UNIVERSIDADE DE SÃO PAULO

ESCOLA DE ENGENHARIA DE SÃO CARLOS

DEPARTAMENTO DE HIDRÁULICA E SANEAMENTO

MARINA BATALINI DE MACEDO

TÉCNICAS DESCENTRALIZADAS PARA RECICLAGEM DE ÁGUAS DE DRENAGEM URBANA VISANDO A SEGURANÇA HÍDRICA-ENERGÉTICA-ALIMENTAR

VERSÃO CORRIGIDA

SÃO CARLOS

2020

MARINA BATALINI DE MACEDO

TÉCNICAS DESCENTRALIZADAS PARA RECICLAGEM DE ÁGUAS DE DRENAGEM URBANA VISANDO A SEGURANÇA HÍDRICA-ENERGÉTICA-ALIMENTAR

Tese de doutorado apresentada à Escola de Engenharia de São Carlos, da Universidade de São Paulo, como parte dos requisitos para obtenção do título de Doutora em Ciências: Engenharia Hidráulica e Saneamento

Orientador: Prof. Dr. Eduardo Mario Mendiondo

VERSÃO CORRIGIDA

SÃO CARLOS

2020

Ficha catalográfica elaborada pela Biblioteca Prof. Dr. Sérgio Rodrigues Fontes da EESC/USP com os dados inseridos pelo(a) autor(a).

Batalini de Macedo, Marina
B141t Técnicas descentralizadas para reciclagem de águas de drenagem urbana visando a segurança hídrica- energética-alimentar / Marina Batalini de Macedo; orientador Eduardo Mario Mendiondo. São Carlos, 2020.

Tese (Doutorado) - Programa de Pós-Graduação em Engenharia Hidráulica e Saneamento e Área de Concentração em Hidráulica e Saneamento -- Escola de Engenharia de São Carlos da Universidade de São Paulo, 2020.

1. Bioretenção. 2. Técnicas Compensatórias. 3. Manejo de águas pluviais. 4. Cidades resilientes. 5. Objetivos de Desenvolvimento Sustentável. I. Título.

Eduardo Graziosi Silva - CRB - 8/8907

FOLHA DE JULGAMENTO

Candidata: Engenheira **MARINA BATALINI DE MACEDO.**

Título da tese: "Técnicas descentralizadas para reciclagem de águas de drenagem urbana visando a segurança hídrica-energética-alimentar".

Data da defesa: 23/11/2020.

Comissão Julgadora	Resultado
Prof. Dr. **Eduardo Mario Mendiondo** **(Orientador)** (Escola de Engenharia de São Carlos/EESC-USP)	APROVADO
Prof. Dr. **Vladimir Caramori Borges de Souza** (Universidade Federal de Alagoas/UFAL)	APROVADO
Profa. Dra. **Rutinéia Tassi** (Universidade Federal de Santa Maria/UFSM)	APROVADO
Prof. Dr. **Marllus Gustavo Ferreira Passos das Neves** (Universidade Federal de Alagoas/UFAL)	APROVADO
Prof. Dr. **Nilo de Oliveira Nascimento** (Universidade Federal de Minas Gerais/UFMG)	APROVADO

Coordenador do Programa de Pós-Graduação em Engenharia Hidráulica e Saneamento:
Prof. Dr. **Eduardo Mario Mendiondo**

Presidente da Comissão de Pós-Graduação:
Prof. Titular **Murilo Araujo Romero**

“Sem um fim social, o saber será a maior das futilidades”

Gilberto Freyre

AGRADECIMENTOS

Primeiramente gostaria de agradecer ao meu orientador, prof. Dr. Eduardo Mario Mendiondo, por ter aceitado me orientar no início da minha jornada acadêmica em um tema tão desafiador como drenagem urbana alternativa. Muito obrigada pelo apoio, pela confiança, pela parceria, por sempre me fazer pensar para além do tradicional e por desenvolver o espírito científico e inovador em mim. Espero que a gente siga colaborando em muitos mais trabalhos promissores com impacto social.

Também gostaria de agradecer o prof. Dr. David McCarthy, que me recebeu em seu laboratório e grupo de pesquisa na Monash University. Desde o início dos meus estudos sobre drenagem urbana alternativa li vários estudos desenvolvidos pelo seu grupo e sempre foi um sonho poder colaborar com vocês. Muito obrigada por ter me dado essa oportunidade, por toda a hospitalidade e gentileza que você e o grupo tiveram comigo durante esse período.

Tenho um enorme agradecimento também a todos os professores do SHS, que sempre que tinha alguma dúvida nas diferentes áreas de hidráulica, hidrologia ou saneamento estiveram disponíveis para me receber, me escutar e me ajudar a pensar em diferentes soluções, sendo um exemplo para mim do que significa trabalho colaborativo. Também tenho muito a agradecer aos técnicos do laboratório de saneamento, laboratório de hidráulica e laboratório de hidráulica computacional por terem me auxiliado nos trabalhos experimentais, me emprestarem ferramentas e me ajudarem a pensar soluções baratas para os meus experimentos. Um agradecimento especial a Sá e a Priscilla por todo o suporte burocrático, sempre com muito carinho e atenção.

Não poderia deixar de dar um agradecimento especial ao meu amigo Narumi Abe, que sempre esteve presente para as risadas, para os momentos difíceis e de angústia, para beber junto e, mais do que tudo, teve paciência para me ensinar programação, eletrônica, inteligência artificial, ficção científica e todas as outras coisas que mudaram minha vida nesses últimos seis anos.

Também devo um enorme obrigada aos meus colegas do “Grupo TC” do NIBH/WADI lab, Thalita Oliveira, Tassiana Halmenschlager, Marcus Nóbrega, César do Lago, Artur Brasil pela parceria e pelo trabalho conjunto. Com certeza sem vocês essa pesquisa não teria saído. Obrigada por todas as discussões, pelas revisões nos artigos, pela ajuda no trabalho experimental e pela paciência e persistência. A todos os outros colegas do NIBH/WADI também agradeço por sempre trocarmos experiências nas diferentes áreas que atuamos, contribuírem com ótimas ideias, e sobretudo pela amizade que construímos esses anos. Que ainda tenhamos muitos “Projetos XIBH” pela frente. Um agradecimento especial também a Maria Clara Fava e a Carol Buarque, por toparem fazer um trabalho extra que juntasse todas as nossas pesquisas e, principalmente, pela amizade, festas e chazinhos que tivemos juntas.

Gostaria de agradecer a todos os meus amigos de São Carlos e Uberlândia, que sempre me fizeram viver o melhor de ter um lar em duas cidades. Com certeza, com vocês ando melhor!

Aos meus pais e minha irmã, um obrigada eterno e imenso por serem sempre meu porto seguro, por terem me instigado o gosto em pesquisa, por terem lido e revisado vários dos meus trabalhos, por sempre acreditarem na minha competência e por me ajudarem a manter o pé no chão e não viver só no mundo das ideias. Obrigada por me nutrirem com muita comida boa e com muito amor.

Por fim, agradeço a Universidade de São Paulo pela infraestrutura cedida, ao CNPq pelo financiamento inicial da minha bolsa de doutorado e pelo financiamento do INCT Mudanças Climáticas, e a FAPESP pela concessão da minha bolsa de doutorado (processo 2017/15614-5), pela minha bolsa de estágio de pesquisa no exterior (processo 2019/05129-8) e pelo projeto regular que possibilitou o financiamento dos materiais para o trabalho experimental (processo 2017/21940-2). Mais do que nunca devemos fortalecer o conhecimento científico em nosso país, conectar ciência com ganhos sociais, e lutar contra os cortes na pesquisa e educação.

ACKNOWLEDGMENTS

First, I would like to thank my advisor, prof. Dr. Eduardo Mario Mendiondo, for having accepted to guide me at the beginning of my academic journey on a topic as challenging as alternative urban drainage. Thank you so much for the support, for the trust, for the partnership, for always making me think beyond the traditional and for developing the scientific and innovative spirit in me. I hope that we will continue collaborating on many more promising works with a social impact.

I would also like to thank prof. Dr. David McCarthy, who received me in his laboratory and research group at Monash University. Since the beginning of my studies on alternative urban drainage I have read several studies developed by your group and it has always been a dream to be able to collaborate with you. Thank you so much for giving me this opportunity, for all the hospitality and kindness that you and the group had with me during this period.

I am also very grateful to all the Professors and Lecturers of SHS, who whenever I had any questions in the different areas of hydraulics, hydrology or sanitation were available to welcome me, listen to me and help me think about different solutions, being an example for me of which means collaborative work. I also have a lot to thank the technicians of the sanitation laboratory, the hydraulics laboratory and the computational hydraulics laboratory for helping me with the experimental work, lending me tools and helping me to think about cheap solutions for my experiments. Special thanks to Sá and Priscilla for all the bureaucratic support, always with great care and attention.

I also have a special thanks to my friend Narumi Abe, who was always present for laughter, for difficult times and anguish, for drinking together and, more than anything, had the patience to teach me programming, electronics, artificial intelligence, talked to me about science fiction and all the other things that have changed my life in the past six years.

I also owe a huge thanks to my colleagues in the “TC Group” of the NIBH/WADI lab, Thalita Oliveira, Tassiana Halmenschlager, Marcus Nóbrega, César do Lago, Artur Brasil for the partnership and for the joint work. Surely without you, this research would not have come out. Thank you for all the discussions, for the text revisions, for your help with the experimental work and for your patience and persistence. I would also like to thank all the other colleagues at NIBH/WADI for always exchanging experiences in the different areas, contributing with great ideas, and above all for the friendship we have built over the years. Hope we still have many “Projetos XIBH” ahead of us. Special thanks also to Maria Clara Fava and Carol Buarque, for agreeing to do an extra job that would connect all our research together and, mainly, for the friendship, parties, and teas we had together.

I would like to thank all my friends from Sao Carlos and Uberlandia, who have always made me live the best of having a home in two cities. For sure, I’m better with you!

To my parents and my sister, an eternal and immense thanks for always being my safe place, for instigating my taste in research, for having read and reviewed several of my works, for always believing in my competence and for helping me to maintain down to earth and not living only in the world of ideas. Thank you for nourishing me with a lot of good food and a lot of love.

Finally, I would like to thank the University of Sao Paulo for the infrastructure provided, to CNPq for the initial financing of my doctoral scholarship and for the financing of the INCT Mudanças Climáticas, and FAPESP for the granting of my doctoral scholarship (process 2017/15614-5), for my grant in research internship abroad (process 2019/05129-8) and for the regular project that made it possible to finance materials for experimental work (process 2017/21940-2). More than ever, we must strengthen scientific knowledge in our country, connect science with social gains, and fight cuts in research and education.

RESUMO

MACEDO, M. B. (2020). **Técnicas descentralizadas para reciclagem de águas de drenagem urbana visando a segurança hídrica-energética-alimentar**. Tese, Escola de Engenharia de São Carlos, Universidade de São Paulo.

Técnicas compensatórias de drenagem urbana sustentável (TCs) são alternativas de adaptação de baixo custo que auxiliam os sistemas clássicos de drenagem, visando a mitigação dos riscos de extremos hidrológicos. Neste projeto, uma nova nomenclatura é utilizada para TCs frente a estes extremos de cenários futuros de mudanças e suas adaptações. Por exemplo, quando são adaptadas para mudanças de uso e ocupação do solo, p. ex. da urbanização, são denominadas Técnicas Compensatórias de 1ª geração (TCs-1G). Quando incorporam adaptações de uso do solo e mudanças climáticas, são denominadas de 2ª geração (TCs-2G). Ainda, quando a adaptação incorpora a reciclagem de recursos para segurança hídrica-energética-alimentar, são denominadas de 3ª geração (TCs-3G). Contudo, vazios científicos ainda permanecem, porque: poucas pesquisas avaliam a eficiência quali-quantitativa, de forma combinada, das TCs-1G e TCs-2G. Também, para as TCs-3G, existe um conhecimento limitado quanto ao emprego de cenários de mudanças climáticas e reciclagem de recursos para redução de riscos de segurança hídrica-energética-alimentar. Ainda, estes vazios ainda são maiores em áreas de clima subtropical e quando aplicadas de forma descentralizada, espacialmente dentro da bacia hidrográfica. Assim, esta pesquisa de doutorado aprimora um marco teórico-experimental, a partir do desenvolvimento conceitual e experimental sobre novas TCs-3G para uso descentralizado e para segurança do nexo água-energia-alimentos. A metodologia aborda três etapas: (1) análise de dimensionamento incorporando cenários futuros sob mudanças de urbanização e clima, (2) novos critérios de operação, manutenção e monitoramento quali-quantitativo de TCs visando terceira geração, em escala de campo e em escala de laboratório, (3) proposição e estudo de novos coeficientes de avaliação visando reciclagem de recursos, contribuição para segurança hídrica-energética-alimentar e sustentabilidade do local. Os resultados foram discutidos sob condições de: (a) clima subtropical, (b) padrões brasileiros de urbanização e (c) locais com demandas sociais por segurança hídrica-energética-alimentar.

Palavras chave: Bioretenção; Técnicas Compensatórias; Manejo de águas pluviais; Cidades resilientes; Objetivos de Desenvolvimento Sustentável.

ABSTRACT

MACEDO, M. B. (2020). **Decentralized Urban Runoff Recycling Facility addressing the security of the Water-Energy-Food Nexus**. Doctoral thesis, São Carlos School of Engineering, University of Sao Paulo, Sao Carlos.

Low Impact Development (LID) practices are low-cost adaptation alternatives to assist traditional drainage systems, aiming at mitigating the risks of hydrological extremes. In this project, a new nomenclature is used for LID, facing the extremes due to change in future scenarios and their adaptations. For example, when they are adapted for changes in land use and occupation, e.g. urbanization, they are called 1st generation LID (LID-1G). When incorporating adaptations of land use and climate change, they are called 2nd generation (LID-2G). Also, when the adaptation incorporates the recycling of resources for water-energy-food security, they are called 3rd generation (LID-3G). However, scientific gaps still remain because: few studies evaluate the combined qualitative and quantitative efficiency of LID-1G and LID-2G; for LID-3G, there is limited knowledge about incorporating climate change scenarios and resource recycling to reduce water-energy-food security risks; these gaps are still larger in areas of subtropical climate and when applied in a decentralized way. Therefore, this doctoral research enhances a new theoretical-experimental framework on a new LID-3G of bioretention, for decentralized use. The methodology addresses three stages: (1) analysis of design incorporating future scenarios with drivers of change of urbanization and climate; (2) new criteria for operation, maintenance and runoff volume and water quality monitoring for 3rd generation, in field scale and laboratory scale, (3) proposition and study of new evaluation coefficients aiming at resources recycling, contribution to water-energy-food security and local sustainability. The results were discussed under conditions of: (a) subtropical climate, (b) Brazilian urbanization standards, and (c) social demands for water-energy-food security.

Key-words: Bioretention; Low Impact Development; Stormwater management; Resilient cities; Sustainable Development Goals.

LIST OF FIGURES

LIST OF TABLES

LIST OF ABBREVIATIONS

ABNT	Associação Brasileira de Normas Técnicas *Brazilian Association of Technical Standards*
API	Antecedent Precipitation Index
BMP	Best Management Practices
CN	Curve number
COD	Chemical Oxygen Demand
CT	Compensatory Tecnhiques
DEAP	Distributed Evolutionary Algorithms in Python
DOC	Dissolved Organic Carbon
EDA	Exploratory Analsysis of Data
EMBRAPA	Empresa Brasileira de Pesquisa Agropecuária *Brazilian Agricultural Research Corporation*
EMC	Event Mean Concentration
EPA	United States Environmental Protection Agency
ET	Evapotranspiration
GCM	Global circulation model
GHG	Greenhouse gas
HAC	Hierarchical Agglomerative Clustering
IBGE	Instituto Brasileiro de Geografia e estatística *Brazilian Institute of Geography and Statistcs*
IDF	Intensity-Duration-Frequency
IEAG-SDGs	Inter-Agency and Expert Group on SDG Indicators
IPCC	Intergovernmental Panel on Climate Change
LCA	Life Cycle Analysis
LID	Low Impact Development
MD	Mapping distribution
NbS	Nature-based Solutions
NPD	Non potable demand
NS	Neighborhood scale
NSE	Nash–Sutcliffe coefficient

OAT	One-at-time (sensitivity analysys)
PDF	Precipitation-Duration-Frequency
PS	Property scale
PT	Power transformation
RCM	Regional climate model
RCP	Representative Concentration Pathway
RP	Return period
SCM	Stormwater Control Measures
SDG	Sustainable Development Goals
SNIS	Sistema Nacional de Informações sobre Saneamento *National Sanitation Information System*
SS	Street scale
STDRM	Space-time dynamic resilience measure
SUDS	Sustainable Urban Drainage Systems
SWAT	Soil and Water Aseessment Tool
SWMM	Storm Water Management Model
SZ	Saturated zone
TC	Total Coliforms
TN	Total Nitrogen
TOC	Total Organic Carbon
TP	Total Phosphorus
UN	United Nations
UNSTATS	United Nations Statistical Commission
USZ	Unsaturated zone
WQV	Water quality volume
WSUD	Water Sensitive Urban Design

TABLE OF CONTENTS

1 GENERAL INTRODUCTION

Since the 50s, Brazil went through an accelerated urbanization process with poor planning or its inaccurate implementation, leading to structural, social and environmental impacts in the urban space. As consequence, there has been a significant increase in runoff, causing the extremes of the natural hydrological cycle to become an urban problem. Extreme precipitation and drought events are precursor elements of risks to the population (Santos, 2007; Young et al., 2015), with greater vulnerability to floods and water stress. These risks may worsen with climate change scenarios (Debortoli et al., 2017; Valverde & Marengo, 2010; Marengo et al., 2010).

To mitigate the risks caused by the extremes in the urban drainage, the concept of Low Impact Development (LID) practices to urban drainage emerges (Fletcher et al., 2013). LID practices are based on the reestablishment of the natural hydrological cycle, or pre-urbanization, focusing on source control, runoff retention, infiltration to the groundwater, diffuse pollution control, landscape integration, and non-transfer of impacts to downstream, from a multidisciplinary approach with environmental education and social participation. Therefore, facing the impacts caused by land use and climate changes, these practices can be used as adaptation measures to increase the urban drainage system resilience and decrease of water insecurity.

Historically, the LID concept has evolved with new paradigms and challenges (Fletcher et al., 2015, Eckart et al., 2017), from simple flood mitigation and water quality control to ecological services and stormwater reuse. In this study we introduce the concepts of 1st, 2nd and 3rd generation of LID practices, according to their main mitigation purposes (Macedo et al., 2017). First generation LID aims to mitigate changes caused by land use, due to increasing urbanization. In this scenario, the change in the infiltration capacity of the soil significantly increases the runoff generation and the pollutant washed off in the catchment area to be further mitigated by the LID practices. Currently, most LID practices aim at 1st generation purposes (LID-1G).

Climate change and its non-stationary behavior in the hydrological cycle can also increase the occurrence of extreme precipitation events (Ambrizzi & Mangaña, 2016). Therefore, another scenario is given by the additional parcel of runoff and pollutant excesses on the urban drainage systems due to changes not only in the land use, but also in the magnitude, intensity and depth of future rainfall in the catchment. The LID practices that deal with this second paradigm are here called as 2nd generation (LID-2G).

Finally, we can insert the LID practices as units that integrate the urban hydrological cycle and their non-stationarities with catchment life cycles linked to water, food and energy security, the so-called *water-energy-food nexus*, from a perspective of integrated co-management of resources, contributing to the green economy and sustainability (Hoff, 2011). Within the premises for the security of the *water-energy-food nexus*, the aim is to increase resource productivity, with innovative technologies for social well-being. In this sense, LID practices can be adapted to allow resources recycling for multiple uses in the urban life cycle, originated from the total or partial reuse of the LID effluent pollutants mass or drained volumes. When LID practices incorporate the co-management and reuse of resources, they are here called as 3rd generation (LID-3G)

However, there are scientific gaps about LID-3G concerning their real, practical and low-cost viability. The effects of climate change in the LID practices design methodologies and its performance over time still need a more comprehensive assessment. Additionally, there are still some interrogations about how to evaluate their contribution to resources recycling and co-management. These gaps are intensified in Brazilian cities, since the largest amount of studies is conducted in areas of temperate climate, in which geoclimatic, sanitary and social conditions are quite different from those in areas of Brazilian subtropical climate. Therefore, the study of adaptations and monitoring of LIDs for tropical and subtropical regions is still a subject that needs further attention.

Considering these questions, this doctoral thesis aimed to evaluate the design, construction, use and maintenance of LID practices applied in different scales in the control of water quantity and quality on urban drainage, mitigating the extremes of urbanization and climate in the urban hydrological cycle. It also aimed to explore the possibility of recycling resources, based on water reuse, nutrient cycling, and energy use by these devices, guaranteeing the water-energy-food security.

This doctoral thesis advances on conclusions and recommendations of studies supported by FAPESP 08/58161-1 “Assessment of impacts and vulnerability to climate change in Brazil and strategies for adaptation option”, of water security at watershed scale and with LID-1G developed by Macedo (2017) (FAPESP 2015/20979-7) and Rosa (2016) (FAPESP 13/06611-1). Also, the study incorporates experimental and modeling elements for the current INCT

Climate Change-II (FAPESP 2014/50848-9 and CNPq 465501/2014-1), co-led by CEMADEN/MCTIC and IAG/USP (Marengo & Ambrizzi, 2016).

1.1 Research hypothesis

The use of LID practices as adaptation measures improve the decentralized stormwater management, reducing the impacts of extreme events on the urban drainage infrastructure and urban hydrology, in terms of runoff quantity (floods) and water quality (contamination of streams by diffuse pollution). In addition, the decentralized use of LID practices contributes to resources co-management, resources recycling and the increase of water-energy-food security, aiming at the development of resilient cities under changing scenarios of urbanization and climate.

1.2 Purpose

1.2.1 General purpose

Improve the scientific framework on a new generation of LID practices, more specific to bioretention, called 3rd generation (LID-3G), based on its conceptual and experimental development. Based on the experimental monitoring in field and laboratory scales, to evaluate the bioretention efficiency for runoff retention, pollutant removal and water-energy-food security of future resilient cities under subtropical climate.

1.2.2 Specific purpose

- Incorporate scientific and technological elements for a new terminology of LID practices, more specif to bioretention, aiming at actions to adapt to the risks of urban drainage extremes, based on the current state of the art and conceptions of LID practices in operation in Brazil and abroad;
- Evaluate the current design methods and propose a modular-adapted implementation of bioretention that incorporates future scenarios with drivers of change;

- From experimental bioretention in real and laboratory scale, to evaluate the operation, maintenance, monitoring and resources recycling for water-energy-food security, under subtropical climate conditions;

- Evaluate the efficiency of bioretention-3G based on new proposed coefficients, to reestablish the water balance of pre-urbanization and resource recycling, incorporating drivers of change, with urbanization growth, climatic variability and consumption habits.

1.3 Text organization

This doctoral thesis is organized in 7 chapters, containing a **first chapter** with the general introduction, hypothesis, and purposes of this study. In **chapter two** it is presented a literature review of the studies with LID practices aiming at urbanization mitigation, climate change adaptation and additional purposes of resources recycling and co-management. In this chapter, we introduce the concept of generations of LID and present different pathways to their contribution to Sustainable Development Goals (SDG). In **chapter three** an assessment of the different pre-design methods generally used for bioretention practices is made, identifying the most sensitive parameters and input variables related to urbanization and climate change that must be considered when sizing these practices for future scenario. Additionally, the modular sizing is presented as an alternative to maintaining the performance over time for future scenarios in a more cost-accessible way. **Chapter four** presents the results of experimental monitoring of a bioretention in street scale, conceived as practice of 2nd generation, and their potential to water reuse integration. **Chapter five** presents the experimental results of a bioretention of 3rd generation in laboratory scale, assessing it performance for different configuration and operation conditions and quantifying it contribution to different SDG by proposed metrics. In **chapter six**, a bioretention model for water flow and nitrogen removal was developed, which will support future studies on identifying optimal configuration for nutrient recycling in bioretention practices. Finally, **chapter seven** presents a general conclusion and recommendations for future studies. The general thesis structure is presented in the flowchart in Figure 1.1.

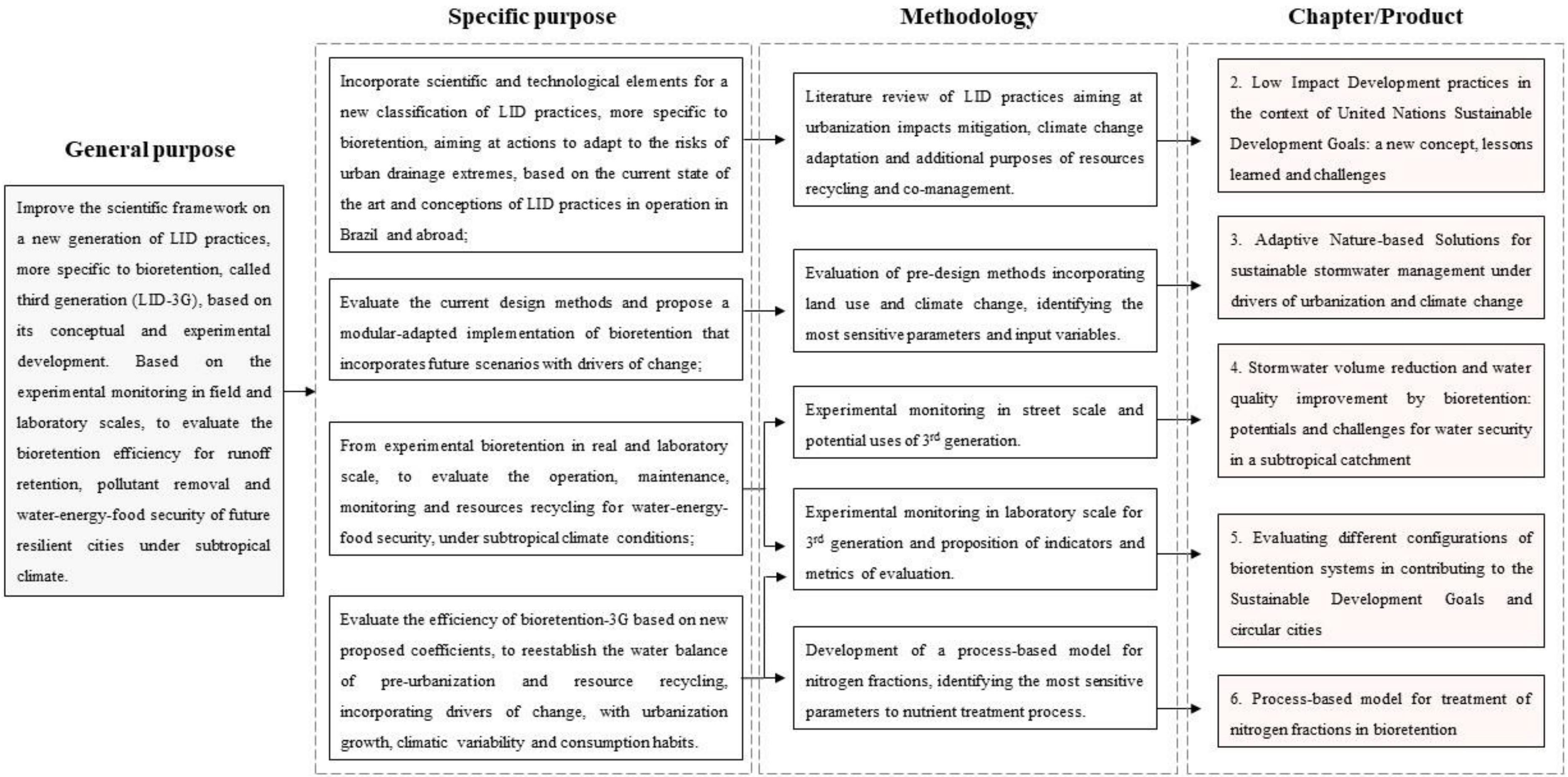

Figure 1.1 – Flowchart representing the thesis structure

References

Ambrizzi, T., & Magaña, V. (2016). *Mudanças Climáticas e Projeções Futuras de Eventos Extremos. Em Desastres - Múltiplas Abordagens e Desafios*. 1st edition. Elsevier.

Debortoli, N. S., Camarinha, P. I. M., Marengo, J. A., & Rodrigues, R. R. (2017). An index of Brazil's vulnerability to expected increases in natural flash flooding and landslide disasters in the context of climate change. *Natural hazards*, 86(2), 557-582.

Eckart, K., McPhee, Z., & Bolisetti, T. (2017). Performance and implementation of low impact development–a review. *Science of the Total Environment*, 607, 413-432.

Fletcher, T. D., Andrieu, H., & Hamel, P. (2013). Understanding, management and modelling of urban hydrology and its consequences for receiving waters: A state of the art. *Advances in water resources*, 51, 261-279.

Fletcher, T., Shuster, W., Hunt, W., Ashley, R., Butler, D., Arthur, S., . . . Mikkelsen, P. (2015). SUDS, LID, BMPs, WSUD and more – The evolution and application of terminology surrounding urban drainage. *Urban Water Journal*, 12(7), 525-542.

Hoff, H. (2011). Understanding the nexus: background paper for the Bonn2011. *Nexus Conference*

Macedo, M., Lago, C., Rosa, A., & Mendiondo, E. (2017). Técnicas compensatórias de bioretenção para cidades resilientes: integração com nexus–água, energia, alimento. [Low Impact Development practices of bioretention to resilient cities: integration with water-energy-food nexus]. *XXII Simposio Brasileiro de Recursos Hidricos*. Florianopolis, Brazil. (In Portuguese)

Marengo, J. A., Ambrizzi, T. (2016). *Programa INCT Mudanças Climáticas (INCT-MC-II)*, CNPq 465501/2014-1.

Marengo, J. A.; Schaeffer, R.; Zee, D.; Pinto, H. S. (2010) *Mudanças climáticas e eventos extremos no Brasil*. [Climate change and extreme events in Brazil] Available in: http://www. fbds. org. br/cop15/FBDS_MudancasClimaticas. pdf. Accessed in October, 2010. (in Portuguese).

Santos, R. F. (2007). *Vulnerabilidade ambiental: desastres naturais ou fenômenos induzidos?* [Environmental vulnerability: natural disasters or induced phenomena?]. MMA (in Portuguese).

Valverde, M. C.; Marengo, J. A. (2010). Mudanças na circulação atmosférica sobre a América do Sul para cenários futuros de clima projetados pelos modelos globais do IPCC AR4. [Changes in atmospheric circulation over South America for future climate scenarios projected by IPCC AR4 global models]. *Revista Brasileira de Meteorologia*, 25(1), 125-145. (in Portuguese)

Young, C. E. F., Aguiar, C., Souza, E., 2015. *Valorando tempestades: custo econômico dos eventos climáticos extremos no Brasil nos anos de 2002 – 2012* [Assessing storms:

economic cost of extreme weather events in Brazil in the years 2002 – 2012], first ed. Observatório do clima, São Paulo (in Portuguese).

2 LOW IMPACT DEVELOPMENT PRACTICES IN THE CONTEXT OF UNITED NATIONS SUSTAINABLE DEVELOPMENT GOALS: A NEW CONCEPT, LESSONS LEARNED AND CHALLENGES

A modified version of this chapter has been submitted as: Marina Batalini de MACEDO, Marcus Nóbrega GOMES JUNIOR, Thalita Raquel Pereira de OLIVEIRA, Marcio H. GIACOMONI, Maryam IMANI, Kefeng ZHANG, César Ambrogi Ferreira do LAGO, Eduardo Mario MENDIONDO. **Low Impact Development Practices in the context of United Nations Sustainable Development Goals: A new concept, lessons learned and challenges**. Critical reviews in Environmental Science and Technology.

Abstract

Low Impact Development (LID) practices can be used as a tool to achieve more resilient cities and communities by integrating runoff retention, water quality improvement, stormwater harvesting, water reuse with social impacts, health, well-being and economic incentives. For this purpose, the LID practices can be understood as an infrastructure related to nature-based solution (NbS) towards UN SDG. First, a new concept of generations is presented to address different LID purposes and potential contribution to the SDGs. When LID practices are adapted for changes in land use, e.g. urbanization, they are called 1st generation LID (LID-1G). When incorporating adaptations with time-flexible and modular setups to tradeoff effects of changes in scenarios of climate change altogether, they are called 2nd generation (LID-2G). When the adaptation incorporates recycling strategies, recovering and reuse of resources for water-energy-food security, they are called 3rd generation (LID-3G). Further, the review identified the absence of guidelines that incorporate urbanization and climate timescales, clear methodology for identifying pollutant flows in soil, vegetation and air, and well-established metrics for assessing the contribution to SDG and resilience as the major challenges yet found.

Keywords: Resilience; Stormwater Harvesting; Carbon sequestration; Climate Change; Water-energy-food nexus.

2.1 Introduction

Several cities worldwide experience problems related to hydrological extremes, e. g. flood events due to intense rainfall and high periods of droughts. These events affect the water security of communities. On the one hand, the increasing urbanization, land use, and paving contributes to a higher runoff generation and, combined with insufficient urban planning and lack of urban drainage structures, aggravates the problems caused by hydrological extremes, intensifying the frequency of flood events (Lucas & Sample, 2015; Guan et al., 2015). In the occurrence of floods, the allocation of people in risk areas, such as river floodplains and hill slopes, in many megacities leads to significant economic losses and injury risks for the population (Sun et al., 2017; Carter, et al., 2015; Douglas, et al., 2010). On the other hand, the high population density in the cities leads to high water demands, while high periods of droughts and urban river contamination by sewage disposal and diffuse pollution threaten the availability of reliable water resources (Fletcher et al., 2013).

Climate change scenarios predicted by recent Intergovernmental Panel on Climate Change (IPCC) reports indicate that the increase of global temperature will affect rainfall patterns in a way that extreme events, both of drought and flood, tend to get worse (Rosinger, 2018; Mohor & Mendiondo, 2017; Kirchhoff et al., 2016; Ambrizzi & Magaña, 2016; Carter et al., 2015). In this sense, the previous problems faced by the cities will be intensified. Therefore, it is necessary to adopt measures aiming to increase the cities resilience, ensuring population access to a healthy, safe and fare environment.

Many approaches, techniques and policies have been presented to increase society and urban resilience. In 2015, the United Nations presented the Sustainable Development Goals (UN SDG) (UN, 2020) as an agenda to be met by countries until 2030 in order to move towards a resilient and prosperous global society for the people and the planet. This agenda presents 17 goals that require urgent action to be taken by all countries to ensure greater social welfare, health, education, reduce inequalities and, at the same time, preserve natural resources and combat climate change.

Specifically when it comes to stormwater management to control the risks generated by flood and drought extremes (intensified by climate change), the alternative urban drainage systems concept emerges around the world as a strategy that facilitates resilience implementation regarding urban drainage systems (Fletcher et al., 2013). Different nomenclatures can be used for this strategy depending on the geographical location and influence. They are known as Best Management Practices (BMP), Low Impact Development (LID), Green Infrastructure or Stormwater Control Measures (SCM) in the USA, Sustainable Urban Drainage Systems (SuDS) in Europe, Compensatory Techniques (CT) in France and Brazil, Water Sensitive Urban Design (WSUD) in Australia, and Sponge City in China (Eckart et al., 2017; Jun, et al., 2017; Fletcher et al., 2015). Some authors present different conceptualization for each nomenclature, however, they have been used as synonyms in international academic papers (Fletcher et al., 2015). In this paper, the LID practice nomenclature will be adopted.

LID practices are based on restoring the natural hydrological cycle, or pre-urbanization cycle, focusing on water infiltration and integrated efficiency in runoff retention and improvement of water quality (The Prince George's County, 2007). Therefore, LID practices can vary from non-structural measures, such as the implementation of policies to reduce the runoff generation, and structural measures aiming at the induced infiltration, retention, (bio)filtration, runoff control at the source, urban landscape integration, and non-transference of the impacts downstream (Fletcher et al., 2015). These practices should have a multidisciplinary approach, with environmental education and social participation.

Due to the multiple purposes of these systems in flood control, water treatment and pathogens removal, stormwater harvesting, carbon sequestration, among others, they can potentially cooperate for different UN SDGs, such as (6) clean water and sanitation (Fletcher et al., 2008, Jing et al., 2017), (13) climate action (Brudler et al., 2016; Zahmatkseh et al., 2015), (7) affordable and clean energy (Ramos et al., 2013; Nair et al., 2014), (3) good health and well-being (Chandrasena et al., 2016) and (11) sustainable cities and communities (Moore & Hunt, 2012), as well as to reduce flood and water insecurity risk exposure.

The aim of this study is to investigate the potentials of LID practices to contribute to different UN SDGs and increase resilience in cities, facing drivers of change in urbanization and climate. Two strategies were adopted to this purpose: (1) to propose a new concept of generations to classify LID practices according to different purposes of mitigation, considering future scenarios, different drivers of change (such as urbanization and climate), and their potential contribution to the UN SDGs; (2) to present a review of the literature of the studies already developed in each LID generation, identifying the gaps and potentials that still need to be explored. Finally, in order to comparatively evaluate new studies developed on this field, new metrics were proposed and a method for evaluating the dynamic resilience of urban watersheds was presented.

2.2 New concept: 1st, 2nd and 3rd generation LIDs - changes in urbanization, climate and integration with UN SDG

Historically, LID practices have evolved with new paradigms and challenges, such as initially water quality improvement linked to flood mitigation, and, recently, water recycling and water security (Fletcher et al., 2015). Guides and manuals for building LID practices in various countries present several classical design purposes, for example, maintenance of recharge volume (for re-establishment of the water cycle), improvement of water quality (first flush treatment), channel protection, reduction of excess runoff volume to protect against channel overflow (flood), and peak flows amortization (Waterways, 2005; The Prince George's County, 2007; Council, 2007; McAuley, 2009). Recently, stormwater harvesting, ecosystem services, and carbon sequestration are starting to be considered as design purposes (Ge et al., 2016; Moore & Hunt, 2012).

However, designing these practices to meet different purposes is not enough to ensure site resilience if the timescale of its application is not considered, i.e. considering only the current characteristics of the catchment. There are drivers of change in the cities affecting considerably the runoff production and pollutant generation, such as changes in land use due to the increasing urbanization and climate change (Liu et al., 2016; Liu et al., 2017). Therefore, the timescale including the future scenarios with drivers of change should also be considered in the conception and design of a LID practice.

In order to integrate these new paradigms, a new concept of LID’s generation is introduced in this paper (Figure 2.1). The generations are differentiated according to the timescale, drivers of change considered, and resilience purpose, evaluating an event with the same return period for different scenarios. This classification aims to clarify that there is an advance in the problems to be addressed in flood management and the need to integrate recent studies and technologies with the demands of the UN SDG. Therefore, the adoption of LID practices from a new paradigm of resource recycling, stormwater harvesting, watershed life cycle and sustainable communities enables to increase the resilience of a natural-social environment that cannot be afforded by other classic structural measures.

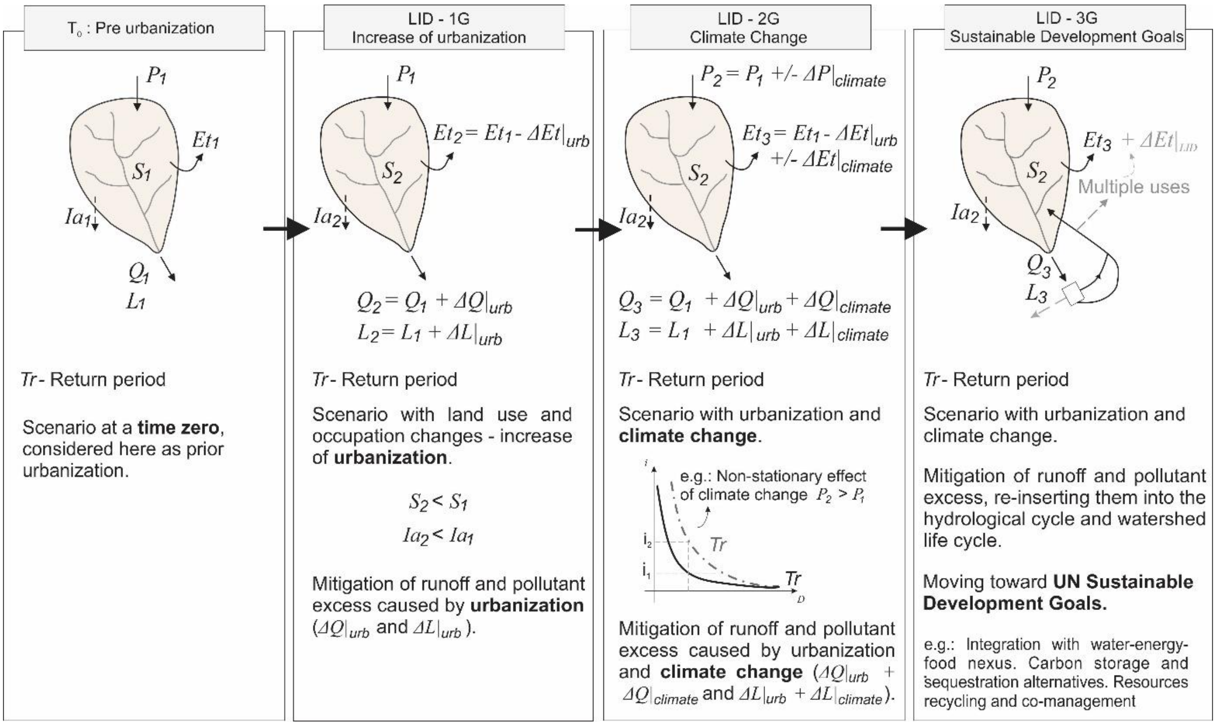

Figure 2.1 - Short-term water balance (during a rainfall event) for LID practices of first, second and third generation, presenting the differences in the excesses to be mitigated for the same return period. In the figure, P_1, Q_1, L_1, S_1, Ia_1 and Tr_1 represent, respectively, rainfall, runoff, pollutant load, soil storage capacity, infiltration and return period to base scenario of pre-urbanization. For the other scenarios there is an additional parcel of each of the variables due to changes in land use and climate patterns (adapted from Macedo et al., 2017).

In Figure 2.1 it is presented a scheme for a better understanding of the generations concept. The initial time T_0 is considered as the period prior to urbanization, before significant anthropogenic changes in the watershed occur. This is considered as the base scenario. In this scenario, any rain (P_1) that falls on the watershed is separated between infiltration, soil storage, runoff and evapotranspiration (ET). However, in this case it is considered only the short-period water balance, during the rain event, so the effect of ET is considered negligible. With changes in soil characteristics caused by urbanization (increased paving and change in slope) there is a reduction in both the soil storage and infiltration capacity, so that the same rainfall P_1 generates a larger volume of runoff relative to the base scenario (named as exceeding runoff $\Delta Q|_{urb}$, which is responsible for flood problems in the cities) (Leopold, 1968; Konrad & Booth, 2005; Wong & Eadie, 2000; Stovin et al., 2013). In addition to the consequence of the amount of runoff generated, the urbanization also leads to pollution problems, so that waste and other pollutants deposited on the surface are carried to the water bodies with the runoff (named as exceeding load $\Delta L|_{urb}$, which is responsible for the contamination of urban rivers) (EPA, 1983). Therefore, the stormwater quality and quantity issues caused by urbanization have led to the emerging of the first generation of LID (LID-1G).

However, nowadays the medium and long-term strategic planning, i.e. incorporating timescale, must also be considered in order to make cities more resilient to the changings in urban space. Therefore, the future scenarios must be addressed, considering all the drivers of change, such as urbanization and climatic patterns. Global climate change also becomes a regional and local problem, changing rainfall depth, intensity and frequency of events, contributing to the increase of droughts and flood extremes (Gersonius et al., 2012; Arnone et al., 2013; Chou et al.; 2014). Therefore, there is an additional rainfall depth for a rain P_1 ($\Delta P|_{climate}$), which generates a new volume of exceeding runoff ($\Delta Q|_{climate}$). This additional volume must be considered in the design of LID structures regarding long-term flood mitigation. The $\Delta P|_{climate}$ will also lead to different process of pollutant build up and wash off, affecting the pollutant load in the runoff ($\Delta L|_{climate}$) (Liu et al., 2016; Liu et al., 2017; Lago, Macedo & Mendiondo, 2018). In this case, where both urbanization and climate are considered, the LID practices are called of 2nd generation (LID-2G).

Finally, changes in future scenarios threats natural resources availability and social environments. In terms of natural resources, the climate change affects the rainfall patterns and contribute to increase rainfall and drought extremes, reducing water security. Simultaneously other resources insecurity also increases, such as energy and food (the link between these resources is presented by Hoff (2011), as the *water-energy-food nexus*). Measures that help fix carbon in the soil and biosphere also contribute to the environment sustainability. It is, therefore, necessary to think of new approaches that are able to consider circular mitigation, where the exceeded runoff volume or pollution is seen not only as something to be eliminated, but as a possibility of resource to be reinserted into the watershed life cycle, moving toward sustainable and resilient communities. Therefore, for the last scenario, the exceeding runoff is mitigated using LIDs and the runoff is reinserted in the watershed life cycle (by stormwater harvesting, nutrient recycling, carbon sequestration) to achieve multiple sustainability purposes (proposed in the UN SDG). In this last scenario, the LID practices are called as 3rd generation (LID-3G).

In order to integrate the new concept with previous studies, the Figure 2.2 presents the LID generations concept according to evolution of LID practices presented by Fletcher et al. (2015). The origin of the urban drainage concept was thought only in flood mitigation, later integrated to water quality, which is defined as LID-1G purposes. These purposes are also listed in the Bioretention Manual, e.g., developed in the USA by The Prince George's County, and in the WSUD Guidelines, developed in Australia by the South Eastern Councils (The Prince George's County, 2007, Waterways, 2006). Currently, most of the alternative systems applied serves to 1st generation purposes (LID-1G).

According to Fletcher et al. (2015), in 2013, aspects of 3rd generation were already involved in the LID systems design (even though the terminology was not used) such as urban harvesting (stormwater as resource), ecosystem ecology and resilience (along with microclimate), that is, integrating nature-based solutions, and targeting different sustainability purposes (UN SDG: clean water and sanitation, climate action, affordable and clean energy, good health and well-being). However, Fletcher et al. (2015) does not present temporal scaling and future changes of the hydrological cycle that occurs also by the climate changes and land use changes. This timescale should not be forgotten during a planning or design of a LID system. Therefore, in Figure 2.2, the urban drainage system evolution presented by Fletcher et al. (2015) has been adapted according to the proposed concept of generations and incorporating the aspects of climate change and land use changes.

Although there are already many studies addressing the new purposes of LID that meets the UN SDG, there is still no systematization of nomenclatures that incorporates these new approaches with the usual purposes of runoff retention and water quality improvement, i.e. the impact of non-stationary effects of climate change, modulation and integration with water-energy-food security, climate action and sustainable cities and communities. In addition, the classification of LID generations indexes a temporal efficiency attribute – unlike usual studies that approach purposes based on static criteria – allowing the maximization of the resilience of these systems over time. Therefore, the concept of LID generations is a form of categorization that helps to visualize and compare different studies, discriminating and analyzing the different approaches, and it evidences the advances on flood management issues to be addressed and a new resource cycling paradigm to increase mitigation and resilience to extremes, moving towards the UN SDG.

In the further sections it is presented a review of papers that corresponds to studies in LID-1G, LID-2G and LID-3G. This review aims to exemplify the studies developed for each purpose of each generation, stating the lessons learned, and identifying challenges that remain for integrate LID practices with the UN SDG.

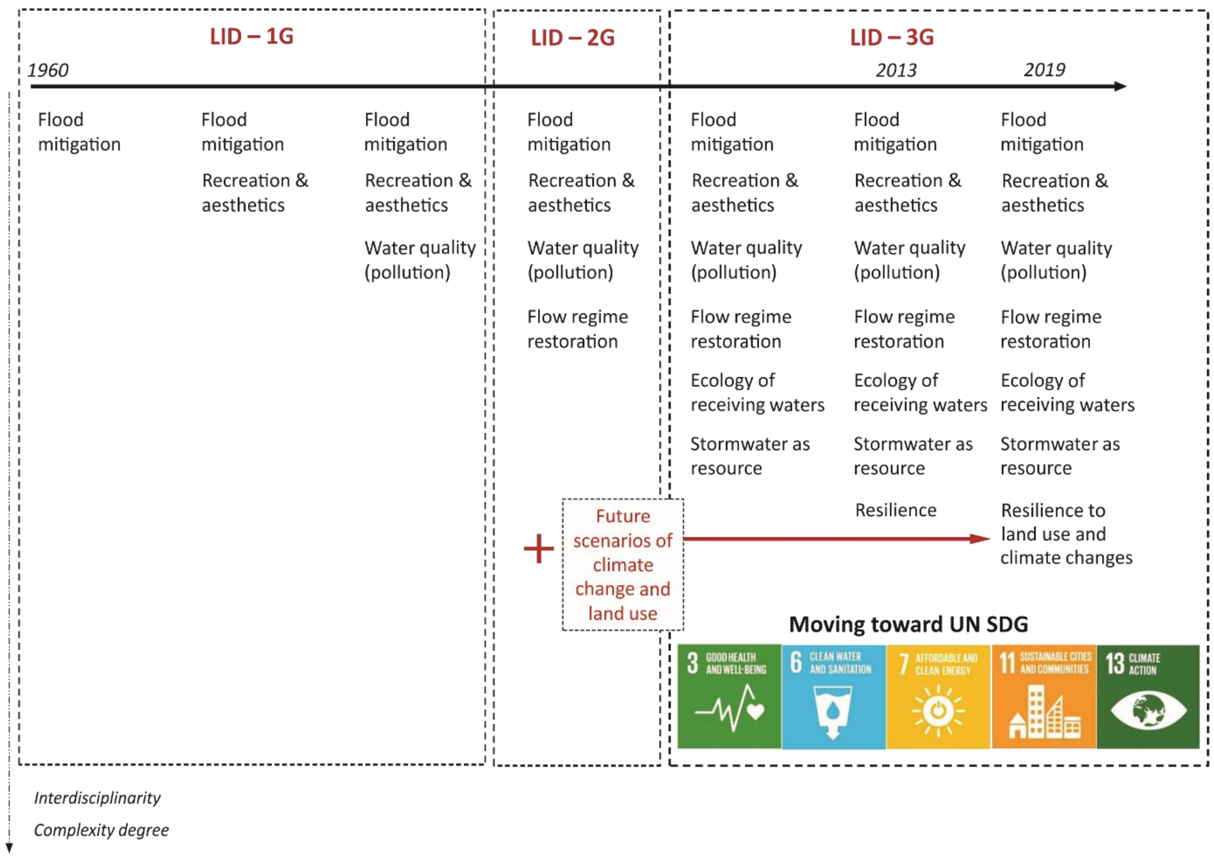

Figure 2.2- New concept: Incorporation of aspects of the hydrological cycle, water quality and watershed ecohydrology into the urban drainage concept over time and its evolution within the concept of LID generations. Adapted from Fletcher et al. (2015).

2.3 LID - 1G – Increase of urbanization

Studies involving applications of LID practices to mitigate urbanization excesses can be separated into two types: static studies and time scale studies. In this section, it is first presented the static studies, as they are the most developed to date. At the end, new perspectives to incorporate time scale in the evaluation of urbanization are presented.

2.3.1 Static evaluation of performance of LID practices to urbanization impacts

One of the LID practices that has been extensively studied is bioretention, due to its ability to both mitigate floods and promote pollutant removal. Table 2.1 presents a summary of studies developed with 1st generation bioretention, both in relation to runoff retention, as well as water quality treatment.

Regarding flood control purposes, the results presented in Table 2.1 show different performance results for each practice evaluated but with a trend in the capacity of flood peaks mitigation and runoff volumes reduction. However, the variability of results indicates the complexity in the general assessment of LID structures, since local factors such as degree of urbanization, soil type, filtering media, as well as climatic characteristics (rainfall, drought time, and rainfall intensity) act jointly on the devices efficiency. This complexity of factors acting on the bioretention performance was evaluated by Macedo et al. (2019a), for a subtropical climate locality, obtaining the antecedent soil moisture and runoff generation rate as the main environmental factors affecting the performance during the dry period, while the rainfall depth and intensity have the greatest influence on the rainy season.

The diversity of results obtained in the different studies shows that there is still a need to better understand the influence of environmental, climatic and constructive factors of LID practices on their performance. From this understanding, the design guidelines and manuals should be updated with recommendations for different combinations of factors. Most current guides address only temperate climate locations and consider constructive aspects without major variations, such as local soil, vegetation, filter media.

Regarding pollutant control, the results presented in Table 2.1 show that there is still a great variability in nutrient removal rates (nitrogen and phosphorus). Soil type, filter media, vegetation (Litern et al., 2011; Pivetta et al., 2019), structural configuration (Pivetta et al., 2019) and climate (Mangangka et al., 2015) are some of the main factors affecting this removal. Configurations including a vegetation layer and an anaerobic zone are recommended to optimize nutrient removal (Glaister et al., 2016; Sun et al., 2017; Wan et al., 2017). However, nutrient species responded differently to changes in inflow volume and dry weather antecedence, making nutrient removal optimization still a challenge (Glaister et al., 2016).

As the nutrient removal efficiency is related to soil characteristics and previous dry period (which affect the nitrogen cycle, for example), it is also necessary to expand the studies to tropical and subtropical regions and evaluate the effect of the climate in their efficiency. In tropical regions it is common to have long drought periods, affecting the biological behavior inside the LID practices and the vegetation survivor. Once the vegetation and biological treatment plays an important role in the pollutant removal, it is necessary to evaluate how these long drought periods will affect the pollutant control.

Table 2.1 - Summary of the results obtained for hydrological and pollutant removal performance on bioretention studies worldwide

Author	Obs.:	Peak flow attenuation (%)	Runoff retention (%)	Pollutant removal (%)										
				Cu	Pb	Cd	Zn	TSS	TP	PO_4	TN	NH_3	$NO_{2,3}$	Micro-pollutants***
Bratieres et al. (2008)*	Soil, sand, gravel with none vegetation							99	81		-204			
Chahal, Shi & Flury (2016)*	Sand and compost			-260 to 60									-53 to -1100	
Daniel Junior (2013)		100	100											
Davis (2007 and 2008)		44 to 63	55 to 70	57	83		62	47	76				83	
Ferreira (2017)			89											
Hatt, Fletcher & Deletic (2009)		37 to 96	15 to 83	67	80	-	84	76	-		-	64		
Liu et al. (2014)*									96 to 98.8					
Lucke and Nichols (2015)		79.5 to 93.6	32.7 to 84.3					-1295 to 100	-8820 to 100		-426 to 100			
Macedo et al. (2019a)	Dry period		70				76.2			61.1		67.7	1 to 69.5	
Mangangka et al. (2015)**	Dry period		22 to 90					80.8	75.3	73.4	47.9	82.2	65.0	
	Rainy period		12 to 38					41.8	36.4	37.8	38.7	49.3	23.2	
Moura (2013)				98		99	98	96 to 99			97			
Shrestha et al. (2018)		86 to 96	48 to 96					93	-35 to -285		-24 to 67		-272 to 77	
Wan, Li & Shi (2017)*	Layered/wood chips										83		81	
Wang et al. (2017)*	Stepped / *Medicago sativa*										-28.8 to -123.0			
	Stepped / *Vetiveria zizanoides and others*										52.8 to 84.2			
Winston, Luell & Hunt (2011)	Undersized							25.3	-38.4		40.2	23.5	62.6	
	Full sized							50.4	3.2		47.6	54.8	75.6	
Winston, Dorsey & Hunt (2016)		24 to 96	36 to 59											
Zhang et al. (2014)	Loamy sand with no submerged zone													40.6 to 99.6
	Sand with submerged zone													6 to 99.6

* lab scale, ** simulation, *** micropollutants analyzed: glyphosate, atrazine, prometryb, simazine, chloroform
Cu - Copper; Pb - Lead; Cd - Cadmium; Zn - Zinc; TSS - Total suspended solids; TP - Total phosporus; PO_4 - Phosphate; TN - Total nitrogen; NH_3 - Amonium - $NO_{2,3}$ - Nitrite and nitrate

2.3.2 New perspectives for time scale evaluation in LID practices for urbanization impacts

The studies presented above are examples of LID practices implementation with mitigation purposes considering only the effects of increased urbanization on the hydrological cycle and pollutants generation. However, these studies are static, not considering the effects of increasing urbanization for future scenarios. To ensure long-term resilience of cities, it is necessary to incorporate drivers of changes in the assessment and sizing of the devices. New studies are incorporating the efficiency evaluation from the simulation of future scenarios in models.

Liu et al. (2016) evaluated through modeling the effect of land use change on the generation of runoff volume and pollutant loads for an urban catchment between 2001 and 2050, resulting in increases between 8% and 17.9% of the total runoff volume generated. From these generated excesses, the authors estimated the amount of green infrastructure to be implemented for their mitigation. Liu et al. (2017) achieved a 1% increase in curve numbers for future urbanization scenarios (from 2001 to 2050), which led to a 1.2 to 17.5% increase in runoff volume in their study area.

Wang et al. (2016) evaluated the cost-effectiveness of bioretention practices under future urbanization scenarios (varied from the Shared Socio-economic reference Pathways – SSP). As a result, urbanization has more effect on surface runoff quality (total suspended solids – TSS loading) than on runoff peak. In addition, the major costs of bioretention are associated with maintenance and transportation activities. However, it is important to highlight this is a site-specific result and it can vary depending on the catchment characteristics.

It is possible to observe that there are still few studies that evaluate the effects of urbanization for future scenarios in the preliminary conception of LID practices, and this factor is not yet presented in the guidelines and manuals for building LIDs. However, to ensure efficiency and long-term resilience maintenance, it is necessary to incorporate timescale aspects right into the LID framework design step. One of the possibilities to consider future urbanization scenarios into the classical designs is by modifying the surface runoff separation constants, according to the method to be used. For example, updating the curve number (Liu et al., 2017) or the runoff coefficient is a simple and easy-to-use way for pre-sizing and design of

LID structures or continuous simulations (e.g. using Storm Water Management Model – SWMM).

2.4 LID - 2G – Climate change

Besides urbanization, another driver of change that should be considered in time-scale studies and design is the change in the future climate pattern. In this section, it is presented studies that address the consequences of climate change in urban catchments and perspectives of their incorporation into design guidelines.

2.4.1 Climate impact mitigation through LID systems

Figure 2.3 presents the results of studies that assessed the impacts of climate change on urban catchment and their drainage systems in different regions of the world, as well as the mitigation capacity provided by LID practices. Impacts vary from regions, some of which has increased rainfall in the wet month (Carter, et al., 2015), and there is a reduction in some (Liuzzo et al., 2015; Arnone et al., 2013; Lyra et al., 2018), but overall, climate change tends to increase rainfall extremes (more intense rainfall). As seen in the previous section, rainfall intensity is one of the factors that most affect the performance of LID practices in the rainy season. In addition, the decrease in total rainfall leads to reduced water availability. Additionally, the increased drought period leads to higher pollutant build up and wash off (Liu et al., 2016; Liu et al., 2017; Lago et al., 2018).

In watershed and city scale, Brudler et al. (2016) used a lifecycle approach to quantify the environmental impacts of climate change in the classic drainage system when compared to system integrated with LID practices, in the city of Copenhagen in Denmark. They have concluded that the classical systems have up to 5x more impacts on the environment than the adaptive measures using LID. The studies of Dudula & Randhir (2016) and Paola et al. (2015) evaluated the effectiveness of LID practices implantation from hydraulic and hydrological models. Common results show that exceeding runoff generated by climate change can be mitigated by the correct use of LID practices in the urban drainage system, if carefully designed and maintained. In this same sense, Zahmatkseh et al. (2015) showed that, while average increase in historical annual runoff volume under climate change was of approximately 48%, the LID controls could provide an average reduction of 41% in annual runoff volume. Application of the LID controls also reduced peak flow rates by an average of 8% to 13%.

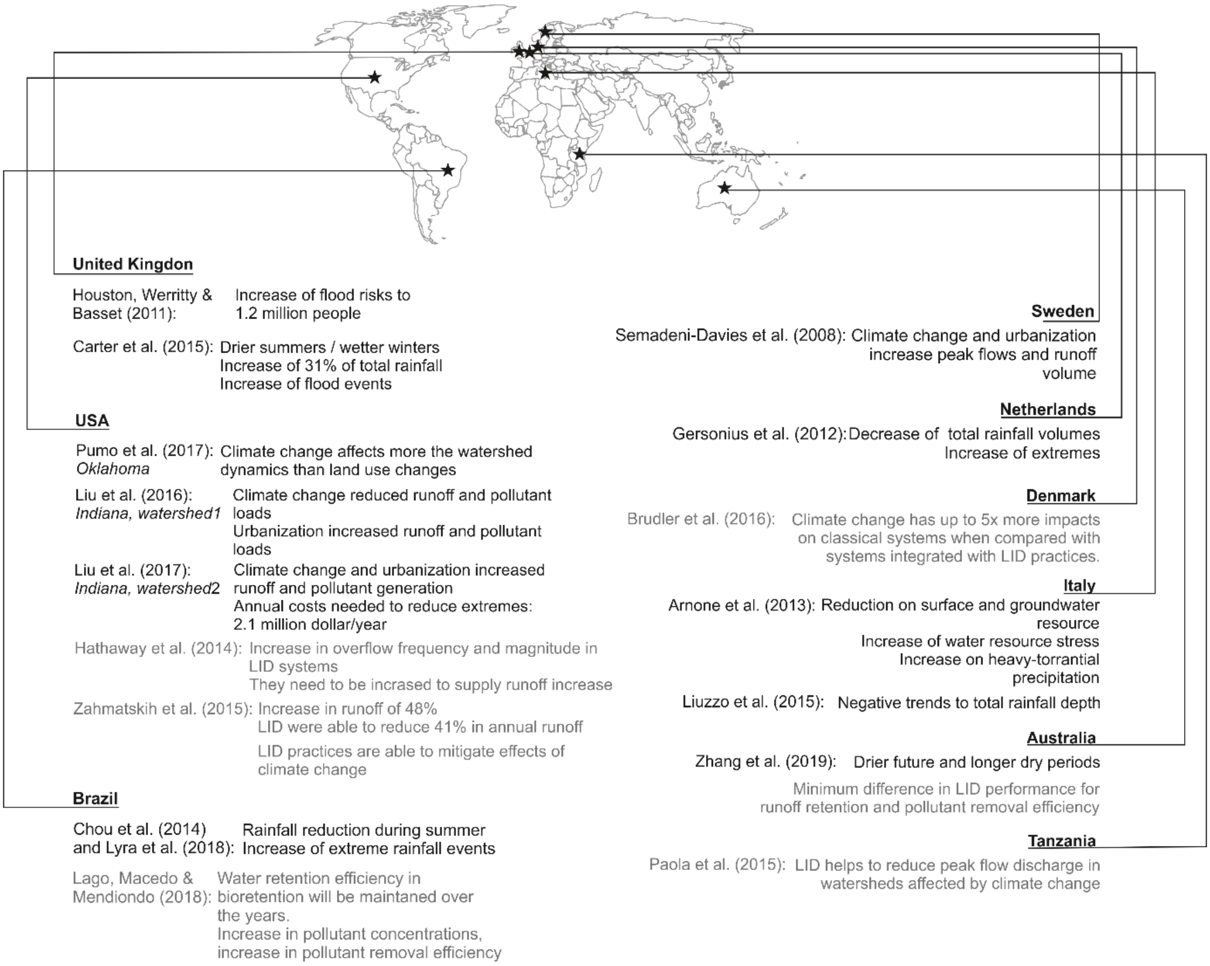

Figure 2.3 - Climate change studies over the world: Impacts on climate patterns, hydrology, and LID systems. In black: impacts of climate change in the rainfall patterns and watershed. In grey: Impacts of climate change in LID practices.

2.4.2 New perspectives for incorporating climate change in LID design

LID practices have a potential to help in the resilience of urban drainage systems and flood and pollutant control due to climate changes. However, their efficiencies are affected by the changes in rainfall patterns. Therefore, it is necessary to consider the effects of future change in urbanization and climate during the design. The studies presented have evaluated the effects of climate changes in the LID practices efficiencies but did not propose new design methods or adaptations.

Many of the guidelines and manuals for design LID practices (Waterways, 2006; The Prince George's County, 2007; COUNCIL, 2007; McAuley, 2009) recommend the use of synthetic design storms obtained using the Intensity-Duration-Frequency (IDF) curves, with different return periods. Therefore, one of the options for incorporate the non-stationarity of climate in the design of LID practices is to update the IDF curves considering the climate change scenarios projected by the IPCC, generating, then, design storms more compatible with the future scenario (Madsen et al., 2009; Soro et al., 2010; Mailhot & Duchesne, 2010). The update of IDFs for drainage structure and flood management design has been recommended in several studies (He et al., 2006; Wang et al., 2013; Madsen et al., 2014) and has already been adopted in guidelines of the New York State and Belgium (Willems, 2013; DeGaetano & Castellano, 2017).

Methods to IDF update are presented by Willems & Vrac (2011), Willems (2013), Wang Hagen & Alizad (2013) and Srivastav et al. (2014). These methods consist in performing spatial and temporal downscaling from Global Circulations Models (GCM) or Regional Climate Models (RCM), followed by bias correction. Climate change models have great uncertainties, and these should be considered in hydrological simulations and construction of new IDFs, even when performing downscaling and bias correction methods. Willems & Vrac (2011) propose that instead of quantifying statistical uncertainties it would be possible to deal with uncertainty scenarios, using various climate models, emission scenarios and applying different bias correction methods.

The implementation of LID practices considering future change scenarios requires a great initial investment, since structures tend to have larger area and volume, which can become a disadvantage of their application. In order to overcome this adversity, Rosa (2016) and Loiola et al. (2018) propose a similar idea of modular design for LID systems (bioretention and green roofs, respectively). The LID devices are designed for future scenarios, but their implementation is made through modular expansion, so that its construction (and hence costs) are distributed over the years, for a better adaptation to changes and exceeding runoff volumes and pollutant loads. The modular expansion, considering the drivers of change, should be incorporated in new design guidelines.

2.5 Contribution of LID practices in moving toward UN SDG (LID - 3G)

In addition to incorporating the timescale and drivers of change in LID projects, moving towards a more resilient society also requires a systematic and holistic view of stormwater management, integrating measures that help the whole of a balanced and fair environment. From this conception, LID practices can be used to meet SDG.

Within the systematic view from which the SDG emerges, in the natural environment it is necessary to understand that the use of natural resources and its impacts on the environment are correlated with each other and have complex relationships of exchange and interdependence. It is from this systemic view in the conception of the SDG that the *water-energy-food nexus* also emerges. This approach is based on the vision that the security of these resources and the system resilience can only be guaranteed by an integrated management, explaining all the relations and connections between the production, operation and distribution of each water, energy and food resource among each other's (Hoff, 2011). *Water-energy-food nexus* is already being used and widespread in the water and energy production sectors, with little insertion in stormwater management studies, despite its potential integration with alternative urban drainage measures.

Therefore, concerning the stormwater management, the development of resilient cities and encouraging the water recycling through alternative urban drainage systems as an urban harvesting process cooperate to increase water security at its application site. Within the *water-energy-food nexus* approach, stormwater harvesting also reduces the energy demand of supply systems by producing water near the point of consumption and the systems can be used to cycle nutrients present in the stormwater as pollutants to be used in agricultural systems. By reducing demands for energy and resource production, alternative drainage systems also have positive impacts on reducing greenhouse gas (GHG) emissions (Novotny, 2010). Moreover, in addition to the indirect contribution to GHG reduction, vegetated LIDs also have carbon sequestration capacity through the assimilation of organic matter into the filter media and vegetation growth (Kavehei et al., 2018). Therefore, a new “carbon” component can also be incorporated into the nexus (Nair et al., 2014).

For a better presentation of the review, in this section, the contribution of LID practices to the safety of each of the nexus components is presented separately, as well as their contribution to different SDG. However, it should be remembered that the nexus and the SDG are systemic approaches, where their correlations are greater than the evaluation of each separate component. In addition, LID practices may contribute to other SDG than those presented in this review.

2.5.1 Water security

Already in 2008, Fletcher et al. (2008) made a review of LID practices to harvest urban drainage water, also indicating the greatest needs so this could be done. They noted that risk management is still a gap in studies, often due to lack of relevant guidance and standards. They also found a lack of studies involving the implantation and operation costs, but found that, in general, these tend to be inversely proportional to the application scale.

Aiming to respond to this gap, Karim et al. (2015) evaluated the reliability and economic saving of using stormwater harvesting systems in Bangladesh megacity. Results indicated that about 15–25% reliability in the alternative supply system can be achieved under the wet climatic condition and for catchment sizes varying from 140 m^2 to 200 m^2, 250 m^3 to 550 m^3 of rainwater can be harvested each year. They also noted that a monetary saving of approximately 2000 BDT (Bangladesh taka, the currency for Bangladesh, ~23.6 USD in January 2021) can be obtained for a 140m^2 basin using a 40m^3 tank under average annual weather and that monetary economies increase with catchment size.

As a study to assess the impacts of runoff harvesting systems on contributing to city resilience, Burns et al. (2015) evaluated the use of domestic rainwater tanks to reduce runoff associated with the demand reduction for non-potable domestic water uses. They have realized that these systems can lead to a significant reduction of the other domestic demands for tap water. Also, Chandrasena et al. (2016) investigated the possibility of applying urban harvesting in bioretention, evaluating the pathogens removal, so that the water quality would reach values allowing its use for irrigation. Mitchel et al. (2008) have developed models to evaluate the stormwater harvesting capacity by LID practices, listing the most sensitivity design parameters and variables as length of rainfall record, inter-annual variability of seasonal demand, and storage surface type.

In terms of how rainwater harvesting impacts water supply and demand, Petit-Boix et al. (2018), in their study for cities in Spain and USA, estimated that in general cisterns were able to supply 75% of the rainwater demand for laundry and toilet flushing. In this same sense, Clark et al. (2015) modeled the demand reduction using rainwater harvesting in a city in South Australia, concluding that an annual demand equating to 12.8% of catchment rainfall could be met with 99.5% of volumetric reliability. Macedo et al. (2019) evaluated how a bioretention practice in field can contribute to increase water security during the dry period in a Brazilian city, when integrating this system with water reuse, specifically for tropical climate. The results show that reusing the total volume of stormwater stored in the bioretention has a potential to reduce the demand for tap-water in its half, during the dry season. The stormwater stored was also evaluated in terms of water quality and the results suggest that it can be reused for non-potable demands, and do not represent a contamination risk if in contact with humans. The water supply by stormwater and rainwater harvesting is highly dependent on the local climatic characteristics and consumption profile. Therefore, before the application on the catchment a specific assessment should be made to verify the actual potential and viability of the project.

One of the difficulties about implementing stormwater reuse is the lack of specific legislation establishing the limit values of water quality parameters, to be used for different reuse types (Fletcher et al., 2008). Therefore, countries should move forward in developing their own legislation, based on values already adopted elsewhere and adapting it to their environmental reality.

In Brazil, two new guidelines and standards were approved in 2019 for the management of alternative sources of water in buildings, instituted by the Brazilian Association of Technical Standards (ABNT): "NBR 16782/2019: water conservation in buildings - requirements, procedures and guidelines" and "NBR 16783/2019: use of alternative sources of non-potable water in buildings", which establish standards and norms for the conservation and efficient use of water and present techniques to reduce water demands, including reuse. The norm "NBR 15.527/2019: stormwater – coverings utilization in urban areas for non-potable purposes - requirements" was also updated, which establishes parameters for the stormwater reuse for non-potable purposes, such as in sanitary basins discharge, patios cleaning and gardens irrigation, not considering other more restrictive uses. The approval of these new guidelines and standards in Brazil shows an advance in the understanding of the necessity to incorporate resource recycling and decentralized water management in public policies.

The most developed countries regarding to the adequacy of specific legislation currently are Australia and the United States. In the case of Australia, in 2008 the Australian Guidelines for Recycling Water document was prepared, which presents a complete discussion on the principles of water recycling, including action policies for its adoption, monitoring routine and operation of the systems (NRMMC, 2008). In this document, the same values from the Australian Drinking Water Guidelines are used, but with a greater discussion on pathogens, and other chemicals such as medicines and pesticides.

In the USA, the current values of reuse water quality are given by the "2012 Guidelines for Water Reuse", produced by the Environmental Protection Agency – EPA, (EPA, 2012). One of the motivations for water reuse presented in this document is the advance of urbanization that increases water scarcity. With a guideline for water reuse, the EPA aims to meet the *water-energy nexus*, in a way to optimize the use of these two resources, ensuring their safety.

2.5.2 Food security

As for the nutrient cycling through plant production and future reuse, three works have been developed in Australia in recent years. Richards et al. (2015) evaluated the possibility of making bioremediation systems with edible vegetable production used as a form of urban agriculture. For this, they used water collected from roofs to make a sub irrigation in crops. This form of application was chosen both to reduce plant stress and to avoid direct contact with contaminants. The yields of this system were similar to those of common irrigation, also contributing to runoff volume and frequency reduction in more than 90%. Daniel Junior (2013) evaluated Brazilian typical edible plants to use as the vegetation layer of a mixed infiltration trench in a southern state in Brazil, and their viability along the entire hydrological year. The plants selected for this study were Banana tree (*Musa velutina*), Taioba (*Xanthosoma sagittifolium*), Yam (*Alocasia*), Canna paniculate (*Canna*), Turnip (*Raphanus sativus L.*). The turnip did not survive low temperatures and died. Banana tree, Taioba and Yam had the best responses, resisting up to 101 days without rainfall and surviving the entire hydrological year.

Following this perspective of use, Ng et al. (2018) studied whether the plants used in conjunction with biofiltration/bioretention systems would be undergoing metals accumulation. As a result, they observed that there was an accumulation in the edible parts larger than the standards recommended by the World Health Organization (WHO). Therefore, they see a perspective on the possibility of retaining these metals in the soil, so that they are not transferred to the plants. Tom, Fletcher & McCarthy (2014) also conducted a study evaluating the contamination by metals in plants irrigated with runoff water, obtaining similar results to those of Ng et al. (2018).

However, nutrients can be recycled in addition to food production. Based on the analyses of accumulated nutrient removal in vegetation over time, Ge et al. (2016) proposed the strategy of harvesting the tissues of the plants used in the systems, considering the period that usually dies more.

2.5.3 Energy security

Regarding integration with energy security, Ramos et al. (2013) analyzed how flood drainage systems (more specifically retention ponds) can be used as water storage volumes to damp floods and simultaneously produce energy, constituting innovative solutions to be integrated in future smart water grid's designs. In the analyzed solutions, the higher the water level in the pond for the same volume the better the production of energy, due to the higher available for the same turbine discharge. Also, Hashemi et al. (2015) noted that green roofs decrease energy consumption by saving on heating/cooling, also collaborating to increase energy security. Studies of life cycle assessment should be done to investigate the impacts of food production near the consumption and nutrient cycling, reducing the demands for artificial fertilizers, in the reduction of energy consumption.

2.5.4 Carbon sequestration and storage

Due to the presence of a vegetation layer in different types of LID practices (e.g. bioretention, green roof, wetlands etc.) and their potential to reduce energy demands, LID practices can also be exploited in their ability to carbon sequester and storage as a mean of mitigating GHG emissions and climate action (Novotny, 2010; Nair et al., 2014).

Kavehei et al. (2018) have made a systematic review of studies with carbon sequestration and LID practices. According to this review, they were able to quantify the carbon footprint related to the life cycle of different types of LID practices and the carbon sequestration during the lifetime. They observed that the main contribution for the carbon footprint of this systems is associated to the implementation phase. Also, the vegetated systems have more potential on amortizing the carbon footprint during the lifetime, e.g. bioretention basins were able to mitigate approximately 70% of carbon emissions, while stormwater ponds only mitigate 8%.

The study by Getter et al. (2009) evaluated the carbon sequestration capacity in green roofs. In addition to incorporating carbon for vegetation growth, this study also explored carbon sequestration in the substrate by incorporating plant litter into the soil. The authors evaluated 12 green roofs vegetated with *sedum species*. As a result, green roofs stored in the range of 64 to 239 gC/m^2 in aboveground biomass (plant tissue) and 37 to 185 gC/m^2 in belowground biomass (plant litter).

Bouchard et al. (2013) and Moore & Hunt (2012) have used a similar methodology of quantifying the accumulation of carbon in the soil to evaluate the carbon sequestration in roadside vegetated filter strips, swales, constructed wetlands and ponds. As a result, the vegetated systems (filter strips and constructed wetlands) had more capacity in accumulate carbon in soil. However, the authors state that the interpretation of the results is limited by the lack of long-term data and the fluxes of carbon in inflow and outflow. Additionally, this methodology does not account with the GHG emissions in the systems.

More recently, D'Acunha & Johnson (2019) have evaluated the water quality through dissolved organic carbon (DOC) and NO_3 and GHG fluxes (carbon, methane and nitrous oxide) for a constructed wetland. They concluded that the outflow of the evaluated system still contained high values of DOC (latter decomposed and transformed in carbon emissions) and the water was supersaturated with carbon and methane, leading to evasion fluxes. These emissions must be considered in the studies of LID practices contributions on carbon sequestration and climate action.

Therefore, to better study the carbon sequestration and storage potentials (processes here called decarbonization) of LID practices, it is necessary to develop clearer methodologies to identify the carbon flux from the atmosphere to vegetation and soil, and from organic carbon to vegetation, soil and atmosphere. Already in 1999, Schlesinger (1999) stated that the carbon cycle in soils is the least well known of all the carbon cycles. In addition, watershed life cycle analysis methodologies, proposed as a challenge to quantify the gains of energy expenditure brought by LIDs, can also be incorporated with carbon emissions to identify indirect reductions.

In addition to the direct effects on carbon sequestration, Pataki et al. (2006) noted the indirect effects that the implementation of LID practices may have in reducing GHG emissions. For example, green roofs increase thermal comfort and, thus, reduce energy costs with cooling and heating. These energy costs can be supplied from fossil fuels, reducing carbon emissions for electricity generation. However, many of these calculations were made from models with untested assumptions regarding urban vegetation and surface process and should be further explored.

Nair et al. (2014) and Novotny (2010) have discussed the incorporation of GHG in the *water-energy nexus*, mainly when approaching decentralized urban water systems, such as LID practices for water reuse. Not only in reducing the energies demands by secondary benefits as cooling and heating (as presented by Pataki et al., 2006), the decentralized water systems also reduce the energy demands for distribution, and this should be considered in terms of costs and GHG emissions, when talking about climate action.

2.5.5 Comparative analysis of LID practices to different SDG

Currently, there are several types of LID practices employed in sustainable urban drainage, according to the different mitigation purposes aimed by the decision makers and physical limitations of the catchment (Jia et al., 2013; Pour et al., 2020). These practices can be divided into vegetated (bioretention systems, rain garden, green roof) or non-vegetated (porous pavement, sand filter, detention ponds), infiltration-based (swales, infiltration trenchs, sand filter, rain garden) and retention-based (green roofs, detention ponds, rain barrel) (Erickson, Weiss & Gulliver, 2013; Eckart, McPhee & Bolisetti, 2017). Due to the different mechanisms employed by each of them and the main benefits provided, they have different levels of contribution to the multiple SDGs.

An overview on the most commonly used LID practices and their limitations in terms of spatial application and their contribution to the different SDG is presented in Table 2.2. The contributions to each SDG were classified as low to high according to the characteristics of each technique in terms of treatment mechanisms for runoff and water quality, main mitigation purposes and scale. E.g. vegetated techniques have potential for carbon sequestration and nutrient cycling, therefore with medium to high potential in contribute to climate action and zero hunger; techniques with water storage per se or that can be coupled to reservoirs have the ability to reuse water, and therefore medium to high contribution to clean water and sanitation; techniques that have greater treatment capacity can contribute to clean water and sanitation and good health and well being; techniques capable of runoff retention and/or detention contribute to reduce flood events and therefore can contribute medium to high for climate action and sustainable cities and communities; techniques with the possibility of landscape integration contribute medium to high for good health and well being; techniques with the production of local resources or that assist in thermal comfort contribute to reduce energy demands, and can be classified as medium to high for affordable and clean energy. In addition, the potential for use already explored in the studies reviewed in this paper was considered in the classification.

Table 2.2 – Description of commonly used LID practices, limitations to their spatial application and benchmark selection to multiple SDG

LID practice	Description	Limitations	Contribution to SDGs					
			SDG 2 *Zero hunger*	**SDG 3** *Good health and well-being*	**SDG 6** *Clean water and sanitation*	**SDG 7** *Affordable and clean energy*	**SDG 11** *Sustainable cities and communities*	**SDG 13** *Climate action*
Green(blue) roof	Roofs covered with a vegetated layer	Small to medium catchment areas and small to medium storms. Regular inspection	Medium	High	High	High	High	Medium to high
Porous pavement	Permeable surface used in roads and pathways that allow subinfiltration	Strongly dependent on hydraulic conductivity, soil infiltration and slope. Small to medium storms	Low	Medium	Medium	Low	Medium	Low
Bioretention / Rain gardens	Vegetated concave filled with a filtering media designed to store, infiltrate and treat stormwater	Strongly dependent on hydraulic conductivity and slope. Small to medium storms. Regular inspection	High	High	High	Medium to high	High	Medium to high
Sand filter	Concave divided in two layers, one of sand and one of gravel to allow infiltration and runoff treatment. They can be vegetated or not.	Strongly dependent on hydraulic conductivity and slope. Small to medium storms	Low	Medium	Medium	Low	Medium	Low
Constructed wetlands	An artificial wetland to treat stormwater	Require soils with low infiltration rate. Annual maintenance	Low	High	Medium to high	Low	Medium to high	Medium to high
Infiltration trenchs	Chanel made of gravel to allow storage and infiltration and can be covered by soil and vegetation	Strongly dependent on hydraulic conductivity and slope. Small to medium storms. Regular cuttings if vegetated.	Low	Medium to high	Medium	Low	Medium	Low to medium
Stormwater detention ponds	An artificial depression in the soil to store stormwater/runoff for a longer period	Does not allow infiltration and can increase disease dissemination	Low	Medium	Low to medium	Low to medium	Medium to high	Low
Rain barrel / Rainwater tank	Surface tanks to store rainwater from rooftops	Small to medium catchment areas and small to medium storms. Does not allow infiltration	Low to medium	Medium to high	Medium to high	Medium to high	Medium to high	Medium
Swales	Shallow open channels grassed of vegetated with mild side slopes and flat bottom	Strongly dependent on slope. Small to medium storms. Regular cuttings.	Low	Medium	Medium	Low	Medium	Low to medium

2.5.6 Future perspectives

Despite all the studies in how the LID practices can be used to stormwater reuse, cycling nutrients, producing and saving energy, and contribution to carbon sequestration (LID-3G purposes), it still lacks integration and knowledge on how to address all the purposes together. In Table 2.3 it is presented the lessons learned in previous studies and the challenges that remain to LID-3G development. Clear metrics to quantify the fluxes of resources in the LID practices still needs to be stated, allowing the evaluation of their contribution to the SDG and the impact on the resilience of the application site.

Also, increasing the application of LID practices focusing on 3rd generation purposes requires clear guidance of how to incorporate these aspects into design and sizing of this practices. Despite assessing the use of LID for this purpose, the studies presented did not present sizing guidelines that incorporate nutrient cycling, energy production or reduced energy expenditure and carbon sequestration and storage in the catchment. Thus, further studies should focus on responding to these gaps.

Table 2.3 - How LID practices can contribute to UN SDG: suggestions and challenges

Contribution to UN SDG	How LID contribute to this UN SDG?	Suggestions in LID design	Spatial scale	Time scale	Challenges	Main references
3. Good health and well-being	Runoff volume retention	Design storms for higher return periods	Catchment	Short-term	Need to incorporate changes in urbanization and climate in design guides and manuals; quantification of maintenance costs	Davis (2008); Winston, Dorsey & Hunt (2016)
	Peak flow reduction			Short-term		
	Decrease flood risks			Mid-term		
	Pollutant removal from runoff	Anaerobic zone to denitrification (increase nutrient removal)	Catchment	Short-term	Results in nutrient removal still present great ranges variation; micropollutants and pathogens removal studies are still incipients	Davis (2007); Hatt, Fletcher & Deletic (2009); Glaister et al. (2016)
	Decrease risks of urban rivers contamination			Mid-term		
	Stormwater harvesting and water reuse	Underdrain to collect treated water and storage tanks to future reuse	Individual/ Catchment	Short and mid-term	Lack of standards to water reuse; estimate of operation costs; need of additional treatment (e.g. pathogens removal)	Fletcher et al. (2008); Karim, Bashar & Imteaz (2015); Chandrasena, Deletic & McCarthy (2016)
	Decrease risks of water scarcity			Mid-term		
6. Clean water and sanitation	LID systems allow runoff treatment and less pollutant loads in urban rivers *(one of the main causes of urban river*	Anaerobic zone to denitrification (increase nutrient removal)	City	Mid-term	Results in nutrient removal still present great ranges variation; micropollutants and pathogens removal studies are still incipients	Davis (2007); Hatt, Fletcher & Deletic (2009); Glaister et al. (2016)
	Groundwater recharge	Permeable walls and bottom to allow exfiltration	Catchment	Mid-term	Avoid groundwater contamination	
7. Affordable and clean	Energy saving by cooling and heating	Green roofs and walls	Individual	Mid-term	Need of methodology to quantification	Ramos et al. (2013)
	Energy production	Integration of turbines with retention ponds. Allow higher water level (available head to turbines)	Catchment	Short-term	Develpment of new technologies to other LID practices types; integration with city energy grid	Hashemi, Mahmud & Ashraf (2015)
	Recycling resources (construction materials, biomass as fertilizers)		City	Long-term	Lack of guidelines and standards to resource recycling	
	Reduction of energy expenditures with water and food transportation	Implementation close to housing	City	Long-term	Need of methodology to quantification	
11. Sustainable cities and communities	Local water, energy and food production	Implementation close to housing	Individual/ Catchment	Mid-term		
	Decrease water-energy-food insecurity	For food production: Vegetable practices (e.g. bioretention) and soil with high capacity of metal sorption	Catchment	Mid-term	For food production: metals and other contaminants in plant tissues;	Richards et al. (2015); Ng et al. (2018); Ge et al. (2016)
	Urban resilience to rainfall and drought extremes under changing scenarios of		City	Mid and long-term	New metrics to system performance linked with UN SDG	Simonovic and Peck (2013)
13. Climate action	Decarbonization	Vegetable practices	Not possible to restrict scale (gas emission)	Long-term	Need of methodology to quantification	Kavehei et al. (2018)

In this paper it was explored the possibility of integrating LID practices with the UN SDG by using the *water-energy-food-GHG nexus* approach. However, the application of LID-3G is not restricted to water reuse, resource recycling and carbon storage, and new ways of integrating with UN SDG should be studied.

2.6 Evaluation of LID practices contribution to resilience of cities and communities

A way to evaluate LID systems is from their contribution to increase on-site resilience. The concept of resilience is linked to the ability of a system, population or society to return to initial conditions prior to a disturbance (Meerow et al., 2016). The effort in this area has been to develop ways of measuring the resilience level of a system. Several authors have proposed static resilience index as a form of quantification (Hashimoto et al., 1982; Kjeldsen & Rosbjerg, 2004). Simonovic & Peck (2013) and Simonovic (2016) criticize the time-independence static resilience measure because it is an abstract attribute, which does not describe the behavior and state of the system after stress, being inefficient for planning actions. Therefore, Simonovic & Peck (2013) propose a space-time dynamic resilience measure (STDRM), based on the concepts of system performance level and adaptive capacity, over time.

The disturbance events can generate different impacts in the systems (e.g. physical, social, economic, health, among others), and these impacts not always has the same unit. Therefore, the system performance needs to be measured for each impact in the correspondent impact unit. The dynamic resilience is then presented as uniform unit measure representing the loss of system performance, i.e. graphically represented as the area under the system performance level between the beginning of the disturbance and the end of the system recovery (Figure 2.4a and Eq. 2.1) and is also variant in time and space. For resilience to be presented in a uniform unit for different impacts, the loss of system performance is normalized dividing it by maximum performance (Eq. 2.2, Simonovic & Pack, 2013). The integrated spatial-time dynamic resilience over all impacts is calculated according to Eq. 2.3 (Simonovic & Pack, 2013).

$$\rho^i(t,s) = \int_{t_o}^{t} \left[P_0^i - P^i(t,s)\right] dt\,, \quad where\ t \in [t_0, t_r] \tag{2.1}$$

$$r^i(t,s) = 1 - \left(\frac{\rho^i(t,s)}{P_0^i \cdot (t-t_0)}\right) \tag{2.2}$$

$$R(t,s) = \left\{\prod_{i=1}^{M} r^i(t,s)\right\}^{\frac{1}{M}} \quad (2.3)$$

where: M is the total number of impacts; P_0^i is the maximum system performance level for impact i (at time t_0); $P^i(t, s)$ is the system performance level, at time t and space s; $r^i(t, s)$ is the resilience to impact i, at time t and space s; R(t, s) is the integrated system resilience, in time t and space s; s is the space variable; t is the time variable; t_0 is the disturbing initial time; t_r is the end of the recovery time; $\rho^i(t, s)$ is the loss of system performance to impact i, at time t and space s. Equations obtained from Simonovic and Peck (2013).

Therefore, the first step in quantifying the dynamic resilience is to determine how much the system performs over time. For 3rd generation objectives, e.g. integration with watershed sustainability and *water-energy-food nexus* safety, in this study, Eq. 2.4 and 2.5 are proposed as performance evaluation measures of LID practices over time. For the assessment of LID-3G, focusing on water-energy-food security, disturbances are considered as flood events and drought periods (Figure 2.4b). As proposed by Simonovic & Arunkumar (2016), the resilience for each system performance curve can be integrated into a single curve to quantify the overall system resilience (Figure 2.4b).

$$System\ performance_{runoff\ retention} = \frac{V_{in}(t) - V_{out}(t)}{V_{in}(t)} \quad (2.4)$$

$$System\ performance_{water\ reuse} = 1 - \left(\frac{WC(t) - \pi_{rec,W:E:F}(t)}{WC(t)}\right) \quad (2.5)$$

where: Where: Vin is the total runoff volume that enters the LID practice, V_{out} is the total water volume that exits the systems and return to the catchment as runoff (directly to rivers or to the conventional drainage systems), WC is the water consumption per household; $\pi_{rec,W:E:F}$ is the volume of water recovered/stored to future reuse for the water-energy-food security. All variables are time dependent.

New equations should be proposed to meet the requirements of each location and expand for evaluation of other forms of resource cycling, carbon storage, and meeting different UN SDG. Performance curves are then constructed by continuous simulation. To consider aspects of urbanization changes and climate change (1G and 2G), simulations can be made by varying runoff coefficient values and future rainfall data from GCMs or RCMs.

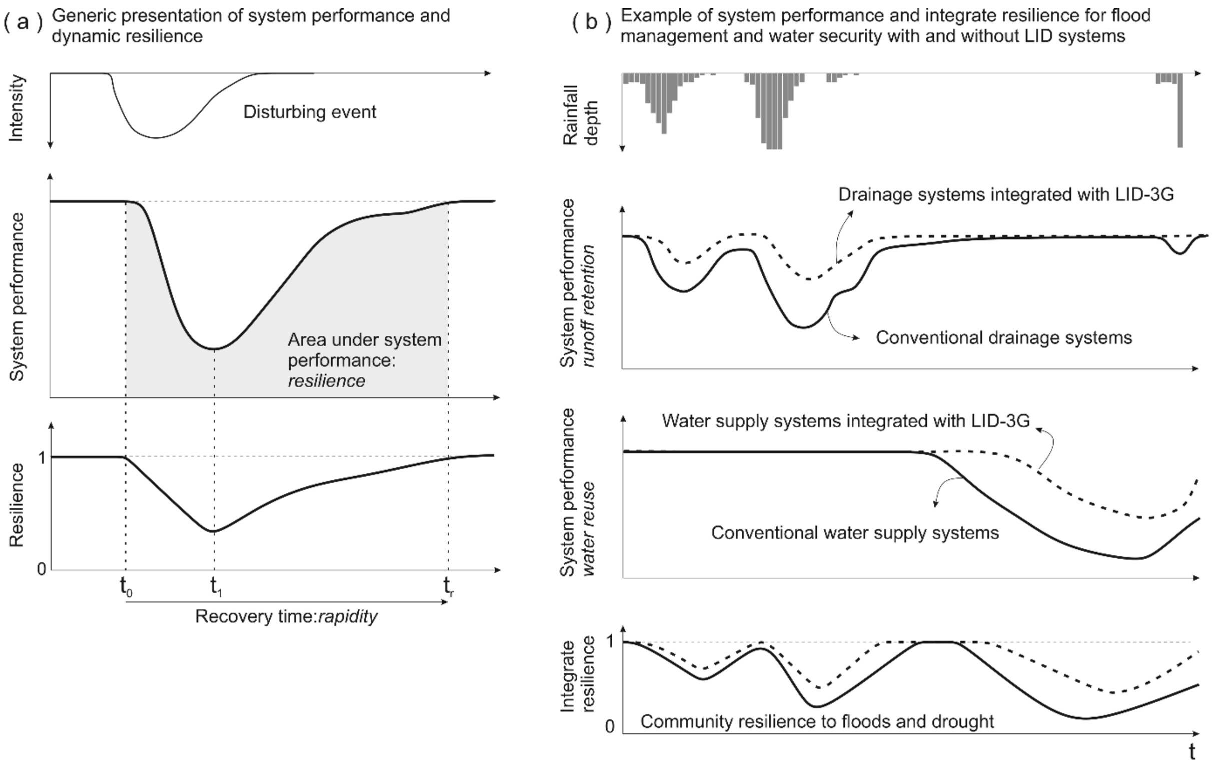

Figure 2.4 - (a) Generic presentation of a system performance and resilience under an extreme rainfall event (disturbing event). Adapted from Simonovic & Peck (2013), (b) Generic presentation of an urban drainage and water supply system performance with and without integration with LID practices and the respective integrate resilience.

2.7 Conclusion

The characterization of LIDs in generations aims to show how their benefits and complexity evolves according to their purposes. For purposes of only runoff control and water quality improvement in the current scenario of urbanization, they are called LID-1G. When considering changes in land use and non-stationary effects of climate change to future planning, they are called LID-2G. Finally, these practices can incorporate the aspects of the *water-energy-food-GHG nexus* and increase of urban resilience to meet the UN SDG in the catchment, and are called LID-3G. Grouping them according to these characteristics will help readers to identify the level of difficulty to design the LID, its sophistication, and degree of urban resilience. In addition, this new proposed terminology can also be used as an advertisement to promote design of higher generation LIDs.

Several researches have already been developed observing these potentialities of the LID practices. Here we present the main lessons and challenges that remain to move toward the integration of LID practices with the UN SDG:

- Future scenarios of urbanization and climate change need to be considered in the planning stage of implementation of a LID practices. Studies showed increases up to 20% of runoff generation due to the increase of urbanization and climate change, together.

- Changes in runoff separation coefficients and update IDF curves with climate change predictions are suggestions of how to include future scenarios on design guidelines. However, Global and regional climate models, and hydrological models, present numerous uncertainties that need to be quantified and included in decision-making analysis of public managers.

- New designs that allow an optimization of urban harvesting in LID practices (integrating runoff reuse with nutrient cycling, and energy production and saving) are still incipient. There is still a need of clear methodology to state the co-relations between these resources among each other and the catchment area.

- Methodologies for quantifying soil, vegetation and atmosphere carbon fluxes and life cycle analysis studies in the watershed, considering GHG emissions, need to be established to study the capacity of LID practices in decarbonization.

- Metrics well-stablished to identify and quantify clearly the contribution of LID systems to the increase of urban resilience and the achievement of UN SDG.

References

Ambrizzi, T., & Magaña, V. (2016). Mudanças Climáticas e Projeções Futuras de Eventos Extremos. [Climate change and future projections of extreme events]. In *Desastres - Múltiplas Abordagens e Desafios*. 1st edition. Elsevier. (in Portuguese)

Arnone, E., Pumo, D., Viola, F., Noto, L., & La Loggia, G. (2013). Rainfall statistics changes in Sicily. *Hydrology and Earth System Sciences*, 17(7), 2449 - 2458.

Bouchard, N., Osmond, D., Winston, R., & Hunt, W. (2013). The capacity of roadside vegetated filter strips and swales to sequester carbon. *Ecological Engineering*, 54, 227-232.

Bratieres, K., Fletcher, T., Deletic, A., Alcazar, L., Coustumer, L., & McCarthy, D. (2008). Removal of nutrients, heavy metals and pathogens by stormwater biofilters. *Proceeding of 11th International Conference on Urban Drainage*. Porto Alegre, Brazil.

Brudler, S., Ambierg-Nielsen, K., Hauschlid, M., & Rygaard, M. (2016). Life cycle assessment of stormwater management in the context of climate change adaptation. *Water research*, 106, 394-404.

Burns, S., Arnbjerg-Nielsen, K., Hauschild, M., & Rygaard, M. (2015). The performance of rainwater tanks for stormwater retention and water supply at the household scale: An empirical study. *Hydrological Processes*, 29(1), 152-160.

Carter, J., Cavan, G., Connelly, A., Guy, S., Handley, J., & Kazmierczak, A. (2015). Climate change and the city: building capacity for urban adaptation. *Progress in Planning*, 95, 1-66.

Chahal, M., Shi, Z., & Flury, M. (2016). Nutrient leaching and copper speciation in compost-amended bioretention systems. *Science of the Total Environment*, 556, 302 - 309.

Chandrasena, G., Deletic, A., & McCarthy, D. (2016). Biofiltration for stormwater harvesting: comparison of Campylobacter spp. and Escherichia coli removal under normal and challenging operational conditions. *Journal of Hydrology*, 537, 248-259.

Chou, S., Lyra, A., Mourao, C., Derecynski, C., Pilotto, I., Gomes, J., . . . Campos, D. (2014). Assessment of climate change over South America under RCP 4.5 and 8.5 downscaling scenarios. *American Journal of Climate Change*, 3(5), 5121 - 525.

Clark, R., Gonzalez, D., Dillon, O., Charles, S., Cresswell, D., & Naumann, B. (2015). Reliability of water supply from stormwater harvesting and managed aquifer recharge with a brackish aquifer in an urbanising catchment and changing climate. *Environmental Modelling & Software*, 72, 117-125.

Council, G. C. (2007). *Water Sensitive Urban Design Guidelines*. Gold Coast City Council, Queensland.

D'Acunha, B., & Johnson, M. (2019). Water quality and greenhouse gas fluxes for stormwater detained in a constructed wetland. *Journal of Environmental Management*, 231, 1232-1240.

Daniel Júnior, J. J. (2013). Avaliação de uma Biorretenção como estrutura sutentável de drenagem urbana. [Evaluation of Bioretention as a sustainable urban drainage structure]. Thesis (Master) Civil Engineering Department, Universidade Federal de Santa Maria. (In Portuguese)

Davis, A. (2008). Field performance of bioretention: Hydrology impacts. *Journal of Hydrologic Engineering*, 13(2), 90-95.

Davis, A. P. (2007). Field performance of bioretention: Water quality. *Environmental Engineering Science*, 24(8), 1048-1064.

DeGaetano, A., & Castellano, C. (2017). Future projections of extreme precipitation intensity-duration-frequency curves for climate adaptation planning in New York State. *Climate Services*, 5, 23-35.

Douglas, I., Garvin, S., Lawson, N., Richards, J., Tippett, J., & White, I. (2010). Urban pluvial flooding: a qualitative case study of cause, effect and nonstructural mitigation. *Journal of Flood Risk Management*, 3(2), 112 - 125.

Dudula, J., & Randhir, T. (2016). Modeling the influence of climate change on watershed systems: adaptation through targeted practices. *Journal of Hydrology*, 541, 703-713.

Eckart, K., McPhee, Z., & Bolisetti, T. (2017). Performance and implementation of low impact development–a review. *Science of the Total Environment*, 607, 413-432.

EPA. (2012). *2012 Guidelines for Water Reuse*. Washington, D. C.: U. S. Environmental Protection Agency, Office of Wastewater Management, Office of Water.

EPA, E. P. (1983). *Results of the National wide urban runoff program: Volume 1 – Final Report*. Washington, DC.: Water Planning Division, U.S. EPA.

Erickson, A., Weiss, P., & Gulliver, J. (2013). *Optimizing stormwater treatment practices: a handbook of assessment and maintenance*. New York: Springer.

Ferreira, L. T. L. M. (2017). Avaliação do funcionamento de um dispositivo de biorretenção: monitoramento e modelagem. [Evaluation of the functioning of a bio-retention device: monitoring and modeling]. Thesis (Master). Department of Water Resources and Sanitation. Universidade Federal de Alagoas. (In Potuguese).

Fletcher, T., Deletic, A., Mitchell, V., & Hatt, B. (2008). Reuse of urban runoff in Australia: a review of recent advances and remaining challenges. *Journal of Environmental Quality*, 37(5), s-116 - s127.

Fletcher, T., Andrieu, H., & Hamel, P. (2013). Understanding, management and modelling of urban hydrology and its consequences for receiving waters: A state of the art. *Advances in Water Resources*, 51, 261-279.

Fletcher, T., Shuster, W., Hunt, W., Ashley, R., Butler, D., Arthur, S., . . . Mikkelsen, P. (2015). SUDS, LID, BMPs, WSUD and more – The evolution and application of terminology surrounding urban drainage. *Urban Water Journal*, 12(7), 525-542.

Ge, Z., Feng, C., Wang, X., & Zhang, J. (2016). Seasonal applicability of three vegetation constructed floating treatment wetlands for nutrient removal and harvesting strategy in urban stormwater retention ponds. *International Biodeterioration & Biodegradation*, 112, 80-87.

Gersonius, B., Nasruddin, F., Ashley, R., Jeuken, A., Pathirana, A., & Zevenbergen, C. (2012). Developing the evidence base for mainstreaming adaptation of stormwater systems to climate change. *Water research*, 46(20), 6824 - 6835.

Getter, K., Rowe, D., Robertson, G., Cregg, B., & Andresen, J. (2009). Carbon sequestration potential of extensive green roofs. *Environmental Science & Technology*, 43(19), 7564-7570.

Glaister, B., Fletcher, T., Cook, P., & Hatt, B. (2016). Intra-event nutrient removal dynamics in stormwater biofilters: the influence of system design. Traitement de la pollution/Pollution treatment. *Proceeding of 9th Novatech*. Lyon, France.

Guan, M., Sillanpaa, N., & Koivusalo, H. (2015). Modelling and assessment of hydrological changes in a developing urban catchment. *Hydrological Process*, 29(13), 2880-2894.

Hashemi, S., Mahmud, H., & Ashraf, M. (2015). Performance of green roofs with respect to water quality and reduction of energy consumption in tropics: a review. *Renewable and Sustainable Energy Reviews*, 52, 669-679.

Hashimoto, T., Stedinger, J., & Loucks, D. (1982). Reliability, resiliency, and vulnerability criteria for water resources system performance evaluation. *Water Resources Research*, 18, 14-20.

Hathaway, J., Brown, R., Fu, J., & Hunt, W. (2014). Bioretention function under climate change scenarios in North Carolina, USA. *Journal of Hydrology*, 519, 503-511.

Hatt, B., Fletcher, T., & Deletic, A. (2009). Hydrologic and pollutant removal performance of stormwater biofiltration systems at the field scale. *Journal of Hydrology*, 365(3), 310 - 321.

He, J., Valeo, C., & Bouchart, F. (2006). Enhancing urban infrastructure investment planning practices for a changing climate. *Water Science and Technology*, 53(10), 13-20.

Hoff, H. (2011). Understanding the nexus: background paper for the Bonn2011. *Nexus Conference*.

Houston, D., Werritty, A., & Bassett, D. (2011). *Pluvial (rain-related) Flooding in Urban Areas: the Invisible Hazard*. Joseph Rowntree Foundation.

Jing, X., Zhang, S., Zhang, J., Wang, Y., & Wang, Y. (2017). Assessing efficiency and economic viability of rainwater harvesting systems for meeting non-potable water demands in four climatic zones of China. *Resources, Conservation and Recycling*, 126, 74-85.

Jun, X., Yong, Z., LiHua, X., Shan, H., LongFeng, W., & ZhongBo, Y. (2017). Opportunities and challenges of the Sponge City construction related to urban water issues in China. *Science China Earth Sciences*, 60(4), 652.

Karim, M., Bashar, M., & Imteaz, M. (2015). Reliability and economic analysis of urban rainwater harvesting in a megacity in Bangladesh. *Resources, Conservation and Recycling*, 104, 61-67.

Kavehei, E., Jenkins, G., Adame, M., & Lemckert, C. (2018). Carbon sequestration potential for mitigating the carbon footprint of green stormwater infrastructure. *Renewable and Sustainable Energy Reviews*, 94, 1179-1191.

Kirchhoff, C. J., Lara-Valencia, F., Brugger, J., Mussetta, P., & Pineda-Pablos, N. (2016). Towards joint consideration of adaptive capacity and water security: lessons from the arid Americas. *Current Opinion in Environmental Sustainability*, 21, 22-28.

Kjeldsen, T., & Rosbjerg, D. (2004). Choice of reliability, resilience and vulnerability estimators for risk assessment of water resources systems. *Hydrologic Sciences Journal*, 49(5), 755-797.

Konrad, C., & Booth, D. (2005). Hydrologic changes in urban streams and their cological significance. *American Fisheries Society Simposium*, 47, 157 - 177.

Lago, C., Macedo, M., & Mendiondo, E. (2018). Modelling climate changes impacts in a subtropical urban drainage. *Proceedings 11th International Conference on Urban Drainage Modelling*. Palermo, Italy.

Leopold, L. (1968). *Hydrology for urban land planning: A guidebook on the hydrological effects of urban land use*. Circ. No. 554. Washington, DC: Geological Survey.

Litern, A., Daly, E., Duncan, H., Hatt, B., Fletcher, T., & Deletic, A. (2011). Key design characteristics that influence the performance of stormwater biofilters. *Proceeding 12th International Conference on Urban Drainage*. Porto Alegre, Brazil.

Liu, J., Sample, D., Owen, J., & Evanylo, G. (2014). Assessment of selected bioretention blends for nutrient retention using mesocosm experiments. *Journal of Environmental Quality*, 43(5), 1754 - 1763.

Liu, Y., Engel, B., Collingsworth, P., & Pijanowski, B. (2017). Optimal implementation of green infrastructure practices to minimize influences of land use change and climate change on hydrology and water quality: case study in Spy Run Creek watershed, Indiana. *Science of the Total Environment*, 601, 1400 - 1411.

Liu, Y., Theller, L., Pijanowski, B., & Engel, B. (2016). Optimal selection and placement of green infrastructure to reduce impacts of land use change and climate change on hydrology and water quality: An application to the Trail Creek Watershed, Indiana. *Science of the Total Environment*, 553, 149-163.

Liuzzo, L., Noto, L., Arnone, E., Caracciolo, D., & La Loggia, G. (2015). Modifications in water resources availability under climate changes: a case study in a Sicilian Basin. *Water Resources Management*, 29(4), 1117-1135.

Loiola, C., Mary, W., & da Silva, L. (2018). Hydrological performance of modular-tray green roof systems for increasing the resilience of mega-cities to climate change. *Journal of Hydrology*, 573, 1057-1066.

Lucas, W. C., & Sample, D. J. (2015). Reducing combined sewer overflows by using outlet controls for Green Stormwater Infrastructure: Case study in Richmond, Virginia. *Journal of Hydrology*, 520, 473 - 488.

Lucke, T., & Nichols, P. (2015). The pollution removal and stormwater reduction performance of street-side bioretention basins after ten years in operation. *Science of the Total Environment*, 536, 784 - 792.

Lyra, A., Tavares, P., Chou, S., Sueiro, G., Dereczynski, C., Sondermann, M., . . . Giarolla, A. (2018). Climate change projections over three metropolitan regions in Southeast Brazil using the non-hydrostatic Eta regional climate model at 5-km resolution. *Theoretical and Applied Climatology*, 132(1-2), 663 - 682.

Macedo, M., Lago, C., & Mendiondo, E. (2019). Stormwater volume reduction and water quality improvement by bioretention: Potentials and challenges for water security in a subtropical catchment. *Science of the Total Environment*, 647, 923-931.

Macedo, M., Lago, C., Mendiondo, E., & Giacomoni, M. (2019a). Bioretention performance under different rainfall regimes in subtropical. *Journal of Environmental Management*, 248, 109266.

Macedo, M., Lago, C., Rosa, A., & Mendiondo, E. (2017). Técnicas compensatórias de bioretenção para cidades resilientes: integração com nexus–água, energia, alimento. [Low Impact Development practices of bioretention to resilient cities: integration with the water-energy-food nexus]. *Proceedings of XXII Simposio Brasileiro de Recursos Hidricos*. Florianopolis, Brazil. (in Portuguese).

Madsen, H., Arnbjerg-Nielsen, K., & Mikkelsen, P. (2009). Updated of regional intensity–duration–frequency curves in Denmark: tendency towards increased storm intensities. *Atmospheric Research*, 92(3), 343 - 349.

Madsen, H., Lawrence, D., Lang, M., Martinkova, M., & Kjeldsen, T. (2014). Review of trend analysis and climate change projections of extreme precipitation and floods in Europe. *Journal of Hydrology*, 519, 3634 - 3650.

Mailhot, A., & Duchesne, S. (2009). Design criteria of urban drainage infrastructures under climate change. *Journal of Water Resources Planning and Management*, 136(2), 201-208.

Mangangka, I., Liu, A., Egodawatta, P., & Goonetilleke, A. (2015). Performance characterization of a stormwater treatment bioretention basin. *Journal of Environment Management*, 150, 173 - 178.

McAuley, A. (2009). *Technical Design Guidelines*. Nothern Territory Department of Planning and Infrastructure.

Meerow, S., Newell, J., & Stults, M. (2016). Defining urban resilience: A review. *Landscape and Urban Planning*, 147, 38-49.

Mitchell, V. G., McCarthy, D. T., Deletic, A., & Fletcher, T. D. (2008). Urban stormwater harvesting–sensitivity of a storage behavior model. *Environmental Modelling & Software*, 23(6), 782-793.

Mohor, G., & Mendiondo, E. (2017). Economic indicators of hydrologic drought insurance under water demand and climate change scenarios in a Brazilian context. *Ecological Economics*, 140, 66-78.

Moore, T., & Hunt, W. (2012). Ecosystem service provision by stormwater wetlands and ponds–A means for evaluation? *Water Research*, 46(20), 6811-6823.

Moura, N. C. B. de. (2013). Biorretenção - Tecnologia ambiental urbana para manejo das águas de chuva [Bioretention - Urban environmental technology for rainwater management]. Thesis. Faculdade de Arquitetura e Urbanismo da Universidade de São Paulo. (In Potuguese).

Nair, S., George, B., Malano, H., Arora, M., & Nawarathna, B. (2014). Water–energy–greenhouse gas nexus of urban water systems: Review of concepts, state-of-art and methods. *Resources, Conservation and Recycling*, 89, 1-10.

Ng, K., Herrero, P., Hatt, B., Farrelly, M., & McCarthy, D. (2018). Biofilters for urban agriculture: Metal uptake of vegetables irrigated with stormwater. *Ecological Engineering*, 122, 177-186.

Novotny, V. (2010). Urban Water and Energy Use from Current US Use to Cities of the Future. *Proceedings of the Water Environment Federation*, 2010(2), 118-140.

NRMMC, E. (2008). *AHMC, Australian guidelines for water recycling: managing health and environmental risks (Phase 2): Augmentation of drinking water supplies*, Environment Protection and Heritage Council, National Health and Medical Research Council. Canberra: Natural Resource Management Ministerial Council.

Paola, F., Galdiero, E., Giugni, M., & Pugliese, F. (2015). Sustainable development of storm-water systems in African cities considering climate change. *Procedia Engineering*, 119, 1181-1191.

Pataki, D., Alig, R., Fung, A., Golubiewski, N., Kennedy, C., McPherson, E., . . . Romero-Lamkaoss, P. (2006). Urban ecosystems and the North American carbon cycle. *Global Change Biology*, 12, 1-11.

Petit-Boix, A., Deykota, J., Phillips, R., VargasParra, M., Josa, A., Gabarrell, X., . . . Apul, D. (2018). Life cycle and hydrologic modeling of rainwater harvesting in urban neighborhoods: implications of urban form and water demand patterns in the US and Spain. *Science of the Total Environment*, 621, 434-443.

Pumo, D., Arnone, E., Francipane, A., Caracciolo, D., & Noto, L. (2017). Potential implications of climate change and urbanization on watershed hydrology. *Journal of Hydrology*, 554, 80-99.

Ramos, H., Teyssier, C., Samora, I., & Schleiss, A. (2013). Energy recovery in SUDS towards smart water grids: A case study. *Energy policy*, 62, 463-472.

Richards, P., Farrell, C., Tom, M., Williams, N., & Fletcher, T. (2015). Vegetable raingardens can produce food and reduce stormwater runoff. *Urban Forestry & Urban Greening*, 14(3), 646-654.

Rosa, A. (2016). *Bioretention for diffuse pollution control in SUDS using experimental-adaptive approaches of ecohydrology*. Sao Carlos: Thesis. Sao Carlos School of Engineering, University of Sao Paulo.

Rosinger, A. (2018). Household water insecurity after a historic flood: diarrhea and dehydration in the Bolivian Amazon. *Social Science & Medicine*, 197, 192-202.

Schlesinger, W. H. (1999). Carbon sequestration in soils. *Science*, 284, 725-730.

Semadeni-Davies, A., Hernebring, C., Svensson, G., & Gustafsson, L. (2008). The impacts of climate change and urbanization on drainage in Helsingborg, Sweden: Suburban stormwater. *Journal of Hydrology*, 350(1-2), 114-125.

Shrestha, A., Babel, M., Weesakul, S., & Vojinovic, Z. (2017). Developing Intensity–Duration–Frequency (IDF) curves under climate change uncertainty: the case of Bangkok, Thailand. *Water*, 9(2), 145.

Simonovic, S. P. (2016). From risk management to quantitative disaster resilience–a paradigm shift. *International Journal of Safety and Security Engineering*, 6(2), 85-95.

Simonovic, S., & Arunkumar, R. (2016). Comparison of static and dynamic resilience for a multipurpose reservoir operation. *Water Resources Research*, 52(11), 8630-8649.

Simonovic, S., & Peck, A. (2013). Dynamic resilience to climate change caused natural disasters in coastal megacities quantification framework. *International Journal of Environment and Climate Change,* 378 - 401.

Soro, G., Goula-Bi, T., Kouassi, F., & Srohourou, B. (2010). Update of intensity duration frequency curves for precipitation of short durations in tropical area of west Africa (Côte D'ivoire). *Journal of Apllied Sciences*, 10(9), 704-715.

Srivastav, R., Schardong, A., & Simonovic, S. (2014). Equidistance quantile matching method for updating IDF Curves under climate change. *Water Resources Management*, 28(9), 2539-2562.

Stovin, V., Moore, S., Wall, M., & Ashley, R. (2013). The potential to retrofit sustainable drainage systems to address combined sewer overflow discharges in the Thames tideway catchment. *Water and Environment Journal*, 27(2), 216-228.

Sun, Y., Zhang, D., & Wang, Z. (2017). The potential of using biological nitrogen removal technique for stormwater treatment. *Ecological Engineering*, 106, 482 - 495.

The Prince George's County. (2007). *Bioretention Manual*. Environmental Services Division, Department of Environmental Resources.

Tom, M., Fletcher, T., & McCarthy, D. (2014). Heavy metal contamination of vegetables irrigated by urban stormwater: a matter of time?. *PloS one*, 9(11), e112441.

UN – United Nations. (2020). *The 17 Goals*. Available in: < https://sdgs.un.org/goals>. Acess in: Oct 2020.

Wan, Z., Li, T., & Shi, Z. (2017). A layered bioretention system for inhibiting nitrate and organic matters leaching. *Ecological Engineering*, 107, 233-238.

Wang, D., Hagen, S., & Alizad, K. (2013). Climate change impact and uncertainty analysis of extreme rainfall events in the Apalachicola River basin, Florida. *Journal of Hydrology*, 480, 125 - 135.

Wang, M., Zhang, D., Adhitvan, A., Ng, W., Dong, J., & Tan, S. K. (2016). Assessing cost-effectiveness of bioretention on stormwater in response to climate change and urbanization for future scenarios. *Journal of Hydrology*, 543, 423-432.

Wang, S., Lin, X., Yu, H., Wang, Z., Xia, H., An, J., & Fan, G. (2017). Nitrogen removal from urban stormwater runoff by stepped bioretention systems. *Ecological Engineering*, 106, 340-348.

Waterways, M. (2005). *Water sensitive urban design: technical design guidelines for South East Queensland*. Moreton Bay Waterways and Catchment Partnership.

Willems, P., & Vrac, M. (2011). Statistical precipitation downscaling for small-scale hydrological impact investigations of climate change. *Journal of Hydrology*, 402(3-4), 193-205.

Willens, P. (2013). Revision of urban drainage design rules after assessment of climate change impacts on precipitation extremes at Uccle, Belgium. *Journal of Hydrology*, 496, 166-177.

Winston, R., Dorsey, J., & Hunt, W. (2016). Quantifying volume reduction and peak flow mitigation for three bioretention cells in clay soils in northeast Ohio. *Science of the Total Environment*, 553, 83 - 95.

Winston, R., Luell, S., & Hunt, W. (2011). Evaluation of undersized bioretention stormwater control measures for treatment of highway bridge deck runoff. *Water Science and Technology*, 64(4), 974-979.

Wong, T., & Eadie, M. (200). Water Sensitive Urban Design —a paradigm shift in urban design. *Proceedings of the Xth World Water Congress*, 12 - 16.

Zahmatkesh, Z., Burian, S., Karamouz, M., Tavakol-Davani, H., & Goharian, E. (2014). Low-impact development practices to mitigate climate change effects on urban stormwater runoff: Case study of New York City. *Journal of Irrigation and Drainage Engineering*, 141(1), 04014043.

Zhang, K., Manuelpillai, D., Raut, B., Deletic, A., & Bach, P. (2019). Evaluating the reliability of stormwater treatment systems under various future climate conditions. *Journal of Hydrology*, 568, 57-66.

Zhang, K., Randelovic, A., Page, D., McCarthy, D., & Deletic, A. (2014). The validation of stormwater biofilters for micropollutant removal using in-situ challenge tests. *Ecological Engineering*, 67(1), 1e10.

3 ADAPTIVE NATURE-BASED SOLUTIONS FOR SUSTAINABLE STORMWATER MANAGEMENT UNDER DRIVERS OF URBANIZATION AND CLIMATE CHANGE

A modified version of this chapter has been submitted as: Marina Batalini de MACEDO, Marcus Nóbrega GOMES JUNIOR, Vivian JOCHELAVICIUS, Thalita Raquel Pereira de OLIVEIRA, Eduardo Mario MENDIONDO. **Adaptive Nature-based Solutions for sustainable stormwater management under drivers of urbanization and climate change**. Hydrological Science Journal.

Abstract

Adaptive Nature-based Solution (NbS) for urban drainage considering future scenarios under change can contribute to prevent the increase of flood-related hazards and water insecurity. This study incorporates land use and climate change into existing design methods and performs a sensitivity analysis to identify parameters causing most uncertainties. A case study is presented for the city of Sao Carlos – SP, Brazil, for a bioretention structure, in three application scales and three temporal scenarios. The choice of the design method was the factor with greatest influence on the final bioretention performance, as it considerably affected the sized areas, followed by the hydraulic conductivity of the media, representing the structure aging. The runoff coefficient (C) and the daily precipitation with 90% probability (P90) was identified as the most sensitive parameters. Due to the modular design, the performance of the bioretention has remained almost constant in future periods.

Keywords: Modulation; IDF curves; Bioretention; LID; Adaptive design; Subtropical climate.

3.1 Introduction

The rapid urbanization causes structural and environmental changes in urban catchments, reducing soil infiltration and increasing the amount of pollutants build up (Leopold, 1968; Konrad and Booth, 2005; Wong & Eadie, 2000). As a result of these changes, there is a significant increase in runoff, converting the natural hydrological cycle into an urban problem. Extreme rainfall events are precursors of risk to the population (Santos, 2007; Young et al., 2015), with greater vulnerability to floods and landslides.

Considering the climate change scenarios, extreme events and their consequences tend to become more frequent (Debortoli et al., 2017; Marengo et al., 2010). Increases in rainfall volume, rainfall intensity and natural disaster incidents are consequences predicted by the studies (IPCC, 2007). The additional stress on infrastructure, construction, environmental conditions, as well as the large population concentration in urban centers, make cities one of the main locals where occur the climate change impacts. As an example, for the United Kingdom, Houston et al. (2011) estimate that the combination of climate change and population growth in cities will cause an increase in rainfall flood risk for over 1.2 million people.

After discussing the importance of cities in the climate change context, both as a contributor and suffering the consequences, the 40 largest cities in the world formed the C40 group to discuss and exchange public management actions and policies aiming to reduce the impacts generated and suffered for them. In the report released by the group in 2014 (C40, 2014), 90% of the cities that are part of the group indicate that climate change represents significant risks to their cities, the main ones associated with flooding and water stress. In addition, they also pointed drainage as the key to flood risk management, in which adaptive Nature-based Solutions (NbS) for urban drainage systems (such as Low Impact Development – LID practices) occupy third place in the most performed actions by the group.

Knowing that both urbanization and climate change generate an exceeding runoff in cities, several studies (Semadeni-Davies et al., 2008; Pyke et al., 2011; Liu et al., 2016; Liu et al., 2017) have been conducted to investigate the magnitude of each of these changes in urban drainage systems. In this sense, Liu et al. (2016) and Liu et al. (2017) made a joint analysis of the effects of urbanization and climate change on the LID practices for two watersheds in Indiana, USA, using the L-THIA-LID 2.1 model. For the watershed 2, both urbanization and climate changes contributed to increase the runoff and pollutant generated, being intensified in the scenario with the two drivers of change acting together.

Pyke et al. (2011) used a simple stormwater model, SGWATER, to assess the runoff and pollutant loads sensitivity due to changes in impervious cover, rainfall volume and intensity. The results obtained suggested that the runoff generation is more sensitive to increasing urbanization than changes in rainfall patterns. Similar analyzes were obtained by the study of Semadeni-Davies et al. (2008), which was conducted to a city in Sweden, with simulations through 15 months. The results obtained were that climate change and urbanization, jointly and separately, increases the peak flows and runoff volume in a city, increasing the flood risk.

Adaptive urban drainage projects need to consider both drivers of change since the conception and design. In some locations, guidelines already propose the update of design storm through Intensity-Duration-Frequency (IDF) curves incorporating the non-stationary effects of climate change, aiming infrastructure sizing, such as in Europe and New York State (Willems, 2013; Madsen et al., 2014, DeGaetano and Castellano, 2017). However, there are still doubts concerning what are the main parameters needing greater attention and efforts during the design stage of adaptive NbS for urban drainage systems, when aiming to address future scenarios with drivers of change. Currently, there are several design methods suggested in guidelines (such as the Water Quality Volume – WQV; envelope curve, volume retention, simulation routine), accounting with different input variables, initial parameters and different characteristics of design storm, besides uncertainties in the determination of some physical parameters, generating a variability in the results of the systems performance. Therefore, there is a need for further studies that help to clarify these doubts and reduce the uncertainties in the design stage.

This paper aims to present a methodological sequence for the incorporation of climate and urbanization changes in the design of adaptive urban drainage structures, more specific in a bioretention system. Considering that it is necessary to design structures to reduce risks to the population while minimizing construction costs, a modular design approach is also presented here. Changes in rainfall patterns are considered since the design from the updating of IDF curves for the future climate, changing the design storm. On the other hand, scenarios of future land use are incorporated through changes in the values of the rainfall-runoff transformation coefficients, in the infiltration methods. The methodology was applied to the Minheirinho Creek catchment, in the city of Sao Carlos (one of the representatives mid-size Brazilian city) located in southeastern Brazil, under subtropical climate.

In addition, as a way of assessing the uncertainties that exist in the different methods, and contributing to identify the parameters and input variables requiring more attention in the systems considering future scenarios, a sensitivity analysis was performed. The parameters and input variables involved in different sizing methods evaluated were chosen considering urbanization (runoff coefficient, curve number, % impervious area, soil type), climate (IDF, rain duration, temporal distribution pattern) and structure aging (hydraulic conductivity). Finally, a simulation of future scenarios was carried out.

3.2 Methodology

The study methodology follows the sequence shown in Figure 3.1: (1) Selection of discrete ranges for the application site and design methods, allowing to identify the input variables and initial design parameters related to urbanization and climate patterns; (2) Selection of discrete ranges regarding future urbanization, climate patterns and infrastructure aging; (3) To perform global sensitivity analysis and identify the main parameters and input variables affecting the performance of the system; (4) To evaluate the efficiency of the adaptive urban drainage structure, by simulating scenarios in the process-based model developed by Randelovic et al. (2016) and adapted by Shen et al. (2018), considering future scenarios with different drivers of change and intervention.

3.2.1. Parameters and input variables representing application site and design methods

3.2.1.1. Study area and application scales

The design of adaptive urban drainage practices, considering the drivers of change in future scenarios, was proposed for the Mineirinho Creek catchment, one of the catchments that forms the urban watershed of the city of Sao Carlos City, SP, Brazil (Figure 3.2), located in the southeast of the country.

For the city of Sao Carlos, the climate of the region according to the Köppen-Geiger climate classification is Cfa (humid summer subtropical climate), with an annual average rainfall of 1361.6 mm, and average daily temperature of 21.5ºC (Weatherbase, 2018). The city has its urban limits within the Monjolinho watershed, having a total area of 76.8 km^2 and population density of 194.53 inh./km^2. The city counts with several points of recurrent flooding, the main ones located in areas of high commercial density (Fava et al., 2018, Abreu, 2019). Therefore, the floods cause significant economic damage to the city (Abreu, 2019). One of the recurring flooding points is in the Mineirinho Creek catchment outlet (Figure 3.3). In this point business such as car dealership, shopping mall, restaurants and construction store are located.

In addition, the Mineirinho Creek catchment presents itself as a diverse peri-urban area, with sites reserved for the preservation of its springs, and at the same time with predicted expansion of housing land division (PMSC, 2016). Due to its diverse characteristics and contributing to flooding problems in the city of Sao Carlos, this area was selected to design adaptive drainage structures with drivers of change.

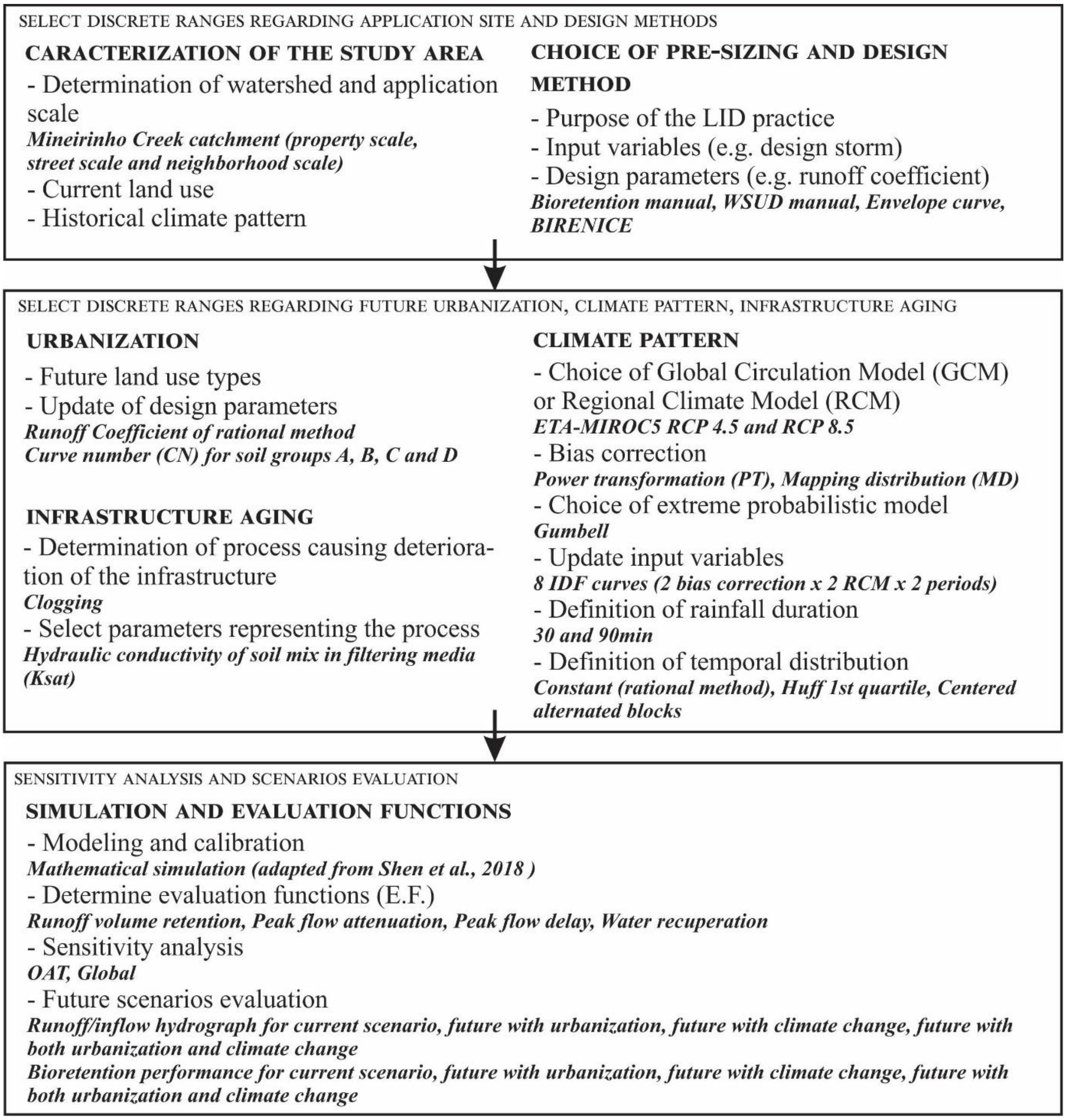

Figure 3.1 - Methodology flowchart. Highlighted: the location, data, parameters, and methods chosen for this study.

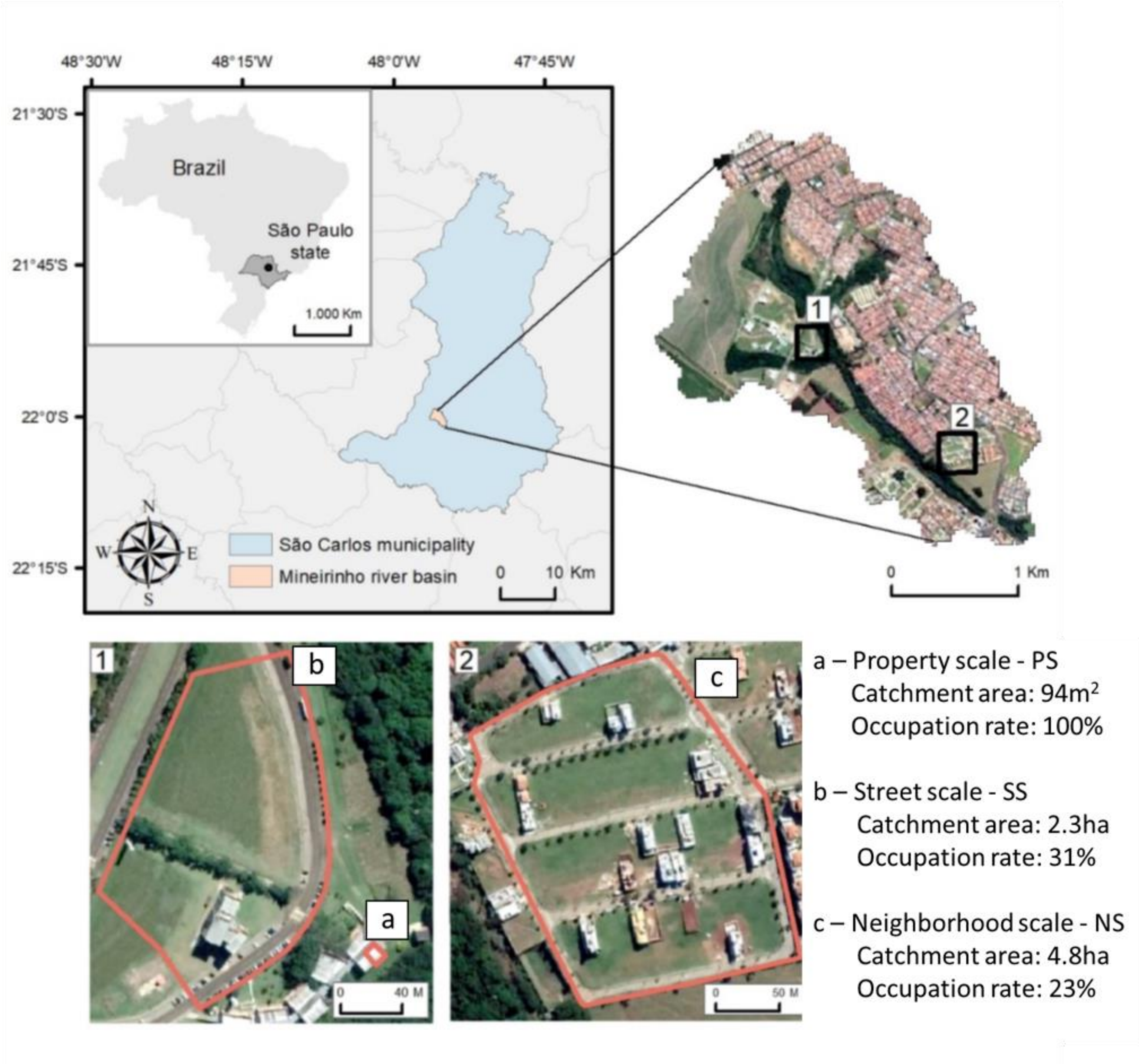

Figure 3.2 - Geographic location of the study area: Mineirinho river catchment and implementation sites of LID practices at property scale, street scale and neighborhood scale. Satellite photos from 2019.

When it comes to adaptive urban drainage measures, they can be applied at three major scales: property/lot scale (PS), street scale (SS) and neighborhood scale (NS) (Marsalek and Schreier, 2009; Waterways, 2005). In this study, three bioretention structures were designed for the three application scales (PS, SS and NS) with predicted increase in urbanization level (Figure 3.2). The PS – area a is a roof, therefore completely occupied, the SS – area b presents current occupation of 31% (as shown in Figure 3.2) and NS – area c presents current occupation of 23% (as shown in Figure 3.2).

Figure 3.3 - Pictures of the flood disaster and economical losses occurred in 14 Jan 2020 in Sao Carlos, SP, Brazil (unknow source, wide circulation images on the internet)

The mitigation purposes used for the design were increase of the resilience to floods (runoff retention, peak flows amortization, increase of the delay in the occurrence of peak flow), as well as increase of the resilience to drought (recovery of runoff treated by bioretention for local non-potable use), since climate change can arise water scarcity problems to the city of Sao Carlos. Regarding this last purpose, the bioretention was designed as a lined structure with an underdrain and a saturated zone (Figure 3.4), to promote greater removal of pathogens and nutrients.

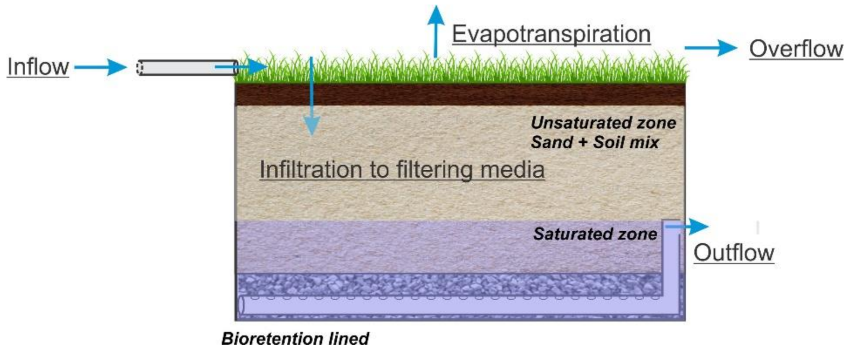

Figure 3.4 – Structure of bioretention designed in this study to meet purposes of flood control and stormwater harvesting

3.2.1.2. Simplified LID practices design and pre-sizing

Several sizing methods for bioretention have been proposed in the USA, Australia and other countries, and are often used internationally. The most commonly methods used in Brazil and a summary of the design purposes and key parameters for each method are presented in Table 3.1. Details of each method can be found in their references.

From Table 3.1, it is possible to observe that the design methods with the purpose of mitigating rainfall extremes, such as runoff volumes and peak flows, present the design storm as main input data (obtained from the IDF curve, defining a duration and temporal distribution). In addition, the main parameters include rain-flow transformation coefficients, namely: surface runoff coefficient (C) of the Rational Method and curve number (CN) of the SNCS – CN method. These parameters vary depending on the land use type and are assigned to each location based on its level of urbanization.

The modular design concept is presented in the proposal of BIRENICE (Rosa, 2016), which consists of sizing the adaptive urban drainage practice for long-term future scenarios, but in a way that its implementation is made modularly, for predefined time intervals. With this approach it is possible to guarantee the safety of the structure, reduce the risk for the population, and, at the same time, amortize the construction and operational costs. To this end, the input data and parameters of each of the sizing methods can be updated to match future scenarios of both urbanization and rainfall patterns changes to predefined expansion intervals.

In this study, the sizing method was considered as an initial variable in the sensitivity analysis, i.e. all of the sizing methods presented were used to determine the bioretention surface areas and their respective performances, jointly varying the parameters and input variables of each method, according to future urbanization scenarios, climate patterns and infrastructure aging. The bioretention dimensions other than the area were kept constant. For the methods that is necessary to adopt the peak of the ouflow/overflow, it was adopted the peak flow for the catchment calculated through rational method, for the property scale, or the unitary hydrograph of the SNCS – CN method, for the street and neighborhood scale, adopting the values of the infiltration methods relative to pre urbanization scenarios (underbrush).

Table 3.1 - Comparison between internationally used sizing methods

Method	Design storm	Mitigation purpose	Main parameters	Variables assumed	Source
Bioretention manual	Q90	Water quality Exceedent volume* Peak flow**	Ac; WQV; Rv; I; K	hb; db; tb; FS, filtering media	The Prince George's County (2007)
WSUD technical guidelines	IDF	Water quality Peak flow*	Ac, C, K, tc	hb; db	Waterways (2005); COUNCIL (2007); McAuley (2009)
Envelope curve/rain method	IDF; PDF	Exceedent volume Peak flow	Ac; C	Qs; qs, filtering media	Urbonas and Stahre (1993); Silveira and Goldenfun (2007)
BIRENICE	IDF	Exceedent volume Peak flow** Water quality***	Ac; CN, n	Bioretention dimensions (flexible), filtering media	Rosa (2016)

Ac - Catchment area; WQV - Water quality volume; Rv - Linear runoff coefficient; I - Percentage of impervious area; K - Permeability coefficient of the filtering media; hb - Ponding depth; db - Filtering media depth; tb -Emptying time; FS - Safety factor; C - Runoff coefficient; Qs - Outflow; L - Length of the saturated area; Ab - Bioretention surface area; CN - Curve number coefficient; tc - time of concentration.

* additional purpose ** complementary simulation *** restriction

3.2.2 Parameters and input variables representing drivers of change for future scenarios

3.2.2.1 Future urbanization: changes in land use

One of the main drivers of change for future scenarios to be considered in the design of urban drainage structures is the expansion of the urbanized area and, consequently, changes in land use. These modifications alter the soil water storage capacity, mostly contributing to the increase of the runoff generation. Therefore, the urbanization process alters peak flows and total runoff volume and should be considered in urban drainage structure designs to maintain efficiency over their lifetime. To incorporate these changes, an alternative is to update the value of runoff coefficient, curve number (CN) or soil infiltration capacity (according to the infiltration method adopted). In the WSUD Technical Guidelines (Waterways, 2005), for example, a correction factor in the value of the runoff coefficient of the Rational Method (C) is presented as a function of the return period (RP) used for the design storm. However, this update of the C value for higher RP are related to the increase of previous soil moisture and decrease of infiltration, due to higher rainfall volumes in a same event duration, when considering events with higher intensity.

For this study, we propose to change the C and CN values from the predicted land use types and expansion rates in different locations, based on the master plan of Sao Carlos city (PMSC, 2016). For each of the catchments, the final value of C or CN, for each time interval in the modular expansion, is obtained from the weighting of the land use types with its respective area. The values of CN for the different land use types and hydrological soil groups were obtained from Canholi (2014).

The level of detail of the Brazilian soil maps and the costs for soil analysis sometimes leads to errors in the most correct classification of the soil type and its hydrological group, leading thus to errors in the CN estimation. Allasia (2002) observed that for each unity of error in the CN estimation leads to 8% of error in the runoff calculation. Therefore, the CN value was also considered as a parameter of the sensitivity analysis.

3.2.2.2 Climate change

In order to incorporate changes in rainfall patterns in the site due to climate changes, it is proposed to update the IDF curves to generate design storm more compatible with future scenarios (Madsen et al., 2009; Soro et al., 2010; Mailhot and Duchesne, 2009). Several studies (Wang et al., 2013; Madsen et al., 2014, de Paola et al., 2015) has already recommended updating IDFs for this purpose. This procedure have been adopted in New York State and Belgium guidelines (Willems, 2013; DeGaetano and Castellano, 2017).

In this study, the IDFs were updated using future data projected by a regional climate model (RCM) developed by INPE-PROJETA (Chou et al., 2014 and Lyra et al., 2018) for the region of Sao Carlos, Brazil, Eta -MIROC5. This model has been used in more than 100 studies in South America. The maximum daily rainfall data for the RCP 4.5 and 8.5 radioactive forcing scenarios was used, representing a more optimistic and pessimistic scenario of climate change.

The RCM used has spatial resolution of 20km grid box. Therefore, the IDF curves generated from rain gauges data (punctual data) cannot be directly compared with data obtained from the RCMs (Wang et al., 2013). In addition, urban drainage problems have small spatial scales and typically need answers ranging from hours to minutes, while global and regional models typically have resolutions from days to hours (Willems and Vrac, 2011). Thus, a bias correction procedure is required.

The bias correction procedure was performed using the CMHyd program, as it is an open source and is widely used for climate change hydrological modelling, mainly linked to the Soil and Water Assessment Tool (SWAT). This program presents different bias correction methods to be employed, namely: linear scaling, local intensity scaling, power transformation, variance scaling, distribution transfer and delta-change approach. The choice of the method to be used varies according to the type of data used (rainfall or temperature) and the final purpose of the study.

To assist in the choice, Teutschbein and Seibert (2012) compared the different methods for hydrological simulation purposes in different watersheds. As a result, the power transformation (PT) and mapping distribution (MD) methods presented better adjustments considering the statistical characteristics and the variability intervals. However, the PT did not accurately correct the probability of dry days, maintaining a large bias in the frequency of dry days. As this study aimed at the maximum annual precipitation values per year, both methods were chosen for bias correction.

Power transformation allows differences in the variance to be corrected and use a non-linear correction in an exponential form (a.Pb). Distribution mapping is also known as quantile-quantile mapping or statistical downscaling. It corrects the distribution function of RCM-simulated precipitation values to fit the observed distribution function, assumed to be the gamma distribution to precipitation events.

Finally, the bias correction was made using 44 years of observed historical overlapping data (obtained by the weather station of the National Institute of Meteorology – INMET for the city of Sao Carlos) with historical data obtained by the employed models, using the interval from 1961 to 2005. For both models, the adjustment for future scenarios was made from the

2015 to 2099 range. In Figure 3.5 it is possible to see the monthly difference between the average rainfall observed and modeled, with and without the bias correction, highlighting the importance of this procedure.

Even when performing bias correction, there is still a great uncertainty in global and regional climate models, as well as in the bias correction method itself. These uncertainties should be considered in hydrological simulations. Willems and Vrac (2011) propose that instead of quantifying statistical uncertainties it would be possible to deal with uncertainty scenarios, using various climate models and emission scenarios and applying different bias correction methods. Therefore, this study analyzed the variability of scenarios considering the RCM used for the RCP 4.5 and 8.5 radioactive forcing and the two bias correction methods employed.

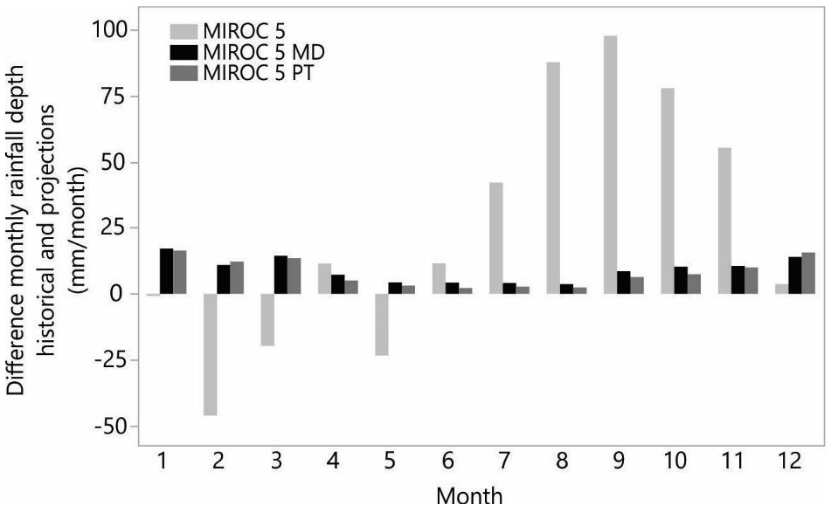

Figure 3.5 - Difference between the observed and modelled monthly rainfall average data with and without bias correction for climate models used.

The city of Sao Carlos presents itself as a partially gauged site – there is no sub daily rainfall data collection with a timescale needed to construct the IDFs, which makes unfeasible the temporal downscaling of the maximum daily rainfall data obtained by climate change models. Therefore, the sub-daily rainfall depths were obtained using disaggregation factors obtained the data observed by Barbassa (1991) (Table 3.3). Similar approaches were performed by Koutsoyiannis et al. (1998), Lehmann et al. (2016) and Muller et al. (2008): They have also used estimates of local distribution obtained for sites with sub daily records to calculate parameters for future climate projections at sites where only daily rainfall is observed, considering the assumption of scale-invariant of rainfall intensity and duration. The disaggregation factors used in this study were proposed for Gumbel distribution model.

For the IDF curve construction, it was used a probabilistic distribution model for extreme values that suited the data set of future scenarios. The theoretical probability distribution was evaluated using the Kolgomorv-Smirnov test, with p-value hypothesis test and coefficient of determination, using Excel 2016 (Levine et al., 2008). In addition, empirical probability distributions (those that aim to explain the population rather than its projection) can also be evaluated against the theoretical distributions. Among the empirical distributions, the proposals of Weibull, Cunnane, Gringorten are highlighted. Regarding the theoretical distributions, the distribution of Gumbel, Log-Normal, Normal, Log-Pearson type 3, Generalized extreme value (GEV), are the most used. Based on the aptitude of the empirical and theoretical combination regarding the statistical indicators and hypothesis tests, the Gumbel's model was chosen, since it passed on the hypothesis and statistical indicators test for all scenario combinations (Table 3.2) and is the most commonly used for building historical IDFs in Brazil. Although other distributions point to greater aptitude for the calculation of IDFs, such as GEV (Cheng & AghaKouchack, 2014; Lima et al., 2018), we adopted the Gumbell model since we used disaggregation factors obtained for the historical IDF of Sao Carlos, built from the Gumbell distribution model.

Table 3.2 - Results obtained for the statistical adherence test for the Gumbel Weibull distribution model for all combinations of RCM and bias correction.

RCM	RCP	Bias-correction	R^2	D-KS	p-value
		2015 - 2050			
Eta - MIROC5	4.5	MD	0.956	0.136	0.51
		PT	0.976	0.123	0.31
	8.5	MD	0.986	0.099	0.10
		TP	0.975	0.147	0.59
		2051 - 2099			
Eta - MIROC5	4.5	MD	0.988	0.111	0.29
		PT	0.991	0.109	0.27
	8.5	MD	0.984	0.134	0.53
		PT	0.99	0.117	0.35

Table 3.3 - Disaggregation factors for sub daily rainfall data for Sao Carlos - SP

Disagregation factors proposed by Barbassa (1991)		Absolute disagregation factors to daily rainfall	
5min/30min	0.34	5 min	0.12
10min/30min	0.54	10 min	0.19
15min/30min	0.70	15 min	0.25
20min/30min	0.81	20 min	0.29
25min/30min	0.91	25 min	0.32
30min/1h	0.74	30 min	0.35
1h/24h	0.42	60 min	0.48
6h/24h	0.72	360 min	0.82
8h/24h	0.78	480 min	0.89
10h/24h	0.82	600 min	0.93
12h/24h	0.85	720 min	0.97
24h/1dia	1.14	1440 min	1.14

The design storm also requires the adoption of a rainfall duration and a pattern of temporal distribution. Adopting different values of both can also lead to changes in the final performance of urban drainage systems. Therefore, variations in these parameters were also adopted to assess sensitivity. For the rainfall duration it was adopted the values of 30 and 90 minutes (generally adopted in Brazilian urban drainage manuals) and constant temporal distribution (according to the use of the rational method), Huff 1st quartile, and centralized alternated blocks, for intervals of 5min.

3.2.2.3 Structure aging

When thinking about mitigating impacts for future scenarios, the aging of the structure is a factor that should be considered since the design stage. Many of the adaptive urban drainage structures are based on infiltration principles to re-establish the hydrological cycle and filtration-based water treatment. Over time, the infiltration capacity of the filtering media is reduced due to clogging. If this process is not considered in the design, the structure may not work with the same performance for future scenarios or can even become obsolete.

In this study, the importance of the filtering media clogging was evaluated from the sensitivity of the sized area and performance of the bioretention to the hydraulic conductivity of the filtering media. Two hydraulic conductivity values were adopted, one representing the new structure $K_{sat,n}$ = 468 mm/h, theoretical value obtained by the weighted average of the hydraulic conductivity for each type of material used in the filtering media, and other representing the old structure, i.e. years after use ($K_{sat,o}$ = 195 mm/h, value obtained experimentally for a bioretention after years of use). The sensitivity analysis was performed considering the combinations: (design parameter, simulation parameter) = ($K_{sat,n}$, $K_{sat,n}$), ($K_{sat,n}$ $K_{sat,o}$), ($K_{sat,o}$, $K_{sat,n}$) and ($K_{sat,o}$, $K_{sat,o}$).

3.2.3 Simulation model

The simulation of bioretentions for the different application areas was made in two steps: the first consisted of estimating the runoff hydrographs that enters the bioretention and the second consisted of simulating the hydrological and hydraulic processes that occur within the bioretention structure. In order to estimate the runoff hydrographs (that serves as inputs in the second step), the design storm constructed from the IDFs for the city of Sao Carlos for the current and future scenarios was used as input data, as previously explained in section 3.2.2.2. The transformation of rainfall into runoff was made using two methods, depending on the catchment area: the simple Rational Method was used for the area a - PS, since it is a 94m^2 roof area, therefore very small in length and time of concentration; the SNCS - CN Unit Hydrograph Method was used for areas b - SS and area c - NS, due to the greater extension and longer concentration time.

In addition, in order to simplify the analysis, all areas were considered with constant time of concentration throughout the urbanization, as they are small catchments with drainage infrastructure already consolidated. For the PS, as it is a small roof, a time of concentration of 5 min was adopted. For SS and NS, time of concentration of 15 and 20 min were adopted, respectively (values estimated by field evaluation).

For the simulation of hydrological and hydraulic processes that occur within the bioretention structure, the water flow module of the mathematical model proposed by Randelovic et al. (2016) and adapted by Shen et al. (2018). This model was also used for the evaluation of the system's performance, quantification of the sensitivity analysis evaluation functions (presented in section 3.2.4) and simulation of future scenarios. This model uses a process/physically based approach and has been proved to be efficient enough to simulate water flows in bioretention, for different configurations (with and without saturated zone, lined and unlined) and for different mitigation purposes. Further details of the model and each of the equations representing the water mass balance and the state variables are presented in Chapter

6. A summary of the parameters required in the model and the adopted values are presented in Table 3.4. In this study, the model was implemented in Python 3.8.

For this study, an automatic calibrator for the model was developed using genetic algorithms (from the Distributed Evolutionary Algorithms in Python library - DEAP 1.3.1) using the maximization of the average Nash-Sutcliffe coefficient (NSE) to outflow through the underdrain and the height of the water level in the ponding zone as the objective function. Six synthetic events monitored for ponding depth and outflow during the year 2019 in a bioretention box in laboratory scale were used for the calibration process (three used for calibration and three for validation). The synthetic events represented an event with constant intensity, effective rainfall of 31mm and 53mm, with a total duration of 30min (Table 3.4. These events simulated the runoff generation for the current urbanization and climate conditions at the Mineirinho creek catchment. This bioretention box is used for research at the University of Sao Paulo, in the city of Sao Carlos and it was constructed with the same materials and configurations (soil mix, lined, with saturated zone) proposed in the three application scales of this study (Figure 3.2 and Figure 3.4). More details about the bioretention box and the monitoring data can be seen in Chapter 5. The final NSE values obtained for calibration was 0.74 and for validation it was 0.62, demonstrating a good fit to the model (Moriasi, 2007). The parameters used in the calibration and its final value can be seen in Table 3.5.

For the design of bioretention devices and simulation at different scales, a single bioretention structure was considered at the outlet of the conventional urban drainage structure of each catchment area, which receives the total runoff volume from the respective areas. The entry for each system follows the structure format shown in Figure 3.5. Each bioretention of each assessed catchment area was simulated separately for the different scenarios.

Table 3.4 – Description of monitored events in bioretention box used to calibration and validation of bioretention model

	Date	P (mm)	d (min)	API (mm)	Dry days	V_{in} (L)	V_{out} (L)	V_{over} (L)	$V_{storage}$ (L)	Q_{in} (L/h)	$Q_{peak,out}$ (L/h)	$Q_{peak,over}$ (L/h)
Calibration	6/26/2019	31	31.1	7.4	15	747.9	520.7	0	227.2	1442.89	498	0
	8/6/2019	31	28.8	5.0	40	500	392.2	0	107.8	1040.46	456	0
	9/2/2019	31	20.5	10.8	11	240.48	214	0	26.48	1442.89	336	0
Validation	2/4/2019	31	30.4	29.5	2	735.95	633.55	0	102.4	1451	792	0
	3/19/2019	53	30.8	37.7	4	1162.98	860	177.8	125.18	2268	864	1188
	9/30/2019	31	30.0	6.7	27	240.48	265.2	0	-24.72	1442.89	348	0

Figure 3.6 – Example of bioretention applied in the outlet of conventional drainage system, for an application in street scale catchment

Table 3.5 - Parameters adopted or calibrated for mathematical simulation and in design methods

Parameter	Description	Value	Unit	Acquisition
hp	Height of the ponding zone	0.3	m	Adopted in design
H_{sm}	Height of soil mix layer	1.0	m	Adopted in design
H_g	Height of gravel layer	0.2	m	Adopted in design
n_{sm}	Soil mix porosity	0.32	-	Adopted in design
n_g	Gravel porosity	0.4	-	Adopted in design
t_b	Emptying time	24	h	Adopted in design
K_{sat}	Hydraulic conductivity	See section 2.2.3	mm/h	Adopted in design
FS	Safety factor	2	-	Adopted in design
Kc	Evapotranspiration constant for plants	1.38	-	Calibrated
K_{weir}	Weir coefficient	1.3	-	Adopted in design
Exp_{weir}	Weir exponent	2.5	-	Adopted in design
sh	Hydroscopic soil moisture	0.036	-	Calibrated
sw	Wilting point moisture	0.120	-	Calibrated
sfc	Field capacity	0.435	-	Calibrated
ss	Plant stress moisture	0.482	-	Calibrated
h_{pipe}	Height of underdrain pipe	0.2	m	Adopted in design
d_{pipe}	Underdrain pipe diameter	32	mm	Adopted in design
Cd	Discharge coefficient for the pipe	0.33	-	Calibrated
Δt	Time-step	5	min	Adopted in simulation

3.2.4 Sensitivity analysis and evaluation functions

A global sensitivity analysis was performed using Morris screening method (Song et al., 2015). This method aims to identify the input variables and parameters that contribute significantly to the variations and uncertainties of the output, other than to determine the exact sensitivity of the model to a specific parameter or variable. In the Morris method, a discrete number of values are used for each parameter, instead of acquiring directly from its distribution functions. This fits the case evaluated in this paper, since most parameters related to the drivers of change have discrete intervals by nature.

According to Morris (1991) and Campolongo et al. (2007), for a vector of base input parameters/variables X = (x_1, x_2, … x_k), the determination of the elementary effect of the i-th parameter or input variable in the deviation of the evaluation function is given by Eq. 3.1. Subsequently, the mean (μ) and standard deviation (σ) of the elementary effects of each range of parameters or input variables are computed, according to Eq. 3.2 (adapted by Campolongo et al. (2007)) and Eq. 3.3. The μ estimates the general effect of each parameter on the model output (in this case, in order to evaluate the influence on the performance of the bioretention from different design methods), and σ estimates higher order effects, as non-linearity and interaction with other parameters (Song et al., 2015). For parameters that have no numerical value (such as rain distribution, soil type, IDF period, urbanization period and urbanization level), only the differences were computed, without delta weighting.

$$d_i(X) = \frac{y(x_1,\dots,x_{i-1},x_i+\Delta,x_{i+1},\dots,x_k)-y(X)}{\Delta} \tag{3. 1}$$

$$\mu_i = \frac{1}{r}\sum_{j=1}^{r}|d_i(j)| \tag{3. 2}$$

$$\sigma_i = \sqrt{\frac{1}{r-1}\sum_{j=1}^{r}\left[d_i(j) - \frac{1}{r}\sum_{j=1}^{r} d_i(j)\right]^2} \tag{3. 3}$$

where: $d_i(X)$ is the elementary effect of the i-th parameter; Δ is the difference between the base parameter or input variable and the evaluated value; y(X) is the evaluation function for the base parameters or input values; μ_i is the mean of the elementary effects of each parameter; r is the number of sample points in the parameter or input value space; $d_i(j)$ is the elementary effect for input i using the j-th sample point; σi is the standard deviation of the elementary effects of each parameter.

Four evaluation functions were proposed (Eq. 3.4 – 3.7), in order to determine the effects of parameters and input variables on different design purposes of urban drainage structures, such as runoff retention, peak attenuation, peak delay and water reuse (representing the amount of water recovered by the underdrain, which can be reused in the future). These functions were used both for sensitivity analysis and to compare the performance of bioretention in future scenarios with different drivers of change, for different purposes of flood control and water reuse.

$$Eff_{rr} = \frac{V_{in} - V_{over}}{V_{in}} \tag{3.4}$$

$$Eff_{peak} = \frac{Q_{peak,in} - Q_{peak,over}}{Q_{peak,in}} \tag{3.5}$$

$$Eff_{time} = \frac{t_{peak,in} - t_{peak,over}}{t_{peak,in}} \tag{3.6}$$

$$Eff_{wr} = \frac{V_{out}}{V_{in}} \tag{3.7}$$

where: Eff_{rr} [-] is the runoff retention efficiency; V_{in} [L^3] is the total inflow volume; V_{over} [L^3] is the total overflow volume; Eff_{peak} [-] is the peak attenuation efficiency; $Q_{peak,in}$ [L^3T^{-1}] is the maximum inflow value; $Q_{peak,over}$ [L^3T^{-1}] is the maximum overflow value; Eff_{time} [-] is the time delay efficiency; $t_{peak,in}$ [T] is the duration of the event until the $Q_{peak,in}$; $t_{peak,over}$ [T] is the duration of the event until the $Q_{peak,over}$; Eff_{wr} [-] is the water reuse efficiency; V_{out} [L^3] is the total outflow volume.

2.3 Results and discussion

2.3.1 Changes in urbanization and land use

The three areas selected to evaluate the implementation of adaptive urban drainage present heterogeneity concerning the application scale and land use type. The area *a* (Figure 3.2) is a property scale (PS), collecting water from a roof, i.e. land use already consolidated, not changing in future periods. For areas *b* and *c* (Figure 3.6), these represent street scale (SS) and neighborhood scale (NS), respectively, still under urbanization process. The quantification of future land use for the intervals 2015-2050 and 2050-2100 is presented in Figure 3.6. The SS will present an urbanization of 30% for the interval 2015-2050 and 80% by 2100 (PMSC, 2016). For the NS, a faster urbanization is expected, with occupation of 80% by 2050 and 100% urbanized by 2100.

In Table 3.6 are presented the runoff coefficients (C) for the rational method and curve numbers (CN) for the future scenarios of urbanization used in the design calculations. For the PS, since the bioretention receives water from a roof, there is no change in C and CN along time, therefore, the values C = 0.9 and CN = 98 were the same for all periods. For the SS and NS, the effects of the urbanization in the C and CN, and therefore in the runoff generation, are more expressed. It is also possible to notice the influence of the soil type in the final CN values – for soils in hydrological group D, i.e. soils with less permeability, the impact of urbanization is lower than in soils with higher permeability – which suggests the importance of correctly determining the type of soil (the influence in the bioretention design and performance are verified by the sensitivity analysis and simulation).

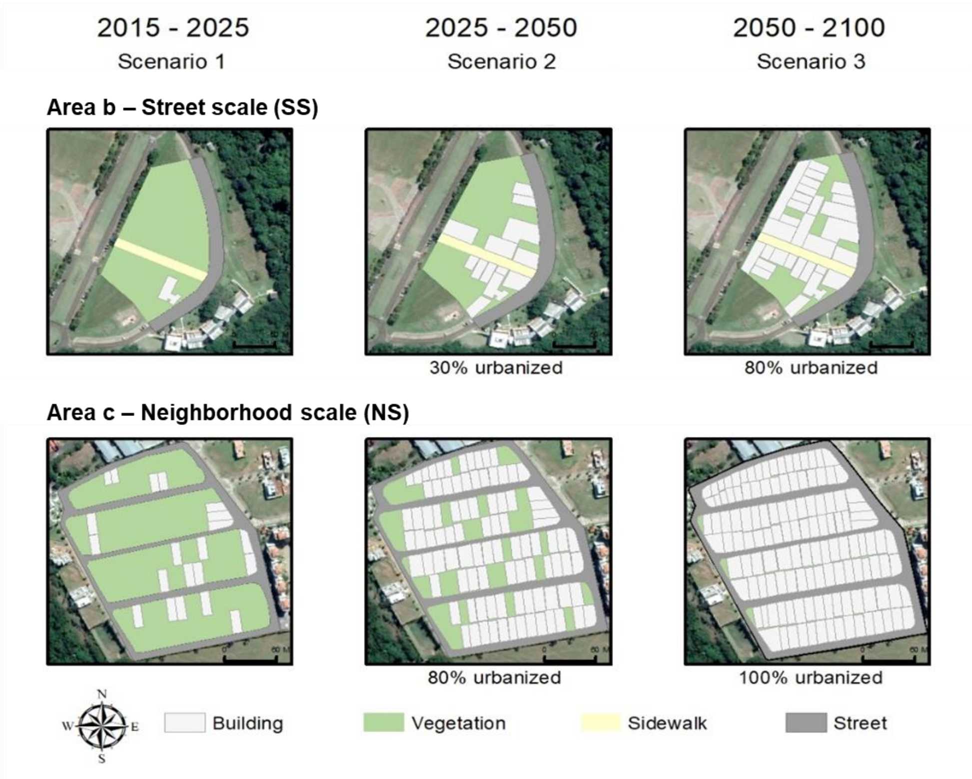

Figure 3.7 - Land use for LID practices at street and neighborhood scale for the current and future scenarios, according to Sao Carlos master plan.

Table 3.6 - Runoff coefficient for current and future scenario according to established land use

	Street scale			Neighborhood scale		
	Current	2015 - 2050 (50% urbanization)	2050 - 2100 (80% urbanization)	Current	2015 - 2050 (80% urbanization)	2050 - 2100 (100% urbanization)
$A_{underbrush}$ (m^2)	15973	11500	3514	36789	11809	1104
A_{roof} (m^2)	573	5046	13032	4877	29856	40562
$A_{sidewalk}$ (m^2)	1910	1910	1910	-	-	-
A_{street} (m^2)	4550	4550	4550	5940	5940	5940
A_{total} (m^2)	23006	23006	23006	47606	47606	47606
C	0.4	0.6	0.8	0.4	0.7	0.9
CN_A	62	72	89	60	86	97
CN_B	77	82	92	76	91	97
CN_C	84	88	94	83	93	98
CN_D	88	90	95	87	95	98

Index in CN represents the hydrological soil group.

2.3.2 Changes in rainfall pattern

Regarding the future scenario of climate change, IDF curve updates were made for the intervals 2015-2050 and 2050-2100 aiming the modular sizing of adaptive urban drainage structures.

First, the historical IDF adopted in this study for the city of Sao Carlos was the one proposed by Gomes Junior (2019) (Table 3.7, current), as an update of the IDF curve proposed by Barbassa (1991), with more recent observed data for maximum daily rainfall. The methodology used by Gomes Junior (2019) to update the IDF for current scenario was the same used in this study. This IDF was used to size the bioretention structures in the current scenario. Further, the IDFs for the city of Sao Carlos were updated considering the climate change scenarios for the intervals 2015-2050 and 2050-2100.

The updated IDFs, with their range of variation considering different future scenario combinations, can be observed in Figure 3.7 and Figure 3.8 and the numerical curve coefficients are presented in Table 3.7. One way to assess the change in rainfall intensities for future scenarios is to observe what would be the historical RP equivalent for a fixed design event. Therefore, considering a design rainfall of 5-years RP and a fixed duration of 30 min for the interval 2015-2050, its historical equivalent is of 2.5-years to 3.5-years RP (range considering the variability in the future scenario) and for the interval 2050-2100 its historical equivalent is of 1.4-year to 1.9-year RP. The same assessment was made for a design rainfall of 50-years RP and a fixed duration of 30 min. It was obtained a historical equivalent of 8.8-year to 11.5-years RP for the interval 2015-2050 and 4.7-years to 8.4-years RP for the interval 2050-2100. This change in the RP represents an increase in the frequency of occurrence for a design rainfall for RP 5.30 up to 2 and 3.6 times and for RP 50.30 up to 5.7 and 10.6 times, for the respective future intervals.

In order to observe if this behavior have significant changes if changing the duration of the rainfall event, the same assessment was made for daily events (fixed duration of 1440min). Increases up to 1.6 and 3.2 times for RP 5 years and up to 5.5 and 10 times for RP 50 years were noted, for the respective future intervals, i.e., same behavior is observed for longer durations.

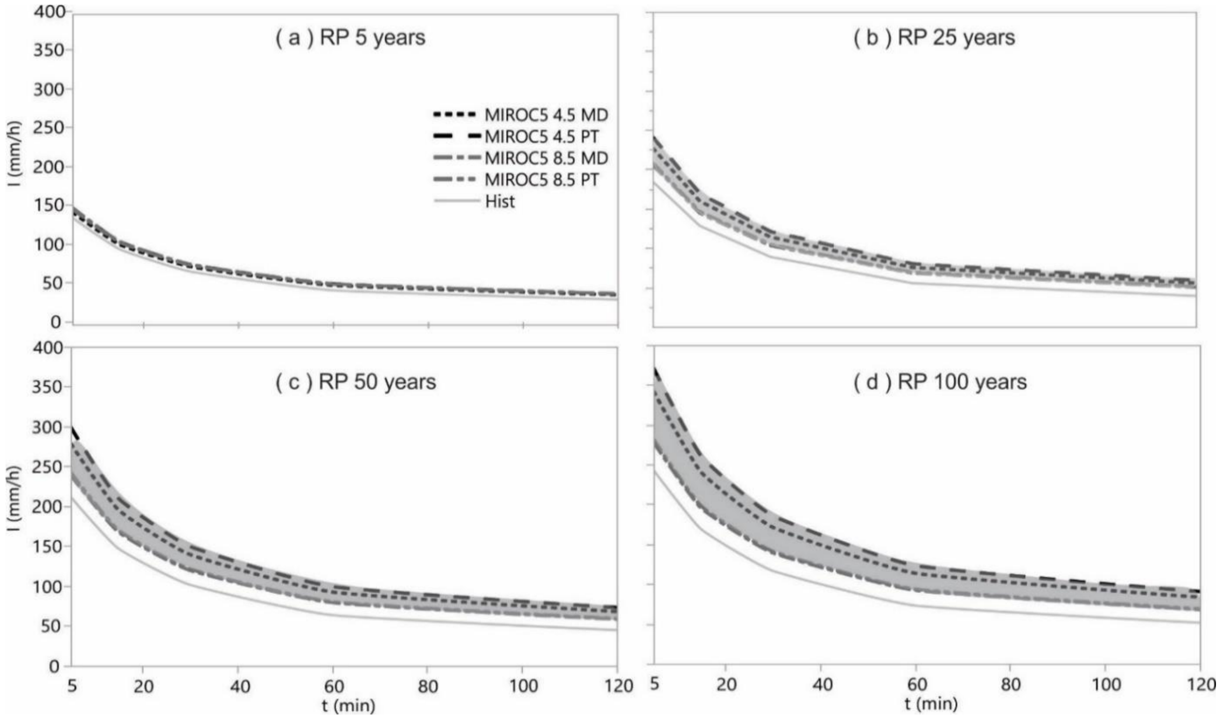

Figure 3.8 - Projected future IDF curves for the period 2015-2050 for the city of Sao Carlos – SP, Brazil and return periods of (a) 5 years, (b) 25 years, (c) 50 years and (d) 100 years.

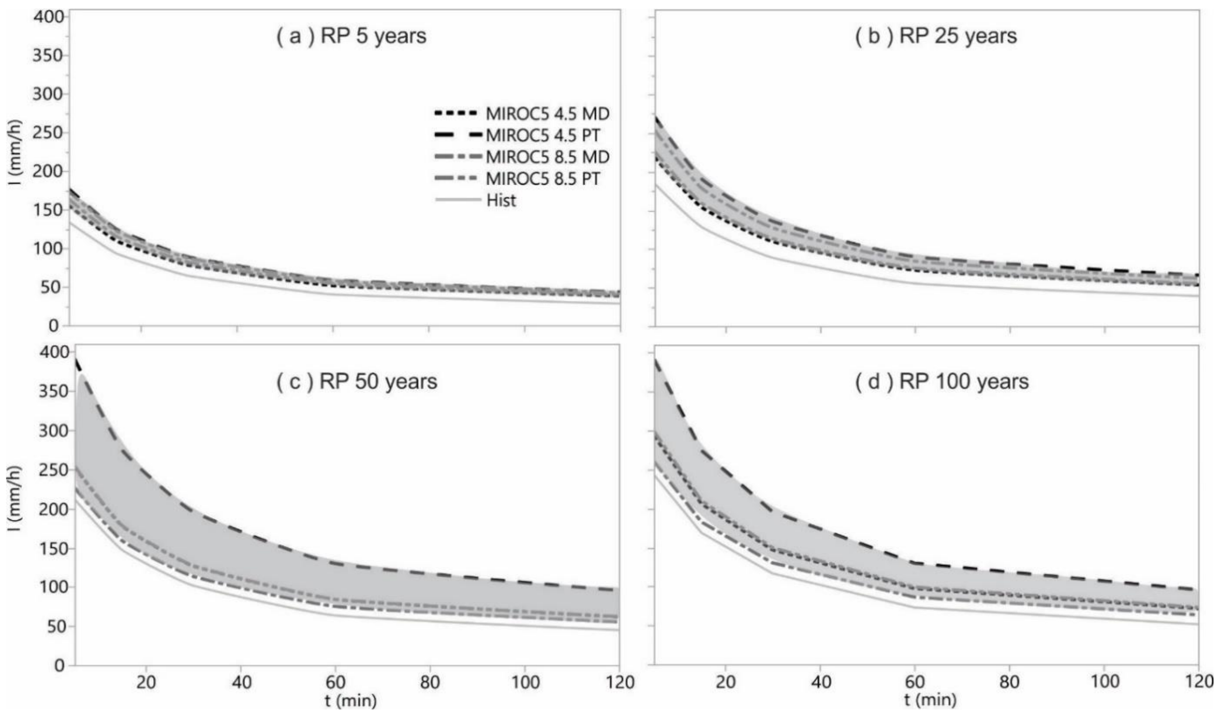

Figure 3.9 - Projected future IDF curves for the period 2050-2100 for the city of Sao Carlos – SP, Brazil and return periods of (a) 5 years, (b) 25 years, (c) 50 years and (d) 100 years.

Table 3.7 - Numerical values of the parameters of the IDF curves updated with climate change patterns, for the interval ranges analyzed

	Current	MIROC5 4.5 PT	MIROC5 4.5 MD	MIROC5 8.5 PT	MIROC5 8.5 MD
			2015 - 2050		
K	819.67	772.4	764.56	899.82	890.51
m	0.138	0.311	0.2956	0.2182	0.2176
t_0	10.77	12	12	12	12
n	0.75	0.764	0.764	0.764	0.764
			2050 - 2100		
K	819.67	1007.77	965.93	1036.49	1034.01
m	0.138	0.2645	0.2113	0.2356	0.2007
t_0	10.77	12	12	12	12
n	0.75	0.764	0.764	0.764	0.764

$$I = \frac{K \times TR^m}{(t + t0)^n}$$

2.3.3 Sensitivity analysis

The parameters and input variables evaluated in the sensitivity analysis, and their respective ranges are shown in Table 3.8. In total, 124417 combinations were evaluated, representing bioretention structures sized for three application scales considering the current period and future scenarios with modular design, i.e. increase of area per period.

To assist in the interpretation of the sensitivity analysis results and in the simulated hydrographs for the current and future scenarios, Figure 3.9 presents the values of Pearson's linear correlation coefficients (r) obtained between the parameters representing the drivers of change for urbanization (CN, Urbanization level - UL) and climate (RP, rainfall intensity - i_{rain}), application scale (catchment area - A_{cat}) and sized area (bioretention area - A_b), with the efficiencies of runoff retention (Eff_{rr}), peak flow attenuation (Eff_{peak}), time delay (Eff_{time}) and water reuse (Eff_{wr}), used as functions to evaluate the system's performance, in addition to their own efficiencies. As expected, it is possible to notice that the efficiencies are correlated with each other: Eff_{rr} and Eff_{peak} presenting the stronger correlation (r = 0.96), and lower correlation for Eff_{wr} with the others (ranging from 0.29 to 0.45).

Regarding the parameters representing the future drivers of change, none of them showed significant correlation with the measures of the bioretention performance (runoff retention efficiency, peak flow attenuation and water reuse efficiency), when evaluated individually. The influences of the drivers of change on the performance of adaptive drainage measures must occur jointly (due to synergistic effects), or non-linearly. An analysis of the influence of these parameters considering all their combinations is presented later. Finally, it was observed a positive correlation between the sized area, Eff_{rr} and Eff_{peak} (r = 0.48 and 0.51, respectively) and negative correlation with Eff_{wr} (r = -0.54). The greater the area of the bioretention, the greater the infiltrated volume, thus, the greater the runoff retention, also reducing overflow peaks. However, in relation to water reuse, the negative value of the correlation does not necessarily indicate a lower volume of water recovered by the underdrain, but rather a lower relationship between recovered volume and total inflow volume. This aspect will be further discussed in section 3.4.

Table 3.8 - Parameters and input values evaluated, and their respective base values and interval ranges analyzed

Parameter/input variable	Unit	Base values PS	Base values SS	Base values NS	Range
Catchment area	(m^2)	94	23000	47600	94, 23000, 47600
Method			Each method		Bioretention manual, BIRENICE, Envelope curve, WSUD manual
Urbanization period			Current		Current, 2015-2050, 2050-2100
IDF period			Current		Current, 2015-2050, 2050-2100
IDF coefficients			IDF current period		See table 5
Soil type			A		A, B, C, D
Urbanization level	(%)	100	30	22	22, 30, 50, 80, 100
Rain duration simulation	(min)		30		30, 90
Return period	(Yrs)		5		5, 10, 25, 50
$K_{sat,sim}$	(mm/h)		195		195, 468
$K_{sat,dim}$	(mm/h)		195		195, 468
Rain intensity	(mm/h)		63.44		Calculated according to IDF and rain duration
CN	(-)		CN current period soil A		See table 4
C	(-)	0.9	0.4	0.4	0.4, 0.6, 0.7, 0.8, 0.9
Rain distribution			Alt. blocks		Alt. blocks, Huff 1st quartil, Rational
P90	(mm)		32.5		Calculated according to daily rainfall in future climate projections

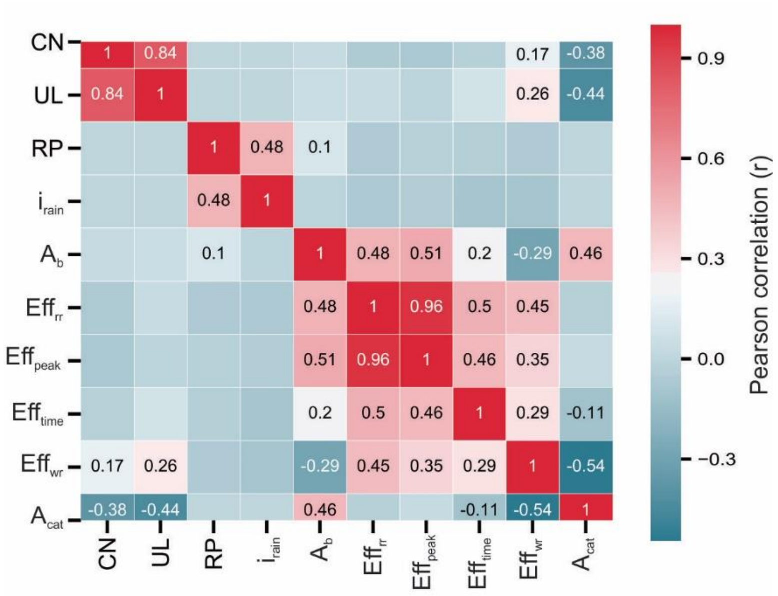

Figure 3.10 - Pearson correlation coefficients between parameters and input variables representing the drivers of change, sized areas, and evaluation functions for the performance of the bioretention.

For all combinations evaluated, Figures 3.10, 3.11 and 3.12 show diamond plots for the values of runoff retention efficiencies, peak flow attenuation, water reuse, time delay and sized area, respectively, grouped for each of the parameters or input variable analyzed. The sizing method presented the greatest variation in efficiency and in the sized areas. For the other parameters, little differences in the variation ranges were observed with come being more in the median.

Evaluating Eff_{rr}, Eff_{peak} and Eff_{wr} from Figure 3.10, the second parameter that causes more variations in the efficiencies was the design K_{sat} (represented in the "Infrastructure Aging Design" group). When adopting the value of $K_{sat,n}$ in the initial design of the bioretention structure, a reduction in the upper limit of the efficiency range was observed, as well as an even greater reduction in the median value. Higher K_{sat} values will require a smaller sized area, consequently resulting in lower efficiency values. In addition, during the aging of the structure, the K_{sat} value decreases, reducing infiltration and, consequently, the amount of runoff retained.

Other parameters with less influence are the catchment area, which will increase the lower limit of efficiencies range as the area increases by moving the median further down, due to the greater runoff generation. The soil type also causes a change in efficiency, so that soils with less infiltration capacity have lower central values. The lower infiltration also leads to greater runoff generation, consequently having more total inflow volume reaching the bioretention. Increasing the RP also reduces efficiencies, since higher RPs represent more extreme events, with greater transformation of rainfall into runoff. The future periods of IDF and urbanization have low impact in efficiencies, due to the capacity of modular design in compensating variations in efficiency over time (in Figure 3.12, the influence of these factors in the areas can be observed). Finally, the different rainfall durations chosen for the design storm applied in the simulations did not generate major changes in efficiency, and for a longer duration, there was even a greater median. Design storms with longer duration are less intense and have a more homogeneous temporal distribution, reducing the difference between the rainfall rate and the infiltration rate in the filtering media of the bioretention, allowing greater runoff retention over time.

In Figure 3.11, it is shown that other than the design method and temporal rainfall distribution, the variation of parameters did not have much influence on the Eff_{time} response, presenting itself as a more static value. The initial runoff retention is the main process involved in the time delay of the overflow peak. Both the initial runoff retention and time delay are mainly influenced by the bioretention surface area, which changes according to the design method. Once an overflow has occurred (regardless of its magnitude), the time for its occurrence is similar. In addition, it is possible that the little variation in the rainfall duration analyzed in this study does not allow the observance of greater differences in the time delay.

The sizing method, the future periods (both urbanization and climate), the urbanization level, the catchment area, the RP and K_{sat} (less important) are the parameters that have greater influence on the sized area (Figure 3.12). The future period of urbanization is the factor with the greatest influence because it is the one with the greatest capacity of increasing the runoff generation. The soil type, on the other hand, had little influence on the sized areas, because there is a proportional increase in runoff generation due to the increase in CN for pre-urbanization and urbanized catchments.

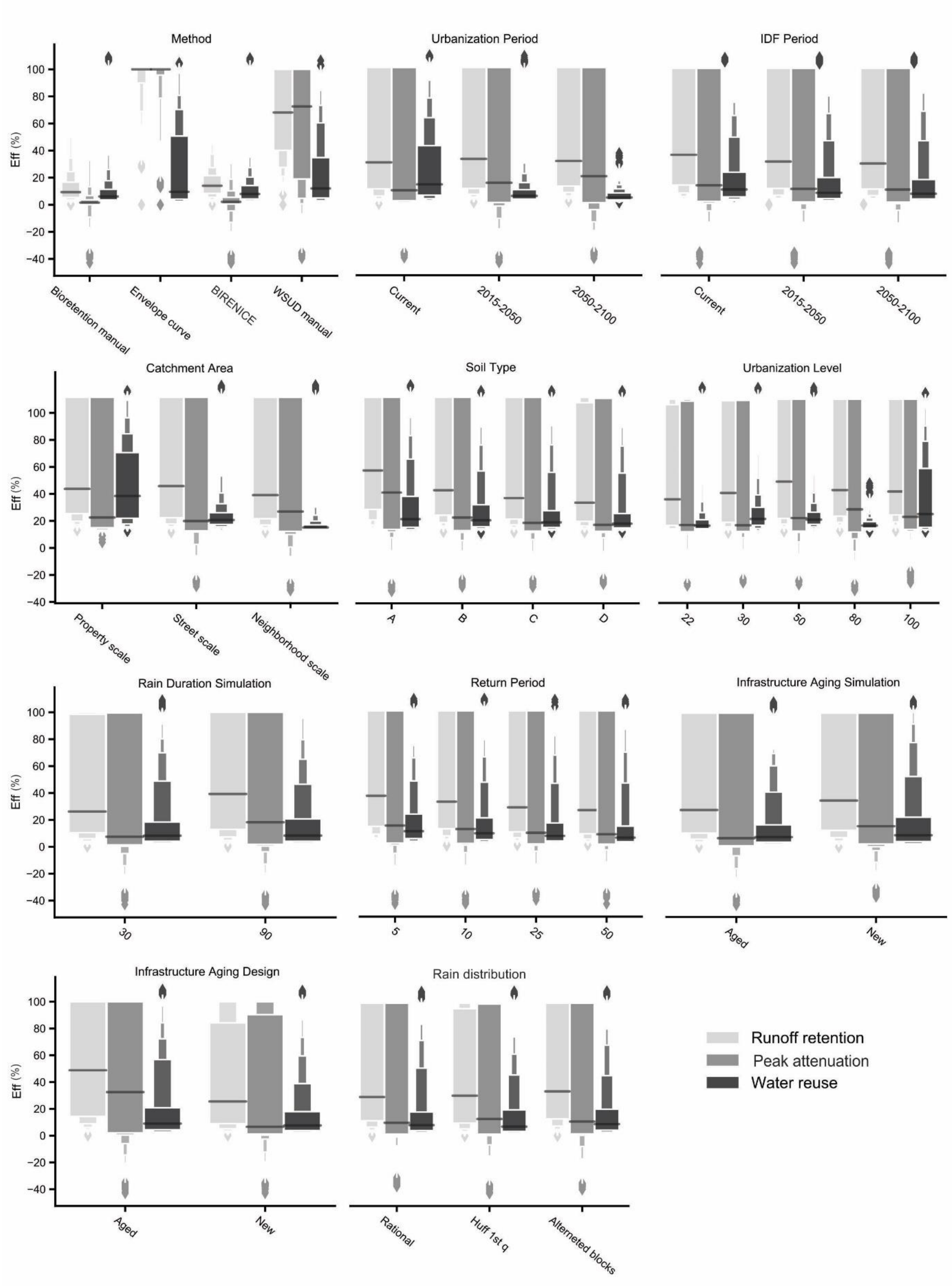

Figure 3.11 - Diamond plot for runoff retention, peak flow attenuation and water reuse efficiency, different parameters and input variables

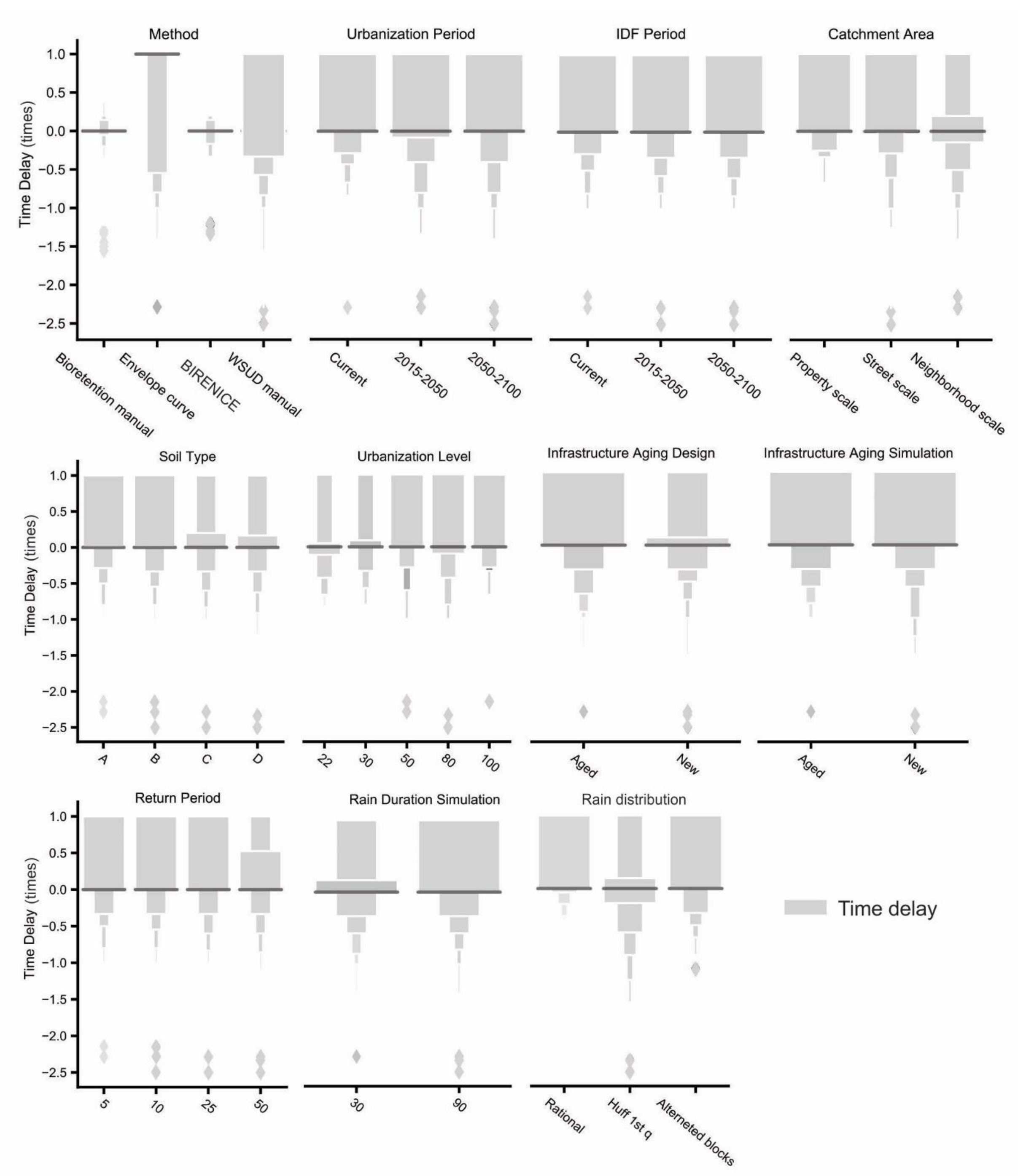

Figure 3.12 - Diamond plot for time delay efficiency, different parameters and input variables

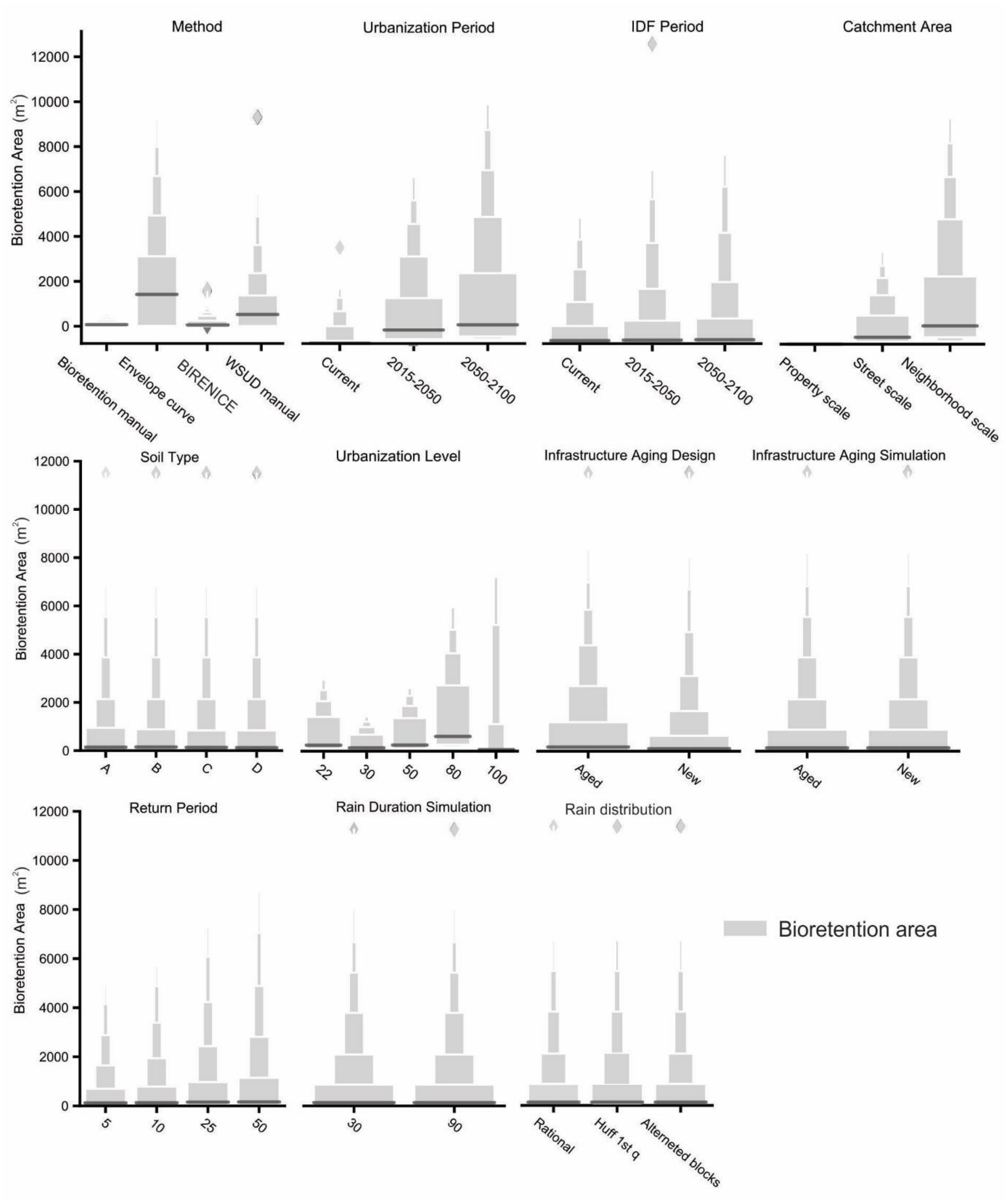

Figure 3.13 - Diamond plot for sized areas, different parameters and input variables

Since the sizing method is one of the factors that most affect the surface area and the final bioretention performance, a one-at-time sensitivity analysis was performed for each method separately, according to the methodology proposed by Morris (1991) and Campolongo et al. (2007).

Figures 3.13, 3.14, 3.15 and 3.16 show the result of the sensitivity analysis one-at-time, for the methods Bioretention manual, BIRENICE, Envelope curve and WSUD manual, respectively. In general, the methods present greater sensitivity to the same parameters, with the main difference occurring in the magnitude of the sensitivity, both in the mean and in the standard deviation. For Eff_{rr} and Eff_{peak}, the parameters with the highest sensitivity are the runoff coefficient (C) and daily precipitation with 90% probability (P90).

The C represents the catchment imperviousness level, which is related to the level of urbanization and, consequently, to the urbanization period. With exception of BIRENICE, it was observed a sensitivity to the urbanization period for all methods, with less important effects in the Bioretention manual for the NS. In all cases, C did not show high σ values, which indicates few higher-order effects. P90 was the parameter with the greatest sensitivity, with higher values for NS. It also presented the highest σ values for all methods (around 1600) indicating strong higher-order effects that may be explained by its interaction with other parameters, such as the future climate period and rainfall intensity.

For the Eff_{wr} of water reuse, in general, there is little sensitivity to the parameters since it will be more influenced by the infiltration rate and size of the underdrain (further discussion in section 3.3.4). Eff_{time} is also not very sensitive to parameters in general (as previously discussed).

The main differences between the design methods sensitivity can be observed in the sized area. For the Bioretention manual and BIRENICE methods, the sized areas are more sensitive to C (μ >200) and to the urbanization period (μ> 50). These two parameters are related with each other, leading to high σ values for C, in both methods. For the Envelope curve and WSUD manual methods, the greatest sensitivity is observed to C (μ >1000), followed by P90 (μ > 500), and subsequently the urbanization period (μ > 200) and urbanization level (μ > 200, for property scale). Higher-order effects are also more significant for C and P90.

The most sensitive parameters are related to the drivers of urbanization and climate change, such as total rainfall volume and runoff coefficient. Both parameters are directly associated with the total runoff volume generated and reaching the bioretention structure, i.e. the design is much more influenced by the total volume of runoff generated, than how it occurs over time.

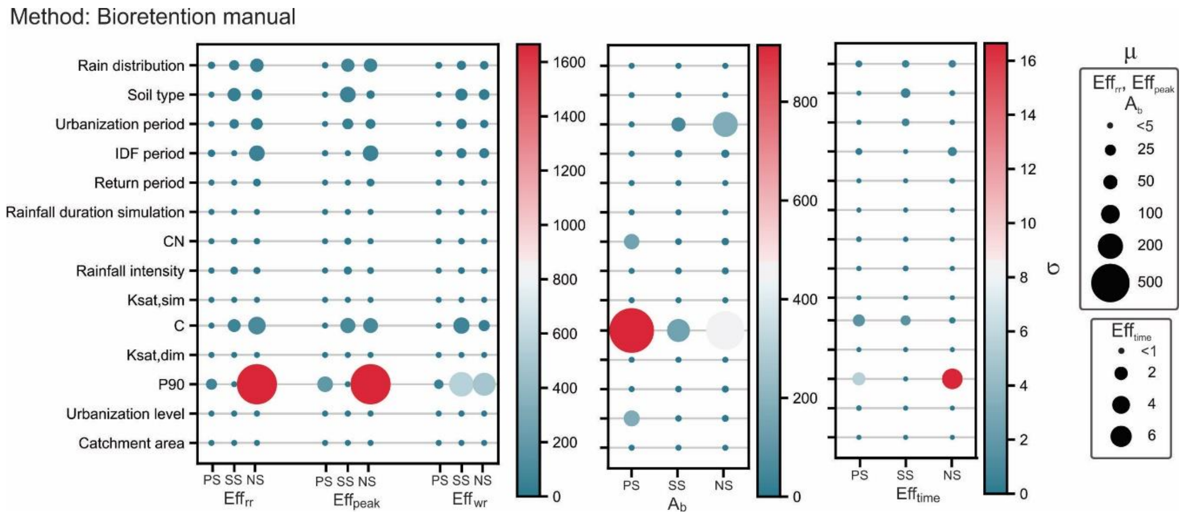

Figure 3.14 - Sensitivity analysis of Bioretention manual method

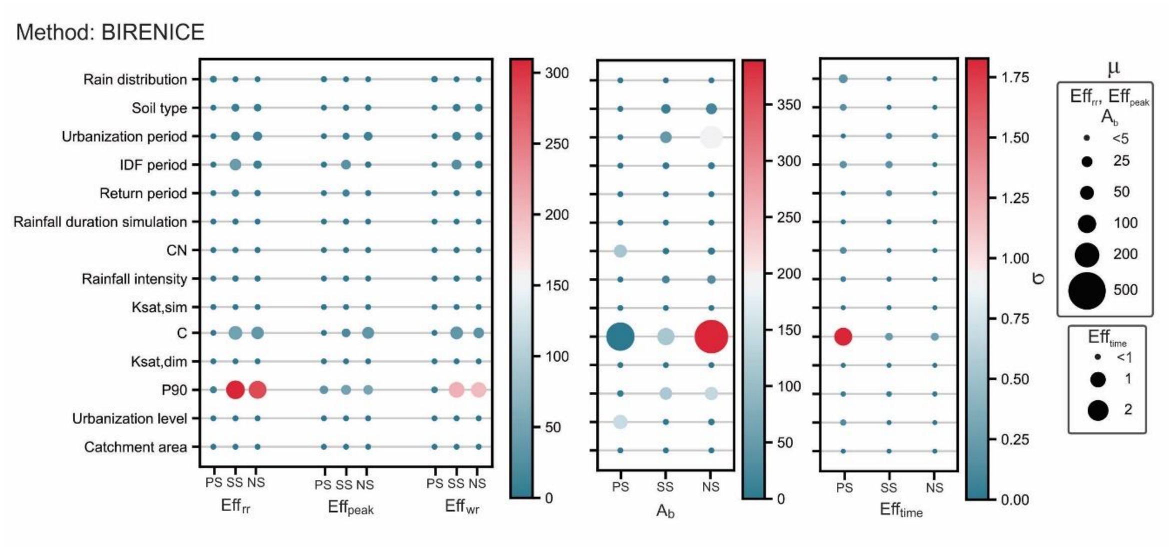

Figure 3.15 - Sensitivity analysis of BIRENICE method

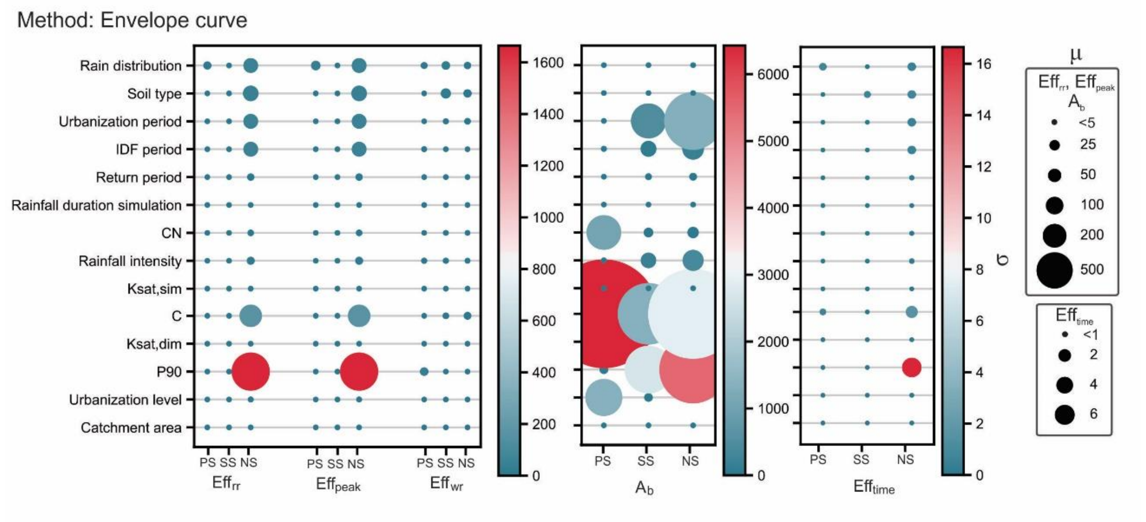

Figure 3.16 - Sensitivity analysis of Envelope curve method

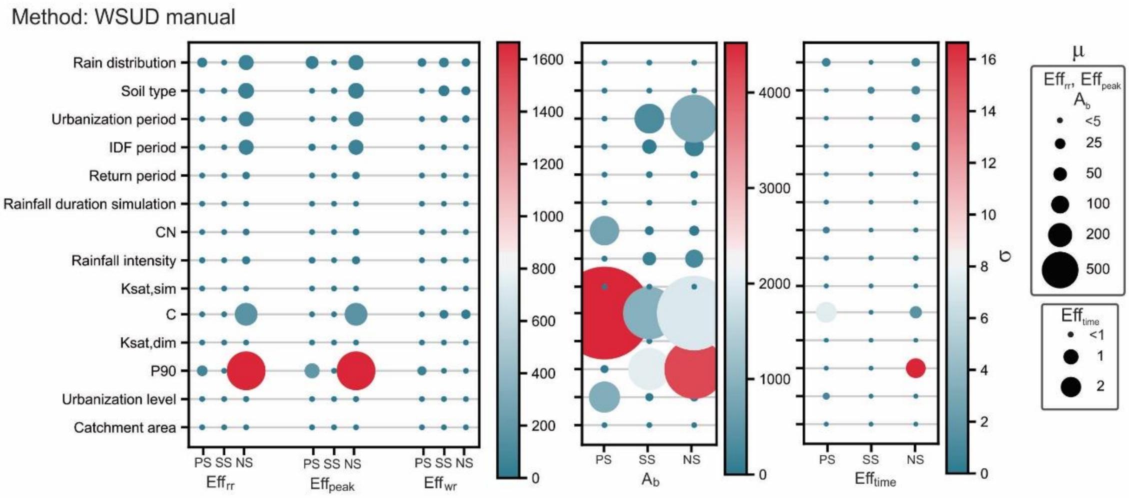

Figure 3.17 - Sensitivity analysis of WSUD manual method

3.2.3.4 Impacts of the changes in the catchment hydrological behavior and in bioretention performance

Figures 3.17, 3.18 and 3.19 show the simulated hydrographs for the different future scenarios and the combination of drivers of change for the three application sites evaluated. The hydrographs of all combinations of parameters and input variables are presented, generating a range of variability, as proposed by Willems and Vrac (2011). In the PS scale (Figure 3.17) there is no difference in hydrographs for the urbanization periods, as it is a consolidated area (no variation in land use). However, climate changes may even double the inflow peak (when considering larger RPs).

When comparing to PS, for SS and NS (Figures 3.18 and 3.19) urbanization has greater influences in the hydrographs when analyzing its effect alone. For SS, the inflow can increase by 1.5x, while in NS, this value increases by up to 3x, for more advanced future scenarios. For these application scales, climate change also has a great influence, increasing inflow up to 4x. Dudula and Randhir (2016), Ghazal et al. (2014), also observed significant effects of climate change on the catchment hydrology, requiring a more careful design that is able to consider this driver of change. Hathaway et al. (2014), evaluated the performance of bioretention structures under future climate change scenarios in North Carolina and noticed the need for increasing storage capacity to maintain efficiency, which in this study it was incorporated through modular design. However, studies by Liu et al. (2016), Liu et al. (2017) and Pyke et al. (2011) jointly evaluated the effects of urbanization and climate change and obtained urbanization as a more important factor, which, in a first moment, may seem contradictory to this study.

However, the analysis of future scenarios in this study presents a zone of variability, considering different factors, including RP (which varies from 5 to 50 years). The MIROC5 model shows a great increase in extreme events in the city of Sao Carlos, which can also be seen from the analysis of the new IDFs built for the city, in which the same event with a RP in the current scenario of 50 years, reduced to 8.8 and 4.7 years for the future periods of 2015-2050 and 2050-2100, respectively. Therefore, the importance of climate change is much more pronounced for greater RP. In addition, climate change can have different effects for each region, and it may be that this greater contribution of climate change to this study area is due to local characteristics, which are not repeated in other studies.

The external limit of the variability range represents the most extreme events (which tend to become even more extreme) and the internal limit are related to more recurring RPs (5 to 10 years), which are the most frequently used in the design of adaptive urban drainage systems. The studies by Liu et al. (2016) and Liu et al. (2017) Pyke et al. (2011), used rainfall events with greater recurrence (daily rainfall over a period of 30 years), which are comparable with the hydrograph internal limits. Thus, there is no disagreement in the results when analyzing the hydrograph simulated for more recurrent events (lower RP) in this study.

Regarding the efficiencies, for SS and NS, when the bioretention is designed only for future climate changes, there is a drop in the upper limit of efficiencies range. This probably happens because the design is more sensitive to the factors related to the urbanization level then the climate periods. When considering only the last it results in smaller sized areas, which leads in less performance. In PS, as there are no changes in the runoff coefficients (C and CN), the drop of upper limit in efficiency does not occur.

The different design methods resulted in considerable differences in the sized areas, which probably is the most important factor for the large variability range presented in the hydrographs. There are 100% Eff_{rr} even for the most advanced scenarios in the future, which happened predominantly for the design with the Envelope curve method (which tends to over-design the areas). When regarding the sized areas at NS for the Envelope curve method there is a variation 6.1 to 35.6% of the impervious catchment area, depending on the future scenarios evaluated. The implementation of bioretention with these great sizes may not be viable due to both availability of land and costs of implementation and maintenance. Therefore, it is up to the decision maker to balance between the required efficiencies and the available resources to decide which alternative is the best. The design methods Bioretention manual and BIRENICE are the ones that result in smaller sized areas (varying from 0.3 to 1.8% and 0.1 to 3.4% of the impervious catchment area, respectively), however, they are the ones with worst Eff_{rr} and Eff_{peak}. The method WSUD manual has good efficiency values and has a range of sized areas from 1.5 to 25.4% of the impervious catchment area.

The areas sized and required by the design methods that resulted in the highest efficiencies (Envelope curve and WSUD manual) are very large and usually unavailable in the consolidated urban areas. With this in mind, it is important to emphasize the importance of advance urban planning, reserving areas for future interventions, e.g. for drainage infrastructure. These areas can be thought of for a dual purpose and types of use, such as green and entertainment areas during the dry periods and for flood control through Nature-based Solutions, during the rainfall events. When the catchment already presents a consolidated urbanization and there are no more vacant areas to implement the total amount of bioretention area required from the design methods, the design must be adapted so that as much of the existing areas as possible can be used for intervention. Even without achieving the desired project efficiencies, any intervention aimed at amortizing the flood peaks and the flow volumes already contributes to the flood control and increase the local resilience.

As areas dimensionadas pelos métodos que resultaram em maiores eficiências (Envelope curve e WSUD manual) são muito grandes e normalmente indisponíveis nas áreas urbanas já consolidadas. Tendo isso em vista, é importante ressaltar a importância do planejamento urbano antecipado, se reservando áreas para futuras intervenções necessárias, neste caso, para obras de drenagem. Essas áreas podem ser pensadas a partir de um duplo propósito e tipos de uso, como por exemplo áreas verdes de lazer quando em períodos secos e antes da necessidade de sua utilização para as obras de drenagem, e para amortecimento de cheias a partir de soluções baseadas na natureza, em períodos de cheia. Quando a urbanizaçao da área já estiver consolidada e não haver mais áreas livres para implantação do total de bioretention area required from the design methods, deve-se adaptar o design para que se possa utilizar o máximo possível das áreas existentes para intervenção. Mesmo sem atingir as eficiencias de projeto desejadas, qualquer intervenção visando amortização dos picos de cheia e dos volumes de escoamento gerado já contribuem para o controle de cheias.

Many manuals recommend bioretention surface areas around 1 to 5% of the impervious catchment area (The Prince Geroge's County, 2007; Waterways, 2005; COUNCIL, 2007). On the other hand, Dussailant et al. (2004) obtained that for rain gardens optimal exfiltration and groundwater recharge occur in rates between the structure area and the directly connected impermeable area from 10 to 20%. However, from the results obtained it is possible to see that this metric does not take into account future urbanization and climate scenarios and may lead to areas that are smaller than necessary to good performances.

Regarding the outflow (the volume recovered by the underdrain to be reused) a softer release of the retained volume over time is observed, with smaller peaks and with longer duration. Regarding the performance of water recovery, there is a reduction in Eff_{wr} for larger scales of application and future scenarios (i.e. grater runoff generation). However, it does not indicate a reduction in the volume of water recovered, but rather in the ratio between recovered volume and inflow volume.

For PS, in the current urbanization scenario and for soil type D, there is a volume of $1.5m^3$ of recovered water. This volume increases to $82m^3$ when analyzing the same conditions for SS, which represents a significant increase due to the bigger catchment area. However, for NS (twice the SS area), the recovered volume remains constant at $82m^3$. Assessing the different future scenarios for both SS and NS, it is possible to notice that the recovered volumes always remain within a range of 82 to $85m^3$ (for the SS – 2.3ha and NS – 4.8 ha), i.e. there is a moment when the recovered volume becomes practically constant. This volume is controlled mainly by the infiltration capacity of the filtering media and secondly by the underdrain capacity, which does not depend on factors related to the catchment area. Therefore, once the maximum capacity is reached there is no major changes in recovered volume.

For future studies it is recommended to evaluate the change in the size of the underdrain pipe, in order to increase the efficiency of water recovery (since Eff_{wr} was not very sensitive to the filtering media hydraulic conductivity). Although Eff_{wr} has not increased for the larger application sites, the volume of recovered water can contribute to the resilience of communities to water scarcity during drought periods, if these systems are integrated with stormwater harvesting techniques. Thus, thinking about means to increase and maintain Eff_{wr} for larger areas and future scenarios, it can further contribute even more to the resilience of communities to drought.

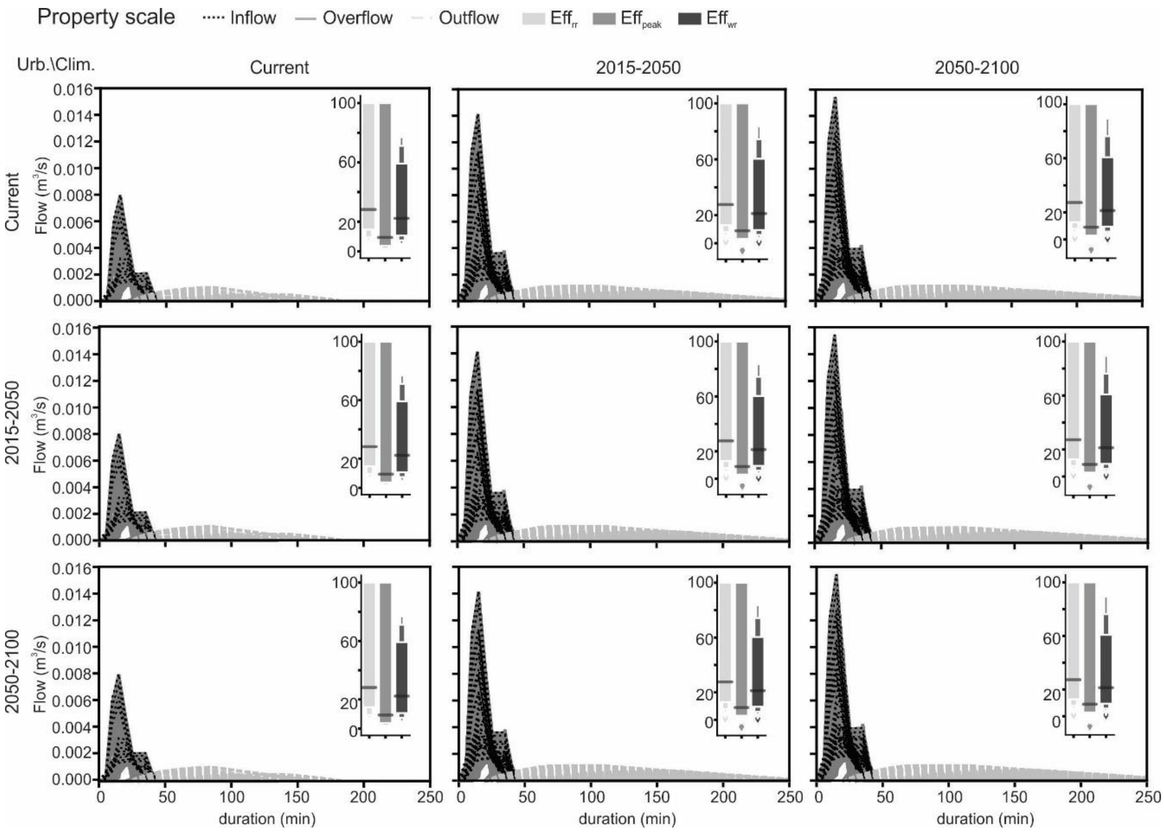

Figure 3.18 - Bioretention hydrographs and performance for future scenarios of climate and urbanization for property scale

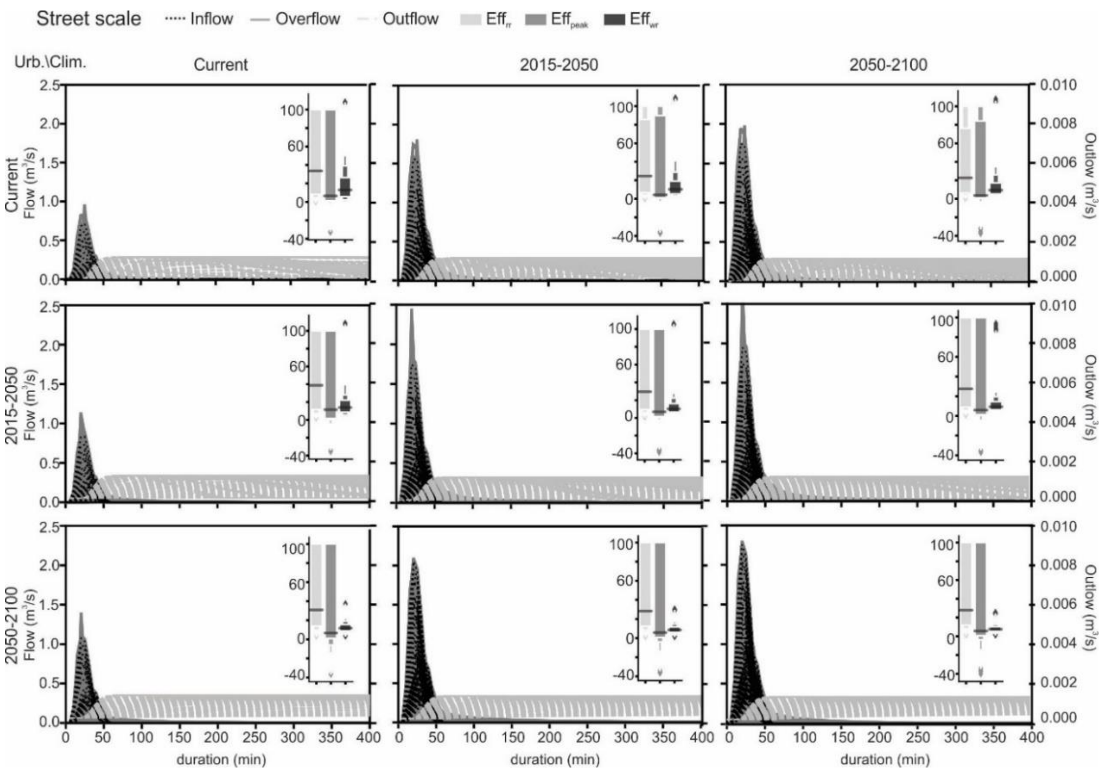

Figure 3.19 - Bioretention hydrographs and performance for future scenarios of climate and urbanization for street scale

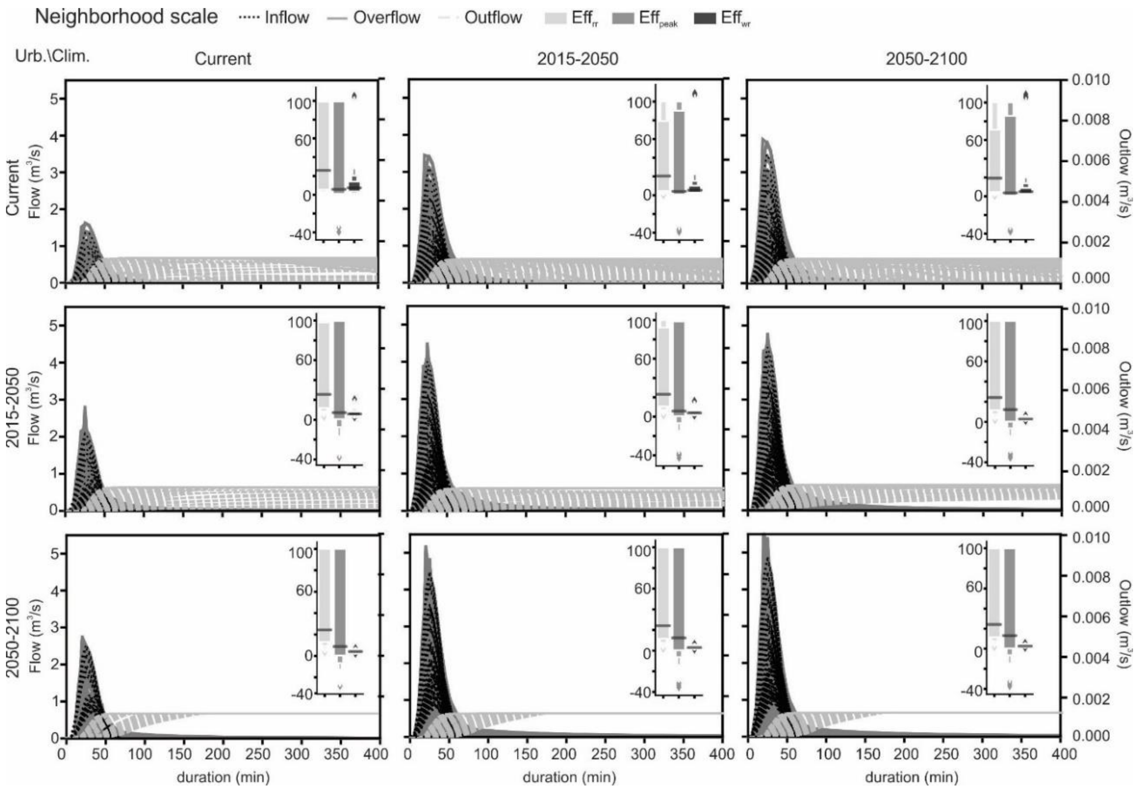

Figure 3.20 - Bioretention hydrographs and performance for future scenarios of climate and urbanization for neighborhood scale

3.4 Conclusion

Given the intensification of impervious areas and climate changes, floods are expected to increase in the coming years if adaptive measures are not taken, increasing the risk for the population. From an initial planning that incorporates these drivers of change from the design of adaptive measures (as LID practices), it is possible to increase the efficiency of these structures and reduce the risks caused by flooding to the population, and even increase the resilience to droughts (by water recuperation). This study presented a methodology for incorporating the drivers of change from the planning, performed a sensitivity analysis to help identify the main parameters that cause uncertainties in determining the performance of the systems, requiring more attention. It was also adopted the modulation design as an alternative to amortize the construction and operational costs for the adaptive urban drainage structures designed for future scenarios. As main conclusions, it was obtained:

- Changes in climate patterns can be estimated from using GCMs or RCMs, to the application site. Even with bias correction, these models still have a lot of uncertainty, and it should be considered as ranges of possibilities in the calculations. The final choice of which value to adopt within the uncertainty range is based on restrictive project criteria.

- The area has a positive correlation with the efficiencies of runoff retention (r = 0.48) and peak flow attenuation (r = 0.51).
- Evaluating all the combinations between design methods, parameters, and input variables, it was possible to observe that the design method and the structure aging are the parameters that most affect the performance of bioretention systems.
- The parameter related to the climate that most affect the efficiency of the bioretention is the Return Period. The parameters and input variables related to urbanization that most affect efficiency are the soil type and the runoff coefficient.
- When all methods and parameters are evaluated together, future periods of urbanization and climate, the urbanization level and the return period are those which most affect the sized area. Due to their influence in the area, when incorporating modular design, efficiency is little affected in future periods.
- For the one-at-time sensitivity analysis, similarity was noted in the sensitivity of all design methods, with the most sensitive parameters being the runoff coefficient and P90, with different magnitudes for the different methods. The P90 has more high-order effects.
- After reaching the limit of the underdrain or the infiltration capacity, the recovered volume remains constant. Future studies about modulation should consider the water recovery capacity, which may increase the efficiency of water reuse over time.
- There are indications that modulation helps to prevent loss of efficiency over time, helping to maintain runoff retention and peak flow attenuation efficiencies along time.

In this study, we have done a general assessment of the different pre-design methods for bioretention considering the uncertainties that the future scenarios bring. The results shows that the design method adopted is the main factor that affects the efficiency of the device and the surface area required. There are some conflicts in choosing what method should be employed, since the methods that generates bioretention more efficient also requires more surface area, leading to greater costs and the need of available area in the catchment, which are generally scarce in the urban environment. Therefore, we recommend that future studies assess the long term efficiency of bioretention devices and validate the different design methods and the project efficiency, allowing to provide more assertive recommendation about the methods more suitable to the Brazilian context of subtropical climate and flash floods.

References

Abreu, F. G. (2019). *Quantificação dos Prejuízos Econômicos à Atividade Comercial Derivados de Inundações Urbanas* [Quantification of Economic Losses to Commercial Activity Derived from Urban Flooding]. Thesis. Escola de Engenharia de São Carlos, Universidade de São Paulo, São Carlos. (In Portuguese)

Barbassa, A. P. (1991). *Simulação do efeito da urbanização sobre a drenagem pluvial da cidade de São Carlos – SP* [Simulation of the effect of urbanization on the storm drainage of the city of São Carlos - SP]. Thesis. Escola de Engenharia de São Carlos, Universidade de São Paulo. São Carlos. (In Portuguese)

C40. (2014), *Climate Action in Megacities: C40 Cities Baseline and Opportunities.* Volume 2.0. Feb, 2014. Access in: 21 out 2016. Available in: <http://issuu.com/c40cities/docs/c40_climate_action_in_megacities/149?e=10643095/6541335> .

Campolongo, F., Cariboni, J., & Saltelli, A. (2007). An effective screening design for sensitivity analysis of large models. *Environmental modelling & software*, 22(10), 1509-1518.

Canholi, A. P. (2014). *Drenagem Urbana e Controle de Enchentes* [Urban Drainage and Floods Control]. 2nd edition, Oficina de Textos, São Paulo. (in Poruguese)

Cheng, L., & AghaKouchak, A. (2014). Nonstationary precipitation intensity-duration-frequency curves for infrastructure design in a changing climate. *Scientific reports*, 4(1), 1-6.

Chou, S. C., Lyra, A., Mourão, C., Dereczynski, C., Pilotto, I., Gomes, J., ... & Campos, D. (2014). Assessment of climate change over South America under RCP 4.5 and 8.5 downscaling scenarios. *American Journal of Climate Change*, 3(5), 512-525.

COUNCIL, Gold Coast City. (2007). *Water Sensitive Urban Design Guidelines*. Gold Coast City Council, Queensland.

De Paola, F., Galdiero, E., Giugni, M., & Pugliese, F. (2015). Sustainable development of storm-water systems in African cities considering climate change. *Procedia Engineering*, 119, 1181-1191.

Debortoli, N. S., Camarinha, P. I. M., Marengo, J. A., & Rodrigues, R. R. (2017). An index of Brazil's vulnerability to expected increases in natural flash flooding and landslide disasters in the context of climate change. *Natural hazards*, 86(2), 557-582.

DeGaetano, A. T., & Castellano, C. M. (2017). Future projections of extreme precipitation intensity-duration-frequency curves for climate adaptation planning in New York State. *Climate Services*, 5, 23-35.

Dudula, J., & Randhir, T. O. (2016). Modelling the influence of climate change on watershed systems: Adaptation through targeted practices. *Journal of Hydrology*, 541, 703-713.

Dussaillant, A. R., Wu, C. H., & Potter, K. W. (2004). Richards equation model of a rain garden. Journal of Hydrologic Engineering, 9(3), 219-225.

Fava, M. C., Abe, N., Restrepo-Estrada, C., Kimura, B. Y., & Mendiondo, E. M. (2018). Flood modelling using synthesised citizen science urban streamflow observations. *Journal of Flood Risk Management*. 12(S2), e12498.

Ghazal, R., Ardeshir, A., & Rad, I. Z. (2014). Climate change and stormwater management strategies in Tehran. *Procedia Engineering*, 89, 780-787.

Gomes Junior, M. N. (2019). *Aspectos Hidrológicos-Hidráulicos e Avaliação da Eficiência de Biorretenções: Modelos, Princípios e Critérios de Projeto de Técnicas Compensatórias de 3ª Geração* [Hydrological-Hydraulic Aspects and Evaluation of Bioretention Efficiency: Models, Principles and Design Criteria for 3rd Generation LID]. Thesis. Escola de Engenharia de São Carlos, Universidade de São Paulo, São Carlos. (in Portuguese)

Hathaway, J. M., Brown, R. A., Fu, J. S., & Hunt, W. F. (2014). Bioretention function under climate change scenarios in North Carolina, USA. *Journal of Hydrology*, 519, 503-511.

Houston, D., Werritty, A., Bassett, D., (2011). *Pluvial (rain-related) Flooding in Urban Areas: the Invisible Hazard*. Joseph Rowntree Foundation. Available in: <Retrieved from: http://www.jrf.org.uk/sites/files/jrf/urban-flood-risk-full.pdf>. Access in: 10 mar 2019.

IPCC. (2007). *Contribution of working groups I, II and III to the fourth assessment report of the intergovernmental panel on climate change*. Geneva, Switzerland, p. 104.

Konrad, C. P., Booth, D. B. (2005) Hydrologic changes in urban streams and their Ecological significance. *American Fisheries Society Symposium*. 47, 157–177.

Koutsoyiannis, D., Kozonis, D., & Manetas, A. (1998). A mathematical framework for studying rainfall intensity-duration-frequency relationships. *Journal of Hydrology*, 206(1-2), 118-135.

Lehmann, E. A., Phatak, A., Stephenson, A., & Lau, R. (2016). Spatial modelling framework for the characterisation of rainfall extremes at different durations and under climate change. *Environmetrics*. 27(4), 239-251.

Leopold, L.B. (1968) *Hydrology for urban land planning: A guidebook on the hydrological effects of urban land use*. Circ. No. 554. U.S. Geological Survey, Washington, DC.

Levine, D. M., Berenson, M. L., Stephan, D. (2008). *Estatística - Teoria e Aplicações* [Statistics - Theory and Applications]. 5. ed. Rio de Janeiro: LTC. (in Portuguese)

Lima, C. H., Kwon, H. H., & Kim, Y. T. (2018). A local-regional scaling-invariant Bayesian GEV model for estimating rainfall IDF curves in a future climate. *Journal of Hydrology*, 566, 73-88.

Liu, Y., Engel, B. A., Collingsworth, P. D., & Pijanowski, B. C. (2017). Optimal implementation of green infrastructure practices to minimize influences of land use change and climate change on hydrology and water quality: case study in Spy Run Creek watershed, Indiana. *Science of the Total Environment*, 601, 1400-1411.

Liu, Y., Theller, L. O., Pijanowski, B. C., & Engel, B. A. (2016). Optimal selection and placement of green infrastructure to reduce impacts of land use change and climate change on hydrology and water quality: An application to the Trail Creek Watershed, Indiana. *Science of the Total Environment*, 553, 149-163.

Lyra, A., Tavares, P., Chou, S. C., Sueiro, G., Dereczynski, C., Sondermann, M., ... & Giarolla, A. (2018). Climate change projections over three metropolitan regions in Southeast Brazil using the non-hydrostatic Eta regional climate model at 5-km resolution. *Theoretical and applied climatology*, 132(1-2), 663-682.

Madsen, H., Arnbjerg-Nielsen, K., & Mikkelsen, P. S. (2009). Update of regional intensity–duration–frequency curves in Denmark: tendency towards increased storm intensities. *Atmospheric Research*, 92(3), 343-349.

Madsen, H., Lawrence, D., Lang, M., Martinkova, M., & Kjeldsen, T. R. (2014). Review of trend analysis and climate change projections of extreme precipitation and floods in Europe. *Journal of Hydrology*, 519, 3634-3650.

Mailhot, A., Duchesne, S. (2009). Design criteria of urban drainage infrastructures under climate change. *Journal of Water Resources Planning and Management*, 136(2), 201-208.

Marengo, J. A.; Schaeffer, R.; Zee, D.; Pinto, H. S. (2010). *Mudanças climáticas e eventos extremos no Brasil* [Climate change and extreme events in Brazil]. Available in: <http://www. fbds. org. br/cop15/FBDS_MudancasClimaticas.pdf.> Access in: 10 oct 2010. (in Portuguese)

Marsalek, J.; Schreier, H. (2009). Innovation in Stormwater Management in Canada: The Way Forward, *Water Quality Resources*, 44(1), v-x.

McAuley, A. (2009). *Technical Design Guidelines.* Northern Territory Department of Planning and Infrastructure.

Moriasi, D. N., Arnold, J. G., Van Liew, M. W., Bingner, R. L., Harmel, R. D., & Veith, T. L. (2007). Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. *Transactions of the ASABE*, 50(3), 885-900.

Morris, M. D. (1991). Factorial sampling plans for preliminary computational experiments. *Technometrics*, 33(2), 161-174.

Muller, A., Bacro, J. N., & Lang, M. (2008). Bayesian comparison of different rainfall depth–duration–frequency relationships. *Stochastic Environmental Research and Risk Assessment*, 22(1), 33-46.

PMSC – Prefeitura Municipal de São Carlos. (2016). *Plano Diretor do Município de São Carlos. Lei no 18.053, de 19 de dezembro de 2016* [Master Plan of the Municipality of São Carlos. Law 18.053 of December 19, 2016]. Available in: < https://leismunicipais.com.br/plano-diretor-sao-carlos-sp>. Access in: 11 sep 2019. (in Portuguese)

Pyke, C., Warren, M. P., Johnson, T., LaGro Jr, J., Scharfenberg, J., Groth, P., ... & Main, E. (2011). Assessment of low impact development for managing stormwater with changing precipitation due to climate change. *Landscape and Urban Planning*, 103(2), 166-173.

Randelovic, A., Zhang, K., Jacimovic, N., McCarthy, D., & Deletic, A. (2016). Stormwater biofilter treatment model (MPiRe) for selected micro-pollutants. *Water research*, 89, 180-191.

Rosa, A. (2016). *Bioretention for diffuse pollution control in SUDS using experimental-adaptive approaches of ecohydrology*. Thesis. Escola de Engenharia de São Carlos, Universidade de São Paulo, São Carlos.

Santos, R. F. (2007). *Vulnerabilidade ambiental: desastres naturais ou fenômenos induzidos?* [Environmental vulnerability: natural disasters or induced phenomena?]. MMA (in Portuguese).

Semadeni-Davies, A., Hernebring, C., Svensson, G., & Gustafsson, L. G. (2008). The impacts of climate change and urbanisation on drainage in Helsingborg, Sweden: Suburban stormwater. *Journal of Hydrology*, 350(1-2), 114-125.

Shen, P., Deletic, A., Urich, C., Chandrasena, G. I., & McCarthy, D. T. (2018). Stormwater biofilter treatment model for faecal microorganisms. *Science of the total environment*, 630, 992-1002.

Silveira, A. L. L.; Goldenfum, J. A. (2007). Metodologia Generalizada para Pré-Dimensionamento de Dispositivos de Controle Pluvial na Fonte [Generalized Methodology for Pre-Dimensioning of Rainwater Control Devices at Source]. *RBRH*, 12(2), 57-168. (in Portuguese)

Song, X., Zhang, J., Zhan, C., Xuan, Y., Ye, M., & Xu, C. (2015). Global sensitivity analysis in hydrological modeling: Review of concepts, methods, theoretical framework, and applications. *Journal of hydrology*, 523, 739-757.

Soro, G.E., Goula-Bi, T.A., Kouassi, F.W., Srohourou, B. (2010). Update of intensity duration frequency curves for precipitation of short durations in tropical area of West Africa (Côte D'ivoire). *Journal of Applied Sciences*. 10 (9), 704–715.

Teutschbein, C., Seibert, J. (2012). Bias correction of regional climate model simulations for hydrological climate-change impact studies: Review and evaluation of different methods. *Journal of Hydrology*, 456, 12-29.

The Prince George's County (2007). *Bioretention Manual*. Environmental Services Division, Department of Environmental Resources. Available in: <https://www.ct.gov/deep/lib/deep/p2/raingardens/bioretention_manual_2009_version.pdf>. Access in: 25 sep 2018.

Urbonas, B., Stahre, P. (1993). *Stormwater: Best Management Practices and Detention for Water Quality, Drainage, and CSO Management*. Prentice Hall.

Wang, D., Hagen, S. C., Alizad, K. (2013). Climate change impact and uncertainty analysis of extreme rainfall events in the Apalachicola River basin, Florida. *Journal of Hydrology*, 480, 125-135.

Waterways, M. B. (2005). *Water sensitive urban design: technical design guidelines for South East Queensland.* Moreton Bay Waterways and Catchment Partnership. Available in: < https://www.brisbane.qld.gov.au/planning-and-building/planning-guidelines-and-tools/brisbane-city-plan-2014/superseded-brisbane-city-plan-2000/superseded-subdivision-and-development-guidelines/water-sensitive-urban-design/engineering-guidelines-superseded>. Access in: 28 aug 2020.

Weatherbase. (2018). *Climate summary*. Availabe in: <http://www.weatherbase.com>. Access in: 04 may 2018.

Willems, P. (2013). Revision of urban drainage design rules after assessment of climate change impacts on precipitation extremes at Uccle, Belgium. *Journal of Hydrology*, 496, 166-177.

Willems, P., & Vrac, M. (2011). Statistical precipitation downscaling for small-scale hydrological impact investigations of climate change. *Journal of Hydrology*, 402(3-4), 193-205.

Wong, T.H.F., Eadie, M.L. (2000) Water sensitive urban design—a paradigm shift in urban design. *Proceedings of the Xth World Water Congress*, 12:16. http://gabeira.locaweb.com.br/cidadesustentavel/biblioteca/%7B30788FE6-98A8-44E5-861E-996D286A78B3%7D_Wong1.pdf.

Young, C. E. F., Aguiar, C., Souza, E., 2015. *Valorando tempestades: custo econômico dos eventos climáticos extremos no Brasil nos anos de 2002 – 2012* [Assessing storms: economic cost of extreme weather events in Brazil in the years 2002 – 2012], first ed. Observatório do clima, São Paulo (in Portuguese).

4 STORMWATER VOLUME REDUCTION AND WATER QUALITY IMPROVEMENT BY BIORETENTION: POTENTIALS AND CHALLENGES FOR WATER SECURITY IN A SUBTROPICAL CATCHMENT

This chapter is published as: Marina Batalini de MACEDO, Cesar Ambrogi Ferraira do LAGO, Eduardo Mario MENDIONDO. **Stormwater volume reduction and water quality improvement by bioretention: potentials and challenges for water security in a subtropical catchment**. Science of the Total Environment, 647, 923-931, 2019.

Abstract

Climate change scenarios tend to intensify extreme rainfall events and drought in Brazil threatening urban water security. Low impact development (LID) practices are decentralized alternatives for flood mitigation and prevention. Recently, their potential has increasingly been studied in terms of stormwater harvesting. However, there is still a lack of knowledge about their potentialities in subtropical climate regions. Therefore, this study evaluated the behavior of a bioretention cell in a Brazilian city, during the dry period, which is critical in terms of pollutant accumulation and water availability. In addition to the runoff reduction and pollutant removal efficiency, this paper analyzed the potential for water reuse in terms of the stored volume and water quality guidelines. The results obtained show an average runoff retention efficiency of 70%. Considering only the water availability aspects, the potential stored runoff could be reused for non-potable purposes, reducing the water demand in the catchment by at least half during the dry season. On the other hand, the bioretention presented two different conditions for pollutant removal: Condition A – the concentration values are within the recommended limits for water reuse. The parameters found in this condition were NO_3, NO_2, Zn, Mn, Cu, Cr; Condition B – the pollutant concentrations are above the guideline limits for water reuse and cannot be directly used for different purposes. The parameters found in this condition were Fe, Pb, Ni, Cd and color. Considering water reuse, an additional treatment is required for parameters in this second condition. Further studies should evaluate the design aspects that can allow collection of LIDs effluent, additional treatment if necessary, and reuse in the catchment.

Keywords: Bioretention; Stormwater reuse; Water security; Stormwater harvesting; Pollutant removal

4.1 Introduction

Rapid urbanization has caused structural and environmental changes in urban basins, increasing paving, reducing soil infiltration and increasing pollutant deposition (Konrad & Booth, 2005; Leopold, 1968; Stovin et al., 2012; Wong & Eadie, 2000). It has also changed social conditions making the population more vulnerable to risks. As a consequence of these changes, there is an important increase in surface runoff, turning natural hydrological cycle risks into urban problems. Extreme rainfall events are precursors of risks to the population (Santos, 2007; Young et al., 2015), who are more vulnerable to floods and landslides. These can be made worse by climate change (Debortoli et al., 2017; Marengo et al., 2010).

Concerning the Brazilian scenario, research carried out by the Brazilian Institute of Geography and Statistics (IBGE) found that more than half of the municipalities in Brazil experienced floods between 2008 and 2012. Among these, the metropolitan region of Sao Paulo was the third city with the highest number of occurrences with a total of 704 floods (IBGE, 2013). During this period, there were deaths in 25% of the flood events in the southeast region. From 2014 to 2016, an extreme drought affected southeast Brazil and the rainfall from January to March was 54% lower than the 1961–1990 reference period (Cemaden, 2015), which caused an unprecedented water crisis in Sao Paulo state. The main supply system in the Sao Paulo metropolitan area, Cantareira, operated at the levels of its dead volume affecting the water security of about 8.8 million people (Escobar, 2015; Tafarello et al., 2016). These extreme droughts also led to other water-related impacts, such as increases in the price of electricity and food (Richards et al., 2015).

Due to the fact that cities are facing these environmental problems and knowing that they tend to become worse with climate change scenarios, the largest cities in the world created the C40 group to discuss and exchange public management actions and policies aimed at reducing the impacts generated and felt by them. In 2014, this group released a diagnostic report and evaluation of its proposed actions. In this report (C40, 2014), 90% of the cities that comprise the group indicated that climate change presents significant risks to their cities; the main ones related to floods and water stress. In addition, they also point to urban drainage as a key to flood risk management, where alternative techniques and systems rank in third place in the group's most accomplished actions. Therefore, the importance of urban drainage can be observed as an adaptation measure to make cities more resilient (Carter et al., 2015). Considering that water stress will become increasingly frequent in these scenarios, alternative drainage techniques that reuse stormwater as a form of urban harvesting (Agudelo-Vera et al., 2012) contributes to increasing urban resilience, as well as water, food and energy security.

These alternative techniques have various nomenclatures which are used worldwide, depending on the region and country where they are used. The most used is Low Impact Development (LID) practices, Stormwater Control Measures (SCM) and Best Management Practices (BMP) in the USA, Water Sensitive Urban Design (WSUD) in Australia and Sustainable Urban Drainage Systems (SUDS) in Europe (Eckart et al., 2017; Fletcher et al., 2013). In this study, we will adopt the LID terminology. LID practices aim to reestablish the natural hydrological cycle of pre-urbanization, focusing on water infiltration and integrated efficiency in the runoff amount and pollutant control (Council, 2007; Fletcher et al., 2013; Prince George's County, 2007). Research centers in Melbourne (Australia) and Santa Monica (USA) are pioneers in integrating LID practices in stormwater reuse from stormwater harvesting.

Many studies further evaluate the benefits of separate water retention and flood attenuation (Davis, 2008; Winston et al., 2016) from pollutant treatment and water quality improvement (Bratieres et al., 2008; Davis, 2007), making it difficult to integrate assessments for stormwater harvesting (Lucke & Nichols, 2015; Hatt et al., 2009). This gap is even larger in subtropical regions as most of the studies are conducted in temperate regions, where geoclimatic, sanitary and social conditions are very different from those in subtropical climate areas. Therefore, studying adaptations and monitoring LID practices for tropical and subtropical regions is still a shortcoming, and some questions still remain:

1. Does using stormwater harvesting techniques increase water security in cities?

2. Does only the direct reuse of stored stormwater contribute to the increase in water security?

3. Does the effluent of the LIDs systems have the appropriate quality standard for water reuse?

Aiming to answer these questions, in this study we evaluated the performance of a LID practice of bioretention already installed in an urban subtropical climate basin, designed for flood mitigation purposes. Based on runoff monitoring (volume, flow and pollution), we considered the potential of adapting these techniques to stormwater harvesting, concerning the direct reuse of water and its contribution to increasing water safety.

4.2 Methodology

4.2.1. Study site

The bioretention analyzed in this study was created and has been in operation since 2015 at the University of Sao Paulo (USP/SC campus 2) in the city of Sao Carlos. This area is representative of other cities with medium to fast urbanization rates and is classified as Cfa in the Köppen climate classification having a total annual rainfall of 1361.6mm and an average daily temperature of 21.5 ºC. The rainy season occurs from November to April and January has the most rainfall (274.7 mm and average daily temperature of 23.4ºC). The dry season occurs from May to October and July has the least rainfall (28.3 mm and an average daily temperature of 18.5ºC) (EMBRAPA, 2017).

USP/SC Campus 2 is located in the Mineirinho river basin. It was inaugurated in 2005 and is still in an expansion process (in 2015 only 15% of its total area was occupied). Therefore, the influence of land use and occupation changes on the long-term bioretention performance can be evaluated. Moreover, the area is a development axis of Sao Carlos city, mainly with a population of low income and popular housing. The Mineirinho basin presents environmental fragility, with points of irregular sewage deposition (Benini, 2005).

The bioretention catchment has a total area of 2.3 ha representing an urban drainage system on a neighborhood level scale (terminology from Marsalek and Schereier (2009)) with runoff reaching the Mineirinho river directly. The main contribution to runoff comes from pedestrian paths, roads and classroom buildings (Figure 4.1), totalizing 25% of the catchment. The other 75% is mostly grassland.

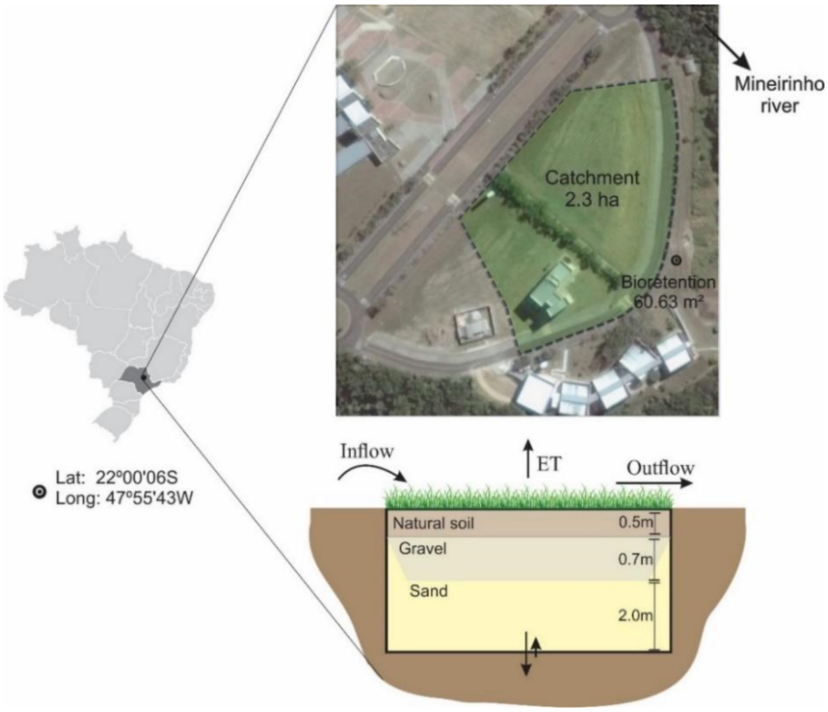

Figure 4.1 - Study site: Campus 2 USP/SC and bioretention scheme

As for the bioretention device, it has a total surface area of 60.63 m² and is 3.2 m deep. Its interior has a filter media composition divided into three layers - soil, gravel and sand - with an average porosity of 35% (Figure 4.1). The top layer is composed of natural soil from the region, which is characterized as dark brown with organic matter and a main composition of medium sized sand (40%), 25% fine sand and 16% clay, and it has a hydraulic conductivity of 5.83 $mm.h^{-1}$. It is important to highlight that despite the soil is natural because it was obtained directly from the site, it has his structural properties modified by excavation. This layer has a depth of 50cm and is covered by four different plant types (*Brachiaria sp.*, *Sorghum sudanense*, *Sansevieria trifasciata* and *Cyperus papyrus*) responsible for landscape integration, soil fixation and helps to improve pollutant removal (Hunt et al., 2015).

The intermediate layer is a 70 cm gravel layer, with a diameter of 5 cm and porosity of 40%. The bottom layer is 2 meters deep comprising coarse sand, with 1mm diameter and porosity of 30%. The gravel and sand layers together are responsible for the greater retention of surface runoff volume and qualitative treatment, totaling a volume of approximately 58 m³. The configuration presented was chosen to achieve the qualitative treatment of sedimentary solids.

The bioretention practice is located at the outlet of the local urban drainage pipe. The water is directed to the device using a rectangular channel, with decanter functions for retaining larger solid particles, equipped with a rectangular-triangular composite section weir, functioning as the inlet structure. Concerning the outlet structure, the system only has a triangular weir for the surface runoff, ensuring a ponding depth of 30 cm. For the subsurface flow, the designed system does not have underdrains for water collection, and therefore all stored water percolates to the ground or is lost by evapotranspiration.

4.2.2 Analysis of the LID performance in water retention and pollutant control for water security

Data were collected in the bioretention in the field over three years (2015 – 2017) during rainfall events in the dry season. This drought period is critical in terms of pollutant accumulation due to the greater deposition of pollutant load in the catchment surface, which is washed-off by the runoff during low intensity rainfall events. This period is also critical in terms of water security because the few rainfall events, which can affect the water supply systems.

The data corresponding to the water balance collected in the field were: inflow, outflow, storage and rainfall. For the inflow and outflow, water level sensors (HOBOU20L-02; Onset; detection limit of 4mm) were used coupled to the inlet and outlet weir, located above the ponding zone to control the exceeding runoff. For the storage, water level sensors were installed in piezometers along the length of the bioretention basin. The precipitated depth was obtained from a rain gauge located at the site. For each of these points and variables, data were collected every minute.

As for the evapotranspiration (ET), three potential or reference ET models (i.e. Hamon, Priestley-Taylor, and FAO Penman-Monteith) were used to obtain daily mean values for each event (Hamon 1961; Priestley and Taylor 1972; Allen et al. 1998). The potential ET models of Hamon and Priestley-Taylor indicates the evaporative demand of the environment, considering only meteorological conditions such as daily air temperature for Hamon, and daily air temperature and solar radiation for Priestley-Taylor (Hamon 1961; Priestley and Taylor 1972 On the other hand, FAO Penman-Monteith model represents an index of the hypothetical ET for a reference surface without any water deficiency, with 0.12m of uniform grass covering the entire surface, a constant albedo of 0.23 and surface resistance of 70 s m−1. In this model, the meteorological parameters used are solar radiation, vapor pressure deficit, wind speed, and daily air temperature (Allen et al., 1998). This ET models were also used in other studies evaluating bioretention performance (Nocco et al., 2016). The meteorological data were obtained from the Sao Carlos weather station of the Brazilian Agricultural Research Corporation - EMBRAPA, which monitors climatic parameters in several Brazilian cities.

Finally, the amount of water percolated to the ground was obtained by water balance (Eq. 4.1, adapted from Erickson et al., 2013). The total volume obtained for each of the water balance variables were quantified in terms of equivalent depth related to the catchment area.

$$S = (P + V_{in}) - (V_{out} + V_I + V_{ET}) = (\frac{Q_{in}(t).t}{A_w} + P) - (\frac{Q_{out}(t).t}{A_w} + V_I + \frac{ET\ .\ Ab}{A_w}) \quad (4.1)$$

where: S = Storage volume in the bioretention basin [mm]; P = Direct precipitation over the bioretention basin [mm]; V_{in} = Total inflow volume [mm]; V_{out} = Total outflow volume [mm]; V_I = Total percolated volume [mm]; V_{ET} = Total evapotranspirated volume [mm]; $Q_{in}(t)$ = Inflow discharge [m³/s]; t = Analyzed time interval [s]; A_w = Catchment surface [1000 m²]; $Q_{out}(t)$ = Outflow discharge [m³/s]; ET = Evapotranspiration over the bioretention surface [mm]; A_b = Bioretention surface [1000 m²].

To complement the analyses, the 30-day Antecedent Precipitation Index (API_{30}) was calculated for each event, according to Eq. 4.2 (Kohler and Linsley, 1951). This index evaluates the previous humidity of the site.

$$API_{30} = \sum_{i=1}^{30} \left[\left(\frac{1}{i} \right) . Pi \right] \quad (4.2)$$

where: API_{30} = Antecedent Precipitation Index for 30 days [mm]; i = total days of the i-eth period before the event in question; Pi = Total accumulated precipitation depth corresponding to the i-eth period [mm].

Concerning the water quality improvement, the data were collected for the critical period of pollutant accumulation, corresponding to the dry season until the beginning of the rainy season. A total of 12 parameters representing contamination by organic matter, nutrients and metals were analyzed, normally used for rainwater characterization and frequently found in Sao Carlos (Galavoti, 2011). These are: Turbidity, pH, color, Chemical Oxygen Demand (COD), phosphate (PO_4), nitrite (NO_2), nitrate (NO_3), ammonia (NH_3), sedimentary solids (SS), Total Organic Carbon (TOC) iron (Fe), zinc (Zn), lead (Pb), nickel (Ni), manganese (Mn), copper (Cu), chromium (Cr), cadmium (Cd). The analysis of these parameters is important to evaluate nutrient cycling, soil and water contamination.

For the quantification of these pollutants in the runoff and the water quality improvement provided by the system, samples were collected for the inlet and outlet structures, and inside the piezometers, corresponding to the inflow, outflow and storage, respectively. The samples were collected every 5 min for the inlet channel and 20 min for the outlet weir and piezometers. The total sampling time was up to 2h (which represents 6x the time of concentration of the catchment). The laboratory analysis of each parameter was based on the methodology proposed in the Standard Methods for Examination of Water and Wastewater (APHA et al., 2015). All pollutants were above the detection limit of the methods used.

These pollutants were analyzed in terms of concentration in order to compare them with the water quality standards and guidelines. However, the water quality improvement evaluation from the concentration assessment does not adequately reflect the effect of volume reductions in pollution control. Thus, we also adopted a load approach as a different flow for each time interval influences the total pollutant mass transferred downstream (Lago et al., 2017). The load value was calculated from Eqs. 4.3 and 4.4. Finally, the bioretention performance was evaluated according to the indicators presented in Eqs. 4.5 and 4.6.

$$EMC = \frac{\int_0^{t_1} C(t)Q(t)\,dt}{\int_0^{t_1} Q(t)\,dt} = \frac{\sum_0^{t_1} C(t)Q(t)\,\Delta t}{\sum_0^{t_1} Q(t)\,\Delta t} \tag{4.3}$$

$$Load = \int C(t)Q(t)\,dt = EMC.V_{total}/1000 \tag{4.4}$$

$$PR = P_{percolated}/Pe \tag{4.5}$$

$$Eff = \pi_{ret}/(Pe + Pi) \tag{4.6}$$

$$\pi_{ret} = \frac{(Q_{in}(t) - Q_{out}(t)).t}{A_w} \tag{4.7}$$

where: EMC = Event Mean Concentration [mg/L]; C(t) = Concentration at time t [mg/L]; Q(t) = Water flow at time t [L/min]; t_1 = Total event duration [min]; Δt = Considered time interval [min]; V_{total} = Total volume of input or output [L]; PR = Percolation ratio; $P_{percolated}$ = total equivalent depth percolated to the underlying soil [mm]; Eff = Water retention efficiency; Pe = total equivalent of runoff arriving at the bioretention input [mm]; Pi = total rainfall depth directly on the bioretention; π_{ret} = total equivalent depth retained by the bioretention [mm], calculated according to Eq. 4.7.

After collecting the data, the mean value of contamination level in each water balance variable of the bioretention was raised to evaluate the water quality improvement to the runoff. These values were compared with the water quality standards in Brazil. However, a current problem is the lack of specific regulations for water reuse worldwide. The Australian Guidelines for Water Recycling (NRMMC, 2008) were systematized only in 2008, and the Guidelines for Water Reuse produced by EPA (EPA, 2012) were updated in the USA in 2012. As for the Brazilian regulation, the water quality was evaluated compering with standards related to river framing and suitable use (Resolution CONAMA 357/420, Brazil MMA (2005)), and rainwater reuse (ABNT, 2019).

4.3 Results and Discussion

4.3.1 Stormwater volume reduction

A total of 14 rainfall events, scattered throughout the dry seasons in Brazil were analyzed for three years. The dry season is critical in terms of pollutant accumulation due to the large antecedent dry period or even due to less rainfall, which is not able to completely wash off the soil. Moreover, this period is also critical in terms of water security due to the small amount of rainfall, which affects the water reservation systems used both for human supply and for energy production (considering that in Brazil most of the electricity comes from hydroelectric power plants) and for food production, which depends on irrigation.

Figure 4.2 shows a characterization and water balance of all monitored events. For these events, the total rainfall depth (P_{total}) was low to medium, with Event 3 standing out as the most intense monitored event, with a maximum intensity of 120 mm/h. Moreover, for Event 3 the percolation ratio was one of the largest, but with a low volume stored inside the bioretention device, with a storage peak reaching 0.41mm (~ 10m^3). This result demonstrates that the bioretention device is working below its total retention capacity. After investigating the causes of this event, we observed soil erosion on the surface of the device and a small amount of established plants (Macedo et al., 2017). Therefore, for the next events, the amount of vegetation was increased. This measure helps increase the water infiltration into the device and the lag time, and also helps to reduce the peak flow as the roots create preferential paths into the soil and the aerial part of the plants increases the resistance to the flow, slowing down the runoff (functioning as an energy disperser).

For the following events, an increase in the storage peak was noted (with the exception of Event 8), varying from 0.55 to 2.17mm (~13 – 50m^3). This result indicates that the vegetation cover has a positive impact because with the increase of the peak storage there was also a rise in the efficiency of water retention.

Regarding the outflow (Figure 4.2), the average pattern shows a noticeable runoff reduction transferred downstream. It could also be observed that the output volume has a relation with the storage, but it is not limited by it. Typically, higher storage peak values are associated with higher retained volume and, consequently, lower outflow. However, two points should be raised: (1) outflow is observed even without the stored volume reaching the total bioretention device capacity, concluding that for the monitored events, the infiltration on the vegetated layer was the limiting factor to the outflow; (2) high storage peaks were observed for some events with low water retention efficiency. In these cases, the stored volume is more related with the low percolation into the ground than with the bioretention capacity of runoff volume control.

It is also important to compute the losses by ET over time. The daily ET values were obtained from the average between potential and reference ET models (i.e. Hamon, Priestley-Taylor and FAO Peanman-Monteith). The events that resulted in higher ET values were 4, 5 and 6, which correspond to the days with higher solar radiation. Regarding Event 4, the daily ET occurred directly on the bioretention surface reached the highest value (4.6 mm), corresponding to an equivalent depth in the water balance of VET = 0.012 mm. This amount corresponds to less than 0.2% of the total precipitation, less than 1.5% of the inflow depth and less than 12% of the total percolation estimated for this event. Regarding all the events, the mean values of daily ET corresponded to 0.07 ± 0.05% for the total precipitation depth, 0.41 ± 0.35% for the total inflow depth, and 2.5 ± 3.38% for the total percolated depth. Therefore, during the dry season (corresponding to the austral winter season, with less solar radiation) we noticed that ET was negligible when comparing to the other components of the water balance.

To evaluate the bioretention device performance, we also calculated the water retention efficiency (Eff). The values obtained ranged from 33% to 100%, with an average value of 70 ± 26% (the intervals are presented in terms of standard deviation) (Figure 4.2). The volume reduction obtained in this study is higher than that compared with studies developed in other regions (Davis, 2008; Hatt et al., 2009; Lucke & Nichols, 2015; Winston et al., 2016). However, it should be remembered that the monitored events occurred during the dry season, and have a low intensity; therefore, more studies need to be carried out for the rainy season.

An assessment of groundwater replenish was conducted by obtaining the percolation rates in order to evaluate if the bioretention systems also helps to reestablish the water balance prior urbanization, other than just control the runoff peak and volume. The percolation was obtained by the water balance, represented by the Eq. 4.1.

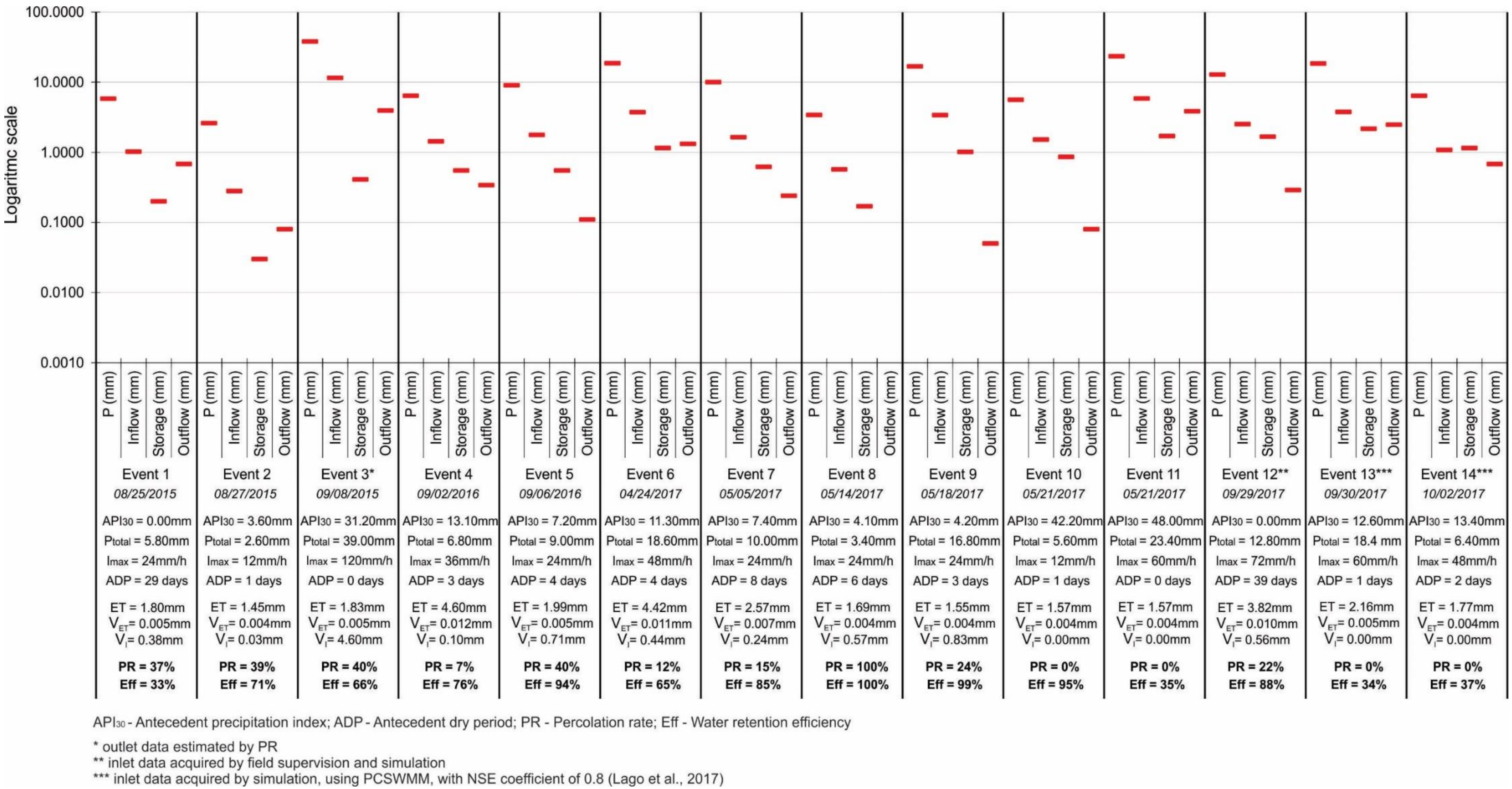

Figure 4.2 – Water balance per event

The results obtained show that, generally, the total volume percolated to the ground during the events is even smaller than the outflow volume. This aspect may be associated with ground soil saturation, leading to an increase in the storage, as mentioned previously. The storage will percolate into the ground over time, even after the rainfall has ended, helping to replenish groundwater. The percolation ratio (PR) presented an average value of 24 ± 27%. This variation is related to the soil moisture content and the inflow depth. As for the storage, on the other hand, the values presented a low variation. However, there were extremes of low storage for some events, as Events 1, 2 and 3, when there was still not much vegetation cover.

Figure 4.3 shows the hydrographs of events with two different types of behavior. In Figure 4.3a, Events 5 and 9 represent the behavior of large peak flow reduction and an increase in lag time, while there is a large reduction in the outflow volume (94% and 99% respectively), called pattern 1. On the other hand, in Figure 4.3b the behavior represented by Events 6 and 11 have little or no reduction in the peak flow, and therefore the total runoff retention (65% and 34% respectively) is more important in flood mitigation. This behavior is called pattern 2.

In Figure 4.2, these two types of behavior are more explicit: and for the events with pattern 1, the bars between inflow and outflow are more spaced, and usually the outflow is smaller than the storage. Events 4, 5, 7, 8, 9, 10 and 12 correspond to this first behavior. Concerning pattern 2, the inflow and outflow bars are closer together, with a storage value lower than the outflow. Events 1, 2, 3, 6, 11, 13 and 14 correspond to the second behavior. This difference in the patterns seems to be associated with the antecedent dry period, and consequently, the antecedent soil moisture condition. The events of pattern 1 have higher values of antecedent dry periods and lower API_{30}, while those of pattern 2 presented API_{30} values above 11mm and small antecedent dry periods.

However, some exceptions can be observed. Events 1 and 2 have antecedent dry periods and API_{30} characteristics of pattern 1, but the water balance behavior of pattern 2, which may be associated with a lack of vegetation cover in the soil layer, as previously mentioned. Events 4 and 10, on the other hand, have antecedent dry periods and API_{30} value characteristics of pattern 2, but a water balance behavior relative to pattern 1. This could be explained when analyzing the total rainfall that also influences the water retention efficiency. For these events, besides the higher API_{30} and lower antecedent dry periods, the total rainfall volume was low enough so that the bioretention could retain a greater amount of it.

Therefore, despite the observation of two different behaviors that can be explained by the variables associated with the soil moisture condition, the different climatic variables along with soil characteristics influence the efficiency of the device, and it is not possible to isolate them completely. Therefore, when analyzing only one of the factors it is not possible to regard a clear relation between them and Eff or PR.

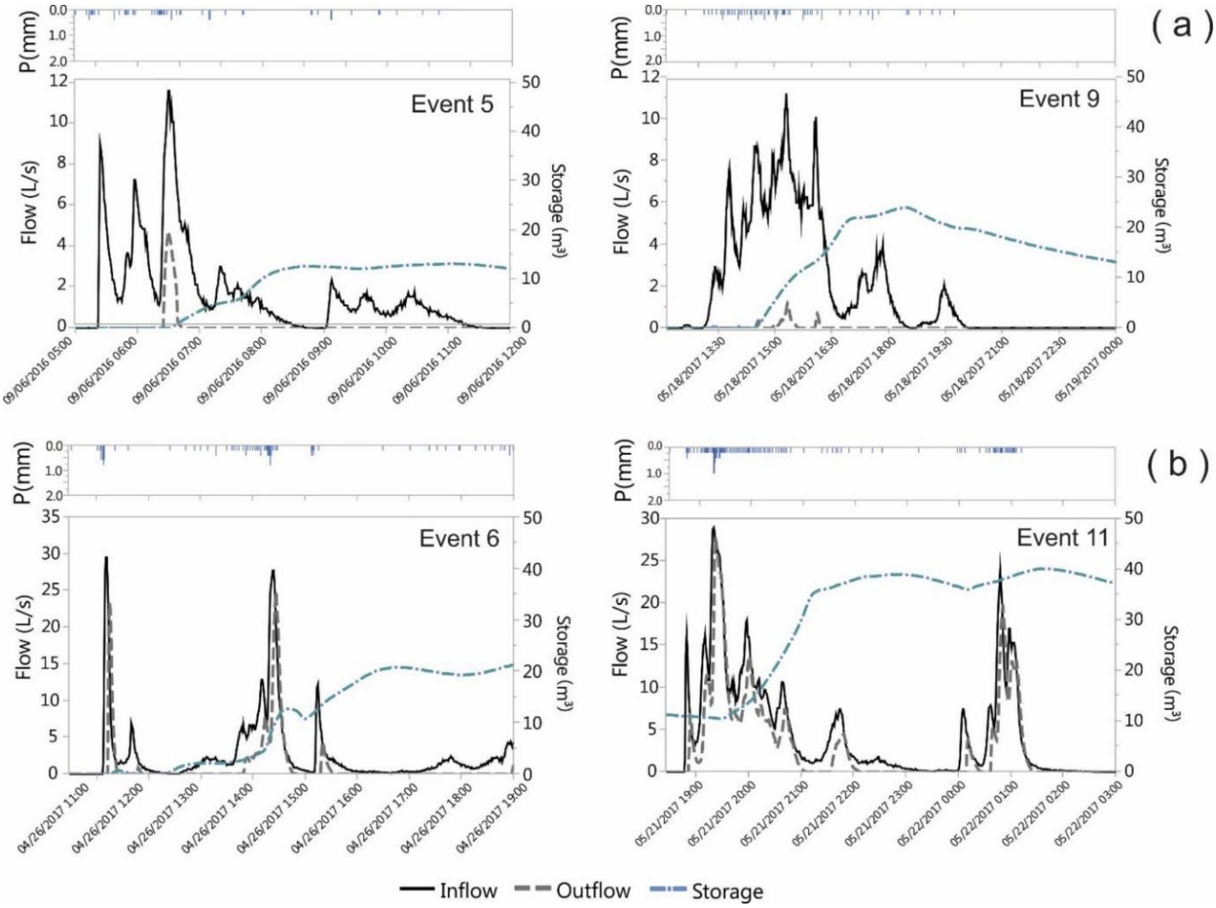

Figure 4.3 - Hydrographs representing different types of behavior: (a) pattern 1 and (b) pattern 2

4.3.2 Pollutant removal

In Figure 4.2, Events 1 and 12 are the ones with higher antecedent dry periods, leading to high pollutant accumulation in the surface. However, Event 1 has a low total rainfall depth and the pollutant wash off continues to occur in the following events. Therefore, for the pollutant removal analysis, Events 1, 2, 3 and 12 were monitored.

Figure 4.4 shows the EMC results obtained for all events at the device inflow, storage and outflow. In terms of concentration, it can be observed that there is no noticeable variation between the three points and the concentration remains practically constant. Only for some pollutants, the outflow value is lower than the inflow, such as COD, TOC, representing organic matter contamination, and metals, Cu and Mn. To better evaluate the pollution level found in the runoff, the concentration values obtained were compared with legislations and guidelines used in Brazil and around the world.

In Brazil, there is still no specific legislation and standards regarding water reuse. The standards and guidelines normally used correspond to specific water uses. For this study, to compare the pollutant level with a standard, the CONAMA resolution 357/420 was used, which presents quality standards for rivers and effluent discharge. The standard values considered were relative to "river class 2", which means rivers that may be for human consumption after conventional treatment, primary contact recreation, irrigation of vegetables and fruit, or any other direct human contact. The standard values for each pollutant are represented by a dotted line in Figure 4.4. When there is no dotted line, there is no limit value specified.

In general, for the analyzed catchment, the pollutants representing organic and nutrient contamination in the runoff are within the established standards. Metals, however, are all above the standards, with the exception of Mn and Cr. For Cu, the inflow has a value above the standard, but in outflow and storage the concentration is reduced, complying with the standard. Finally, the color is also above the limit established by the resolution.

To compare with the specific legislations and recommendations for water reuse that are already used worldwide, guidelines from the USA and Australia were also evaluated. Concerning guidelines from the USA, considering unrestricted urban use with the exception of drinking water, for the analyzed parameters only turbidity presents a guideline value, and for this case the runoff is worse than the limit. Regarding Australian guidelines, the standard values for water recycling are the same as those for drinking water. For this case, the turbidity and NH_3 are out of limits, as well as Fe, Pb, Ni, Cd and color. On the other hand, different from Brazilian standards, Zn and Cu comply with the Australian guideline values.

With these results based only on a concentration approach, the bioretention does not appear to be able to treat or assist the pollutant removal. However, it is important to mention the following: (1) despite the fact that the soil layer has a good pollutant removal capacity due to the adsorption process and plant assimilation (Laurenson et al., 2013), the main treatment takes place inside the bioretention device in the sand and gravel layers, where the biofilter and phytoremoval will be established. This happens because inside the bioretention device, the hydraulic retention time is higher than on the surface, favoring non-conservative chemical processes of (bio)degradation, such as denitrification (Erickson et al., 2013; Mangangka et al., 2015). Therefore, the outflow value is expected to remain at the same concentration or slightly lower than the inflow due to a moderate settling (for pollutants bound to particles) and adsorption process in the soil layer. For this configuration, the percolation represents the water actually treated by the bioretention device. (2) Analysis of pollutant removal capacity only by the concentration approach does not take into account the effects of volume attenuation provided by the LID practice, thus presenting an incomplete analysis (Lago et al., 2017).

Therefore, this study also includes the pollutant load approach in order to incorporate the water balance effects to improve the water quality. These results are shown in Figure 4.5, in load per square meters. This unit was chosen to present a load unit that could be compared with other bioretentions with different catchment areas.

From the pollutant load approach, almost all the pollutants present an average load reduction, mainly between the inlet and outlet. For storage, this variation is even more noticeable as the infiltrated volume accounts for 12% – 40% (with an exception of 100%) of the inflow volume. As an exception, Fe and NO_3 presented no reduction in average loads, rather, an increase in loads was observed. The NO_3 exports are due to the nitrification process that occurs after long drought periods, converting nitrite to nitrate (Bratieres et al., 2008; Davis et al., 2006; Mangangka et al., 2015; Payne et al., 2014). Regarding Fe, this behavior can be explained due to region soil characteristics and a strong erosive process that occurred in Event 3. In the soil layer, natural soil (Red Oxisoil) was used whose main characteristic is the high presence of iron oxide (hematite). In the erosion process in the bioretention, the soil was carried out with the outflow, leading to an export of Fe (Macedo et al., 2017).

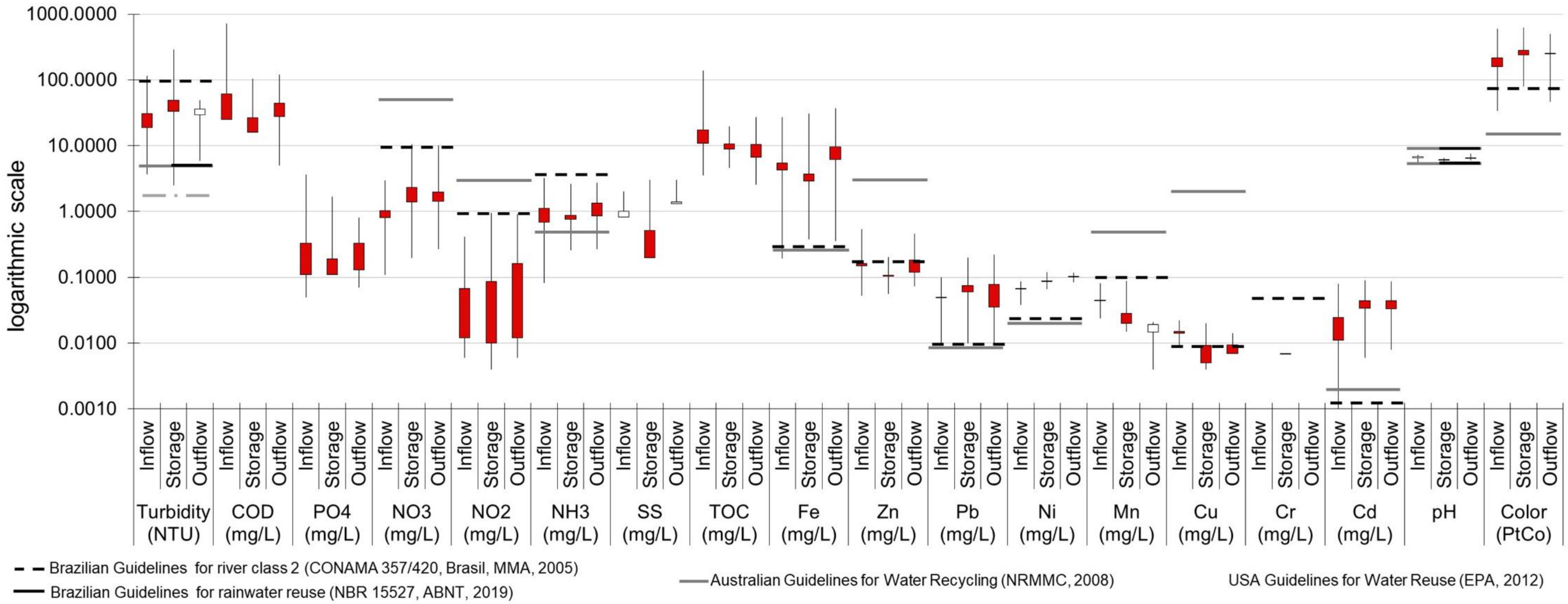

Figure 4.4 - Event Mean Concentration in each water balance variable of bioretention for n = 100. Dotted lines represent the values of water quality standards and guidelines.

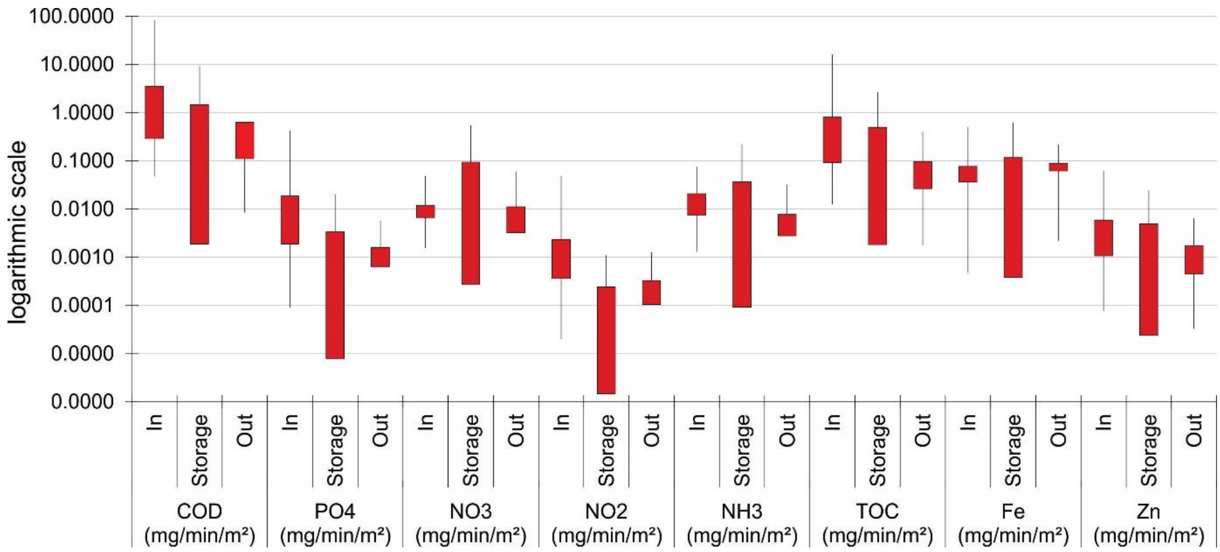

Figure 4.5 - Pollutant mass balance, for n = 100

4.3.3 Potential for stormwater harvesting and reuse

Considering the possibility of adapting LID practice to the dual purpose of flood protection and water reuse, through stormwater harvesting, an adaptive design needs to be developed that allows volume storage inside the bioretention device to be collected by lower drains and transferred to individual water reservoirs. In this case, the increasing percolation function provided by the LID practices will be reduced. However, reserving stormwater for multiple uses helps increase water security in the dry season and, consequently, city resilience to climate change. For the bioretention presented in this study, if all stored water were collected by drains, an annual average of 14.8 ± 11m^3 per storm would be reserved during the dry season, which is equivalent to 0.22% of all the rainfall depth during the wettest month.

To better assess the contribution of stormwater reuse to increasing water security, the total amount of stored water should be compared to water demands (which need to be quantified). As an example of application, we have assessed the water demands for cleaning purposes close to the catchment area (for the study site, the cleaning is done in the classroom building next to the bioretention). For this purpose, a one-week survey was conducted with the cleaning staff, quantifying all the water expenses required for this type of activity (cleaning bathrooms, and the indoor and outdoor areas). As a result, an average expense was obtained by type of activity. The survey also determined the cleaning routine (what activities are done on which days, how many times a week, and how many times a month). Finally, the calculation of the monthly water demand and during the entire dry season was made by multiplying the expenses by activity per the total amount of activities carried out in the analyzed period.

For this type of water use in the site, there is an average daily demand of 150 L, totaling 27 m³ in the total dry season. The stormwater volume stored by the bioretention device per event corresponds to half of the cleaning demand, i.e., with two events occurring during this season, the water needs for cleaning purposes can be met. Therefore, the demand for tap and drinking water from public supply systems can be reduced, increasing the local water security during the dry season. However, to meet the demand, stormwater needs to meet water quality standards.

Evaluating the water quality values at the bioretention outflow and storage, the potential water reuse directly from the LID practice can be affected by the pollution level. Two conditions may be found for the water quality improvement through the bioretention for each pollutant parameter: Condition A - the water in the outflow or in the percolated flow reaches the quality standards evaluated, and therefore it can be reused directly after storage; Condition B – the water quality in the outflow and percolated flow is less than the quality standards. Therefore, the guidelines for each type of intended use should be considered in order to choose the reuse that best suits the quality of the water obtained, generating no risks to human health or the environment. One other option is to adapt the system with additional treatment modules to achieve the appropriate quality for the chosen water use.

For this study, we observed that the parameters NO_3, NO_2, Zn, Mn, Cu and Cr can be classified in condition A for all the guidelines and standards analyzed, therefore there is no need for further treatment concerning this pollutants. However, for parameters Fe, Pb, Ni, Cd and color, they were above the limits for all the evaluated guidelines, and were suitable for condition B. Concerning turbidity and NH_3, these parameters are within the limits for the Brazilian legislation, but not for the Australian one. Thus, we can conclude that the additional treatment must be done focusing on metal removal as condition B mainly comprises this class of pollutants. It is important to highlight that in this study, we only evaluated the pollution level at the inflow, storage and outflow. However, for the case of stormwater reuse, the water will be collected by underdrain, which should have a better quality.

Additional treatment proposals have already been presented in studies carried out by Mitchell et al. (2007) from other LID types, such as wetlands, conjugated to post disinfection. Moreover, the Santa Monica SMURFF facility uses physio-chemical treatments, more commonly used for water supply and wastewater treatment facilities: coarse and fine screening, dissolved air flotation (DAF), microfiltration and ultraviolet (UV) disinfection (Boyle Engineering Corporation, 1999). For adaptive designs focusing on stormwater reuse, we propose to include studies of effluent treatability during the sizing stage, choosing the best system between biological and physio-chemical treatments. We present the biological treatment with tubular horizontal bed reactors as viable technologies, adapted from Sarti et al. (2006), Zaiat et al. (2000), and Zaiat (2003) and physio-chemical treatments of slow filtration, adapted from Sabogal Paz (2000) and Reali et al. (2013).

Stormwater can be reused and recycled in different ways to increase water security during the dry season: (1) All the stormwater treated by the bioretention cell during the wet season (mainly) can be collected by underdrains and stored in a reservoir, to be directly used in different demands; (2) The overflow of the bioretention cell, which has a better quality then the runoff, will be directed downstream and after going through the process of mixing and dilution in the river, the water can be collected again and used in some of the demands; (3) Part of the stormwater treated by the bioretention cell infiltrates to the ground supplying the water sheet. This can increase the river flow during the dry season, amortizing the extreme drought.

4.4 Conclusion

This study showed that the bioretention device presented a good volume reduction capacity, with average efficiencies of 70%, and the peak flow attenuation for events with a longer antecedent dry period and lower soil moisture was also large (pattern 1). The values of the percolation ratio showed that during rainfall events the percolation was low and the groundwater replenishment occurred mainly after the event, with a transfer of the stored volume to the ground over time. Therefore, the LID practice contributed to reestablishing part of the prior urbanization water balance.

Regarding water quality, the bioretention contribution to the reduction of pollutant concentration in the device outflow and storage was low. For metals, these values were not within the quality standard (CONAMA 357/420, Brazil – MMA, 2005). On the other hand, from the load analysis, which considers the effect of volume reduction, the pollutant removal was more remarkable.

The potential stormwater reuse directly from the LID practice storage can be affected by its quality. It is necessary to choose the reuse that corresponds to the quality value achieved, or adapt the system with additional treatment modules to achieve the standards of the attempted reuse.

Therefore, the LID practices can contribute to increasing the city resilience, both from reducing flood risks and pollutant contamination, as well as increasing water availability and reducing demand for potable uses. However, in order for these systems to achieve this double purpose for these systems, we must ask some key questions during the design phase: how much stormwater can be harvested? How reliable is this supply source? How large a store is required? (Mitchell et al., 2008). Additionally, it may exist a conflict in the purposes of flood control, where we need to have the device empty as quick as we can, and with the purpose of water reuse, where we want to store water for future use. This may be solved by integrating an additional storage tank connected with the bioretention outlet structure, however is an additional cost in the projects.

This study addressed a newly established bioretention in an expanding city in a subtropical climate during the dry season (May to October), which is a critical period regarding pollutant accumulation. To better evaluate its performance in flood risk control, further studies need to incorporate the critical period in terms of runoff, which for this subtropical condition is summer (November to April). In addition, adaptations should be made so that it can be used as a stormwater harvesting system, or implement new systems, answering to the key questions previously proposed.

References

Agudelo-Vera, C. M., Leduc, W. R., Mels, A. R., & Rijnaarts, H. H. (2012). Harvesting urban resources towards more resilient cities. *Resources, conservation and recycling*, 64, 3-12.

Allen, R. G., Pereira, L.S., Raes, D., Smith, M. (1998). *Crop evapotranspiration: Guidelines for computing crop water requirements*. FAO Irrigation and Drainage Paper. GAO, Rome.

APHA - American Public Health Association. (2015). *Standard methods for the examination of water and wastewater.* American Public Health Association (APHA): Washington, DC, USA.

Benini, R. M. (2005). *Cenários de ocupação urbana e seus impactos no ciclo hidrológico na bacia do córrego do Mineirinho*. [Urban occupation scenarios and their impacts on the hydrological cycle in the Mineirinho waterstream]. Master Thesis. Escola de Engenharia de São Carlos, Universidade de São Paulo, São Carlos. (In Portuguese)

Boyle Engineering Corporation (1999). *Engineering Report for Dry-Weather Runoff, Reclamation, Storage, Pumping, Distribution, and Nonpotable Water Use Area Facilities For City of Santa Monica*, Sept 1999, FR-C77-100-02.

Brasil, MMA (2005). *Resolução nº 357/420*. Ministério do Meio Ambiente, Conselho Nacional do Meio Ambiente, Brasília. (In Portuguese)

Bratieres, K.; Fletcher, T. D.; Deletic, A.; Alcazar, L.; Coustumer, L.; Mccarthy, D. T. (2008). Removal of nutrients, heavy metals and pathogens by stormwater biofilters. *Proceedings of 11th International Conference on Urban Drainage*.

C40 (2014). *Climate Action in Megacities: C40 Cities Baseline and Opportunities.* Volume 2.0. Acess in: 21 out 2016. Available in: <http://issuu.com/c40cities/docs/c40_climate_action_in_megacities/149?e=10643095/6541335> .

Carter, J. G.; Cavan, G.; Connelly, A.; Guy, S.; Handley, J.; Kazmierczak, A. (2015). Climate change and the city: Building capacity for urban adaptation. *Progress in Planning*, 95, 1-66.

Centro Nacional de Monitoramento e Alertas de Desastres Naturais [CEMADEN]. (2015). *Cantareira System: Current situation and hydrological prediction* (Full Report updated on 10 Fev 2015). Retrieved February 17, 2015, from http://www.cemaden.gov.br/cantareira/relatoriosistema-cantareira-10-022015.php. (In Portuguese)

COUNCIL, Gold Coast City. (2007). *Water Sensitive Urban Design Guidelines*. Gold Coast City Council, Queensland. Retrieved October, v. 6, p. 2009.

Davis, A.P., Shokouhian, M., Sharma, H. And Minami, C. (2006). Water quality improvement through bioretention media: Nitrogen and phosphorus removal. *Water Environmental Research*, 78(3), 284-293.

Davis, A. P. (2007). Field performance of bioretention: Water quality. *Environmental Engineering Science*, 24(8), 1048-1064.

Davis, A. P. (2008). Field performance of bioretention: Hydrology impacts. *Journal of Hydrologic Engineering*, 13(2), 90-95.

Debortoli, N. S., Camarinha, P. I. M., Marengo, J. A., & Rodrigues, R. R. (2017). An index of Brazil’s vulnerability to expected increases in natural flash flooding and landslide disasters in the context of climate change. *Natural hazards*, 86(2), 557-582.

Eckart, K.; Mcphee, Z.; Bolisetti, T. Performance and implementation of low impact development – a review. *Science of The Total Environment*, 607, 413-432.

EMBRAPA. (2017). *Condições Meteorológicas Estação da Embrapa Pecuária Sudeste.* [Meteorological conditions station of Southern Embrapa Pecuária]. Available in: <http://www.cppse.embrapa.br/meteorologia/index.php?pg=caracterizacao>. Accessed in: 01 jun. 2017. (In Portuguese)

EPA. (2012). *2012 Guidelines for Water Reuse*. U.S. Environmental Protection Agency, Office of Wastewater Management, Office of Water, Washington, D.C.

Erickson, A. J. ; Weiss, P. T. ; Gulliver, J. S. (2013). *Optimizing Stormwater Treatment Practices : A handbook of assessment and maintenance.* Springer, New York.

Escobar, H. (2015). Drought triggers alarms in Brazil's biggest metropolis. *Science*, 347(6224), 812–812.

Fletcher, T. D., Andrieu, H., & Hamel, P. (2013). Understanding, management and modelling of urban hydrology and its consequences for receiving waters: A state of the art. *Advances in water resources*, 51, 261-279.

Galavoti, R. C. (2011). *Proposta de um modelo de gestão integrada de águas urbanas em escala de lote residencial: alcances e limitações.* [Proposal for an integrated urban water management model on a residential lot scale: scope and limitations]. Thesis. Escola de Engenharia de São Carlos, Universidade de São Paulo, São Carlos. (In Portuguese)

Hamon, W. R. (1961). Estimating potential evapotranspiration. *Journal of Hydraulics Division ASCE*, 87(HY3), 107-120.

Hatt, B. E.; Fletcher, T. D.; Deletic, A. (2009). Hydrologic and pollutant removal performance of stormwater biofiltration systems at the field scale. *Journal of Hydrology*, 365(3), 310 – 321.

Hunt, W. F., Lord, B., Loh, B., & Sia, A. (2015). *Plant selection for bioretention systems and stormwater treatment practices*. Springer.

IBGE – Brazilian Institute of Geography and Satistics (2013). *Profile of Brazilian cities*. Avaiable in < https://ww2.ibge.gov.br/home/estatistica/economia/perfilmunic/2013/default.shtm>. Acess in: 04 july 2018.

Kohler, M. A., & Linsley, R. K. (1951). Predicting the runoff from storm rainfall (Vol. 30). US Department of Commerce, Weather Bureau.

Konrad, C.P., Booth, D.B. (2005). Hydrologic changes in urban streams and their ecological significance. *American Fisheries Society Symposium*, 47, 157–177.

Lago, C. A. F.; Macedo, M. B. ; Rosa, A. ; Mendiondo, E. M. (2017). Calibração e validação quali-quantitativa do escoamento superficial - Estudo de caso na USP/São Carlos. [Calibration and quali-quantitative validation of runoff - Case study at USP / São Carlos]. *Proceedings of XXII Simpósio Brasilieiro de Recursos Hídricos*, 2017, Florianópolis. (In Portuguese).

Laurenson, G.; Laurenson, S.; Bolan, N.; Beechan, S.; Clark, I. (2013). *Advances in Agrononmy:* Chapter Four – The Role of Bioretention Systems in the Treatment of Stormwater. Elsevier.

Leopold, L.B. (1968). *Hydrology for urban land planning: A guidebook on the hydrological effects of urban land use*. Circ. No. 554. U.S. Geological Survey, Washington, DC.

Lucke, T.; Nichols, P. W. B. (2015). The pollution removal and stormwater reduction performance of street-side bioretention basins after ten years in operation. *Science of the Total Environment*, 536, 784 – 792.

Macedo, M. B.; Rosa, A.; Do Lago, C. A. F.; Mendiondo, E. M.; De Souza, V. C. Borges. (2017). Learning from the operation, pathology and maintenance of a bioretention system to optimize urban drainage practices. *Journal of Environmental Management*, 204, 454-466.

Mangangka, I. R., Liu, A., Egodawatta, P., & Goonetilleke, A. (2015). Performance characterisation of a stormwater treatment bioretention basin. *Journal of Environmental Management*, 150, 173-178.

Marsalek, J.; Schreier, H. (2009). Innovation in Stormwater Management in Canada: The Way Forward. *Water Quality Resources*, 44(1), v-x.

Marengo, J. A.; Schaeffer, R.; Zee, D.; Pinto, H. S. (2010). *Mudanças climáticas e eventos extremos no Brasil.* [Climate change and extreme events in Brazil]. Available in: http://www. fbds. org. br/cop15/FBDS_MudancasClimaticas. pdf. Accessed in October, 2010. (in Portuguese)

Mitchell, V., Deletic, A., Fletcher, T. D., Hatt, B. E., & Mccarthy, D. T. (2007). Achieving multiple benefits from stormwater harvesting. *Water Science and Technology*, 55(4), 135-144.

Mitchell, V. G; Mccarthy, D. T.; Deletic, A.; Fletcher, T. D. (2008). Urban stormwater harvesting–sensitivity of a storage behaviour model. *Environmental Modelling & Software*, 23(6), 782-793.

Nocco, M. A., Rouse, S. E., & Balster, N. J. (2016). Vegetation type alters water and nitrogen budgets in a controlled, replicated experiment on residential-sized rain gardens planted with prairie, shrub, and turfgrass. *Urban ecosystems*, 19(4), 1665-1691.

NRMMC, E. (2008). AHMC, *Australian guidelines for water recycling: Managing health and environmental risks (Phase 2): Augmentation of drinking water supplies*, Environment Protection and Heritage Council, National Health and Medical Research Council. Natural Resource Management Ministerial Council, Canberra.

Payne, E. G. I.; Pham, T.; Cook, P. L. M.; Fletcher, T. D.; Hatt, B. E.; Deletic, A. (2014). Biofilter design for effective nitrogen removal from stormwater–influence of plant species, inflow hydrology and use of a saturated zone. *Water Science and Technology*, 69(6), 1312-1319.

Prince George's Couty. (2002). *Bioretention manual*. Prince George's County (MD) Government, Department of Environmental Protection. Watershed Protection Branch, Landover, MD.

Priestley, C. H. B., & Taylor, R. J. (1972). On the assessment of surface heat flux and evaporation using large-scale parameters. *Monthly weather review*, 100(2), 81-92.

Reali, M. A. P., Sabogal Paz, L. P., Daniel, L. A. (2013). *Tratamento de Água para Consumo Humano.* [Water treatment to human consuption] in: Engenharia Ambiental: Conceitos, Tecnologia e Gestão. Elsevier, Rio de Janeiro. (In Portuguese)

Richards, R. C., Rerolle, J., Aronson, J., Pereira, P. H., Gonçalves, H., & Brancalion, P. H. S. (2015). Governing a pioneer program on payment for watershed services: Stakeholder involvement, legal frameworks and early lessons from the Atlantic forest of Brazil. *Ecosystem Services: Science, Policy and Practice,* 16, 23–32.

Sabogal Paz, L. P. (2000). *El riesgo sanitário y la eficiência de los sistemas de tratamiento en la selección de tecnologías para la potabilizacion del água* [The sanitary risk and the efficiency of the treatment systems in the selection of technologies for the purification of the water]. Monography. Cali: Universidad del Vale. (In Spanish)

Santos, R. F. (2007). *Vulnerabilidade ambiental: desastres naturais ou fenômenos induzidos?* [Environmental vulnerability: natural disasters or induced phenomena?]. MMA (in Portuguese).

Sarti, A., Sakamoto, I. K., Varesche, M. B. A., Zaiat, M., & Foresti, E. (2006). Tratamento de esgoto sanitário utilizando reatores anaeróbios operados em bateladas sequenciais (escala piloto). [Sewage treatment using anaerobic reactors operated in sequential batches (pilot scale)]. *Engenharia Sanitária e Ambiental*, 73-82. (In Portuguese)

Stovin, V. R., Moore, S. L., Wall, M., & Ashley, R. M. (2013). The potential to retrofit sustainable drainage systems to address combined sewer overflow discharges in the T hames T ideway catchment. *Water and Environment Journal*, 27(2), 216-228.

Tafarello, D.; Mohor, G. S.; Calijuri, M. C.; Mendiondo, E. M. (2016). Field investigations of the 2013–14 drought through quali-quantitative freshwater monitoring at the headwaters of the Cantareira System, Brazil. *Water International*, 41(5), 776 – 800.

Winston, R. J.; Dorsey, J. D.; Hunt, W. F. (2016). Quantifying volume reduction and peak flow mitigation for three bioretention cells in clay soils in northeast Ohio. *Science of the Total Environment*, 553, 83 – 95.

Wong, T. H. F., Eadie, M. L. (2000). Water sensitive urban design—a paradigm shift in urban design. *Proceedings of the Xth World Water Congress*, 12:16. http://gabeira.locaweb.com.br/cidadesustentavel/biblioteca/%7B30788FE6-98A8-44E5-861E-996D286A78B3%7D_Wong1.pdf.

Young, C. E. F., Aguiar, C., Souza, E., 2015. *Valorando tempestades: custo econômico dos eventos climáticos extremos no Brasil nos anos de 2002 – 2012* [Assessing storms: economic cost of extreme weather events in Brazil in the years 2002 – 2012], first ed. Observatório do clima, São Paulo (in Portuguese).

Zaiat, M.; Passig, F. H.; Foresti, E. (2000). Treatment of domestic sewage in horizontal-flow anaerobic immobilized biomass (HAIB) reactor. *Environmental Technology*, 21(10), 1139-1145.

Zaiat, M. (2003). *Desenvolvimento e análise de biorreatores anaeróbios contendo células imobilizadas para tratamento de águas residuárias*. [Development and analysis of anaerobic bioreactors containing immobilized cells for wastewater treatment]. Habilitatoin Thesis. Universidade de São Paulo. (In Portuguese)

5 EVALUATING DIFFERENT CONFIGURATIONS OF BIORETENTION SYSTEMS IN CONTRIBUTING TO THE SUSTAINABLE DEVELOPMENT GOALS AND CIRCULAR CITIES

A diferente version of this chapter was submited as: Marina Batalini de MACEDO, Thalita Raquel OLIVEIRA, Tassiana Halmenschlager OLIVEIRA, Marcus Nóbrega GOMES JUNIOR, José Artur BRASIL, Cesar Ambrogi Ferreira do LAGO, Eduardo Mario MENDIONDO. **Evaluating different configurations of bioretention systems in contributing to the Sustainable Development Goals and circular cities**. Ecological Engineering.

Abstract

Nature-based Solutions (NbS) as alternative measures of urban drainage can be used within the approach of circular cities and contribute to the Sustainable Development Goals (SDG) based on the recycling and co-management of resources. This study aimed to evaluate the configuration of a bioretention system for the recycling of resources from the experimental monitoring of a system in the laboratory. Additionally, an assessment of the consequent contribution to circular cities was made by the extrapolation of results to household and watershed scale, quantifying the indicators of water demand reduction, energy demand reduction and carbon emission reduction from water hybrid systems. The laboratory results indicates that the use of a bioretention with a submerged zone can improve the quality of the water recovered for reuse, while maintaining the efficiency of runoff retention and peak flow attenuation. However, when comparing the parameters of water quality of the bioretention effluent with the Brazilian standards and guidelines for water supply and stormwater reuse, the color, turbidity, E. coli and metals presents values above the limits, indicating the necessity of a better treatment to solids particles and disinfection. Expanding the analysis to watershed scale, the bioretention helped to reduce non-potable water demands up to 45%, leading to a reduction in energy demand and carbon emission from the centralized water supply system. Additional studies should be done using more extreme events in laboratory scale and modeling to watershed scale.

Keywords: Water-energy-greenhouse gases nexus; Nature-based Solutions; Low Impact Development; Runoff retention; Water quality; Exploratory analysis.

5.1 Introduction

The United Nations Conference on Sustainable Development (also known as Rio + 20) discussed the challenges that still exist for the achievement of sustainable development in all nations and its aggravating factors in this new era. At this conference, it was established the need for a new agenda (2030 agenda), in which the 17 Sustainable Development Goals (SDG) were presented as the framework of results to be delivered until 2030. These goals integrate aspects of economic growth, social justice, balanced environment, and international cooperation to achieve sustainability (UN, 2020). Although the goals are presented separately, their assessment must always be done in an integrated and holistic manner, since sustainable development can only be achieved when all objectives are met.

At this conference, it was possible to note the growing concern of countries regarding climate change and its consequences, so that a specific goal (SDG 13) was established for urgent actions to combat climate change and adaptations to its impacts on society. Within the targets related to goal 13, actions are listed such as strengthening resilience and the capacity to adapt to climate-related risks in different locations, integrating measures for climate change into national and local policies, strategies and planning, institutional and human capacity in mitigation, adaptation and reduction of impacts and targets for reducing carbon emissions by the signatory countries (UN, 2020; UNSTATS, 2020). It is worth remembering that due to the current high rate of urbanization worldwide (about 55% of the population lives in urban areas in the world in 2017, already reaching over 85% in the case of Brazil (Our World in Data, 2019)), and due to the high population density and high level of paving in large urban centers, there is a drop in local resilience so that most of these impacts are felt in the cities (Carter et al., 2015).

In this sense, the 40 largest cities in the world came together to create the C40 group, aiming at cooperation between large urban centers[1]. In 2014, this group launched a diagnostic and evaluation report of its proposed actions. In this report (C40, 2014), 90% of the cities that make up the group indicate that climate change presents significant risks to their localities, the

[1] Currently, the C40 has 96 affiliated cities.

main ones being associated with floods and water stress. Furthermore, they also point to drainage as a key to flood risk management, in which alternative urban drainage systems occupy the third place in the actions most performed by the group. Therefore, it is noticeable the importance of urban drainage as adaptation measures to make cities more resilient.

Alternative urban drainage systems, known as Low Impact Development (LID) techniques[2], have a different approach than conventional urban drainage systems, using adaptive Nature-based Solutions (NbS) to control runoff. Initially, these systems aim to reintegrate excess runoff into the hydrological cycle, using techniques that increase the infiltration of water into the soil, decrease runoff velocity and reduce the pollutant load (Fletcher et al., 2013). These techniques are also applied in a decentralized manner, prioritizing the runoff control at the source (Fletcher et al., 2015; Eckart et al., 2017). Although its initial objective was to control runoff (and consequent flood control), re-establish the natural hydrological cycle, and reduce pollution in urban rivers, new studies have been developed based on systemic approaches, such as the *water-energy-food nexus* (Macedo et al., 2017) and *water-energy-greenhouse gases nexus*, integrating other mitigation and adaptation objectives for the LID practices. As an example, we can mention the runoff recycling for non-potable uses (and even more ambitious for potable use) (Burns et al., 2015; Chandrasena et al., 2016; Petit-Boix et al., 2018) , food cultivation (Richards et al, 2015; Ng et al., 2018), nutrient recycling (Ge et al., 2016), production and/or reduction of energy demands (Hashemi et al., 2015) and carbon sequestration (Getter et al., 2009; D'Acunha & Johnson, 2019; Charalambous et al., 2019). In this sense, we understand that from these cycles the LID techniques (called at 3rd generation LID, according to Macedo et al., 2017) have the capacity to contribute to some SDGs and their targets in urban centers, which can be evaluated from the attribution of own metrics and indicators on a local scale. The use of LID practices aimed at meeting SDGs is based on the phrase “Think globally, act locally”, increasingly used in the lexicon of sustainable development (Charlesworth, 2010).

[2] These systems are also known as Sustainable Urban Drainage Systems (SUDS), Water Sensitive Urban Design (WSUD), Sponge cities, Best Management Practices (BMP), Compensatory Techniques (CT), depending on the location.

In Figure 5.1, we present an example of the possible recycling pathways in the watershed urban cycle using a lined bioretention technique and how each pathway can contribute to different SDGs. The inflow runoff arrives at the bioretention with a high load of pollutants due to wash-off of surfaces (streets, roofs, sidewalks, etc.), including solids, nutrients (nitrogen and phosphorus), organic matter, metals (Litern et al. , 2011; Eckart et al., 2017), among others. The bioretention systems, which are composed of a vegetated layer, a filtering media and a lower drainage layer, are responsible for storing part of the water during rainfall events from the retention in the ponding zone and in the pores of the filtering media, releasing stored water (outflow) through a underdrain over a longer period of time and amortizing peak flows (Eckart et al., 2017; The Prince George's County, 2007, Waterways, 2006). New studies aim to recover the outflow (which has a higher quality than the runoff) for later non-drinking water demands, in order to increase the water security of the population in the dry season, a process known as stormwater harvesting (Mitchel et al., 2008; Fletcher et al., 2008); The contribution to the water security of the population in these systems will depend on factors such as rainfall pattern in the location, adequate water quality in the outflow, capacity of the storage reservoir and demand of the population (Karim et al., 2015).

Additionally, when the system is not able to retain all the runoff, the overflow process occurs, in which the runoff passes only through the ponding zone (with less water treatment processes), returning directly to the stream. In this case, there is a reduction in the pollutant loads being transported to the stream by two processes, reduction of pollutant concentration by simplified treatment processes such as sedimentation and interception by vegetation, and by the reduction of volumes, which lead to less transport of pollutants (Macedo et al., 2019a). These two processes that occur with outflow and overflow contribute to an improvement in the water supply and sanitation system. Outflow recycling increases decentralized water production and reduces drinking water supply demands in classic systems, increasing their resilience (Petit-Boix et al., 2018; Clark et al., 2015). The treatment and reduction of overflow contribute to a reduction of urban rivers contamination, contributing to more simplified and economical water treatment systems. Thus, we can say that there is a contribution to SDG 6 (clean water and sanitation).

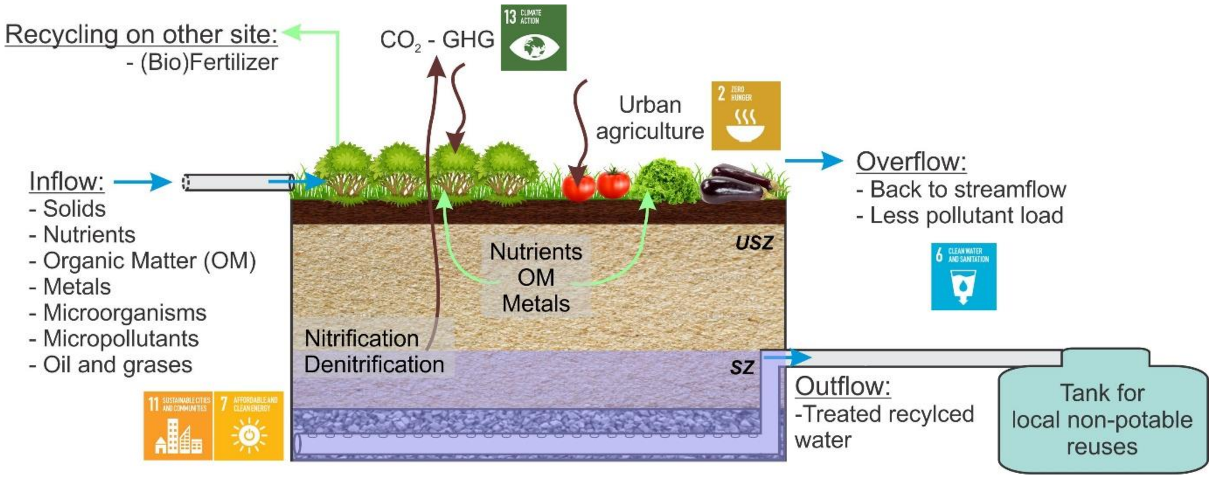

Figure 5.1 - Resource recycling pathways in a lined bioretention aiming at Sustainable Development Goals (SDGs). In the figure, USZ represents the unsaturated zone and SZ represents the saturated zone, the blue arrows represent the runoff pathways, the brown arrows represent the carbon pathways and the green arrows represent the nutrient pathways through plant uptake.

Since the bioretention systems (as well as other LID techniques) are vegetated, and one of the main pollutants present in the runoff are nutrients, new studies have explored the food production capacity on the surface of these systems, contributing to urban agriculture. The food cultivation allows nutrient recycling, which once free in water can lead to eutrophication processes, but when absorbed by plants, they contribute to their growth. The study by Richards et al. (2015) evaluated the food production in a bioretention practice, irrigated by runoff from a roof catchment. They obtained a food production capacity similar to a common irrigation system, additionally contributing to a reduction in the overflow frequency by more than 90%. The system proposed a sub-irrigation to reduce the direct contact between crops and pollutants, reducing contamination risks. Ng et al. (2018) also evaluated food production in bioretention systems, however, the presence of metals in the runoff makes this process tricky, due to their possible accumulation in the edible parts of the plants above the risk limits established by the WHO. In view of this challenge in direct food production, an alternative for nutrient cycling is the reuse of plant biomass as a biofertilizer in another location (Ge et al., 2016), which allows the management with a proper dosage so that there is no toxicity by metals to the plants or consumers. In this perspective, bioretentions can contribute to urban agriculture and sustainable agriculture, less intensive in the use of chemical fertilizers and in the use of land extension, contributing to SDG 2 (zero hunger), within its target of food security, volume and percentage of production under sustainable agriculture.

Another advantage of vegetated systems is their ability to assimilate carbon both by directly absorbing CO_2 from the atmosphere in plant tissues, as well as by fixing organic matter in the soil. Several studies are currently being conducted exploring the potential of carbon sequestration in vegetated techniques. Getter et al. (2009) studied the carbon sequestration capacity in green roofs and obtained values of 375gC/m^2 additional sequestration in these systems. Kavehei et al. (2018) and Kavehei et al. (2019) observed that bioretention systems have the capacity to sequester 70% of their carbon footprint in the production, transport and construction process. However, despite the capacity of these systems to sequester carbon by plant biomass, internal treatment processes can also contribute to the generation of GHG, mainly linked to anaerobic processes. The decomposition of organic matter, the processes of nitrification and denitrification result in the production of CO_2, CH_4 and N_2O, depending on the oxygen saturation in the medium (D'Acunha & Johnson, 2019; VO et al., 2014). Within the perspective of carbon sequestration, LID practices can contribute to SDG 13 (climate action).

From a general view of LID practices in urban infrastructure, they can integrate hybrid water supply systems (Sapokta et al., 2015), from decentralized water supply and treatment of diffuse pollution, and decentralized drainage systems. Thus, there is a reduction in energy demands in centralized supply systems, increasing the integrated resilience of water and energy systems, contributing to SDG 7 (affordable and clean energy). Consequently, from the view of the *water-energy-greenhouse gas nexus* (Nair et al., 2014), the reductions in energy demand in hybrid supply systems also contribute to the reduction of GHG emissions (Arora et al., 2015). These integrated studies, however, require more complete Life Cycle Analysis (LCA) studies (Petit-Boix et al., 2015; Petit-Boix et al., 2017).

Finally, due to its principles of interconnection, integration of materials, energy and water flows, decentralization to increase diversity and resilience, renewable flows, among others, LID practices can be considered within the "Infrastrcuture Ecology" approach (Pandit et al., 2017). These practices can increase cities resilience to disasters related to extremes of rainfall (floods) and water scarcity, incorporate society participation in the co-management of stormwater, and allow retrofit of urban space. In this way, the LIDs contribute to both SDG 11 (sustainable cities and communities) and SDG 13 (climate action).

It is possible to verify that most of the studies involving LID practices and their contributions to the SDGs do not make this connection directly and do not present metrics that allow quantify their contribution to each target, in addition to presenting isolated studies for each of the processes that contribute to SDGs. Therefore, this study aims to present an initial and integrated assessment of the contribution of a real system of bioretention to the SDGs. To this end, this study was separated into two main stages: (1) Formulation and presentation of local indicators to assess the contribution of urban drainage systems to SDGs 2, 6, 7 11 and 13, and their interconnections with regional and broader global and; (2) Exploratory analysis of the water balance and water quality data of a bioretention system evaluated in the laboratory with controlled events, identifying the connections between its variables and parameters, allowing the identification of better configurations and processes for the optimization of SDG indicators.

5.2 Materials and methods

5.2.1 Case study for a bioretention system

In this study, a laboratory prototype of bioretention system was evaluated (dimensions and configurations are shown in Figure 5.2), representing a real system at property scale. The field system collects water drained from a 94m2 roof, located on Campus 2 of the University of Sao Paulo, situated in the Mineirinho watershed, in the city of Sao Carlos, SP, Brazil. The sizing of the bioretention structure on a full scale was done by simulation in the TC-Hydro model (Gomes Junior, 2019), which combines synthetic hydrological models of temporal distribution, infiltration and rainfall-runoff to simulate the inflow. The flow propagation in the bioretention is done using a 1-D infiltration model (Green-Ampt) and the mass balance in the bioretention cell is resolved by the PULS method. As model inlet parameters and variables, a design storm for RP 10 years, duration of 30min and temporal distribution Huff 1st quartile was used (the design and evaluation of the structure in the field is presented in detail in Gomes Junior, 2019).The laboratory prototype was made on a 1:2 scale with the field system.

For Sao Carlos, the climate is classified as Cwa - dry-winter humid subtropical climate (Koeppen Climate classification), with two well-defined seasons, dry winters and hot wet summers. Sao Carlos has an average daily temperature of 21.5 oC, an average daily relative humidity of the air of 74.3%, and an average annual rainfall of 1361.6 mm, with higher rain concentration in January and lower concentration in July and August (EMBRAPA, 2017). The daily precipitation that occurs at 90% frequency (P90) for the city is 32.5 mm.

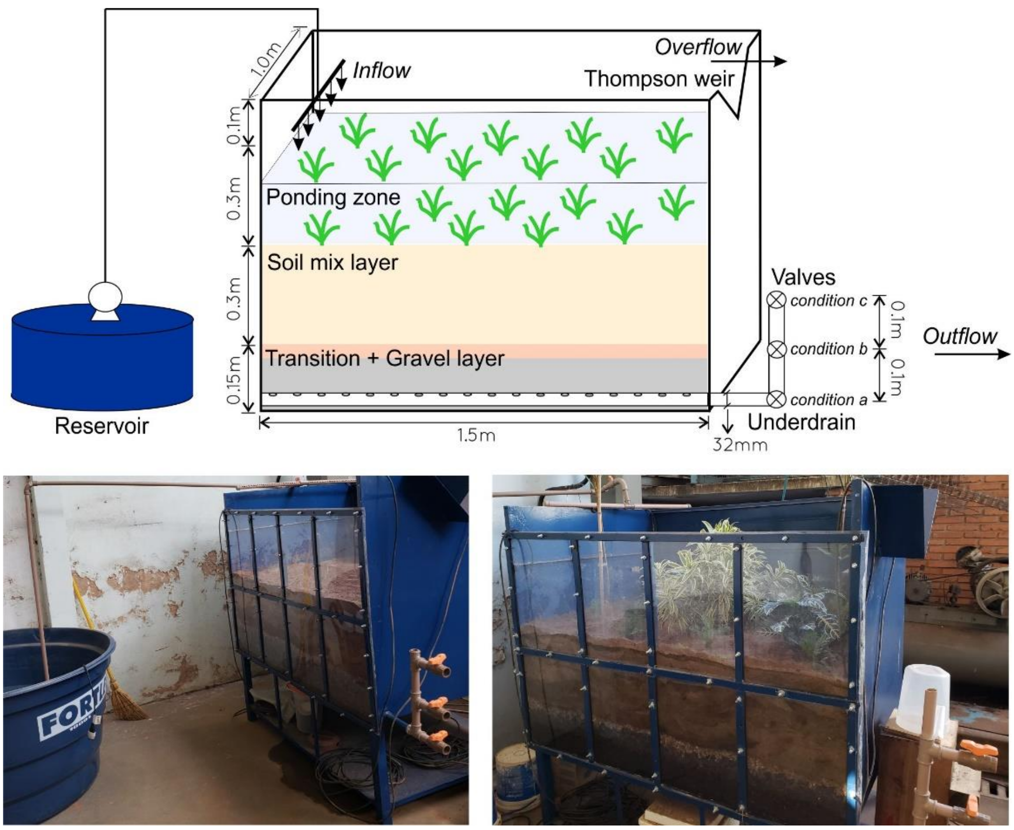

Figure 5.2 - Prototype in laboratory scale: Bioretention box scheme. For this bioretention system, the filtering media is composed by 20% of natural soil and 80% of coarse sand, and drainage media is formed by medium size gravel, according to recommendations of The Prince George's County (2007).

The laboratory prototype was built in the form of a bioretention box, according to the one proposed by Davis et al. (2006) and Macedo et al. (2018). The bioretention box allows keeping physical similarities as close as possible to the real system in the field to ensure that the hydraulic and treatment processes occurring in the field are replicated in the laboratory. In this regard, the scale of 1:2 was used to guarantee geometric similarity, and same construction materials, filtering and drainage media were used so the roughness, porosity and infiltration capacity was as close as possible to the real scale, in order to have a good representation of the hydraulic and treatment processes occurring in the field.

The evaluation of the system in the laboratory has some advantages in relation to the real system in the field (Macedo et al., 2018): (1) The environmental conditions of the events can be controlled to obtain a more appropriate investigation, both in terms of the total runoff volume and peak flows, as in pollutant concentration; (2) It allows the evaluation of different configurations and factors to optimize the design and operation. For this study, different configurations of saturated zones were evaluated for the treatment of nitrogen and pathogens and underdrain head loss (Figure 5.2); (3) There is no temporal dependence on the occurrence of rainfall events, allowing a greater number of analyzes in less time.

The design of the synthetic events was made based on the conditions of rainfall, height of the saturated zone and underdrain head loss (Table 1): Rainfall equivalent to two design storms of constant duration (30 min) was proposed for the return periods of 5 years (condition 1), representing more recurring events, and 50 years (condition 2), representing extreme events; Three saturated zone heights were assessed (condition a - without saturated zone, valve height equal to zero, condition b - with saturated zone regulated by a 0.1m valve height, condition c - with saturated zone regulated by a 0.2m valve height); and two underdrain conditions were evaluated, with more or less head loss, simulated by opening the output valve (condition I - valve completely open and condition II, valve half open).

Table 5.1 – Description of the conditions evaluated in the synthetic events in bioretention box prototype

Physical parameter	Condition	Description
Rainfall characteristics	Condition 1	RP = 5 years; P = 31 mm; d = 30 min; i = 62 mm/h
	Condition 2	RP = 50 years; P = 53 mm; d = 30 min; i = 106 mm/h
Saturation of filtering media	Condition a	Without saturated zone - valve height = 0m
	Condition b	With saturated zone - valve height = 0.1m
	Condition c	With saturated zone - valve height = 0.2m
Underdrain head loss	Condition I	Valve completely open
	Condition II	Valve half open

RP - Return period; P - rainfall depth; d - duration; i - rainfall intensity

The events were designed assuring equivalence to real events in terms of rainfall in the catchment area - Pequivalent (mm), duration of the event (d), operational factors of flow rate (FR) (m/h) and application rate (AR) (%), and the design factor equivalent net depth (Hequivalent) (m), as proposed by Macedo et al. (2018) for comparison between laboratory and real events. The equations defining each of the variables and the calculation of equivalence for this study are presented in Eq. 5.1 to Eq. 5.8. Details about the parameters here used and their application can be found in Macedo et al. (2018).

$$P_{DesignStorm} = i(RP, d) * d \tag{5.1}$$

$$V_{runoff,cat} = \frac{P_{DesignStorm}\, A_{cat}\, C}{1000} \tag{5.2}$$

$$P_{equivalent,bio,field} = \left(\frac{V_{runoff,A_{cat}}}{A_{bio,field}}\right)1000 \tag{5.3}$$

$$P_{equivalent,bio,field} = P_{equivalent,bio,lab} \tag{5.4}$$

$$V_{equivalent,bio,lab} = \frac{P_{equivalent,bio,lab}\,A_{bio,lab}}{1000} \tag{5.5}$$

$$Q_{runoff,lab} = \frac{V_{equivalent,bio,lab}}{d/60} \tag{5.6}$$

$$FR = \frac{\overline{Q_{runoff}}}{A_{bio}} = \frac{Q_{runoff,lab}}{A_{bio,lab}} \tag{5.7}$$

$$AR = \left(\frac{V_{runoff}}{V_{bio,storage}}\right)100 = \left(\frac{V_{equivalent,bio,lab}}{V_{bio,lab,storage}}\right)100 \tag{5.8}$$

where: $P_{DesignStorm}$ is the total rainfall of the design storm calculated based on Intensity-Duration-Frequency curve [mm]; d is the duration of the rainfall event [min]; $V_{runoff,cat}$ is the total runoff volume for the catchment area [m^3]; A_{cat} is the total catchment area [m^2]; C is the runoff coefficient [-]; $P_{equivalent,bio,field}$ is the rainfall depth calculated based on the bioretention area in field equivalent to the runoff volume of the catchment area [mm]; $A_{bio,field}$ is the surface area of the bioretention in field, $P_{equivalent,bio,lab}$ is the rainfall depth calculated based on the bioretention area in laboratory equivalent to the runoff volume of the catchment area [mm]; $V_{equivalent,bio,lab}$ is the volume equivalent to the $P_{equivalent,bio,lab}$ based on the bioretention area in laboratory [m^3]; $A_{bio,lab}$ is the surface area of the bioretention in laboratory [m^2]; $Q_{runoff,lab}$ is the runoff inflow in the bioretention in laboratory [m^3/h]; FR is the flow rate [m/h]; $\overline{Q_{runoff}}$ is the average inflow in a generic bioretention device [m^3/h]; A_{bio} is the surface area of a generic bioretention device [m^2]; AR is the application rate [%]; V_{runoff} is the total inlet runoff volume of a generic bioretention device [m^3]; $V_{bio,storage}$ is the total storage volume of a generic bioretention device [m^3]; $V_{bio,lab,storage}$ is the total storage volume of the bioretention in laboratory.

In total, 26 synthetic events were monitored between January 2019 and February 2020 (Table 1). To assess the water quality and treatment capacity of the system, 10 events were evaluated, in which the following parameters were analyzed: Chemical Oxygen Demand (COD), nitrogen series (nitrite - NO_2, nitrate - NO_3, and ammonia NH_3), phosphate - PO_4, apparent color, pH, turbidity, Total Coliforms - TC, E. coli, metals (cadmium - Cd, chromium - Cr, copper - Cu, iron - Fe, manganese - Mn, nickel - Ni, lead - Pb and zinc - Zn), sedimentable solids - SS. The synthetic inflow for experiments 21, 22 and 23 was made from dosages of nitrogen salts in clean water without chlorine, specific to analyze the reaction of the intra-event nitrogen series. For the other events, synthetic water was made from a mixture of clean water without chlorine and solid particles collected from the same catchment area (where the roof is located to obtain a similar dry deposition), using the sweeping method, proposed by Maglionico (1998).

Regarding the monitoring of water balance variables, the inflow was kept constant, according to the designed storm of the event, and the variables overflow through weir, outflow through underdrain and ponding depth were monitored every 5 minutes. A total of 10 water samples were collected to assess the quality of the overflow and outflow every 5 min. For the inflow, the concentration was kept constant throughout the event, by continuous mixing. For exact quantification of the inflow concentration, samples were collected for analysis in the beginning and end of the event. At the end, it was possible to obtain hydrographs and pollutographs for all monitored events.

The efficiencies regarding runoff volume and peak were calculated according to Eq. 5.9 to Eq. 5.11. For a bioretention aiming to contribute to the SDG from water recycling, the outflow is conducted to a reservoir for future non-potable reuse. Therefore, to evaluate the increase of water stress resilience, the indicators of runoff conversion into water for reuse the Eq. 5.12 was used.

$$Eff_{rr} = \frac{V_{in} - V_{over}}{V_{in}} \tag{5.9}$$

$$Eff_{peak} = \frac{Q_{peak,in} - Q_{peak,over}}{Q_{peak,in}} \tag{5.10}$$

$$Eff_{time} = \frac{t_{peak,in} - t_{peak,over}}{t_{peak,in}} \tag{5.11}$$

$$Eff_{wr} = \frac{V_{out}}{V_{in}} \tag{5.12}$$

where: Eff_{rr} [-] is the runoff retention efficiency; V_{in} [L^3] is the total inflow volume; V_{over} [L^3] is the total overflow volume; Eff_{peak} [-] is the peak attenuation efficiency; $Q_{peak,in}$ [L^3T^{-1}] is the maximum inflow value; $Q_{peak,over}$ [L^3T^{-1}] is the maximum overflow value; Eff_{time} [-] is the time delay efficiency; $t_{peak,in}$ [T] is the duration of the event until the $Q_{peak,in}$; $t_{peak,over}$ [T] is the duration of the event until the $Q_{peak,over}$; Eff_{wr} [-] is the water reuse efficiency; V_{out} [L^3] is the total outflow volume.

In addition, the system's treatment capacity and pollutant removal efficiencies were evaluated in terms pollutant load (Eq. 5.13) and Event Mean Concentration (EMC) (Eq. 5.14). The system efficiencies related to runoff quantity and quality can be used as individual indicators to the SDGs.

$$Eff_{pr,load} = \frac{M_{in} - M_{out}}{M_{in}} \tag{5.13}$$

$$Eff_{pr,EMC} = \frac{EMC_{in} - EMC_{out}}{EMC_{in}} \tag{5.14}$$

where: $Eff_{pr,load}$ [-] is the pollutant removal efficiency, in terms of pollutant load; M_{in} [M] is the inflow pollutant mass; M_{out} [M] is the outflow pollutant mass; $Eff_{pr,EMC}$ [-] is the pollutant removal efficiency, in terms of pollutant event mean concentration (EMC); EMC_{in} [ML^{-3}] is the event mean concentration in the inflow; EMC_{out} [ML^{-3}] is the event mean concentration in the outflow.

5.2.2 Statistical design

The statistical evaluation of the results of the bioretention system was made in two main stages, the first was an exploratory analysis of the data (EDA) and the second was a clustering evaluation of the results. Statistical analysis was performed using the Python library SciPy 1.5.0 and SciKit 0.6.1 and scikit-learn 0.23.2.

For the EDA, the data were grouped at first in three different ways: (1) With respect to the water balance (the different variables of the water balance and their related indicators); (2) Regarding the removal of pollutants (concentration and load of different pollutants related to the different variables of the water balance and its indicators); and (3) Regarding the configuration of bioretention (heights of the saturated zone). For the EDA, descriptive statistical measures of the results and their distribution were performed, correlation measures using Pearson's linear correlation coefficients and Spearman's monotonic correlation between all the parameters of the groups, and hypothesis testing for comparison of central measures of runoff retention, water reuse, and pollutant removal efficiency for different independent groups, using the Kruskal-Wallis Test and Dunn's test. Since hydrological data generally presents great skewness, almost never having a normal distribution, these tests were chosen because they are non-parametric, presenting greater power of description in non-normal data (Helsel et al., 2020). All statistical tests were evaluated for significance level α = 0.10.

For the second stage, the clustering was carried out in order to identify patterns in the groups according to the different indicators (runoff retention, water reuse and pollutant removal) that allow to optimize the design of the bioretention systems for the different evaluation objectives. The hierarchical agglomerative clustering (HAC) method was used. The choice of this method was due to the fact that it is not necessary to specify the number of clusters, since we want to identify the best system configurations or input variables that affect performance based on the characteristics of the clusters, without assumptions, and because it is not sensitive to choice of metric distance. After the construction of the dendogram, the groups were separated from a greater distance of vertical line (visual) and Elbown method (mathematician).

5.2.3 Contribution to SDGs in watershed scale

The use of bioretention cells, as decentralized drainage systems that allow water quality improvement and water reuse, can integrate the approach of hybrid water supply systems (Sapokta et al., 2015). The hybrid systems allow a reduction in energy demands in centralized supply systems, increasing the integrated resilience of water and energy systems, contributing to SDG 7 (affordable and clean energy). Consequently, from the view of the water-energy-greenhouse gas nexus (Nair et al., 2014), the reductions in energy demand in hybrid supply systems also contribute to the reduction of GHG emissions (Arora et al., 2015). In this study, we propose to evaluate the contribution of the bioretention systems to the water-energy-greenhouse gas nexus using the metrics presented in Eq. 5.15 to 5.18, at the Mineirinho watershed (Figure 5.3).

$$WSR = 1 - \left(\frac{WD - V_{out,rec}}{WD}\right) \tag{5.15}$$

$$WDR_{hs} = \frac{\sum(WD - V_{out,rec})}{\sum A_b} / IA_c \tag{5.16}$$

$$EDR_{hs} = \frac{ED_{cs} - ED_{hs}}{\sum A_b} / IA_c \tag{5.17}$$

$$CER_{hs} = \frac{CE_{cs} - CE_{hs}}{\sum A_b} / IA_c \tag{5.18}$$

where: WSR [-] is the Water Stress Reduction index; WD [L^3] is the total water demand from the centralized water supply system; $V_{out,rec}$ [L^3] is the recovered outflow volume available to non-potable reuse; WDR_{hs} [$L^3L^{-2}L^{-2}$] is the Water Demand Reduction in hybrid systems, in terms of volume per alternative drainage systems area per impervious catchment area; IA_c [L^2] is the impervious catchment area; EDR_{hs} [$ML^2T^{-2}L^{-2}L^{-2}$] is the Energy Demand Reduction in hybrid systems, in terms of energy per alternative drainage systems area per impervious catchment area; ED_{cs} [ML^2T^{-2}] is the energy demand in the conventional water supply system; ED_{hs} [ML^2T^{-2}] is the energy demand in the hybrid system; CER_{hs} [$ML^{-2}L^{-2}$] is the Carbon Emission Reduction of hybrid systems, in terms of carbon mass per alternative drainage systems area per impervious catchment area.

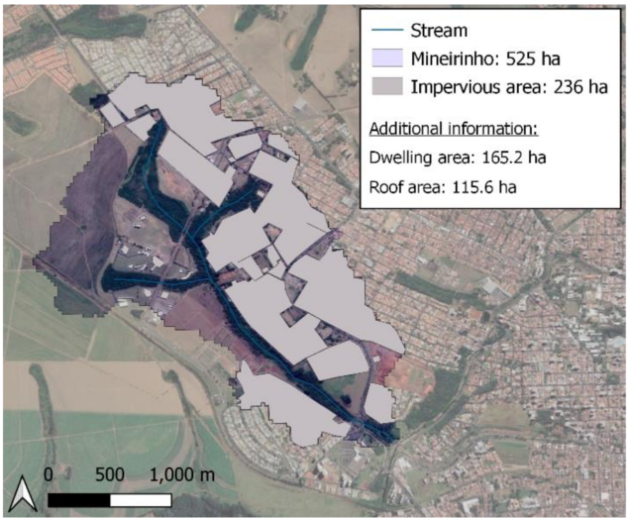

Figure 5.3 - Mineirinho watershed area and occupation characteristics

To quantify the reduction in the centralized water demand in households, the following data were collected:

(1) Average demand of tap water per household, adopting the average data of consumption per economy, micro-measured consumption per economy and consumption of billed water by economy, available in the National Sanitation Information System (SNIS) for the year 2018.

(2) Water demand profile and identification of non-potable demand for households in Brazil. A water demand profile of households in different Brazilian region was made in the studies by Silva (2013), Barreto (2008), Sant'Ana et al. (2013), Duarte et al. (2018) and Cesar (2016), for the cities of Vitoria - ES, Sao Paulo - SP, Brasília - DF, Belem do Para - PA and Rio de Janeiro

- RJ, respectively. It was considered for non-potable demand uses of irrigation and other outdoor uses, flushing toilets and washing machine.

(3) Volume stored in the reuse reservoir. The volume stored in the reservoir was obtained by the equivalent average outflow volume from the bioretention box in laboratory to field scale (calculated according to Eq. 3 – 5), reaching a maximum volume stored of the reservoir capacity. For this study, it was adopted a modulated reservoir with capacity of 1m3 due to space availability in the residential lots.

(4) Average monthly rainfall volume and average monthly dry days, for normal climatological conditions from 1989 to 2018, constructed with historical data from the National Meteorological Institute (INMET) for the city of Sao Carlos. Finally, the cost of saving water monthly was obtained, considering the current tariff for the city of Sao Carlos (ARES-PCJ, 2019).

After the evaluation for one household, an optimistic scenario of the use of bioretention structures coupled to reservoirs for water reuse in all residences in the Mineirinho watershed was raised. The total number of residences in the watershed was calculated using image classification in QGIS 3.12 software and the following steps:

(1) Quantification of the total watershed area.

(2) Quantification of the total impervious area. All the impervious area (excluding the area of the University of Sao Paulo) was considered as residences.

(3) Quantification of the total area of roofs, considering the maximum percentage of construction per property of 70%, according to maximum occupancy coefficients defined in the Master Plan of the City of Sao Carlos for Zones 1 and 2 (which fall within the Mineirinho basin) (PMSC, 2016).

(4) Quantification of the total number of residences, dividing the total area of roofs in the watershed (determined by the step 3) by the average roof area of one household for the city of Sao Carlos.

Finally, the water demand for non-potable uses and the water recovery from the NbS systems for the watershed was obtained from the extrapolation of data from one household to the total number of houses allocated in the watershed. Then, it was possible to obtain a total reduction in water demands from the central supply system in the entire watershed area.

To quantify the energy demands and carbon emissions for the central and hybrid systems, we obtained: (1) the average energy demand of the water supply networks for the city of Sao Carlos for the year 2018 (SNIS, 2018): 1.2 kWh/m3 of water distributed, and; (2) the average monthly CO2 emission value per unit of energy produced for the National Interligated System (MCTIC, 2020).

5.3 Results and discussion

5.3.1 Runoff quantity results

Table 5.1 presents the 26 monitored events and their descriptions in terms of the equivalence variables to real events in the field and previous drought conditions. The monitored events cover a wide range of variability in relation to the previous drought condition, ranging from evaluating events with less than a day of spacing, to up to 4 months of drought (condition that occurs in the city of Sao Carlos, during the dry winters). In addition, 4 events with a shorter duration were monitored, classified as a1, a2, a3 and a4. These events have the same rainfall intensity for design storms of 5 years RP, but with 10 min duration, representing events of lesser magnitude. It is worth mentioning that the total rainfall measured considering RP 5 years and duration of 30 min is close to P90 (32.5mm), that is usually used as total rainfall volume for bioretention design in other sizing methods (The Prince George's County, 2007).

Figure 5.4 presents the average hydrographs and their confidence intervals for each type of configuration evaluated, aggregated according to the intensity of the event. For events with greater recurrence (RP = 5 years and d = 30min, condition 1), no overflow was observed for any of the configurations, reaching a maximum ponding depth of 30cm, representing an imminence for overflow through weir (since the weir is 31cm high). For this condition, the increased head loss in the underdrain (condition II) and the presence of a saturated zone (condition b and c) do not affect the runoff retention capacity, at the same time that they present greater amortization in the outflow. If there is no recovery of water for reuse, a more amortized outflow contributes to the reduction of flood events. If water is recovered for reuse, an amortized outflow means longer water retention within the filtering media, which may result in greater treatment efficiency (hypothesis later evaluated with water quality analysis).

Table 5.2 - Description of the monitored events

Event	Date	Type	FR (m/h)	AR (%)	d (min)	API (mm)	Dry days	Quality monitoring	V_{in} (L)	V_{out} (L)	V_{over} (L)	$V_{storage}$ (L)	Q_{in} (L/h)	$Q_{peak,out}$ (L/h)	$Q_{peak,over}$ (L/h)	Group EMC	Group Load
1	1/29/2019	1.a.I	1.0	239	29.0	-	-	No	701.32	618.15	0	83.17	1451	1320	0	-	-
2	1/30/2019	1.a.II	1.0	240.0	29.2	31.0	0	No	705.35	656.85	0	48.5	1451	822	0	-	-
3	1/31/2019	1.b.I	1.0	243.0	29.6	46.5	0	No	714.62	549.78	0	164.84	1451	864	0	-	-
4	2/1/2019	1.b.II	1.0	239.0	29.0	56.8	0	No	701.32	590.7	0	110.62	1451	736	0	-	-
5	2/4/2019	1.c.I	1.0	251.0	30.4	29.5	2	No	735.95	633.55	0	102.4	1451	792	0	-	-
6	2/6/2019	1.c.II	1.0	249.0	30.3	35.2	1	No	731.95	667.7	0	64.25	1451	680	0	-	-
7	3/11/2019	2.a.I	1.6	405.0	31.4	5.0	32	No	1187	975.6	6	205.4	2268	1128	72	-	-
8	3/12/2019	2.a.II	1.6	392.0	30.4	57.8	0	No	1150.38	865.1	121.3	163.98	2268	852	924	-	-
9	3/13/2019	2.b.I	1.6	440.0	34.2	84.2	0	No	1290.87	985.8	159	146.07	2268	1110	930	-	-
10	3/14/2019	2.b.II	1.6	391.0	30.4	101.8	0	No	1147.86	881.4	147	119.46	2268	816	1092	-	-
11	3/19/2019	2.c.I	1.6	396.0	30.8	37.7	4	No	1162.98	860	177.8	125.18	2268	864	1188	-	-
12	3/28/2019	2.c.II	1.6	412.0	32.0	23.0	8	No	1209.6	881.2	191.7	136.7	2268	924	1206	-	-
13	5/27/2019	1.a.I	1.0	246.0	30.0	6.1	59	Yes	721.44	592.55	0	128.89	1442.89	840	0	0	0
14	6/10/2019	1.a.II	1.0	249.0	30.4	7.4	13	Yes	731.03	655.65	0	75.41	1442.89	666	0	0	0
15	6/26/2019	1.c.I	1.0	255.0	31.1	7.4	15	Yes	747.9	520.7	0	227.2	1442.89	498	0	0	0
16	8/6/2019	1.c.II	0.7	170.0	28.8	5.0	40	Yes	500	392.2	0	107.8	1040.46	456	0	1	1
17	8/19/2019	1.a.I	1.0	238.0	30.2	6.8	12	Yes	697.31	456.5	0	240.81	1384.62	744	0	0	0
a1	8/20/2019	a.I	1.0	87.0	10.6	37.5	0	No	473.08	455.3	0	17.78	1384.6	564	0	-	-
a2	8/20/2019	a.II	1.0	82.0	10.1	48.5	0	No	692.31	582	0	110.31	1384.6	648	0	-	-
a3	8/21/2019	c.I	1.0	82.0	10.0	43.9	0	No	255.71	224.3	0	31.41	1442.89	536	0	-	-
a4	8/21/2019	c.II	1.0	82.0	10.0	54.9	0	No	242.89	234	0	8.89	1442.89	585	0	-	-
18	9/2/2019	1.c.I	1.0	161.0	20.5	10.8	11	Yes	240.48	214	0	26.48	1442.89	336	0	0	1
19	9/30/2019	1.c.I	1.0	236.0	30.0	6.7	27	Yes	240.48	265.2	0	-24.72	1442.89	348	0	0	0
20	1/21/2020	1.c.I	1.0	232.0	29.5	3.1	112	Yes - N serie	680.77	550.7	0	130.07	1384.62	588	0	-	-
21	1/28/2020	1.c.I	0.9	236.0	30.0	7.5	6	Yes - N serie	692.31	642.5	0	49.81	1348.6	618	0	-	-
22	2/11/2020	1.cI	0.9	236.0	30.0	6.5	13	Yes - N serie	692.3	656	0	36.3	1348.6	666	0	-	-

FR - Flow rate, AR - Aplication rate, d - Duration, API - Antecedent precipitation index, Dry days - days with 0mm of precipitation, $V_{storage}$ - final stored volume, Q_{in} - inflow, $Q_{peak,out}$ - outflow peak

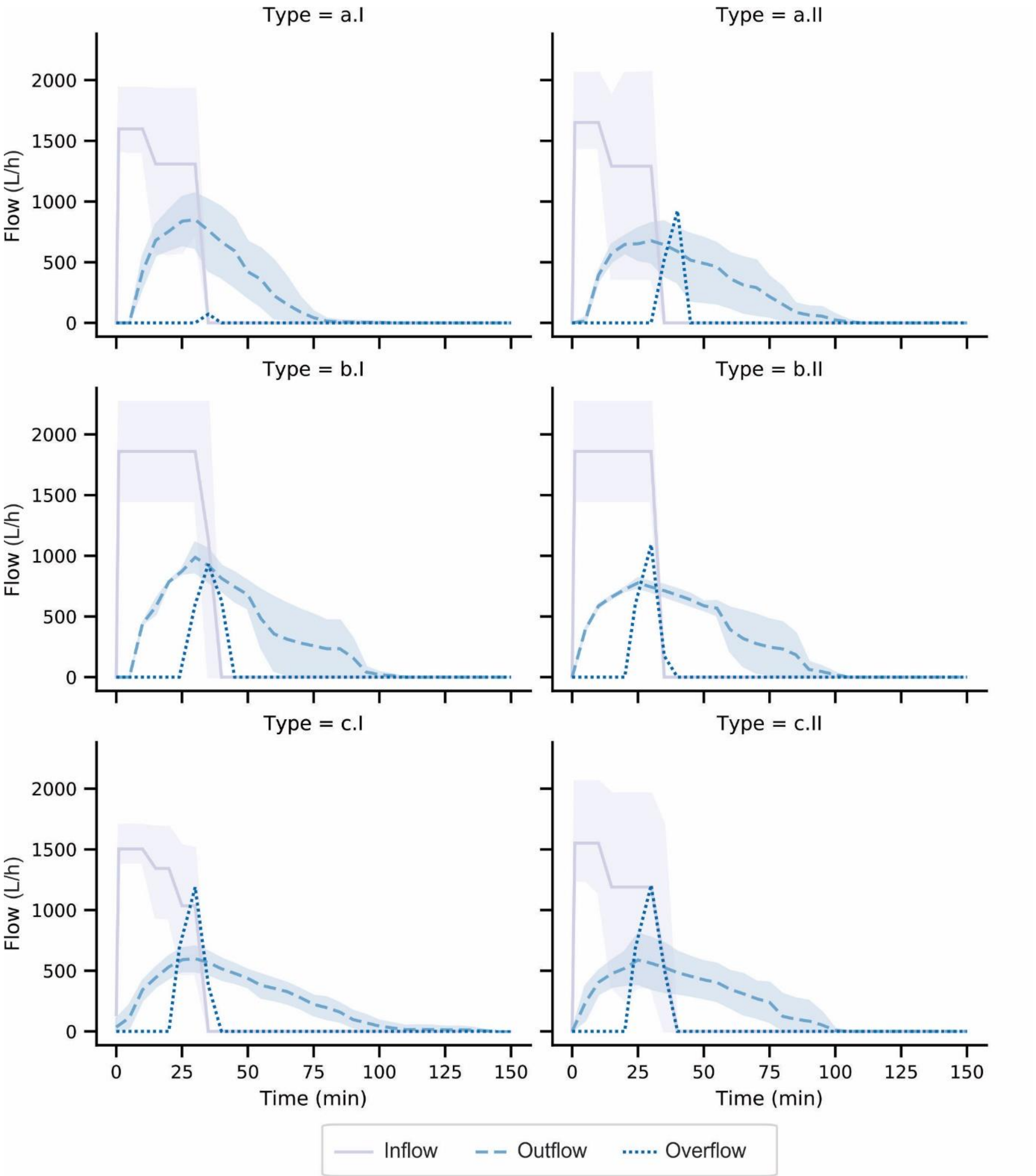

Figure 5.4 - Hydrographs for all events evaluated, grouped by different configurations in terms of saturated zone and underdrain head loss, for RP of 5 years and 50 years.

However, for events with less recurrence and more extremes (RP = 50 years and d = 30min, condition 2), the presence of a saturated zone and greater head loss in the underdrain leads to a greater overflow and a slight reduction in outflow. For the condition II this behavior can be explained by the restriction of the maximum flow supported by the underdrain due to the insertion of the head loss, limiting the infiltration into the filtering media and, therefore, increasing ponding depth. In the case of condition b and c, the presence of the saturated zone reduces the initial useful storage volume in the filtering media, leading to a greater overflow volume. Conditions b and c, however, are not much affected by the insertion of additional head loss in the underdrain.

Figure 5.5a shows the intervals for the different water balance variables, considering the different systems. When comparing the "configurations I and II", it is possible to note that "configuration II" in general result in greater overflow, less outflow and less storage, which is in agreement with previous discussions. For "configurations a, b and c", there is an increase in storage when comparing "configurations a and b", and the storage is kept constant when comparing "configurations a and c". Although this result seems controversial in previous discussions, the storage considered in this water balance jointly considers the water stored in the filtering media and in the ponding zone. Therefore, as almost all flow is retained for type 1 events there is no big difference in total storage. Type b events have higher storage values because they also have a higher central inflow value.

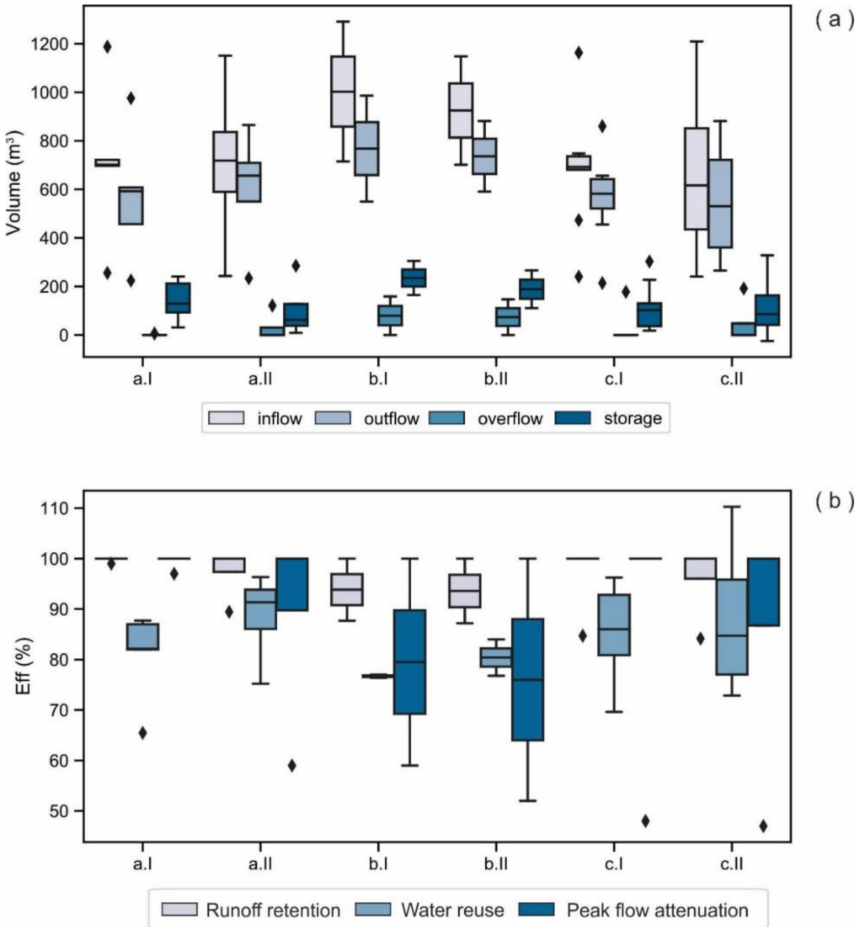

Figure 5.5 - Boxplot for different configurations considering: (a) water balance variables and (b) efficiencies of runoff retention, water reuse and peak flow attenuation

Although the boxplot with the water balance variables present a good overview of the behavior of bioretention for different configurations, the comparisons between them are difficult due to the different range of inflow evaluated in each one. Despite remaining within the acceptable limit to still be considered events of the same magnitude for type 1 and type 2, there is an impossibility to always place the same flow in the distribution pump between different events, leading to variability in the inflow. Therefore, in Figure 5.5b we present the boxplots with the runoff retention, peak attenuation and water reuse efficiency values, as a more uniform scale measure for the different configurations and we evaluated the difference between them using the Kruskall-Wallis distribution test. The distribution test for runoff retention efficiency (statistics = 2.313 and p-value = 0.804), water reuse efficiency (statistcs = 3.715 and p-value = 0.591) and peak flow attenuation (statistcs = 2.215 and p-value = 0.819) failed to reject the null hypothesis, concluding that there is no difference in the median of the efficiencies for the different configurations evaluated.

This result seems contradictory when comparing with the discussions and behaviors presented in the hydrographs in Figure 5.4. However, as shown in Table 5.1, type 2 events (RP = 50 years), which are responsible for producing overflow, were carried out in lower number when compared to type 1 events, mainly for “configurations a and c”, then considered outliers for these configurations (Figure 5.5b). Since the median is a central value measure that is more resistant to outliers (Helsel et al., 2020), type 1 events have greater influence on the distribution and central value and, therefore, the Kruskall-Wallis test failed in reject the null hypothesis.

The choice of the best configuration used for a bioretention practice must be made according to the main design purpose. When designed for the purpose of mitigating extreme rainfall events, the runoff retention and peak flow attenuation efficiencies should be assessed. For this mitigation purpose the choice of bioretention without a saturated zone (type a) seems to be more appropriate, considering the results shown in the hydrographs (Figure 5.4). However, in this study the number of extreme events evaluated was limited. Therefore, we recommend future evaluations incorporating more extreme events, which allows to perform a test of central values in more representative distributions, in order to assess whether in fact the adoption of a saturated zone leads to large losses in peak flow attenuation efficiency.

5.3.2 Runoff quantity results

The improvement of water quality by the adoption of bioretention practices was also assessed. For this purpose, 10 events were evaluated, for “configurations a and c”, as they presented the greater difference regarding water balance variables. A previous evaluation for all configurations did not show any difference in the water quality from different retention times when comparing the type I and II configurations. Therefore, the experiments continued to be conducted only with condition I, as this proves to be more efficient when assessing the water balance.

The average pollutographs with their confidence intervals obtained for configurations a and c can be seen in Figure 5.6 in terms of concentration. For Cr, Cu, Pb, Mn, Ni, Cd the samples had concentrations lower than the detection limits, thus, no pollutographs were constructed for these pollutants. In general, for the “configuration a” (Figure 5.6 - Type = a) in terms of concentration, we can observe that almost all pollutants have peak values for outflow concentration higher than the inflow. Exceptions can be noted for COD, PO_4, NO_2, Zn and TC in terms of average value, however, even for these pollutants the upper confidence interval sometimes exceeds or equals the inflow concentrations. Also noteworthy are the parameters that present the greatest export: NO_3, NH_4, Fe and *E. coli*, in addition to the turbidity and color, which are not measured in concentration.

For the nitrogen series, several studies have also obtained export of total nitrogen or its fractions in bioretention systems without a saturated zone (Chahal et al., 2016; Davis et al., 2006; Mangangka et al., 2015; Payne et al., 2014a). The export of this nutrient in vegetated systems and with water storage occurs mainly due to two processes, which prevails depending on the characteristics of the filtering media and the system configuration: (1) The first hypothesis is due to the initial composition of the filtering media and presence of plants; There may be initial amounts of high nutrients or release of nitrogen due to the death of the plants, which leached from the soil along with the water movement (Payne et al., 2014b). Because nitrogen is more mobile than phosphorus (mainly the NO_3 fractions when compared to PO_4), due to its higher solubility, low adsorption and low sedimentation, the leaching process is more remarkable for this nutrient (Laurenson et al., 2013; Roy-Poirier et al., 2010); (2) The second hypothesis is due to the natural processes of the nitrogen cycle that occur intra-events. During periods of drought and in the presence of aerobic environments, there is the transformation of NH_4 into NO_2 and later NO_3, which accumulates in the water and are later released. The presence of a saturated zone can assist in the denitrification process, converting the residual

NO_3 into nitrogen gas, removing it permanently from the system (Payne et al., 2014a; Payne et al., 2014b).

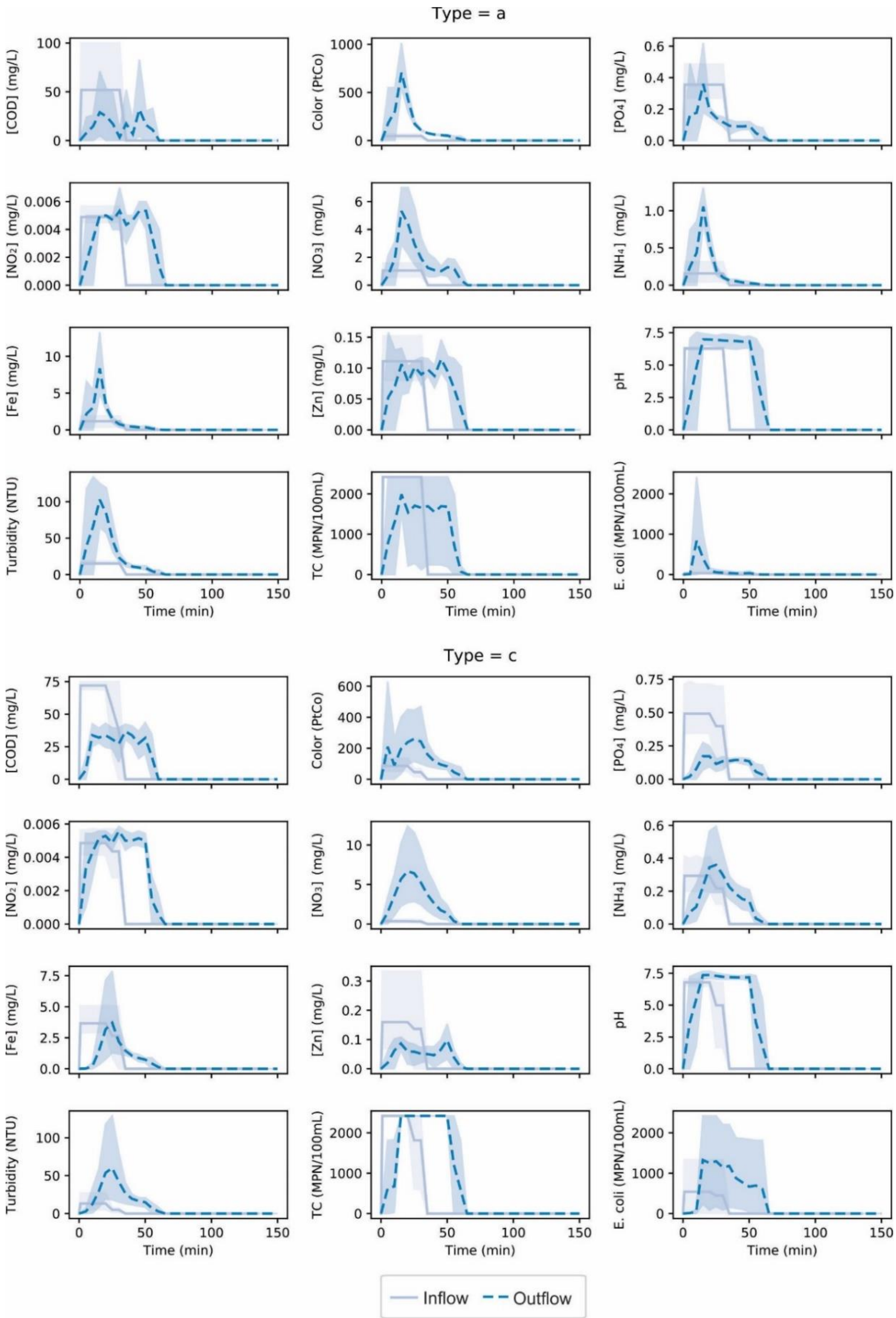

Figure 5.6 - Pollutographs in terms of concentration for the different water quality parameters evaluated, considering different the different types of configuration

The export of iron can be explained due to the composition of the filtering media. For this bioretention system, the filtering media is composed by 20% of natural soil and 80% of coarse sand, according to recommendations of The Prince George's County (2007). The natural soil of the Sao Carlos region, however, is predominantly of the type Red-Yellow Oxisoil (Macedo et al., 2019a), presenting large amounts of ferrous oxide goethite (FeO) and hematite (Fe_2O_3), and clay texture. Evaluating the results of color and Turbidity together, we can notice an increase of these two parameters in the outflow when compared to the inflow, indicating a greater amount of dissolved and suspended solids in the output of the systems. However, in the SS evaluation, for all events and configurations the value of this parameter in the outflow was null, showing that the solid particles in the outflow are colloidal, characteristic of soils with a clay texture. From this, we can conclude that the export of iron in these systems is due to the transport of soil particles along with the outflow. Similar results were observed by Macedo et al. (2019a), in the evaluation of bioretention applied in the field, on a neighborhood scale. They notice export of Fe in the overflow mainly in the occurrence of erosion of the top vegetated layer also composed by Red-Yellow Oxisoil.

Regarding *E. Coli*, the increased presence of this microorganism in the outflow may be related to the movement of animals on the surface of the bioretention (mainly birds) and their excrement, which are then moved along with the surface water for outflow. Another possibility is desorption after long drought events (Shen et al., 2018). However, due to the sensitivity of this parameter measurement, external contamination during the sampling process, or proliferation between collection and analysis, is also a possibility.

For “configuration c” (Figure 5.6 - Type = c), it is possible to observe a general improvement for all pollutants. For COD, and PO_4, the average values and upper limits for outflow concentration no longer exceed the inflow concentrations, and for Fe the average outflow value also decreases when compared to the inflow, having significant less export than for “configuration a”. The color and turbidity values are also lower when compared to “configuration a”, although they are still higher than the outflow. This improvement can be explained by the presence of the saturated zone providing a longer detention time for a portion of water retained between events, favoring the occurrence of physical-chemical-biological processes.

This water retention process between events is also explained by Shen et al. (2018), presenting this differentiation based on the terms "old water" - the portion retained from the previous event due to the presence of the saturated zone, and "new water" - the portion of the outflow that correspond only to the current monitored event. In "old water" the processes of sedimentation, adsorption and biological degradation are more significant, due to longer settling time, longer contact time between particle-filtering media and particle-biofilm, and higher plant uptake and evapotranspiration fluxes of the water/pollutant mixture. This process can be visually noticed when comparing samples with and without saturated zone, for apparent color (Figure 5.7). It is possible to observe the further displacement of solid peaks for “configuration c”, which corresponds to the water volume retained in the saturated zone at the beginning of the event.

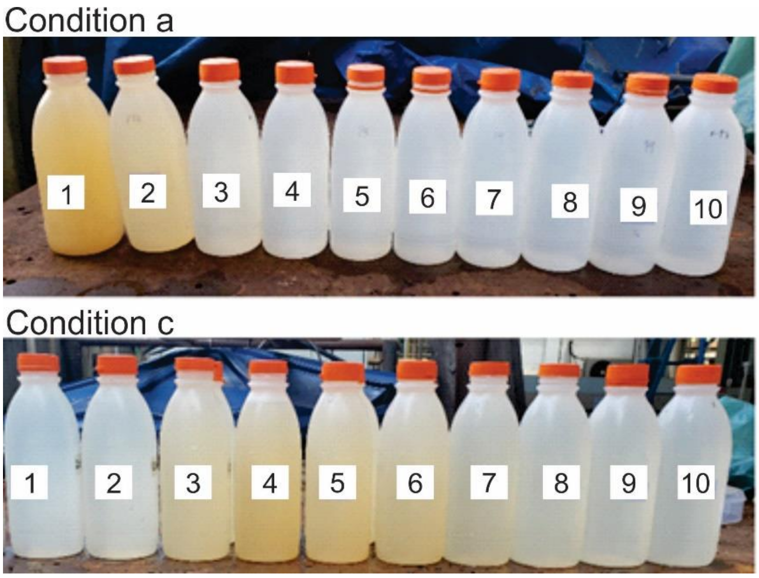

Figure 5.7 - Apparent color over samples collected at 5min intervals for (a) “configuration a” - without saturated zone and (b) “configuration c”, with saturated zone

The inclusion of the saturated zone is indicated mainly to assist in the removal of NO_3, due to the creation of an anaerobic zone allowing the denitrification process to occur. In the case of this study, it was possible to observe that the presence of the saturated zone helped to reduce NH_4, so that the longer detention time favored the occurrence of nitrification reactions and/or plant uptake. However, it was not possible to observe an improvement in the treatment of NO_3 for “configuration c”, and it was observed even an increase in its export. Two hypotheses can explain this behavior: (1) The export of NO_3 occurs mainly due to the presence of a large amount of nitrogen fractions in the filtering media and the leaching rates exceeds the denitrification rates; (2) Denitrification occurs at low rates due to a lack of sufficient dissolved carbon to serve as a energy source for denitrifying bacteria. From Figure 5.6, it is noticeable a low amount of DOC in the outflow for both configurations, and even smaller for “configuration c”. In addition, an internal carbon source was not used in this study, which is usually indicated to assist in denitrification (Payne et al., 2014b; Kim et al., 2003). In this study we aimed to evaluate the system without an additional carbon source to assess the possible contribution of denitrification in removing organic matter in the runoff and acting as a pathway to carbon sequestration.

It was also possible to observe an average increase in the concentrations of TC and *E. Coli*, for "configuration c". Stott et al. (2017) noted less microbial retention in systems with saturated zones. On the other hand, Soberg et al. (2019) found a reduction in the concentration of bacteria in the saturated zone, however, the increase in temperatures also increased the outflow concentrations. For this study, there is no conclusion due to the possibility of sample contamination.

In addition to assessments of pollutant removal in terms of concentration, assessments in terms of load are also recommended, since the effect of reducing volumes also contributes to reducing the pollution washed off downstream of the watershed, which is not verified by analysis of concentrations over time or EMC (WWEGC, 2007; Jones et al., 2008, Lago et al., 2017). Therefore, Figure 5.8 also present the pollutographs in terms of load. Comparing the inflow load values with the outflow, it is possible to notice that for almost all pollutants there is a significant reduction in the pollutant peaks, with the exception of NO_3, NH_4 and Fe for "configuration a", and only for NO_3 in "configuration c", different then when analysing only concentration values.

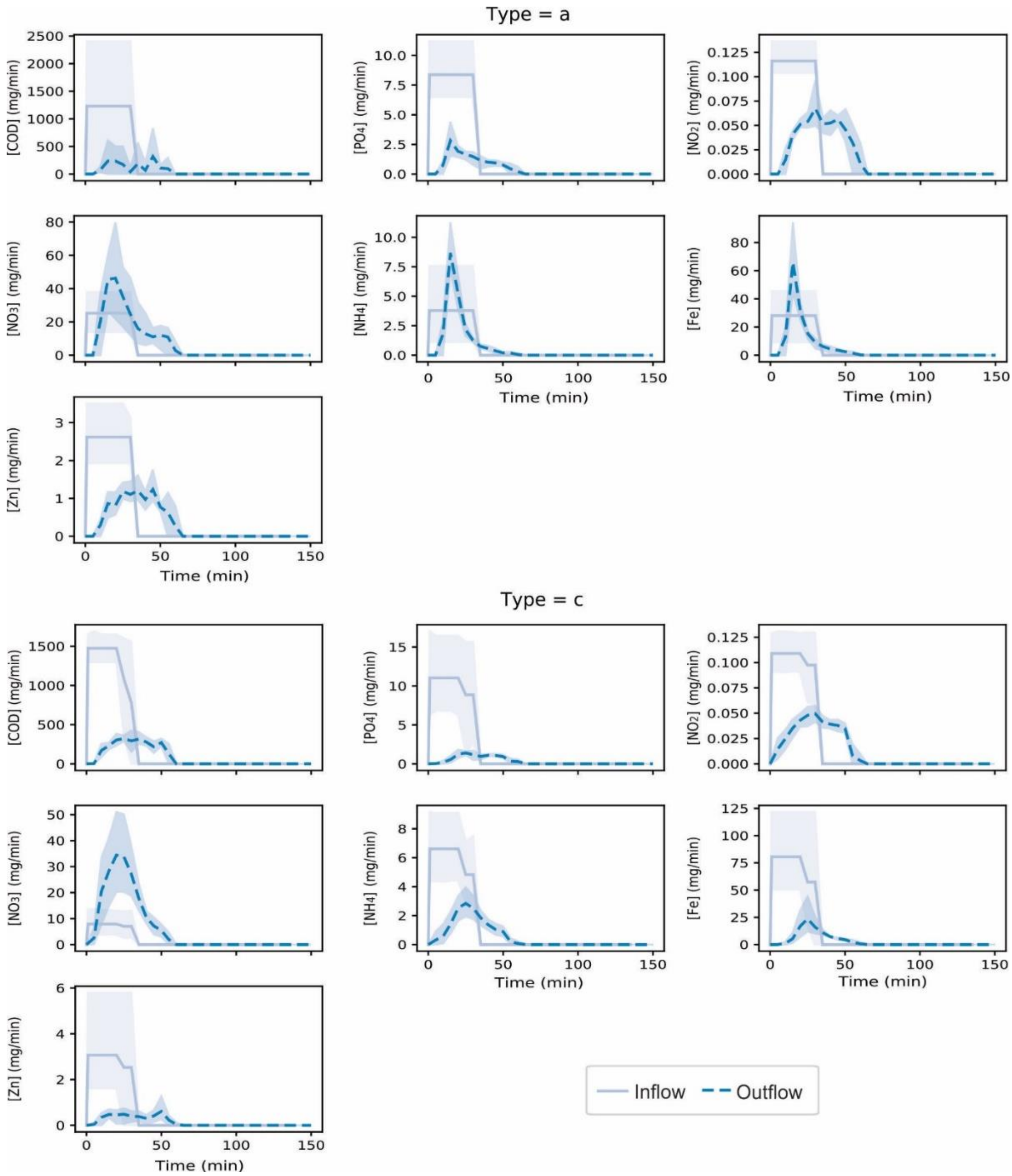

Figure 5.8 - Pollutographs in terms of load for the different water quality parameters evaluated, considering different the different types of configuration

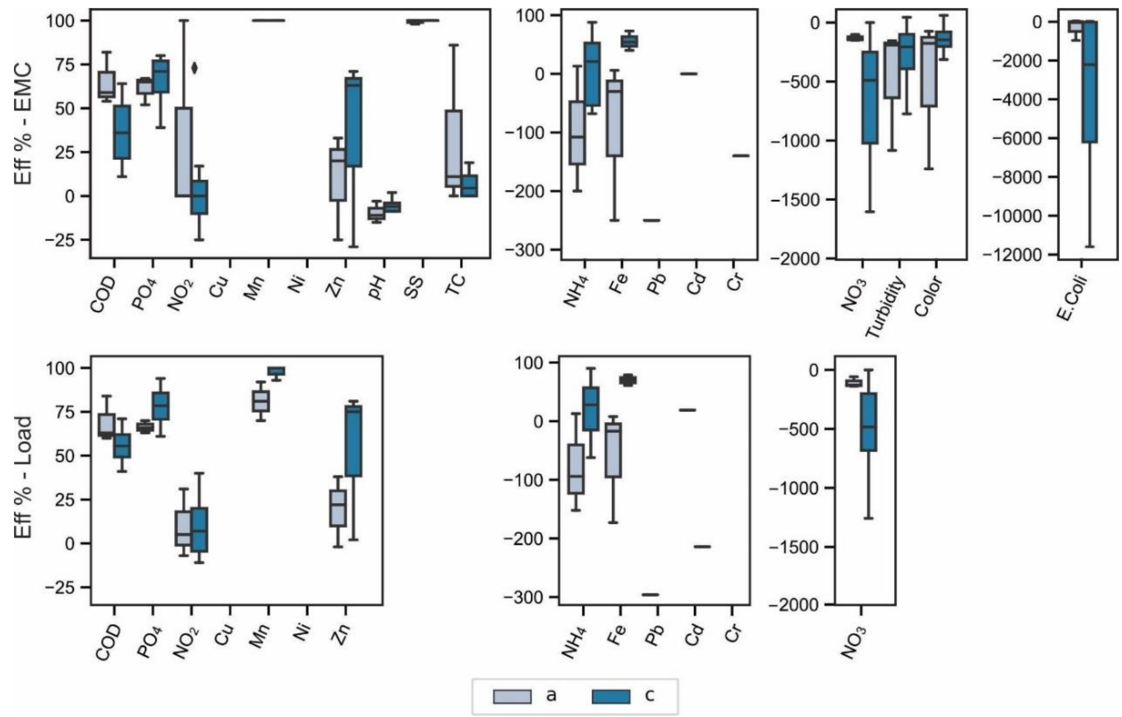

Figure 5.9 - Boxplot for pollutant removal efficiencies by configurations and by type of evaluation (EMC or load)

To assist in the interpretation of the results regarding the differences brought by the adoption of the two different configurations in the pollutant removal efficiencies, in Figure 5.9 are presented boxplots of the efficiencies in terms of EMC and load for all monitored parameters. The efficiencies for "configuration c" are in general greater than for "configuration a", except for NO_3, *E. Coli* and TC, as already observed by the pollutographs. The results of the Kruskall-Wallis test in general for all parameters showed p-value higher than the significance level for both EMC and load, failing to reject the null hypothesis, that is, presenting the same distributions, with the exception of NO_3 and NH_4 (statistics = 2.92 and p-value = 0.087) for nutrients, and Fe (statistics = 3.86 and p-value = 0.05) for metals, obtaining better efficiencies in "configuration a" for NO_3 and in "configuration c" for NH_4 and Fe.

However, the lower efficiency for NO_3 in the "configuration a" does not necessarily indicate less removal capacity caused by the presence of the saturated zone, but rather a greater conversion of NH_4 to NO_3 (since the efficiency of NH_4 removal was increased), therefore the two configurations are equally inefficient in removing this pollutant. In other words, the presence of the saturated zone without the insertion of an internal carbon source did not favor the denitrification process, as expected, demonstrating the importance of the internal carbon source.

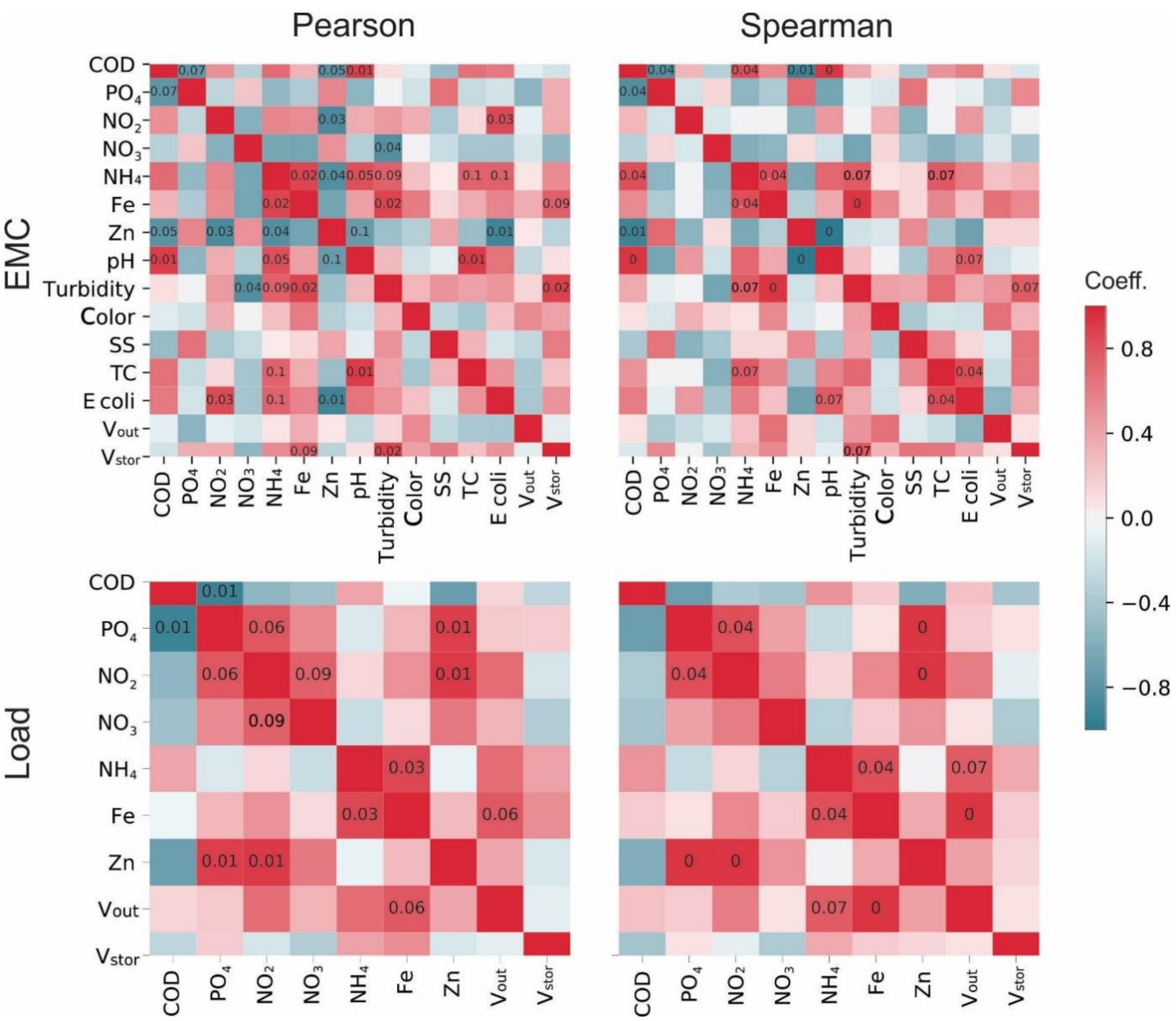

Figure 5.10 - Pearson and spearman correlation – the colors represent the Pearson and Spearman coefficients and the numbers inside the squares show significant p-value values

Finally, a correlation analysis was carried out between the different pollutants and with the stored and outflow volumes, in order to identify possible cross-influence between the variables (Figure 5.10). Pearson's correlation indicates linear correlations, while Spearman's correlation indicates diverse monotonic correlations, not necessarily linear. However, the results obtained do not seem to have physical sense and can be explained by coincident efficiencies for the different parameters.

5.3.3 Identification of clusters and their characteristics

A clustering analysis of the events was performed based on their runoff retention and water reuse efficiencies (Eff_{rr} and Eff_{wr}), EMC based pollutant removal efficiency ($Eff_{pr,EMC}$) and load based pollutant removal efficiency ($Eff_{pr,load}$). The clustering analysis aimed to identify possible configuration or hydro-meteorological patterns that determine similar water quality and water balance behaviors.

Regarding the clusters based on water balance, three main groups were identified (Figure 5.11): Group 0 = [Event 1, 2, 5, 6, 14, 18, 21, 22, a1, a2, a3, a4]; Group 1 = [Event 3, 4, 7, 13, 15, 16, 17, 19, 20] and; Group 3 = [Event 8, 9, 10, 11, 12]. By jointly evaluating the characteristics of the events from each of the groups, it is possible to observe three different hydro-meteorological patterns for each of the groupings (Table 5.1). Group 0 is formed by recurring events (5 years RP), with little dry period between them, ranging from 0 to 13 days, and APIs ranging from 6.5 to 54.9 mm, but with a greater predominance of values above 29 mm. This pattern resulted in events with intermediate outflow and Eff_{wr} values. Group 2 also has a well-defined pattern, characterized by extreme events (50 years RP), with little dry period between them (<8 days) and high APIs, ranging from 23 to 101.8 mm. This pattern resulted in overflow events and high outflow values. Finally, Group 1 has more distinct characteristics between the events. In general, the observed pattern is of events with 5 years RP, i.e. more recurrent, with large periods of previous drought, ranging from 12 to 59 days and with low API values, ranging from 3.1 to 5.0 mm. However, some exceptions are observed, events 3 and 4 have high API (46.5 and 56.8 mm, respectively) and zero previous drought days. However, these two events are type b, so the presence of a saturated zone and the day's weather conditions must have led to smaller outflows and higher Eff_{rr} and Eff_{wr}. In addition, event 7 is also an exception, as it is an event with 50 years RP, however, this event occurred after a long period of drought (112 days) with low API (3.1 mm), leading to practically no formation of overflow and great water retention in the pores, resulting in low outflow (behavior that can be noticed in Figure 5.4 for type a.I event).

As for the clustering based on $Eff_{pr,EMC}$, it was identified two main groups (Figure 5.11): Group 0 = [Event 13, 14, 15, 17, 18, 19] and Group 1 = [Event 16]. For event 16, there is a high nitrogen export in terms of concentration, a behavior that differs significantly from other events. The other variables describing the events (Table 5.1) do not present clear distinct patterns to characterize different clusters. Therefore, the high nitrogen export on event 16 is due to operation factors instead of its hydro-meteorological characteristics.

Finally, for the clustering based on $Eff_{pr,load}$ two groups were also identified (Figure 5.11): Group 0 = [Event 13,14, 15, 17, 19] and Group 1 = [Event 16, 18]. The two groups have similar characteristics regarding periods of drought and API, ranging from 12 to 59 dry days and API of 6.1 to 7.4 mm for Group 0, and 11 to 40 dry days and API of 5.0 to 10.8 mm for

Group 1. Additionally, both groups have rainfall equivalent to 5 years RP. The main differentiation between the two groups occurs in terms of AR, so that Group 1 has applied volumes of 161 and 170% of the total volume available in the technique, while group 0 has AR ranging from 236 to 255%. Smaller AR leads to smaller outflows, and consequently lower polluting loads due to volumes retention (WWEGC, 2007; Jones et al., 2008). In addition, both events are type c, which has been shown previously to have slightly lower concentrations and loads over time, and higher efficiencies than type a (Figure 5.6, 5.8 and 5.9).

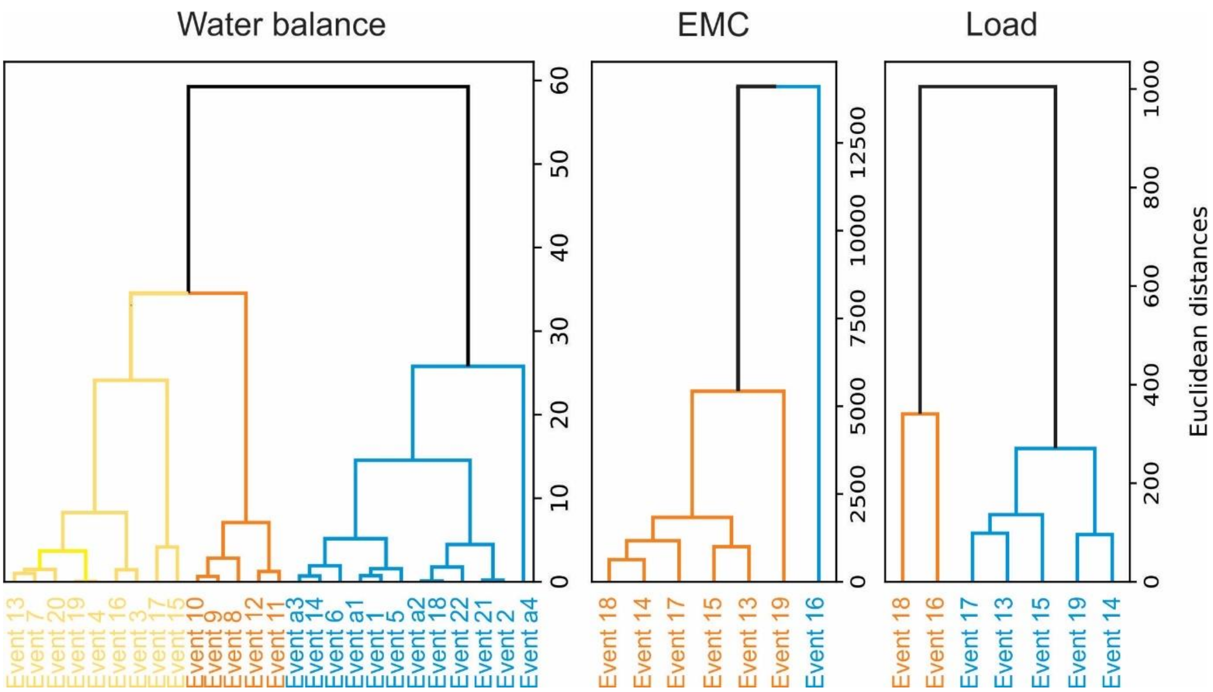

Figure 5.11 - Dendogram of the Hierarchical Agglomerative Clustering and the different clusters identified

Regarding $Eff_{pr,EMC}$ and $Eff_{pr,load}$, the Kruskall-Wallis test showed a difference between the configurations for NO_3, NH_4 and Fe pollutants. However, in the cluster analysis the efficiencies for all pollutants are assessed in an integrated manner. Therefore, the configuration of each event was not decisive in the formation of the groups. The event characteristic with the greatest influence for the cluster formations for water quality was the AR, which represents a measure of the event magnitude regarding the bioretention useful volume.

5.3.4 Contribution to SDGs from the water-energy-greenhouse gas nexus

For an initial quantification of the contribution of bioretentions to the SDGs, in addition to the individual assessment, it was raised reductions in water demand at residential and watershed scale, and reductions in energy demands and carbon emissions for watershed scale.

Table 3 shows the results of average non-potable water demand for the city of Sao Carlos, calculated based on the average demand profile for different cities in Brazil (Barreto, 2008; Sant'Ana et al., 2013; Silva, 2013; Cesar, 2016; Duarte et al., 2018) and the average water demand of 14m^3/household in Sao Carlos city (SNIS, 2018). Two types of possible reuse for households were raised, relying on the demands for water for washing machines, external use and sanitary discharge - Non potable demand (NPD) Type 1, and only external use and sanitary discharge – NPD Type 2. For the NPD-Type 1, a demand of 6.3 m^3/month was obtained, and for NPD-Type 2, a demand of 2.8 m^3/month was obtained.

Table 5.3 – Quantification of non-potable water demands for the city of Sao Carlos - SP

Type of use	Profile (%)	Volume (m^3)
External use (irrigation and other uses)	5	0.7
Sanitaty discharge	15	2.1
Washing machine	25	3.5
Total NPD-Type 1	*45*	*6.3*
Total NPD-Type 2	*20*	*2.8*

Average water demand for Sao Carlos city = 14m^3

NDP-Type1: Non-potable demands for irrigation and other outdoor uses, flushing toilets and washing machine

NDP-Type2: Non-potable demands for irrigation and other outdoor uses and flushing toilets

The quantification of the monthly volume of water recovered by the bioretention was made considering the average roof area of the residences as 94m2, in order to extrapolate the result obtained on a laboratory scale (on a scale of 1: 2) to a real scale of application. Since this study presents an initial assessment of the contribution of bioretention devices for the SDG, by quantifying the indicators proposes, we chosen to adopt the extrapolation of the water balance values obtained experimentally for the locality, instead of perform a modelling. For each rainfall event of 31 mm (equivalent to a rain of 5 years RP and 30 min duration and P90), around 2.3 m3 of water is recovered (conversion of the median of the laboratory results for the real scale).

The monthly demand values were contrasted with the volume of recovery water by the bioretention, identifying the months in which the NPD-Type 1 and Type 2 are completely supplied and how this translates into the Water Stress Reduction (WSR) indicator proposed in this study (Table 5.4). For the city of Sao Carlos, there is a strong variation between total rainfall volume in the wetter and drier months, as well as dry days, reaching average values of up to 28 days without rain in the driest month. This pattern in rainfall and dry period also leads to a big difference in the volume of recovered water. For the month of January, up to 23.4m3 of water can be recovered, which significantly exceeds non-potable demands, for both Type 1 or Type 2. However, the reuse reservoir only has a capacity of 1m3, preventing the storage of the entire recovered volume for future dry months. The excess of recovered water per event returns to the watershed through an infiltration to groundwater or as runoff to the drainage system or directly to the stream.

For the months of May to September there was a deficit in the total volume recovered water for NPD-type 1, and in July and August for NPD-type 2, i.e. the amount of water recovered is not sufficient to meet these demands. An alternative would be the installation of additional reservoir modules to store the water of the wettest months for use in the following months. With the adoption of one more module it would be possible to supply the NPD-Type2 for the entire year. For NPD-Type1, 15 more modules would be needed, which is impractical due to space and cost. However, there is already a 45% savings for 7 months of the year with reuses Type 1, which contribute to the general security of the water supply system.

Monetary savings per household were also raised by replacing the non-potable demands of the central supply system with the bioretention reuse system. For the city of Sao Carlos, the tariff currently practiced by the water supply company is 6.3 R$/m3 of water (equivalent to U$ 32.87 for the year 2020), for demand ranges from 11 to 15m3 (ARES-PCJ, 2019). Therefore, the water reuse from bioretention systems can lead to savings of up to R$ 385 per residence per year (Table 5.4).

Regarding the quality of recovered water (outflow), an assessment was made based on the resolution CONAMA 357/430 (BRAZIL – MMA, 2011), which provides quality standards for water bodies and effluent discharge. Comparing the EMC values over the events with the reference values for water quality of the water bodies, the water recovered from the bioretention was classified as Class 3, for fresh water. Class 3 waters can be destined for human demand supply, after conventional or advanced treatment, irrigation of trees, cereal and forage crops, recreation of secondary contact and animals’ consumption. Therefore, it is necessary to improve the water quality before it can be used for NPD-Type1. Improvement in the configuration (varying filtering media or saturated zone) or addition of subsequent treatment steps to improve color, TC and metal removal are necessary before the recovered water can be intended to use that require direct contact. As for the overflow or excess of recovered volume returning to the stream, the water quality agrees with the effluent discharge standards established by the resolution.

Another important standard to be evaluated in the Brazilian case is NBR 15527 (ABNT, 2019) which establishes the quality standards of rainwater for non-potable reuse, based on the parameters E. coli (<200 MPN/100mL), turbidity (<5 NTU) and pH (> 6 and <9). Considering the EMC values for the events, it is possible to notice that the events have values within the norm for pH (all events within the established range). For E. coli, 4 events had values above the norm, as well as the general average between them. As for turbidity, all events presented EMC above the norm (ranging from 15.16 NTU to 61.2 NTU). These results also demonstrate the need to improve the treatment factors internal to bioretention or the need for additional treatment to remove solids and disinfection.

Table 5.4 - Quantification of water recovery, water stress reduction, and monetary savings for a unitary bioretention system over a typical year

Month	P (mm)	Dry days	Recovered water (m3)	R$ saved NPD-Type1	R$ saved NPD-Type2	WSR NPD-Type1	WSR NPD-Type2
Jan	320.4	11	23.4	39.9	17.7	0.45	0.20
Fev	233.12	12	17.0	39.9	17.7	0.45	0.20
Mar	175.16	17	12.8	39.9	17.7	0.45	0.20
Apr	89.42	23	6.5	39.9	17.7	0.45	0.20
May	56.19	25	4.1	25.9	17.7	0.29	0.20
Jun	39.45	26	2.9	18.2	17.7	0.20	0.20
Jul	32	28	2.3	14.7	14.7	0.17	0.17
Aug	35.13	27	2.6	16.2	16.2	0.18	0.17
Sep	66.7	23	4.9	30.7	17.7	0.35	0.20
Oct	117.64	21	8.6	39.9	17.7	0.45	0.20
Nov	155.76	18	11.4	39.9	17.7	0.45	0.20
Dec	242.9	14	17.8	39.9	17.7	0.45	0.20

NDP-Type1: Non-potable demands for irrigation and other outdoor uses, flushing toilets and washing machine = 6.3 m^3 (per house)

NDP-Type2: Non-potable demands for irrigation and other outdoor uses and flushing toilets = 2.8 m^3 (per house)

Dry days: days with 0mm of precipitation

Colors - Light red: Deficit for NPD-Type1; Red: Deficit for NPD-Type1 and NPD-Type2

For the evaluation on watershed scale, its land use characteristics were raised (Figure 3) in order to evaluate the reductions in water demands, energy demand and carbon emission, in unitary measurements. Table 5 shows the results obtained for the respective indicators, considering NPD-Type 1 and Type 2. For WDR, values closer to 0 represents better system performance, while for EDR and CER, values closer to 1 represents better the system performance. These values establish an initial basis for future comparison with other studies.

Table 5.5 – Quantification of the indicators for water demand reduction rates, reduction of energy demand and reduction of carbon emissions for the hybrid system in the Mineirinho watershed over a typical year

Month	WDR_{hs} [1] NPD-Type1	WDR_{hs} [1] NPD-Type2	EDR_{hs} [2] NPD-Type2	EDR_{hs} [2] NPD-Type1	CER_{hs} [3] NPD-Type1	CER_{hs} [3] NPD-Type2	GHG emissions [4]
Jan	0.53	0.77	0.52	0.23	0.28	0.12	0.54
Fev	0.53	0.77	0.52	0.23	0.29	0.13	0.56
Mar	0.53	0.77	0.52	0.23	0.28	0.12	0.54
Apr	0.53	0.77	0.52	0.23	0.26	0.12	0.51
May	0.68	0.77	0.34	0.23	0.18	0.12	0.53
Jun	0.76	0.77	0.24	0.23	0.13	0.13	0.56
Jul	0.80	0.80	0.19	0.19	0.11	0.11	0.56
Aug	0.78	0.78	0.21	0.21	0.12	0.12	0.59
Sep	0.63	0.77	0.40	0.23	0.23	0.13	0.58
Oct	0.53	0.77	0.52	0.23	0.30	0.13	0.58
Nov	0.53	0.77	0.52	0.23	0.28	0.12	0.54
Dec	0.53	0.77	0.52	0.23	0.28	0.12	0.54

NDP-Type1: Non-potable demands for irrigation and other outdoor uses, flushing toilets and washing machine = 78750 m^3 (watershed)

NDP-Type2: Non-potable demands for irrigation and other outdoor uses and flushing toilets = 35000 m^3 (watershed)

[1] $10^{-6} m^3/m^2/m^2$

[2] $10^{-6} kWh/m^2/m^2$

[3] $10^{-6} kgCO_2/m^2/m^2$

[4] MCTIC (2020) - $kgCO_2/kWh$

These indicators were accounted by the extrapolation of an individual bioretention practice evaluate for a watershed scale. However, for better evaluation, hydrological modelling of the watershed should be carried out coupled with individual bioretention modelling for each system.

5.4 Conclusion

This study evaluated the effect of different configurations of bioretention on their performance in relation to flood control and pollutant removal, aiming to contribute to the SDGs. From the exploratory analysis it was possible to observe that the systems with saturated zone have greater ponding zone formation, which can result in greater overflow for more extreme events. In addition, the saturated zone also leads to a greater initial flow retention, reducing the outflow. From the monitored events, it was not possible to observe a significant reduction in runoff retention and peak flow attenuation efficiency by using the saturated zone. For the events only with USZ the Effrr varied from 84.45% to 100%, the Effpeak varied from

59% to 100% and the Effwr varied from 65.46% to 96.33%. As for the events only with 0.2m SZ, the Effrr varied from 84.15% to 100%, the Effpeak varied from 47% to 100% and the Effwr varied from 69.62% to 110.27%. Even though the stastical tests did not demonstrated significant differences in the efficiencies for configurations with USZ and SZ, it was possible to observe a tendency of higher peak flows in the events for the configuration with SZ. We recommend future evaluations incorporating more extreme events, which allows to perform a test of central values in more representative distributions, in order to assess whether in fact the adoption of a saturated zone leads to large losses in peak flow attenuation efficiency.

In addition, the saturated zone contributes to an improvement in pollutant removal efficiency. Although it was used to increase the removal of nitrate, favouring the nitrification process, this effect was not noticed possibly due to the absence of a carbon source in the filtering media. When comparing the results of water quality in the outflow with the guidelines and standards for freshwater and stormwater reuse in Brazil, the parameters color, turbidity, E. coli and metals were above the limits established. Therefore, it is necessary investigation of different configurations that improve the treatment process or an additional treatment aiming at solids removal and disinfection.

For the statistical survey, a cluster analysis was also carried out, which resulted in the formation of three groups regarding the efficiencies related to the water balance, and two groups to assess the efficiency of pollutant removal. The events characteristics related to hydro-meteorological condition, such as API and dry days, had more influence on the formation of groups than the different configurations of the bioretention. For the removal of pollutants, the application rates were decisive in the formation of the groups.

Finally, the evaluation of the contribution to the SDGs was made from extrapolation of the results obtained in the laboratory for household and watershed scales. Reductions in water demand of up to 45% from the centralized supply system were obtained in less restrictive non potable demands and up to 17.7% for more restrictive demands. Reductions in energy demand (EDR ranging from 0.19 to 0.52 to NPD Type 1 and 0.12 to 0.23 to NPD Type 2) and carbon emissions (CER ranging from 0.12 to 0.3 to NPD Type 1 and 0.11 to 0.13 to NPD Type 2) by the use of hybrid systems have also been quantified, serving as an initial value for comparison for future studies. This evaluation was made by extrapolating the results obtained in the laboratory for the watershed scale, which has some limitations in terms of the rainfall temporal distribution and runoff generation. Therefore, we recommend that future studies be performed through hydrological-hydraulic modeling over typical hydrological years.

References

ARES-PCJ – Agência Reguladora dos Serviços de Saneamento das Bacias dos Rios Piracicaba, Capivari e Jundiaí. (2019). *Resolução ARES-PCJ no. 291, de 20 de maio de 2019.* [ARES-PCJ Resolution no. 291, of May 20, 2019]. Available in: < https://www.saaesaocarlos.com.br/docs/ares/2019/20190702resolucao.pdf>. Access in: Oct 2020. (In Portuguese)

Arora, M., Malano, H., Davidson, B., Nelson, R., & George, B. (2015). Interactions between centralized and decentralized water systems in urban context: A review. *Wiley Interdisciplinary Reviews: Water*, 2(6), 623-634.

Barreto, D. (2008). Perfil do consumo residencial e usos finais da água. [Profile of residential consumption and end uses of water]. *Ambiente Construído*, 8(2), 23-40. (In Portuguese)

Brazil. (2020). *Relatório dos Indicadores para os Objetivos de Desenvolvimento Sustentável*. [Indicators Report for Sustainable Development Goals]. Available in: <https://odsbrasil.gov.br/relatorio/sintese>. Access in: Oct 2020. (In Portuguese)

Brazil, MMA – Ministério do Meio Ambiente, CONAMA – Conselho Nacional de Meio Ambiente. (2011). *Resolução no 430, de 13 de maio de 2011*. [Resolution 430, of May 13, 2011]. Available in: <http://www2.mma.gov.br/port/conama/legiabre.cfm?codlegi=646>. Access in: Oct 2020. (In Portuguese)

Burns, S., Arnbjerg-Nielsen, K., Hauschild, M., & Rygaard, M. (2015). The performance of rainwater tanks for stormwater retention and water supply at the household scale: An empirical study. *Hydrological Processes*, 29(1), 152-160.

C40. (2014), *Climate Action in Megacities: C40 Cities Baseline and Opportunities*. Volume 2.0. Feb, 2014. Access in: 21 out 2016. Available in: <http://issuu.com/c40cities/docs/c40_climate_action_in_megacities/149?e=10643095/6541335> .

Carter, J. G.; Cavan, G.; Connelly, A.; Guy, S.; Handley, J.; Kazmierczak, A. (2015). Climate change and the city: Building capacity for urban adaptation. *Progress in Planning*, 95, 1-66.

Cesar, B. S. S. G. (2016). *Diagnóstico do perfil de consumo e proposta de plano de contingência para edifícios residenciais em cenários de crise hídrica*. [Consumption profile diagnosis and contingency plan proposal for residential buildings in water crisis scenarios]. Graduation Thesis. Rio de Janeiro: Escola Politécnica da Universidade Federal do Rio de Janeiro. (In Portuguese)

Chahal, M., Shi, Z., & Flury, M. (2016). Nutrient leaching and copper speciation in compost-amended bioretention systems. *Science of the Total Environment*, 556, 302 - 309.

Chandrasena, G., Deletic, A., & McCarthy, D. (2016). Biofiltration for stormwater harvesting: comparison of Campylobacter spp. and Escherichia coli removal under normal and challenging operational conditions. *Journal of Hydrology*, 537, 248-259.

Charalambous, K., Bruggeman, A., Eliades, M., Camera, C., & Vassiliou, L. (2019). Stormwater retention and reuse at the residential plot level—Green roof experiment and water balance computations for long-term use in cyprus. *Water*, 11(5), 1055.

Charlesworth, S. M. (2010). A review of the adaptation and mitigation of global climate change using sustainable drainage in cities. *Journal of Water and Climate Change*, 1(3), 165-180.

Clark, R., Gonzalez, D., Dillon, O., Charles, S., Cresswell, D., & Naumann, B. (2015). Reliability of water supply from stormwater harvesting and managed aquifer recharge with a brackish aquifer in an urbanising catchment and changing climate. *Environmental Modelling & Software*, 72, 117-125.

D'Acunha, B., & Johnson, M. (2019). Water quality and greenhouse gas fluxes for stormwater detained in a constructed wetland. *Journal of Environmental Management*, 231, 1232-1240.

Davis, A.P., Shokouhian, M., Sharma, H. And Minami, C. (2006). Water quality improvement through bioretention media: Nitrogen and phosphorus removal. *Water Environmental Research*, 78(3), 284-293.

Duarte, J. M., da Silva, F. S., Brito, F. S. L., & Senado, M. C. (2018). Perfil do consumo residencial e usos finais da água em Belém do Pará. [Profile of residential consumption and end uses of water in Belém do Pará]. *Proceedings of Encontro Técnico AESABESP, 29º Congresso Nacional de Saneamento e Meio Ambiente*. São Paulo. (In Portuguese)

Eckart, K.; Mcphee, Z.; Bolisetti, T. Performance and implementation of low impact development – a review. *Science of The Total Environment*, 607, 413-432.

EMBRAPA. (2017). *Condições Meteorológicas Estação da Embrapa Pecuária Sudeste.* [Meteorological conditions station of Southern Embrapa Pecuária]. Available in: <http://www.cppse.embrapa.br/meteorologia/index.php?pg=caracterizacao>. Accessed in: 01 jun. 2017. (In Portuguese)

Fletcher, T., Deletic, A., Mitchell, V., & Hatt, B. (2008). Reuse of urban runoff in Australia: a review of recent advances and remaining challenges. *Journal of Environmental Quality*, 37(5), s-116 - s127.

Fletcher, T., Andrieu, H., & Hamel, P. (2013). Understanding, management and modelling of urban hydrology and its consequences for receiving waters: A state of the art. *Advances in Water Resources*, 51, 261-279.

Fletcher, T., Shuster, W., Hunt, W., Ashley, R., Butler, D., Arthur, S., . . . Mikkelsen, P. (2015). SUDS, LID, BMPs, WSUD and more – The evolution and application of terminology surrounding urban drainage. *Urban Water Journal*, 12(7), 525-542.

Ge, Z., Feng, C., Wang, X., & Zhang, J. (2016). Seasonal applicability of three vegetation constructed floating treatment wetlands for nutrient removal and harvesting strategy in urban stormwater retention ponds. *International Biodeterioration & Biodegradation*, 112, 80-87.

Getter, K., Rowe, D., Robertson, G., Cregg, B., & Andresen, J. (2009). Carbon sequestration potential of extensive green roofs. *Environmental Science & Technology*, 43(19), 7564-7570.

Hashemi, S., Mahmud, H., & Ashraf, M. (2015). Performance of green roofs with respect to water quality and reduction of energy consumption in tropics: a review. *Renewable and Sustainable Energy Reviews*, 52, 669-679.

Helsel, D.R., Hirsch, R.M., Ryberg, K.R., Archfield, S.A., and Gilroy, E.J., 2020, *Statistical methods in water resources: U.S. Geological Survey Techniques and Methods*, book 4, chapter A3, 458 p., https://doi.org/10.3133/tm4a3. [Supersedes USGS Techniques of Water-Resources Investigations, book 4, chapter A3, version 1.1.]

Indicator Monitoring System. (2018). Portal ODS. [SDG portal]. Available in: <http://rd.portalods.com.br/>. Access in: Oct 2020. (In Portuguese).

Karim, M., Bashar, M., & Imteaz, M. (2015). Reliability and economic analysis of urban rainwater harvesting in a megacity in Bangladesh. *Resources, Conservation and Recycling*, 104, 61-67.

Kavehei, E., Jenkins, G., Adame, M., & Lemckert, C. (2018). Carbon sequestration potential for mitigating the carbon footprint of green stormwater infrastructure. *Renewable and Sustainable Energy Reviews*, 94, 1179-1191.

Kavehei, E., Jenkins, G. A., Lemckert, C., & Adame, M. F. (2019). Carbon stocks and sequestration of stormwater bioretention/biofiltration basins. *Ecological Engineering*, 138, 227-236.

Kim, H., Seagren, E. A., & Davis, A. P. (2003). Engineered bioretention for removal of nitrate from stormwater runoff. *Water Environment Research*, 75(4), 355-367.

Lago, C. A. F.; Macedo, M. B. ; Rosa, A. ; Mendiondo, E. M. (2017). Calibração e validação quali-quantitativa do escoamento superficial - Estudo de caso na USP/São Carlos. [Calibration and quali-quantitative validation of runoff - Case study at USP / São Carlos]. *Proceedings of XXII Simpósio Brasilieiro de Recursos Hídricos*, 2017, Florianópolis. (In Portuguese).

Laurenson, G.; Laurenson, S.; Bolan, N.; Beechan, S.; Clark, I. (2013). *Advances in Agrononmy:* Chapter Four – The Role of Bioretention Systems in the Treatment of Stormwater. Elsevier.

Litern, A., Daly, E., Duncan, H., Hatt, B., Fletcher, T., & Deletic, A. (2011). Key design characteristics that influence the performance of stormwater biofilters. *Proceeding 12th International Conference on Urban Drainage*. Porto Alegre, Brazil.

Macedo, M., Lago, C., Rosa, A., & Mendiondo, E. (2017). Técnicas compensatórias de bioretenção para cidades resilientes: integração com nexus–água, energia, alimento. [Low Impact Development practices of bioretention to resilient cities: integration with the water-energy-food nexus]. *Proceedings of XXII Simposio Brasileiro de Recursos Hidricos*. Florianopolis, Brazil. (in Portuguese).

Macedo, M. B. D., Lago, C. A. F. D., Mendiondo, E. M., & Souza, V. C. B. D. (2018). Performance of bioretention experimental devices: contrasting laboratory and field scales through controlled experiments. *RBRH*, 23.

Macedo, M., Lago, C., & Mendiondo, E. (2019a). Stormwater volume reduction and water quality improvement by bioretention: Potentials and challenges for water security in a subtropical catchment. *Science of the Total Environment*, 647, 923-931.

Macedo, M., Lago, C., Mendiondo, E., & Giacomoni, M. (2019b). Bioretention performance under different rainfall regimes in subtropical. *Journal of Environmental Management*, 248, 109266.

Maglionico, M. (1998*). Indagine sperimentale e simulazione numerica degli aspetti qualitativi dei deflussi nelle reti di drenaggio urbano*. [Experimental investigation and numerical simulation of the qualitative aspects of outflows in urban drainage networks]. 1998. 217 f. Doctoral Thesis - Università di Bologna, Bologna. (In Italian)

Mangangka, I., Liu, A., Egodawatta, P., & Goonetilleke, A. (2015). Performance characterization of a stormwater treatment bioretention basin. *Journal of Environment Management*, 150, 173 - 178.

MCTIC – Ministério da Ciência Tecnologia e Inovações. (2020). *Arquivos dos Fatores de Emissão da Margem de Operação Pelo Método da Análise de Despacho*. [Operating Margin Emission Factors Files by Dispatch Analysis Method]. Available in: < http://antigo.mctic.gov.br/mctic/opencms/ciencia/SEPED/clima/textogeral/emissao_despacho.html>. Access in: Oct 2020. (In Portuguese)

Mitchell, V. G., McCarthy, D. T., Deletic, A., & Fletcher, T. D. (2008). Urban stormwater harvesting–sensitivity of a storage behavior model. *Environmental Modelling & Software*, 23(6), 782-793.

Nair, S., George, B., Malano, H., Arora, M., & Nawarathna, B. (2014). Water–energy–greenhouse gas nexus of urban water systems: Review of concepts, state-of-art and methods. *Resources, Conservation and Recycling*, 89, 1-10.

Ng, K., Herrero, P., Hatt, B., Farrelly, M., & McCarthy, D. (2018). Biofilters for urban agriculture: Metal uptake of vegetables irrigated with stormwater. *Ecological Engineering*, 122, 177-186.

Jones, J. Clary, J.; Strecker, E.; Quigley, M. 15 reasons why you should think twice before using percent removal to assess BMP performance. *Stormwater*, Guest Editorial, January/February, 2008.

Our World in Data. (2019). *Urbanization across the world today*. Available in: <https://ourworldindata.org/urbanization#number-of-people-living-in-urban-areas>. Access in: Oct 2020.

Pandit, A., Minné, E. A., Li, F., Brown, H., Jeong, H., James, J. A. C., ... & Yang, P. (2017). Infrastructure ecology: an evolving paradigm for sustainable urban development. *Journal of Cleaner Production*, 163, S19-S27.

Payne, E. G. I.; Pham, T.; Cook, P. L. M.; Fletcher, T. D.; Hatt, B. E.; Deletic, A. (2014a). Biofilter design for effective nitrogen removal from stormwater–influence of plant species, inflow hydrology and use of a saturated zone. *Water Science and Technology*, 69(6), 1312-1319.

Payne, E. G., Fletcher, T. D., Russell, D. G., Grace, M. R., Cavagnaro, T. R., Evrard, V., ... & Cook, P. L. (2014b). Temporary storage or permanent removal? The division of nitrogen between biotic assimilation and denitrification in stormwater biofiltration systems. *PloS one*, 9(3), e90890.

Petit-Boix, A., Sevigné-Itoiz, E., Rojas-Gutierrez, L. A., Barbassa, A. P., Josa, A., Rieradevall, J., & Gabarrell, X. (2015). Environmental and economic assessment of a pilot stormwater infiltration system for flood prevention in Brazil. *Ecological engineering*, 84, 194-201.

Petit-Boix, A., Sevigné-Itoiz, E., Rojas-Gutierrez, L. A., Barbassa, A. P., Josa, A., Rieradevall, J., & Gabarrell, X. (2017). Floods and consequential life cycle assessment: Integrating flood damage into the environmental assessment of stormwater Best Management Practices. *Journal of cleaner production*, 162, 601-608.

Petit-Boix, A., Deykota, J., Phillips, R., VargasParra, M., Josa, A., Gabarrell, X., . . . Apul, D. (2018). Life cycle and hydrologic modeling of rainwater harvesting in urban neighborhoods: implications of urban form and water demand patterns in the US and Spain. *Science of the Total Environment*, 621, 434-443.

PMSC – Prefeitura Municipal de São Carlos. (2016). *Plano Diretor do Município de São Carlos. Lei no 18.053, de 19 de dezembro de 2016* [Master Plan of the Municipality of São Carlos. Law 18.053 of December 19, 2016]. Available in: < https://leismunicipais.com.br/plano-diretor-sao-carlos-sp>. Access in: 11 sep 2019. (in Portuguese)

Richards, R. C., Rerolle, J., Aronson, J., Pereira, P. H., Gonçalves, H., & Brancalion, P. H. S. (2015). Governing a pioneer program on payment for watershed services: Stakeholder involvement, legal frameworks and early lessons from the Atlantic forest of Brazil. *Ecosystem Services: Science, Policy and Practice*, 16, 23–32.

Roy-Poirier, A., Champagne, P., & Filion, Y. (2010). Bioretention processes for phosphorus pollution control. *Environmental Reviews*, 18(NA), 159-173.

Sant'Ana, D., Boeger, L., & Monteiro, L. (2014). Aproveitamento de águas pluviais e o reúso de águas cinzas em edifícios residenciais de Brasília–parte 1: reduções no consumo de água. [Use of rainwater and the reuse of gray water in residential buildings in Brasília - part 1: reductions in water consumption]. *Água e Sociedade*, (10), 77-84. (In Portuguese)

Sapkota, M., Arora, M., Malano, H., Moglia, M., Sharma, A., George, B., & Pamminger, F. (2015). An overview of hybrid water supply systems in the context of urban water management: Challenges and opportunities. *Water*, 7(1), 153-174.

Shen, P., Deletic, A., Urich, C., Chandrasena, G. I., & McCarthy, D. T. (2018). Stormwater biofilter treatment model for faecal microorganisms. *Science of the total environment*, 630, 992-1002.

Silva, L. M. (2013). *Estudo da demanda de água não potável e desenvolvimento de um sistema de reuso de água cinza para habitações de interesse social*. [Study of non-potable water demand and development of a gray water reuse system for social housing]. Master Thesis. Vitória: Universidade Federal do Espírito Santo. (In Portuguese)

SNIS – Sistema Nacional de Informações sobre Saneamento. (2013). Consumo de água: Veja indicadores por cidade do Brasil - SÃO CARLOS, SP. [Water consumption: See indicators by city in Brazil - SÃO CARLOS, SP]. Available in: < http://www.deepask.com/goes?page=sao-carlos/SP-Consumo-de-agua:-Veja-indicadores-da-sua-cidade>. Access in: Oct 2020. (In Portuguese)

SNIS – Sistema Nacional de Informações sobre Saneamento. (2018). Série Histórica. [Historical Serie]. Available in: < http://app4.mdr.gov.br/serieHistorica/>. Access in: Oct, 2020.

Søberg, L. C., Viklander, M., Blecken, G. T., & Hedström, A. (2019). Reduction of Escherichia coli, Enterococcus faecalis and Pseudomonas aeruginosa in stormwater bioretention: Effect of drying, temperature and submerged zone. *Journal of Hydrology* X, 3, 100025.

Stott, R., Tondera, K., Blecken, G. T., & Schreiber, C. (2018). Microbial loads and removal efficiency under varying flows. *In Ecotechnologies for the treatment of variable stormwater and wastewater flows* (pp. 57-74). Springer, Cham.

The Prince George's County. (2007). *Bioretention Manual*. Environmental Services Division, Department of Environmental Resources.

UN – United Nations. (2020). *The 17 Goals*. Available in: < https://sdgs.un.org/goals>. Acess in: Oct 2020.

UNSTATS. (2020). *Global indicator framework adopted by the General Assembly* (A/RES/71/313), annual refinements contained in E/CN.3/2018/2 (Annex II), E/CN.3/2019/2 (Annex II), and 2020 Comprehensive Review changes (Annex II) and annual refinements (Annex III) contained in E/CN.3/2020/2. Available in: <https://unstats.un.org/sdgs/indicators/indicators-list/>. Access in: Oct 2020.

VO, P. T., Ngo, H. H., Guo, W., Zhou, J. L., Nguyen, P. D., Listowski, A., & Wang, X. C. (2014). A mini-review on the impacts of climate change on wastewater reclamation and reuse. *Science of the Total Environment*, 494, 9-17.

Waterways, M. (2005). *Water sensitive urban design: technical design guidelines for South East Queensland*. Moreton Bay Waterways and Catchment Partnership.

WWEGC - Wright Water Engineers and Geosyntec Consultants (2007). Frequently Asked Questions Fact Sheet for the International Stormwater BMP Database: Why does the International Stormwater BMP Database Project omit percent removal as a measure of BMP performance? (as posted on www.bmpdatabase.org)

6 PROCESS-BASED MODEL FOR TREATMENT OF NITROGEN FRACTIONS IN BIORETENTION

A diferent version of this chapter was submited as: Marina Batalini de MACEDO, Eduardo Mario MENDIONDO, David MCCARTHY. **Process-based model for treatment of nitrogen fractions in biofilters**. 15th International Conference on Urban Drainage.

Abstract

LID practices can be used to mitigate extremes of floods and pollution, and new perspectives also include their use to contribute to circular cities and to the *water-energy-food nexus*. However, there is still a gap in relation to the use of these practices for nutrient cycling and contribution to the "food" component of the nexus. Studies on the nutrient removal in vegetated LIDs, in particular bioretention, present a wide range of variability in the treatment efficiencies, remaining difficult to understand the key factors that influence nutrient removal and treatment pathways. One way of assessing the key factors for the treatment processes and identifying the main parameters that deserve more attention in design and in monitoring studies is the use of process-based modeling followed by a sensitivity analysis, which allows to identifying the most sensitive parameters. There are several studies in the literature aiming at the modeling of nutrients (especially nitrogen) in vegetated techniques such as bioretention, wetlands and green roofs. However, most models have simplified approaches (not considering all processes involved in nitrogen treatment) or are not process-based (using linear models, correlation or artificial intelligence). Therefore, this study aimed to build a process-based modeling for nitrogen fractions. The model was based on representing the bioretention in a three-bucket approach and is divided between a water flow module and the nitrogen quality module. The initial sensitivity analysis showed that the parameters related to the nitrification process are those with the highest sensitivity, which are related to the filtering media and biofilm formation. This study presented an initial assessment of the model, which still needs to be further investigated.

Keywords: Low Impact Development; Bioretention modelling; Nitrate; Ammonia; Sensitivity analysis

6.1 Introduction

Extreme hydrological events, such as droughts and floods, are one of the main causes of disasters worldwide, and it tends to be aggravated by climate and land use changes (Carter et al., 2015). Consequently, risks to the population increases, once flood events in urban centres become more frequent and the parallel between higher demand and resources scarcity contributes to increase water, energy and food insecurity. In this context, the approach *water-energy-food nexus* emerges, integrating these three resources to turn the society more resilient and increase sustainable communities, aiming to the UN Sustainable Development Goals (SDG) (Hoff, 2011). Low Impact Development (LID) practices can be used as a tool to achieve more resilient cities and communities, if integrating purposes of runoff retention, water quality improvement and stormwater harvesting, for others then just water reuse (Fletcher et al., 2015, Macedo et al., 2017).

There are still limitations on LID real, practical and low-cost employability facing climate change, mitigation, adaptation and re-use for water-energy-food security risks. Considering the design and operation, the key-factors influencing in the water quality improvement, stormwater harvesting, and runoff retention still need to be explored better. After identified these factors, they should be equated to allow multi-objective optimization for water reuse and flood control. Moreover, these gaps intensify especially in cities with a subtropical climate, since the most of studies are conducted in temperate areas, where geoclimatic, sanitary and social conditions are very different from those in subtropical climate. Thus, adapting LID systems for tropical and subtropical regions is still a shortcoming.

One of the ways to explore different future design and operation scenarios, allowing to identify its key factors, is from the modelling of the systems. Several models have already been developed to simulate the hydraulic-hydrological behaviour in different LID systems. However, there are still few models for nutrients, especially for total nitrogen or its fractions. Therefore, in this study we aim to present an initial approach to a process-based model to simulate the ammonia and nitrate fractions in a biofilter system, with the processes of nitrification, denitrification, adsorption and plant uptake.

Initially, a literature review was made in order to identify the main process involved in the nitrogen treatment in a bioretention device, the equations used to describe this process and other models developed to nitrogen treatment in LID practices.

Laurenson et al. (2013), Chen et al. (2013) and Payne et al. (2014) presents as the main transformations occurring in bioretention devices regarding nitrogen treatment the following: nitrification and denitrification, adsorption (most important to NH_4), filtration (most important to NH_4), plant and microbial uptake, dissimilatory nitrate reduction. However, not all of this reactions and process have the same importance in the treatment. In order to develop a process-based model, it is necessary to simplify the treatment, by identifying the governing process. Therefore, for the nitrogen fractions, the governing process to me modeled would be nitrification and denitrification, adsorption and nutrient uptake. Payne et al. (2014) found that nutrient uptake is more important for nitrogen removal then denitrification, even in practices with saturated zones. The saturated zone, instead of favoring the occurrence of denitrification, increases the retention time, the water movement through plant roots, and, consequently, uptake by plants.

The nitrogen cycling is composed by two main process, the nitrification (that can be divided in ammonia oxidation and nitrite oxidation, (Eq. 6.1), which is an aerobic reaction and causes acidification in the soil (Eq. 6.2), and the denitrification (Eq. 6.3), which is an anaerobic process that results in the release of carbon dioxide and nitrogen gas. These chemical reactions can be represented as first or zero order reactions, according to the oxidation degree, or by Michaelis-Mentem equations, developed firstly to simulate enzymatic reactions (Barber, 1984).

$$NH_4^+ \rightarrow NO_2^- \leftrightarrow NO_3^- \qquad \text{(Eq. 6.1)}$$

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + H_2O + 2H^+ \qquad \text{(Eq. 6.2)}$$

$$C_6H_{12}O_6 + 4NO_3^- \rightarrow 6CO_2 + 6H_2O + 2N_2 \qquad \text{(Eq. 6.3)}$$

The nitrification process was represented by different equations in different studies. Palfy et al. (2017) have developed a model of NH_4 removal for wetlands, that can be adapted to bioretention. In this model, the nitrification is represented by first-order kinetic equations (Eq. 6.4), but with decay rates depending on temperature (Eq. 6.5). Also, the nitrification is considered an inter-event process in wetlands (due to the necessity of O_2).

$$NH_4N_{(t+1)} = NH_4N_{(t)}e^{-d_{i,t}} \qquad \text{(Eq. 6.4)}$$

$$d_{i,t} = d_{i,t}.e^{\frac{T-20}{Rc}} \qquad \text{(Eq. 6.5)}$$

where: NH_4N is the mass of NH_4N at time step t, $d_{i,t}$ is the rate constant. The $d_{i,t}$ constant is corrected in function of temperature T (˚C) and the temperature sensibility constant Rc.

Kirk and Kronzucker (2005) simulated the nitrogen uptake by plants in wetland soils. They also present a formulation for nitrification but using Michaelis-Menten equation (Eq. 6.6).

$$R_{nit} = Vm_{nit} \frac{[O_2]_L}{Km_{nit1} + [O_2]_L} \cdot \frac{[NH_4]_L}{Km_{nit2} + [NH_4]_L} \qquad \text{(Eq. 6.6)}$$

where: R_{nit} is the rate of the nitrification process, Vm_{nit} is the maximum rate of nitrification, Km_{nit1} is rhe Michaelis constant for nitrification, regarding O_2 consumption, Km_{nit2} is the Michaelis constant for nitrification, regarding NH_4 consumption.

Another model commonly used for lake modeling is the AED model (Hipsey et al., 2013). In this model they present the mass balance equations according to Eq. 6.7, and the process parametrization is made using Michaelis-Menten equations to simulate nitrification process in the lakes, with a correction factor for temperature (Eq. 6.8 and 6.9).

$$\frac{dNH_4}{dt} = f_{sed}^{NH4} + f_{miner}^{NH4} + f_{nitri}^{NH4} - uptake\ from\ phytoplancton^{NH4} \qquad \text{(Eq. 6.7)}$$

$$f_{nitri}^{NH4} = R_{nitr} \frac{[O_2]}{K_{nitri} + [O_2]} (\theta_{nitr})^{T-20} [NH_4] \qquad \text{(Eq. 6.8)}$$

where: f_{sed} is the rate of NH4 sedimanted, f_{miner} is the rate of NH_4 mineralized, f_{nitri} is the rate of NH_4 nitrified, R_{nitri} is the maximum rate of nitrification, K_{nitri} is the Michaelis constant for nitrification, regarding O_2 consumption, θ_{nitr} is is a temperature multiplier for temperature dependence of nitrification and T is temperature.

For the denitrification process, Lynn et al. (2017) have developed a model of denitrifying in biofilters, to be coupled with SWMM. They have adopted a first order PFR kinetics equation, to represent denitrification (Eq. 9). In this model, they aim to include the release of DOC from a carbon source as one of the factors to be analyzed. Therefore, k_1 is dependent of dissolved organic carbon (DOC) and dissolved oxygen (DO) (Eq. 6.10 and Eq. 6.11). The DOC is provided by the release of woodchips (Eq. 6.12). The NO_3 and bDOC effluent concentrations were then obtained by mass balance equations, for each time interval.

$$N_{Ri} = N_{Ii}\, e^{(-k_{1i} V_P / 3600 Q_i)} \qquad \text{(Eq. 6.9)}$$

$$k_{1i} = k . O_{fi} . bDOC_{fi} \qquad \text{(Eq. 6.10)}$$

$$O_{fi} = \frac{K_{O2}}{K_{O2} + O_{2Ii}} \qquad \text{(Eq. 6.11)}$$

$$bDOC_{fi} = \frac{bDOC_{Ei-1} + bDOC_d \Delta t}{bDOC_{Ei-1} + bDOC_d \Delta t + K_{bDOC}} \qquad \text{(Eq. 6.12)}$$

where: N_{Ri} is the nitrate concentration after denitrifying, N_{Ii} is the influent nitrate concentration, k_{1i} is the first order denitrification constant, V_P is the biofilter pore volume, Q_i is the flow volume, k is the maximum denitrification rate constant, O_{fi} is oxygen inhibition factor, $bDOC_{fi}$ is DOC limitation factor, K_{O2} is oxygen inhibition coefficient for denitrification, O_{2Ii} is influent DO concentration, $bDOC_{Ei-1}$ is the biofilter pore water bDOC concentration from the previous time step, $bDOC_d$ is the dissolution rate, Δt is the time step, and K_{bDOC} is the bDOC half maximum rate concentration for denitrification.

Kirk and Kronzucker (2005) on the other hand, present a different formulation for denitrification. The denitrification process is considered in the reaction component in the transport equations, using a Michaelis-Menten equation (Eq. 6.13). To this model, it is also necessary to quantify the movements of O_2 in the soil. This model will be further presented with more details.

$$R_{denit} = I_{denit} Vm_{denit} \frac{[NO_3]_L}{K_{mdenit} + [NO_3]_L} \quad \text{(Eq. 6.13)}$$

$$I_{denit} = \begin{cases} if\ [O_2] \geq\ Km_o, I_{denit} = 0 \\ if\ [O_2] < Km_{o,}\ I_{denit} = 1 - \frac{[O_2]}{Km_o} \end{cases} \quad \text{(Eq. 6.14)}$$

where: I_{denit} is an adjustment variable for denitrification regarding aerobic conditions, Vm_{denit} is the maximum rate of denitrification, K_{mdenit} is the Michaelis constant for denitrification, Km_o is the Michaelis constant for O_2 consumption.

AED model (Hipsey et al., 2013) also presents equations to simulate the denitrification fluxes in lakes, using mass balance and Michaelis-Menten equations with temperature adjustement (Eq. 6.15 and Eq. 6.16).

$$\frac{dNO_3}{dt} = -f_{sed}^{NO3} + f_{nitri}^{NH4} - f_{denitri}^{NO3} -\ uptake\ from\ phytoplancton^{NO3} \quad \text{(Eq. 6.15)}$$

$$f_{dentri}^{NO3} =\ R_{denitr}\ \frac{K_{denitri}}{K_{denitri} + [O_2]}\ (\theta_{denitr})^{T-20}\ [NO_3] \quad \text{(Eq. 6.16)}$$

where: f_{sed} is the rate of sedimentation for NO_3, $f_{denitri}$ is the rate of denitrification, R_{denitr} is the maximum rate of denitrification, K_{denitr} is the Michaelis constant for denitrification, θ_{denitr} is a temperature multiplier for temperature dependence of denitrification.

In addition to the reactions that occur in the nitrogen cycle process, the soil sorption process is also important for the ammoniacal fraction of nitrogen. This process can be considered based on the use of isotherms, from simpler models such as linear isotherms, to more complex ones, such as Freundlich and Langmuir. Randelociv et al. (2016), in their model for micropollutants uses linear isotherms with additional equations to represent the empty and available spaces in the soil to be considered in the linear equation. In the adaptation by Shen et al. (2018) to *E. Coli*, it was adopted the use of simple linear isotherms, but simulating the adsorption and desorption processes separately in different equations, because they assume that these two processes occur at different speeds and times in the bioretention soil mix.

The process of nutrients absorption in plants is usually represented by the Michaelis-Menten equation and is used as boundary conditions in nutrient uptake models, in agronomical studies (plants nutrition, soil nutrient bioavailability, among others). The models are built with transport equations (diffusion and mass flow). For plant nutrition in normal soils, Barber (1984) present the mechanistic model in Eq. 6.17.

$$\frac{\partial C_l}{\partial t} = \frac{1}{r}\frac{\partial}{\partial r}\left(rD_e\,\frac{\partial C_l}{\partial r} + \frac{r_0 v_0 C_l}{b}\right) \qquad \text{(Eq. 6.17)}$$

where: D_e is the effective diffusion coefficient, r is the radial distance, C_l is the concentration of ions in the soil solution, v_0 is the rate of water flux into the root, r_0 is the root radius and b is the relation between the concentration of ions in the solid phase and liquid phase.

The assumptions of this model are: the soil is homogenous and isotropic, moisture conditions are maintained constant near field capacity, nutrient uptake occurs only from nutrients in solution, root exudates or microbial activity does not affect nutrient flux, nutrients are moved to the roots by a combination of mass flow and diffusion, the relation between net influx and concentration can be described by Michaelis-Menten kinetics, the roots are assumed to be smooth cylinders, D_e and b are assumed independent of concentration, influx characteristics are not changed by root age, influx is independent of the rate of water absorption. The initial and boundary conditions are:

- For t = 0: $C_{li} = C_{l0}$
- For r = r_0 and t > 0, the uptake follows Michaelis-Menten kinetics: $J_r = D_e\frac{\partial C_s}{\partial r} + v_0 C_l = \frac{I_{max} C_l}{K_m + C_l}$
- For r = r_1 and t > 0, and roots do not compete for nutrients: $C_l = C_{li}$

- For r = r_1 and t > 0, and concentration gradients extending from adjacent roots do overlap: $J_r = 0$

Kirk and Kronzucker (2005) have made an application of Barber (1984) model to quantify the fluxes of O_2, NH_4 and NO_3 in an anoxic flooded soil near a cylindrical root, to represent the process occurring in a wetland. In Kirk and Kronzucker (2005) model, the equations representing the nitrification (Eq. 6.6) and denitrification (Eq. 6.13 and Eq. 6.14) are also added in Eq. 6.17, when evaluating NH_4 and NO_3. The plant uptake equations by Michaelis-Menten (J_r) equation continue to be used as boundary conditions.

In addition to the studies already presented so far, which have considered the processes of nitrification, denitrification and plant uptake in their models, some other studies have been developed in modelling nitrogen treatment in bioretention. Li et al. (2018) used the Hydrus-1D model to simulate the parameters COD, NO_3, NH_4, TN and TP. Hydrus-1D uses the solute transport equation in the soil, which simulates the processes of diffusion and advection by water flow for the liquid fraction, and sorption for the soil/liquid interaction. The model can consider a source sink time, but this term for each of the analyzed parameters was not specified in the study by Li et al. (2018). Christianson et al. (2004) also modelled the effectiveness of bioretention cells in North America, and they used Green-Ampt infiltration model to water flow simulation and Freundlich isotherms to water quality assessment. Also, Christianson et al. (2013) modelled the TN load reduction in using a simple linear equation correlating detention time with nitrogen removal.

It is possible to observe a lack of process-based models considering all the governing treatment process in bioretention devices, specially nitrification, denitrification, and plant uptake. The use of process-based model has the advantage of even in the absence of data for the evaluation site, the adoption of literature values for the constants can provide a good idea of the behavior expected. In addition, by performing sensitivity analysis in the design-related parameters, it is possible to identify the key-factors in design regarding the treatment process. Therefore, in this chapter, we propose to adapt a process-based model using solute transport equations in soil to NH_4 and NO_3 treatment in bioretention devices.

6.2 Materials and methods

6.2.1 Process-based model development

Based on previous studies by Shen et al. (2018), Randelovic et al. (2016), the model developed employs the “three-bucket” approach (Figure 6.1), in which each bucket represents the main zones found in a biofilter: ponding zone, unsaturated zone and saturated zone. As a complement to the previous models, in this study we added the possibility to simulate a configuration without saturated zone (since it is one common configuration in Brazil), which can be defined by the height of the underdrain outlet structure. The model is also separated into two modules: water flow module and nitrogen quality module. The water flow module is an independent module, while the nitrogen quality module depends on the results of water balance state variables in their calculation. Here, we will present the two modules separately.

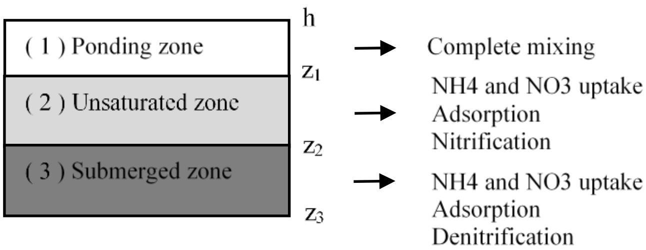

Figure 6.1 - Three-bucket considered to the model and the governing process in each of them

6.2.1.1 Water flow module

The water flow module aims to describe the hydraulic-hydrological behaviour of a bioretention system, in a continuous simulation. The main simulated processes consist of infiltration from the ponding zone to the unsaturated zone, infiltration from the unsaturated zone to the saturated zone, outflow from an underdrain, overflow from a weir, evapotranspiration and capillary rise. This module have used the same equations proposed by Shen et al. (2018), with a few changes to incorporate the possibility of simulations without saturated zone. Additionally, the code for this module was updated to fix problems in water balance. In this section we present all the equations used to this module. The variables for all the equations are presented in Table 6.1 and the parameters are described in Table 6.2.

Table 6.1 - Variables of the water flow module

Symbol	Meaning	Dimensions
$Q_{in,i}$	Biofilter inflow, in time i	m^3/s
$Q_{inf,p,i}$	Infiltration to surrounding soil, in time i	m^3/s
$S_{pz,i}$	Storage in ponding zone, in time i	m^3
$Q_{over,i}$	Overflow through weir, in time i	m^3/s
$Q_{pf,i}$	Infiltration to USZ, in time i	m^3/s
$S_{usz,i}$	Storage in soil mix, in time i	m^3
$h_{usz,i}$	Depth of the unsaturated zone, in time i	m
$Q_{fs,i}$	Infiltration to SZ, in time i	m^3/s
$h_{sz,i}$	Depth of the saturated zone, in time i	m
$Q_{et,i}$	Total evapotranspiration flow, in time i	m^3/s
$Q_{et,usz,i}$	Total evapotranspiration flow in USZ, in time i	m^3/s
$Q_{et,sz,i}$	Total evapotranspiration flow in SZ, in time i	m^3/s
$Q_{hc,i}$	Capillary rise flow, in time i	m^3/s
$Q_{pipe,i}$	Drainage pipe flow, in time i	m^3/s
$Q_{inf,sz,i}$	Infiltration to bottom surrounding soil, in time i	m^3/s
$h_{pz,i}$	Ponding depth, in time i	m
$n_{usz,i}$	Porosity in unsaturated zone	
$n_{sz,i}$	Porosity in saturated zone	
$\theta_{usz,i}$	Soil water fraction by volume in unsaturated zone	
$\theta_{sz,i}$	Soil water fraction by volume in saturated zone	

Table 6.2 - Parameters of the water flow module

Symbol	Meaning	Dimensions
A_b	Bioretention surface area	m^2
Ap	Filter surface area	m^2
nf	Soil mix porosity	
nt	Transition layer porosity	
ng	Gravel porosity	
Df	Soil mix height	m
Dt	Transition layer height	m
Dg	Gravel height	m
P_v	Weir height	m
kexp	Weir exponential coefficient	
kweir	Weir coefficient	
S_{fc}	Field capacity/effective moisture	
S_h	Hygroscopic point moisture	
S_w	Wilting point moisture	
S_s	Saturation as threshold for plants to reach potential ET	
γ	Saturated curve parameter	
k_s	Hydraulic conductivity coefficient of soil mix	m/s
Cd	Discharge coefficient	
d_{orif}	Hole diameter	mm
Kf	Hydraulic conductivity coefficient of surrounding soil	m/s
g	Gravity aceleration	m/s^2
h_{pipe}	Height of underdrain	m
Kc	Plant coefficient for evapotranspiration	

The equations used to represent each of the hydrological/hydraulic process are described above (Eq. 6.1 to Eq. 6.25)

Evapotranspiration:

$$Q_{et}{}^{i} = \begin{cases} 0\,, & if\ S_t{}^{i} \leq S_w \\ A_b K_c ET_0 \frac{S_t{}^{i} - S_w}{S_s - S_w}\,, & if\ S_w < S_t{}^{i} \leq S_s \ \text{(mm/min)} \\ A_b K_c ET_0\,, & if\ S_s < S_t{}^{i} \leq 1 \end{cases} \quad \text{(Eq. 6.1)}$$

$$Q_{et}{}^{i} = Q_{et}{}^{i} / (1000.60)\ (\text{m}^3/\text{s}) \quad \text{(Eq. 6.2)}$$

$$S_t{}^{i} = \frac{S^{i}\, n_{usz}{}^{i}\, h_{usz}{}^{i} + n_{sz}{}^{i}\, h_{sz}{}^{i}}{n_{usz}{}^{i}\, h_{usz}{}^{i} + n_{sz}{}^{i}\, h_{sz}{}^{i}} \quad \text{(Eq. 6.3)}$$

Ponding zone:

Water mass balance

$$\frac{\partial S_{pz}}{\partial t} = Q_{in}{}^{i} + Q_{rain}{}^{i} - Q_{pf}{}^{i} - Q_{over}{}^{i} - Q_{inf,p}{}^{i} \qquad \text{(Eq. 6.4)}$$

State variables equations

$$S_{pz}{}^{i} = h_{pz}{}^{i} . A_b \qquad \text{(Eq. 6.5)}$$

$$Q_{pf}{}^{i} = \min\left(\frac{k_s A_p\left(h_{pz}{}^{i} + h_{usz}{}^{i}\right)}{h_{usz}{}^{i}},\ \frac{h_{pz}{}^{i} A_b}{dt} - Q_{inf,p}{}^{i},\frac{\left(1 - S^{i}\right) n_{usz}{}^{i}\, h_{usz}{}^{i}\, A_p}{dt}\right) \qquad \text{(Eq. 6.6)}$$

$$Q_{over}{}^{i} = \begin{cases} 0 & ,\ if\ h_{pz}{}^{i} \leq P_v \\ \min\left(\frac{A_p\left(h_{pz}{}^{i} - P_v\right)}{dt},\ kweir\ 2g^{0.5}\left(h_{pz}{}^{i} - P_v\right)^{kexp}\right), & if\ h_{pz}{}^{i} > P_v \end{cases} \qquad \text{(Eq. 6.7)}$$

$$Q_{inf,pz}{}^{i} = K_f\left[\left(A_b - A_p\right) + C_s P_p h_{pz}{}^{i}\right] \qquad \text{(Eq. 6.8)}$$

$$h_{pz}{}^{i} = h_{pz}{}^{i-1} + \frac{\left(Q_{in}{}^{i} + Q_{rain}{}^{i} - Q_{pf}{}^{i} - Q_{over}{}^{i} - Q_{inf,p}{}^{i}\right)dt}{A_b} \qquad \text{(Eq. 6.9)}$$

Unsaturated zone (USZ):

Water mass balance

$$\frac{\partial S_{usz}}{\partial t} = Q_{pf}{}^{i} + Q_{hc}{}^{i} - Q_{fs}{}^{i} - Q_{et,usz}{}^{i} \qquad \text{(Eq. 6.10)}$$

State variables equations

$$Q_{et,usz}{}^{i} = Q_{et}{}^{i} \frac{S_{usz}{}^{i}\, n_{usz}{}^{i}\, h_{usz}{}^{i}}{n_{usz}{}^{i}\, h_{usz}{}^{i} + n_{sz}{}^{i}\, h_{sz}{}^{i}} \qquad \text{(Eq. 6.11)}$$

$$Q_{hc}{}^{i} = \begin{cases} A_p Cr\left(S_{usz}{}^{i} - S_s\right)\left(S_{fc} - S_{usz}{}^{i}\right), & if\ S_s \leq S_{usz}{}^{i} \leq S_{fc} \\ 0 & \end{cases} \qquad \text{(Eq. 6.12)}$$

$$Cr = \frac{4 K_c ET_0}{2.5\left(S_{fc} - S_s\right)^2} \qquad \text{(Eq. 6.13)}$$

$$Q_{fs}{}^{i} = \begin{cases} min\left(\frac{k_s A_p\left(h_{pz}{}^{i} + h_{usz}{}^{i}\right)}{h_{usz}{}^{i}}\, S^{i\gamma}, \frac{\left(S_{usz}{}^{i} - S_{fc}\right) A_p h_{usz}{}^{i} n_{usz}{}^{i}}{dt}\right), & if\ S_{usz}{}^{i} \geq S_{fc} \\ 0 & ,if\ S_{usz}{}^{i} < S_{fc} \end{cases} \qquad \text{(Eq. 6.14)}$$

$$h_{usz}{}^{i} = L - h_{sz}{}^{i-1} \qquad \text{(Eq. 6.15)}$$

$$n_{usz}{}^{i} = \begin{cases} \frac{n_f D_f + n_g\left(D_g - h_{sz}{}^{i-1}\right)}{h_{usz}{}^{i}} = \ , & if\ h_{sz}{}^{i-1} < D_g \\ n_f\ , & if\ h_{sz}{}^{i-1} \geq D_g \end{cases}$$ (Eq. 6.16)

$$\theta_{usz}{}^{i} = S_{usz}{}^{i} n_{usz}{}^{i}$$ (Eq. 6.17)

$$S_{usz}{}^{i} = max\left[min\left(1,\ S_{usz}{}^{i-1} h_{usz}{}^{i-1} n_{usz}{}^{i-1} + \frac{\left(Q_{pf}{}^{i} + Q_{hc}{}^{i} - Q_{fs}{}^{i} - Q_{et,usz}{}^{i}\right)dt}{A_p h_{usz}{}^{i} n_{usz}{}^{i}}\right), S_h\right]$$ (Eq. 6.18)

Saturated zone (SZ):

Water mass balance

$$\frac{\partial S_{sz}}{\partial t} = Q_{fs}{}^{i} - Q_{hc}{}^{i} - Q_{et,sz}{}^{i} - Q_{pipe}{}^{i} - Q_{inf,sz}{}^{i}$$ (Eq. 6.19)

$S_{sz} = 1$ (constant)

State variables equations

$$Q_{et,sz}{}^{i} = Q_{et}{}^{i} - Q_{et,usz}{}^{i}$$ (Eq. 6.20)

$$Q_{pipe}{}^{i} = \begin{cases} min\left(\frac{\left(h_{sz}{}^{i} - h_{pipe}\right)A_p n_{usz}{}^{i}}{dt} - Q_{inf,sz}{}^{i},\ C_d A_{pipe}\left[\left(h_{sz}{}^{i} - h_{pipe}\right)2g\right]^{1/2}\right), & if\ h_{sz}{}^{i} \geq h_{pipe} \\ 0\ , & if\ h_{sz}{}^{i} < h_{pipe} \end{cases}$$

(Eq. 6.21)

$$Q_{inf,sz}{}^{i} = K_f\left(A_p + C_s P_p h_{sz}{}^{i}\right)$$ (Eq. 6.22)

$$h_{sz}{}^{i} = h_{sz}{}^{i-1} + \frac{\left(Q_{fs}{}^{i} - Q_{hc}{}^{i} - Q_{et,sz}{}^{i} - Q_{pipe}{}^{i} - Q_{inf,sz}{}^{i}\right)dt}{A_p\, n_{sz}{}^{i-1}}$$ (Eq. 6.23)

$$n_{sz}{}^{i} = \begin{cases} \frac{n_g D_g + n_t D_t + n_f\left(h_{sz}{}^{i} - D_g - D_t\right)}{h_{sz}{}^{i}}, & if\ D_t + D_g < h_{sz}{}^{i} \leq L \\ \frac{n_g D_g + n_t\left(h_{sz}{}^{i} - D_g\right)}{h_{sz}{}^{i}}, & if\ D_g < h_{sz}{}^{i} \leq D_t + D_g \\ n_g\ , & if\ h_{sz}{}^{i} \leq D_g \end{cases}$$ (Eq. 6.24)

$$\theta_{sz}{}^{i} = n_{sz}{}^{i}$$ (Eq. 6.25)

6.2.1.2 Nutrient quality module

For the quality module, each of the buckets is simulated considering the predominant processes in each of the zones. For the ponding zone, it is considered that there is a complete mixing occurring, with denitrification reactions predominating (for high flooding periods). The mass balance in this zone follows Eq. 6.27. For the unsaturated zone and saturated zone, the mass balance is made for different layers within each zone, using the solute transport equation in soils, adding a source sink and plant uptake terms (Eq. 6.28). The process of interaction between soil – liquid phase of the pollutant is considered only for ammonia since nitrate is highly soluble. This process is simulated through a linear adsorption equation (Eq. 6.29), but considering a desorption factor separately, once these processes predominate in different times in the biofilter media (Shen et al., 2018). The source sink term is calculated according to reactions occurring in each zone, represented by the Eq. 6.34 to Eq. 6.48. For plant uptake, the model uses the Michalis-Menten equation (Eq. 3.5.54 to Eq. 6.61).

The advection term in is calculated by upwind or central differences in space and forward in time, depending on the value of Peclet number, while the dispersion term is approximated by central differences in space and forward in time (Randelovic et al., 2016). In this section we present all the equations used to this module. The variables for all the equations are presented in Table 6.3 and the parameters are described in Table 6.4.

Table 6.3 - Variables of the nitrogen quality module

Symbol	Meaning	Dimensions
$C_{l,i}$	Solute concentration in water phase, in layer l and time i	mg/L
$C_{s,i}$	Solute concentration in soil phase, in layer l and time i	mg/Kgsolo
$Q_{in,i}$	Biofilter inflow, in time i	m3/s
$Q_{inf,p,i}$	Infiltration to surrounding soil, in time i	m3/s
$Q_{over,i}$	Overflow through weir, in time i	m3/s
$Q_{pf,i}$	Infiltration to USZ, in time i	m3/s
$Q_{fs,i}$	Infiltration to SZ, in time i	m3/s
$Q_{et,i}$	Total evapotranspiration flux, in time i	m3/s
$Q_{hc,i}$	Capillary rise flow, in time i	m3/s
$Q_{pipe,i}$	Drainage pipe flow, in time i	m3/s
$Q_{inf,sz,i}$	Infiltration to bottom surrounding soil, in time i	m3/s
$q_{l,i}$	unit water flux, in layer l and time i	m/s
$Rx_{l,i}$	Reactions rate, in layer l and time i	mg/L/s
A	Biofilter area	m^2
h_i	Ponding depth, in time i	m

Table 6.4 - Parameters of the nitrogen quality module

Symbol	Meaning	Dimensions
θi	Soil water fraction by volume	L/L
ρ	Soil bulk desisty	kg/L
D_e	Solute diffusion coefficient in water, for each type of solute	m2/s
f	Soil diffusion impedance factor	
k_{ads}	Adsorption rate	1/s
k_{des}	Desorption rate	1/s
$k_{microbial}$	Constant for microbial reaction in soil	1/s
K_{O2}	Oxygen inhibition coefficient for denitrification	mg/L
k_{O2}	Constant for oxygen consumption in pore water	1/s
f_{bDOC}	Fraction of inflow DOC biodegradable	
bDOCd	DOC dissolution rate	mg/Ls
K_{bDOC}	bDOC half-maximum rate concentration for denitrification	mg/L
k_{DOC}	Constant for DOC consumption in pore water	1/s
k_{nit}	Maximum nitrification rate constant	1/s
k_{denit}	Maximum denitrification rate constant	1/s

For ponding depth, complete mixing assumed and reactions of denitrification and settling occurring:

$$M_{pz}^{i} = M_{in}^{i} - M_{pf}^{i} - M_{over}^{i} \quad \text{(Eq. 6.26)}$$

$$\frac{\partial (C_{pz}^{i+1}\, h_{pz}^{i+1}\, A_b)}{\partial t} = C_{in}^{i} Q_{in}^{i} - C_{pz}^{i-1}\left(Q_{pf}^{i} + Q_{over}^{i} + Q_{inf,p}^{i}\right) + R_{pz}^{i} \quad \text{(Eq. 6.27)}$$

For USZ and SZ, mass balance equations for a given solute C, represented by solute transport process:

$$\frac{\partial \theta_{usz}^{i+1}\, C_l^{i+1}}{\partial t} + \rho \frac{\partial C_{s_l}^{i+1}}{\partial t} = \frac{\partial}{\partial z}\left(D\, \theta_{usz}^{i} \frac{\partial C_l^{i}}{\partial z}\right) - \frac{\partial\, q_l^{i}\, C_l^{i}}{\partial z} + R_l^{i} + \emptyset\, J_l^{i} \quad \text{(Eq. 6.28)}$$

$$\rho \frac{\partial C_{s_l}^{i+1}}{\partial t} = \theta_{usz}^{i}\, k_{ads,usz}\, C_l^{i} - \rho\, k_{des,usz} C_{s_l}^{i} + R_{s_l}^{i} \quad \text{(Eq. 6.29)}$$

The linear adsorption and linear desorption are considered separately, according to Shen et al. (2018).

$$q_l^i = \frac{\alpha_l({Q_{pf}}^i - {Q_{et1}}^i) + \beta_l({Q_{fs}}^i - {Q_{hc}}^i)}{A_b}, \quad \text{or} \quad V_l^i = \frac{q_l^i}{\theta_{usz}^i} \qquad \text{(Eq. 6.30)}$$

$$\frac{\partial \theta_{sz}^{i+1}\, C_j^{i+1}}{\partial t} + \rho \frac{\partial C_{s_j}^{i+1}}{\partial t} = \frac{\partial}{\partial z}\left(D\, \theta_{sz}^i \frac{\partial C_j^i}{\partial z}\right) - \frac{\partial\, q_j^i\, C_j^i}{\partial z} + R_j^i + \emptyset J_j^i \qquad \text{(Eq. 6.31)}$$

$$\rho \frac{\partial C_{s_j}^{i+1}}{\partial t} = \theta_{sz}^i\, k_{ads,sz}\, C_j^i - \rho\, k_{des,sz} C_{s_j}^i + R_{s_j}^i \qquad \text{(Eq. 6.32)}$$

$$q_j^i = \frac{\alpha_j({Q_{fs}}^i - {Q_{hc}}^i - {Q_{et}}^i) + \beta_j({Q_{pipe}}^i + {Q_{inf,sz}}^i)}{A_b}, \quad \text{or} \quad V_j^i = \frac{q_j^i}{\theta_{sz}^i} \qquad \text{(Eq. 6.33)}$$

Boundary conditions:

α + β = 1, upper boundary: α = 1, lower boundary: β = 1

Definition of reactions, according to the different layers and different pollutants:

$$R_{pz}^i = -R_{denit,pz}^i - R_{set}^i \qquad \text{(Eq. 6.34)}$$

$R_{set}^i = can\ be\ estimeted\ by\ a\ settling\ equation\ for\ particels\ with\ nitrogen\ adsorb,$ *p. ex. Stokes' law.*

$$R_{denit,pz}^i = -k_{denit,pz}\, [NO_3]_{pz}^i \qquad \text{(Eq. 6.35)}$$

$$R_l^i = {R_{nit}}_l^i,\ {R_O}_l^i,\ {R_{DOC}}_l^i \qquad \text{(Eq. 6.36)}$$

$$ {R_{nit}}_l^i = \pm k_{nit}\, [NH_4]_l^i\ , R_{nit} = \begin{cases} +, if\ \ NO_3 \\ -, if\ NH_4 \end{cases} \qquad \text{(Eq. 6.37)}$$

$$ {R_O}_l^i = -k_{o2}[O_2]_j^i - 2\, k_{nit}\, [NH_4]_j^i \qquad \text{(Eq. 6.38)}$$

$$ {R_{DOC}}_l^i = -k_{DOC}[DOC]_l^i \qquad \text{(Eq. 6.39)}$$

$$R_{s_l}^i = {R_{NH4}}_{s_l}^i, {R_{DOC}}_{s_l}^i \qquad \text{(Eq. 6.40)}$$

$$ {R_{NH4}}_{s_l}^i = -k_{mb\_nh4,usz}[NH_4]_{s_l}^i \qquad \text{(Eq. 6.41)}$$

$$ {R_{DOC}}_{s_l}^i = -k_{mb\_doc,usz}[DOC]_{s_l}^i \qquad \text{(Eq. 6.42)}$$

$$R_j^i = {R_{nit}}_j^i,\ {R_{nit}}_j^i + {R_{denit}}_j^i,\ {R_O}_j^i,\ {R_{DOC}}_j^i \qquad \text{(Eq. 6.43)}$$

$$R_{nit}{}_{j}^{i} = \pm\, k_{nit,sz}\, [NH_4]_j^i \;\;, R_{nit} = \begin{cases} +\,, if\;\; NO_3 \\ -, if\; NH_4 \end{cases} \quad \text{(Eq. 6.44)}$$

$$R_{denit}{}_{j}^{i} = \, - \, k_2^i\, [NO_3]_j^i \quad \text{(Eq. 6.45)}$$

$$k_2^i = k\; O_f^i\; bDOC_f^i \quad \text{(Eq. 6.46)}$$

$$O_{f_j}^i = \frac{K_{O2}}{K_{O2} + [O_2]_{j-1}^i} \quad \text{(Eq. 6.47)}$$

$$bDOC_f^i \;\; = \frac{bDOC_E^{i-1} + bDOC_d\, \Delta t}{bDOC_E^{i-1} + bDOC_d\, \Delta t + K_{bDOC}} \quad \text{(Eq. 6.48)}$$

If considering a carbon source, there must be considered an additional inflow DOC, from the released of the carbon source. In the paper of Lynn et al. (2017) they have adopted the concentration equation (instead of rate) and have obtained the effluent concentration by mass balance equations.

$$R_{O}{}_{j}^{i} = \, -k_{o2}[O_2]_j^i - \; 2\; k_{nit}\; [NH_4]_j^i \quad \text{(Eq. 6.49)}$$

$$R_{DOC}{}_{j}^{i} = \, -k_{DOC}[DOC]_j^i + bDOCd \quad \text{(Eq. 6.50)}$$

$$R_{s_j}^i = R_{NH4}{}_{s_j}^{i} \,, R_{DOC}{}_{s_j}^{i} \quad \text{(Eq. 6.51)}$$

$$R_{NH4}{}_{s_j}^{i} = -k_{mb\_nh4,sz}[NH_4]_{s_j}^i \quad \text{(Eq. 6.52)}$$

$$R_{DOC}{}_{s_j}^{i} = -k_{mb\_doc,sz}[DOC]_{s_j}^i \quad \text{(Eq. 6.53)}$$

The plant uptake follows Michaelis-Menten kinetics, close to the root. To simplificate the model and avoid a second transport equation to be solved in the limits of the root zone influence, a fraction of the roots in the soil is used to account the plant uptake (ø) considering the boundary condition of r = 0 in the Barber (1987) nutrient uptake by plants model.

$$J_{\,l}^{\,i} = \, J_{NH4}{}_{l}^{i} \,, J_{No3}{}_{l}^{i} \,, J_{O2}{}_{l}^{i} \quad \text{(Eq. 6.54)}$$

$$J_{NH4}{}_{l}^{i} = \, Fm_{NH4}\, \frac{\theta_{usz}^i\, [NH_4]_l^i}{Km_{NH4} + \theta_{usz}^i\, [NH_4]_l^i} \quad \text{(Eq. 6.55)}$$

$$J_{NO3}{}_{l}^{i} = \, Fm_{NO3}\, \frac{\theta_{usz}^i\, [NO_3]_l^i}{Km_{NO3} + \, \theta_{usz}^i\, [NO_3]_l^i} \quad \text{(Eq. 6.56)}$$

$$J_{O2}{}_{l}^{i} = \, \lambda\big(\theta_{usz}^i [O_2]_{root}^i - \theta_{usz}^i [O_2]_l^i\big) \quad \text{(Eq. 6.57)}$$

$$J_j^i = J_{NH4}{}_j^i, J_{No3}{}_j^i, J_{O2}{}_j^i \quad \text{(Eq. 6.58)}$$

$$J_{NH4}{}_j^i = Fm_{NH4} \frac{\theta_{sz}^i [NH_4]_j^i}{Km_{NH4} + \theta_{sz}^i [NH_4]_j^i} \quad \text{(Eq. 6.59)}$$

$$J_{NO3}{}_j^i = Fm_{NO3} \frac{\theta_{sz}^i [NO_3]_j^i}{Km_{NO3} + \theta_{sz}^i [NO_3]_j^i} \quad \text{(Eq. 6.60)}$$

$$J_{O2}{}_j^i = \lambda\left(\theta_{sz}^i [O_2]_{root}^i - \theta_{sz}^i [O_2]_j^i\right) \quad \text{(Eq. 6.61)}$$

When solving the differential equation of the transport equation at the same time that using non-differential equations there are some problems of continuity that reflect in errors in the pollutant mass balance. Therefore, it is necessary to add a corrective step in the numerical solution. Therefore, the routine for calculation is presented here.

Ponding zone:

$$C_{pz}^{i+1} = C_{pz}^i\, h_{pz}^i\, A_b + \left[\frac{C_{in}^i Q_{in}^i - C_{pz}^{i-1}\left(Q_{pf}^i + Q_{over}^i + Q_{inf,p}^i\right) + R_{pz}^i}{h_{pz}^{i+1}\, A_b}\right] \Delta t \quad \text{(Eq. 6.62)}$$

USZ and SZ – method used: Adapted forward time central differences in space

$$L = h_{usz}^0 + h_{sz}^0 \quad \text{(Eq. 6.63)}$$

$$n = number\ of\ layers$$

$$\partial z = \frac{L}{n} \quad \text{(Eq. 6.64)}$$

$$m_{usz} = round\left(\frac{h_{usz}^0}{dz}, 1\right) \quad \text{(Eq. 6.65)}$$

$$m_{sz} = n - m_{usz} \quad \text{(Eq. 6.66)}$$

$$\alpha_l = \left(\frac{m_{usz} - 1 - l}{m_{usz} - 1}\right),\ \beta_l = \left(\frac{l}{m_{usz} - 1}\right) \quad \text{(Eq. 6.67)}$$

$$\alpha_j = \left(\frac{m_{sz} - 1 - j}{m_{sz} - 1}\right),\ \beta_j = \left(\frac{j}{m_{sz} - 1}\right) \quad \text{(Eq. 6.68)}$$

1st step – predictive:

If l == 0 (first layer):

$$\frac{\partial C_l^{i+1}}{\partial t} = \frac{1}{\theta_{usz}^{i+1}}\left[-\theta_{usz}^i\, k_{ads,usz}\, C_l^i + \rho\, k_{des,usz} C_{s_l}^i + \theta_{usz}^i \left(D\, \frac{C_{l+1}^i - 2\, C_l^i + C_{pz}^i}{\partial z^2} - \frac{V_l^i \left(C_l^i - C_{l-1}^i\right)}{\partial z}\right) + R_l^i + \emptyset\, J_l^i\right] \quad \text{(Eq. 6.69)}$$

Else if l == m - 1 (last layer):

$$\frac{\partial C_l^{i+1}}{\partial t} = \frac{1}{\theta_{usz}^{i+1}}\left[-\theta_{usz}^i\, k_{ads,usz}\, C_l^i + \rho\, k_{des,usz} C_{s_l}^i + \theta_{usz}^i \left(D\, \frac{C_l^i - 2\, C_l^i + C_{l-1}^i}{\partial z^2} - \frac{V_l^i \left(C_l^i - C_{l-1}^i\right)}{\partial z}\right) + R_l^i + \emptyset\, J_l^i\right] \quad \text{(Eq. 6.70)}$$

Else: (middle):

$$\frac{\partial C_l^{i+1}}{\partial t} = \frac{1}{\theta_{usz}^{i+1}}\left[-\theta_{usz}^i\, k_{ads,usz}\, C_l^i + \rho\, k_{des,usz} C_{s_l}^i + \theta_{usz}^i \left(D\, \frac{C_{l+1}^i - 2\, C_l^i + C_{l-1}^i}{\partial z^2} - \frac{V_l^i \left(C_l^i - C_{l-1}^i\right)}{\partial z}\right) + R_l^i + \emptyset\, J_l^i\right] \quad \text{(Eq. 6.71)}$$

$$C_{s_l}^{i+1} = C_{s_l}^i + \left(\frac{\theta_{usz}^i}{\rho} . k_{ads,usz} . C_l^i - k_{des,usz} C_{s_l}^i + R_{s_l}^i\right) \Delta t \quad \text{(Eq. 6.72)}$$

$$C_l^{i+1^*} = C_l^i + \frac{\partial C_l^{i+1}}{\partial t} \Delta t \quad \text{(Eq. 6.73)}$$

If j == 0 (first layer):

$$\frac{\partial C_j^{i+1}}{\partial t} = \frac{1}{\theta_{sz}^{i+1}}\left[-\theta_{sz}^i\, k_{ads,sz}\, C_j^i + \rho\, k_{des,sz} C_{s_j}^i + \theta_{sz}^i \left(D\, \frac{C_{j+1}^i - 2\, C_j^i + C_{m-1}^i}{\partial z^2} - \frac{V_j^i \left(C_j^i - C_{j-1}^i\right)}{\partial z}\right) + R_j^i + \emptyset\, J_j^i\right] \quad \text{(Eq. 6.74)}$$

Else if j == m - 1 (last layer):

$$\frac{\partial C_j^{i+1}}{\partial t} = \frac{1}{\theta_{sz}^{i+1}}\left[-\theta_{sz}^i\, k_{ads,sz}\, C_j^i + \rho\, k_{des,sz} C_{s_j}^i + \theta_{sz}^i \left(D\, \frac{C_j^i - 2\, C_j^i + C_{j-1}^i}{\partial z^2} - \frac{V_j^i \left(C_j^i - C_{j-1}^i\right)}{\partial z}\right) + R_j^i + \emptyset\, J_j^i\right] \quad \text{(Eq. 6.75)}$$

Else: (middle):

$$\frac{\partial C_j^{i+1}}{\partial t} = \frac{1}{\theta_{sz}^{i+1}}\left[-\theta_{sz}^{i}\, k_{ads,sz}\, C_j^{i} + \rho\, k_{des,sz} C_{s_j}^{i} + \theta_{sz}^{i}\left(D\, \frac{C_{j+1}^{i} - 2\, C_j^{i} + C_{j-1}^{i}}{\partial z^2} - \frac{V_j^{i}\left(C_j^{i} - C_{j-1}^{i}\right)}{\partial z}\right) + R_j^{i} + \emptyset\, J_j^{i}\right] \quad \text{(Eq. 6.76)}$$

$$C_{s_j}^{i+1} = C_{s_j}^{i} + \left(\frac{\theta_{sz}^{i}}{\rho}.k_{ads,sz}.C_j^{i} - k_{des,sz} C_{s_j}^{i} + R_{s_j}^{i}\right)\Delta t \quad \text{(Eq. 6.77)}$$

$$C_j^{i+1^*} = C_j^{i} + \frac{\partial C_j^{i+1}}{\partial t}\Delta t \quad \text{(Eq. 6.78)}$$

2nd step – corrective:

$$M_{stor}^{i+1^*} = \sum \overline{C_l^{i+1}}\, \frac{A_b\, h_{usz}^{i+1}}{m_{usz}}\, \theta_{usz}^{i+1} + \sum \overline{C_j^{i+1}}\, \frac{A_b\, h_{sz}^{i+1}}{m_{sz}}\, \theta_{sz}^{i+1} \quad \text{(Eq. 6.79)}$$

$$M_{stor,MB}^{i+1} = M_{stor,MB}^{i} + Q_{in}^{i+1} C_{in}^{i+1} - \left(h_{pz}^{i+1} A_b + Q_{over}^{i+1}\right) C_{pz}^{i+1} - \left(Q_{pipe}^{i+1} + Q_{inf,sz}^{i+1}\right)\overline{C_{m_{sz}-1}^{i+1}} - Q_{et}^{i+1}\overline{C_{l=0}^{i+1}} - M_s^{i+1} - M_R^{i+1} - M_J^{i+1} \quad \text{(Eq. 6.80)}$$

$$\Delta C_{usz}^{i+1} = \frac{M_{stor,MB}^{i+1} - \overline{M_{stor}^{i+1}}}{n\, \frac{A_b\, h_{usz}^{i+1}}{m_{usz}^{t}}\theta_{usz}^{i+1}} \quad \text{(Eq. 6.81)}$$

$$C_l^{i+1} = C_l^{i} + \frac{\partial C_l^{i+1}}{\partial t}\Delta t + \Delta C_{usz}^{i+1} \quad \text{(Eq. 6.82)}$$

$$\Delta C_{sz}^{i+1} = \frac{\overline{M_{stor}^{i+1}} - M_{stor,MB}^{i+1}}{(n-1)\frac{A_b h_{sz}^{i+1}}{m_{sz}^{t}}\theta_{sz}^{i+1}} \quad \text{(Eq. 6.83)}$$

$$C_j^{i+1} = C_j^{i} + \frac{\partial C_j^{i+1}}{\partial t}\Delta t + \Delta C_{sz}^{i+1} \quad \text{(Eq. 6.84)}$$

6.2.2 Development of automatic calibrator

For the water flow module, an automatic calibrator for the model was developed using genetic algorithms (from the Distributed Evolutionary Algorithms in Python library - DEAP 1.3.1) using the maximization of the average NSE to outflow through the underdrain and the height of the water level in the ponding zone as the objective function. The parameters calibrated were Kc, Ks, S_h, S_w, S_{fc}, S_s, Kf, Cd.

As for the water quality module, it was also developed an automatic calibrator with DEAP, but according to the type of data monitored, the objective functions can vary. When the data have concentrations over time, being able to build a pollutograph, the calibration is made by maximization of NSE to outflow pollutograph. When the data only have final EMC concentrations, the calibration is made by minimization of standard errors for outflow EMC. The parameters calibrated for this module were k_{ads}, k_{des}, k_{nit}, k_{denit} and D.

6.2.3 Bioretention systems monitored in Brazil and in Australia

Data from two bioretention systems were used to evaluation of the model, representing two different climatic conditions, one located in Sao Carlos, Brazil, and the second one located in Melbourne, Australia. The system located in Brazil is a bioretention box with the configuration presented in Chapter 5. 26 events were monitored from February 2019 to February 2020. The system located in Australia are several bioretention columns with the configuration presented in Figure 6.2. 12 events were monitored from November 2010 to May 2012.

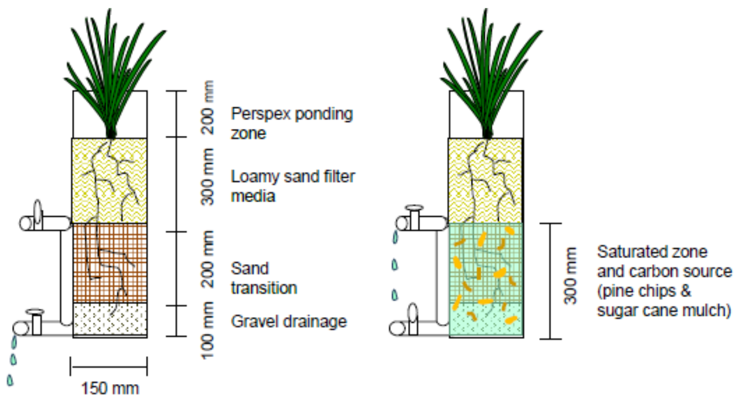

Figure 6.2 - Bioretention columns scheme in Melbourne, Australia (Payne, 2013)

6.2.4 Sensitivity analysis

A preliminary sensitivity analysis was performed in the nutrient quality module in order to evaluate the main process affecting the final outflow concentration. This sensitivity analysis has the purpose of identify the main process that require further attention in refining the model prediction. Therefore, it was performed a sensitivity analysis one-at-time (OAT), varying the same parameters used in calibration for the nutrient quality module in ranges of +100% to -100%, with increases of 15%. As evaluation functions, it was used the NSE and the difference in the outflow mass observed and predicted.

6.3 Results and discussion

The code developed in Python for the model and the automatic calibrators is freely available for use in research on the GitHub platform at the following link: https://github.com/mabatalini/Process-based-Nitrogen-Model.git.

In this section, it is presented the preliminary results obtained for the model developed. First, the results for the water flow module are presented, followed by the results for the nitrogen quality module.

The water flow module obtained directly from Shen et al. (2018) allowed different configurations of a bioretention device: bioretention area and soil mix area with same or different sizes and lined or unlined. In this new version it was added a new possibility of configuration, with or without saturated zone (varying the height of the underdrain height). In addition, the code routine developed previously by Shen et al. (2018) was giving water balance errors of around 2% in the final variables. Therefore, it was made a correction in the saturation estimation in the code, which reduced the error to less than 0.0001%.

After the adaptations were made, this module was evaluated from controlled monitored events in laboratory, in the bioretention box, located in Sao Carlos, Brazil. Adopting literature values for the constants, the model returned a good prediction of the behavior of the bioretention (good representations of when the flow peaks occur and the flow durations), however, the final water balance and the intensity of the flows were not well predicted. Therefore, it was performed a calibration of the model.

Six events monitored during the year 2019 in a bioretention box in laboratory scale were used for the calibration process (three used for calibration and three for validation). The final NSE values obtained for calibration was 0.74 and for validation it was 0.62, demonstrating a good fit to the model (Moriasi, 2007). In Figure 6.3 it is possible to see the hydrographs predicted by the model and monitored for two controlled events. The events showed in Figure 6.3 have a saturated zone included, in order to evaluate the denitrification process and influence of saturated zone in nitrogen treatment.

After the water flow module was adapted for the new conditions, the errors in the water balance were eliminated, and the calibration returned in a good fit to the monitored events, the nitrogen quality module was evaluated.

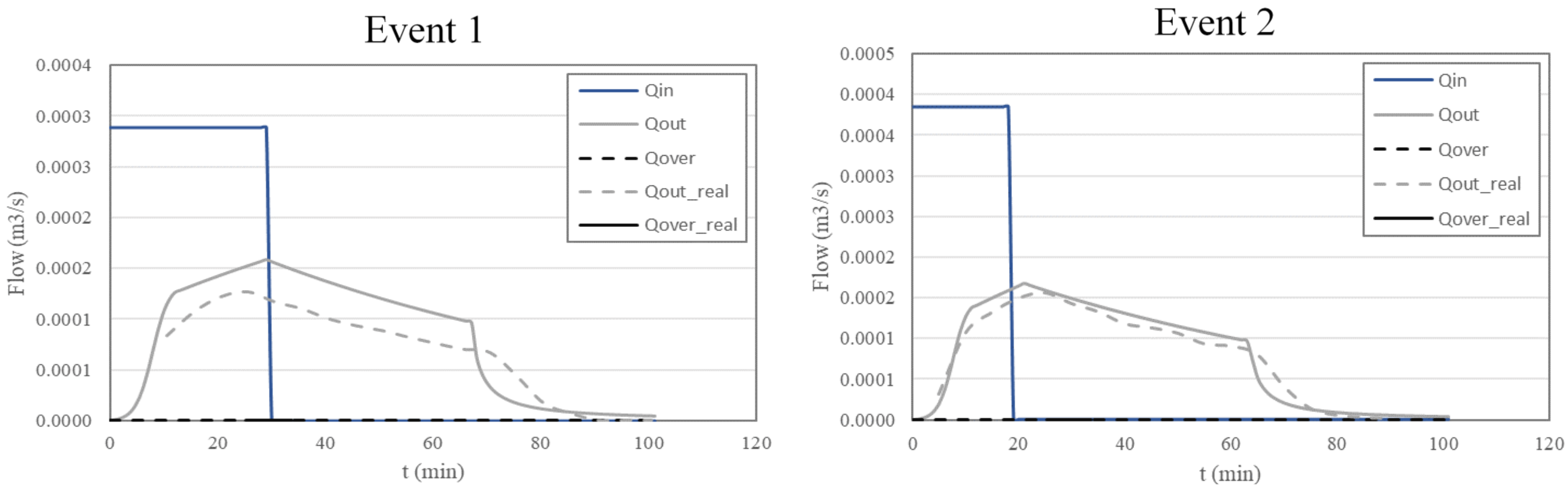

Figure 6.3 - Modeled and observed hydrographs for two controlled events in a bioretention with saturated zone

The first try of implementation of the nitrogen quality module was made by using only the predictive step in the numerical methods to solve the differential transport equation in the bioretention soil mix. In this first try, the pollutant mass balance was assessed for the ammonia and nitrate fractions and errors in order of 1 to 10% were obtained. The error was proportional to the intensity of the inflow rates (higher inflows was generating higher errors). Therefore, it was assumed that the errors were related to the advection component of the transport equation and with the outflow generated in the model. As a solution, a corrective step in the numerical solution was proposed. With this, the errors in the mass balance were reduced to less than 0.01%.

After solving the problem of the pollutant mass balance, the pollutographs obtained by the model were compared with the values monitored in the bioretention box for Event 1 and Event 2. Figure 6.4 shows the results for ammonia and nitrate for both events, after a first calibration the nitrogen quality module. It is possible to see from Figure 6.4 that the model was able to predict the general behavior of the nitrogen fractions (the occurrence of the peaks and the duration), however, it does not present a good fit, specially for nitrate. The values of nitrate along time monitored in the bioretention for both events are much higher than the values predicted by the model. Additionally, with the parameters values obtained from calibration, it was not even possible to obtain a nitrate exportation in the bioretention (which was observed in almost all the events monitored in the bioretention box).

An additional test was made by increasing the k_{nit} value until all the ammonia was converted to nitrate and considering that no denitrification was occurring ($k_{denit} = 0$) (Figure 6.5). In this case it is possible to observe an export of nitrate by the bioretention, however, the load values predicted by the model are still very distant from the ones monitored. Some hypotheses must be further evaluated to give a better fit of the model: (1) The initial concentration of nitrogen fractions in the soil were considered zero, different initial concentrations must be evaluated; (2) The nitrification and denitrification process are not considering initially the influence of oxygen and dissolved carbon concentrations, which must be incorporated.

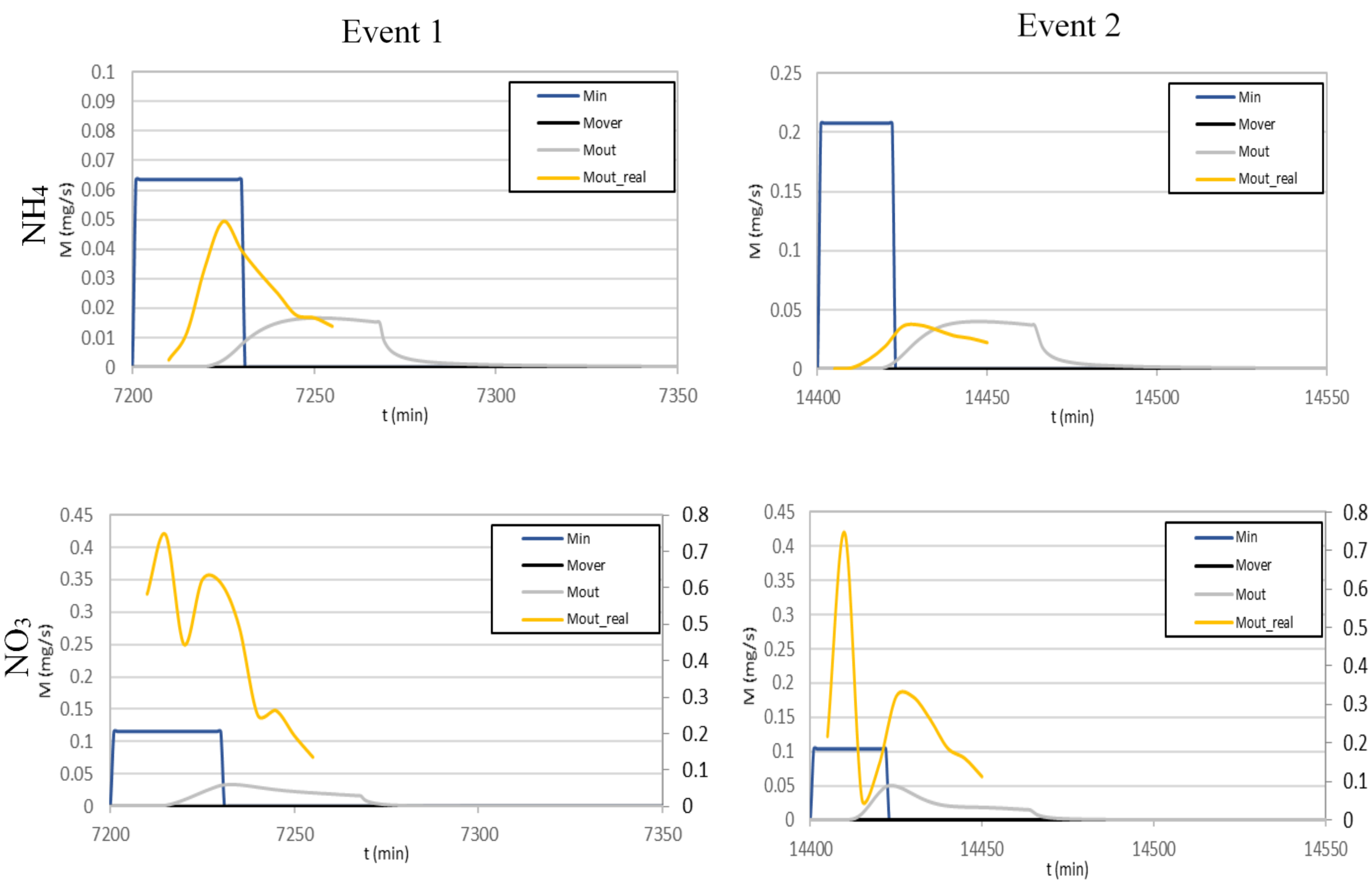

Figure 6.4 - Modeled and observed pollutographs for two controlled events in a bioretention with saturated zone. The observed mass values for nitrate are presented in a secondary scale, due to the differences of mass magnitude.

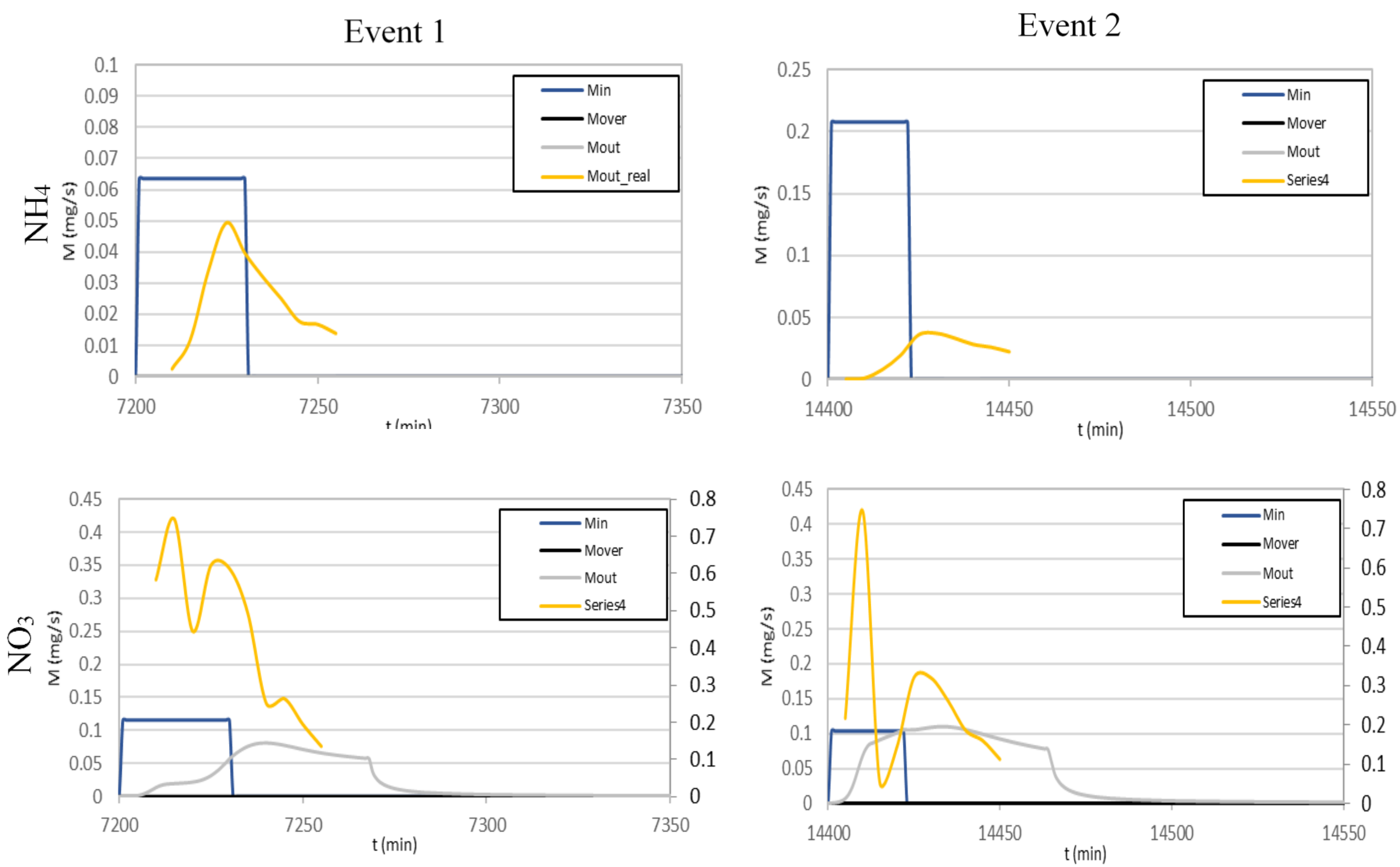

Figure 6.5 - Modeled and observed pollutographs for two controlled events in a bioretention with saturated zone, increasing knit and considering kdenit = 0. The observed mass values for nitrate are presented in a secondary scale, due to the differences of mass magnitude.

To complement the analysis and identify the process that have more potential to generate differences in the results of the model, a preliminary sensitivity analysis was performed. The results can be observed in Figure 6.6, for NSE. The parameters influencing the most the evaluate function (NSE and simple difference in final concentrations) were the k_{nit} (representing the nitrification process) and k_{denit} (representing the denitrification process). The variation on the other parameters resulted in differences of less then 0.05 in the NSE, and therefore were not considered as sensitive.

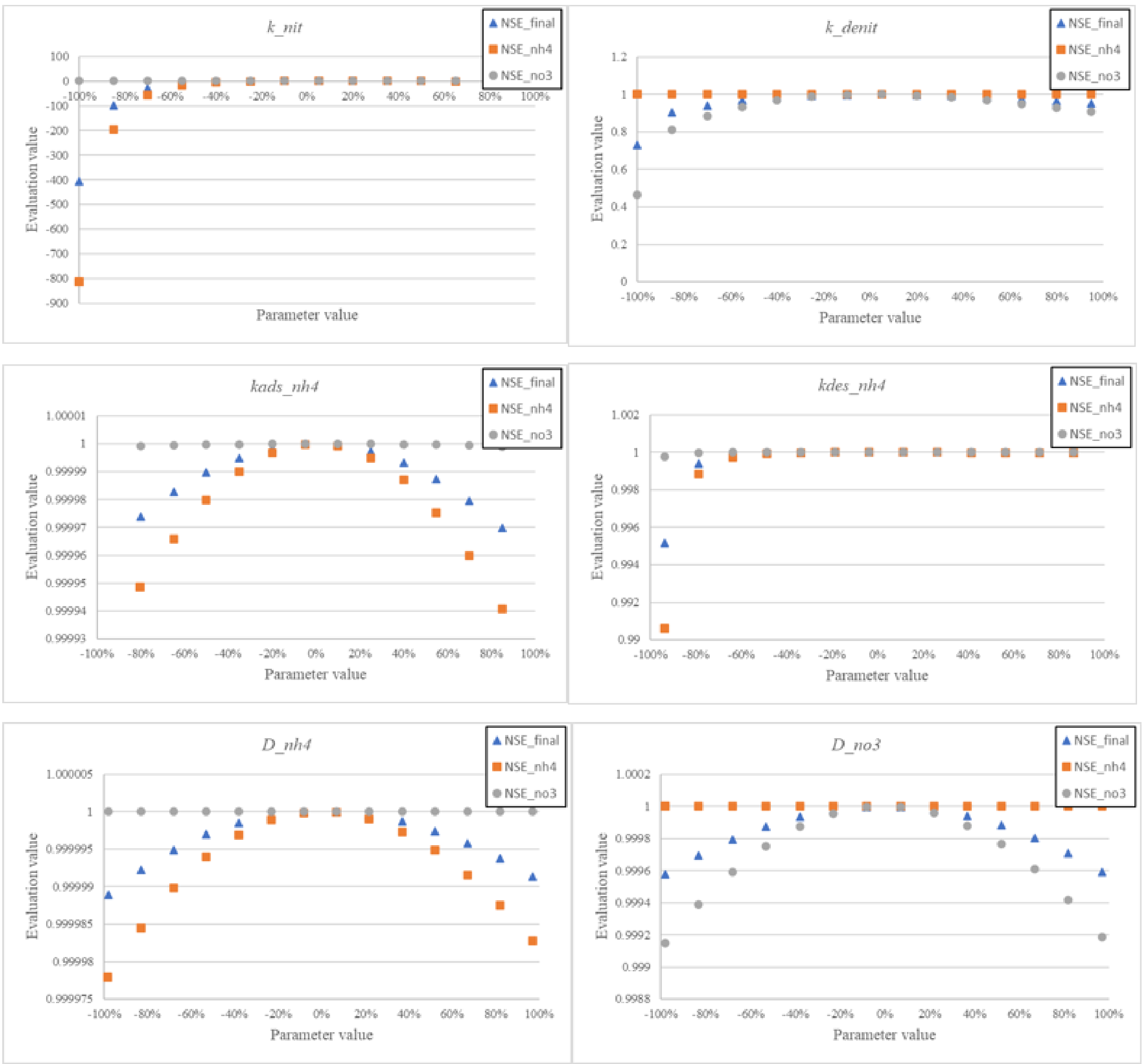

Figure 6.6 - Sensitivity analysis results for NSE evaluation function

For the k_{nit}, it is possible to see that lower values have the most influence on the results of the model, generating differences in NSE of up to -850 (high influence). When comparing with the second most sensitive parameter, k_{denit}, lower values also have more influence in the results of the model, however the NSE reach values up to 0.4, which is a great difference with the values for k_{nit}, showing the importance of the nitrification process.

The next steps of the research is to: (1) Test the model for different initial concentrations in the soil and; (2) To test the model for the bioretention columns in Australia climate.

6.5 Conclusion

The developed model is composed of two main modules, the water flow module and the water quality module. The water flow module presented a good fit with the results obtained in experimental monitoring, for the bioretention device evaluated in Brazil (NSE values of 0.74 to calibration and 0.62 to validation, three events each). The bioretention evaluated in Brazil was used for the initial tests due to the availability of flow data over time, making it possible to compare the actual hydrographs and polygraphs with those obtained by the model, and not just final data of accumulated volumes and event mean concentration. So far, the model has been tested for 20-day intervals of continuous monitored data, with low processing time (less than one minute). Processing time is important when aiming at a later implementation for systems with real-time control. The water balance of the model was also verified, with residual errors smaller than 0.0001% of the total volume of entry into the system.

As for the quality module, the results are promising so far, returning expected behaviors for the nitrogen fractions and errors less than 0.01% in the mass balance, after incorporating a corrective step in the model. However, a good fit has not yet been obtained with the experimental results collected for the monitored bioretention device in Brazil. The change in initial parameters, such as initial concentrations of nitrogen in the soil, and additional treatment reactions should be tested. To assist in refining the modeling of physical processes, an initial sensitivity analysis one-at-time was carried out, which showed a greater influence of nitrification processes on final concentrations, followed by the denitrification process.

The next steps in the research consist of evaluating the results based on the data monitored in Australia's bioretention columns (once the monitored data contains the results of nitrogen fractions in each of the treatment pathways, helping to identify which processes need to be improved by the model) and testing different initial conditions. After finalizing the model, a global sensitivity analysis will be performed, identifying the key-factors for the nitrogen elimination process and the parameters that cause greater uncertainty in the model.

References

Barber, S. A. (1984). *Soil nutrient bioavailability: a mechanistic approach*. John Wiley & Sons.

Carter, J., Cavan, G., Connelly, A., Guy, S., Handley, J., & Kazmierczak, A. (2015). Climate change and the city: building capacity for urban adaptation. *Progress in Planning*, 95, 1-66.

Chen, X., Peltier, E., Sturm, B. S., & Young, C. B. (2013). Nitrogen removal and nitrifying and denitrifying bacteria quantification in a stormwater bioretention system. *Water Research*, 47(4), 1691-1700.

Christianson, L., Christianson, R., Helmers, M., Pederson, C., & Bhandari, A. (2013). Modeling and calibration of drainage denitrification bioreactor design criteria. *Journal of irrigation and drainage engineering*, 139(9), 699-709.

Christianson, R. D., Barfield, B. J., Hayes, J. C., Gasem, K., & Brown, G. O. (2004). Modeling effectiveness of bioretention cells for control of stormwater quantity and quality. In *Critical Transitions in Water and Environmental Resources Management* (pp. 1-7).

Fletcher, T., Shuster, W., Hunt, W., Ashley, R., Butler, D., Arthur, S., . . . Mikkelsen, P. (2015). SUDS, LID, BMPs, WSUD and more – The evolution and application of terminology surrounding urban drainage. *Urban Water Journal*, 12(7), 525-542.

Hipsey, M. R., Bruce, L. C., & Hamilton, D. P. (2013). *Aquatic Ecodynamics (AED) Model Library Science Manual*. The University of Western Australia Technical Manual, Perth, Australia, 34.

Hoff, H. (2011). Understanding the nexus: background paper for the Bonn2011. *Nexus Conference*

Kirk, G. J. D., & Kronzucker, H. J. (2005). The potential for nitrification and nitrate uptake in the rhizosphere of wetland plants: a modelling study. *Annals of botany*, 96(4), 639.

Laurenson, G.; Laurenson, S.; Bolan, N.; Beechan, S.; Clark, I. (2013). *Advances in Agrononmy: Chapter Four – The Role of Bioretention Systems in the Treatment of Stormwater*. Elsevier.

Li, J., Zhao, R., Li, Y., & Chen, L. (2018). Modeling the effects of parameter optimization on three bioretention tanks using the HYDRUS-1D model. *Journal of environmental management*, 217, 38-46.

Lynn, T. J., Nachabe, M. H., & Ergas, S. J. (2017). Modeling denitrifying stormwater biofilters using SWMM5. *Journal of Environmental Engineering*, 143(7), 04017017.

Macedo, M., Lago, C., Rosa, A., & Mendiondo, E. (2017). Técnicas compensatórias de bioretenção para cidades resilientes: integração com nexus–água, energia, alimento. [Low Impact Development practices of bioretention to resilient cities: integration with water-energy-food nexus]. *XXII Simposio Brasileiro de Recursos Hidricos*. Florianopolis, Brazil. (In Portuguese)

Moriasi, D. N., Arnold, J. G., Van Liew, M. W., Bingner, R. L., Harmel, R. D., & Veith, T. L. (2007). Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. *Transactions of the ASABE*, 50(3), 885-900.

Pálfy, T. G., Gourdon, R., Meyer, D., Troesch, S., Olivier, L., & Molle, P. (2017). Filling hydraulics and nitrogen dynamics in constructed wetlands treating combined sewer overflows. *Ecological Engineering*, 101, 137-144.

Payne, E. G. (2013). *The influence of plant species and water dynamics on nitrogen removal within stormwater biofilters*. Monash University. Thesis.

Payne, E. G., Fletcher, T. D., Russell, D. G., Grace, M. R., Cavagnaro, T. R., Evrard, V., ... & Cook, P. L. (2014). Temporary storage or permanent removal? The division of nitrogen between biotic assimilation and denitrification in stormwater biofiltration systems. *PloS one*, 9(3), e90890.

Randelovic, A., Zhang, K., Jacimovic, N., McCarthy, D., & Deletic, A. (2016). Stormwater biofilter treatment model (MPiRe) for selected micro-pollutants. *Water research*, 89, 180-191.

Shen, P., Deletic, A., Urich, C., Chandrasena, G. I., & McCarthy, D. T. (2018). Stormwater biofilter treatment model for faecal microorganisms. *Science of the total environment*, 630, 992-1002.

7 GENERAL CONCLUSION

The conclusion will be presented in the form of answers to the general and specific purposes, and a section of recommendations for future works.

6.1 Conclusions

This research had as general purpose “To improve the scientific framework on a new generation of LID practices, more specific to bioretention, called 3rd generation (LID-3G), based on its conceptual and experimental development. Based on the experimental monitoring in field and laboratory scales, to evaluate the bioretention efficiency for runoff retention, pollutant removal and water-energy-food security of future resilient cities under subtropical climate.” Therefore, in the initial phase of this doctoral research, a bibliographic review was performed, allowing to identify the studies that addressed the purposes of recycling and co-management of resources, and consequent contribution to the SDGs. Based on the bibliographic review made, it was possible to: (1) identify the main challenges in the design of bioretention practices considering future change scenarios, (2) raise the potentials of 3rd generation in the bioretention practices already in operation by the WADI lab and (3) test different bioretention configurations through laboratory experiments aiming at better mitigation and resource recycling. The conclusions on each of these activities are presented below, responding to each of the specific purposes.

The first specific purpose of this research was “To incorporate scientific and technological elements for a new classification of LID practices, aiming at actions to adapt to the risks of urban drainage extremes, based on the current state of the art and conceptions of LID practices in operation in Brazil and abroad ”. The bibliographic review presented in chapter 2 aimed to meet this purpose. In the initial proposition of this doctoral research, we aimed to integrate aspects related to the *water-energy-food nexus* in the operation of bioretention systems, looking at the excesses of urban drainage not only as problems to be controlled but as resources to be reincorporated in the watershed for the increased water-energy-food security. This idea came due to the growing number of studies with stormwater harvesting, but also with some new attempts to produce food on the surface of vegetated LID practices. However, during the bibliographic review, it was possible to observe new studies incorporating different aspects of sustainability, beyond just the *water-energy-food nexus*. New studies have evaluated the potential for integration with ecosystem ecology, hybrid water supply systems, life cycle analysis of the practices, carbon cycle (both for sequestration and for the production of GHGs), energy production by rotating turbines with the movement of water. Therefore, we expanded our initial assessment only to the *water-energy-food nexus* for the UN SDGs, identifying possible ways of contributing the SDGs 2 - zero hunger, 6 - clean water and sanitation, 7 - affordable and clean energy, 11 - resilient cities and communities and 13 - climate action.

From the bibliographic review, it was also possible to notice the importance of adopting time scales in the design and evaluation of bioretention systems and LID practices in general, mainly in places with changes in urbanization. Therefore, the second specific purpose assessed was “To evaluate the current design methods and propose a modular-adapted implementation of bioretention that incorporate future scenarios with drivers of change”. For this specific purpose, in the second chapter we evaluated three methods of pre-design that are commonly used in Brazil and the method proposed by Rosa (2016) in his initial work with bioretention in the city of Sao Carlos, considering changes in the input variables and the parameters over time due to drivers of urbanization and climate change, and due to infrastructure aging (causing loss of hydraulic conductivity of the filtering media). The sensitivity analysis showed that the performance of bioretention is more sensitive to parameters related to land use, such as coefficients of the infiltration methods for the watershed, and to the total precipitated volumes (represented by P90) that will change according to future climate scenarios. The adoption of modular design over time was able to avoid decreases in performance for future scenarios of greater rainfall volumes and less infiltration.

In addition, two experimental bioretention systems designed for different scales and mitigation purposes were evaluated, according to the specific purpose “From experimental bioretention in real and laboratory scale, to evaluate the operation, maintenance, monitoring and resources recycling for water-energy-food security, under subtropical climate conditions”. In chapter 4 and 5 the experimental results for street scale and property scale bioretention systems were presented evaluating their potential for water reuse. For both systems, the water volume recovered by overflow and outflow is able to supply less restrictive non-potable demands for wet periods and have an excess of water returning to the watershed. However, both systems and their different configurations present water quality as a limitation for its reuse, due to the non-compliance with the CONAMA 357/420 standard for uses with direct human contact, and NBR 15.527/2019: stormwater – coverings utilization in urban areas for non-potable purposes – requirements, mainly in terms of color, metals and TC. Chapter 5 evaluated the use of a bioretention with a saturated zone to improve water quality. Although it was possible to notice a higher pollutant removal efficiency for this configuration, it was not sufficient to meet CONAMA 357/420 class 2 standards.

In addition to recycling resources through the water reuse, LID practices can also contribute to other SDGs and to co-manage resources, as seen in chapter 2. Therefore, to incorporate this assessment into experimental work, we proposed in chapter 5 individual and local indicators related to the global and national SDGs indicators, according to the specific purpose “To evaluate the efficiency of bioretention-3G based on new proposed coefficients, to re-establish the water balance of pre-urbanization and resource recycling, incorporating drivers of change, with urbanization growth, climatic variability and consumption habits”. A first assessment of these indicators for the co-management of the water-energy-greenhouse gas nexus was presented in chapter 5, quantifying the reductions in energy demands and carbon emissions by area of bioretention and by watershed area.

To complement the experimental work, a process-based model for nitrogen fractions was proposed, allowing to evaluate more quickly and from the physical-chemical-biological processes the efficiency of different bioretention configurations to remove nitrogen. By the modelling it is possible to identify the key factors in the nitrogen treatment (from the sensitivity analysis), and to evaluate different design and management scenarios. Until now, the efficiency of the system has been found to be more sensitive to the nitrification constant, followed by the denitrification constant, which are related to the choice of the filtering media and the establishment of biofilm.

These results are important because they extend the practical knowledge in real scale for the use of LID practices, with a focus on Brazil. Only with the practical and detailed knowledge, it is possible to increase the application of these systems as public policies actions by the managers.

6.1 Recommendations for future studies

From the identification of gaps remaining, this doctoral research suggests the following points as a recommendation for future work:

- Dynamic assessment of sustainability and resilience indicators through continuous simulation. The design of LID practices can also be verified and improved through the dynamic assessment of these indicators.

- Investigation of vegetated LID practices for direct food production. We propose the combined use of edible plant species and plants with capacity of bioaccumule metals, to assess whether with the presence of bioaccumulator plants there would be less concentration of metals in edible plants tissues.

- Quantification of plant biomass production and assessment of metals accumulation in different parts of plants and in the soil.

- Improve the bioretention filtering media or a post-treatment systems to improve the quality of water stored to future reuse. The post-treatment systems must focus on removing color, microorganisms and metals, since these parameters were identified as the most critical in relation to CONAMA 357/420 and NBR 15.527/2019: stormwater – coverings utilization in urban areas for non-potable purposes - requirements. We suggest as possible treatments: Disinfection processes by applying chlorine in a reuse reservoir; Use of a filter membrane in the underdrain outlet (however, it is necessary to assess how much this would affect the hydraulic efficiency of the system). New studies also incorporate the use of activated carbon in the filtering media, which can be an alternative to improve metals removal.

- Expand the development of process-based models for other pollutants then nitrogen, using the conceptual basis of the solute transport equation in the soil and the Python code already developed, calibrating the sorption and diffusion parameters, and adapting the non-conservIative equations for each pollutant.

- Real-time monitoring and control systems to optimize decentralized urban drainage systems to flood control and hybrid water supply systems to non potable demands reduction.

In addition to the recommendations made for new studies, in improving the state of the art in alternative urban drainage techniques and technologies that contribute to the different SDGs and increasing the resilience of communities, we would also like to highlight contributions to new water management policies rainwater, in view of the current discussion involved with the approval of the Law 14026/2020 that updates the guidelines of the Sanitation Legal Framework in Brazil. Sanitation incorporates the areas of distribution and treatment of drinking water, sewage collection and effluent treatment, solid waste management and urban drainage. However, even with the improvement of the Legal Framework for sanitation, there is still little legal and public policy approach to the treatment of diffuse pollution present in runoff, flood control through sustainable and decentralized urban drainage and the incorporation of alternative sources of water (like rainwater and stormwater). The Law 14026/2020 establishes goals for universal sanitation (99% of the population with access to drinking water and 90% with access to sewage collection and treatment), which with the climate change scenarios, the simple coverage of services does not guarantee the supply, since the extremes worsen and the scarcity of resources becomes an increasingly frequent problem. We recommend that future studies on the implementation, updating and feasibility of sanitation infrastructure, which should be expanded after the approval of the Law 14026/2020, incorporate alternative techniques of urban drainage and hybrid systems of water supply and rainwater management as sustainable alternatives that contribute to the resilience of communities.