	02:15:35

Source: Authors (2026).

Under the design storm conditions, the integrated scenario proved the most effective in reducing flood magnitude and volume. A significant attenuation of the peak flow was observed, with a reduction of 48,4% (from 114,24 m³/s to 58,90 m³/s) the most direct indicator of a reduced risk of flooding downstream. This was accompanied by consistent reductions in average flow (from 6,579 m³/s to 4,067 m³/s) and total runoff volume (38,2%).

The response during the actual rainfall event was slightly less effective, due to saturation and outflow from the NBS. The average flow saw a drastic reduction of 67,9% (from 23,34 m³/s to 7,50 m³/s). While the peak discharge was mitigated by 33,9%, the total volume was reduced by 25,2%, confirming the intervention's ability to retain a substantial volume of stormwater. Thus, the integrated NBS strategy proved

effective in reducing the downstream hydraulic load, mitigating overflow risks by altering the water balance at the catchment scale.

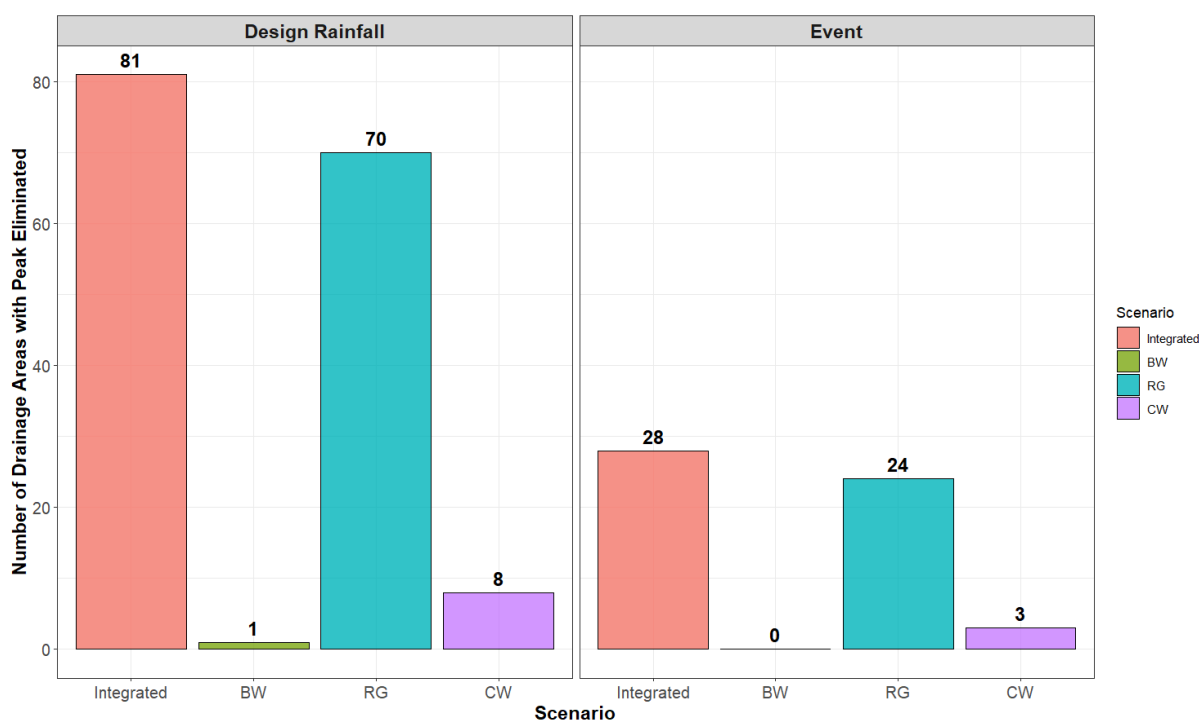
Regarding the time to peak, a discrete delay in peak discharge was observed in the Integrated Scenario for both rainfall events, as well as in the CW Scenario for the real event. In the design rainfall, the CW Scenario promoted a delay of approximately 10 minutes compared to the Baseline Scenario. Conversely, scenarios involving rain gardens and bioswales exhibited a subtle anticipation of the time to peak. This behavior stems, primarily, from the greater discretization of the model into 330 drainage areas, which reduces flow lengths and local response times. Furthermore, this phenomenon is explained by a hydrological rearrangement: by retaining and infiltrating the volume fractions that would compose the main body of the original hydrograph (first flush and intermediate volumes), the NBS promote a flow remodeling where the remaining maximum discharge, with lower magnitude, is dictated by rapid-response contributions from impervious areas near the outlet or by unit overflow upon reaching early saturation. Therefore, the observed anticipation does not indicate an acceleration of global runoff, but rather an effect of selective attenuation and volume subtraction, where the new remaining peak manifests at a distinct time interval without compromising the system's attenuation efficiency.

5.3.2 Spatial Variation of Impact Across Drainage Areas

This section assesses the spatial distribution of hydrological impacts across the 330 drainage areas. By comparing the baseline scenario with integrated and isolated interventions, it was determined how different characteristics of the drainage areas and precipitation patterns influence the reduction in surface runoff.

Initially, the analysis focused on the total elimination of peak flow. As shown in Figure 5-2, the Integrated Scenario yielded a higher number of drainage areas with zero runoff (81 catchments for the design rainfall and 28 for the real event) compared to the mathematical sum of the isolated NBS scenarios (79 for the design rainfall and 27 for the real event). While subtle, this difference confirms a positive synergistic effect, suggesting that the combined spatial arrangement of NBS provides a slightly more robust defense against runoff than the mere sum of its parts.

Figure 5-2: Number of drainage areas with total elimination of peak flow across study scenarios and rainfall events.



Source: Authors (2026).

However, a significant reduction in the number of 'zero-runoff' drainage areas was observed when comparing the design rainfall to the monitored event. This performance drop indicates that the higher intensity or different temporal distribution of the monitored event exceeded the storage and infiltration capacity of the NBS, highlighting the physical threshold of the system under more critical hydraulic loads.

To further understand the effects of runoff coefficient reduction, the drainage areas were categorized based on their specific values into 'High', 'Medium', and 'Low' runoff levels, in accordance with the criteria established in the methodology. Tables 5-11 and 5-12 present the distribution of these catchments across each studied scenario and rainfall type, highlighting the shift in permeability patterns.

Table 5-11: Distribution of drainage areas by runoff coefficient classification under the Design Rainfall scenario.

Scenario	Highly impermeabilization areas	Medium impermeabilization areas	Low impermeabilization areas
Baseline Scenario	268	50	12
Integrated Scenario	78	125	127
BW Scenario	261	56	13

RG Scenario	94	120	116
CW Scenario	260	48	22

Source: Authors (2026).

The analysis of Table 5-10, regarding the Design Rainfall scenario, reveals a significant reduction in the number of drainage areas classified as highly impermeable, dropping from 268 in the Baseline Scenario to 78 in the Integrated Scenario, indicating an effective synergy between the applied techniques. Among the individual NBS scenarios, the most prominent results were observed in the RG Scenario. With 244 units deployed, this intervention reduced the number of highly impermeable drainage areas to 94 and enabled 116 areas to reach the low impermeability classification.

In contrast, the BW Scenario, comprising only 11 units, shows a smaller absolute impact but a notable relative efficiency at the local level: among the areas that received this intervention, 6 migrated from the high to the medium impermeability category, and 1 reached the low category. This demonstrates that, although the limited number of structures restricts their influence at the watershed scale, the technique was effective where applied. Similarly, in the CW Scenario, with 17 units, 8 drainage areas migrated from high impermeability and 2 from medium impermeability to the low impermeability classification.

Table 5-12: Distribution of drainage areas by runoff coefficient classification under the Monitored Real Event.

Scenario	Highly impermeabilization areas	Medium impermeabilization areas	Low impermeabilization areas
Baseline Scenario	284	41	5
Integrated Scenario	157	95	78
BW Scenario	282	42	6
RG Scenario	167	93	70
CW Scenario	275	43	12

Source: Authors (2026).

The analysis of Table 5-11, regarding the Monitored Real Event scenario, reveals that nearly all drainage areas in the Baseline Scenario (284 units) exhibit high impermeabilization. With the implementation of the Integrated Scenario, this

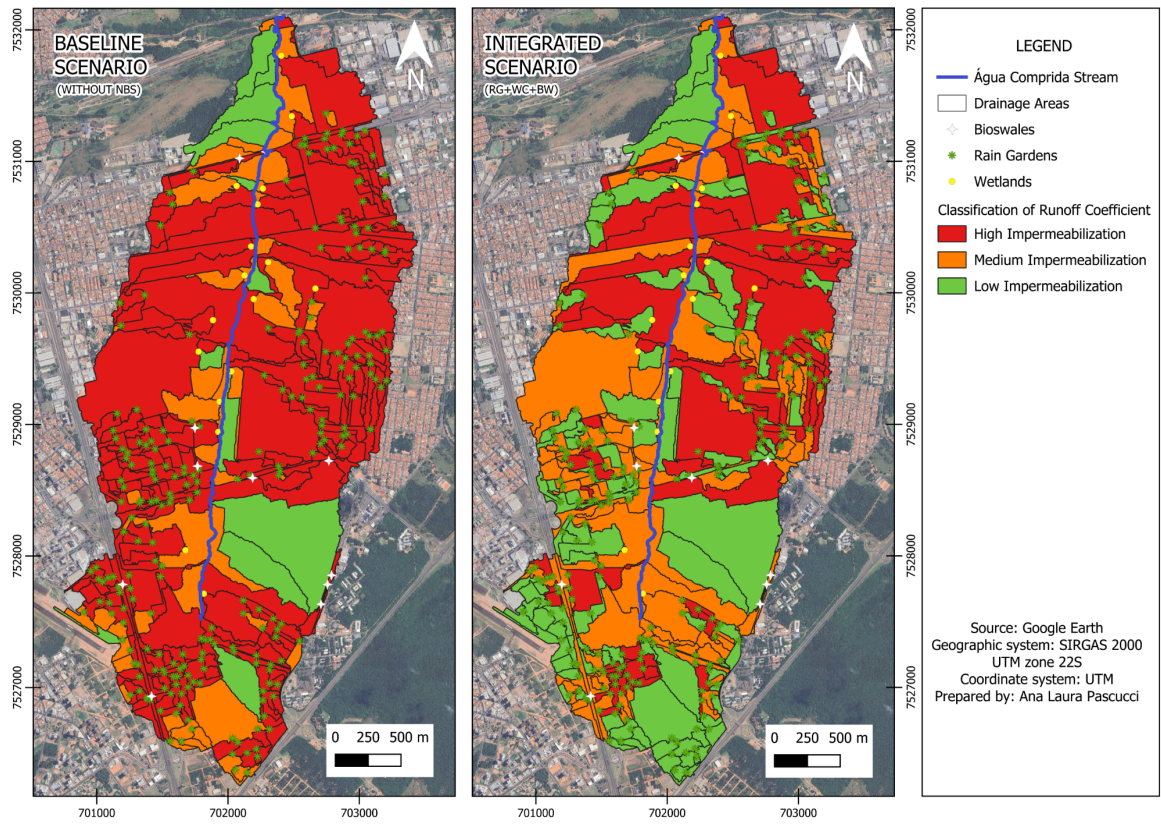
number drops to 157, demonstrating that, even under real rainfall conditions, the combination of techniques succeeds in reclassifying a significant portion of the areas into the medium and low categories. Consistent with the findings from the design rainfall, the primary individual driver is the RG Scenario, which, with its 244 units, reduced highly impermeable drainage areas to 167 and enabled 70 areas to reach the low impermeability classification.

In contrast, the Bioswales and Wetlands Scenarios showed more modest impacts in absolute terms. In the BW scenario (11 units), only 2 drainage areas migrated from the high to the medium impermeability category and 1 to the low category, indicating that the small deployment scale limits the systemic response during a monitored real event. Similarly, the CW scenario (17 units) resulted in the migration of 9 areas from the high impermeability category to lower-impact levels (7 to medium and 2 to low). These results reinforce that, while the techniques are locally effective, the transformation of hydrological behavior at a macro scale depends on the density of the installed units.

A comparison of the two scenarios shows that the effectiveness of NbS is greater under the design rainfall scenario than under the actual monitored event. While in the design rainfall scenario the reduction in areas of high impermeability in the integrated scenario was approximately 71%, in the actual event this reduction was 45%. This disparity occurs because actual events often have prolonged durations or intensity patterns that lead to early saturation of the soil and infiltration devices, reducing their retention capacity compared to theoretical design models.

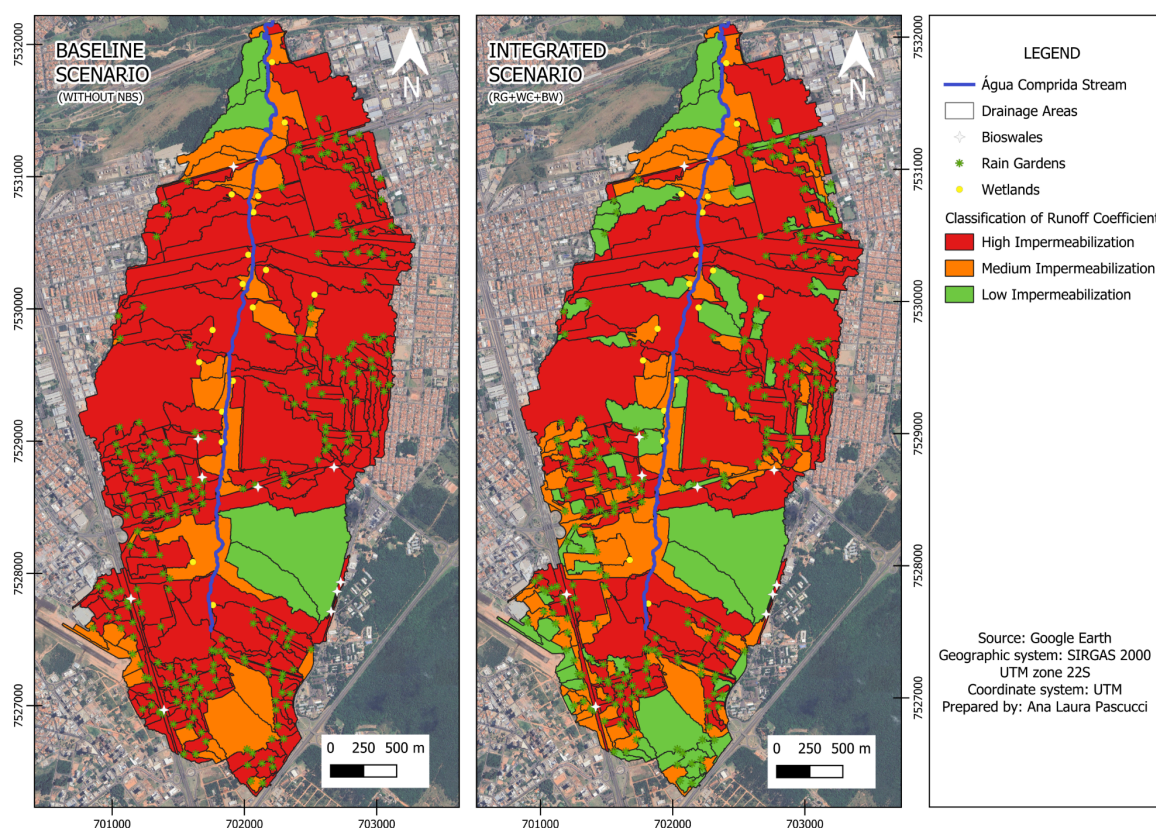
To illustrate the spatial dynamics of these changes in imperviousness classification, Figure 5-3 and Figure 5-4 present comparative maps of the drainage areas' behavior under the Baseline Scenario and the Integrated Scenario, for the design rainfall and the actual event, respectively. In the figures, the reduction in imperviousness is represented by the transition of colors, where the green areas (low imperviousness) expand over the yellow areas (medium) and, primarily, over the extensive red areas (high imperviousness) observed in the Baseline Scenario.

Figure 5-3: Spatial variability of the surface runoff coefficient under the baseline and integrated scenarios for the design rainfall.



Source: Authors (2026).

Figure 5-4: Spatial variability of the surface runoff coefficient under the baseline and integrated scenarios for the real event.



Source: Authors (2026).

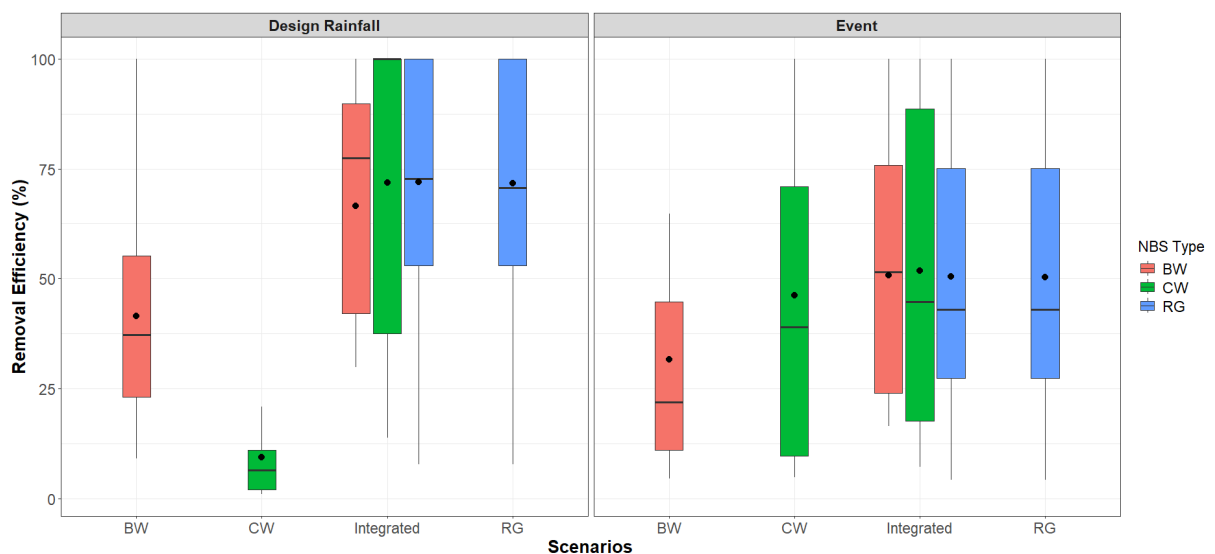
The spatial visualization provided by Figures 5-3 and 5-4 confirms that the effectiveness of the integrated system is distributed non-uniformly across the watershed, highlighting critical aspects of its hydrological operation. On the other hand, there is a notable indirect positive effect: some drainage areas, located downstream of drainage areas with NBS, were reclassified to lower levels of impermeability, even without having NBS directly allocated to them. This phenomenon can be attributed to the capacity of upstream NBS to retain and infiltrate significant volumes of water, reducing the flow reaching lower areas and lowering the runoff coefficient of the aggregate drainage area. Some drainage areas were also analyzed that, despite having SBNs, were not reclassified, such as the northeastern portion of the watershed, which exhibits intense and consolidated urbanization. Thus, the non-reclassification of some areas may be associated with the characteristics of the drainage area itself and/or insufficient SBN area for the total flow.

Finally, an analysis of Figure 5-5 allows for a comparison of the variability and effectiveness of surface runoff removal across different scenarios. In the Design

Rainfall, removal efficiency is higher and more consistent for the Rain Gardens (RG) and the Integrated Scenario, both of which have averages above 70% and drainage areas that reach the maximum removal limit of 100%. However, the results suggest that the combination of NBS not only achieves peaks in overall efficiency but also increases the average performance and median removal rate across the drainage area. This trend indicates a preliminary synergistic effect, where the integration of different types of NBS compensates for the individual limitations of each technique, raising the overall level of stormwater control. On the other hand, CWs exhibit the lowest isolated performance under design rainfall, with a median efficiency close to 10%.

In the Real Event, however, there is a general decline in average efficiency and a significant increase in data dispersion. This behavior indicates a heterogeneous response of the system to the variability of actual precipitation. The CWs show a relative improvement in performance during the actual event compared to the design rainfall, with a median removal rate of nearly 40%. This suggests that, during intense events, the detention capacity of these structures is better utilized. However, in both rainfall regimes, the Integrated Scenario and the RG Scenario remain the most resilient solutions, although the total variability in the actual event reveals that some watersheds exhibit minimal performance, reaching values close to zero, likely due to the increase in inflow volume and prior soil saturation in the initial moments of rainfall until it reached its peak intensity at 2 hours.

Figure 5-5: Comparison of runoff removal efficiency (infiltration and storage) by NBS scenario for Design Rainfall and Real Event.



Source: Authors (2026).

Although the descriptive analysis confirms the superiority of the Integrated Scenario and Rain Gardens (Figure 5-5), the high dispersion observed suggests that the performance of NBS is influenced by variables intrinsic to each drainage area. The variability in the results indicates that removal efficiency is not uniform, requiring further investigation into the determinants of this performance. To isolate the actual effect of each typology and understand the physical interactions involved, the next section presents a mixed-effects model for a statistically controlled comparative analysis of the proposed scenarios.

5.3.3 Comparative Performance of NBS Typologies and Scenarios

This section examines the efficiency per unit of NbS from two analytical perspectives: the intrinsic efficiency of the typologies (RG, BW, and CW) and the strategic efficiency of the scenarios (Integrated Scenario, RG Scenario, BW Scenario, and CW Scenario) by comparing the removal volume, infiltration volume, storage volume, reduction in maximum peak flow, and peak delay.

Intrinsic typological efficiency assesses the nominal potential of each technique in isolation, focusing on the device's ability to process the hydraulic load it receives directly. Scenario-based strategic efficiency, on the other hand, examines the average performance of the units when integrated into specific watershed configurations, capturing the effects of deployment density, connectivity, and the systemic resilience of the entire system.

Prior to applying the hydrological model data to the Linear Mixed-Effects Models (LMM), the results of the Shapiro-Wilk test were analyzed, as shown in Table 5-13.

Table 5-13: Normality Test Results (Shapiro-Wilk) for Raw Data and Model Residuals.

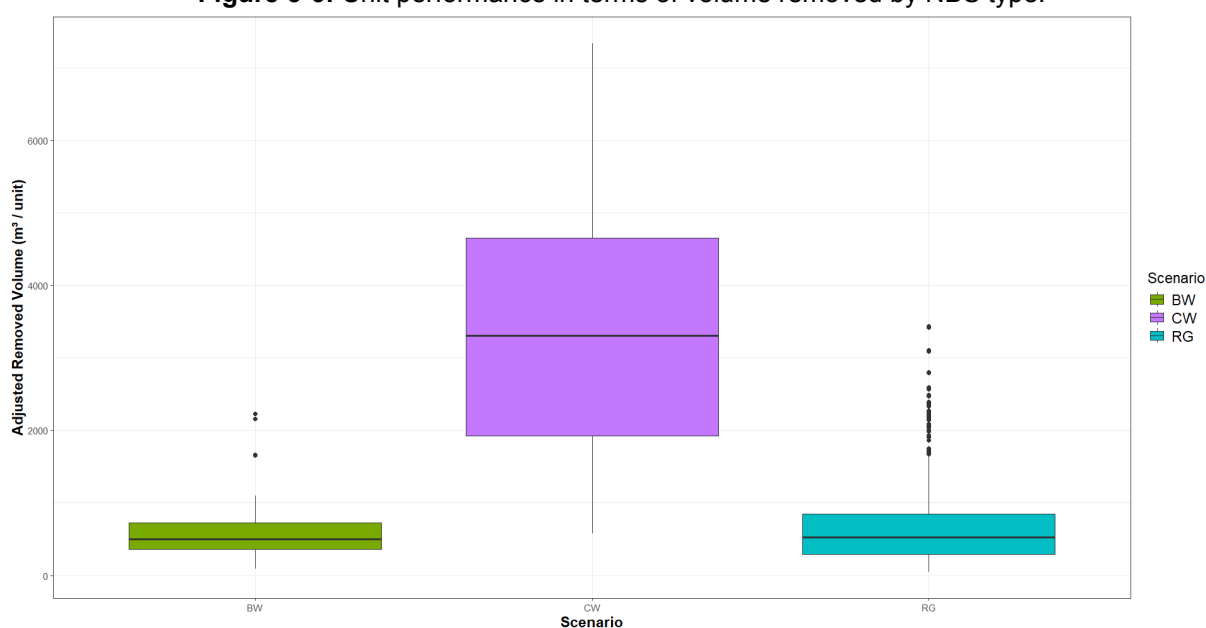
Variable/Model	Data Type	W-statistic	p-value
Removed Volume	Raw Data	0,634	$< 2,2 \cdot 10^{-16}$
	Residuals	0,964	$8,53 \cdot 10^{-16}$
Infiltration Volume	Raw Data	0,737	$< 2,2 \cdot 10^{-16}$
	Residuals	0,713	$< 2,2 \cdot 10^{-16}$
Storage Volume	Raw Data	0,440	$< 2,2 \cdot 10^{-16}$
	Residuals	0,943	$< 2,2 \cdot 10^{-16}$
—Peak Flow Reduction—	Raw Data	0,433	$< 2,2 \cdot 10^{-16}$

Peak delay	Residuals	0,939	$< 2,2 \cdot 10^{-16}$
	Raw Data	0,724	$< 2,2 \cdot 10^{-16}$
	Residuals	0,855	$< 2,2 \cdot 10^{-16}$

Source: Authors (2026).

Although both the raw data and the residuals deviated from normality according to the Shapiro-Wilk test ($p < 0,05$), due to the test's high sensitivity in large samples, graphical analysis of the residuals, following logarithmic transformation, revealed homoscedasticity (Residual vs. Fitted Plot showing a random distribution around zero) and a symmetric distribution (Q-Q Plot, with most residuals aligning with the theoretical normal distribution), conditions that allow for statistical inference in Linear Mixed-Effects Models with large datasets. Thus, by applying the LMM, it was possible to analyze the individual performance of each variable examined.

The results of the mixed-effects model revealed that the volumetric removal efficiency per unit area was influenced by both the type of NbS and the rainfall regime. Overall, it was observed that constructed wetlands (CW) exhibited the highest average volumetric removal efficiency per unit area among all the techniques analyzed (Figure 5-6). This superiority is statistically confirmed by the post-hoc analysis (Table 5-14), where CW was isolated in Group "a" with an adjusted mean of $2,128 \text{ m}^3$, a value significantly higher than those of bioswales and rain gardens (475 m^3 and 391 m^3 , respectively), both belonging to Group "b." This disparity indicates that, in terms of intrinsic efficiency, the wetland structure offers a significantly more robust retention capacity than the other individual solutions.

Figure 5-6: Unit performance in terms of volume removed by NBS type.

Source: Authors (2026).

Table 5-14: Post-hoc comparative analysis of Intrinsic Typological Efficiency for removed volume across different NBS techniques.

NBS Type	Adjusted Average Removal Volume (m³)	Standard Error (SE)	Confidence interval (95%)	Group
CW	2128	384,0	1493-3035	a
BW	475	106,0	306-736	b
RG	391	20,7	353-434	b

Source: Authors (2026).

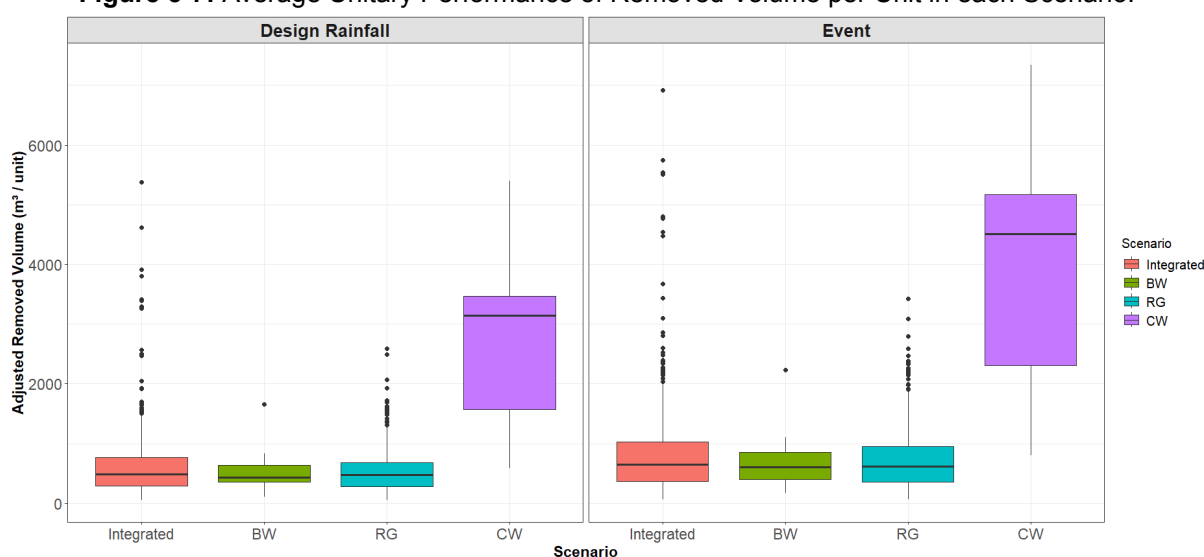
Regarding the average unit performance for each rainfall scenario (Figure 5-7), a change in the response dynamics of the solutions is observed. Under the Design Rainfall regime, the model demonstrates remarkable homogeneity: all configurations (BW Scenario, CW Scenario, RG Scenario, and Integrated Scenario) exhibit statistically equivalent performance (Group "a") in Table 5-15. This suggests that, for theoretical and controlled rainfall, the choice of arrangement (isolated vs. integrated) does not result in statistically significant differences in volume removed per unit. However, in the Event, the removal averages increased, and the BW configurations achieved the best average performance (1050 m³, Group "a"), statistically outperforming the Integrated and RG scenarios (Group "b").

The performance inversion or parity observed in higher magnitude events, where the CW scenario is no longer uniquely superior (classified as 'ab'), is justified by the stability of the operational yield per unit of scale of the bioswales. While

Wetlands achieve high absolute volumes, their marginal efficiency tends to stabilize under critical loads, exhibiting greater statistical variability, as represented by the standard error in Table 5-15. In contrast, bioswales maintain a proportionally robust response relative to their scale, allowing their relative efficiency to match or exceed that of large-scale solutions under the Linear Mixed-Effects Model adjustment. Therefore, although CWs present the highest absolute retention potential, their unitary performance is more sensitive to boundary conditions and the specific dynamics of each rainfall event.

Finally, it is important to note that the “Integrated” implementation scenario did not demonstrate statistical superiority over the standalone use of BW or CW, achieving a level of efficiency similar to that of rain gardens (RG), with both configurations being classified into the same statistical group (“b”). This behavior suggests a dampening of the overall average performance of the integrated scenario, caused by the greater numerical representation of RG units, which have the lowest average removal capacity among the evaluated typologies.

In this context, the majority of the units within these scenarios operate at a moderate efficiency level, thereby limiting the system's overall average performance. This phenomenon is likely due to the fact that, in both the Integrated and RG scenarios, the high fragmentation of the drainage areas resulted in an underutilization of the units' storage and treatment capacities. Consequently, although the integrated arrangement results in significant volumetric removal in absolute terms (total volume in m³ in the watershed), this gain did not translate into an increase in the central tendency of average efficiency. On the contrary, the high internal variance of the scenario caused episodes of high unit performance to manifest only as outliers (Figure 5-7).

Figure 5-7: Average Unitary Performance of Removed Volume per Unit in each Scenario.

Source: Authors (2026).

Table 5-15: Post-hoc comparative analysis of Strategic Scenario Efficiency for removed volume across implementation scenarios, grouped by rainfall event type.

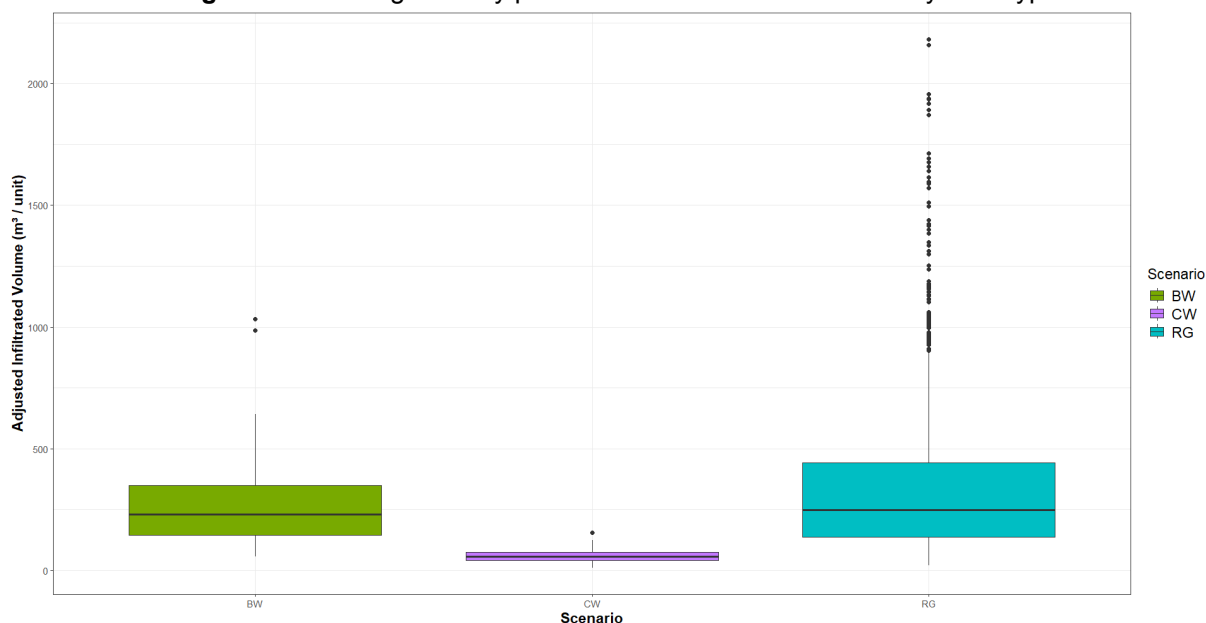
Rainfall	Scenario	Adjusted Average Removal Volume (m³)	Standard Error (SE)	Confidence interval (95%)	Group
Design Storm	BW	645	72,2	518-804	a
	CW	564	61,1	456-698	a
	RG	559	56,1	459-681	a
	Integrated	555	55,5	456-675	a
Event	BW	1050	116,0	845-1305	a
	CW	940	100,0	762-1159	ab
	Integrated	873	85,3	720-1058	b
	RG	869	85,3	716-1054	b

Source: Authors (2026).

Regarding the intrinsic unitary performance of infiltrated volume across NBS types, the model results indicated that RG and BW typologies (Group 'a') achieved statistically superior infiltration capacity compared to wetlands (Group 'b'), as shown in Figure 5-8 and Table 5-16. Although the central tendency of rain gardens is statistically equivalent to that of BWs, Figure 5-8 reveals an extensive series of upper outliers for rain gardens. The presence of these outliers indicates that specific RG units reached infiltration performances far exceeding the scenario average. This

behavior highlights the technique's sensitivity to sub-catchment boundary conditions, suggesting that when positioned in areas of high hydraulic load, rain gardens manifest an infiltration potential significantly higher than their nominal average performance. In contrast, the low variance observed in wetlands for this metric confirms their intrinsic infiltration limitation, operating more consistently but restricted to low infiltration volumes.

Figure 5-8: Average unitary performance of infiltrated volume by NBS type.



Source: Authors (2026).

Table 5-16: Post-hoc comparative analysis of Intrinsic Typological Efficiency for infiltrated volume across different NBS techniques.

NBS Type	Adjusted Average Infiltration Volume (m³)	Standard Error (SE)	Confidence interval (95%)	Group
BW	243,3	59,6	150-393,8	a
RG	234,6	12,8	211-261,1	a
CW	43,2	8,7	29-64,1	b

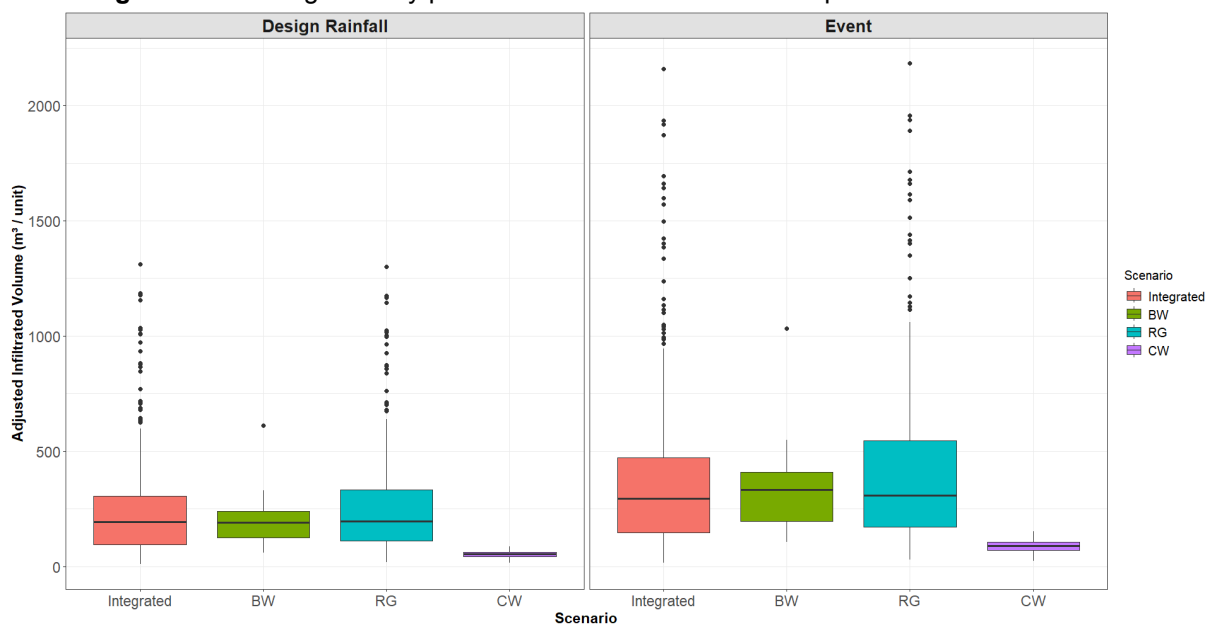
Source: Authors (2026).

In contrast, analyzing the unitary performance of infiltrated volume by scenario (Table 5-17) isolates Constructed Wetlands Scenario as the configuration with the highest adjusted mean efficiency under both rainfall regimes. This interpretation, however, is challenged by the model's residuals, where the Shapiro-Wilk test yielded a remarkably low W value of 0,713 (Table 5-13). This severe violation of the normality

assumption reveals that the Linear Mixed Model (LMM) is forcing a single, homogenous distribution onto NBS behaviors that are fundamentally distinct.

Confronting these Estimated Marginal Means (EMMeans) with the diagnostic flaw suggests that the apparent statistical superiority of the CW Scenario may be an artificial distortion caused by the model's structure, rather than a direct reflection of physical reality. In practice, the actual infiltration capacity of CW is substantially inferior to that of BW and RG. By forcing a uniform distribution across highly disparate hydraulic dynamics, the LMM tends to smooth out the high internal variance and the distinct, scale-dependent behaviors of the other typologies. Consequently, instead of capturing the system's precision, the model might be imposing a similar, artificial behavior across all scenarios. This mathematical leveling potentially overestimates the CW Scenario, positioning it as the top performer despite its lower absolute volumes, and likely masking the true operational differences between the evaluated NBS.

Figure 5-9: Average unitary performance of infiltrated volume per unit in each scenario.



Source: Authors (2026).

Table 5-17: Post-hoc comparative analysis of Strategic Scenario Efficiency for infiltrated volume across implementation scenarios, grouped by rainfall event type.

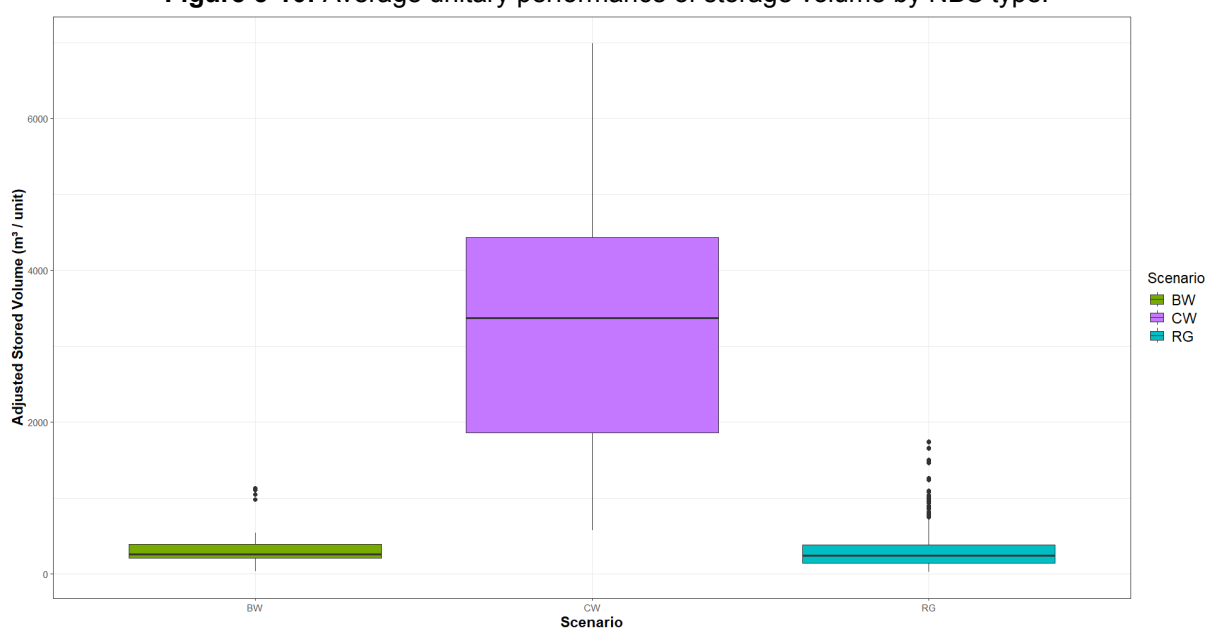
Rainfall	Scenario	Adjusted Average Removal Volume (m³)	Standard Error (SE)	Confidence interval (95%)	Group
Design Storm	CW	126,4	14,1	101,5-157	a

	BW	96,8	11,0	77,4-121	b
	Integrated	94,6	10,3	76,5-117	b
	RG	93,7	10,2	75,7-116	b
Event	CW	230,7	25,4	185,7-287	a
	BW	178,1	20,0	142,8-222	b
	RG	160,4	17,2	129,9-198	bc
	Integrated	158,6	17,0	128,5-196	c

Source: Authors (2026).

The mixed-effects model results for storage volume position Constructed Wetlands (CW) at a distinct performance level, exclusively comprising Group 'a' with an adjusted mean of 2053 m³ (Table 5-18). This magnitude is significantly higher than the means observed for bioswales (248 m³) and rain gardens (179 m³), both situated in Group 'b'. Such a disparity indicates that CW operate as large-scale accumulation units within the drainage system, while BW and RG function as limited-capacity micro-reservoirs. The visual analysis of unitary performance (Figure 5-10) reinforces this distinction in hydraulic behavior.

While CW presents a high operational range and a robust confidence interval (1393–3025 m³), BW and RG exhibit reduced dispersion with values concentrated near the bottom of the graph. This configuration highlights that wetlands possess greater storage elasticity, being capable of absorbing runoff peaks of varying magnitudes. In contrast, the low variability in the volume stored by BW and RG suggest that these techniques quickly reach their physical filling limits.

Figure 5-10: Average unitary performance of storage volume by NBS type.

Source: Authors (2026).

Table 5-18: Post-hoc comparative analysis of Intrinsic Typological Efficiency for storage volume across different NBS techniques.

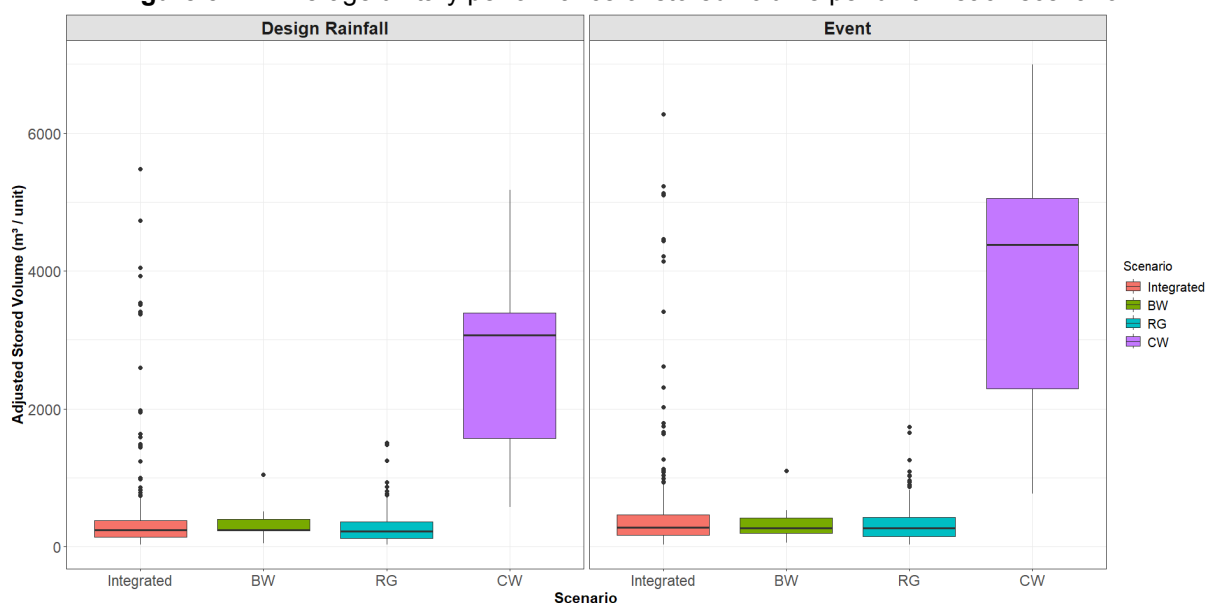
NBS Type	Adjusted Average Stored Volume (m³)	Standard Error (SE)	Confidence interval (95%)	Group
CW	2053	404,0	1393-3025	a
BW	248	59,9	154-399	b
RG	179	12,0	157-204	b

Source: Authors (2026).

While the typological analysis highlights the volumetric superiority of Constructed Wetlands, the scenario-based analysis (Table 5-19) suggests a homogeneous performance across strategies when the model adjusts variables to the scenario scale. Under this scale, all configurations are classified into Group "a" for both the Design Storm and the Real Event. Although this uniformity initially suggests a functional equivalence in capturing hydraulic loads, this statistical behavior is primarily driven by the substantial data dispersion observed within the CW scenario, resulting in an elevated standard error (SE) and a total overlap of the confidence intervals. Consequently, this widespread overlap limits the model's capacity to reject the null hypothesis, indicating that the apparent statistical equivalence among scenarios stems from data variability rather than identical physical performances. Figure 5-11 complements this view by demonstrating the operational variability of each strategy. The CW scenario exhibits a superior storage

range, which directly contributes to the high dispersion and subsequent inflation of the system's standard error. Conversely, the BW and RG strategies operate close to their capacity limits, showing low variability. Thus, from a systemic perspective, the CW scenario acts not only as a volume accumulator but as a strategy with greater design resilience and hydrological safety against rainfall uncertainties. Additionally, Figure 5-11 details that the Integrated scenario presents a notable number of upper outliers under both rainfall types. This indicates that integrating different typologies allows specific units to reach exceptional storage peaks, even though the overall system average remains balanced with the isolated strategies. In short, the results in Table 5-19 indicate that the choice between scenarios should not be based solely on the identical classification in Group "a", as this similarity is a reflection of statistical overlap caused by high variance. The real distinction lies in how this stored volume is subsequently processed, whether through infiltration or retention, and in the resilience of each structure when facing local conditions and varying hydraulic loads.

Figure 5-11: Average unitary performance of stored volume per unit in each scenario.



Source: Authors (2026).

Table 5-19: Post-hoc comparative analysis of Strategic Scenario Efficiency for storage volume across implementation scenarios, grouped by rainfall event type.

Rainfall	Scenario	Adjusted Average Removal Volume (m³)	Standard Error (SE)	Confidence interval (95%)	Group
Design Storm	BW	452	68,9	335-609	a
	RG	360	41,5	287-452	a

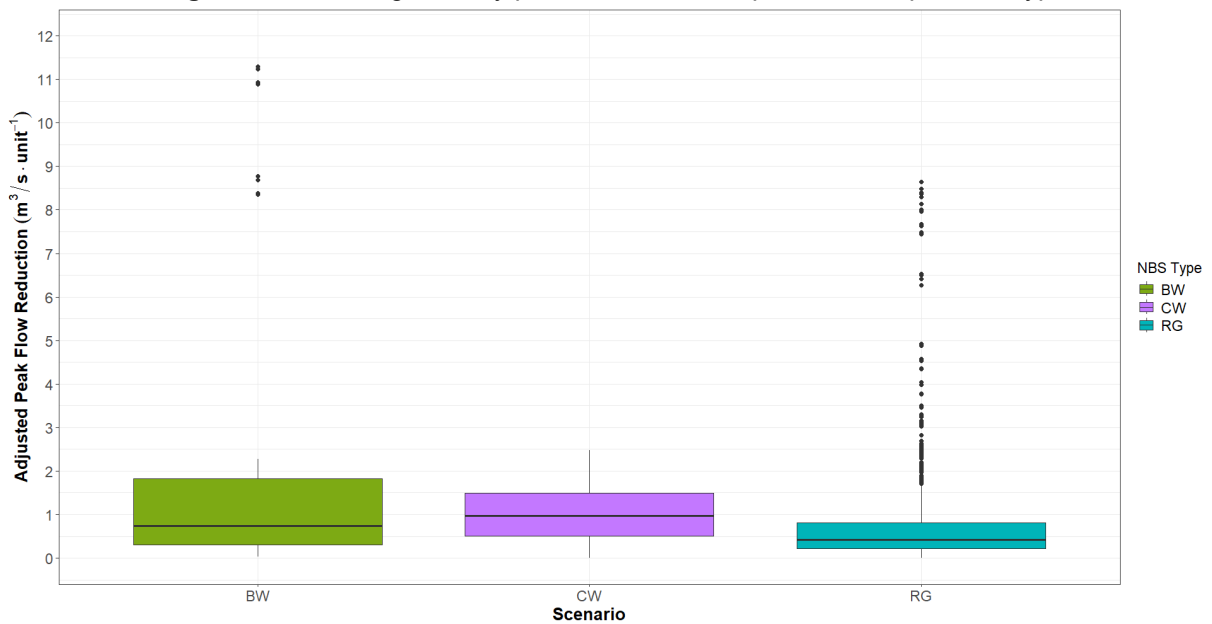
	Integrated	357	40,6	285-447	a
	CW	342	48,2	259-451	a
Event	BW	582	87,2	434-781	a
	CW	573	78,6	438-750	a
	Integrated	506	54,7	409-626	a
	RG	503	55,1	406-624	a

Source: Authors (2026).

Unlike the results observed for stored volume, the analysis of peak unit flow reduction (Table 5-20) ranks bioswales as the type with the highest statistical performance (Group 'a'), with an adjusted average reduction of 1.49 m³/s. This result suggests that, although bioswales have lower gross volumetric capacity, their operational configuration and proximity to runoff-generating surfaces give them superior effectiveness in the immediate attenuation of floods.

The CWs showed intermediate performance (Group 'ab', average of 1.12 m³/s), indicating that their large storage capacity does not necessarily translate into a proportionally greater reduction in peak flow compared to decentralized techniques. RG, on the other hand, showed the lowest relative performance (Group 'b'), reducing the peak by 0,72 m³/s.

The observation in Figure 5-12 corroborates this hierarchy, highlighting that, although BW and RG exhibit several higher outliers (indicating units with exceptional performance in specific cases), the adjusted marginal average favors the response of bioswales.

Figure 5-12: Average unitary peak flow reduction performance per NBS type.

Source: Authors (2026).

Table 5-20: Post-hoc comparative analysis of Intrinsic Typological Efficiency for peak flow reduction across different NBS techniques.

NBS Type	Adjusted Peak Flow Reduction (m³/s)	Standard Error (SE)	Confidence interval (95%)	Group
BW	1,490	0,280	0,995-2,107	a
CW	1,128	0,195	0,777-1,550	ab
RG	0,726	0,051	0,629-0,829	b

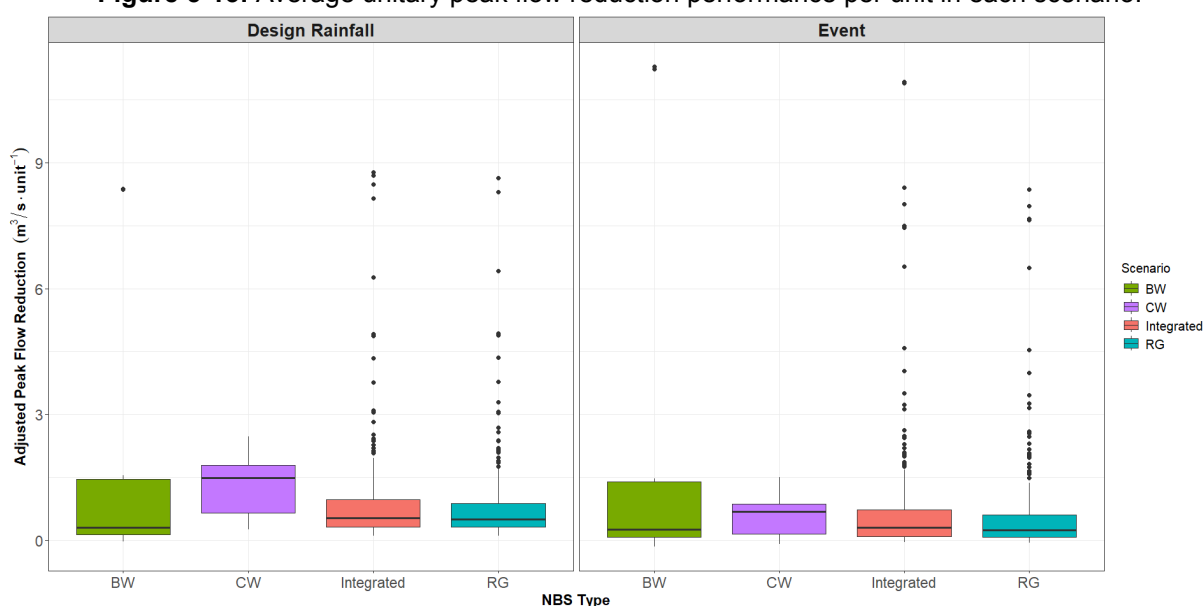
Source: Authors (2026).

The analysis of scenario-based peak flow reduction (Table 5-21) reveals a fundamental distinction between intrinsic technical potential and systemic performance. During the Design Storm, the CW Scenario, RG Scenario, and Integrated Scenario showed superior performance (Group 'a'), while the BW Scenario demonstrated significantly lower attenuation capacity (Group 'b'). This divergence reflects the sensitivity of the mixed-effects model to sampling density and spatial arrangement: while individual BW units are highly efficient in unitary terms (Table 5-20), the low unit density in the scenario ($n=11$) limits their representativeness within the systemic mean. Because the scenario-based post-hoc evaluates the average performance per installed unit within each strategy's context, the BW's performance is diluted by higher statistical uncertainty and lower spatial coverage compared to high-density scenarios like RG ($n=244$).

However, a crucial inversion occurs during the Event: the Integrated and RG scenarios consolidated as the most effective strategies (Group 'a'), while the CW scenario shifted to Group 'b'. Figure 5-13 provides the visual evidence for this phenomenon. The statistical drop of the CW scenario highlights its operational instability when facing a real hydrological event, despite its larger individual volumetric capacity.

In contrast, the Integrated and RG scenarios maintain a notable number of upper outliers in Figure 5-13. This indicates that the superior efficacy of these strategies during the Real Event does not stem from a uniform performance across all units, but rather from their spatial distribution. These configurations successfully intercept the runoff at multiple points along its flow path simultaneously. In summary, the hydraulic robustness of the Integrated scenario demonstrates that the spatial complementarity between different typologies outperforms the isolated implementation of large-scale storage structures when facing real-world hydrological uncertainties.

Figure 5-13: Average unitary peak flow reduction performance per unit in each scenario.



Source: Authors (2026).

Table 5-21: Post-hoc comparative analysis of Strategic Scenario Efficiency for peak flow reduction across implementation scenarios, grouped by rainfall event type.

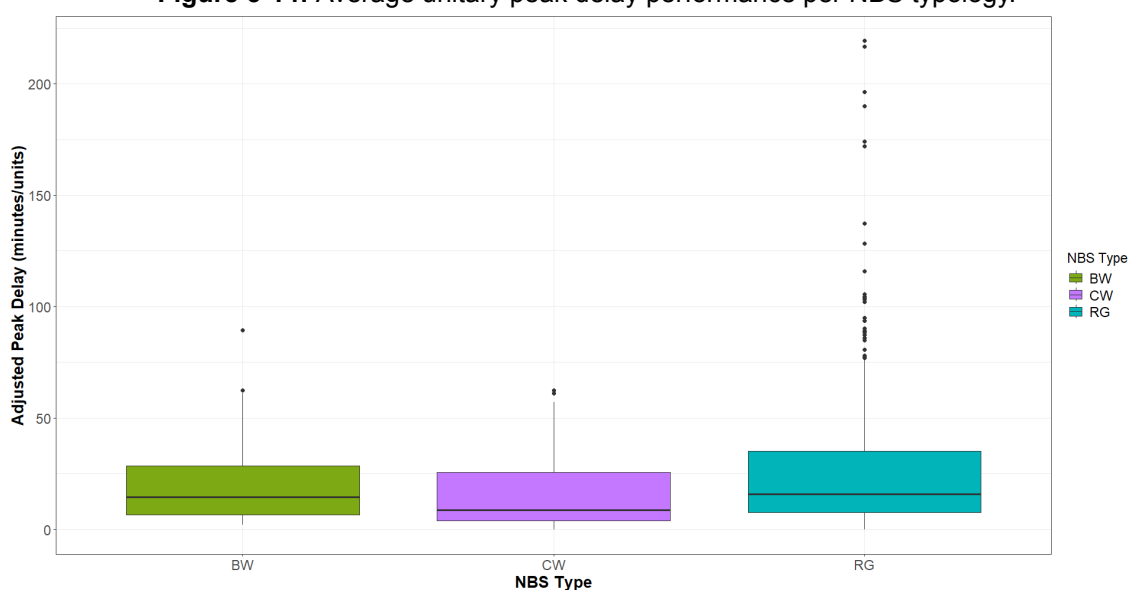
Rainfall	Scenario	Adjusted Peak Flow Reduction (m ³ /s)	Standard Error (SE)	Confidence interval (95%)	Group
Design Storm	CW	1,733	0,167	1,425-2,082	a

	RG	1,602	0,137	1,347-1,885	a
	Integrated	1,598	0,135	1,345-1,879	a
	BW	0,830	0,119	0,610-1,079	b
Event	Integrated	1,010	0,101	0,821-1,218	a
	RG	0,992	0,101	0,803-1,200	a
	CW	0,721	0,103	0,530-0,935	b
	BW	0,572	0,101	0,386-0,782	b

Source: Authors (2026).

Regarding peak flow delay (Table 5-22), the post-hoc analysis did not identify any statistically significant differences among the types of NbS, classifying all techniques into Group 'a'. Although the Rain Garden (RG) had the highest adjusted mean (36,8 min), the high standard errors observed for CW (SE=10,6) and BW (SE=9,3) indicate high internal variability that prevents a hierarchical distinction between the solutions.

Figure 5-14 corroborates this parity, revealing similar medians among the techniques. However, the presence of pronounced outliers in the RG scenario stands out, with units reaching delays exceeding 200 minutes. This behavior suggests that, for the peak delay metric, the specific location of the unit within the watershed and its individual configuration may be more influential determinants than the type of NbS itself, resulting in similar average unit performance but with greater potential for exceptions in the RG scenario.

Figure 5-14: Average unitary peak delay performance per NBS typology.

Source: Authors (2026).

Table 5-22: Post-hoc comparative analysis of Intrinsic Typological Efficiency for peak flow delay across different NBS techniques.

NBS Type	Adjusted Peak Flow Delay (min)	Standard Error (SE)	Confidence interval (95%)	Group
RG	36,8	3,72	30,1-44,8	a
CW	31,4	10,60	16,0-60,9	a
BW	24,5	9,34	11,4-51,5	a

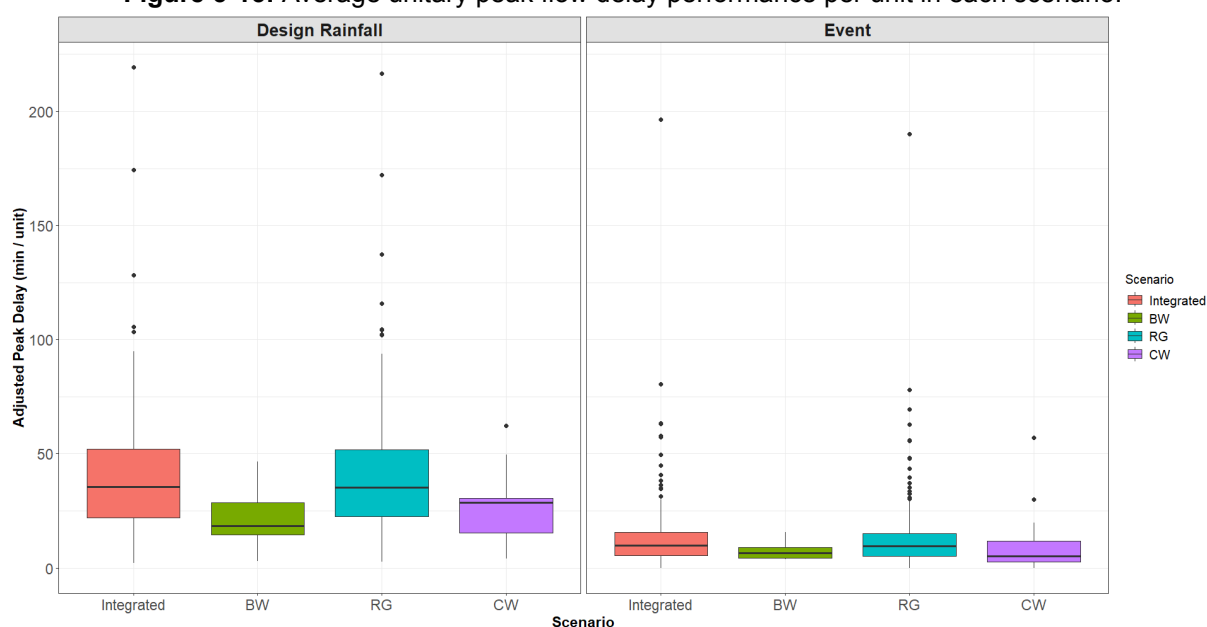
Source: Authors (2026).

Regarding the scenario-based peak flow delay (Table 5-23), it is important to first note that the Shapiro-Wilk test on the residuals yielded a lower value ($W = 0,855$; Table 5-13). This violation of the normality assumption suggests that the model may be smoothing out highly distinct hydraulic processes to fit a single distribution. Under the Design Storm regime, within this modeled context, the CW, Integrated, and RG scenarios established themselves as superior, sharing Group "a", while the BW scenario demonstrated a significantly lower performance, falling into Group "b". This behavior indicates that, to delay the flood wave in high-magnitude events, the system relies on two distinct mechanisms: either the large individual volumetric storage capacity of the CW, or the dense network of the RG and Integrated scenarios that delays the runoff at multiple points along its flow path simultaneously.

When transitioning to the Real Event, a generalized reduction in delay times is observed, yet the efficiency hierarchy persists. The Integrated and RG scenarios

exhibited the greatest temporal resilience, maintaining their position in Group "a". This reinforces that the spatial distribution of multiple units in the RG and Integrated scenarios allows them to successfully intercept and delay the runoff at various points of the catchment trajectory simultaneously. Figure 5-15 corroborates this behavior, demonstrating that although delay medians are reduced in real events, the high capillarity of the RG and Integrated scenarios enables the maintenance of high-performance outliers. Meanwhile, the BW scenario remains constrained by the low numerical representativeness of its units and the lack of integration with other interventions.

Figure 5-15: Average unitary peak flow delay performance per unit in each scenario.



Source: Authors (2026).

Table 5-23: Post-hoc comparative analysis of Strategic Scenario Efficiency for peak flow delay across implementation scenarios, grouped by rainfall event type.

Rainfall	Scenario	Adjusted Peak Flow Delay (min)	Standard Error (SE)	Confidence interval (95%)	Group
Design Storm	CW	107,93	22,80	71,16-163,4	a
	Integrated	105,87	19,00	74,37-150,5	a
	RG	104,59	18,80	73,37-148,9	a
	BW	35,25	7,34	23,36-52,9	b
Event	Integrated	14,45	2,61	10,08-20,5	a
	RG	13,96	2,54	9,70-19,9	a

	CW	11,42	2,39	7,50-17,1	a
	BW	6,57	1,48	4,16-10,1	b

Source: Authors (2026).

5.3.4 Sensitivity analyses

The results of the sensitivity analysis reveal selective sensitivity, where the system's equilibrium is governed by a limited set of critical variables, with particular emphasis on the impermeabilization percentage and the berm height of the edge of the Rain Garden units, while most of the design parameters of Nature-Based Solutions (NbS) have a marginal impact on the final outfall (Table 5-24). The system's response to impervious surfaces was asymmetric, indicating that the watershed is more sensitive to de-imperviousing measures than to incremental urban densification, possibly because many drainage areas already have a high percentage of impervious surfaces. Furthermore, the Manning coefficient of permeable areas had a greater influence on peak flow and infiltration than that of impervious areas, since the roughness of natural soil more drastically alters the watershed's concentration time.

Table 5-24: Qualitative assessment of parameter influence on runoff and storage variables.

Parameter	Maximum Peak Flow	Total Volume	Infiltration	Storage	Outflow
% Imperv	High*	High	High	High	High
Width	Medium	Low	Low	Low	Low
Minimum Infiltration Rate	Medium	Medium	High	Medium	Medium
Maximum Infiltration Rate	Low	Low	Low	Low	Low
Berm Height - Rain Gardens	Medium*	High*	High*	High*	High*
Berm Height - Bioswales	Low	Low	Low	Low	Low
Soil Thickness - Rain Gardens	Low	Low	Low	Medium*	Low
Soil Thickness - Bioswales	Low	Low	Low	Low	Low
Area - Wetlands	Low	Low	Low	Low	Low
Initial Depth - Wetlands	Low	Low	Null	Low	Low
Manning's roughness (n-imperv)	High	Low	Low	Low	Low
Manning's roughness (n-perv)	High	Medium	High	Medium	Medium

* Asymmetry

High: > 10% | Medium: 5-10% | Low: < 5% | Null: 0%

Source: Authors (2026).

Regarding green infrastructure, the berm height of rain gardens has proven to be a critical and asymmetric parameter; reductions in this storage capacity result in significant impacts on surface runoff, suggesting that current design operates close to the overflow threshold. On the other hand, variables such as the soil depth of Rain Gardens act primarily as regulators of internal storage, with limited influence on the final outflow. Parameters related to Wetlands and Bioswales showed predominantly low sensitivity, exerting marginal control over the overall water balance. In summary, the results indicate that the system's efficiency depends critically on the connectivity of impervious surfaces and the physical integrity of the accumulation zones within the bioretention units. Table 5-25 provides a summary of the main variables affected by changes in the analyzed parameters, broken down by sensitivity level.

Table 5-25: Summary of the sensitivity analysis results.

Sensibility	Parameters	Summary of the main variables affected
High	% Imperv, Berm Height (Rain Gardens)	These are the only ones showing a variation of more than 10% in nearly all metrics, affecting Maximum Peak Flow, Total Volume and Outflow. These parameters alter the global water balance of the system.
Medium	Manning's n (imperv/perv), Width, Min. Infiltration Rate	The Manning coefficient (n-imperv) acts as a point regulator of peak flow. In contrast, n-perv and the infiltration rate exhibit intermediate distributed sensitivity, influencing the infiltration balance without reaching critical levels of variation in total volume.
Low	Soil Thickness (Rain Gardens) Max. Infiltration Rate, Berm Height (Bioswales), Soil Thickness (Bioswales), Area (Wetlands), initial Depth (Wetlands)	Parameters with predominantly low variations (<5%). They do not exert significant control over runoff or total volume, showing sensitivity only in very limited indicators of storage or infiltration.

Source: Authors (2026).

5.4 Discussions

The analysis of the results revealed the different effects of Nature-based Solutions (NbS) across multiple scales within the studied hydrological watershed. It was evidenced that the volumetric efficiency of the scenarios observed at the outlet scale may conceal significant impacts occurring at the drainage area level. This discrepancy reveals that the watershed's hydrological effectiveness is governed by a binomial relationship between the individual magnitude of the interventions and their strategic distribution across the territory. While the gross storage capacity ensures volumetric control at downstream control sections, it is the allocation and intrinsic potential of each technique that promote the recovery of hydrological functions in the regions upstream of the watershed outlet.

One example was the BW Scenario, which, despite presenting a reduced systemic impact at the outlet due to the small number of units, was decisive at the local level, reclassifying six areas from 'high' to 'medium' imperviousness and one to 'low'. This result corroborates studies such as those by Loperfido et al. (2014) and Tansar, Duan & Mark (2022), which emphasize the importance of strategic distribution to maximize hydrological benefits.

Complementing the spatial distribution, the Integrated Scenario exhibited a hydraulic synergy effect, presenting a number of runoff-free sub-watersheds slightly greater than the sum of the individual scenarios. This dynamics suggests a mutual aid effect, where an NbS positioned upstream alleviates the hydraulic load for the subsequent unit, optimizing the system's total retention. Another notable finding of this study was that the volumetric magnitude of only 17 Wetlands (centralized strategy) matched the capillary network of 244 Rain Gardens (decentralized strategy). This volumetric balance demonstrates that planning can achieve similar storage goals through different design pathways (Garbanzos and Maniquiz-Redillas, 2022), acknowledging that performance depends not only on the number of units, but on the contributing area and storage capacity of each typology (Wu et al., 2018).

The post-hoc analysis for the NbS typologies reveals that the units operate under distinct hydraulic specializations. The CW established themselves as large-scale accumulation units, holding isolated statistical superiority in removed volume and storage. In contrast, BW and RG exhibit superior intrinsic effectiveness in peak flow reduction and nominal infiltration potential. When translating these results

to the scenario scale, an inversion between nominal potential and systemic resilience is observed. Although CW possess lower unit infiltrative potential, the CW Scenario demonstrated greater operational stability for infiltration, suggesting that large-scale storage provides a constant hydraulic head that outweighs the storage of the RG. Regarding peak delay, the statistical parity between typologies indicates that flow retardation was more influenced by the local boundary conditions of the sub-watershed than by the individual technology.

Furthermore, the transition from design rainfall to the actual event revealed the operational limits of the infrastructures. Although the absolute volume removed was higher in the actual event due to the high hydraulic head, the relative performance declined because of porous media saturation. The post-hoc ranking of the 'best scenario' is dynamic and may change according to the return period: an ideal solution for frequent events may lose efficiency in extreme events, where infiltration becomes the limiting factor (He et al., 2022). This evidence suggests that NbS effectiveness should not be evaluated solely by its average capacity, but by its stability under hydrological stress, reinforcing the need to use long-term historical series (Garbanzos and Maniquiz-Redillas, 2022).

5.4.1 Study Limitations and Future Perspectives

The apparent homogeneity resulting from the loss of statistical significance observed in some results of the post-hoc analyses stems, in part, from data insufficiency and unbalanced sampling in certain categories. This is especially evident in the less representative typologies (CW and BW), whose reduced sample sizes may have prevented the manifestation of significant differences in metrics where high internal variability limits the power of the mixed-effects model to correctly isolate the superiority of the scenarios. As pointed out by Bai et al. (2018), this variability reflects the fact that NbS effectiveness depends drastically on site-specific conditions, where different types of LID (Low Impact Development) exhibit varied performances depending on the specific location within the watershed.

Uncertainties are inherent to NbS performance, with retention rates ranging from 11% to 99% (Kõiv-Vainik et al., 2022). To overcome the lack of significance in traditional p-values, the application of decision-making approaches based on fuzzy logic, as proposed by Ni et al. (2025), would allow for a more faithful interpretation. Through the normalization of indicators and the use of algorithms such as TOPSIS, it

would be possible to identify incremental improvement trends and evaluate operational stability under different hydraulic loading conditions (Ni et al., 2025).

In this context, the simulation of only two precipitation scenarios acts as an additional restrictive factor. Considering a greater number of rainfall conditions and conducting further experiments are fundamental to obtaining clearer and more reliable results. The inclusion of rainfall events covering different intensities and frequencies would allow a reduction in uncertainties, given that the performance of NbS and scenarios is non-linear and varies significantly according to rainfall duration, intensity, and depth.

Additionally, the variation of imperviousness parameters (% Imperv) and Manning's roughness (n-imperv) decisively affected the peak flow and total volume (Cavalcanti, 2020; Silveira et al., 2022). Furthermore, the observed sensitivity of Width and infiltration/storage parameters aligns with the findings of Jiang et al. (2025), reinforcing that these are the governing mechanisms for volumetric reduction in the SWMM model.

In addition, the sensitivity of the SWMM model to infiltration, storage, and surface roughness parameters highlights the importance of calibration procedures using local experimental data. The incorporation of monitoring data from pilot NbS areas within the study watershed could substantially improve the reliability and representativeness of future simulations.

Finally, future studies should incorporate long-term hydrological series to evaluate the resilience and consistency of NbS performance over time, especially under extreme rainfall events and climate variability conditions. Such advancements would be particularly relevant to support urban watershed management and decision-making regarding the planning and implementation of nature-based interventions.

5.5 Conclusions

The results demonstrated that the effectiveness of Nature-based Solutions (NbS) in urban watersheds depends on both the magnitude of interventions and their strategic spatial distribution. Analyses focused only on watershed outlets may overlook important local hydrological benefits, reinforcing the importance of multi-scale planning approaches.

The Integrated Scenario showed a hydraulic synergy effect, with combined NbS arrangements promoting greater runoff retention than isolated interventions. NbS performance also proved to be dynamic and sensitive to rainfall conditions, especially under intense events where porous media saturation reduced infiltration efficiency.

Finally, the study highlighted uncertainties associated with limited rainfall scenarios, unbalanced typologies, and the sensitivity of SWMM parameters such as imperviousness, Width, Manning's roughness, infiltration rate, and Berm Height. Future studies should incorporate field monitoring, local calibration, and long-term hydrological analyses to improve model reliability and support more robust NbS planning in urban watersheds. Such advancements are fundamental to provide more robust guidelines to support planning and prioritization of areas for NbS implementation in consolidated urban contexts.

5.6 References

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Chapter 6:

Conclusions and Recommendations

This study demonstrates that the resilience of urban watersheds is governed by a synergy between individual nature-based interventions and their strategic territorial distribution. The geoenvironmental and hydrological analysis of the Água Comprida stream watershed reveals that its environmental challenges are deeply rooted in physical constraints coupled with unplanned urban occupation. This research demonstrates that establishing a resilient urban environment requires planning measures strictly aligned with the watershed's intrinsic characteristics. By integrating geoenvironmental diagnosis, automated computational routines, and hydrological modeling, this study provides a comprehensive framework for the strategic implementation of Nature-based Solutions (NbS).

A significant methodological advancement was the development of the automated preprocessing routine, which addressed a critical gap in NbS literature regarding the need for systematic hydrological preprocessing at the watershed scale. By enabling the automated extraction of model input parameters, for 330 drainage units, the tool ensures efficiency and reproducibility. Future perspectives for this tool should focus on the incorporation of machine learning methods to enhance land use classification and the development of graphical user interfaces to make the routine accessible to a broader range of users. Additionally, integration with real-time data from hydrometeorological sensors could enhance dynamic environmental monitoring and management.

Regarding implementation, while NbS are technically feasible, their application in consolidated urban environments faces significant spatial and regulatory hurdles. Rain gardens emerged as the most adaptable typology, whereas wetlands and bioswales were heavily constrained by urban density. Consequently, there is an urgent need to integrate NbS into urban governance, updating building codes and urban master plans to resolve technical incompatibilities, such as standard sidewalk configurations versus green infrastructure requirements. Municipalities should also promote financial incentives for private property adoption and participatory governance mechanisms that involve the community in the design and maintenance of these systems.

Furthermore, the hydrological modeling results lead to the conclusion that NbS effectiveness is governed by the relationship between individual magnitude and strategic territorial distribution. A key finding is that downstream volumetric efficiency often conceals significant local-scale improvements in hydrological functions. The integrated scenario revealed a clear hydraulic synergy, where the serial arrangement of units optimized total retention beyond the simple sum of individual contributions. Therefore, planning and flood control policies should move away from reductionist outlet-only analyses toward a multi-scale vision that prioritizes local recovery and evaluates infrastructure stability under severe stress rather than just average performance.

Finally, access to quality spatial and environmental information was fundamental to this research, although the need for finer scales to capture subtle spatial variations remains. Future research should prioritize the creation and subsequent long-term monitoring of NbS pilot areas, as the current absence of these structures limits the empirical validation of their real-world efficiency. It is indispensable to obtain high-resolution data and in situ measurements to enhance model accuracy and refine sensitive parameters, such as imperviousness and berm height. Furthermore, to overcome the rigidity of traditional statistical p-values, it is recommended to significantly expand the number of simulated precipitation events, encompassing a broader range of return periods and rainfall patterns. Integrating these expanded datasets with decision-making approaches like Fuzzy Logic will allow for a better capture of incremental improvement trends and a more nuanced understanding of system resilience. By combining spatial analysis, automated preprocessing, and strategic planning, this research establishes a transferable framework to support more resilient and sustainable urban water management strategies.

Chapter 7:

Academic background of the author

7.1 Academic Profile

The author is a master's student in the Graduate Program in Civil and Environmental Engineering at São Paulo State University (UNESP), with a focus on Environmental Sanitation. Holding a degree in Civil Engineering, the author has continued to deepen technical and scientific knowledge through graduate-level coursework in Unit Operations Applied to Water and Wastewater Treatment, Solid Waste, Rehabilitation of Degraded Areas, and Water Pollution, totaling 32 credits.

7.2 PAADES Program

Through the PAADES (Graduate Student Improvement Program), the author contributed as an instructor for Civil Engineering students in the Hydrology and Basic Sanitation courses. In Hydrology (22 hours, first semester of 2024), the activities involved designing and delivering practical classes focused on the use of the software QGIS tools to connect data obtained from government platforms such as TOPODATA and DATAGeo. Students were guided to delineate watersheds, extract basin characteristics (e.g., thalweg, elevation difference, area), and generate maps as part of their assignments. To support these practical activities, the author provided dedicated office hours for clarifying student doubts. Additionally, the author was responsible for administering and grading the first (P1) and second (P2) scheduled exams.

In the Basic Sanitation course (32 hours, second semester of 2024), the author developed and taught theoretical lectures covering the components of water supply systems, population estimation, and the calculation of design flow rates for each system segment. Practical activities were also organized to reinforce theoretical topics, with guidance provided to students in solving applied exercises. Within this context, the author conducted the application and grading of the second scheduled exam (P2) and the final examination.

7.3 Research Projects

As part of academic training, the author collaborated on the SACRE project (Integrated Solutions for Resilient Cities), developed by the University of São Paulo (USP) and funded by FAPESP. Activities involved conducting fieldwork, such as topographic surveys and groundwater sampling, in support of studies on Riverbank Filtration (RBF) and Phytoremediation (WP2). These efforts contributed to the development of applied skills in groundwater sampling, data collection and environmental monitoring.

7.4 Conference Presentations

Research findings have been presented at both national and international scientific events. At the XXIII Brazilian Congress on Groundwater (ABAS 2024), the topic addressed was the impact of municipal sanitation management on urban water security. A geo-environmental assessment of the Água Comprida watershed was presented at the Mediterranean Geosciences Union Annual Meeting (MedGU 2024), focusing on diagnostic and prognostic analyses. Recognition was also received at Women's Research Engineers Network (WREN) Symposium 2025, where the author earned the Best Presentation Award for a systematic review on aquifer monitoring methodologies in areas impacted by solid waste disposal. Earlier research was also shared at the Latin American Symposium on Geotechnologies and Environmental Studies, under the title: "Geoprocessing as a Tool for Predicting the Occurrence of Linear Water Erosion Based on Surface Runoff Concentration Patterns at Hillslope Scale."

7.5 Publications

The author's scientific output includes publications in peer-reviewed journals. In the *National Journal of City Management* (2024), geospatial tools were applied to predict linear water erosion. Previous studies developed during undergraduate research were published in *Holos* (2023) and *Green Cities* (2022), both focusing on the spatiotemporal behavior of erosive processes in urban landscapes.

7.6 Additional Activities

Additional professional development in 2024 includes certifications in *2D HEC-RAS Modeling* (offered by the Brazilian National Water Agency – ANA) and *Environmental Risk Management* (UN Environment/PEDRR). In 2025, the author was

part of the organizing committee for SIPAS (São Paulo International Groundwater Week), contributing to the coordination of a large-scale academic and professional event.

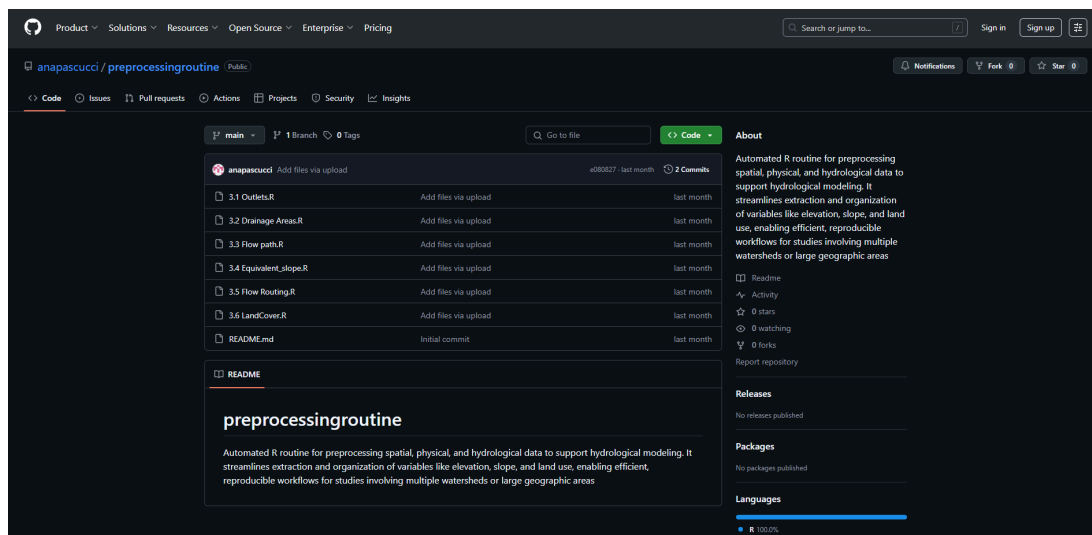
(Certificates and receipts are attached in the Appendix.)

Chapter 8: Appendix

8.1 Supplementary Material relating to Chapter 3 - Extracting, Structuring and Organizing Spatial, Physical and Hydrological Data for Modeling: an Approach Using Rstudio

GitHub is an online platform focused on source code storage and version control, based on the Git system, which tracks the history of changes in files, allowing users to monitor all modifications made. With GitHub, developers and teams can collaborate efficiently on projects, sharing code, managing changes, fixing bugs, and developing new features in an organized manner.

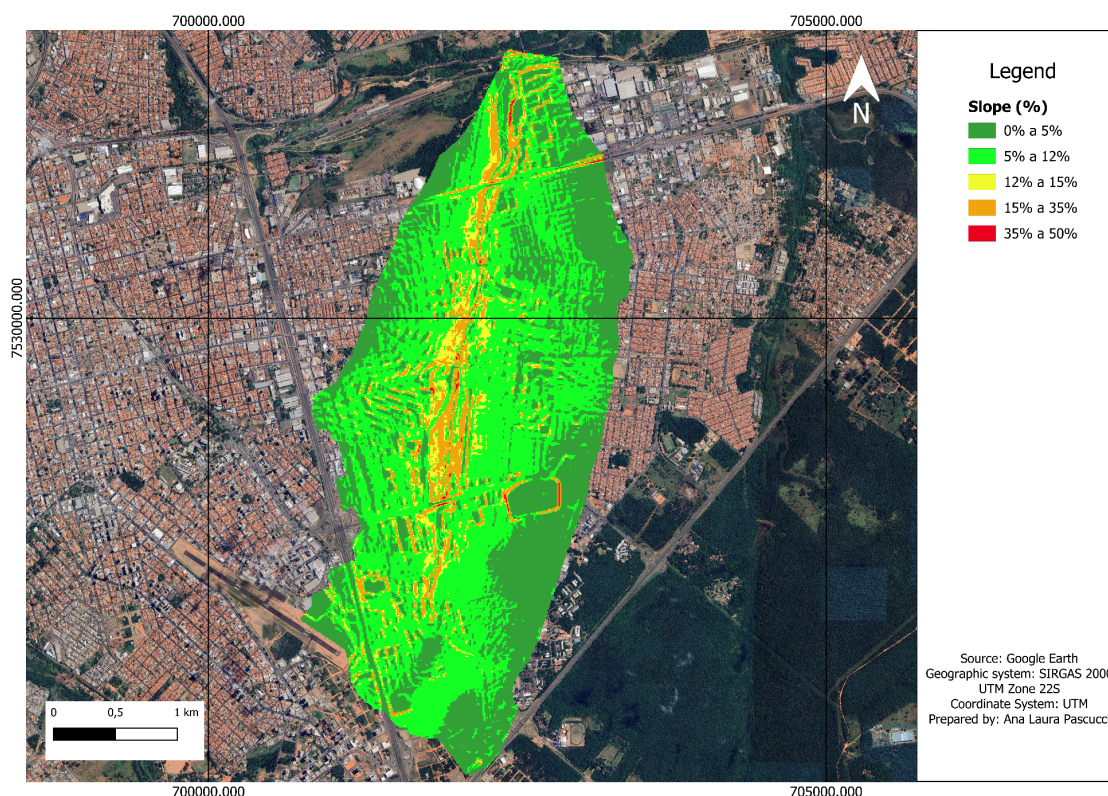
Figure 8-S1. Screenshot of the preprocessing routine repository interface on GitHub. Available at <https://github.com/anapascucci/preprocessingroutine>.



Source: Authors (2025).

8.2 Supplementary Material relating to Chapter 4 - Nature-based Solutions for Surface Water Management in an Urban Watershed: Criteria for Site Selection and Automated Data Pre-processing for Hydrological Modeling

Figure 8-S2. Slope map of the Água Comprida stream watershed.



Source: Authors (2025).

Table 8-S1. Characterization of drainage areas: total drainage area values, type of NbS located, the maximum and minimum limits recommended for each type of solution, and the area available for implementation in each drainage area. The available area values that do not meet the minimum required criteria have been highlighted in red.

Drainage Area ID	Drainage Area (m ²)	NbS Type	Maximum Drainage Area (m ²)	Minimum Drainage Area (m ²)	NbS Available Area (m ²)	NbS Minimum Area (m ²)
S1	10175,87	Rain Garden	20234,3	-	2887,03	508,79
S2	5215,78	Rain Garden	20234,3	-	1501,64	260,79
S3	20162,12	Rain Garden	20234,3	-	1775,12	1008,11
S4	2109,31	Rain Garden	20234,3	-	512,774	105,47
S5	12368,28	Rain Garden	20234,3	-	2199,02	618,41
S6	8296,87	Rain Garden	20234,3	-	756,594	414,84

Drainage Area ID	Drainage Area (m²)	NbS Type	Maximum Drainage Area (m²)	Minimum Drainage Area (m²)	NbS Available Area (m²)	NbS Minimum Area (m²)
S7	19465,15	Rain Garden	20234,3	-	4993,19	973,26
S8	18728,86	Rain Garden	20234,3	-	3148,1	936,44
S9	8759,08	Rain Garden	20234,3	-	1327,44	437,95
S10	9860,99	Rain Garden	20234,3	-	1184,85	493,05
S11	10478,79	Rain Garden	20234,3	-	3725,27	523,94
S12	13170,87	Rain Garden	20234,3	-	2600,13	658,54
S13	17179,23	Rain Garden	20234,3	-	2767,77	858,96
S14	20039,23	Rain Garden	20234,3	-	1504,87	1001,96
S15	16770,58	Rain Garden	20234,3	-	871,306	838,53
S16	11713,07	Rain Garden	20234,3	-	1516,14	585,65
S17	16347,69	Rain Garden	20234,3	-	925,086	817,38
S18	5587,30	Rain Garden	20234,3	-	343,854	279,36
S19	168133,96	-	-	-	-	-
S20	20184,02	Rain Garden	20234,3	-	1653,96	1009,20
S21	8403,04	Rain Garden	20234,3	-	468,476	420,15
S22	8756,50	Rain Garden	20234,3	-	441,515	437,82
S23	10698,61	Rain Garden	20234,3	-	710,412	534,93
S24	7823,77	Rain Garden	20234,3	-	979,074	391,19
S25	6978,02	Rain Garden	20234,3	-	436,669	348,90
S26	11119,91	Rain Garden	20234,3	-	565,324	556,00
S27	12491,25	Rain Garden	20234,3	-	797,65	624,56
S28	5985,63	Rain Garden	20234,3	-	1398,23	299,28
S29	6460,83	Rain Garden	20234,3	-	2542,28	323,04
S30	11917,99	Rain Garden	20234,3	-	701,497	595,90
S31	20122,43	Rain Garden	20234,3	-	1354,48	1006,12
S32	4528,42	Rain Garden	20234,3	-	939,923	226,42

Drainage Area ID	Drainage Area (m²)	NbS Type	Maximum Drainage Area (m²)	Minimum Drainage Area (m²)	NbS Available Area (m²)	NbS Minimum Area (m²)
S33	10467,16	Rain Garden	20234,3	-	842,6	523,36
S34	17612,55	Rain Garden	20234,3	-	1069,58	880,63
S35	6167,75	Rain Garden	20234,3	-	626,767	308,39
S36	3426,35	Rain Garden	20234,3	-	117,593	171,32
S37	22576,94	Bioswales	40468,6	-	1839,08	225,77
S38	19161,98	Rain Garden	20234,3	-	1005,19	958,10
S39	8215,34	Rain Garden	20234,3	-	1036,73	410,77
S40	5658,80	Rain Garden	20234,3	-	330,177	282,94
S41	97195,37	-	-	-	-	-
S42	6758,27	Rain Garden	20234,3	-	923,905	337,91
S43	10746,70	Rain Garden	20234,3	-	2293,45	537,33
S44	2186,43	Rain Garden	20234,3	-	588,49	109,32
S45	8495,59	Rain Garden	20234,3	-	2175,13	424,78
S46	9202,06	Rain Garden	20234,3	-	358,295	460,10
S47	6057,51	Rain Garden	20234,3	-	655,804	302,88
S48	20201,72	Rain Garden	20234,3	-	1185,27	1010,09
S49	4494,17	Rain Garden	20234,3	-	338,678	224,71
S50	1875,50	Rain Garden	20234,3	-	219,54	93,77
S51	7957,27	Rain Garden	20234,3	-	1378,26	397,86
S52	2066,81	Rain Garden	20234,3	-	647,606	103,34
S53	15146,19	Rain Garden	20234,3	-	2164,4	757,31
S54	17337,85	Rain Garden	20234,3	-	980,464	866,89
S55	13639,38	Rain Garden	20234,3	-	1520,29	681,97
S56	16711,11	Rain Garden	20234,3	-	1010,71	835,56
S57	16586,09	Rain Garden	20234,3	-	12508	829,30
S58	7115,30	Rain Garden	20234,3	-	886,688	355,77

Drainage Area ID	Drainage Area (m²)	NbS Type	Maximum Drainage Area (m²)	Minimum Drainage Area (m²)	NbS Available Area (m²)	NbS Minimum Area (m²)
S59	19417,64	Rain Garden	20234,3	-	1015,1	970,88
S60	12948,60	Rain Garden	20234,3	-	3351,46	647,43
S61	2309,53	Rain Garden	20234,3	-	801,215	115,48
S62	15711,39	Rain Garden	20234,3	-	806,841	785,57
S63	10146,36	Rain Garden	20234,3	-	785,102	507,32
S64	5195,42	Rain Garden	20234,3	-	1020,19	259,77
S65	13773,83	Rain Garden	20234,3	-	2820,54	688,69
S66	9988,43	Rain Garden	20234,3	-	1460,45	499,42
S67	10768,99	Rain Garden	20234,3	-	975,336	538,45
S68	13617,28	Rain Garden	20234,3	-	2325,37	680,86
S69	3903,34	Rain Garden	20234,3	-	989,913	195,17
S70	19448,38	Rain Garden	20234,3	-	1354,47	972,42
S71	7722,86	Rain Garden	20234,3	-	447,873	386,14
S72	16221,87	Rain Garden	20234,3	-	3657,75	811,09
S73	7805,85	Rain Garden	20234,3	-	507,992	390,29
S74	14929,68	Rain Garden	20234,3	-	1313,22	746,48
S75	19588,85	Rain Garden	20234,3	-	2855,38	979,44
S76	14104,07	Rain Garden	20234,3	-	826,427	705,20
S77	11919,13	-	-	-	-	-
S78	125393,07	-	-	-	-	-
S79	13252,43	Rain Garden	20234,3	-	737,825	662,62
S80	16132,66	Bioswales	40468,6	-	1052,93	161,33
S81	2616,21	Rain Garden	20234,3	-	316,752	130,81
S82	11775,65	Rain Garden	20234,3	-	686,527	588,78
S83	20214,91	Rain Garden	20234,3	-	1557,82	1010,75
S84	20227,09	Rain Garden	20234,3	-	3897,07	1011,35

Drainage Area ID	Drainage Area (m²)	NbS Type	Maximum Drainage Area (m²)	Minimum Drainage Area (m²)	NbS Available Area (m²)	NbS Minimum Area (m²)
S85	20151,25	Rain Garden	20234,3	-	2470,57	1007,56
S86	19991,90	Rain Garden	20234,3	-	1608,94	999,59
S87	102017,27	Wetlands	-	20234,3	2579,07	1020,17
S88	15060,86	Rain Garden	20234,3	-	755,228	753,04
S89	12333,31	Rain Garden	20234,3	-	1556,78	616,67
S90	2536,36	Rain Garden	20234,3	-	191,631	126,82
S91	17920,25	Rain Garden	20234,3	-	1604	896,01
S92	10998,55	Rain Garden	20234,3	-	590,556	549,93
S93	120603,48	-	-	-	-	-
S94	20036,12	Rain Garden	20234,3	-	2277,88	1001,81
S95	1128,63	Bioswales	40468,6	-	695,111	11,29
S96	548,51	Rain Garden	20234,3	-	278,816	27,43
S97	6734,60	Rain Garden	20234,3	-	307,309	336,73
S98	56284,50	-	-	-	-	-
S99	18929,56	Rain Garden	20234,3	-	2913,72	946,48
S100	7301,29	Rain Garden	20234,3	-	4238,95	365,06
S101	24530,84	Bioswales	40468,6	-	3456,26	245,31
S102	91016,98	-	-	-	-	-
S103	14683,58	Bioswales	40468,6	-	789,434	146,84
S104	10617,03	Rain Garden	20234,3	-	1210,45	530,85
S105	4303,15	Rain Garden	20234,3	-	422,847	215,16
S106	7828,20	Rain Garden	20234,3	-	524,58	391,41
S107	155031,18	-	-	-	-	-
S108	3073,64	Rain Garden	20234,3	-	187,166	153,68
S109	14538,61	Rain Garden	20234,3	-	761,863	726,93
S110	18385,76	Rain Garden	20234,3	-	1328,35	919,29

Drainage Area ID	Drainage Area (m²)	NbS Type	Maximum Drainage Area (m²)	Minimum Drainage Area (m²)	NbS Available Area (m²)	NbS Minimum Area (m²)
S111	15707,58	Rain Garden	20234,3	-	3647,38	785,38
S112	70835,01	-	-	-	-	-
S113	66341,64	Wetlands	-	20234,3	6331,05	663,42
S114	8300,08	Rain Garden	20234,3	-	4076,36	415,00
S115	8153,11	Rain Garden	20234,3	-	1914,2	407,66
S116	441214,62	-	-	-	-	-
S117	142481,72	-	-	-	-	-
S118	20083,81	Rain Garden	20234,3	-	4117,59	1004,19
S119	1925,61	Rain Garden	20234,3	-	121,557	96,28
S120	3675,18	Rain Garden	20234,3	-	1920,89	183,76
S121	19762,49	Rain Garden	20234,3	-	1166,43	988,12
S122	18968,02	Rain Garden	20234,3	-	1219,44	948,40
S123	2842,17	Rain Garden	20234,3	-	254,568	142,11
S124	17948,41	Rain Garden	20234,3	-	1684,33	897,42
S125	26425,03	-	-	-	-	-
S126	14049,21	Rain Garden	20234,3	-	4582,79	702,46
S127	10278,62	Rain Garden	20234,3	-	5716,66	513,93
S128	6844,09	Rain Garden	20234,3	-	555,415	342,20
S129	9488,20	Rain Garden	20234,3	-	1056,79	474,41
S130	19016,60	Rain Garden	20234,3	-	1036,6	950,83
S131	8198,96	Rain Garden	20234,3	-	628,042	409,95
S132	5809,56	Rain Garden	20234,3	-	1145,83	290,48
S133	12895,15	Rain Garden	20234,3	-	1263,85	644,76
S134	156130,94	-	-	-	-	-
S135	5916,34	Rain Garden	20234,3	-	1496,75	295,82
S136	17934,84	Rain Garden	20234,3	-	5412,67	896,74

Drainage Area ID	Drainage Area (m²)	NbS Type	Maximum Drainage Area (m²)	Minimum Drainage Area (m²)	NbS Available Area (m²)	NbS Minimum Area (m²)
S137	7764,05	Rain Garden	20234,3	-	684,169	388,20
S138	39417,12	Bioswales	40468,6	-	1454,65	394,17
S139	6266,96	Rain Garden	20234,3	-	632,52	313,35
S140	14052,07	-	-	-	-	-
S141	16184,62	-	-	-	-	-
S142	17497,67	Rain Garden	20234,3	-	1145,12	874,88
S143	2061,44	Rain Garden	20234,3	-	342,251	103,07
S144	3784,39	Rain Garden	20234,3	-	1697,4	189,22
S145	12562,82	Rain Garden	20234,3	-	1562,57	628,14
S146	6905,42	Rain Garden	20234,3	-	4222,17	345,27
S147	20094,49	Rain Garden	20234,3	-	1899,64	1004,72
S148	4041,81	Rain Garden	20234,3	-	495,555	202,09
S149	11264,95	Rain Garden	20234,3	-	867,221	563,25
S150	14051,37	Rain Garden	20234,3	-	1502,08	702,57
S151	15105,07	Bioswales	40468,6	-	1026,85	151,05
S152	31482,49	-	-	-	-	-
S153	7757,77	-	-	-	-	-
S154	5018,30	Rain Garden	20234,3	-	456,888	250,92
S155	20112,05	Rain Garden	20234,3	-	2211,43	1005,60
S156	4984,38	Rain Garden	20234,3	-	476,881	249,22
S157	18006,75	Rain Garden	20234,3	-	1020,88	900,34
S158	20192,68	Rain Garden	20234,3	-	3778,14	1009,63
S159	13208,30	Rain Garden	20234,3	-	744,826	660,41
S160	10561,22	Rain Garden	20234,3	-	3303,1	528,06
S161	14046,70	Rain Garden	20234,3	-	592,592	702,34
S162	20221,25	Rain Garden	20234,3	-	4643,82	1011,06

Drainage Area ID	Drainage Area (m²)	NbS Type	Maximum Drainage Area (m²)	Minimum Drainage Area (m²)	NbS Available Area (m²)	NbS Minimum Area (m²)
S163	2313,40	Rain Garden	20234,3	-	151,423	115,67
S164	23611,08	-	-	-	-	-
S165	8426,97	Bioswales	40468,6	-	1224,02	84,27
S166	19165,45	Rain Garden	20234,3	-	1487,38	958,27
S167	5020,35	Rain Garden	20234,3	-	597,384	251,02
S168	12837,67	Rain Garden	20234,3	-	668,317	641,88
S169	3657,61	Rain Garden	20234,3	-	577,254	182,88
S170	4447,74	Rain Garden	20234,3	-	478,949	222,39
S171	6415,61	Rain Garden	20234,3	-	1076,6	320,78
S172	16474,76	Rain Garden	20234,3	-	859,76	823,74
S173	12506,43	Rain Garden	20234,3	-	631,028	625,32
S174	6668,30	Rain Garden	20234,3	-	765,316	333,41
S175	3391,59	Rain Garden	20234,3	-	100,15	169,58
S176	8887,96	Rain Garden	20234,3	-	879,937	444,40
S177	7334,75	Rain Garden	20234,3	-	563,109	366,74
S178	15640,45	Rain Garden	20234,3	-	1012,91	782,02
S179	17804,01	Rain Garden	20234,3	-	2291,94	890,20
S180	23864,71	Bioswales	40468,6	-	1165,7	238,65
S181	7755,33	Rain Garden	20234,3	-	901,98	387,77
S182	31849,26	Wetlands	-	20234,3	10705	318,49
S183	3587,59	Rain Garden	20234,3	-	249,506	179,38
S184	11439,45	Rain Garden	20234,3	-	4004,61	571,97
S185	19504,08	Rain Garden	20234,3	-	1002,46	975,20
S186	66336,72	-	-	-	-	-
S187	20116,31	Rain Garden	20234,3	-	1055,08	1005,82

Drainage Area ID	Drainage Area (m²)	NbS Type	Maximum Drainage Area (m²)	Minimum Drainage Area (m²)	NbS Available Area (m²)	NbS Minimum Area (m²)
S188	19553,98	Rain Garden	20234,3	-	1575,95	977,70
S189	295883,33	-	-	-	-	-
S190	19819,27	Rain Garden	20234,3	-	2965,99	990,96
S191	20078,29	Rain Garden	20234,3	-	212,417	1003,91
S192	12087,17	Rain Garden	20234,3	-	310,39	604,36
S193	3683,20	Rain Garden	20234,3	-	1133,65	184,16
S194	11887,49	Rain Garden	20234,3	-	624,403	594,37
S195	4933,37	Rain Garden	20234,3	-	248,109	246,67
S196	18822,39	Rain Garden	20234,3	-	4715,02	941,12
S197	10229,65	Rain Garden	20234,3	-	113,864	511,48
S198	70431,81	Wetlands	-	20234,3	8214,78	704,32
S199	18071,30	Rain Garden	20234,3	-	537,676	903,57
S200	11119,38	Rain Garden	20234,3	-	907,696	555,97
S201	7421,54	Rain Garden	20234,3	-	190,962	371,08
S202	18486,90	Rain Garden	20234,3	-	448,872	924,34
S203	24020,94	-	-	-	-	-
S204	30832,30	Wetlands	-	20234,3	11433,2	308,32
S205	11369,30	Rain Garden	20234,3	-	616,386	568,47
S206	3470,98	Rain Garden	20234,3	-	260,815	173,55
S207	7358,18	Rain Garden	20234,3	-	372,808	367,91
S208	6279,17	Rain Garden	20234,3	-	267,278	313,96
S209	20186,84	Rain Garden	20234,3	-	2338,51	1009,34
S210	14821,23	Rain Garden	20234,3	-	858,168	741,06
S211	20197,83	Rain Garden	20234,3	-	1325,97	1009,89
S212	11569,97	Rain Garden	20234,3	-	5281,62	578,50
S213	120283,73	-	-	-	-	-

Drainage Area ID	Drainage Area (m²)	NbS Type	Maximum Drainage Area (m²)	Minimum Drainage Area (m²)	NbS Available Area (m²)	NbS Minimum Area (m²)
S214	54870,45	-	-	-	-	-
S215	2242,89	Rain Garden	20234,3	-	136,744	112,14
S216	467077,70	Wetlands	-	20234,3	9496,21	4670,78
S217	7808,04	Rain Garden	20234,3	-	452,84	390,40
S218	20130,99	Rain Garden	20234,3	-	2138,54	1006,55
S219	9825,86	Rain Garden	20234,3	-	586,676	491,29
S220	5641,71	Rain Garden	20234,3	-	352,596	282,09
S221	10460,97	Rain Garden	20234,3	-	190,879	523,05
S222	19271,26	Rain Garden	20234,3	-	1148,28	963,56
S223	13432,73	Rain Garden	20234,3	-	3202,05	671,64
S224	5180,79	Rain Garden	20234,3	-	459,873	259,04
S225	24544,72	-	-	-	-	-
S226	24036,19	-	-	-	-	-
S227	8138,16	Rain Garden	20234,3	-	477,327	406,91
S228	13206,23	Rain Garden	20234,3	-	673,155	660,31
S229	19979,95	Rain Garden	20234,3	-	582,242	999,00
S230	13112,42	Rain Garden	20234,3	-	926,264	655,62
S231	13029,05	-	-	-	-	-
S232	19597,16	Rain Garden	20234,3	-	1022,53	979,86
S233	5598,63	Rain Garden	20234,3	-	1344,42	279,93
S234	6643,26	Rain Garden	20234,3	-	1381,39	332,16
S235	7458,60	Rain Garden	20234,3	-	461,722	372,93
S236	20022,97	Rain Garden	20234,3	-	3031,59	1001,15
S237	1873,07	Rain Garden	20234,3	-	125,505	93,65
S238	138830,90	-	-	-	-	-
S239	20091,78	Rain Garden	20234,3	-	13410,5	1004,59

Drainage Area ID	Drainage Area (m²)	NbS Type	Maximum Drainage Area (m²)	Minimum Drainage Area (m²)	NbS Available Area (m²)	NbS Minimum Area (m²)
S240	2391,65	Rain Garden	20234,3	-	352,688	119,58
S241	9305,07	Rain Garden	20234,3	-	2403,6	465,25
S242	27724,47	Wetlands	-	20234,3	5943,96	277,24
S243	46274,85	-	-	-	-	-
S244	246792,28	Wetlands	-	20234,3	7819,08	2467,92
S245	21605,53	-	-	-	-	-
S246	6781,39	Rain Garden	20234,3	-	500,075	339,07
S247	52410,10	Wetlands	-	-	11763,3	524,10
S248	10349,36	Rain Garden	20234,3	-	7761,22	517,47
S249	231498,85	-	-	-	-	-
S250	19154,52	Rain Garden	20234,3		6224,6	957,73
S251	82112,55	-	-	-	-	-
S252	60359,33	Wetlands	-	20234,3	13461,4	603,59
S253	20196,46	-	-	-	-	-
S254	26547,91	-	-	-	-	-
S255	39247,21	Wetlands	-	20234,3	17057	392,47
S256	9271,99	-	-	-	-	-
S257	103990,76	-	-	-	-	-
S258	132289,73	-	-	-	-	-
S259	103919,08	Wetlands	-	20234,3	3340,12	1039,19
S260	15560,31	Rain Garden	20234,3	-	579,416	778,02
S261	18043,93	Rain Garden	20234,3	-	1006,44	902,20
S262	15990,04	Rain Garden	20234,3	-	1175,76	799,50
S263	4950,03	Rain Garden	20234,3	-	468,957	247,50
S264	6226,01	Rain Garden	20234,3	-	142,345	311,30
S265	16320,30	-	-	-	-	-

Drainage Area ID	Drainage Area (m²)	NbS Type	Maximum Drainage Area (m²)	Minimum Drainage Area (m²)	NbS Available Area (m²)	NbS Minimum Area (m²)
S266	6566,19	Rain Garden	20234,3	-	240,18	328,31
S267	136508,27	-	-	-	-	-
S268	20131,85	Rain Garden	20234,3	-	1262,16	1006,59
S269	20181,32	Rain Garden	20234,3	-	7001,41	1009,07
S270	123892,82	-	-	-	-	-
S271	7347,01	Rain Garden	20234,3	-	1914,35	367,35
S272	2054,30	Rain Garden	20234,3	-	521,95	102,72
S273	4201,07	Rain Garden	20234,3	-	234,958	210,05
S274	15202,58	Rain Garden	20234,3	-	787,292	760,13
S275	20215,69	Rain Garden	20234,3	-	1442,13	1010,78
S276	59242,70	-	-	-	-	-
S277	5000,91	Rain Garden	20234,3	-	2791,48	250,05
S278	119227,36	-	-	-	-	-
S279	20179,40	Rain Garden	20234,3	-	5440,38	1008,97
S280	4173,22	Rain Garden	20234,3	-	2538,57	208,66
S281	16193,77	Rain Garden	20234,3	-	2101,04	809,69
S282	62853,57	Wetlands	-	20234,3	952,206	628,54
S283	7410,48	Rain Garden	20234,3	-	2049,94	370,52
S284	14921,33	Rain Garden	20234,3	-	6138,32	746,07
S285	6491,50	-	-	-	-	-
S286	17032,27	Rain Garden	20234,3	-	1044,21	851,61
S287	27908,34	Wetlands	-	20234,3	2905,25	279,08
S288	69429,53	Wetlands	-	20234,3	8274,34	694,30
S289	182439,50	-	-	-	-	-
S290	5263,62	Rain Garden	20234,3	-	1318,37	263,18
S291	4311,67	-	-	-	-	-

Drainage Area ID	Drainage Area (m²)	NbS Type	Maximum Drainage Area (m²)	Minimum Drainage Area (m²)	NbS Available Area (m²)	NbS Minimum Area (m²)
S292	15550,55	Rain Garden	20234,3	-	7001,38	777,53
S293	11118,18	-	-	-	-	-
S294	5764,00	Rain Garden	20234,3	-	2852,57	288,20
S295	17614,39	Rain Garden	20234,3	-	1374,82	880,72
S296	14054,66	Rain Garden	20234,3	-	1042,97	702,73
S297	11123,31	Rain Garden	20234,3	-	578,387	556,17
S298	21543,25	Bioswales	40468,6	-	497,199	215,43
S299	39352,39	-	-	-	-	-
S300	45908,71	-	-	-	-	-
S301	17869,29	Bioswales	40468,6	-	559,101	178,69
S302	2959,85	Rain Garden	20234,3	-	300,151	147,99
S303	31929,31	-	-	-	-	-
S304	20212,71	Rain Garden	20234,3	-	1134,75	1010,64
S305	9999,85	Rain Garden	20234,3	-	1548,67	499,99
S306	43252,17	-	-	-	-	-
S307	15267,54	Rain Garden	20234,3	-	1460,04	763,38
S308	4802,13	Rain Garden	20234,3	-	749,766	240,11
S309	15655,01	Rain Garden	20234,3	-	2010,76	782,75
S310	12508,20	Rain Garden	20234,3	-	110,809	625,41
S311	18918,50	Rain Garden	20234,3	-	1002,2	945,93
S312	16099,13	-	-	-	-	-
S313	16287,59	Rain Garden	20234,3	-	1558,4	814,38
S314	3023,72	Rain Garden	20234,3	-	1689,26	151,19
S315	22180,39	-	-	-	-	-
S316	94139,09	-	-	-	-	-
S317	6594,15	Rain Garden	20234,3	-	1906,59	329,71

Drainage Area ID	Drainage Area (m ²)	NbS Type	Maximum Drainage Area (m ²)	Minimum Drainage Area (m ²)	NbS Available Area (m ²)	NbS Minimum Area (m ²)
S318	13534,20	Rain Garden	20234,3	-	1264,44	676,71
S319	3501,24	Rain Garden	20234,3	-	1793,61	175,06
S320	4603,00	Rain Garden	20234,3	-	977,437	230,15
S321	24387,55	Wetlands	-	20234,3	1291,29	243,88
S322	7891,42	Rain Garden	20234,3	-	4625,26	394,57
S323	96711,16	-	-	-	-	-
S324	78050,89	-	-	-	-	-
S325	222405,07	Wetlands	-	20234,3	2374,1	2224,05
S326	24276,74	-	-	-	-	-
S327	72482,49	-	-	-	-	-
S328	46421,77	-	-	-	-	-
S329	29698,93	-	-	-	-	-
S330	8801,26	-	-	-	-	-

Source: Authors (2025).

Table 8-S2. Distribution of drainage areas by type of land use. For each drainage area, the total values in square meters (m²) and the respective percentage in relation to the total area of the drainage area are shown.

Drainage Area ID	Type of Use	Total Area (m ²)	Percentage
S1	Urban Voids	4183,787	41,11
S1	Green Areas	2566,569	25,22
S1	Urban Areas	3427,644	33,68
S2	Urban Voids	2625,104	50,32
S2	Green Areas	200,2647	3,84
S2	Urban Areas	2391,506	45,84
S3	Urban Voids	3764,085	18,67
S3	Green Areas	8237,988	40,85
S3	Urban Areas	8164,25	40,48

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S4	Urban Voids	702,8024	33,31
S4	Green Areas	156,9806	7,44
S4	Urban Areas	1249,967	59,25
S5	Urban Voids	3418,893	27,64
S5	Green Areas	2480,725	20,05
S5	Urban Areas	6471,257	52,31
S6	Urban Voids	1079,862	13,01
S6	Green Areas	2465,165	29,71
S6	Urban Areas	4753,598	57,28
S7	Urban Voids	1548,299	7,95
S7	Green Areas	7989,111	41,03
S7	Urban Areas	9931,788	51,01
S8	Urban Voids	2851,704	15,22
S8	Green Areas	8938,551	47,72
S8	Urban Areas	6942,532	37,06
S9	Urban Voids	4606,322	52,58
S9	Urban Areas	4154,553	47,42
S10	Urban Voids	7357,143	74,59
S10	Green Areas	354,9403	3,6
S10	Urban Areas	2150,937	21,81
S11	Urban Voids	1358,849	12,96
S11	Green Areas	4801,831	45,81
S11	Urban Areas	4320,32	41,22
S12	Urban Voids	6818,706	51,76
S12	Green Areas	3259,041	24,74
S12	Urban Areas	3095,84	23,5
S13	Urban Voids	2375,032	13,82

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S13	Green Areas	4838,346	28,16
S13	Urban Areas	9969,497	58,02
S14	Urban Voids	13639,44	68,05
S14	Green Areas	2107,563	10,52
S14	Urban Areas	4296,344	21,44
S15	Urban Voids	11365,63	67,76
S15	Green Areas	382,7925	2,28
S15	Urban Areas	5025,58	29,96
S16	Urban Voids	2640,04	22,53
S16	Green Areas	2273,929	19,41
S16	Urban Areas	6801,578	58,06
S17	Urban Voids	4035,466	24,68
S17	Green Areas	1376,333	8,42
S17	Urban Areas	10939,2	66,9
S18	Urban Voids	166,4007	2,98
S18	Green Areas	284,1837	5,09
S18	Urban Areas	5137,861	91,94
S19	Urban Voids	10375,87	6,17
S19	Green Areas	142761,9	84,89
S19	Urban Areas	15031,17	8,94
S20	Urban Voids	1923,677	9,53
S20	Green Areas	274,0002	1,36
S20	Urban Areas	17990,45	89,11
S21	Urban Voids	55,13933	0,66
S21	Green Areas	180	2,14
S21	Urban Areas	8169,611	97,2
S22	Urban Voids	479,263	5,47

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S22	Green Areas	1663,318	18,99
S22	Urban Areas	6615,769	75,54
S23	Urban Voids	1465,994	13,7
S23	Green Areas	1053,311	9,84
S23	Urban Areas	8181,502	76,46
S24	Urban Voids	594,2559	7,59
S24	Green Areas	1481,563	18,93
S24	Urban Areas	5749,613	73,47
S25	Urban Voids	669,5389	9,59
S25	Green Areas	1722,033	24,67
S25	Urban Areas	4587,879	65,73
S26	Urban Voids	551,2197	4,96
S26	Green Areas	1169,898	10,52
S26	Urban Areas	9401,142	84,53
S27	Urban Voids	263,959	2,11
S27	Green Areas	123,4395	0,99
S27	Urban Areas	12106,4	96,9
S28	Urban Voids	884,1092	14,77
S28	Green Areas	697,4024	11,65
S28	Urban Areas	4405,328	73,58
S29	Urban Voids	2060,106	31,88
S29	Green Areas	1442,421	22,32
S29	Urban Areas	2959,597	45,8
S30	Urban Voids	3143,648	26,37
S30	Green Areas	1366,621	11,46
S30	Urban Areas	7410,119	62,16
S31	Urban Voids	4547,593	22,6

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S31	Green Areas	3936,06	19,56
S31	Urban Areas	11642,83	57,85
S32	Urban Voids	1436,486	31,71
S32	Green Areas	220,4727	4,87
S32	Urban Areas	2872,419	63,42
S33	Urban Voids	1282,109	12,25
S33	Green Areas	1186,08	11,33
S33	Urban Areas	8001,122	76,42
S34	Urban Voids	2307,023	13,1
S34	Green Areas	1055,094	5,99
S34	Urban Areas	14254,01	80,91
S35	Urban Voids	78,12242	1,27
S35	Green Areas	427,3433	6,93
S35	Urban Areas	5663,534	91,81
S36	Urban Voids	171,421	5
S36	Green Areas	600,2479	17,51
S36	Urban Areas	2655,383	77,48
S37	Urban Voids	5504,887	24,38
S37	Urban Areas	17076,61	75,62
S38	Urban Voids	342,9437	1,79
S38	Green Areas	922,95	4,82
S38	Urban Areas	17900,13	93,4
S39	Urban Voids	1485,284	18,08
S39	Green Areas	1236,924	15,05
S39	Urban Areas	5494,829	66,87
S40	Urban Voids	371,9381	6,57
S40	Green Areas	251,2342	4,44

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S40	Urban Areas	5036,825	88,99
S41	Urban Voids	5665,864	5,83
S41	Green Areas	78423,82	80,67
S41	Urban Areas	13126,04	13,5
S42	Urban Voids	454,0188	6,72
S42	Green Areas	10,57187	0,16
S42	Urban Areas	6295,056	93,13
S43	Urban Voids	1929,679	17,95
S43	Green Areas	319,5075	2,97
S43	Urban Areas	8499,67	79,08
S44	Urban Voids	288,6344	13,2
S44	Urban Areas	1898,241	86,8
S45	Urban Voids	141,6921	1,67
S45	Green Areas	2387,938	28,1
S45	Urban Areas	5967,766	70,23
S46	Urban Voids	284,3375	3,09
S46	Green Areas	289,9718	3,15
S46	Urban Areas	8629,655	93,76
S47	Urban Voids	390,6247	6,45
S47	Green Areas	49,6598	0,82
S47	Urban Areas	5618,465	92,73
S48	Urban Voids	548,9274	2,72
S48	Green Areas	248,3738	1,23
S48	Urban Areas	19408,58	96,05
S49	Urban Voids	189,8651	4,22
S49	Green Areas	107,9813	2,4
S49	Urban Areas	4197,279	93,37

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S50	Urban Voids	105,8288	5,64
S50	Green Areas	54,3402	2,9
S50	Urban Areas	1715,714	91,46
S51	Urban Voids	1550,928	19,49
S51	Green Areas	34,10078	0,43
S51	Urban Areas	6373,846	80,08
S52	Urban Voids	484,9818	23,46
S52	Green Areas	38,83413	1,88
S52	Urban Areas	1543,434	74,66
S53	Urban Voids	11557,64	76,29
S53	Green Areas	920,1246	6,07
S53	Urban Areas	2671,457	17,63
S54	Urban Voids	654,9921	3,78
S54	Green Areas	49,65444	0,29
S54	Urban Areas	16636,78	95,94
S55	Urban Voids	539,1964	3,95
S55	Green Areas	8354,922	61,24
S55	Urban Areas	4748,132	34,8
S56	Urban Voids	2692,964	16,11
S56	Green Areas	1547,645	9,26
S56	Urban Areas	12473,89	74,63
S57	Urban Voids	8398,88	50,63
S57	Green Areas	1440,886	8,69
S57	Urban Areas	6749,859	40,69
S58	Urban Voids	1161,177	16,32
S58	Green Areas	777,5143	10,93
S58	Urban Areas	5178,058	72,76

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S59	Urban Voids	1253,744	6,46
S59	Green Areas	855,0083	4,4
S59	Urban Areas	17313	89,14
S60	Urban Voids	2908,784	22,46
S60	Green Areas	991,8325	7,66
S60	Urban Areas	9050,633	69,88
S61	Urban Voids	30,68281	1,33
S61	Green Areas	251,3379	10,88
S61	Urban Areas	2027,979	87,79
S62	Urban Voids	863,5457	5,5
S62	Green Areas	47,7488	0,3
S62	Urban Areas	14803,33	94,2
S63	Urban Voids	6111,753	60,22
S63	Green Areas	145,8497	1,44
S63	Urban Areas	3890,797	38,34
S64	Urban Voids	911,4136	17,54
S64	Green Areas	2718,324	52,31
S64	Urban Areas	1566,763	30,15
S65	Urban Voids	11757,32	85,34
S65	Urban Areas	2019,265	14,66
S66	Urban Voids	434,3247	4,35
S66	Green Areas	251,2813	2,52
S66	Urban Areas	9304,894	93,14
S67	Urban Voids	941,412	8,74
S67	Green Areas	3889,057	36,11
S67	Urban Areas	5940,78	55,15
S68	Urban Voids	1936,626	14,22

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S68	Green Areas	944,6469	6,94
S68	Urban Areas	10738,85	78,85
S69	Urban Voids	772,2566	19,78
S69	Green Areas	159,5485	4,09
S69	Urban Areas	2972,32	76,13
S70	Urban Voids	686,9069	3,53
S70	Green Areas	935,6576	4,81
S70	Urban Areas	17829,94	91,66
S71	Urban Voids	224,9992	2,91
S71	Green Areas	1791,11	23,19
S71	Urban Areas	5708,39	73,9
S72	Urban Voids	1159,201	7,14
S72	Green Areas	3186,895	19,64
S72	Urban Areas	11879,15	73,21
S73	Urban Voids	126,4869	1,62
S73	Green Areas	1282,036	16,42
S73	Urban Areas	6398,988	81,96
S74	Urban Voids	853,1186	5,71
S74	Green Areas	2388,907	16
S74	Urban Areas	11690,83	78,29
S75	Urban Voids	18490,71	94,38
S75	Urban Areas	1102,028	5,62
S76	Urban Voids	13889,11	98,46
S76	Urban Areas	217,7752	1,54
S77	Urban Voids	928,5441	7,79
S77	Green Areas	6356,5	53,32
S77	Urban Areas	4636,552	38,89

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S78	Urban Voids	23495,22	18,73
S78	Green Areas	35834,58	28,57
S78	Urban Areas	66088,88	52,69
S79	Urban Voids	560,5139	4,23
S79	Green Areas	637,6204	4,81
S79	Urban Areas	12057,1	90,96
S80	Urban Voids	1179,062	7,31
S80	Green Areas	5190,56	32,17
S80	Urban Areas	9766,503	60,53
S81	Green Areas	381,7327	14,59
S81	Urban Areas	2235,03	85,41
S82	Urban Voids	332,2501	2,82
S82	Green Areas	7661,02	65,04
S82	Urban Areas	3784,855	32,13
S83	Urban Voids	6976,659	34,51
S83	Green Areas	3657,522	18,09
S83	Urban Areas	9584,812	47,4
S84	Urban Voids	16774,16	82,91
S84	Green Areas	2377,171	11,75
S84	Urban Areas	1079,781	5,34
S85	Urban Voids	14588,58	72,38
S85	Green Areas	156,1532	0,77
S85	Urban Areas	5410,493	26,84
S86	Urban Voids	10712,8	53,57
S86	Green Areas	288	1,44
S86	Urban Areas	8995,106	44,98
S87	Urban Voids	4415,337	4,33

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S87	Green Areas	20435,96	20,03
S87	Urban Areas	77187,25	75,65
S88	Urban Voids	354,7319	2,35
S88	Green Areas	2325,859	15,44
S88	Urban Areas	12383,44	82,21
S89	Urban Voids	5472,946	44,37
S89	Green Areas	1746,375	14,16
S89	Urban Areas	5116,429	41,48
S90	Urban Voids	155,3215	6,12
S90	Urban Areas	2381,553	93,88
S91	Urban Voids	6934,167	38,69
S91	Green Areas	2897,481	16,17
S91	Urban Areas	8092,25	45,15
S92	Urban Voids	3150,016	28,63
S92	Green Areas	1004,78	9,13
S92	Urban Areas	6845,953	62,23
S93	Urban Voids	4173,82	3,46
S93	Green Areas	49786,99	41,27
S93	Urban Areas	66668,01	55,27
S94	Urban Voids	4695,199	23,43
S94	Green Areas	377,2891	1,88
S94	Urban Areas	14967,6	74,69
S95	Urban Voids	204,8556	18,15
S95	Green Areas	28,47926	2,52
S95	Urban Areas	895,5401	79,33
S96	Urban Voids	803,3971	9,66
S96	Urban Areas	7509,25	90,34

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S97	Urban Voids	306,5672	4,55
S97	Green Areas	18,23144	0,27
S97	Urban Areas	6411,146	95,18
S98	Urban Voids	2390,672	4,25
S98	Green Areas	26497,17	47,07
S98	Urban Areas	27408,23	48,69
S99	Urban Voids	2751,289	24,63
S99	Urban Areas	8418,046	75,37
S100	Urban Voids	1658,287	22,71
S100	Urban Areas	5644,458	77,29
S101	Urban Voids	3386,087	13,8
S101	Urban Areas	21149,66	86,2
S102	Urban Voids	7745,621	8,51
S102	Green Areas	26834,21	29,48
S102	Urban Areas	56455,59	62,01
S103	Urban Voids	1308,51	8,91
S103	Green Areas	869,2554	5,92
S103	Urban Areas	12508,98	85,17
S104	Urban Voids	548,0884	5,16
S104	Urban Areas	10071,04	94,84
S105	Urban Voids	431,7335	10,03
S105	Green Areas	63,66062	1,48
S105	Urban Areas	3808,606	88,49
S106	Urban Voids	107,9116	1,38
S106	Green Areas	60	0,77
S106	Urban Areas	7661,838	97,86
S107	Urban Voids	3683,398	2,38

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S107	Green Areas	135617,7	87,46
S107	Urban Areas	15762,74	10,17
S108	Urban Voids	47,62902	1,55
S108	Green Areas	72	2,34
S108	Urban Areas	2954,621	96,11
S109	Urban Voids	80,05099	0,55
S109	Green Areas	76	0,52
S109	Urban Areas	14385,45	98,93
S110	Urban Voids	3000,212	16,31
S110	Green Areas	283,6387	1,54
S110	Urban Areas	15105,62	82,14
S111	Urban Voids	3544,67	22,56
S111	Green Areas	996,61	6,34
S111	Urban Areas	11169,47	71,09
S112	Urban Voids	2269,995	3,2
S112	Green Areas	60586,55	85,51
S112	Urban Areas	7993,205	11,28
S113	Urban Voids	27440,01	41,35
S113	Green Areas	10285,64	15,5
S113	Urban Areas	28629,46	43,15
S114	Urban Voids	1822,315	21,95
S114	Green Areas	634,106	7,64
S114	Urban Areas	5845,317	70,41
S115	Urban Voids	1125,025	13,8
S115	Green Areas	203,8625	2,5
S115	Urban Areas	6825,862	83,7
S116	Urban Voids	7044,738	1,6

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S116	Green Areas	373423,4	84,62
S116	Urban Areas	60840,49	13,79
S117	Urban Voids	19722,55	13,84
S117	Green Areas	54740,05	38,41
S117	Urban Areas	68048,34	47,75
S118	Urban Voids	5023,866	25,01
S118	Green Areas	1473,342	7,33
S118	Urban Areas	13590,66	67,66
S119	Urban Voids	159,518	8,28
S119	Green Areas	61,89959	3,21
S119	Urban Areas	1704,578	88,5
S120	Urban Voids	1646,374	44,79
S120	Green Areas	868,1247	23,62
S120	Urban Areas	1161,416	31,6
S121	Urban Voids	5327,839	26,95
S121	Green Areas	318,7126	1,61
S121	Urban Areas	14119,92	71,43
S122	Urban Voids	1868,365	9,85
S122	Green Areas	216,6533	1,14
S122	Urban Areas	16886,86	89,01
S123	Urban Voids	249,8743	8,79
S123	Urban Areas	2592,876	91,21
S124	Urban Voids	5615,627	31,28
S124	Green Areas	791,5992	4,41
S124	Urban Areas	11544,78	64,31
S125	Urban Voids	1354,222	5,12
S125	Green Areas	18526,3	70,09

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S125	Urban Areas	6549,958	24,78
S126	Urban Voids	1886,952	13,43
S126	Green Areas	351,5349	2,5
S126	Urban Areas	11813,59	84,07
S127	Urban Voids	4810,199	46,79
S127	Green Areas	984,9981	9,58
S127	Urban Areas	4485,494	43,63
S128	Urban Voids	652,5323	9,53
S128	Green Areas	333,1319	4,87
S128	Urban Areas	5859,811	85,6
S129	Urban Voids	838,6069	8,84
S129	Green Areas	88,36299	0,93
S129	Urban Areas	8563,155	90,23
S130	Urban Voids	1049,268	5,52
S130	Green Areas	325,8358	1,71
S130	Urban Areas	17645,32	92,77
S131	Urban Voids	2251,596	27,46
S131	Urban Areas	5949,029	72,54
S132	Urban Voids	1478,464	25,44
S132	Green Areas	167,1572	2,88
S132	Urban Areas	4165,129	71,68
S133	Urban Voids	3706,962	28,74
S133	Green Areas	201,1002	1,56
S133	Urban Areas	8989,659	69,7
S134	Urban Voids	6440,362	4,12
S134	Green Areas	30240,15	19,36
S134	Urban Areas	119483,7	76,51

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S135	Urban Voids	466,7383	7,89
S135	Green Areas	18,00097	0,3
S135	Urban Areas	5432,782	91,81
S136	Urban Voids	2418,751	13,48
S136	Green Areas	1745,173	9,73
S136	Urban Areas	13774,58	76,79
S137	Urban Voids	120,0602	1,55
S137	Green Areas	62,89695	0,81
S137	Urban Areas	7582,668	97,64
S138	Urban Voids	7612,773	19,31
S138	Green Areas	3075,739	7,8
S138	Urban Areas	28736,99	72,89
S139	Urban Voids	716,1222	11,42
S139	Green Areas	1038,888	16,57
S139	Urban Areas	4513,24	72
S140	Urban Voids	1057,425	7,52
S140	Green Areas	10456,5	74,4
S140	Urban Areas	2541,058	18,08
S141	Urban Voids	4022,683	24,85
S141	Green Areas	7918,72	48,92
S141	Urban Areas	4246,589	26,23
S142	Urban Voids	1361,65	7,78
S142	Green Areas	337,103	1,93
S142	Urban Areas	15802,43	90,29
S143	Urban Voids	1277,168	61,94
S143	Green Areas	188,0951	9,12
S143	Urban Areas	596,6119	28,94

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S144	Urban Voids	806,6088	21,31
S144	Green Areas	92,32653	2,44
S144	Urban Areas	2886,252	76,25
S145	Urban Voids	1203,553	9,58
S145	Urban Areas	11361,82	90,42
S146	Urban Voids	1269,297	18,38
S146	Green Areas	130,7488	1,89
S146	Urban Areas	5506,829	79,73
S147	Urban Voids	632,5722	3,15
S147	Green Areas	275,8714	1,37
S147	Urban Areas	19190,06	95,48
S148	Urban Voids	246,3885	6,09
S148	Green Areas	371,4472	9,19
S148	Urban Areas	3424,789	84,72
S149	Urban Voids	3931,079	34,89
S149	Green Areas	308	2,73
S149	Urban Areas	7028,171	62,38
S150	Urban Voids	3668,962	26,11
S150	Green Areas	765,7237	5,45
S150	Urban Areas	9619,689	68,45
S151	Urban Voids	725,2058	4,8
S151	Green Areas	657,2257	4,35
S151	Urban Areas	13725,92	90,85
S152	Urban Voids	1092,255	3,47
S152	Green Areas	2131,981	6,77
S152	Urban Areas	28264,86	89,76
S153	Urban Voids	263,231	3,39

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S153	Green Areas	5263,431	67,83
S153	Urban Areas	2232,708	28,77
S154	Urban Voids	193,1027	3,85
S154	Green Areas	0,839262	0,02
S154	Urban Areas	4825,433	96,14
S155	Urban Voids	12899,03	64,12
S155	Green Areas	236,1509	1,17
S155	Urban Areas	6980,996	34,7
S156	Urban Voids	197,2442	3,96
S156	Green Areas	97,28511	1,95
S156	Urban Areas	4690,846	94,09
S157	Urban Voids	1296,385	7,2
S157	Green Areas	564,4998	3,13
S157	Urban Areas	16149,49	89,67
S158	Urban Voids	391,4365	1,94
S158	Green Areas	788,6384	3,9
S158	Urban Areas	19016,9	94,16
S159	Urban Voids	511,509	3,87
S159	Green Areas	299,6385	2,27
S159	Urban Areas	12399,83	93,86
S160	Urban Voids	1533,362	14,52
S160	Green Areas	999,5647	9,46
S160	Urban Areas	8030,573	76,02
S161	Urban Voids	348,7142	2,48
S161	Green Areas	11,74229	0,08
S161	Urban Areas	13689,04	97,43
S162	Urban Voids	5395,312	26,68

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S162	Green Areas	543,7854	2,69
S162	Urban Areas	14286,28	70,64
S163	Urban Voids	12,48394	0,54
S163	Urban Areas	2301,38	99,46
S164	Urban Voids	9662,118	40,91
S164	Green Areas	10780,28	45,65
S164	Urban Areas	3173,564	13,44
S165	Urban Voids	2594,971	30,79
S165	Green Areas	108,5073	1,29
S165	Urban Areas	5725,216	67,93
S166	Urban Voids	1337,15	6,98
S166	Green Areas	997,3395	5,2
S166	Urban Areas	16835,14	87,82
S167	Green Areas	521,9648	10,39
S167	Urban Areas	4499,466	89,61
S168	Urban Voids	824,6422	6,42
S168	Green Areas	280,6485	2,19
S168	Urban Areas	11735,17	91,39
S169	Urban Voids	183,695	5,02
S169	Green Areas	482,0965	13,18
S169	Urban Areas	2992,562	81,8
S170	Urban Voids	59,10673	1,33
S170	Green Areas	62,0003	1,39
S170	Urban Areas	4327,518	97,28
S171	Urban Voids	659,3475	10,28
S171	Green Areas	941,106	14,67
S171	Urban Areas	4816,546	75,06

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S172	Urban Voids	530,0573	3,22
S172	Green Areas	129,9997	0,79
S172	Urban Areas	15818,01	95,99
S173	Urban Voids	423,598	3,39
S173	Urban Areas	12085,57	96,61
S174	Urban Voids	238,2254	3,57
S174	Urban Areas	6431,4	96,43
S175	Urban Voids	297,7822	8,78
S175	Green Areas	185,5678	5,47
S175	Urban Areas	2908,979	85,75
S176	Urban Voids	768,133	8,64
S176	Green Areas	346,0519	3,89
S176	Urban Areas	7775,565	87,47
S177	Urban Voids	945,4255	12,89
S177	Green Areas	239,7189	3,27
S177	Urban Areas	6151,083	83,85
S178	Urban Voids	2026,362	12,95
S178	Green Areas	930,4128	5,95
S178	Urban Areas	12687,06	81,1
S179	Urban Voids	1244,338	6,99
S179	Green Areas	80	0,45
S179	Urban Areas	16483,29	92,56
S180	Urban Voids	1825,387	7,65
S180	Green Areas	100	0,42
S180	Urban Areas	21944,21	91,93
S181	Urban Voids	213,2789	2,75

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S181	Urban Areas	7543,596	97,25
S182	Urban Voids	14090,04	44,23
S182	Green Areas	11131,4	34,94
S182	Urban Areas	6634,41	20,83
S183	Urban Voids	121,1938	3,38
S183	Green Areas	503,7233	14,04
S183	Urban Areas	2963,458	82,58
S184	Urban Voids	718,561	6,28
S184	Green Areas	1081,529	9,45
S184	Urban Areas	9641,66	84,27
S185	Urban Voids	1652,749	8,47
S185	Green Areas	378,9371	1,94
S185	Urban Areas	17476,63	89,59
S186	Urban Voids	15947,23	24,03
S186	Green Areas	42692,54	64,34
S186	Urban Areas	7710,765	11,62
S187	Urban Voids	640,6989	3,18
S187	Green Areas	139,4835	0,69
S187	Urban Areas	19340,24	96,12
S188	Urban Voids	2289,324	11,71
S188	Green Areas	106,5005	0,54
S188	Urban Areas	17162,05	87,75
S189	Urban Voids	18222,69	6,16
S189	Green Areas	19095,75	6,45
S189	Urban Areas	258627,5	87,39
S190	Urban Voids	1268,274	6,4

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S190	Green Areas	2183,901	11,02
S190	Urban Areas	16371,07	82,59
S191	Urban Voids	326,3258	1,62
S191	Green Areas	144	0,72
S191	Urban Areas	19612,38	97,66
S192	Urban Voids	2,405041	0,02
S192	Green Areas	72,63599	0,6
S192	Urban Areas	12014,58	99,38
S193	Urban Voids	487,6661	13,24
S193	Green Areas	191,1881	5,19
S193	Urban Areas	3005,146	81,57
S194	Urban Voids	312,5374	2,63
S194	Green Areas	193,5086	1,63
S194	Urban Areas	11384,01	95,74
S195	Urban Voids	31,5011	0,64
S195	Green Areas	195,3628	3,96
S195	Urban Areas	4707,511	95,4
S196	Urban Voids	2523,843	13,41
S196	Green Areas	2366,826	12,57
S196	Urban Areas	13935,83	74,02
S197	Urban Voids	309,3242	3,02
S197	Urban Areas	9922,551	96,98
S198	Urban Voids	21008,24	29,82
S198	Green Areas	15697,97	22,28
S198	Urban Areas	33740,08	47,89
S199	Urban Voids	736,2155	4,07

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S199	Green Areas	168	0,93
S199	Urban Areas	17171,05	95
S200	Urban Voids	329,9475	2,97
S200	Green Areas	962,2221	8,65
S200	Urban Areas	9829,591	88,38
S201	Urban Voids	218,0094	2,94
S201	Green Areas	391,2568	5,27
S201	Urban Areas	6813,869	91,79
S202	Urban Voids	358,224	1,94
S202	Green Areas	124	0,67
S202	Urban Areas	18008,72	97,39
S203	Urban Voids	2831,495	11,79
S203	Green Areas	19764,72	82,26
S203	Urban Areas	1429,711	5,95
S204	Urban Voids	15148,3	49,12
S204	Green Areas	6430,583	20,85
S204	Urban Areas	9259,86	30,03
S205	Urban Voids	430,6129	3,79
S205	Green Areas	488,9468	4,3
S205	Urban Areas	10452,22	91,91
S206	Urban Voids	101,4124	2,92
S206	Green Areas	662,902	19,09
S206	Urban Areas	2707,408	77,98
S207	Urban Voids	137,5886	1,87
S207	Green Areas	52	0,71
S207	Urban Areas	7170,208	97,42

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S208	Urban Voids	108,445	1,73
S208	Green Areas	225,5056	3,59
S208	Urban Areas	5946,573	94,68
S209	Urban Voids	2836,787	14,05
S209	Green Areas	350,9596	1,74
S209	Urban Areas	17003,51	84,21
S210	Urban Voids	637,882	4,3
S210	Green Areas	250,0525	1,69
S210	Urban Areas	13936,52	94,01
S211	Urban Voids	1051,138	5,22
S211	Green Areas	938,5658	4,66
S211	Urban Areas	18166,27	90,13
S212	Urban Voids	458,1774	3,96
S212	Green Areas	1383,934	11,96
S212	Urban Areas	9730,315	84,08
S213	Urban Voids	3843,895	3,2
S213	Green Areas	10451,5	8,69
S213	Urban Areas	106014	88,12
S214	Urban Voids	22304,02	40,64
S214	Green Areas	8785,067	16,01
S214	Urban Areas	23792,71	43,35
S215	Urban Voids	75,7202	3,38
S215	Green Areas	194,6192	8,68
S215	Urban Areas	1973,038	87,95
S216	Urban Voids	47001,49	10,06
S216	Green Areas	53533,24	11,46

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S216	Urban Areas	366637,1	78,48
S217	Urban Voids	474,1706	6,07
S217	Green Areas	339,1604	4,34
S217	Urban Areas	6996,429	89,59
S218	Urban Voids	1422,283	7,06
S218	Green Areas	1073,686	5,33
S218	Urban Areas	17639,41	87,6
S219	Urban Voids	508,3115	5,17
S219	Green Areas	207,1222	2,11
S219	Urban Areas	9112,591	92,72
S220	Urban Voids	65,69557	1,16
S220	Green Areas	121,0771	2,15
S220	Urban Areas	5456,149	96,69
S221	Urban Voids	1135,586	10,85
S221	Green Areas	308,8396	2,95
S221	Urban Areas	9018,837	86,2
S222	Urban Voids	625,3346	3,25
S222	Green Areas	1333,92	6,92
S222	Urban Areas	17310,38	89,83
S223	Urban Voids	1973,026	14,69
S223	Green Areas	800	5,95
S223	Urban Areas	10662,6	79,36
S224	Urban Voids	493,1462	9,48
S224	Urban Areas	4710,021	90,52
S225	Urban Voids	2032,919	8,28
S225	Green Areas	21014,12	85,6

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S225	Urban Areas	1502,761	6,12
S226	Urban Voids	1891,64	7,87
S226	Green Areas	7439,649	30,95
S226	Urban Areas	14709,96	61,19
S227	Urban Voids	541,283	6,65
S227	Green Areas	517,0153	6,35
S227	Urban Areas	7081,604	87
S228	Urban Voids	474,3261	3,59
S228	Green Areas	897,1697	6,79
S228	Urban Areas	11837,61	89,62
S229	Urban Voids	885,8155	4,43
S229	Green Areas	454,807	2,28
S229	Urban Areas	18643,71	93,29
S230	Urban Voids	698,9547	5,33
S230	Green Areas	1078,481	8,22
S230	Urban Areas	11337,79	86,45
S231	Urban Voids	1740,435	13,36
S231	Green Areas	4501,243	34,54
S231	Urban Areas	6790,072	52,1
S232	Urban Voids	1681,033	8,58
S232	Green Areas	416,0196	2,12
S232	Urban Areas	17504,32	89,3
S233	Urban Voids	774,5746	13,83
S233	Green Areas	359,4066	6,42
S233	Urban Areas	4465,88	79,75
S234	Urban Voids	683,8239	10,29

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S234	Green Areas	999,6558	15,04
S234	Urban Areas	4961,239	74,66
S235	Urban Voids	93,97376	1,26
S235	Green Areas	95,763	1,28
S235	Urban Areas	7270,513	97,46
S236	Urban Voids	2325,504	11,61
S236	Green Areas	562,0255	2,81
S236	Urban Areas	17139,69	85,58
S237	Urban Voids	9,281902	0,5
S237	Green Areas	96,7438	5,16
S237	Urban Areas	1767,45	94,34
S238	Urban Voids	12963,45	9,34
S238	Green Areas	50983,32	36,72
S238	Urban Areas	74913,38	53,95
S239	Urban Voids	4684,997	23,38
S239	Green Areas	1494,007	7,45
S239	Urban Areas	13862,01	69,17
S240	Urban Voids	416,0665	17,39
S240	Green Areas	1573,239	65,77
S240	Urban Areas	402,8198	16,84
S241	Urban Voids	5536,94	59,49
S241	Green Areas	132	1,42
S241	Urban Areas	3638,128	39,09
S242	Urban Voids	5079,663	18,32
S242	Green Areas	7723,909	27,86
S242	Urban Areas	14917,47	53,81

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S243	Urban Voids	3807,284	8,23
S243	Green Areas	1830,137	3,95
S243	Urban Areas	40646,81	87,82
S244	Urban Voids	17038,47	6,9
S244	Green Areas	6108,481	2,47
S244	Urban Areas	223699,1	90,62
S245	Urban Voids	1010,656	4,68
S245	Green Areas	623,0495	2,88
S245	Urban Areas	19976,17	92,44
S246	Urban Voids	263,0098	3,88
S246	Green Areas	68	1
S246	Urban Areas	6451,74	95,12
S247	Urban Voids	18350,38	35,01
S247	Green Areas	16628,29	31,72
S247	Urban Areas	17442,54	33,27
S248	Urban Voids	7101,456	68,6
S248	Urban Areas	3250,128	31,4
S249	Urban Voids	15500,95	6,69
S249	Green Areas	67085,26	28,97
S249	Urban Areas	148960,3	64,33
S250	Urban Voids	743,8968	3,88
S250	Green Areas	2980,41	15,56
S250	Urban Areas	15434,07	80,56
S251	Urban Voids	4579,728	5,58
S251	Green Areas	19268,41	23,46
S251	Urban Areas	58281,96	70,96

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S252	Urban Voids	4039,253	6,69
S252	Green Areas	28203,64	46,72
S252	Urban Areas	28129,27	46,59
S253	Urban Voids	1332,889	6,6
S253	Green Areas	3531,507	17,48
S253	Urban Areas	15336,5	75,92
S254	Urban Voids	6306,383	23,75
S254	Green Areas	19052,58	71,75
S254	Urban Areas	1194,54	4,5
S255	Urban Voids	8576,614	21,85
S255	Green Areas	10790,7	27,49
S255	Urban Areas	19888,05	50,66
S256	Urban Voids	87,02363	0,94
S256	Green Areas	8725,836	94,09
S256	Urban Areas	461,0753	4,97
S257	Urban Voids	6236,146	6
S257	Green Areas	16867,79	16,22
S257	Urban Areas	80908,26	77,79
S258	Urban Voids	5673,128	4,29
S258	Green Areas	9216,293	6,97
S258	Urban Areas	117425	88,75
S259	Urban Voids	7550,338	7,26
S259	Green Areas	4940,914	4,75
S259	Urban Areas	91449,12	87,98
S260	Urban Voids	764,2116	4,91
S260	Green Areas	254,6901	1,64

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S260	Urban Areas	14544,85	93,45
S261	Urban Voids	840,3629	4,66
S261	Green Areas	1153,253	6,39
S261	Urban Areas	16054,26	88,95
S262	Urban Voids	709,7665	4,44
S262	Green Areas	1280,828	8,01
S262	Urban Areas	14002,91	87,55
S263	Urban Voids	155,0688	3,13
S263	Green Areas	513,0773	10,36
S263	Urban Areas	4282,979	86,51
S264	Urban Voids	165,5837	2,66
S264	Green Areas	64	1,03
S264	Urban Areas	5997,791	96,31
S265	Urban Voids	556,1874	3,41
S265	Green Areas	7793,732	47,74
S265	Urban Areas	7973,831	48,85
S266	Urban Voids	346,6414	5,28
S266	Green Areas	268,4154	4,09
S266	Urban Areas	5952,568	90,64
S267	Urban Voids	5511,585	4,04
S267	Green Areas	11553,42	8,46
S267	Urban Areas	119472,9	87,5
S268	Urban Voids	876,8627	4,35
S268	Green Areas	76	0,38
S268	Urban Areas	19183,43	95,27
S269	Urban Voids	4157,978	20,6

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S269	Green Areas	2897,244	14,35
S269	Urban Areas	13130,19	65,05
S270	Urban Voids	7074,849	5,71
S270	Green Areas	21702,58	17,51
S270	Urban Areas	95141,12	76,78
S271	Urban Voids	296,4565	4,03
S271	Green Areas	1344,874	18,3
S271	Urban Areas	5707,295	77,66
S272	Urban Voids	152,3933	7,42
S272	Urban Areas	1902,357	92,58
S273	Urban Voids	220,0087	5,12
S273	Green Areas	171,9602	4
S273	Urban Areas	3905,031	90,88
S274	Urban Voids	212,7128	1,4
S274	Green Areas	704,9862	4,64
S274	Urban Areas	14288,18	93,96
S275	Urban Voids	466,4521	2,31
S275	Green Areas	667,3654	3,3
S275	Urban Areas	19086,31	94,39
S276	Urban Voids	1872,132	3,16
S276	Green Areas	20576,54	34,72
S276	Urban Areas	36810,27	62,12
S277	Urban Voids	130,611	2,6
S277	Green Areas	1233,75	24,57
S277	Urban Areas	3656,139	72,82
S278	Urban Voids	4481,081	3,76

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S278	Green Areas	40147,89	33,67
S278	Urban Areas	74622,98	62,58
S279	Urban Voids	5092,608	25,23
S279	Green Areas	1737,458	8,61
S279	Urban Areas	13353,43	66,16
S280	Urban Voids	339,9405	8,18
S280	Green Areas	1460,243	35,14
S280	Urban Areas	2355,442	56,68
S281	Urban Voids	294,8326	1,82
S281	Green Areas	1484,749	9,17
S281	Urban Areas	14417,74	89,01
S282	Urban Voids	2876,253	4,58
S282	Green Areas	9428,62	15
S282	Urban Areas	50562,11	80,43
S283	Urban Voids	428,2022	5,78
S283	Green Areas	1202,813	16,23
S283	Urban Areas	5781,104	78
S284	Urban Voids	10269,22	68,81
S284	Green Areas	374,5423	2,51
S284	Urban Areas	4280,606	28,68
S285	Urban Voids	7,37245	0,11
S285	Green Areas	5886,429	90,66
S285	Urban Areas	599,0671	9,23
S286	Urban Voids	522,5005	3,07
S286	Green Areas	836,6007	4,91
S286	Urban Areas	15676,9	92,02

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S287	Urban Voids	2129,366	7,63
S287	Green Areas	1466,321	5,25
S287	Urban Areas	24318,58	87,12
S288	Urban Voids	11087,2	15,97
S288	Green Areas	20998,67	30,24
S288	Urban Areas	37358	53,8
S289	Urban Voids	13121,06	7,19
S289	Green Areas	5993,382	3,28
S289	Urban Areas	163364,5	89,53
S290	Urban Voids	444,3654	8,44
S290	Green Areas	584,3785	11,1
S290	Urban Areas	4236,027	80,46
S291	Urban Voids	60,48984	1,4
S291	Green Areas	3769,926	87,42
S291	Urban Areas	482,1596	11,18
S292	Urban Voids	2861,378	18,4
S292	Green Areas	2143,248	13,78
S292	Urban Areas	10549,25	67,82
S293	Urban Voids	435,6719	3,92
S293	Green Areas	10632,84	95,61
S293	Urban Areas	52	0,47
S294	Urban Voids	773,1794	13,41
S294	Green Areas	2108,094	36,57
S294	Urban Areas	2883,994	50,02
S295	Urban Voids	2261,105	12,83
S295	Green Areas	1581,566	8,98

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S295	Urban Areas	13775,33	78,19
S296	Urban Voids	309,3256	2,2
S296	Green Areas	1023,888	7,28
S296	Urban Areas	12724,53	90,52
S297	Urban Voids	375,4093	3,37
S297	Green Areas	438,5076	3,94
S297	Urban Areas	10311,82	92,68
S298	Urban Voids	1595,799	7,41
S298	Green Areas	4222,228	19,59
S298	Urban Areas	15729,68	73
S299	Urban Voids	3679,101	9,35
S299	Green Areas	30673,93	77,93
S299	Urban Areas	5007,593	12,72
S300	Urban Voids	4011,049	8,74
S300	Green Areas	28471,87	62,01
S300	Urban Areas	13435,52	29,26
S301	Urban Voids	3755,032	21,01
S301	Green Areas	6274,802	35,11
S301	Urban Areas	7843,167	43,88
S302	Urban Voids	83,58071	2,82
S302	Green Areas	220,1828	7,44
S302	Urban Areas	2656,737	89,74
S303	Urban Voids	4829,938	15,12
S303	Green Areas	7268,209	22,76
S303	Urban Areas	19837,97	62,12
S304	Urban Voids	624,3434	3,09

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S304	Green Areas	2062,079	10,2
S304	Urban Areas	17530,67	86,71
S305	Urban Voids	166,0844	1,66
S305	Green Areas	677,2492	6,77
S305	Urban Areas	9158,666	91,57
S306	Urban Voids	2548,952	5,89
S306	Green Areas	21591,46	49,91
S306	Urban Areas	19120,74	44,2
S307	Urban Voids	194,2275	1,27
S307	Green Areas	420,1258	2,75
S307	Urban Areas	14656,52	95,98
S308	Urban Voids	132,8943	2,77
S308	Green Areas	1101,783	22,94
S308	Urban Areas	3568,498	74,29
S309	Urban Voids	862,6687	5,51
S309	Green Areas	946,8582	6,05
S309	Urban Areas	13848,85	88,44
S310	Urban Voids	532,8814	4,26
S310	Green Areas	46,34515	0,37
S310	Urban Areas	11931,72	95,37
S311	Urban Voids	748,2961	3,95
S311	Green Areas	1749,561	9,25
S311	Urban Areas	16424,8	86,8
S312	Urban Voids	272	1,69
S312	Green Areas	14058,26	87,3
S312	Urban Areas	1772,235	11,01

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S313	Urban Voids	1393,018	8,55
S313	Green Areas	454,0731	2,79
S313	Urban Areas	14444,03	88,66
S314	Urban Voids	409,9458	13,55
S314	Green Areas	562,8512	18,61
S314	Urban Areas	2051,578	67,83
S315	Urban Voids	1761,892	7,94
S315	Green Areas	16852,24	75,96
S315	Urban Areas	3570,96	16,1
S316	Urban Voids	2843,821	3,02
S316	Green Areas	77221,15	82,01
S316	Urban Areas	14093,78	14,97
S317	Urban Voids	1487,693	22,56
S317	Green Areas	48,28021	0,73
S317	Urban Areas	5059,594	76,71
S318	Urban Voids	446,3352	3,3
S318	Green Areas	964,9978	7,13
S318	Urban Areas	12125,81	89,57
S319	Urban Voids	404,2018	11,54
S319	Green Areas	13,46872	0,38
S319	Urban Areas	3084,329	88,07
S320	Urban Voids	744,7898	16,18
S320	Green Areas	326,7437	7,1
S320	Urban Areas	3532,466	76,73
S321	Urban Voids	1223,613	5,02
S321	Green Areas	1243,621	5,1

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S321	Urban Areas	21925,53	89,89
S322	Urban Voids	4141,144	52,47
S322	Green Areas	81,0941	1,03
S322	Urban Areas	3670,887	46,51
S323	Urban Voids	1926,499	1,99
S323	Green Areas	86970,9	89,91
S323	Urban Areas	7834,1	8,1
S324	Urban Voids	3499,672	4,48
S324	Green Areas	41404,06	53,04
S324	Urban Areas	33163,79	42,48
S325	Urban Voids	21915,64	9,85
S325	Green Areas	16190,53	7,28
S325	Urban Areas	184347	82,87
S326	Urban Voids	338,1292	1,39
S326	Green Areas	23084,46	95,07
S326	Urban Areas	859,2847	3,54
S327	Urban Voids	4792,09	6,61
S327	Green Areas	59007,82	81,39
S327	Urban Areas	8697,842	12
S328	Urban Voids	2249,156	4,84
S328	Green Areas	5008,646	10,79
S328	Urban Areas	39173,95	84,37
S329	Urban Voids	737,6333	2,48
S329	Green Areas	24091,33	81,1
S329	Urban Areas	4876,284	16,42
S330	Urban Voids	506,5485	5,75

Drainage Area ID	Type of Use	Total Area (m²)	Percentage
S330	Green Areas	6906,015	78,45
S330	Urban Areas	1390,562	15,8

Source: Authors (2025).

Chapter 9: Annexes

Table 9-1: Data, sources and access links to obtain the information used in the study.

Data	Source	Acess
Maps: Land use and cover, Sanitary sewage system and Urban Evolution	Diagnóstico do Plano Diretor de Bauru (2019)	https://pdbauru2019.webflow.io/mapas-diagnostico
Map of flooding points	Municipal Urban and Rural Development Company (Empresa Municipal de Desenvolvimento Urbano e Rural (EMDURB))	https://www.emdurb.com.br/NET/transito/interdicao/enchentes/Mapa%20dos%20pontos%20de%20alagamentos.pdf
Aerial Images (1962, 1972, and 1979)	São Paulo Geographic and Cartographic Institute (Instituto Geográfico e Cartográfico de São Paulo (IGC))	https://www.igc.sp.gov.br/habit_igc/contato_2
Physicochemical water parameters	Bauru Water and Wastewater Department - DAE (Departamento de Água e Esgoto de Bauru)	Collaboration agreement DAE and UNESP
Digital Terrain Model	Bauru City Hall	https://www2.bauru.sp.gov.br/acessoainformacao/
Stream network	Bauru City Hall	https://www2.bauru.sp.gov.br/acessoainformacao/
Land Cover Raster	National Institute for Space Research (Instituto Nacional de Pesquisas Espaciais (INPE))	https://www.dgi.inpe.br/catalogo/explore
Rainfall Data	Meteorological Research Institute (Instituto de Pesquisas Meteorológicas (IPMET))	Request to institution

Source: Authors (2025).