Pluvial Flood Risk Modeling, Assessment, and Management under Evolving Urban Climates and Land Cover

by

Jason Sauer

A Dissertation Presented in Partial Fulfillment
of the Requirements for the Degree
Doctor of Philosophy

Approved April 2022 by the
Graduate Supervisory Committee:

Nancy B. Grimm, Chair
Mikhail V. Chester
Daniel L. Childers
Elizabeth M. Cook
Hallie Eakin

ARIZONA STATE UNIVERSITY

May 2022

ABSTRACT

Pluvial flooding is a costly, injurious, and even deadly phenomenon with which cities will always contend. However, cities may reduce their risk of flood exposure by changing historically dominant patterns of development that have removed natural landscape features and reduce the damages that flooding causes by identifying and supporting vulnerable populations. Accomplishing either goal requires the development and application of appropriate frameworks for modeling or recording flood exposure. In this dissertation, I used modeling and surveying methods for assessing pluvial flood exposure in two cities, first in Valdivia, Chile, and then in Hermosillo, México. I open with a summary on pluvial flood risk in the present day and the threat it may pose under changing climates. In the second chapter, I explored how a form of urban ecological infrastructure (UEI), the wetland, is being wielded in Valdivia toward pluvial flood mitigation, and found that wetland daily, seasonal, and interannual changes in wetland surface and soil water storage alter pluvial flood risk in the city. In the third chapter, I used a mixed methodology, including projections of future land cover generated by cellular automata models with inputs from visioning workshops conducted by the Urban Resilience to Extremes Sustainability Research Network (UREx SRN), and found that wetland loss in future land configurations may lead to increased pluvial flood risk. In the fourth chapter, I combined these land cover models from the third chapter with downscaled climate data on precipitation, also generated by the UREx SRN, and found that wetland conservation can help to mitigate the pluvial flood risk posed by changing patterns of rainfall. In the fifth chapter, I applied the Arc-Malstrøm method for pluvial flood assessment in Hermosillo, México, and compared it with the more traditional rational method for flood assessment, and through accompanying surveys found that perception of flood risk is significantly affected by flood dimensions and impacts. This dissertation concludes with a synthesis of pluvial flood risk assessment, suggestions for improvements to modeling, as well as suggestions for future research on pluvial flood risk assessment in cities. This dissertation advances the understanding of the utility of inland wetland UEI in cities under present and future land cover and climate conditions. It also qualifies the utility of common and new pluvial flood risk assessments and offers research directions for future pluvial flood assessments.

DEDICATION

To my family and friends, who have seen me through my lowest points in this program and celebrated with me at my highest. To my friends and colleagues in Chile, who supported me through this work and who have also taught me much about what is good in life. To my lab mates and my UREx SRN cohort, who have given me the support and opportunities necessary to accomplish the work contained herein and the work I will produce hereafter.

ACKNOWLEDGMENTS

Thank you endlessly to my advisory committee: Dr. Nancy B. Grimm, Dr. Mikhail V. Chester, Dr. Daniel L. Childers, and Dr. Hallie Eakin. An equal amount of thanks is due to Dr. Olga Barbosa and the members of her laboratory, who oriented me to the city of Valdivia, Chile, and who made me feel like a part of the lab and at home.

Thank you as well to my cohort and academic colleagues within the Urban Resilience to Extremes Sustainability Research Network (UREx SRN). Working on urban resilience to climate change was something I thought I had made up back in 2010, when I set out on the long, long journey to where I am now. Thank you for believing that I could contribute meaningfully to this field, and for providing me the financial, academic, and emotional support to do this work.

Thank you to all members of the Grimm Lab past and present, who have been something of a second family to me during this time. I hope my support for you all has been as meaningful as your support has been for me.

Thank you as well to Dr. Agustin Robles-Morua and his Laboratorio de Modelación Ambiental y Sustentabilidad the Instituto Tecnología de Sonora, as well as Dr. Javier Navarro from ITSON, for their hard work and welcome during my research in Hermosillo, México. Thank you as well to the employees of the IMPLAN, particularly Eduardo “Lalo” Hinojosa Robles and Lupita Peñúñuri, who provided much of the logistical support for my work there.

Thank you as well to the team from the Future Cities Podcast, who have helped me reconfigure my perceptions on science communication to various audiences.

This dissertation is based, in part, upon work that was supported by the National Science Foundation through the UREx SRN, the Resilient Latin America International Research Experience for Students project, and the Central Arizona-Phoenix Long Term Ecological Research Program. It is also based, in part, upon work that was supported by Global Development Research Program at the United States Agency for International Development.

TABLE OF CONTENTS

CHAPTER Page

CHAPTER PAGE

CHAPTER Page

CHAPTER Page

LIST OF TABLES

Table Page

LIST OF FIGURES

Figure Page

CHAPTER 1

INTRODUCTION

Discussions over the contours and content of urban resilience are robust and lively (Alberti et al., 2003; Ahern, 2011; Meerow, Newell, & Stults, 2016), but under the course of this dissertation, I have noted a sort of tension and growing professional disconnection in myself between researcher and subject. The widely employed definition provided by Meerow, Newell, & Stults (2016) defines resilience as:

> "*...the ability of an urban system—and all its constituent socio-ecological and socio-technical networks across temporal and spatial scales—to maintain or rapidly return to desired functions in the face of a disturbance, to adapt to change, and to quickly transform systems that limit current or future adaptive capacity.*"

Working on resilience to extreme events has at many times for me during this dissertation become an anodyne matter of examining figures on structural damages, displacement, morbidities and deaths from heat waves, droughts, and floods, and applying resilience practices and methods to figure out a way to get the figures to go down. Much like the Meerow, Newell, & Stults (2016) definition of resilience, the work in my dissertation research has at times felt more like a technical engagement with the "what" and "how" of resilience at the expense of the other more emotional elements, specifically the "where," "for whom," and "why" of the work. To their great credit, Meerow, Newell, & Stults (2016) argue for the importance of considering these latter, more emotional elements in resilience work. In the course of my dissertation and in working on manuscripts for publication, I have identified a pressure to put these concerns at a distance and to find justifications that are a matter of academic intrigue rather than a personal one. In this dissertation and elsewhere, I have found ways to justify my study sites that elide the truth: that these systems are interesting primarily because I lived in them, among hosts and friends and neighbors, and I care about them. The function of this work is to learn lessons that can be useful to them and, after vetting among the scientific community, to share lessons that may be useful to the communities of others.

In this opportunity to reflect on and explain the origins of this work, I begin with a re-engagement with the "why" of everything accomplished here. My personal motivation has been a deep-seated concern about the coerced or forced displacement of people. My focus is not the result of an abundance of research concerned with this outcome, and in fact in my corner of urban resilience I almost never come across it. Rather, I dwell on such consequences because of personal experiences working with and living among migrant and refugee communities. Life as a forced migrant or refugee struck me as an incredibly difficult one and I have been deeply affected by the plights of migrants and refugees close to me. The evidence in the academic literature confirms this. Migration journeys are dangerous and asylum laws, when they exist, may lock in or worsen the precarious condition of displaced peoples once they have arrived in their host geographies (Ertorer, 2021; Oner, Durmaz-Drinkwater, & Grant, 2021; Papatzani, Psallidaki, Kandylis, & Micha, 2021). Even when migrants are able and willing to adapt to the new work and life conditions of their host geographies, they must contend with economic insecurities, bigotries, and laws that ensure their lack of political representation indefinitely (Bentz, 2012; Logie, Daniel, Ahmed, & Lash, 2017).

My own work in resilience is the result of working with refugees in India, which evolved into a more specific concern for people that were referred to as "climate refugees." Climate refugees are a special category of migrants and refugees, whose continued living on their land has been made impossible or highly unfavorable due to the impacts of climate change on their immediate environments (Stern, 2008). Bierman and Boas (2010) define climate refugees "people who have to leave their habitats, immediately or in the near future, because of sudden or gradual alterations in their natural environment related to at least one of three impacts of climate change: sea-level rise, extreme weather events, or drought and water scarcity." While climate refugees are afforded a special distinction from traditional forms of migrants and refugees (Levy and Patz, 2015), I find compelling argumentation that the forces behind displacement are shared with more traditional forms of refugees, which are socioeconomic and political at their core, and often overlap with the causes of mobility among people who would not be considered climate refugees (Munoz, 2021). These common forces behind the creation of all forms of refugees and migrants imply the possibility of interventions that mitigate or erase the stresses of climate drivers without having to contend directly with climate change itself. The examination and manipulation of these common forces are the essential forms of work for resilience scholars, and thus I have maneuvered over time to work in this field.

Of course, the creation of refugees and migrants is only one concerning outcome of a lack of resilience to climate change in cities: there is a range of concerning impacts on communities from climate change-inflicted disturbances, from occasional nuisance to death. The degree and frequency of temporary disruptions to urban life, such as those to transportation networks due to pluvial flooding (Chang et al., 2010), can be affected by the work of resilience scholars. At the more extreme end of possible outcomes, resilience research may aid in the reduction of death tolls of extreme events like heat waves (Hondula, Balling, Vanos, & Georgescu, 2015) and flooding (Falconer et al., 2009), which, depending on our response, may worsen under a changing climate. These possibilities, too, motivate my work in urban resilience.

Our collective and regional responses to climate change may in turn provoke their own problems, and thus the good urban resilience scholar must be cognizant of the literature detailing the impacts of our responses. For example, managed retreat (Siders, Hino, & Mach, 2019), or the displacement of populations resulting from a lack of policies curtailing unmanaged retreat, from areas at elevated risk from the effects of climate change may coerce the mobilization of populations within cities. Additionally, market forces may coerce economically marginalized groups out of areas in cities that are safe from extreme events and climate change, in a process broadly known as “climate gentrification” (Keenan, Hill, & Gumber, 2018). Thus, in pursuing solutions to stop forced or coerced mobility on certain populations, we may burden other populations with the same type of problems we seek to address. Implementation of green infrastructure in cities to combat climate change may increase demands on scarce resources such as water (Pincetl, Bunje, & Holmes, 2012), may create maintenance burdens on populations that already feel stretched thin (Pincetl, 2010), or may be undesirable for more nuanced reasons of landscape preference (Teixiera et al., 2019). The literature of these sort of knock-on effects of climate change and the responses of cities is growing and will certainly become a greater part of resilience research as time goes on.

Thus, my work is motivated by the desire to mitigate the effects of climate change, and the effects of our responses to climate change, on the communities of which I am a part. In the course of completing this dissertation, I have also become focused on the ways in which urban resilience work presents opportunities for reversing many of the deleterious effects than urbanization has had on the environment and the communities that occupy it. Of particular interest to me is combatting urbanization's historical competition with other forms of land cover. Wetlands are one of the more notable victims of such competition, with 87% of global wetland cover being lost since the 1700s, and half of that loss occurring during urbanization and agricultural expansion in the 20th century (Davidson, 2014). Rates of wetland loss differ by region, and so do their causes. Groffman et al. (2003) estimated that the majority of the pre-settlement ecosystems of urban floodplains have been converted to industrialized or arable land, as well as to impermeable forms such as structures used for housing, commerce, and transit. Not captured in these figures on wetland loss are other forms of wetland alteration, such as sedimentation, siltation, eutrophication, changes in prevailing patterns of surface and soil storage of water, introduction of novel species, habitat disconnection, and the loss of charismatic species (Grayson, Chapman, & Underwood, 1999; Shields et al., 2008; Xu, Chen, Yang, Jiang, & Zhang, 2019; Zedler & Kercher, 2005).

Research on the loss and alteration of wetland cover has emerged contemporaneously to predictions about storm intensity and frequency, and consequent flooding. Climate models project an increase in the frequency of present-day 100-year flood events in many parts of the globe (Intergovernmental Panel on Climate Change, 2021; Milly et al., 2002). By the end of the 21st century, 50-year magnitude floods may become 13-year magnitude flood events in the climate of 2071-2100 (Li et al., 2021). Globally, climate and water disasters have increased five-fold between 1970 and 2019, with floods causing an estimated 58,700 deaths and roughly US$ 1.1 trillion in damages during this period (World Meteorological Organization, 2021).

Storm characteristics are not under the control of cities, but many other factors that control flooding are affected by the development decisions that cities make. There are many forms of flooding that cities manage, such as fluvial and coastal flooding; however, this dissertation is concerned only with pluvial flooding. Pluvial flooding is characterized as surface ponding or overland flow that occurs when rates of precipitation exceed the capacity of drainage systems and/or surfaces to remove it (Falconer et al., 2009). Pluvial floods threaten many aspects of urban communities, causing loss of life, damage to property, disruption of transportation networks, and displacement (Chang et al., 2010; Douglas et al., 2010; Falconer et al, 2009; Yin et al., 2016). The ways in which precipitation, urban stormwater management practices, and biophysical characteristics of the urban and peri-urban landscape interact to determine characteristics of pluvial flooding, such as location, duration, and depth, are increasingly better understood (Rosenzweig et al. 2018; Westra et al., 2014). The tools to assess pluvial flood risk are improving but remain in need of further improvement (Rosenzweig et al., 2021).

Consideration of the long-term risk of pluvial flooding is critical for urban communities, as the components (e.g., pipes, canals, drains) of the stormwater management systems of cities typically have expected lifespans on the order of 50-100 years or more (Zhou, Mikkelsen, Halsnæs, Arnbjerg-Nielsen, 2012). Such long-lived infrastructure is then expected to perform under several different climate conditions over the course of its lifetime. And yet, in the United States and many other countries, conventional gray stormwater management infrastructure in urban areas is typically designed for managing two- to fifteen-year design storms (ASCE/Environmental & Water Resources Institute 2006), rendering it unequipped for managing increasingly common extreme precipitation events brought about by climate change (Ashley et al., 2005). Though this gray infrastructure can be designed or upgraded to manage storms of greater magnitude (ASCE/Environmental & Water Resources Institute, 2006), this is a costly and spatially disruptive process that requires considerable time from idea conception to execution.

In the past several decades, urban communities have explored alternatives to traditional forms of stormwater management system design to be more adaptive to changing climates and extreme events beyond the two- to fifteen-year design storm (Kim, Chester, Eisenberg, & Redman, 2019). There has been a particular interest in the use of new forms of infrastructure, particularly urban ecological infrastructure (UEI; Childers et al., 2019; otherwise known as (GI), green stormwater infrastructure (GSI), or, more broadly, nature-based solutions (NBS)), which may provide similar or greater benchmarks of performance, under a broader range of climate conditions, than traditional "gray" infrastructure, while providing other services, known as ecosystem services, that gray infrastructure cannot (Demuzere et al. 2014; McPhearson, 2014). Cities pursuing UEI solutions may expand their stormwater management system by designating new areas as sites of GI, for example by constructing new urban parks to facilitate greater infiltration. Or cities may retrofit portions of the existing system to incorporate more GI elements, which may synergize with other initiatives to enhance degraded urban landscapes and thus provide desired ecosystem services beyond stormwater management (Haaland & van den Bosch. 2015). Additionally, cities may incorporate existing (referred to in the literature as "remnant") wetlands into the urban environment as they expand (Holzer, 2014).

Many urban communities have set their sights on wetland UEI as a potential solution to the threats posed by present-day and future pluvial flood risk, which include injury, displacement, and death. They are thus presented with a serendipitous opportunity to reverse the trend of wetland loss while simultaneously relieving cities of pluvial flood risk and making cities more pleasant habitat for humans. However, critically absent from current studies on the flood mitigation services of wetlands are city-wide studies on whether and how the performance of urban stormwater management systems changes when urban wetlands are constructed, restored, or incorporated. Furthermore, in drainage systems that already include wetland features, there are no city-wide studies on the effects of changes in wetland extent or configuration on drainage system performance. Researchers have previously argued that, at least outside the urban context, system-specific knowledge is necessary to accurately estimate effects of various wetland characteristics on the flood regulation services that wetlands may provide (Acreman & Holden, 2013; Kadykalo & Findlay, 2016). Wetland dimensions, extent, antecedent storage conditions, rates of infiltration and evapotranspiration, and configuration are all likely to influence the performance of any stormwater management systems into which urban wetlands may be included.

Conventionally, urban communities rely in part on models to assess flood risk and the effectiveness of planned or existent stormwater management systems in mitigating pluvial flood risk. Pluvial flood modeling is critical to predicting flood locations and damages, even though model architecture and estimates have their limitations (Rosenzweig et al., 2020). Models in practice in urban watersheds are often two-dimensional and predict areas of elevated runoff (HEC-RAS), one-dimensional and predict flooding locations and volumes within urban drainage areas (EPA SWMM, Arc-Malstrøm (Balstrøm and Crawford, 2018)), and coupled 1D-2D models (PCSWMM) that predict flooding locations and even flood depths. Tu, Wadzuk, and Traver (2019) reviewed all of the major models of these types currently being used to simulate functions of stormwater management systems, with a special focus on those incorporating UEI (in the paper, referred to as GSI), such as GIFMod, MOUSE, and EPA SWMM, and found that none of them had comprehensive abilities to model runoff generation, routing, GSI vadose zone/groundwater, evapotranspiration, and programmable control in urban settings. New, so-called “data-driven” models are of increasing interest among academics (Heonin et al., 2013; Penning-Roswell & Kordenwall, 2019; Wolfs & Willems, 2016) and offer advantages over other models insofar as they are computationally faster and not deterministically constructed (Li & Willems, 2020). The adoption of data-driven models by practitioners in cities is, in my experience, quite low, and older, more deterministic forms of flood modeling still dominate the field.

Developing accurate and useful models of pluvial flooding in urban watersheds requires accurate model inputs and validation, but these data are not presently available to cities. Different models accommodate different model inputs, including digital elevation models (DEMs), stormwater management networks, design storms, land cover typologies and their rates of infiltration, and building footprints, among other elements. In many cities, it is difficult for researchers to provide all of these elements as inputs even though their availability improves the characterization of pluvial floods. If an urban community wishes to assess pluvial flood risk in the future, they commonly will not have access to scenarios of urban development and land cover or to climate model data that have been downscaled to the city. In the U. S., present-day layouts of stormwater management networks are commonly inaccessible due to national security concerns. However, given changing climates in cities, the expectation of long lifespans of stormwater infrastructure, and the various factors that contribute to pluvial flooding, there is a clear rationale for the creation and sharing of such data.

In the case that urban communities have data on future land cover and climates, such data come with substantial uncertainty. Consequently, there is a need to design stormwater management systems to perform well under a range of plausible land cover and climate scenarios (Measham et al., 2011; Fig. 1.1). The development and use of scenarios reflecting a range of future city land cover configurations have increasing purchase among academics and practitioners. Such scenarios are defined as plausible, coherent narratives about the future of a place or a situation and are used to produce anticipatory knowledge (Millennium Ecosystem Assessment, 2005). Scenarios are also characterized by the exploration of desirable configurations of cities (Iwaniec et al., 2020;

McPhearson, Iwaniec, & Bai, 2016), and often incorporate transdisciplinarity and co-production to capture a range of visions from a broad cross-section of sociodemographic characteristics (Cook et al., 2021; Jahn, Bergmann, & Keil, 2012; Lang et al., 2012). These scenarios may be used as inputs to models that estimate land cover change at some time in the future (Iwaniec et al., 2020; Ortiz et al., 2020; Sauer et al., *in revision*), and the outputs of these models can give researchers and urban communities an idea of the range of possible land cover futures.

Model estimates of a range of future climates are necessary to assess expected future storm and climate conditions in which a stormwater management system must perform. Changes in annual rainfall volumes and timing may shift conditions away from those for which stormwater management systems were designed. In the case that annual rainfall diminishes, and the intensity of the average storm also diminishes, cities may experience a reduced number of floods due to a drier stormwater management system and reduced load during storms. However, if annual precipitation increases and/or the intensity of the average storm increases, cities may contend with greater numbers of floods in a year due to a waterlogged stormwater management system and increased load during storms. There are also intermediate scenarios with which to contend, such as increased annual precipitation through an increase in the number of storms of lesser intensities, or reduced annual precipitation with an increase in the intensity of extreme storms, among others–all of which have their potential impacts on flood risk. The direction and degree of these effects may in turn vary depending on the configuration and inventory of a stormwater management system and its urban drainage shed. Where downscaled climate model data are available for cities, they should then be used to assess a range of future climates for the construction of a range of input design storms and for establishing a range of antecedent storage conditions of system elements that may be present.

Current models for estimating pluvial flood risk are recognized as incomplete and/or insufficient for representing all hydrological processes and for assessing impacts that pluvial flooding has on the lives of citizens (Rosenzweig et al., 2021; Tu, Wadzuk, & Traver, 2019). There are other forms of flood risk assessment, such as participatory mapping, that do not rely on modeling but that can nonetheless be used to direct the efforts of modelers to improve their capabilities for flood risk assessment (Mercer et al., 2009). Participatory mapping, whereby individuals or groups from communities are invited to provide geographical information in easy-to-access ways (Cadag & Gaillard, 2012), is common among development workers and researchers (Chambers, 2008; International Fund for Agricultural Development, 2009). It offers the advantages of relative ease of setup and low cost of execution (Cadag & Gaillard, 2012), and has low barriers to deployment and citizen participation compared with other methods of collecting citizen knowledge of hazards that may not be captured in quantitative models (Schumann, Binder, & Greer, 2018). Participatory mapping may also be considered a form of transdisciplinary work, which emphasizes the inclusion of knowledge from non-academics (Scholz, 2017) and the active inclusion of “civil society actors as an external corrective on the ‘blind spots’ in the science system” (Schneidewind, Singer-Brodowski, & Augenstein, 2016).

Thus, both the stakes and the benefits are high to design urban environments and their stormwater management systems to manage pluvial flood risk. Viable pathways toward this goal involve the considered use of models of climate, models of land cover, models of stormwater management systems, and implementation of UEI—and our ways of using them must be improved in the process. The objective of this dissertation is to explore the use of wetland UEI, climate models, scenario models, and participatory mapping to better characterize flood risk from extreme storm events now and into the future. Four dissertation chapters address the following questions:

1. How do characteristic hydrologic functions of wetlands, particularly surface and soil water storage, change across seasons, and how do these changes affect their abilities to mitigate pluvial flood risk during extreme storms?
2. How will pluvial flood risk from extreme storms change across a range of potential urban development scenarios in a city that uses wetland UEI in its stormwater management network?
3. How will pluvial flood risk change in combination with the effects of future land cover change on wetland and contributing watershed area, and climate change on average climate and extreme storm events, in a city that uses wetland UEI in its stormwater management network?
4. How can new forms of pluvial flood risk assessment improve upon our understanding of pluvial flood risk, and what are their limitations?

I address these questions in an interdisciplinary way, using mixed methods, including field and computational modeling methods, to improve upon our understanding of pluvial flood risk more generally and how wetland UEI may be used to mitigate pluvial flood risk. In chapter 1, I use data I collected on surface and soil storage of wetland UEI in Valdivia, Chile, and the city's model of the stormwater management system in EPA SWMM, to demonstrate how intra- and interannual fluctuations in antecedent storage conditions affect pluvial flood risk from extreme storm events (Sauer et al., in preparation-a). In Chapter 2, I combine projected changes in land cover generated from a scenarios workshop and cellular automata-based model to estimate changes in wetland UEI and impervious cover in EPA SWMM, and estimate consequent pluvial flood risk during extreme storm events (Sauer et al., in revision). In Chapter 4, I combine these same models estimating changes in wetland UEI and impervious cover with downscaled climate model data in the city to estimate pluvial flood risk under changing climates and changing urban configurations (Sauer et al., in preparation-b). In chapter 5, I assess the strengths and limitations of using Arc-Malstrøm and participatory mapping methods in estimating areas of pluvial flood risk in the context of Hermosillo, México, and consider the ways these approaches can improve upon our current methods (Sauer et al. in preparation-c). In Chapter 6, I summarize the lessons learned in this process, situating this dissertation in a context of pluvial flood risk assessment more broadly, and suggest pathways forward to developing future cities that are more resilient to a range of possible climates and urban configurations.

Returning for a moment to the motivations of this dissertation, it is my hope that the lessons contained herein are directly useful to communities of Valdivia and Hermosillo, which hosted me during the process of its completion. It is also my hope that other urban communities around the globe can benefit from its findings. Though it may not be explicit in the rationales of each chapter, I have accomplished this work only out of care for my family, friends, and greater community. Publishing on my work will accomplish a general dissemination of lessons among academic communities, but I also seek to work directly with city officials and wetland conservationists in Valdivia toward ensuring its direct utility to citizens.

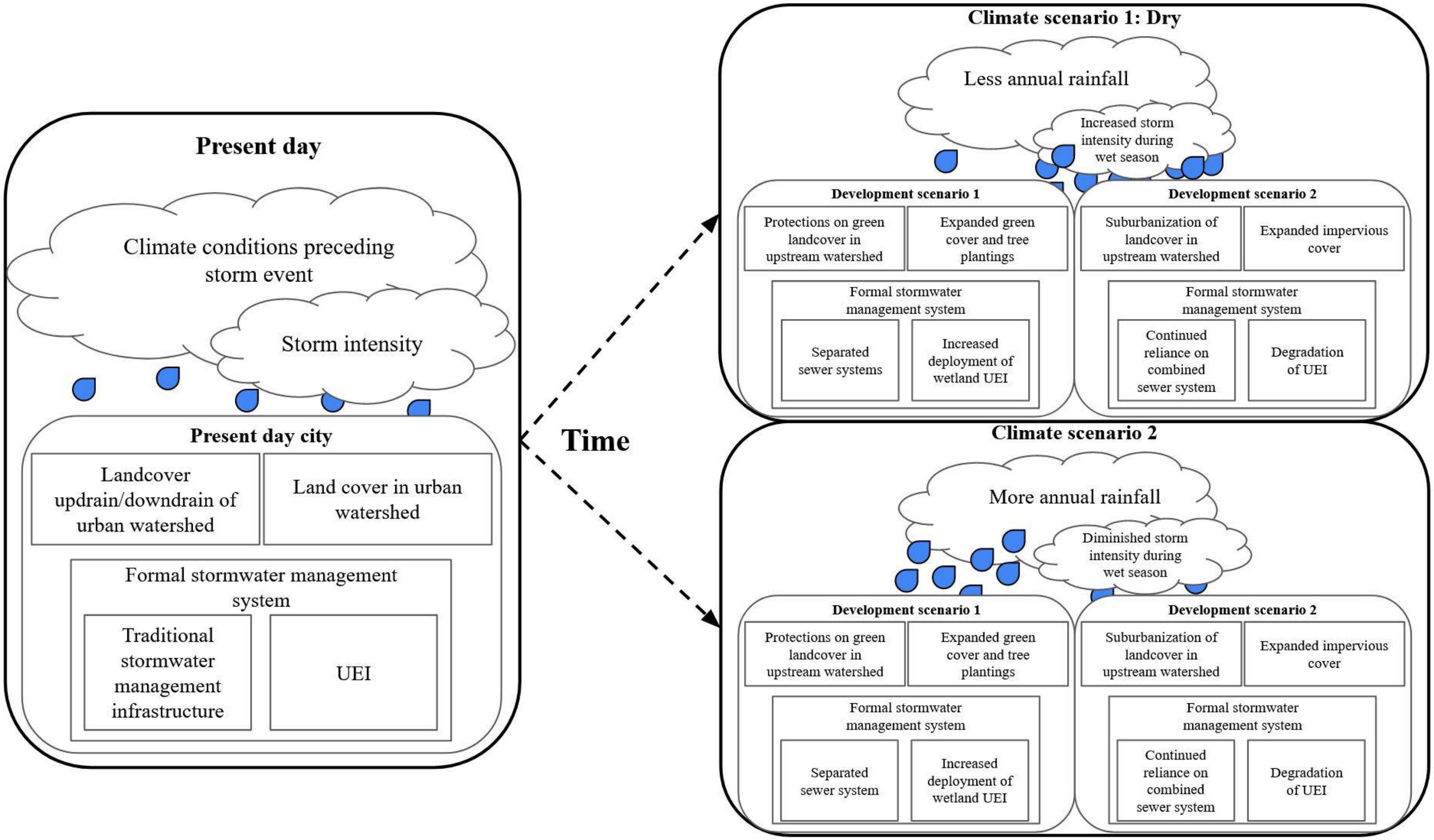

Figure 1.1. Conceptual diagram of important contributing factors of pluvial flooding in cities considered in this dissertation.

CHAPTER 2

WETLAND STORAGE DYNAMICS AND VARIABLE FLOOD RISK MITIGATION SERVICES IN VALDIVIA, CHILE

2.0. Abstract

Pluvial flooding, or flooding that results from precipitation rates that exceed the removal rates of drainage systems and of the permeable landscape, poses a major challenge to cities. Flood risk managers are looking to new forms of infrastructure, particularly green stormwater infrastructure or urban ecological infrastructure, to better manage this problem under present and future conditions. Wetlands, whether natural, modified, or constructed, have been proposed as a particularly promising form of infrastructure for this purpose, but real-world studies of wetland characteristics and flood risk are in short supply, do not consider temporal changes in wetland surface and soil storage conditions, and most often are not conducted at city scales. In the present study, I monitored six wetlands for changes in wetland water levels for two separate years and soil water storage in one year in Valdivia, Chile: a city notable for the presence of wetlands throughout its formal stormwater management system. For each season, I then used EPA SWMM to construct models with no additional storage capacity of soils, models where soil storage is simply added to the surface storage, and models that allowed for infiltration, for four drainage catchments of the city and for each season. I found slight but significant positive relationships between systemic flooding and wetland storage volumes in one drainage catchment, and for the aggregate flood risk of all four catchments. I also found that models that allowed for infiltration had substantially lower amounts of flooding than models that did not, and that this reduction was much greater

than any reduction between wetland water levels between seasons. Broadly, I concluded that wetland characteristics such as surface and soil storage and infiltration play major roles in potential flood risk reduction, but that system- and condition-specific considerations confound narratives that wetlands universally reduce flood risk.

2.1. Introduction

The frequency of pluvial floods is expected to increase with climate change-driven increases in the intensity and frequency of rainfall in cities across the globe (O'Donnell & Thorne, 2020), and from the replacement of natural, more permeable landscape features with less permeable ones as part of dominant patterns of urbanization (Lashford et al., 2019). Pluvial flooding is surface ponding or overland flow that occurs when rates of precipitation exceed the capacity of drainage systems and/or surfaces to remove it (Falconer et al., 2009; Rosenzweig et al., 2018). Pluvial flooding can lead to loss of life, damage to property, and disruption of transportation networks (Chang et al., 2010; Douglas et al., 2010; Falconer et al., 2009; Yin et al., 2016).

In response to the increasing threat and impacts of pluvial flooding, many cities have developed strategies to restore or construct natural landscape features, or alternatively incorporate and maintain them, in order to harness their stormwater management services (Chan et al., 2018; Fletcher et al., 2013). Such natural landscape features are variably referred to as green infrastructure, green stormwater infrastructure (GSI), urban ecological infrastructure, or nature-based solutions. The practice of constructing, restoring, incorporating, and maintaining them may be considered key to low-impact development strategies or sustainable urban drainage systems to reduce pluvial flooding (Chan et al., 2018; Fletcher et al., 2015; Lashford et al., 2019; Li et al., 2020).

Wetlands have been of particular interest to cities in managing their pluvial flood risk (Chan et al., 2018; Elmqvist et al., 2015), though their abilities to actually accomplish this are under-studied. I find no published studies examining the effects of urban wetlands on pluvial flooding at the city scale (Sauer et al., in revision), though one modeling exercise examined the flood mitigation effects of a proposed constructed urban wetland in a smaller urban drainage catchment scale (Kumar et al., 2021). Nonetheless, wetlands have been included as part of a broader low-impact development strategy in cities in China (Zhou, Leng, & Huang, 2020). From my conversations with practitioners in the U.S.A. and Chile, other cities are already managing wetlands to augment their flood mitigation abilities (personal communication).

Many key features of urban wetlands differ from other forms of urban stormwater infrastructure, and affect overall stormwater management system performance (Line et al., 2012). Wetlands store large volumes of water as a function of their basin morphology (Mitsch & Gosselink, 2015) and, in an urban context, depending on the locations and elevations of inlet and outlet pipes. Wetlands may also store or remove water from urban stormwater management systems through evapotranspiration and infiltration to soils and groundwater (Gülbaz & Kazezyilmaz-Alhan, 2016). Both surface and soil storage reduce downstream or downdrain rates of flow and slow the propagation of peak flows compared to artificially or naturally channelized forms of landscape cover (Fletcher et al., 2013; Kadykalo & Findlay, 2016), and these effects may also be controlled in part by wetland soil type (Gülbaz & Kazezyilmaz-Alhan, 2016). Wetland storage levels may change substantially according to year, season, and potentially even daily hydrological cycles (Mitsch & Gosselink, 2015). Wetland storage can also be actively managed through the use of engineered controls on flow (Kumar et al., 2021). The most similar GSI element to a wetland that has been well-studied and deployed widely, the retention pond (Keyvanfar, Shafaghat, Ismail, Mohamad, & Ahmad, 2021; Verstraeten & Poesen, 1999), has several of these hydrological features in common with wetlands, but they may be lined to prevent infiltration and may not be vegetated to the same degree as wetlands.

The abilities of models to fully consider the effects of the presence and characteristics of wetlands on urban hydrology are limited. Tu, Wadzuk, and Traver (2019) reviewed all the major models currently being used to simulate functions of stormwater management systems, particularly those incorporating green stormwater elements, such as GIFMod, MOUSE, and EPA SWMM, and found that none of them had comprehensive abilities to model runoff generation, routing, GSI vadose zone/groundwater, evapotranspiration, and programmable control (such as in flow control gates) in urban settings. The authors (Tu, Wadzuk, & Traver, 2019) focused on the inability of SWMM to model multiple soil layers with differing properties that would affect rates of infiltration and potential storage volumes, particularly when the vadose zone is full. Nonetheless, modeling pluvial flooding in cities remains a critical need for stormwater managers to understand flood risk and evaluate the benefits of infrastructure solutions, green or otherwise (Rosenzweig et al., 2020).

This study investigates the relationship between inland urban wetland storage dynamics and systemic flooding at the city scale. For my study system, I first asked the following question: How does surface and soil water storage change in urban wetlands across days, seasons, and years? I hypothesized that storage would change substantially between years and seasons, and that there may be detectable diurnal patterns of change as well in the drier, warmer seasons due to evapotranspiration. Additionally, I asked: How does systemic flooding change as a result of these changes in water storage across days, years, and seasons? I hypothesized that there would be a positive correlation between water storage in inland urban wetlands and systemic flooding, and that flood risk would be greatest during the wetter, cooler seasons. Finally, I also asked: How do different modeling conceptions of wetland water storage affect estimates of systemic flooding? I hypothesized that models allowing for soil infiltration and storage would estimate lower systemic flood volumes than models that only allowed for surface retention.

2.2. Materials and Methods

2.2.1. Study Site

Valdivia, Chile is a city of approximately 170,000 people in the southern half of Chile, 850 km south of the capital Santiago, in the Región de los Ríos (Fig 2.1). The city experiences high annual precipitation (1760 mm; 69.29 in), with a pronounced wet season that typically begins in March and ends in October. The city center is located about 12 km inland from the Pacific Ocean at the confluence of three major rivers. Its ecosystem is classified as a temperate rainforest, and the low areas of the landscape often feature expansive wetland cover.

In general, wetland soil textures in Valdivia are classified as silty loam (Instituto Nacional de Investigación de Recursos Naturales, 1978). However, soils toward the edges of wetlands may be a mixture of silty loam and other coarser grain types due to intentional or unintentional partial infill. Soils in wetland interiors may also be a mixture of silty loam and other soil textures, due to processes of sediment production and transport characteristic of urban and urbanizing watersheds (Allmendinger, Pizzuto, Moglen, & Lewicki, 2007; Gellis et al., 2017), and from exchange of soils from wetland peripheries and the wetland interior.

Valdivia’s stormwater management system depends heavily on the hydrological function of wetlands that run from the city’s center to its peripheries, and ultimately to the Río Valdivia (Fig. 2.1). A magnitude 9.5 earthquake in 1960 created many of these wetlands, which as of 2015, according to spectral analysis of a 2015 orthophoto of the city conducted by the Universidad Austral de Chile, cover 22% of the city’s total area. Though some proportion of the wetlands has been lost since 1960 due to sanctioned development or unsanctioned infilling, city planners in Valdivia made the uncommon decision to incorporate the wetlands as components of the stormwater management system rather than follow the traditional route of urban development that converts wetlands to other land cover/land use classes and approximates their conveyance function via pipes and canals (Ministerio de Obras Públicas de Chile, 2012). Grassroots organizations have also conserved some of these wetlands primarily for their cultural services and reasons of environmental justice and equity (Correa et al., 2018), though as part of the stormwater management system they nonetheless provide stormwater management services.

2.2.2. Continuous wetland surface water monitoring

Wetland surface storage is a changing and potentially critical factor in determining the flood mitigation services of wetlands. I installed HOBO Water Level Loggers (U20L-04) in six wetland sites across Valdivia to monitor surface water levels at fifteen-minute intervals (Fig. 2.1). I selected sites from a variety of configurations (updrain and downdrain, urban and periurban, riverine and inland) in order to capture a range of considerations that might affect wetland storage conditions. These wetlands are linked to either downdrain wetlands or downdrain pipe systems, in either case via constructed pipes or canals, that ultimately flow into the rivers that surround the city. Outflow pipes from the wetlands were typically situated between 0.25 - 0.5 meters above the wetland thalweg. Depending on wetland stage, updrain wetlands may be variably connected or disconnected via surface flow to downdrain portions of the drainage system.

Sensors were hung from cables strung through the sides of a polyvinyl chloride (PVC) pipe housing, and sensors rested just above the surface of the soil of the wetlands. This PVC housing was perforated to allow water and air to exchange freely across the housing at all wetland water stages.

I collected wetland stage data at wetlands 1-5 across two different field seasons (2017-2018 and 2019-2020), and for wetland 6 in only one field season (2017-2018; Table 1). The sensor in wetland 6 was either destroyed or lost in the construction of a park enhancement that extended into the wetland. Additionally, due to user error, I did not collect data in wetland 5 between the dates of 11/28/2017 and 1/17/2018 (Table 1). Data from the sensors were downloaded on-site to a laptop every two to four weeks, with the exception of the period between December 28, 2017 and January 17, 2018 when data retrieval was delayed.

I divided Valdivia's seasons into four categories: (1) early dry (November and December), (2) late dry (January and February), (3) early wet (March - May), and late wet (June - October). Valdivia's wettest month is typically July, for which I have no data; however, precipitation events may still occur 3-6 days a week through late October, depending on the year.

2.2.3. Estimating daily changes in wetland storage

To explore daily surface-water storage dynamics in my wetlands, I developed a method for detecting the presence of a daily tidal phenomenon, which potentially would have been driven by evapotranspiration (ET; Bois et al. 2017). For each wetland I monitored, I removed from its depth data all days when precipitation was recorded as being greater than or equal to 2 mm, in order to control for daily wetland depth differences due to direct precipitation input and runoff influxes to wetlands. Further, I removed wetland stage data for all days when the daily minimum stage of a wetland was 0 m, indicating that there was no surface water available to monitor for daily stage changes. I also removed data for days when the sensor was installed or removed. I then separated the stage data by day.

In order to examine my daily stage data for diurnal tidal patterns, I converted stage into a metric, "relative stage." That is, for each day, *d*, stage was normalized at a given time step, *t*, by its minimum and maximum recorded daily values, according to the following equation:

$$Relative\ stage_{d,t} = \frac{stage_{d,t} - minimum\ stage_d}{maximum\ stage_d - minimum\ stage_d} \tag{1}$$

Thus, for a given day and time step, relative stage ranges between 0 (the minimum daily stage) and 1 (the maximum daily stage minus the minimum daily stage).

2.2.4. Calculating daily and seasonal minimum, average, and maximum wetland stages

In order to calculate daily minimum and maximum wetland stages, I sorted my stage data by day and selected the minimum and maximum values. To calculate daily mean wetland stages, I calculated the mean for all stage data collected in a given day. For seasonal minimum and maximum wetland stages, I calculated the mean of all daily minimum and maximum values for each season. To calculate seasonal mean wetland stages, I calculated the mean of all stage data collected in each season.

2.2.5. Estimating wetland surface-water storage volumes

To estimate the volumes of wetland surface storage, I first converted a 2016 contour map (1-m resolution) of Valdivia to a triangulated irregular network (TIN). Then, using ArcGIS Pro (ESRI), and its "Calculate TIN Volume" tool, I generated virtual surfaces at 0.25-m intervals from the wetland base to the height of the lowest wetland edge, and calculated the volume underneath the virtual surfaces. I used these calculated volumes and depths in EPA SWMM for wetland volumes.

2.2.6. Soil water storage surveys

I estimated soil water storage using the results of two surveys. In both surveys, I extracted soil cores to a 20-cm depth using a semi-cylindrical gouge auger with a 3.175-

cm diameter. Also in both surveys, soil cores were divided into two segments, representing soils from depths of 0–10 cm and 10–20 cm. Soil segments were wrapped in foil before being placed in an airtight plastic bag and transported to the lab within 45 minutes of collection. Once in the lab, cores were cut to a regular "puck" shape using a loop knife, and placed on a scale to determine their wet mass. Cores were then dried in a drying oven at 105° C for 24 hours. Dried cores were returned to the scale to determine their dry mass. Using the difference between wet and dry mass, a water density of 997 kg/m^3 and the dimensions of the wet sample, I determined the soil water storage of a given core, in units of m^3 water/m^3 soil:

$$Volumetric\ soil\ water\ content = \frac{\frac{m_{wet}-m_{dry}}{\rho_{water}}}{V_{core}} \qquad (2)$$

In my survey to collect saturated cores, which occurred in July of 2017, I collected soil samples from all wetlands within the SWMM that had public access points accessible by vehicle. I also collected at least one saturated soil core from each wetland in which I had installed stage sensors. Soils were considered saturated if they had visible standing water above their surface. I did not collect soil samples in instances where the soil core was too unstable to maintain structural integrity during collection, transport, and pre-oven preparation. Ultimately, I collected soil cores from 27 different wetland sites across Valdivia, including at least one saturated soil core from each of my wetland sites (Fig. 2.2).

In my survey to collect data on unsaturated soils, which I conducted in the driest part of the dry season in February of 2018, I developed a sampling strategy that would allow me to extract unsaturated soils with a range of moisture conditions. Additionally, I surveyed only those wetlands in which I had installed stage sensors. I divided the wetlands into three different zones, referred to as top, middle, and bottom, that represent the top third, middle third, and bottom third of the wetlands in terms of relative elevation using the ArcGIS Pro (ESRI) and the TIN generated from Valdivia's contour map. For each wetland, I generated four random locations in each zone to sample unsaturated soils (Fig 2.3). During the dry season campaign, I was able to collect a total of 20 unsaturated soil samples and 14 saturated soil samples across the six monitored wetlands (Fig. 2.2).

2.2.7. Modeling soil storage in EPA SWMM

Among urban flood models, EPA SWMM (EPA) is one of the most comprehensive programs for modeling GSI. It is commonly used by municipalities worldwide to model the performance of their stormwater management system and to evaluate the utility of GSI for stormwater management systems. To examine the effects of changes in soil storage capacity of my study wetlands on flooding, I constructed nine models in EPA SWMM: one that assumes no soil storage (the "base" model), four that assume wetlands allow for infiltration at different rates according to seasonal soil-moisture conditions (the "infiltration" models), and four that model wetland soil storage as additional surface storage (the "surface storage" model).

EPA SWMM allows for, but does not require, infiltration in wetlands. Furthermore, the EPA SWMM effectively removes water from the system at the point of infiltration, such that water that infiltrates into wetlands will not return as surface or groundwater flow elsewhere in the model unless the model is otherwise modified. Finally, infiltration in EPA SWMM occurs throughout the entire duration of the model run, rather than for a specific period of time or until a specific volume of water has infiltrated the available soil pore space.

Soils in my wetlands, depending on the season and antecedent rainfall conditions, may have high water tables that sit above the base of the wetland. In real-world conditions, a high water table will lead to a reduction in infiltration rates. Consequently, some accuracy in modeling infiltration is lost in my system due to EPA SWMM's modeling mechanisms. Still, infiltration rates for silty loam are typically 6.9 mm/hour, and given that my soil cores represent soil conditions of the top 20 cm of soil, I can reasonably expect infiltration to proceed at around this rate during seasons when the water table is low.

I constructed infiltration models using the soil-moisture data collected during soil surveys. I divided the observed range of soil moisture between the wet and dry seasons into quartiles, increasing in soil moisture content from the late dry season to the early dry season to the early wet season to the late wet season. These values were used in the infiltration models as the "initial deficit" values for soils in my monitored wetlands.

As an alternative to allowing for infiltration processes that permanently remove water from the system, and that proceeded indefinitely at a constant rate, I constructed a model of soil-water storage where I increased surface water storage such that it included the volume of soil pore space available up to 20 cm depth. I called this model the surface storage model. Mechanistically, this conception of soil storage would be equivalent to assuming that water infiltrates instantaneously up to a volumetric limit. This model then removes the temporal aspect of transferring water from surface to soil compartments, but may nonetheless be a more accurate representation of the finite storage volume that wetland soils provide.

Similar to the infiltration models, using the same seasonal gradient of soil moisture described earlier, I increased wetland volume according to available soil-moisture content. I calculated the area of wetlands under water at various stages, multiplying this value by 20 cm (representing the depth of soil available for infiltration), and then multiplying this value by the volumetric soil moisture volume.

2.2.8. Determining relationship between antecedent surface and soil water storage conditions in wetlands and systemic flooding

To examine the relationship between antecedent surface and soil-water storage conditions and systemic flooding, I input seasonally averaged minimum, mean, and maximum wetland stage values in the "initial depth" field in EPA SWMM for the monitored wetlands. In the base models, the differences in these seasonally averaged stages were the only source of difference between models. In the infiltration models, I also allowed for infiltration and altered the "initial deficit" field to reflect seasonal differences in soil water storage. In the surface storage models, I increased wetland volumes to reflect seasonal differences in soil water storage between seasons and did not allow for infiltration.

2.3. Results

2.3.1 Changes in surface storage

Wetland stage varied substantially between wetlands, and within the same wetlands, between seasons and campaigns (Fig. 2.4). In campaign 1, wetland stage generally decreased from September/October through mid-March. An exception to this trend, stage in wetland 2 increased during the early dry season. In campaign 2, wetland stage generally decreased from September/October through January, except for after major storms, such as in late October. Stages in all wetlands, for any given season, were generally lower in campaign 2 than they were in campaign 1 (Fig. 2.4).

Mean, minimum, and maximum daily stages for a given wetland were all greater in campaign 1 than campaign 2 (Table 2). In campaign 1, these values all tended to be greater in the wet seasons than they were in the dry seasons, and for wetlands 2, 3, 4, and 5 were greater in the early wet season than they were in the late wet season. Also in campaign 1, minimum, mean, and maximum stages tended to be lowest in the late dry season, with the exception of wetland 3, which had its lowest values in the early dry season, and wetland 2, which had lower values in the late wet season. In campaign 2, minimum, mean, and maximum values were all highest in the late wet season and declined to their lowest point in the late dry season, when data collection ended. Across both campaigns, wetlands 2, 3, 4, and 5 all had their highest minimum, mean, and maximum stages during the late wet season and declined through the late dry season. Wetlands 3 and 4 tended to have the highest water levels of any wetlands in any season and reduced to 0 m in campaign 2; while wetland 5 tended to have the second lowest minimum, mean and maximum stages of the wetlands in any season or any campaign, yet never reached a stage of 0 m even when wetlands starting with higher stages dried (Table 2). In campaign 1, some wetlands progressively dried from the late wet season to the late dry season, though some did not, whereas in campaign 2, all wetlands dried from the late wet to the late dry season. Wetlands tended to have greater differences in stage between campaigns than they did between seasons (Fig. 2.5; Fig. 2.6). Distribution of wetland stages in campaign 2 was generally lower than in campaign 1 than in campaign 2, though for wetlands 1 and 2, the distribution of wetland stages did overlap between the two campaigns.

2.3.2. Changes in soil storage

Average soil moisture content was 0.386 m^3/m^3 for saturated soils and 0.308 for unsaturated soils. Thus, the soil moisture deficit (the difference between soil porosity and soil moisture content) in the dry season averaged 7.8%. Across seasons, soil moisture deficit was 0.0%, 5.2%, 7.8%, and 2.6% in the late wet, early dry, late dry, early wet seasons, respectively.

2.3.3 Detecting and estimating daily changes in wetland storage

Days when precipitation was greater than 0.002 m or the minimum stage was 0 m were removed from the data, and the remaining data were analyzed for daily changes in wetland stage (Fig. 2.7). For all wetlands, across both campaigns, the average difference between the maximum water level and minimum water level was 0.037 ± 0.029 m, in campaign 1 was 0.038 ± 0.035 m, and in campaign 2 was 0.033 ± 0.011 m, which I have also separated by wetland (Tab. 2.3). Maximum daily difference for all wetlands was

0.315 m in campaign 1 and 0.122 for campaign 2; minimum daily difference was 0.010 in campaign 1 and 0.009 in campaign 2 (Fig. 2.8). The average daily difference in wetland volumes across both seasons equated to 14,800 ± 9,900 m^3 of water between the daily minimum volume and the daily maximum volume.

Raw daily stage data were transformed according to equation 1 to produce values for relative daily stage (Fig. 2.8). For the majority of days, wetlands had higher stages during the early and later parts of the day, with and their lowest stages generally occurring around midday. Drawdown in all wetlands typically began around 04:00. Among monitored wetlands, wetland 5 most frequently diverged from this pattern.

2.3.4. Effects of antecedent surface and soil water storage on flooding

For the base and surface storage models, flood volumes increased with average wetland stage in the southeast sector (base: $R^2 = 0.653$, $p < 0.05$; surface storage: $R^2 = 0.579$, $p < 0.05$) and when data were aggregated across all sectors (base: $R^2 = 0.676$, $p < 0.05$; surface storage: $R^2 = 0.617$, $p < 0.05$; Fig 2.9). This trend resulted in very small increases in flood volume for the southeast sector, and was not significant in any direction for the other individual sectors. All flooding occurred in system elements that were downdrain of the wetlands.

In models that allowed for infiltration at different rates according to antecedent soil moisture conditions, I found that systemic flooding generally increased with average wetland stage in the southeast sector (LW: $R^2 = 0.377$, $p < 0.05$; ED: $R^2 = 0.690$, $p < 0.05$; LD: $R^2 = 0.717$, $p < 0.05$; LW; EW: $R^2 = 0.657$, $p < 0.05$), and when data were aggregated across all sectors (LW $R^2 = 0.263$, $p < 0.05$; ED: $R^2 = 0.500$, $p < 0.05$; LD: R^2

$= 0.651$, $p < 0.05$; EW: $R^2 = 0.549$, $p < 0.05$; Fig. 2.10), but this trend was not significant for any other individual sector. Flood volume was lower for all average wetland stages using the infiltration model, regardless of season, than in the base and surface storage models. All flooding occurred in system elements that were downdrain of the wetlands.

2.3.5. Seasonal differences in systemic flood risk between seasons and between models

In the base and surface storage models, during campaign 1, flood risk decreased from the late wet to the early dry season, increased from the early dry season to the late dry season, and then increased again from the late dry to the early wet season (Fig. 2.11). For both models during campaign 1, flood risk was greatest in the early wet season, and least in the early dry season. In the base and surface storage models, during campaign 2, flood risk decreased from the late wet season to the early dry season, and again from the early dry season to the late dry season. For both models during this campaign, flood risk was greatest in the late wet season and least in the late dry season.

For any given season, for either campaign, flooding was estimated to be lower in the infiltration models than in the base or surface storage models. In the models that allowed for infiltration, for both campaigns, flood volumes were greatest in models with soils with the highest initial moisture content (late wet), and decreased with initial moisture content from late wet, to early wet, to early dry, to late dry. For all models during campaign 1, flood risk was greatest in the early wet season, and least in the early dry season. For all models during campaign 2, flood risk decreased from the late wet season to the early dry season, and again from the early dry season to the late dry season.

2.4. Discussion

For my study system, I first asked the exploratory question: How does surface and soil water storage change in urban wetlands across days, seasons, and years? I hypothesized that storage would change substantially between years and seasons, and that there may be detectable diurnal patterns of change as well in the drier, warmer seasons due to evapotranspiration. My findings support this hypothesis at all three temporal scales, indicating that wetlands are dynamic GSI features that may operate differently than their more static gray counterparts. This dynamism of urban wetland GSI is not represented in the present literature, and, from the findings in this chapter, may be critical toward understanding the range of flood mitigation services that wetlands provide, as well as the conditions in which they may provide them.

2.4.1. Effect of antecedent surface and soil conditions on urban flood risk

I found that antecedent surface and soil storage conditions in wetlands, which were controlled by interannual, seasonal, and perhaps daily forces, had significant positive but small effects on systemic flooding during an extreme storm event, in one watershed and aggregated across all watersheds. In three other watersheds, I detected no such significant effects on flooding. Thus, antecedent conditions may impact flood risk but this likely depends on other characteristics of wetlands, such as dimensions, flow path, outlet/inlet placement, and the characteristics of the stormwater management system to which they are connected, such as conduit slope and diameter. I found that estimates of systemic flood volumes were on average greater in my model that did not account for infiltration, and least in models that allowed for infiltration. Studies on the utility of wetlands toward reducing flood volume should then take care to include this infiltration component or risk missing a major contributing mechanism for flood volume reduction. However, many studies do not take infiltration into account (Kumar et al., 2021; Sauer et al., in revision). Finally, I found that systemic flood volume was generally greater in the seasons when wetlands were at their highest stages and when soil moisture content was higher. In none of my models, under any surface and soil water storage condition, did the monitored wetlands flood. Rather, in all cases, downdrain areas in the system were the locations of changes in flood volumes, indicating that wetlands have distal effects on stormwater management systems, and that flooding and flood mitigation in relation to wetland characteristics in urban systems should be measured at scales that capture all downdrain elements.

Our findings on the relation between wetland storage conditions and systemic flooding complicate the almost uniform characterization of wetlands as reducers of flood risk for cities (Arkema et al., 2013; Bullock & Acreman, 2003; Nicholls et al., 1999; Narayan et al., 2017; Rojas et al., 2019; Li et al., 2020; Van Coppenolle & Temmerman, 2019; Van Coppenolle & Temmerman, 2020). I affirm arguments that system-specific understanding is necessary to understand what, if any, flood risk reduction might be gained by the incorporation, management, restoration, or construction of wetlands (Acreman & Holden, 2013; Sauer et al., in review). It may be that under low storage conditions, some wetlands do reduce flood risk in urban areas compared with traditional gray infrastructure, but under high storage conditions the effect will be lessened. In a review of 28 modeling and empirical studies of the effects of wetlands on flow regimes in rivers, Kadykalo and Findlay (2016) theorized that wetlands could even increase the risk of flooding by causing sustained, elevated downstream/downdrain flow compared to other natural or synthetic elements of the landscape that are designed to maximize rates of flow through the system. The authors developed this theory based on the results of one outlier study (Lundin, 1994), which found that the presence of wetlands in a forested system actually increased the frequency and magnitude of flooding. A recent modeling exercise on the flood mitigation effects of a proposed wetland in an urban drainage catchment found that such a wetland could reduce flood risk, with the caveat that the wetland should be paired with an engineered flapgate to stop backflow between two subcatchments during heavy rainfall (Kumar et al., 2021). Care must then be taken to not make sweeping statements about the utility of wetlands, or cities could risk overestimating the flood risk reductions that wetlands provide, or even potentially increase the risk of flooding in their systems. I recommend modeling wetlands within the system context in which they will exist in order to estimate the direction and magnitude of their effects on flooding.

2.4.2. Changes in surface and soil water storage

I found substantial differences in soil water storage between years (campaigns), seasons, and days. Differences in the surface storage of monitored wetlands between years were likely primarily attributable to annual differences in rainfall. In the eight months leading up to the first monitoring campaign, precipitation records from Valdivia's Pichoy Airport were 1016 mm, compared to 890 mm in the eight months leading up to the second monitoring campaign, and both campaigns occurred during years that were dryer than the 30-year average (1760 mm). Differences in surface storage between seasons were likely due to differences in precipitation, given the relationship between wetland stage and rainfall events (Fig. 2.4). Additionally, I found substantial differences in soil-water storage between seasons, with soils tending to have more storage capacity in the dry season than in the wet season, which is in accordance with seasonal patterns of precipitation.

The source, or sources, of daily changes in wetland surface storage were more difficult to parse. Given previous scholarship on baseflow in urban waterways (Fillo & Bhaskar, 2021; Shepherd, Ellis, & Rivett, 2006), I considered that the patterns I saw in my wetlands were driven by urban effluent and groundwater pumping. Human contributions to urban effluent, such as from tap water usage and watering lawns, would normally occur during daylight hours, when humans are most active (Fillo & Bhaskar, 2021). Yet wetland stage consistently dropped during this period. Conversely, pumping might be more active during daylight hours, to provide water for human consumption. However, a 2017 census shapefile from Valdivia revealed the presence of fewer than ten pumps, used by individual households, near the wetlands (Fig. 2.12). Using triangulated irregular network (TIN) shapefiles of the monitored wetlands and calculating wetland volumes under different wetlands stages, I estimate that a drawdown of several centimeters in monitored wetlands would equate to tens of thousands of cubic meters of surface water. Water in Valdivia is typically drawn from riverine sources that are distant and upstream from the wetlands examined in this study; however, under low flow conditions, Valdivia has relied at least partially on groundwater pumping to meet daily water supply needs. Pumping by residents or by the municipality may be the primary cause of change in wetland stage, but from the available information on pumping, I cannot make conclusions in the affirmative or negative.

Evapotranspiration was also a probable cause of daily changes in wetland stage, but I was again unable to make a definitive case for this. First, contrary to what I expected, I found many of the highest mean values for daily change in stage occurred in the late wet season. If evapotranspiration was the primary driver of daily changes in wetland stage, daily decreases in wetland stage should have been greatest in the drier months when insolation lasts longer each day and temperature is generally higher. Furthermore, driven by longer daylight hours during the spring and summer dry seasons, I would have expected the wavelength (the distance between wave crests) of the stage curves to be greater in these seasons than in the fall and winter wet seasons. But I find no significant differences in the timing of wave crests between seasons. In contrast to my study, previous scholarship on a “biological tide,” driven by evapotranspiration, in an experimental treatment wetland in Phoenix, AZ, found strong seasonal patterns of evapotranspiration, closely following seasonal patterns of wetland biomass increase in the warmer months and decrease in the winter months (Bois et al., 2017). From visual inspection in the monitored wetlands, biomass increased during the summer and decreased during the winter, but I see no such pattern of daily wetland stage differences to reflect these seasonal differences in biomass.

Summarily, I observed a dominant sinusoidal diurnal pattern of changes in relative stage that is most consistent with pumping and/or evapotranspiration. While I reject simple drainage as the cause of observed diurnal changes in wetland stage, I nonetheless leave a definitive diagnosis of the cause(s) for the dominant sinusoidal diurnal pattern I observed in the study wetlands to future research.

2.4.3. Holistic modeling of wetland hydrological processes

The difference in flood volume estimates varied more by model than by storage conditions, with the additional surface storage model producing lower flood volume estimates than the base model, and the infiltration model producing by far the lowest flood volume estimates. These results signal the importance of the inclusion and appropriate conceptualization of soil water storage in wetlands. Nonetheless, I expect that models allowing for infiltration likely produced underestimates of flood volumes because they allowed infiltration to proceed at the same rate once pore space was filled. This effectively bottomless infiltration potential is unrealistic, perhaps especially in this model system, due to changes in soil type with depth and the elevation of the water table. Previous scholarship (Tu, Wadzuk, & Traver, 2019) has identified that no urban flood model is equipped to account for all major hydrologic pathways in urban settings, but that models like EPA SWMM may be modified with some effort to simulate critical hydrologic features like changes in soil type, and one could theoretically simulate the water table as well via a soil with an effective infiltration rate of near zero. Of course, even with such modifications, any model will overestimate rates of water removal and conveyance from some features of the model and underestimate rates from others. While no urban hydrological model is perfect, they are nonetheless useful for estimating ranges of expected flooding values under present and future configurations of hydrologic elements in cities (Rosenzweig et al., 2020; Sauer et al., in review). my findings here from testing different models indicate that cities should collect basic soil data (e.g., soil type, seasonal soil moisture content) on wetlands connected to their stormwater management systems and include in their models the possibility of infiltration to some depth or volume, or risk underestimating the flood risk mitigation services of their wetland GSI.

Table 2.1. Monitoring periods during the two monitoring campaigns, between 2017-2018 and 2019-2020.

Wetland	**Start date 2017**	**End date 2018**	**Start date 2019**	**End date 2020**	**Number of days monitored**	**Missing days in recording period**
1	10/16	05/02	09/13	02/11	351	
2	10/16	05/02	09/14	02/11	351	
3	09/29	05/02	09/14	02/11	367	
4	09/29	05/02	09/14	02/11	367	
5	09/28	05/02	09/14	02/11	313	11/28/2017 - 1/17/2018
6	10/16	05/02			199	

Table 2.2. Average values for minimum, mean, and maximum stages, in meters, across seasons and campaigns, and for the combination of data from campaigns 1 and 2. Blank values indicate a lack of data available for analysis.

Campaign 1	**Late wet**			**Early dry**			**Late dry**			**Early wet**		
Wetland	*Minimum*	*Mean*	*Maximum*	*Minimum*	*Mean*	*Maximum*	*Minimum*	*Mean*	*Maximum*	*Minimum*	*Mean*	*Maximum*
1	0.382	0.397	0.412	0.321	0.334	0.348	0.178	0.195	0.213	0.338	0.366	0.388
2	0.340	0.352	0.364	0.311	0.325	0.339	0.220	0.238	0.256	0.452	0.472	0.492
3	0.607	0.633	0.670	0.525	0.545	0.571	0.572	0.591	0.610	0.755	0.778	0.800
4	0.605	0.625	0.647	0.657	0.672	0.689	0.606	0.652	0.699	0.879	0.898	0.916
5	0.386	0.402	0.420	0.374	0.391	0.406	0.536	0.567	0.592	0.583	0.605	0.626
6	0.603	0.621	0.649	0.531	0.546	0.563	0.447	0.463	0.478	0.471	0.493	0.524
Campaign 2	**Late wet**			**Early dry**			**Late dry**			**Early wet**		
Wetland	*Minimum*	*Mean*	*Maximum*	*Minimum*	*Mean*	*Maximum*	*Minimum*	*Mean*	*Maximum*	*Minimum*	*Mean*	*Maximum*
1	0.081	0.098	0.110	0.014	0.025	0.036	0.000	0.000	0.000			
2	0.186	0.204	0.216	0.105	0.117	0.129	0.000	0.000	0.000			
3	0.263	0.284	0.305	0.172	0.188	0.205	0.054	0.072	0.088			
4	0.201	0.223	0.239	0.159	0.175	0.189	0.020	0.031	0.044			
5	0.161	0.184	0.203	0.092	0.110	0.126	0.042	0.061	0.075			
6												
Combined campaigns	**Late wet**			**Early dry**			**Late dry**			**Early wet**		
Wetland	*Minimum*	*Mean*	*Maximum*	*Minimum*	*Mean*	*Maximum*	*Minimum*	*Mean*	*Maximum*	*Minimum*	*Mean*	*Maximum*
1	0.155	0.172	0.184	0.167	0.180	0.192	0.121	0.133	0.145			
2	0.224	0.241	0.253	0.208	0.221	0.234	0.150	0.162	0.175			
3	0.403	0.426	0.454	0.349	0.367	0.388	0.407	0.426	0.444			
4	0.366	0.387	0.405	0.408	0.423	0.439	0.419	0.455	0.491			
5	0.254	0.274	0.293	0.179	0.196	0.212	0.349	0.375	0.396			
6												

Table 2.3. Mean daily changes in stage for all wetlands in all seasons. Data for the late wet season for wetland 6 was not included as data were available for only three days. Mean daily change in stage for the early wet season is derived only from campaign 1, as I did not collect during campaign 2. Days in which precipitation exceeded 2 mm or the minimum daily stage was 0 were excluded from this analysis.

	Mean daily change in stage (m)			
Wetland ID	*Late wet*	*Early dry*	*Late dry*	*Early wet*
1	0.026 ± 0.007	0.028 ± 0.009	0.031 ± 0.020	0.039 ± 0.025
2	0.030 ± 0.008	0.027 ± 0.007	0.033 ± 0.012	0.033 ± 0.017
3	0.047 ± 0.020	0.038 ± 0.017	0.031 ± 0.014	0.036 ± 0.012
4	0.032 ± 0.011	0.030 ± 0.012	0.085 ± 0.081	0.028 ± 0.013
5	0.033 ± 0.017	0.032 ± 0.013	0.043 ± 0.025	0.034 ± 0.019
6		0.030 ± 0.009	0.028 ± 0.006	0.030 ± 0.012

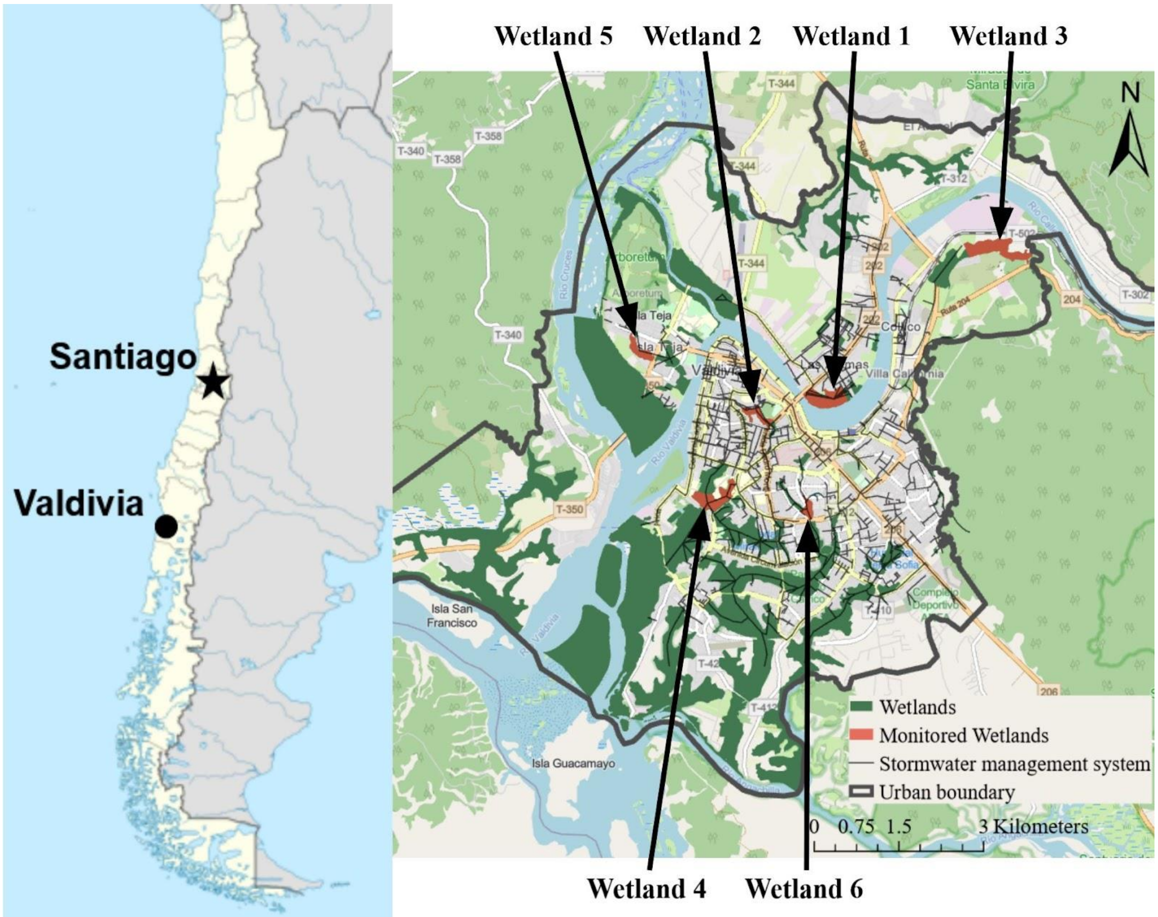

Figure 2.1. Left: Location of study site, Valdivia, in the context of greater Chile (39.8336 S, 73.2154 W; left). Right: Valdivia's main urban area, its wetlands, including the wetlands monitored in this study in red, and its stormwater management system. Valdivia's stormwater management system includes gray elements such as pipes and canals, and also formally includes natural and modified wetlands.

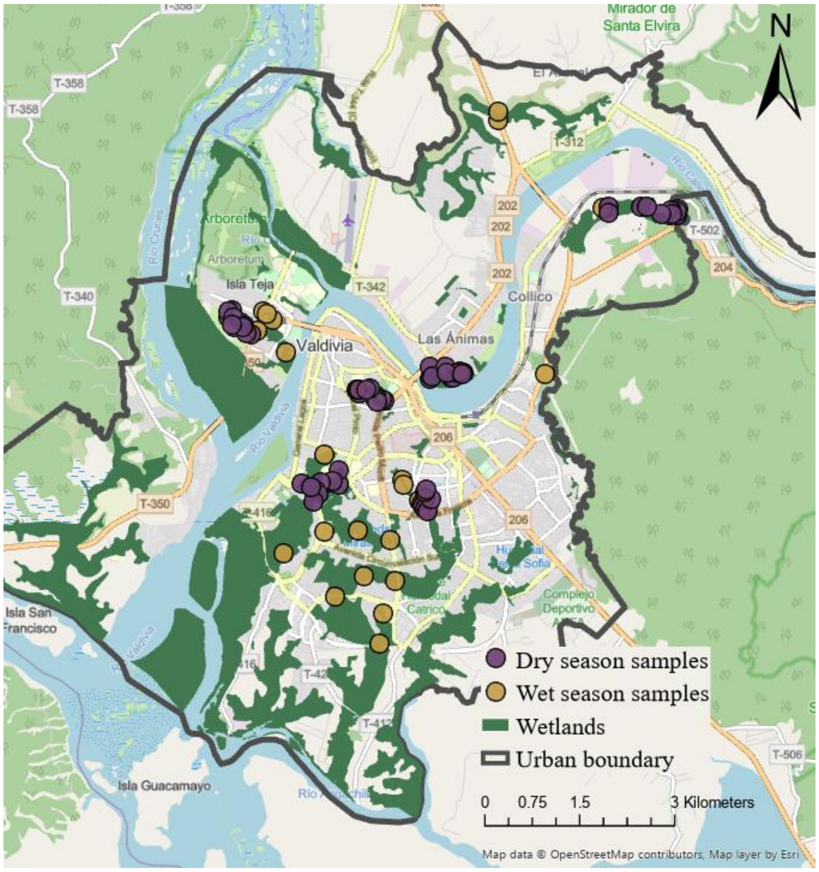

Figure 2.2. Locations where soil samples were collected during dry season and wet season campaigns.

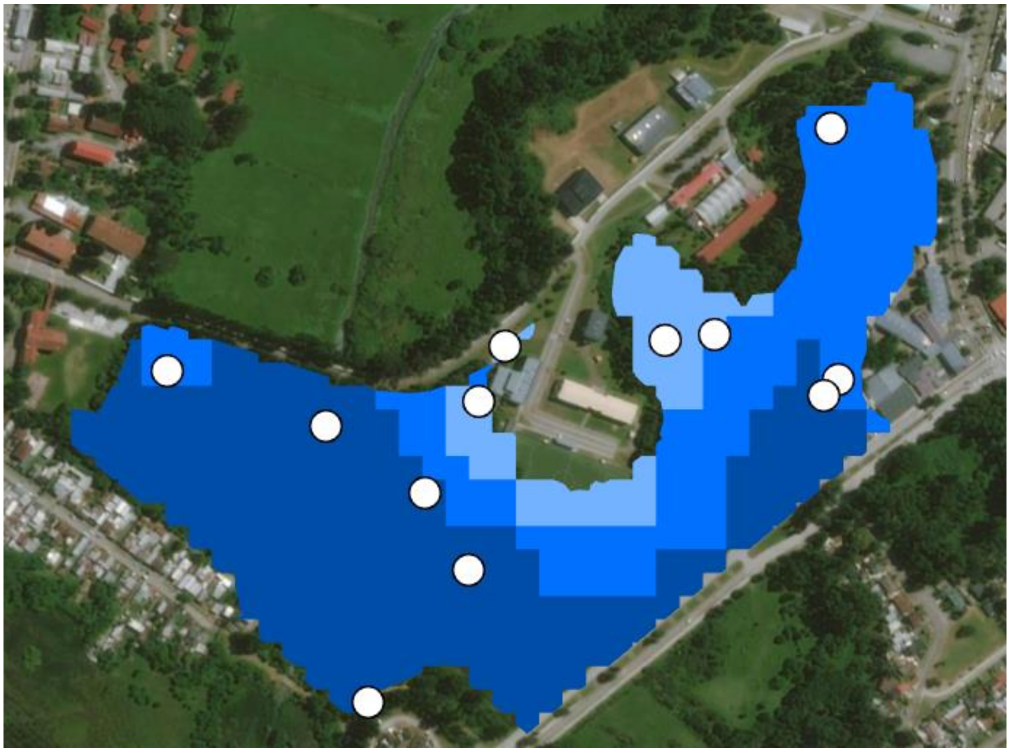

Figure 2.3. Soil sampling locations in a monitored wetland during the dry season. Color gradation from light blue to dark blue indicates increasing wetland depth, from the shallowest third of the wetland to the deepest third of the wetland.

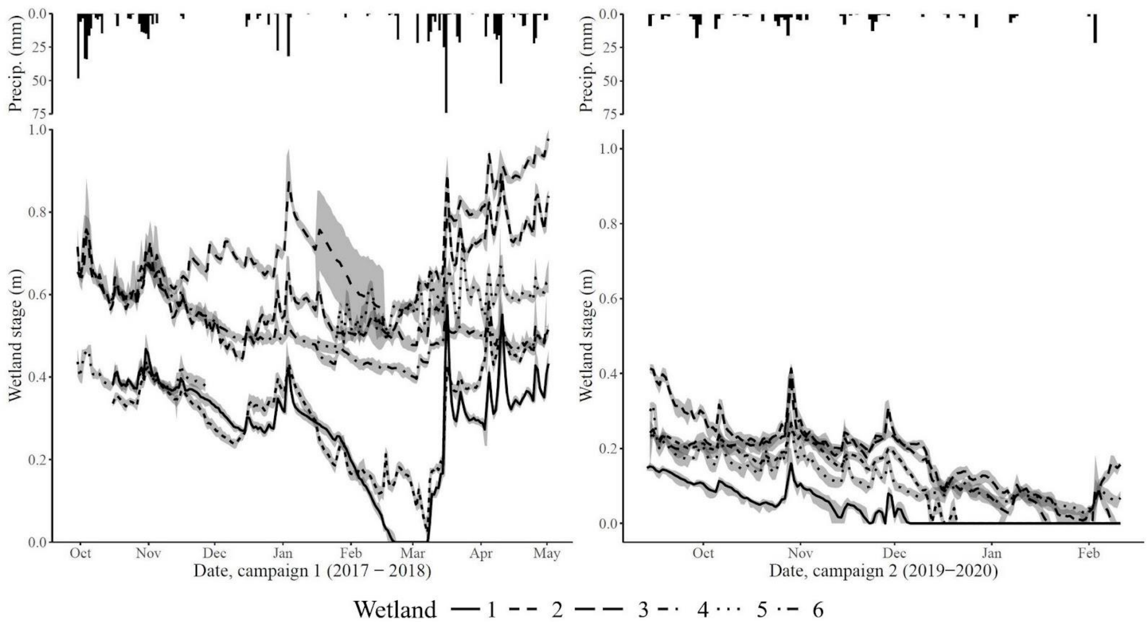

Figure 2.4. Wetland stage and hyetographs in monitored wetlands across the two campaigns. Lines represent daily mean stage, ribbons represent daily maximum and minimum stage, and bars represent daily precipitation. Data collection ended in May of 2018 and February of 2020 in campaigns 1 and 2, respectively.

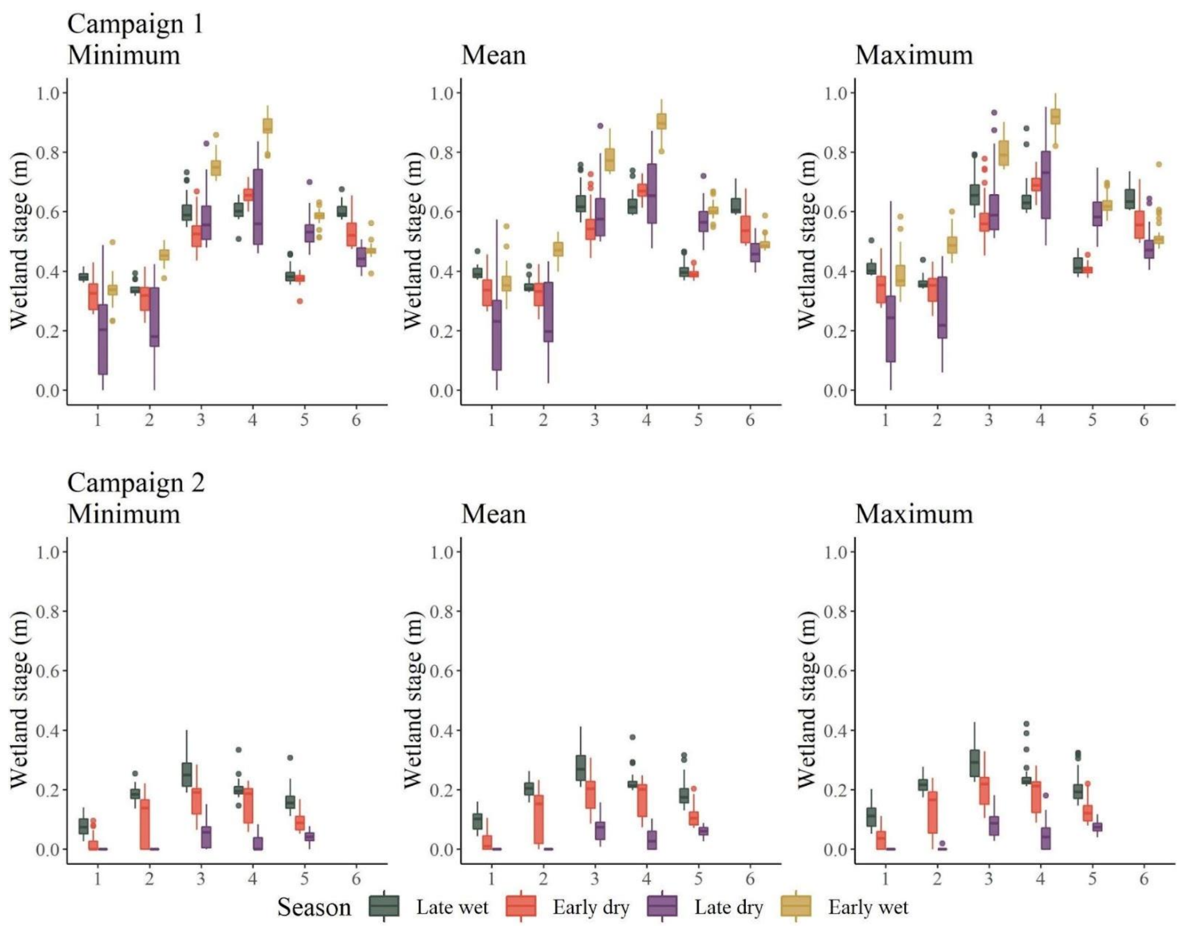

Figure 2.5. Daily minimum, mean, and maximum stages by season by wetland, for (top) campaign 1, and (bottom) campaign 2. Labels on the x axis indicate wetland ID. No data were collected in campaign 2 for the early wet season in any wetland. No data were collected in campaign 2 for any season in wetland 6.

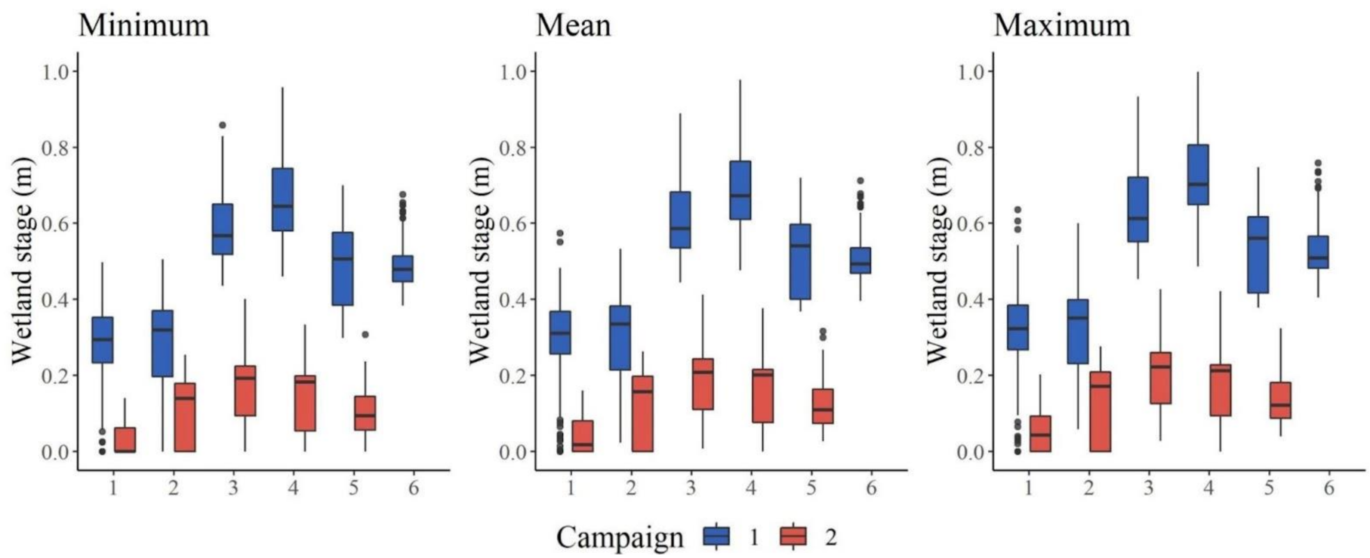

Figure 2.6. Daily minimum, mean, and maximum values by campaign by wetland, where all seasons are combined. Numbers on the x axis indicate wetland ID. No data were collected in campaign 2 for any season for wetland 6.

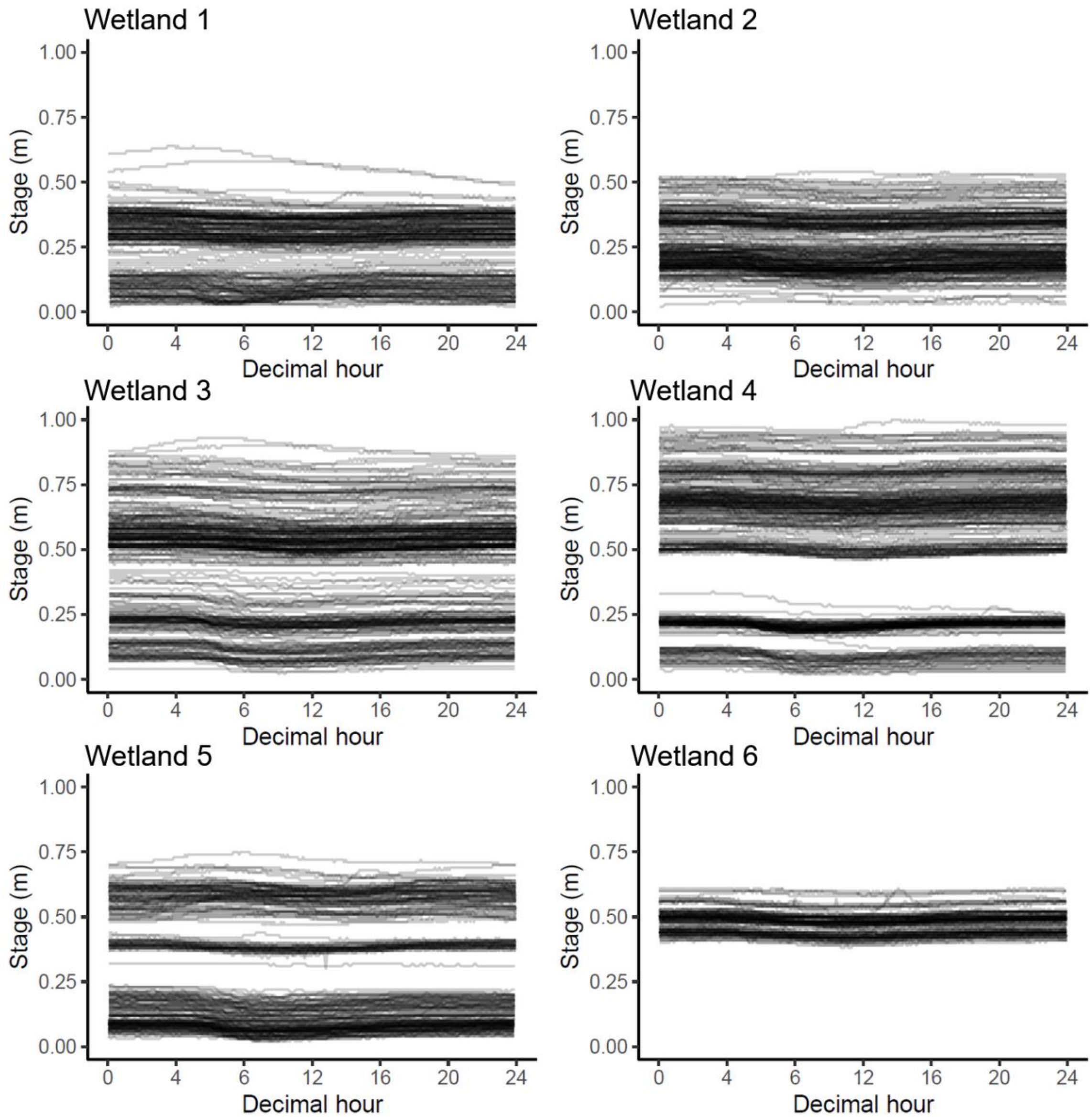

Figure 2.7. Wetland stage versus decimal hour of the day, grouped by day, for both campaigns. Each line represents a single full day of wetland depth measurements. Days in which precipitation in Valdivia exceeded 2 mm or the minimum daily stage was 0 were excluded from this analysis.

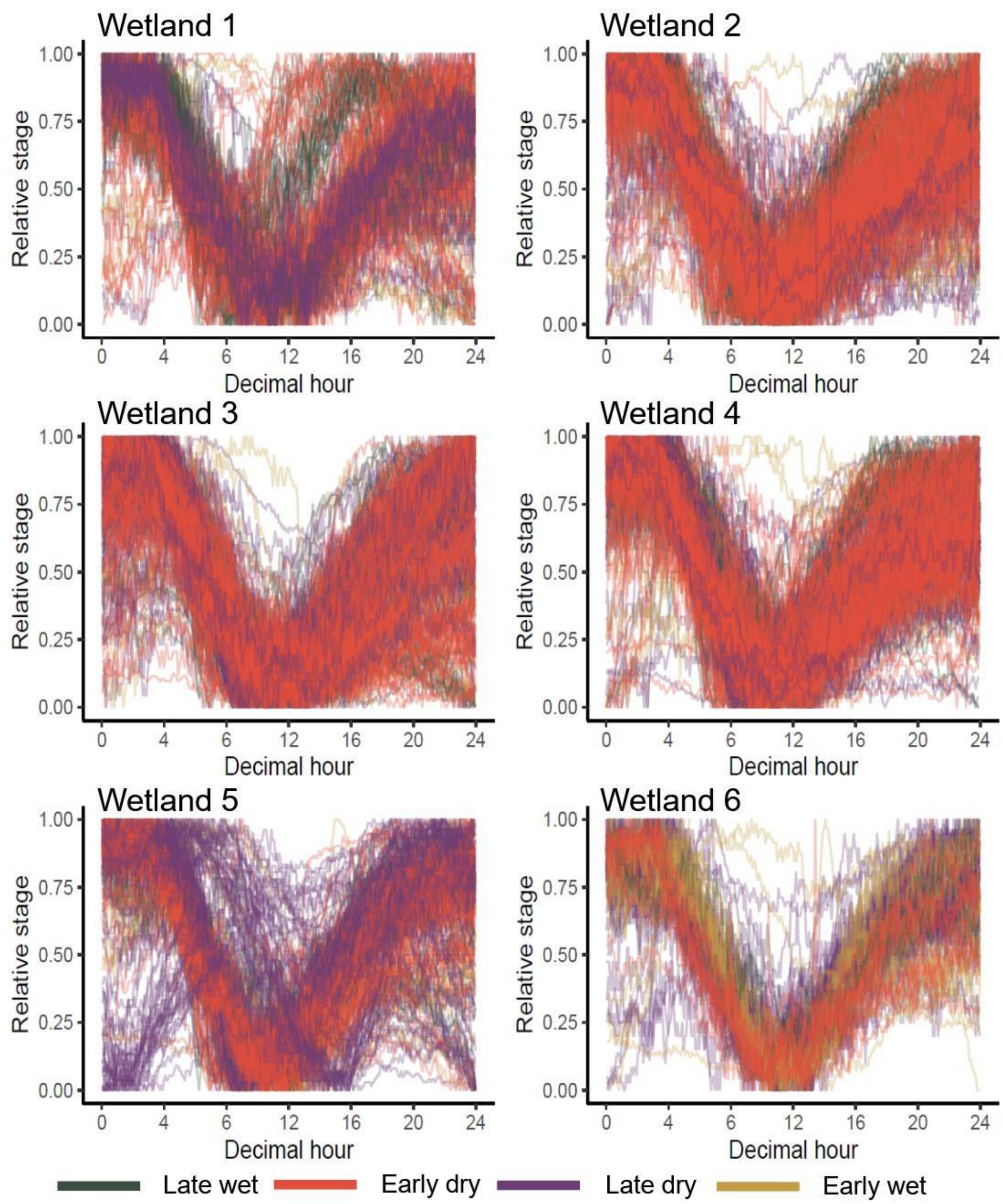

Figure 2.8. Relative wetland stage versus decimal hour of the day, grouped by day, for both campaigns. Raw wetland stage data for a given day were normalized according to that day's maximum and minimum stage values. Maximum relative stage is 1 and minimum daily stage is 0 for all days. Days in which precipitation in Valdivia exceeded 2 mm or the minimum daily stage was 0 were excluded from this analysis.

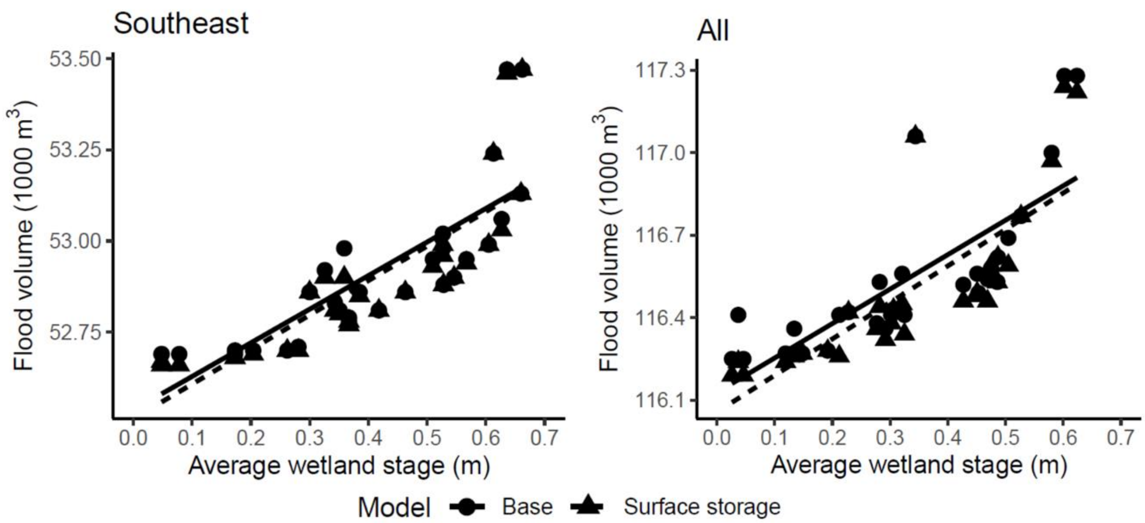

Figure 2.9. Systemic flooding versus average wetland stage in the Southeast sector and across All sectors, for the base model and the model where soil storage was modeled as additional surface storage. Presence of trendline indicates $p < 0.05$, and trends were significant for both models. No other sector showed significant trends.

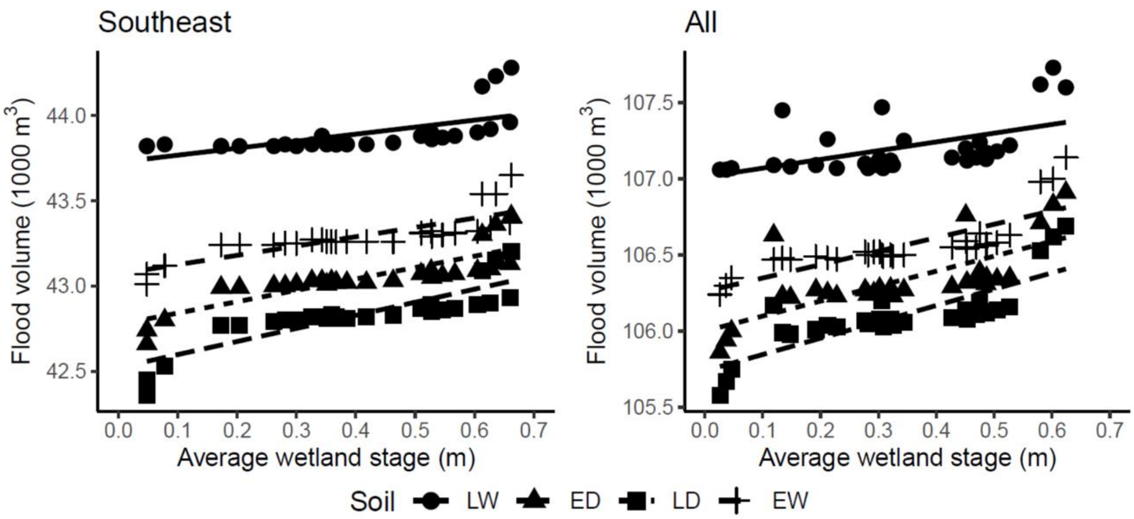

Figure 2.10. Systemic flooding versus average wetland stage in the southeast sector and aggregated across all sectors, for the model with infiltration. Presence of trendline indicates $p < 0.05$, and trends were significant for all soil infiltration rates. No other sector showed significant trends.

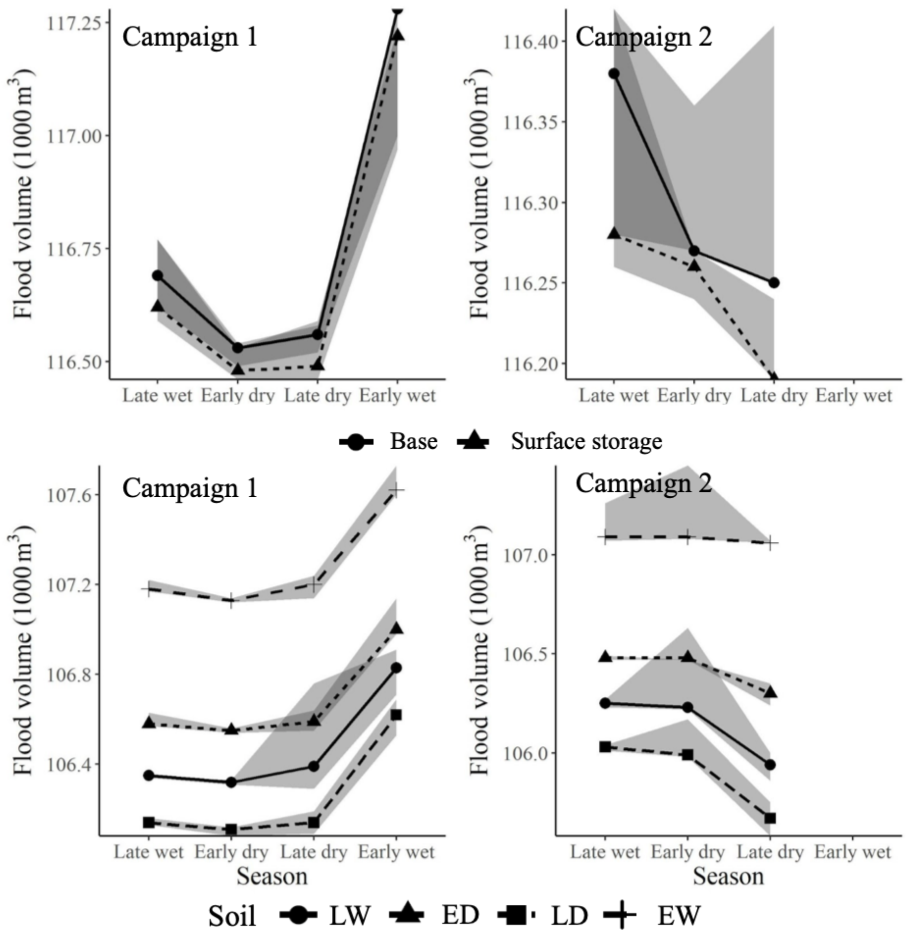

Figure 2.11. Systemic flood volumes for each campaign for (top) base and surface storage models, and (bottom) infiltration models. Gray ribbons represent maximum and minimum flood volumes during maximum and minimum stages in each season.

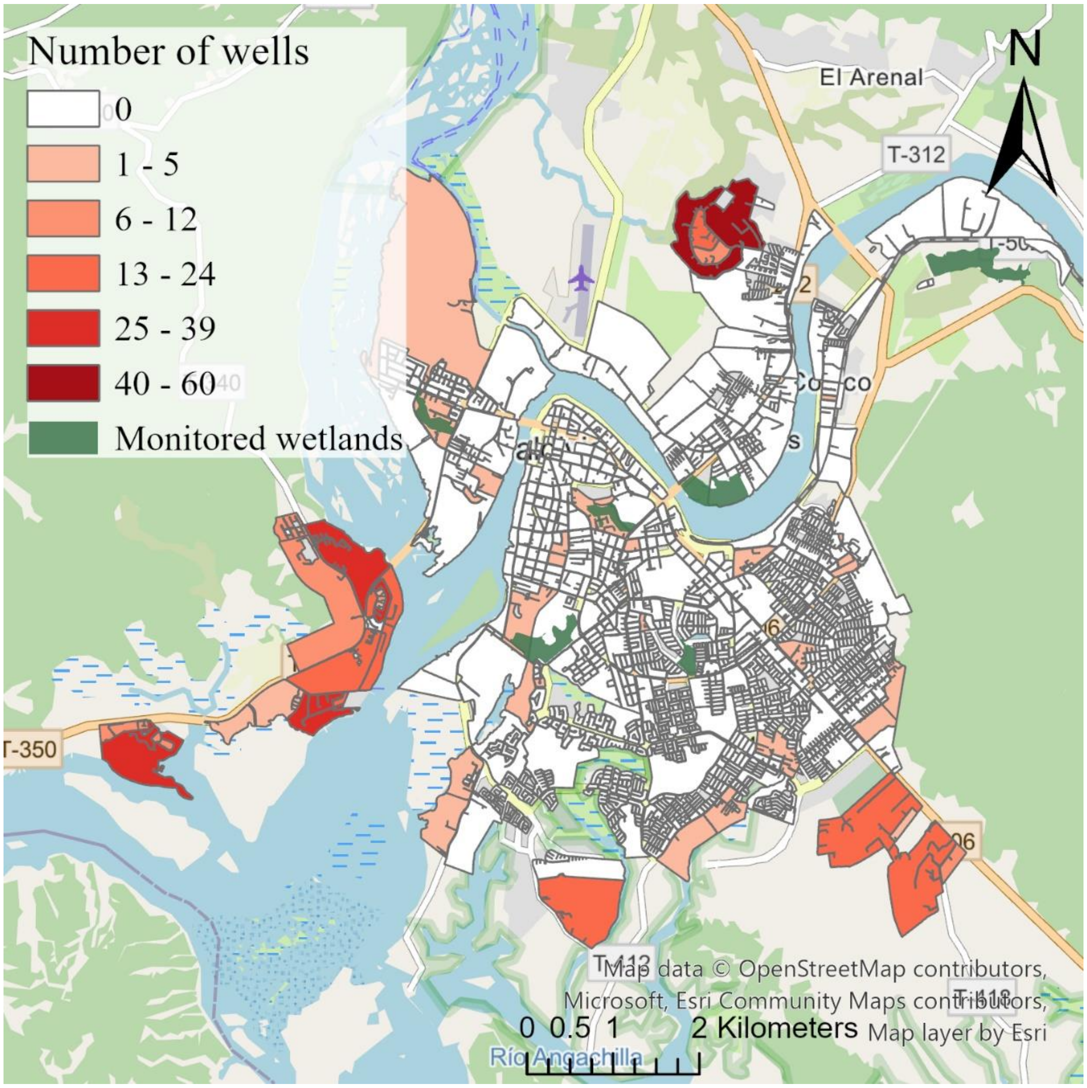

Figure 2.12. Residential well presence in the census blocks of Valdivia and their spatial relationship with wetlands monitored in the study.

CHAPTER 3

IMPACTS OF CHANGING WETLAND COVER ON PLUVIAL FLOOD RISK IN VALIDIVA, CHILE

3.0. Abstract

Globally, the footprint of inland cities is expanding into areas with existing wetland cover, and many cities are planning to restore or construct wetlands in extant urban areas. However, information is lacking about how systemic flood risk in urban stormwater management systems changes with urban wetland areas. I present a case study on changes in urban wetland extent and their impacts on flood risk in an urban drainage system in Valdivia, Chile, where urban wetlands comprise ~22% of municipal area and ~17% of stormwater management system length. I used outputs from stakeholder workshops that determined four plausible scenarios of urban development of Valdivia through 2080, and developed a fifth scenario according to historical trends. I used EPA SWMM and GIS software to model changes to wetland area and volume for each scenario. I examined the effects of changing wetland extent in this system on flood volume, flood duration, and flooding location at the scale of the entire city stormwater management system resulting from a 100-year return period storm event. Such a storm event may become more common by the year 2080 and representative of a more frequent storm event than in the present day. I found that flood volume, the flood duration, and the number of locations that flooded increased with increasing wetland loss. Further, the contribution of wetlands to flood volume and flood duration increased with wetland loss. Finally, downdrain flooding generally increased with updrain wetland loss, indicating an important role of wetland configuration. I discuss the practical implications of inclusion

and stewardship of inland wetlands in urban stormwater management systems and offer suggestions for improving the modeling of wetland hydrological function.

3.1. Introduction

Pluvial flooding is a major concern for urban areas, and can lead to loss of life, damage to property, and disruption of transportation networks (Chang et al., 2010; Douglas et al., 2010; Falconer et al., 2009; Yin et al., 2016). Pluvial flooding is surface ponding or overland flow that occurs when rates of precipitation exceed the capacity of drainage systems and/or surfaces to remove it (Falconer et al., 2009). Pluvial floods threaten many aspects of urban communities, causing loss of life, damage to property, disruption of transportation networks, and displacement (Chang et al., 2010; Douglas et al., 2010; Falconer et al, 2009; Yin et al., 2016).

As a physical phenomenon, pluvial flooding results from interactions between rates of precipitation, urban stormwater management practices, and biophysical characteristics of the urban and peri-urban landscape (Westra et al., 2014). In many cities, one or all three of these interacting factors are changing in ways that exacerbate pluvial flood risk—both in terms of chance of occurrence and potential damages. Subdaily extreme rainfall events have become more frequent and intense due to anthropogenic climate change (Westra et al., 2014; Wuebbels et al., 2014). Yet cities have prioritized mitigating the risks of fluvial and coastal flooding over pluvial flooding (Guerreiro et al., 2017). Pluvial flooding has in recent years captured attention of researchers and planners (Rosenzweig et al., 2018; Goodrich et al., 2020), but my understanding of how to mitigate its causes and effects in urban areas needs improvement.

The construction, use, or conservation of wetlands have all been suggested as measures to mitigate the risk of various forms of flooding in many different ecosystem types. The ability of coastal wetlands to reduce coastal flooding has been explored in depth and in a diverse array of ecosystems (Arkema et al., 2013; Nicholls et al., 1999; Narayan et al., 2017; Rojas et al., 2019; Van Coppenolle & Temmerman, 2019; Van Coppenolle & Temmerman, 2020). The effects of wetland presence on riverine flooding have received notable attention as well. Neri-Flores et al. (2019) modeled the effects of wetland preservation on reducing riverine flooding caused by hurricane storm surges. Pomeroy et al. (2014) modeled how preserved inland wetlands can reduce riverine flooding driven by snowmelt. Yang et al. (2010) modeled how the restoration of wetlands in a Canadian prairie watershed can reduce peak river discharge and flooding. In a review of 28 modeling and empirical studies of the effects of wetlands on flow regimes in rivers, Kadykalo and Findlay (2016) found that wetlands generally reduced the frequency and magnitude of flooding, with one notable exception in a forest wetland system (Lundin, 1994). Historically, declarations about the flood mitigation services of wetlands have their bases in studies in wetlands of these types and in these ecosystems.

Only recently has research explored the abilities of inland urban wetlands to reduce urban pluvial flood risk, or how the incorporation of wetlands in an urban stormwater management system might alter the system's performance. The theory and practice of inland wetland restoration and construction in urban areas to reduce pluvial flood risk is relatively new in academia and among stormwater managers (Chan et al., 2018; Elmqvist et al., 2015), and modeling and empirical studies of the effects of wetland restoration and construction in urban areas are rare. Some cities have added inland wetlands to their portfolios of urban ecological infrastructure (UEI; Childers et al. 2019; otherwise known as green infrastructure (GI), green stormwater infrastructure (GSI), or, more broadly, nature-based solutions (NBS)) or suggested that the construction, restoration, or incorporation of inland wetlands be included in sustainable urban drainage systems (SuDS) or low-impact development strategies (LiDS) to reduce pluvial flooding (Chan et al., 2018; Fletcher et al., 2015; Li et al., 2020).

Wetlands may provide water management services to cities through a variety of hydrologic mechanisms, though cities often only build or manage for one or two mechanisms. Depending on wetland morphology, wetland vegetation, environmental conditions, soil characteristics, water-table depth, and the drainage systems to which wetlands may be connected, wetlands may manage stormwater via some combination of impoundment (the temporary storage of water), infiltration (the removal of surface water via percolation into wetland soils), evapotranspiration (the removal of surface and soil water from the system via evaporation or plant-mediated transpiration), and conveyance (the movement of water through and out of the drainage system via passive flow; Bullock & Acreman, 2003). For many cities considering the use of wetland UEI, the key hydrologic functions of wetlands are those of impoundment and infiltration (Li et al., 2020). Previous scholarship has demonstrated that impoundment of stormwater in wetlands delays or reduces stormwater release to downstream waterways (Kadykalo & Findlay, 2016). Infiltration is of interest to cities correcting course from the proliferation of impervious surfaces, as impervious surfaces increase the relative amount of precipitation that converts to runoff relative to pervious surfaces (Fletcher et al., 2013). This elevated conversion of precipitation to runoff in turn increases peak flows in drainage systems and overwhelms their capacities to remove stormwater from the city surface (Ogden et al., 2011).

Critically absent from studies on the flood mitigation services of wetlands are city-wide studies on whether and how urban stormwater management system performance changes when urban wetlands are constructed, restored, or incorporated. Furthermore, in drainage systems that already include wetland features, there are no city-wide studies on the effects of changes in wetland extent or configuration on drainage system performance. Researchers have previously argued that, at least outside the urban context, system-specific knowledge is necessary to accurately estimate effects of various wetland characteristics on the flood regulation services that wetlands may provide (Acreman & Holden, 2013; Kadykalo & Findlay, 2016). Wetland dimensions, extent, antecedent storage conditions, rates of infiltration and evapotranspiration, and configuration within a stormwater management system are all likely to influence the performance of any stormwater management systems into which urban wetlands may be included.

To detail the links between inland urban wetland dimensions and configuration and stormwater management system performance, I modeled the individual and coupled effects of inland wetland loss and urban watershed expansion on the performance of an urban stormwater management system. For my study system, I asked the following question: How does loss of wetland UEI in an urban stormwater management system alter flood volume, flood duration, and the locations of flooding within the system? I hypothesized that flood volume, flood duration, and the number of locations flooded in the urban stormwater management system would increase with increasing wetland UEI loss. Further, I hypothesized that gray components of the drainage system would experience increasing flood volume and flood duration as wetland UEI loss increased.

Additionally, I asked: in urban drainage systems, to what degree are changes in flood volume, flood duration, and locations flooded caused by wetland loss or watershed area gain resulting from urban growth? I hypothesized that I would see greater increases in flood volume, flood duration, and flood locations in drainage systems from the loss of wetland UEI rather than the gain of impervious watershed area, but that wetland loss combined with watershed gain would result in the greatest increase in these three aspects of flooding.

2. Materials and Methods

2.1. Study site

Valdivia, Chile (area: 93.94 km^2) is a city of approximately 170,000 people in the southern half of Chile, 850 km south of the capital Santiago, in the Región de los Ríos (Fig. 3.1). Citizens and city managers of Valdivia must contend with a high risk of fluvial and pluvial flooding owing to high annual precipitation (1780 mm), a long wet season, and the city's location 12 km inland from the Pacific Ocean, at the confluence of three major rivers. Its ecosystem is classified as a temperate rainforest (Hajek & Ramirez, 1975; Amigo & Ramirez, 1998). Wetlands are a characteristic feature of Valdivia, covering 20.64 km^2 (22.7%) of the municipal area.

Valdivia's stormwater management system is composed primarily of gray infrastructure components (e.g., pipes and canals) and wetlands. As of 2012, Valdivia's stormwater management system consists of roughly 245.7 km of drainage infrastructure, of which 41.19 km (16.8%) is wetland UEI. The origin of most of this wetland cover is a 1960 earthquake of magnitude 9.5, which caused up to 20 m of uplift in some areas (Barrientos & Ward, 2007) and subsidence and rifting in others. Since 1960, the city has deliberately incorporated these wetlands into its stormwater management system (CMOP, 2012). In addition, the continued presence of wetlands in the city is owed in part to local conservation movements to maintain the cultural services of wetlands (Correa et al., 2018) and their function as habitat to charismatic species tied to Valdivian identity (Silva et al., 2015).

The flow of stormwater through gray infrastructure components and wetlands in Valdivia is variable and complex. For example, stormwater may first enter the system as runoff via gray infrastructure components, flow to a wetland, and then to a river outlet. Alternatively, stormwater may first enter the system as runoff via wetlands, which may then flow into gray infrastructure components, which then may flow to river outlets. In a more complex scenario, stormwater may first enter the system via a wetland, then flow through gray infrastructure components for some distance, pass through a second, downdrain wetland, and flow through several other gray infrastructure components and wetlands before reaching a river outlet. The stormwater management system of Valdivia thus depends heavily on the coupled performance of gray infrastructure components and wetlands, and I would expect to see changes in either element affecting the performance of the other.

3.2.2. General approach

I employed mixed methods over four phases of work (Fig. 3.2). Phase 1 involved the convening of an in-person workshop in Valdivia, Chile to co-develop with practitioners the goals and objectives of four different scenarios of development for the city to achieve by the year 2080. Phase 2 involved combining historical data on land-cover change in Valdivia and scenario goals and objectives into rules governing land-cover change in the Dinamica EGO cellular automata-based model (Soares-Filho et al., 2001, 2002). The outputs of this model were land cover maps for the four scenarios developed in the workshop in phase 1, along with an additional "business-as-usual" scenario estimating land cover change in the absence of interventions to the status quo. Phase 3 involved using ArcGIS Pro (ESRI) to estimate changes in wetland land cover and subcatchment area as a result of the changes in land cover in the five outputs from phase 2. Phase 4 involved including the changes to wetland and subcatchment area for each scenario from phase 3 in the Environmental Protection Agency's Stormwater Management Model (EPA SWMM; U.S. EPA) and modeling the changes in flood characteristics for Valdivia's stormwater management system.

3.2.3. Phase 1: UREx SRN scenarios workshop in Valdivia envisioning land cover change

In May of 2017, the Urban Resilience to Extremes (UREx) Sustainability Research Network (SRN) hosted a workshop in Valdivia, Chile, to envision a series of long-term (2080) future scenarios and desirable future pathways of urban development. Participants in the workshop represented a diverse array of Valdivia's stakeholders, such as municipal and regional government employees, university professors, students, and members of community action groups. Participants collaborated to develop a suite of visions and strategies to undertake in order to achieve four unique, plausible scenarios for a future Valdivia: an Inclusive City, a Friendly City, an Eco-Wetland City, and a Flood Resilient City. The scenario themes emerged from the concerns of the citizens of Valdivia and an analysis of Valdivia's governance documents, as well as a publication from the Inter-American Development Bank (IDB, 2015). The visioning and scenario development process in the workshop followed methods described by Iwaniec et al. (2020).

3.2.4. Phase 2: Modeling land cover changes using cellular automata-based model

The qualitative strategies of four scenarios—Inclusive, Flood Resilient, Friendly, and Eco-Wetland—developed in Valdivia's workshops were translated by the UREx SRN modeling team into quantitative rules, spatial and temporal, and introduced into cellular automata-based models of land-use/land cover (LULC). This phase represents an iterative process in which the modeling team gathered feedback from various stakeholders on the four co-produced scenarios, adjusted the quantitative rules based on that feedback, and released updated simulations. Paired with historical information on LULC changes (observed 1983 and 2010 LULC maps) in Valdivia, the cellular automata model generated predictions of LULC configuration in Valdivia in 2080 for each scenario, as well as for a 'Business-as-usual' (BAU) scenario, which assumes LULC proceeded entirely according to historical patterns of development.

3.2.5.1. Phase 3: Estimating changes to wetland area and volume

To simulate the effects of changes in wetland cover based on the cellular automata-based model outputs, I translated changes in wetland area to changes in wetland volume. First, I considered any wetland area in the present day that converts to either low-density urban or high-density urban area in the scenarios to be lost, because such changes necessitate the in-filling and elevating of the area's surface, thus reducing the capacity of the converted area to store water. In contrast, conversion of wetland cover to either pasture/green or forest land cover types would not necessarily be associated with in-filling and reduction of a wetland's overall area or volume. I overlaid the present-day land cover layer with each of the scenario land cover layers in ArcGIS Pro (ESRI), and subtracted the portion of the present-day wetland that was converted to low- or high-density urban land cover. I then identified a subset of wetlands used in the city's official SWMM, and calculated the change in wetland area in each scenario by comparing the wetland area in this subset in the present day with the wetland area in this subset in each scenario. It should be noted that with this methodology that it is possible for change in wetland area in the subset of SWMM wetlands to differ from change in wetland area for the whole city.

To determine changes in wetland volume that resulted from changes in wetland area, I first converted a 2019 contour map (1 m vertical resolution) of Valdivia into a triangulated irregular network (TIN), which represented the three-dimensional topography of the landscape. Then, for each of the wetlands, and for each scenario and the present day, I used the wetland boundaries exhibiting lost wetland area (delineated in the previous step) to generate pseudo-surfaces every 0.25 m from the base of each wetland to the lowest bank of each wetland and calculated the volume of the TIN underneath the surface using the Surface Volume tool in ArcGIS Pro.

3.2.5.2. Phase 3: Translating changes in wetland area, wetland volume, and subcatchment area to inputs to the EPA SWMM

The 28 wetlands included in my base SWMM were modeled as one of two hydrological features: storage units or conduits. Wetlands with relatively small areal extents were generally modeled as storage units, through which stormwater moves in and out instantaneously, according to inlet and outlet heights and slope, and according to the morphology of the wetland. The calculated volume of these wetlands was equivalent to that of the storage unit alone. Wetlands of relatively large areal extents were modeled as one conduit or a series of conduits linked by nodes. Stormwater moves into and out of conduit wetlands similarly to how it would through an irregularly shaped conduit, which requires time and is affected by factors such as slope, roughness, and morphology. The calculated volume of these wetlands was divided proportionally among all its constituent conduits. Thus, in a given conduit wetland, longer conduit segments were attributed greater volumes than were shorter conduit segments, while the sum of the volumes of all conduit segments in the wetland equaled the calculated volume of the entire wetland.

In this part of phase 3, I made a key decision about how to represent cellular automata-based model outputs depicting changes in wetland LULC in EPA SWMM. According to my assumptions about land cover conversion, subcatchments connected to wetlands increase in size as wetlands are lost to low- and high-intensity urban land cover. As such, I decided that subcatchment area in the SWMM would increase by the exact amount of area that wetlands lost in each of my scenarios. In the case that multiple subcatchments were directly connected to a wetland, the amount of area that the wetland lost was distributed proportionally among the subcatchments, according to the relative area of a given subcatchment, such that the sum of the area added to all subcatchments equaled the total area of wetland that was lost to development.

3.2.6.1. Phase 4: Hydrologic modeling in the EPA SWMM

Converting calculated changes in wetland cover in Valdivia to changes in the stormwater management model for each scenario required me to make many decisions on wetland representation and various model assumptions informed in part by the limitations of my data in phase 3. I moved between phases 3 and 4 several times in order to obtain tenable model representation of SWMM structure and function.

3.2.6.2. Phase 4: Characterizing the base stormwater management model and design storm for Valdivia, Chile

The Chilean Ministry of Public Works (CMOP), which is tasked with managing flooding in Valdivia and other cities across Chile, commissioned in 2002 the development of a stormwater management model using EPA SWMM. This model was designed to represent the function of the gray infrastructure (e.g., pipes and canals), as well as the wetlands to which these gray components are connected, given precipitation inputs designed to simulate storms of varying return periods (e.g., 2-year, 10-year, 100-year) and durations (e.g., 6-hour, 8-hour, 24-hour). This model was updated by the CMOP in 2012 to represent a more current configuration of the system, to include a city-wide assessment of subcatchment areas via on-the-ground observation of surface flow paths. The model also uses the dynamic wave option to determine flow routing, which allows for closed conduits (pipes) to become pressurized once they fill with water and, as a result, to flow at more elevated rates than would be otherwise possible by conduit dimensions and roughness.

I used a design storm simulating a rainfall event with a 100-year return period that occurs over the span of six hours, which is the typical duration for an extreme rainfall event in Valdivia. I selected a 100-year return period storm for my analysis in order to simulate particularly hazardous storms in the present-day configuration of the city, that may or may not increase in frequency by the year 2080 (IPCC, 2021). In this design storm, precipitation intensity proceeds in a gaussian manner, increasing until its third hour, and decreasing thereafter. The simulation runs for a total of 36 hours, though rainfall occurs only within the first six. In contrast to the design storm used in my models, the CMOP uses a 2-year return period storm that occurs over the span of six hours in order to determine areas of the city in need of flood resilience interventions. I used the 100-year return period event instead because the intention of this chapter was an exploration of how well the system performs under more extreme precipitation conditions, which may become more common in Valdivia's future given projections of climate change.

3.2.6.3. Phase 4: Modeling initial surface water depth, soil infiltration, and evapotranspiration in wetlands

The base SWMM does not include information on surface water depths for any of its 28 wetlands, and it simulates wetland response to rainfall events based on the assumption that they start with a surface water depth of 0 m. I left unaltered the base SWMM assumption for lack of representative surface water data for the wetlands. I also left unaltered the base SWMM assumption of a water table that is level with or higher than the bottom of the modeled wetlands, which prevents infiltration. Models do not account for rate of water removal due to evapotranspiration, or for other processes that would impede precipitation from reaching wetlands, such as through interception.

3.2.6.4 Phase 4: Detecting flooding in the stormwater management model

The base SWMM considers any volume of water that rises above the maximum elevation of a storage unit wetland or a node in a conduit wetland to be flood water (Fig. 3.3). The base SWMM tallies flood volumes at the location and time that they are generated, and then removes this stormwater from the system at the location where it occurs. Thus, if flooding occurs in an updrain node, that volume of flood water does not move via surface flow or downdrain conduits to downdrain nodes. In the modified SWMMs that simulate whole-system flooding, I allow flooding to be recorded this way. Additionally, in two SWMMs of two subsystems of the city-wide system, I reconstructed features to instead allow for ponding.

3.2.6.5. Phase 4: Isolating the effects of wetland loss, watershed gain, and combined wetland loss and watershed gain

Wetland loss in the cellular automata-based model outputs represented a combination of two phenomena: a loss of wetland area and a gain in contributing watershed area. That is, during typical urban development processes, wetlands are filled in (thereby reducing wetland storage volumes) and replaced with low- and high-density urban land cover (thereby increasing the area of the remaining wetland's subcatchment). I wanted to examine how these two phenomena independently affect flooding and compare the results to their combined effect. Toward this end, I created three separate SWMMs for each scenario: one SWMM characterized by only losses to wetland area, one SWMM characterized by only gains to contributing watershed area, and one SWMM characterized by both losses to wetland area and gains to contributing watershed area; hereafter, WETLOSS, WSHEDGAIN, and LOSS+GAIN, respectively.

3.2.6.6. Phase 4: Exploring two characterizations of model flooding

I applied two methods for tabulating flooding volume and location, depending on whether I were examining systemic flooding or the effects of updrain wetland loss on downdrain flooding in a subsystem. In models of the entire drainage system, modeled flood volume was likely underestimated, because EPA SWMM removes stormwater from the system once it rises above the banks of a wetland or a node. Once EPA SWMM removes stormwater from the system at the points of flooding like this, this stormwater no longer travels through the system, and thus downdrain nodes that may in the real world eventually receive that water instead receive a reduced load. There is no way to toggle off this mechanism of removal, and modelers who want to keep this flood water in the system would have to construct additional elements in EPA SWMM to simulate any surface ponding and surface routing that might occur in the real world—though I have seen no papers that have attempted to do so.

In order to more accurately model the effects of updrain wetland loss on downdrain flooding, I modified models in two subsystems such that flood water was no longer removed at the flood source. My modifications allowed stormwater to pond above wetlands and nodes, with an area equivalent to the subcatchments to which they were attached, to a depth of several meters. Additionally, use of dynamic wave routing caused any ponded water to generate pressure within the closed, connected components of the drainage system. In the model, such pressurization will increase flow rates and potentially alter flood timing and rate of stormwater flux, as pressurized water moves more quickly between nodes linking these closed components. Thus, modifications in these subsystems more accurately account for stormwater volume by not removing it from the system when it is considered flood water, and they allow for more accurate representation the timing of stormwater movement through the system by allowing it to become pressurized where ponding occurs and flow more quickly.

3.3. Results

3.3.1. Wetland land cover characteristics of the five modeled scenarios

The cellular automata-based model yielded land cover maps of the five scenarios—Inclusive, Flood Resilient, Friendly, Eco-wetland, and BAU—that exhibited a gradient of change in city-wide wetland area compared to the present day: -33.9%,

+16.0%, -30.4%, 27.4%, and -49.0% of wetland area, respectively. Within the subset of wetlands used in the SWMM, all scenarios exhibited loss in wetland area compared to the present day: 9.72%, 13.3%, 18.3%, 24.0%, and 37.3% loss in wetland area in the Inclusive, Flood Resilient, Friendly, Eco-wetland, and BAU scenarios, respectively (Fig. 3.4).

3.3.2. Flooding

Systemic flooding generally increased with increasing wetland loss (circle markers, dashed trendline, $R^2 = 0.77$, $p < 0.05$), watershed gain ($R^2 = 0.90$, $p < 0.05$), and both wetland loss and watershed gain (triangle markers, solid trendline, $R^2 = 0.82$, $p < 0.05$; Fig. 3.5a). In the LOSS+GAIN model for the case of 37.3% wetland loss (the BAU scenario), flood volume increased by 131% compared to the Present Day.

The time that nodes were flooding increased in WETLOSS ($R^2 = 0.73$, $p < 0.05$) and LOSS+GAIN models ($R^2 = 0.73$, $p < 0.05$). The WSHEDGAIN model ($R^2 = 0.51$, $p = 0.07$ 5; Fig. 3.5b) was not significant. In the LOSS+GAIN model, for the case of 37.3% wetland loss (the BAU scenario), the time that nodes flooded increased by 118% compared to the Present Day.

The number of flooded nodes increased in WETLOSS ($R^2 = 0.68$, $p < 0.05$) and LOSS+GAIN ($R^2 = 0.68$, $p < 0.05$), but not WSHEDGAIN models ($R^2 = -0.16$, $p \geq 0.05$; Fig. 3.6a). In LOSS+GAIN models, the scenario featuring the greatest amount of wetland loss (BAU scenario, 37.3%) exhibited the following differences and similarities in flooding characteristics compared to the scenario featuring the least amount of wetland loss (Present Day, 0.00%): nine additional locations experienced flooding; 28 locations flooded in both scenarios, but had greater flood volume with greater wetland loss; and 49 locations had equal flooding regardless of wetland loss (Fig. 3.6f). In any scenario, only one location exhibited reduced flood volume and time spent flooding compared to the present day (Fig. 3.6f).

Model estimates for all scenarios indicated that the majority of flooding locations and the greatest flood volume occurs in parts of the drainage system that are not wetlands. However, wetlands generally did produce greater flood volumes as wetland loss increased in WETLOSS ($R^2 = 0.76$, $p < 0.05$), WSHEDGAIN ($R^2 = 0.90$, $p < 0.05$), and LOSS+GAIN models ($R^2 = 0.82$, $p < 0.05$; Fig. 3.6b). Further, the proportion of flooding that occurred at wetland nodes compared to non-wetland nodes generally increased as wetland loss increased in WETLOSS ($R^2 = 0.76$, $p < 0.05$) and LOSS+GAIN models ($R^2 = 0.85$, $p < 0.05$), but not in WSHEDGAIN models ($R^2 = 0.04$, $p \geq 0.05$; Fig. 3.6c). In the LOSS+GAIN model in the Present Day scenario, only 5 of 69 locations that experience flooding occurred within wetlands, and accounted for 38.8% of total system flooding. In the BAU model, 10 of 78 locations that experience flooding occur within wetlands, and account for 43.1% of total system flooding.

Model estimates for all scenarios indicated that the majority of the time the system spent flooding occurred in parts of the system that are not wetlands; however, wetlands generally did spend more time flooding with increasing wetland loss in WETLOSS ($R^2 = 0.83$, $p < 0.05$), LOSS+GAIN ($R^2 = 0.83$, $p < 0.05$), but not WSHEDGAIN models ($R^2 = 0.30$, $p \geq 0.05$; Fig. 3.6d). Further, the proportion of time that wetland nodes flooded compared to non-wetland nodes generally increased as wetland loss increased in WETLOSS ($R^2 = 0.69$, $p < 0.05$) and LOSS+GAIN models (R^2

= 0.68, $p < 0.05$), but not in WSHEDGAIN models ($R^2 = -0.16$, $p \geq 0.05$; Fig. 3.6e). In the LOSS+GAIN model in the Present Day scenario, wetland nodes contributed 5.84% of the total time the system was flooded, while in the BAU scenario their contribution more than doubled, to 13.4% of the total time the system was flooded.

3.3.3. Updrain wetland loss and its effect on flooding in downstream nodes

I modeled the effects of updrain wetland loss on downdrain flooding characteristics in two subsystems of my SWMM. The relevant components of subsystem 1 (Fig. 3.7a) consist of one wetland, modeled as a storage unit; its contributing watershed; three downdrain nodes that experience flooding; and a park where some portion of stormwater may flow between downdrain nodes 2 and 3 once a threshold stage is reached at downdrain node 2, which is below the stage at which downdrain node 2 floods. Proportional wetland loss for the wetland in subsystem 1 was greater than proportional wetland loss of wetlands in the entire SWMM, ranging from 29.2% in the Friendly scenario to 96.5% in the BAU scenario (Table 1).

In subsystem 1, flood volume increased with WETLOSS ($R^2 = 0.53$, $p < 0.05$) and LOSS+GAIN ($R^2 = 0.89$, $p < 0.05$), but not with WSHEDGAIN, where no change in flood volume occurred for any amount of watershed gain (Fig. 3.7b). Flood volume increased in downdrain node 1 with WSHEDGAIN ($R^2 = 0.90$, $p < 0.05$) and LOSS+GAIN ($R^2 = 0.90$, $p < 0.05$; Fig 7c), but not with updrain wetland loss. Flood volume increased in downdrain node 2 with LOSS+GAIN ($R^2 = 0.91$, $p < 0.05$), but not with WETLOSS ($R^2 = 0.71$, $p \geq 0.05$) or WSHEDGAIN ($R^2 = -0.11$, $p \geq 0.05$; Fig. 3.7d). Flood volume in downdrain node 3 did not show any significant trends for any of the three model characteristics ($p > 0.10$; Fig. 3.7e). The park space, though connected to nodes 2 and 3, never experienced flooding under any model characteristic or degree of wetland loss, watershed gain, or combined wetland loss and watershed gain.

Additionally, I explored the differences in flood volume between the models that allowed for ponding and those that did not allow for ponding across all subsystem 1 components (Table 1). I found that the difference in flood volumes between the two models generally increased with increasing wetland loss, and that these correlations were significant for WETLOSS ($R^2 = 0.78$, $p < 0.05$) and LOSS+GAIN ($R^2 = 0.68$, $p < 0.05$), but not with WSHEDGAIN ($p \geq 0.05$)).

The relevant components for subsystem 2 (Fig. 3.8) consist of two updrain wetlands that are downdrain of independent drainage systems, which are connected by pipes to the same downdrain wetland; and the subcatchments that drain into these wetlands. Subsystem 2 (Fig. 3.8) is more complex than subsystem 1, with updrain wetlands serving as downdrain nodes to small drainage subsystems of their own, before draining into the downdrain wetlands of this subsystem. My model results for subsystem 2 were that no flooding occurred in any scenario or in any model characterization.

3.4. Discussion

Although there is enthusiasm among researchers and practitioners for the inclusion of wetlands as UEI in cities to reduce the risk of pluvial flooding, there remains a need for systemic studies of flood risk in stormwater management systems that include wetlands. Further, as cities are not static in their layouts and land cover, and rather are constantly evolving and rearranging, there is a need to study how changes in wetland cover might impact the performance of the stormwater management systems into which they are incorporated. My study represents the first city-wide exploration of the effects of changing wetland areas and volumes, according to various potential scenarios of development, on systemic flooding using an urban stormwater management system model. Further, my study is the first to isolate and compare the effects that increasing wetland areas, decreasing impervious surface area, and both increasing wetland area and decreasing impervious surface area have on flooding. Broadly, I found strong evidence that increasing wetland cover in a stormwater management system does reduce flood risk in urban stormwater management systems.

3.4.1. The effects of wetland loss on flood risk in a stormwater management system

The models projected strong positive and significant relationships between flood volumes, wetland loss, watershed gain, and combined wetland loss and watershed gain. Further, they exhibited strong and significant relationships between time the network spent flooding and wetland loss and coupled wetland loss and watershed gain. Finally, more nodes tended to flood as wetland loss increased. Though some of the increase in flood volume and time the network spent flooding could be attributed to increases in wetland flooding, the majority of the increase of both variables occurred in gray components of the stormwater management system outside of wetland areas. Thus, I find strong support for my first hypothesis that flood volume, flood duration, and the number of locations flooded in the urban stormwater management system would increase with increasing wetland loss. I also found strong support for my second hypothesis, that the gray components of the stormwater management system would experience increasing flood volume and flood duration with increasing wetland loss. My findings indicate the existence of two important features of this network incorporating natural wetland areas: (1) the effects of wetland area loss in a stormwater management system are both local and dispersed, and (2) the wetlands have excess storage and conveyance capacity that is untapped by the modern stormwater management system. I explore feature (2) more extensively in section 4.4.

Toward feature (1), it is crucial to conceive of the stormwater management system used in my analysis as being designed with assumptions of wetland performance. Chile's Ministry of Public Works, which commissioned the original SWMM and which is generally tasked with managing flood risk in cities across Chile, has planned the gray components of the infrastructure with the assumption that wetland performance remains constant through time. As Valdivia's footprint has expanded and additional pipe conduits have been added to the stormwater management system, the reliance of the system on the performance of wetlands has likely increased and become more complex. I explored the downdrain effects of wetland loss using models of two subsystems that allowed for surface ponding to occur, and generally found flooding to increase linearly with wetland loss at downdrain locations until either a wetland or a storage unit was able to intercept stormwater. I advise cities considering incorporating wetlands into their stormwater management systems to balance any performance gains with an imperative to maintain wetland characteristics that are key determinants of wetland function, particularly wetland area and volume.

3.4.2. Prioritizing updrain wetland conservation for urban flood resilience

Our results lead me to recommend that cities prioritize the conservation of upstream wetland areas in order to mitigate risk of flooding in downdrain areas. In study of subcatchment 1 (Figure 9), flood volume in the upstream wetland and downdrain nodes 1 and 2 increased with increasing wetland loss. In the LOSS+GAIN models, which are the most comprehensive representations of the effects of wetland loss, I found positive correlations between combined wetland loss and watershed gain on downdrain flooding. Downdrain node 3, where flooding occurred but was insensitive to changes in wetland or watershed area, was linked to the updrain components of the subsystem in a way that allowed large quantities of stormwater to first route to depression storage at a connected park space (which was formerly a wetland). If this park space and its storage were absent from the subsystem, I would have expected increasing flood volume for at least the WETLOSS and LOSS+GAIN models.

Adding some complexity to this recommendation, my results also suggest that wetland conservation for flood mitigation in a system of interconnected wetlands may be a lower priority than in systems where wetlands are simply updrain of gray infrastructure. In subsystem 1, I found downdrain flooding increased with wetland loss until stormwater was able to enter a park space that was formerly a wetland. In subsystem 2, I found no flooding at all downdrain from these updrain wetlands, which my analysis suggested was the result of excess storage capacity of the wetlands in this subsystem.

The sum of these results suggests that the conservation of updrain wetland extent should be of high importance to stormwater managers, especially for wetlands situated updrain of gray stormwater infrastructure. Wetlands in urban stormwater management systems provide storage for runoff, and also may delay the release of stormwater to parts of the system that are overwhelmed for finite periods of time. Kadykalo and Findlay (2016) theorized in their review of wetland flow regulation literature that these hydrological properties could have beneficial and harmful effects: the beneficial effect of storing flood waters and release them more slowly into the system, which could more evenly distribute the load the system must manage; and the harmful effect of overwhelming downstream (or in my urban setting, downdrain) components of the system with constant elevated flow. However, in my study system, I found without exception that wetlands reduced the strain on downdrain components.

3.4.3. Planning for flood-resilient futures in urban spaces

Our results in this study demonstrate how co-producing scenarios of urban development may be an effective strategy for promoting urban resilience and sustainability, to be used and adapted by other cities across the globe facing similar issues. For all of the scenarios co-produced with local stakeholders in my scenarios workshop, flooding was reduced compared with the scenario where development simply proceeded as it had historically, represented by the BAU scenario. This reduction in flood risk compared with the BAU scenario was present in all co-developed scenarios, in spite of all these scenarios experiencing some degree of wetland loss. A previous study (Iwaniec et al., 2020) has estimated that similarly co-produced scenarios of urban development were likely to reduce heat stress, flood risk, and stress on water supplies during drought in Phoenix, Arizona, USA, and other authors have also recommended co-produced visioning exercises in order to achieve resilience and sustainability goals in cities (Iwaniec et al., 2020; McPhearson et al., 2017; Miller & Wyborn, 2018). As described in the scenario-development process detailed in Iwaniec et al. (2020), the results from my study would ideally be used as part of an iterative revision of the scenario visions with the same workshop participants in Valdivia, to allow for the examination the LULC model outputs, comparison of the flood risk implications of each scenario, and weighing of the tradeoffs of each development pathway. Workshop participants may then, for example, refine objectives in certain scenarios to increase the amount of wetland cover in the city compared to present day, introduce other forms of UEI landcover to increase infiltration, and/or upgrade gray components of the stormwater management system in order to further reduce future flood risk. As part of this iterative process of revision and modeling, Valdivia and other cities using this process may set even more transformative objectives than they previously staked out, and further reduce their flood risk accordingly.

3.4.4. The (lack of) effect of increasing watershed areas on flood risk

I did not find strong relationships between watershed gain alone and most of my response variables. I hypothesized that the effect of watershed gain would be lesser than the effects of wetland loss and combined wetland loss and watershed gain, but would nonetheless be significant. Thus, my hypothesis was only partially supported.

These results may contradict previous scholarship on the effect of increasing impervious area on watershed flooding (Sohn et al., 2020; Westra et al., 2014), but I find them likely to be consistent in the context of the particular drainage system I studied. I theorize that my results provide evidence that wetland storage units and wetland conduits in my drainage system could be considered “overdesigned” in the engineering sense. That is, the naturally formed wetlands in my study featured excess capacity for storing and conveying stormwater, even during extreme precipitation intensities and volumes. In contrast, an engineering firm tasked with constructing or restoring a wetland may be bound in their wetland designs by considerations of parcel size and stormwater management system design criteria, such as that the system is only expected to manage water up to a storm with a 10-year return period occurring over the span of six hours. A wetland designed with these limits may generate local or distal flooding during storms of greater return periods or longer durations, when a natural wetland might not.

Thus, the incorporation of natural or designed wetlands with storage and conveyance capacity in excess of the needs of the modern day may be particularly valuable to cities with high-intensity urban land cover with abundant impermeable pavement or scarce low-impact development practices. Such incorporation would allow flexibility in development of the built environment of cities, and may even align with emerging safe-to-fail infrastructure design practices (Kim et al., 2017).

3.4.5. Accounting for surface routing and ponding in models of stormwater management systems

In my examination of the effects of updrain wetland removal on downdrain flooding in two subsystems, I found that models that allowed for ponding above wetlands and nodes experienced greater flood volumes than models that did not allow for ponding. Further, the differences in flood volume between the models that allowed for ponding and those that did not allow for ponding increased with increasing wetland loss and watershed gain. Outside of model simulations, this water, once ponded and once having achieved a certain depth, may route along the surface, such as along streets and down historic channels of lower relative elevation, and may not re-enter the system at the location it was produced, if it re-enters at all. Other recent studies on urban flood risk that use 1-D stormwater management system models like EPA SWMM for their analysis (Hou et al., 2020; Zhou et al. 2018) do not account for the removal of flood water from the system at the point of flooding and make no effort to account for surface routing or ponding. Based on my simulations, I expect such studies to produce underestimates of systemic flood risk.

Other studies that estimate pluvial flood risk in urban areas have opted to instead use modeling techniques that estimate runoff production and surface routing rather than flooding produced via inadequate drainage system performance (Kaspersen et al., 2020; Verbieren et al. 2013). Though runoff generation is a critical consideration in urban areas, given the prevalence of impervious surfaces, studies in urban areas only considering runoff generation and flow may overestimate pluvial flooding, as the drainage systems will manage some portion of this runoff—as they were designed to do.

Of course, the optimal model for estimating flood risk in urban areas would have robust representation of surface and stormwater management system characteristics. EPA SWMM does allow modelers to estimate runoff generation and routing, but it is much simplified compared to surface runoff generation models with more complex surface routing and runoff generation calculations. In cases where 1-D models like EPA SWMM are used, researchers and practitioners should attempt to account for ponding and surface routing or risk underestimating flood risk. To accomplish this, I recommend that cities generate high-resolution digital elevation models of their urban areas appropriate to estimate with a high degree of accuracy how water is likely to route along the surface.

3.4.6. Note on novel methodologies

Our study is the first, to my knowledge, to relate wetland loss to systemic flooding characteristics for a stormwater management system that incorporates wetlands. I presented a novel methodology for translating changes in wetland land cover to changes in wetland dimensions in EPA SWMM, allowing estimation of the effects of wetland loss on a whole urban drainage system. Further, though this study is focused on the example of one city, the lessons my data provide will allow for other cities considering the construction of new wetlands, restoration of removed or degraded wetlands, or the incorporation of natural wetlands in their stormwater management systems to understand the importance of wetland conservation as their drainage network matures and expands.

3.5. Conclusions

The present study developed unique stormwater management models to simulate the effects that wetland area loss, watershed area gain, and combined wetland area loss and watershed area gain would have on flood volume and durations of flooding in an urban drainage system. The amounts of wetland area loss in these models derived from the outputs of a cellular automata model estimating future LULC in five scenarios of future urban development in the city of Valdivia, Chile. The LULC transition rules for the cellular automata model were based on historical data and objectives developed by city stakeholders in a workshop setting. The stormwater management models are distinguished by both the amount of wetland loss they simulate and whether the model includes only loss of wetland area, gain of watershed area, or both the loss of wetland area and the gain of watershed area.

The following are the key findings of the present study:

- Systemic flooding increased with loss of wetland area, gain of watershed area, and combined loss of wetland area and gain of watershed area.
- The number of locations in the models that flooded increased with increasing wetland area loss. The majority of these flooded locations were outside of wetlands.
- Wetland locations produced increasing flood volume with increasing wetland area loss, and a greater proportion of systemic flood volume was produced in wetlands as wetland loss increased.
- Increasing loss of updrain wetland area was generally correlated with increasing volume of downdrain flooding, until stormwater was allowed to flow to storage.

Future studies on the performance of drainage systems incorporating wetland UEI may include a more accurate representation of wetland function and antecedent conditions to major precipitation events. Wetlands may be either sinks or sources of groundwater, depending on the location of the water table, the elevation of which, in turn, in settings not heavily influenced by humans, may depend on seasonal patterns of precipitation, tidal patterns, or other environmental factors. In urban areas, the elevation of the water table may even depend on the so-called “urban baseflow” of runoff from human water uses. Further, vegetated wetlands may intercept substantial amounts of precipitation on emergent vegetation, and in the process remove this precipitation from the stormwater management system. Vegetation may also remove substantial amounts of surface and/or groundwater from the system through transpiration.

Future studies should also incorporate the ponding and routing of floodwater along the surface of the city and determine how it is routed back into stormwater management systems. My system models removed floodwater from the system. However, in the case of one subsystem, in which I allowed for water to pond at the surface and re-enter the drainage system, I found substantial and significant increases in downdrain flood volume with increasing wetland loss. Further, in this subsystem, I found that the number of downdrain locations that flooded increased with increasing wetland loss. Further studies that account for ponding and surface flow may then more accurately represent flood volume, flood duration, and the more distal effects of wetland loss on the stormwater management system.

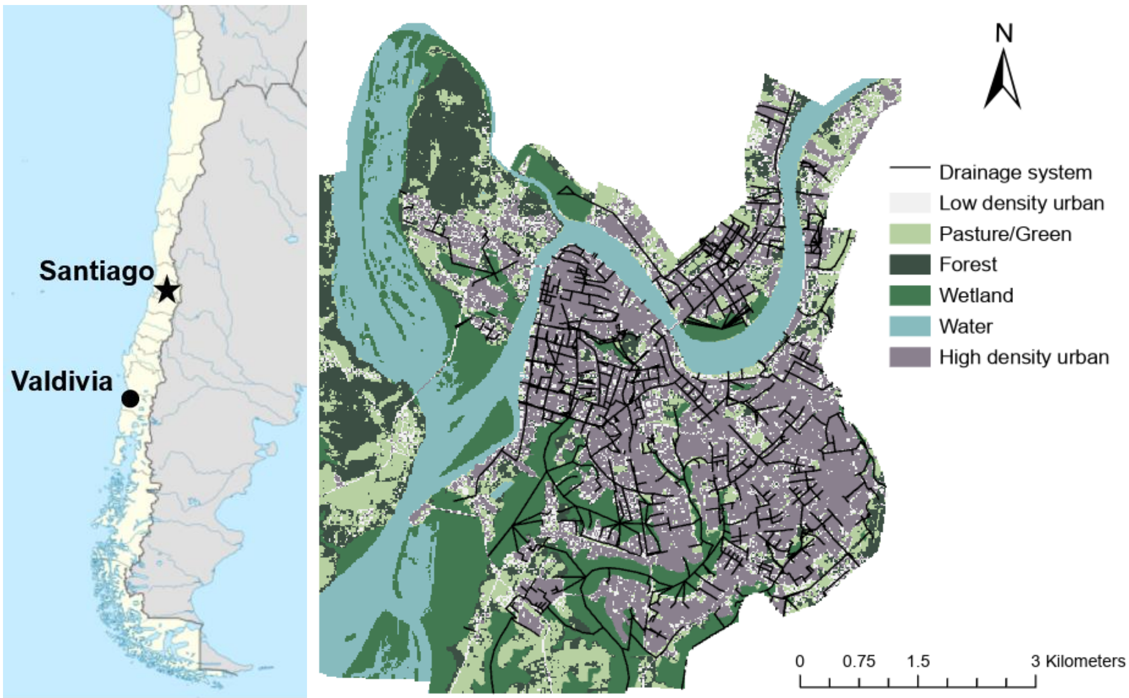

Figure 3.1. Left: Location of study site, Valdivia, Chile (39.8336 S, 73.2154 W). Right: Valdivia's land cover, as delineated via spectral analysis of an 2010 orthophoto, and drainage system, as described in 2012 by the Chilean Ministry of Public Works.

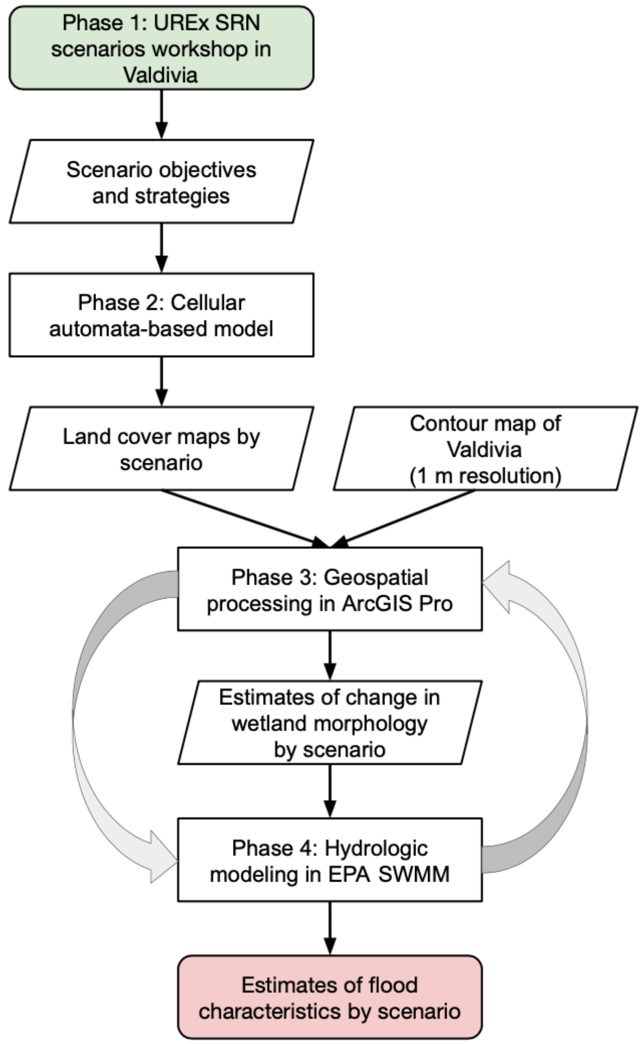

Figure 3.2. The four phases of work conducted in this study: (1) Conducting a workshop with stakeholders to develop scenario goals for Valdivia for the year 2080; (2) Modeling land cover changes for each scenario using historical data and the workshop outputs; (3) Estimating changes to wetland land cover and subcatchment area using geospatial software; (4) Modifying wetlands and subcatchments in the EPA Stormwater Management Model and modeling changes in flood characteristics. Phase 3 required some anticipation of the needs of EPA SWMM, and thus it is depicted as interacting with phase 4.

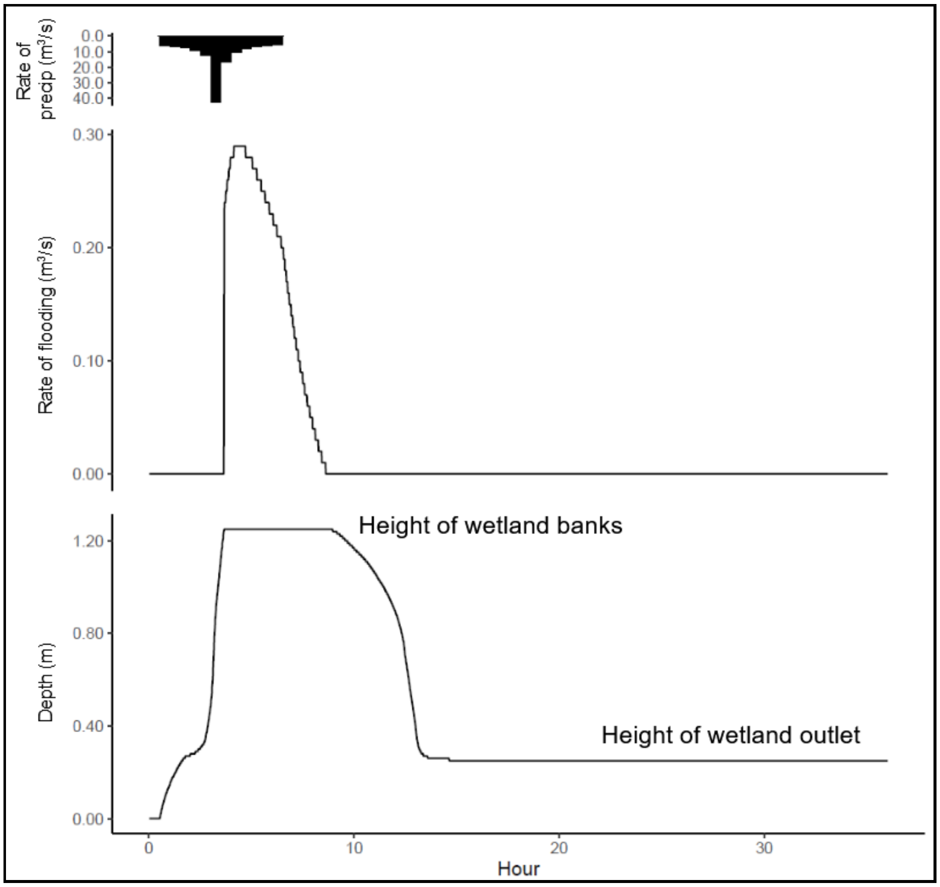

Figure 3.3. Example representation of the relationship between precipitation, surface water depth, and flooding rate for a modeled storage unit wetland. Wetland flooding occurs when surface water depth exceeds the height of the wetland banks. Flood volume is then calculated as the integral of the rate of flooding above 0 m^3/s.

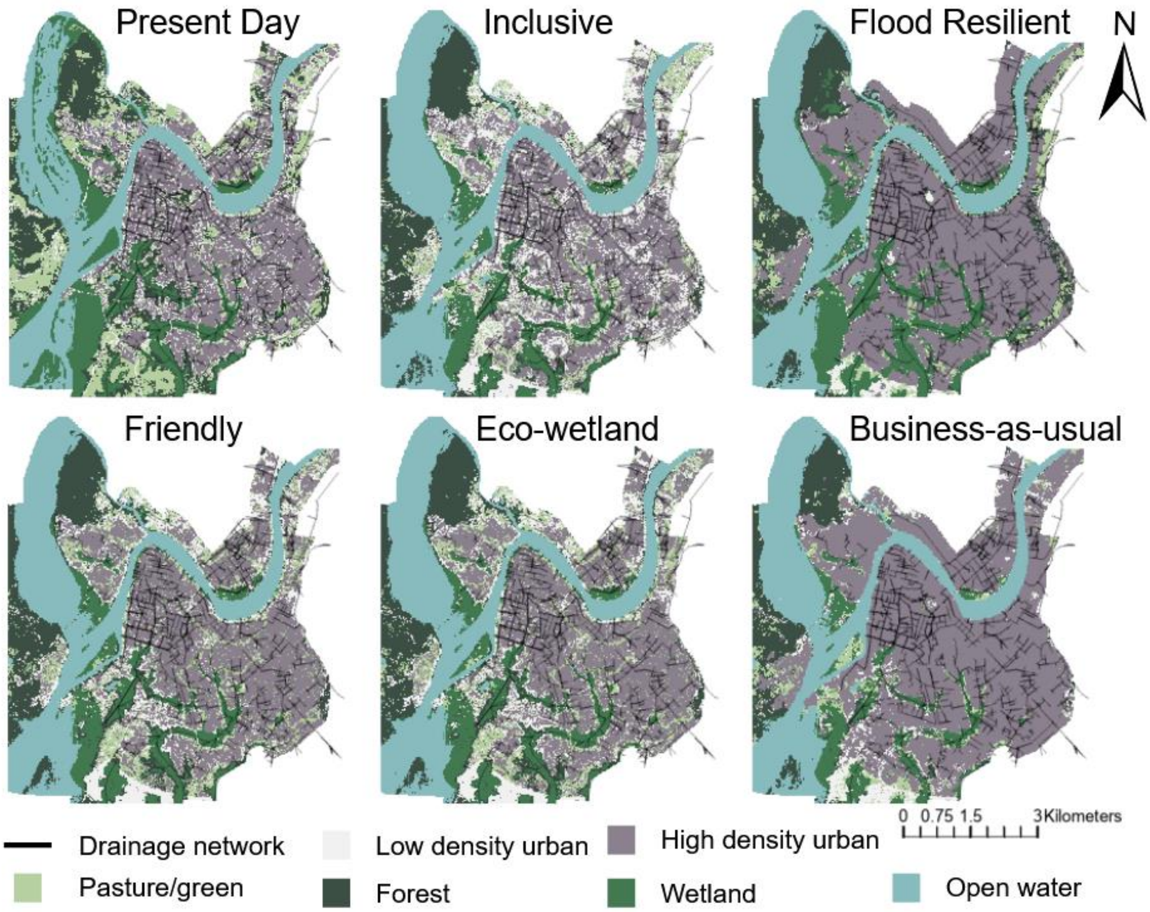

Figure 3.4. Land cover in the present day (2010) and under five scenarios of development by the year 2080. Wetland loss generally increases from left to right, and from top to bottom, compared to the present day. City-wide wetland loss for each scenario was: 9.72% in Inclusive, 13.3% in Flood Resilient, 18.3% in Friendly, 23.98% in Eco-Wetland, and 37.3% in Business-as-usual compared to Present Day wetland coverage.

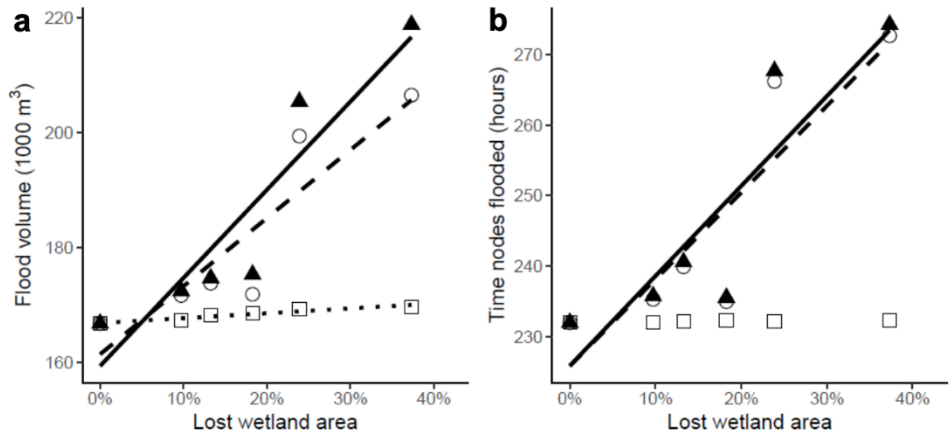

Figure 3.5. Linear regressions of (a) node flood volume (1000 m^3), and (b) time that nodes spent flooding (in hours), as a function of lost wetland area. Results for WETLOSS are indicated by circle markers and dashed trendlines; results for WSHEDGAIN are indicated by square markers and dotted trendlines; results for LOSS+GAIN are indicated by filled triangles and solid trendlines. Presence of trendline indicates a significant regression ($p < 0.05$).

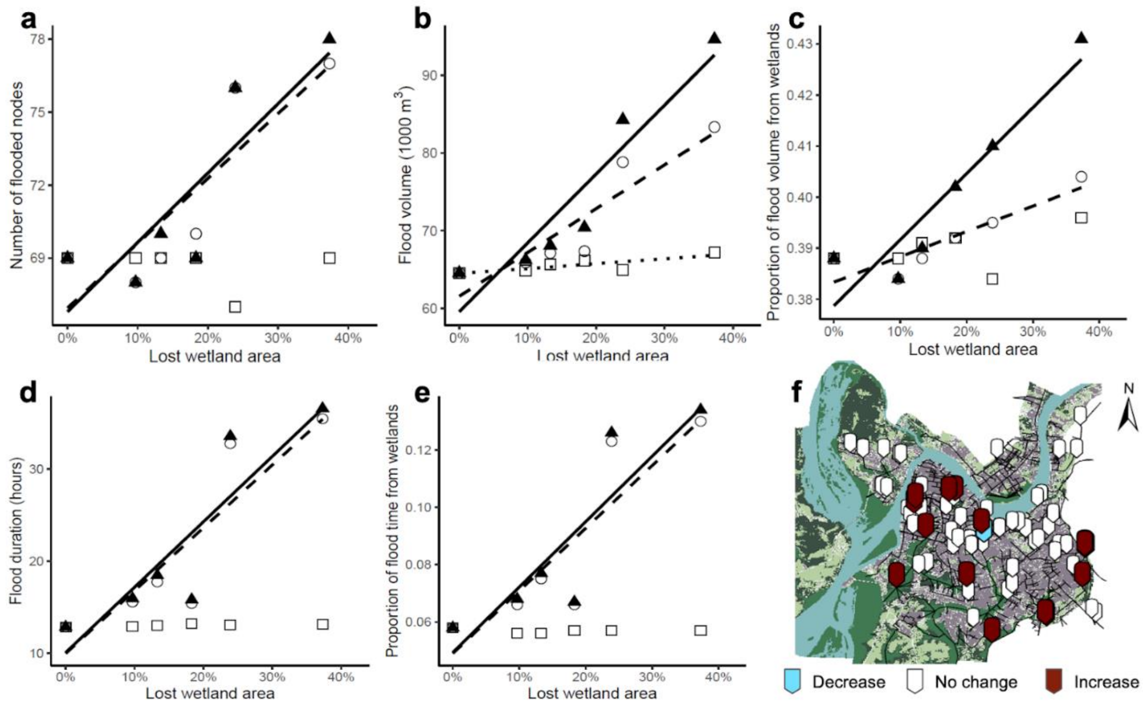

Figure 3.6. (a-e) Linear regressions for (a) total number of flooded nodes, (b) total flood volume for the SWMM, (c) proportion of wetland flood volume contribution to whole SWMM flood volume, (d) flood duration, and (e) proportion of flood time that wetlands contributed to whole SWMM flooding, vs. wetland loss. Wetland loss as in Fig. 3.4, symbols, markers, and trendlines as in Fig. 3.5. Presence of trendline indicates a significant regression ($p < 0.05$). (f) Locations and directions of change in flood volume between the present day and the BAU scenario.

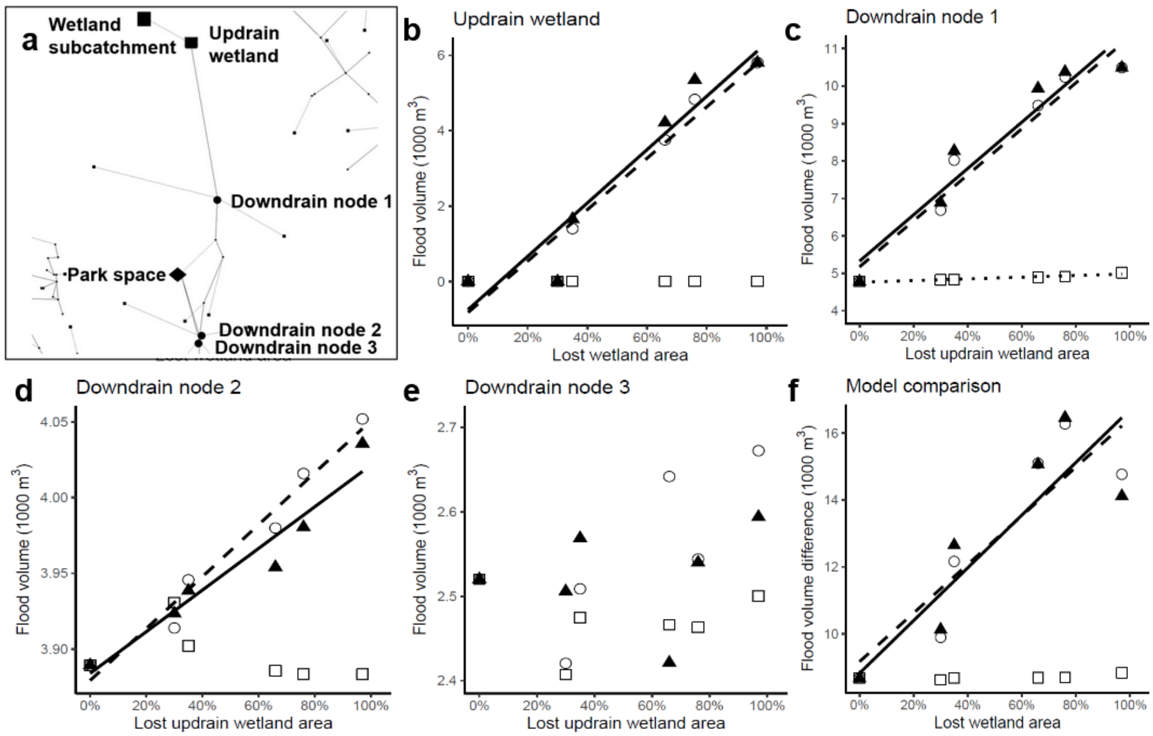

Figure 3.7. Effects of updrain wetland loss on downdrain flooding in subsystem 1. (a) EPA SWMM representation of the updrain wetland, the updrain wetland's subcatchment, the downdrain nodes that experience flooding, and a park space where a portion of stormwater is sent between downdrain nodes 2 and 3. (b-e) Regressions of (b) flood volume in the updrain wetland node as a function of wetland loss, (c) flood volume in downdrain node 1 vs. updrain wetland loss, and (d) flood volume in downdrain node 2 vs. updrain wetland loss, and (e) flood volume in downdrain node 3 vs. updrain wetland loss. (f) Differences in flood volume across all subsystem 1 nodes between the models that allowed for ponding and the models that did not allow for ponding, as a function of wetland loss. Wetland loss as in Fig. 3.4, symbols, markers, and trendlines as in Fig. 3.5. Presence of trendline indicates $p < 0.05$ except in the case of the results for the model characterized by wetland loss in downdrain node 2, where $p = 0.06$.

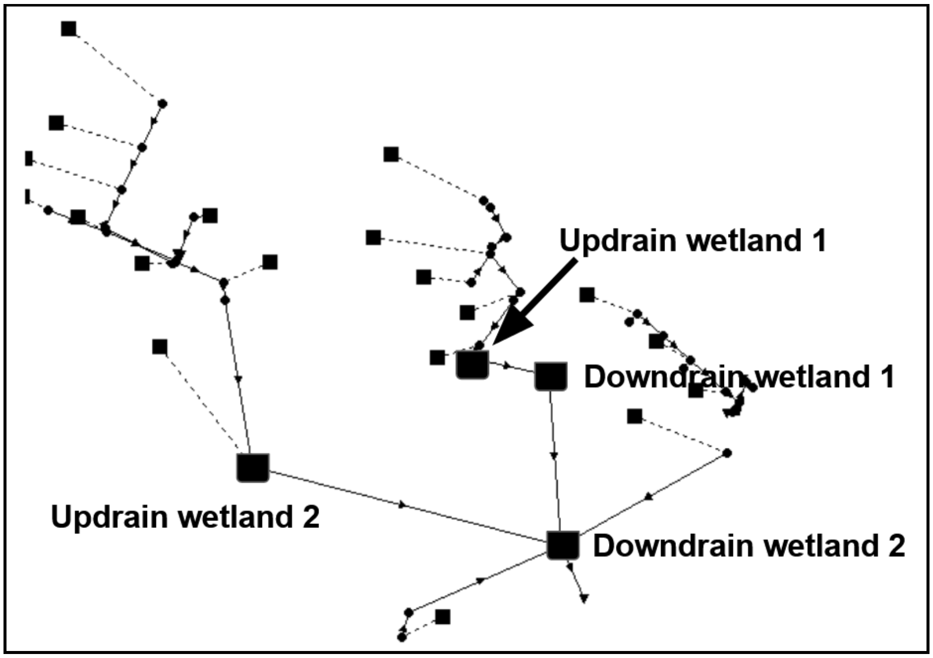

Figure 3.8. EPA SWMM representation of subsystem 2, with two updrain wetlands that ultimately drain to the same downdrain wetland. Updrain wetland 1 first drains to downdrain wetland 1, which in turn drains to downdrain wetland 2, while updrain wetland 2 drains only to downdrain wetland 2. The conduit leading downward from downdrain wetland 2 leads to a system outlet, which is incapable of flooding.

CHAPTER 4

ESTIMATING FUTURE FLOOD RISK DUE TO CLIMATE CHANGE AND LAND COVER CHANGE IN VALDIVIA, CHILE

4.0. Abstract

Cities are struggling to cope with pluvial flooding, which is flooding that results from precipitation rates that exceed the removal rates of drainage systems and of the permeable landscape. This type of flooding has become more prevalent as precipitation intensity has increased, but cities must make management decisions that will meet the needs of future eras as heavy rainfall events become even more common. Stormwater managers are pressing for accessible climate-change data at appropriate scales for cities, but most climate-change models operate at much coarser scales. Further, speculations and models of future city land cover are rare, leaving flood risk managers without the tools to consider how different land-cover configurations will affect flooding. In this study, I combined downscaled climate data generated by an asynchronous regional regression model with estimates of changes in land cover from a cellular-automata model, to assess flood risk under a variety of climate and land-cover scenarios in the South American city of Valdivia, Chile. Climate modeling projected a drier overall climate in Valdivia with more intense storms by the year 2080. In two sectors of the city, as well as for the combined sectors, projected wetland loss significantly increased the risk of flooding under all present and future rainfall volumes during extreme storm events. In sectors where this finding was significant, there was no threshold value of wetland loss beyond which the finding was not true. Equally important, I did not find significant relationships

between wetland loss and increased flooding under any rainfall volume in two other sectors of the city, indicating the need to consider other features of the wetland within the stormwater management system, such as wetland configuration. I identify strong arguments for the conservation and stewardship, and perhaps by extension even the construction, of wetlands in urban stormwater management systems. However, I emphasize that this should only be part of a city's stormwater management strategy. Further, I discuss the practical implications of managing wetland infrastructure under drier climates and under changing legal protections of wetlands.

4.1. Introduction

Pluvial flooding is a major concern for urban areas, and can lead to loss of life, damage to property, and disruption of transportation networks (Chang et al., 2010; Douglas et al., 2010; Falconer et al., 2009; Yin et al., 2016). Pluvial flooding is surface ponding or overland flow that occurs when rates of precipitation exceed the rates of removal by drainage systems and/or rates of infiltration of surfaces (Falconer et al., 2009).

Only recently has research begun exploring with depth the abilities of inland urban wetlands to reduce urban pluvial flood risk, or how the incorporation of wetlands in an urban stormwater management system might alter the system's performance. The theory and practice of inland wetland restoration and construction in urban areas to reduce pluvial flood risk is relatively new in academia and among stormwater managers (Chan et al., 2018; Elmqvist et al., 2015), and modeling and empirical studies of the effects of wetland restoration and construction in urban areas are rare. Some cities have added inland wetlands to their portfolios of urban ecological infrastructure (UEI; Childers et al. 2019; otherwise known as green infrastructure (GI), green stormwater infrastructure (GSI), or, more broadly, nature-based solutions (NBS)) or suggested that the construction, restoration, or incorporation of inland wetlands be included in sustainable urban drainage systems (SuDS) or low-impact development strategies (LiDS) to reduce pluvial flooding (Chan et al., 2018; Fletcher et al., 2015; Li et al., 2020).

The long-term viability of such UEI in mitigating pluvial flood risk has yet to be adequately explored. In previous chapters, it was demonstrated how the flood risk mitigation services of wetland UEI in a stormwater management system are negatively correlated with surface and soil water storage conditions preceding a storm, and positively correlated with wetland volumetric capacity. As such, the ability of wetlands to maintain flood-risk mitigation services in the future at levels comparable to the present day depends on wetland management for these factors. That is, wetland storage conditions can be modified through technological means, such as gating and channelization, and wetland volumetric capacity can be maintained or expanded through city policy.

However, one factor, climate change, has the potential to alter flood risk in cities in ways that are more complex to manage. First, changes in annual rainfall amount and timing may shift conditions away from those for which stormwater management systems were designed. In the case that annual rainfall reduces and the intensity of the average storm also reduces, cities may experience a reduced number of floods due to a drier stormwater management system and reduced load during storms. However, if annual precipitation increases and/or the intensity of the average storm increases, cities may contend with a greater number of floods in a year due to a waterlogged stormwater management system and increased load during storms. There are also intermediate scenarios to contend with, such as increased annual precipitation through an increase in the number of storms of lesser intensities, or reduced annual precipitation with an increase in the intensity of extreme storms, among others–all of which have their potential impacts on flood risk. The direction and degree of these effects may in turn vary depending on the configuration and inventory of a stormwater management system and its urban drainage shed.

In this study, I use projections from long-term, downscaled climate models and land-cover change models in a Chilean city, Valdivia, to estimate the change in flood risk under a range of future city configurations and precipitation conditions, compared to the present day. I asked: What is the range of future flooding conditions in Valdivia, given scenarios of urban development and projections of climate change, and how can the city best manage for one or several of them? My goal was to provide a viable methodology for cities worldwide to bound their future flood risk, and to provide a more specific guide for cities doing so with an inventory of wetlands, either current or proposed, in their stormwater management systems.

4.2. Materials and Methods

4.2.1. Study site

Valdivia, Chile (area: 93.94 km2) is a city of approximately 166,000 people in the southern half of Chile, 850 km south of the capital Santiago, in the Región de los Ríos (Fig. 4.1). Citizens and stormwater managers in Valdivia must contend with a high risk of pluvial flooding owing to high average annual precipitation, a long rainy season, and the city's location 12 km inland from the Pacific Ocean, at the confluence of three rivers. Its ecosystem is classified as a temperate rainforest (Hajek & di Castri, 1975; Amigo & Ramirez, 1998). Wetlands are a characteristic feature of Valdivia, covering 20.64 km^2 (22.7%) of the municipal area.

Although Valdivia's average annual rainfall was approximately 1760 mm between 1985 and 2015, it has in recent years faced problematic drought (Dirección General de Aeronáutica Civil, 2020). Of note, in 2015, rainfall in the region and snowpack in the Andés were low enough that the river from which the city derives its potable water became too saline for treatment due to tidally forced saltwater intrusions from the nearby ocean, and the city was forced to pump nearly all of its supply from groundwater sources (Vargas et al., 2020). In 2021, for the first time since the city began measuring precipitation at the Pichoy Airport meteorological station in 1969, the city registered less than 1,000 mm of precipitation (Sepúlveda, 2021). This recent spate of drought has provoked concern about the changes Valdivia will face in a future with potentially less precipitation (Sepúlveda, 2021; Vargas et al., 2020).

Valdivia's stormwater management system is composed primarily of gray infrastructure components (e.g., pipes and canals) and wetlands. As of 2012, Valdivia's stormwater management system consists of roughly 245.7 km of drainage infrastructure, of which 41.19 km (16.8%) is wetland UEI. The origin of most of this wetland cover is a 1960 earthquake of magnitude 9.5, which caused up to 20 m of uplift in some areas (Barrientos & Ward, 2007) and subsidence and rifting in others. Since 1960, the city has deliberately incorporated these wetlands into its stormwater management system (CMOP, 2012). In addition, the continued presence of wetlands in the city is owed in part to local conservation movements to maintain the cultural services of wetlands (Correa et al., 2018) and their function as habitat to charismatic species tied to Valdivian identity (Silva et al., 2015).

The flow of stormwater through gray infrastructure components and wetlands in Valdivia is variable and complex. For example, stormwater may first enter the system as runoff via gray infrastructure components, flow to a wetland, and then to a river outlet. Alternatively, stormwater may first enter the system as runoff via wetlands, which may then flow into gray infrastructure components, which then may flow to river outlets. In a more complex scenario, stormwater may first enter the system via a wetland, then flow through gray infrastructure components for some distance, pass through a second, downdrain wetland, and flow through several other gray infrastructure components and wetlands before reaching a river outlet. The stormwater management system of Valdivia thus depends heavily on the coupled performance of gray infrastructure components and wetlands, and I would expect to see changes in either element affecting the performance of the other.

4.2.2. General approach

I employed mixed methods that incorporated work previously accomplished for chapter 3 (Fig. 4.2). I included SWMM model outputs from chapter 3 that reflect the stormwater management system under the present day, as well as the stormwater management system in the year 2080 under four desirable development pathways and one business-as-usual pathway. Unique to this chapter, I used daily precipitation estimates of

23 climate models that were downscaled using daily precipitation data from the Pichoy Airport meteorological station to estimate annual rainfall volumes and to estimate the rainfall volume of a 100-year return period, 24-hour duration storm by the year 2080. These outputs were used as design storms in the Environmental Protection Agency's Stormwater Management Models (EPA SWMM; U. S. EPA) from chapter 3 with present-day and possible future wetland extent in order to estimate a range of characteristics of flooding in 2080.

4.2.3. Estimating future land cover scenarios and wetland characteristics

In May of 2017, the Urban Resilience to Extremes (UREx) Sustainability Research Network (SRN) hosted a workshop in Valdivia, Chile, to envision a series of long-term (2080) future scenarios and desirable future pathways of urban development. Participants collaborated to develop a suite of visions and strategies to undertake in order to achieve four unique, plausible scenarios for a future Valdivia: an Inclusive City, a Friendly City, an Eco-Wetland City, and a Flood-Resilient City, according to a visioning and scenario development methods described in Iwaniec et al. (2020). The strategies of these four scenarios were then translated by the UREx SRN modeling team into quantitative rules, spatial and temporal, and introduced into cellular automata-based models of land-use/land cover (LULC), and paired with historical information on LULC transformations, to produce LULC estimates for the year 2080. This process produced LULC map outputs for the scenarios, as well as for a "business-as-usual" (BAU) scenario, which assumed LULC change proceeded entirely according to historical patterns of development (Fig. 4.3).

As detailed in chapter 3, these LULC maps were used to estimate changes in wetland volumes and contributing watershed areas for each of the scenarios by the year 2080 using ArcGIS Pro (ESRI). These changes in wetland volume were then converted into changes in the EPA SWMM model that Valdivia's Ministry of Public Works (Ministerio de Obras Públicas) uses to estimate flood risk in the city. This process produced six SWMMs, one for the present-day land cover, and five for the future land-cover scenarios. Each SWMM can be further reduced into four separate sectors of Valdivia: Southeast, Barrios Bajos, Las Animas, and Isla Teja. In this study, when these sectors and their results are referred to in aggregate, they are identified as the "combined sectors."

4.2.4. Downscaling climate models to Valdivia, Chile

I used a form of asynchronous regional regression models (ARRMs) to downscale precipitation estimates from atmosphere-ocean general circulation models (AOGCMs) to Valdivia, Chile. This technique was demonstrated by Stoner et al. (2012) to be an efficient, robust, and easily generalizable method for producing more regionally appropriate estimates of temperature and precipitation from coarser AOGCMs by training them on local observational data of temperature and precipitation. Using methods detailed in Stoner et al. (2012), I trained 23 AOGCMs (Table 4.1) on observational data between 1969–2015 from the Pichoy Airport meteorological station in Valdivia and produced daily estimates of precipitation localized to the station for the years 1969–2080 (Table 4.2). This station is within the urban boundary and the closest in proximity to the city center of Valdivia of any other meteorological station in the region, and it has a rainfall record beginning in 1969.

4.2.5. Estimating rainfall volume of 100-year, 24-hour storms

Historical precipitation data in this study consisted of daily precipitation reported by the meteorological station located at the Pichoy Airport of Valdivia between the years 1969 and 2021. Rainfall data used for estimating rainfall beyond this historical record were the daily estimates of rainfall produced by the 23 downscaled climate models.

Rainfall of historical and future 100-year, 24-hour storms were estimated using the generalized extreme value (GEV) distribution. The GEV distribution is the combination of three extreme value distributions (the Gumbel ($e^{-e^{-y}}$), Frechet ($e^{-e^{-\gamma}}$), and Weibull ($e^{-(-y)^{\gamma}}$) distributions), and is given by the following equation:

$$G(z) = e^{-\left[1+\xi\left(\frac{z-\mu}{\sigma}\right)\right]_{+}^{-1/\xi}} \quad (1)$$

Where,

G(z) is the probability that the monthly precipitation will be greater than or equal to *z* mm

μ is the location parameter

σ (σ > 0) is the scale parameter

ξ is the shape parameter

The return period T of a rainfall amount greater than *z* is understood to be:

$$T = \frac{1}{P(exceedence)} \quad (2)$$

Where *P(exceedance)* is the probability of event exceeding rainfall amount *z*.

Thus, the return period T can be related to the GEV by:

$$G(z) = 1 - \frac{1}{T} \tag{3}$$

$$e^{-\left[1+\xi\left(\frac{z-\mu}{\sigma}\right)\right]_{+}^{-1/\xi}} = 1 - \frac{1}{T} \tag{4}$$

$$z = \mu + \frac{\sigma}{\xi}\left(\left(-ln\left(1 - \frac{1}{T}\right)\right)^{-\xi} - 1\right) \tag{5}$$

Using equation (5), 1200 months for T (or 100 years), and the μ, σ, and ξ parameters from fitting the GEV model to the data, I was able to estimate the precipitation expected to fall in a 100-year, 24-hour storm in any period of interest. The GEV distribution is commonly employed for modeling extremes in rainfall such as extreme events of various return periods (Bella, Dridi, & Kalla, 2020; Reiss & Thomas, 2007).

Observational and modeled daily precipitation data were first used to determine monthly maximum rainfall. These monthly extremes were then grouped into periods of 30 years (for example, 1986–2015) in R (R Core Team, 2022) and fitted to a GEV distribution using the extRemes package (Gilleland & Katz, 2016). From this fitted model, I was able to estimate the mean rainfall expected during a 100-year event, along with the 95% confidence interval, using my historical data and ensemble average precipitation data for the RCP 4.5 and RCP 8.5 climate scenarios (Table 4.2). Rainfall was distributed along a normal distribution over a 24-hour period, such that rainfall peaked at hour 12 of the model run.

Because annual rainfall and event frequency were both expected to decrease by 2080 compared to the mean values from 2021, I expected that wetland stage would be at or below the levels that I observed in campaign 2 of Chapter 2. In this campaign, annual rainfall in the first year was abnormally low (1071.3 mm in 2019), leading to wetland stage at or near 0 m. A drier climate on average by 2080 would likely reduce surface and soil water stores in the region, leading to low wetlands stage, perhaps at or near 0 m except during the onset and peak of the rainy season. I demonstrated the effects of antecedent wetland surface and soil storage in chapter 2, and so I expect that infiltration in particular may reduce system-wide flooding. However, due to the added complexity of comparing models with and without infiltration, I was unable to model system-wide infiltration in the SWMM in this study. However, I did set wetland stage to 0 m at the start of modeling, reflecting drier surface storage conditions.

In one subwatershed of the southeast sector, where I modeled the effects of downstream flooding in chapter 3, I were able to allow for infiltration in one wetland and one park space (a former wetland; Fig. 4.4). I tested the limits of the effects of infiltration by using as design storm inputs the same rainfall as the mean, lower CI, and upper CI for the 2021 and 2080, RCP 8.5 storms (Table 4.2). I used a soil moisture deficit of 7.8%, which corresponds to the deficit in the driest soils I were able to collect during my dry season campaign in a relatively dry year (2017; see chapter 2), and which would be the most representative soil moisture deficit of those I measured in wetlands in 2080. Initial wetland stages were set at 0 m. All flooding at the wetlands and downdrain nodes in the image were summed to produce estimates of subsystem flooding.

4.**3. Results**

4.3.1. Trends in annual rainfall in the observed and modeled datasets

Based on the prior 30 years of observational data, mean annual rainfall in 2015, which represents the end of the training dataset for the models, was 1750.81 ± 58.58 mm, and modeled mean ensemble average annual rainfall was 1800.62 ± 15.63 mm and 1803.50 ± 14.71 mm in the RCP 4.5 and RCP 8.5 climate scenarios, respectively (Fig. 4.5). Root mean square error (RMSE) of the ensemble average annual rainfall during this period was 331.86 and 318.67 for the RCP 4.5 and RCP 8.5 climate scenarios, respectively. Based on the prior 30 years of observational data, mean annual rainfall in 2021 was 1693.43 ± 64.55 mm, and modeled mean ensemble average annual rainfall was 1765.86 ± 13.46 mm and 1784.04 ± 12.50 mm in the RCP 4.5 and RCP 8.5 climate scenarios, respectively. Root mean square error (RMSE) of the ensemble average annual rainfall during this period was 373.21 and 379.10 for the RCP 4.5 and RCP 8.5 climate scenarios, respectively (Fig. 4.5).

Ensemble average predictions of average annual rainfall between 2051–2080 were projected to be 1581.28 ± 14.09 mm under the RCP 4.5 climate scenario or 1449.86 ± 17.78 mm under the RCP 8.5 climate scenario (Fig. 4.5). Model projections of mean annual precipitation represented a 9.68% and 17.19% reduction in annual precipitation compared with the observed mean annual rainfall in 2015, and a 6.62% and 14.38% reduction in annual precipitation compared with the observed mean annual rainfall in 2021, for the RCP 4.5 and RCP 8.5 climate scenarios, respectively (Fig. 4.5).

Observed mean annual rainfall did not significantly decrease between 1969–2015, nor between the 30-year window of 1986–2015, but did significantly decrease between 1969–2021 ($R^2 = 0.09$, $p < 0.05$; Table 3). Ensemble average annual rainfall decreased significantly between 2016 and 2080 for both the RCP 4.5 ($R^2 = 0.40$, $p < 0.01$) and RCP 8.5 ($R^2 = 0.80$, $p < 0.01$) climate scenarios. The majority of individual models predicted a significant decrease in annual rainfall in the periods 1969–2080 and 2016–2080, but not for the periods 1969–2016 or 1969–2021 (Appendix A).

Rainfall of the mean 100-year, 24-hour duration storm in 2015 was 233.02 ± 3.17 mm in the observed data, and 218.98 ± 1.74 mm and 220.41 ± 1.53 mm from the ensemble averages of the RCP 4.5 and RCP 8.5 climate models, respectively (Fig. 4.6). The RMSE of the mean ensemble average annual rainfall from 1998–2015 was 18.96 and 17.50 for the RCP 4.5 and RCP 8.5 climate scenarios, respectively. In 2021, the return period of the same storm was 231.32 ± 3.32 mm in the observed data, and 221.16 ± 1.62 mm and 222.22 ± 1.39 mm in the RCP 4.5 and RCP 8.5 climate scenarios, respectively. The RMSE of the mean ensemble average annual rainfall from 1998–2021 were 16.93 and 15.97 for the RCP 4.5 and RCP 8.5 climate scenarios, respectively (Fig. 4.6).

Ensemble average model predictions of mean rainfall of a 100-year, 24-hour duration storm between 2051–2080 were 262.92 ± 1.27 mm under the RCP 4.5 climate scenario and 297.14 ± 1.28 mm under the RCP 8.5 climate scenario (Fig. 4.6). These mean estimates of the rainfall of such a storm represented increases of 15.91% and 30.99% in rainfall, for the RCP 4.5 and RCP 8.5 climate scenarios, respectively, compared to the year 2021(Fig. 4.6). The lower 95% CI, mean, and upper 95% CI estimates for rainfall during a 100-year, 24-hour return-period storms used in models can be found in Table 2.

The rainfall from an observed mean 100-year, 24-hour duration storm significantly decreased between 1998 and 2015 ($R^2 = 0.58$, $p < 0.01$), but not for the full observational period between 1998 and 2021 ($p > 0.05$). The ensemble average of the rainfall of a 100-year, 24-hour return period storm significantly increased between 2016–

2080 in the RCP 4.5 ($R^2 = 0.88$, $p < 0.01$) and RCP 8.5 ($R^2 = 0.95$, $p < 0.01$) climate scenarios, and between 2051–2080 in the RCP 4.5 ($R^2 = 0.45$, $p < 0.01$) and RCP 8.5 ($R^2 = 0.79$, $p < 0.01$) climate scenarios. The majority of models predicted significant increases in the rainfall of a 100-year, 24-hour storm over any period examined, and the ensemble average significantly increased over all periods examined (Appendix B).

4.3.2. Flood modeling in EPA SWMM

Flood duration and flood volume increased significantly with wetland loss and with storm intensity in two sectors of the city as well as city-wide (Fig. 4.7; Table 4.3). Mean values of flood duration and flood volume did not overlap between emission scenarios; however, for storms representing the lower and upper 95% CI of each scenario, flood duration and flood volume did overlap between emission scenarios. Flooding significantly increased in two sectors and in the combined sectors with increasing rainfall ($p < 0.05$; Table 4.3).

For the combined sectors, using mean rainfall and holding the present-day land-cover configuration constant, flood volume from an RCP 4.5 and RCP 8.5 100-year, 24-hour return-period storm increased flooding by 40.1% and 80.0%, respectively, compared to a present-day event (Fig. 4.7). Holding the BAU land cover configuration constant, flood volume from an RCP 4.5 and RCP 8.5 100-year, 24-hour return-period storm increased flooding by 36.5% and 72.5%, respectively, compared to a present-day storm. Comparing an RCP 8.5 storm in the BAU land-cover configuration to a present-day storm in the present-day land-cover configuration, flooding increases by 99.1%, which represents the greatest difference in flood volumes between any land-cover configuration for the mean estimated rainfall of the three emissions scenarios.

Additionally, for the Southeast, Barrios Bajos, and combined sectors, there were significant positive correlations between the rate of conversion of rainfall to flooding and wetland loss, indicating that more rainfall volume was converted to flood volume as wetland loss increased (Southeast: 0.0198 ± 0.0025 m^3/mm/% wetland loss, $p < 0.05$; Barrios Bajos: 0.0169 ± 0.00381 m^3/mm/% wetland loss; combined sectors: 0.0369 ± 0.00535 m^3/mm/% wetland loss, $p < 0.01$; Fig. 4.8). However, conversion rate of rainfall to flood duration was not correlated with wetland loss in any sector nor across combined sectors ($p > 0.05$).

4.3.3. Infiltration in subwatershed

Flood volume significantly increased with increasing wetland loss in the subwatershed for the models that did not allow for infiltration and for those that did involve infiltration under all rainfall volumes (Table 4.4; Fig. 4.9). However, the difference between models that allowed for infiltration and those that did not diminished with increasing wetland loss. For a given amount of wetland loss, differences in flood volume were greater with increasing rainfall (i.e., difference between present-day and RCP 8.5 scenarios; Fig. 4.9).

4.4. Discussion

The Intergovernmental Panel on Climate Change is predicting dramatic changes in rainfall amount and timing across the globe (IPCC 2021) that will worsen the load on stormwater systems, many of which are already underperforming, and render many more inadequate. Cities are also facing increasing flood risk due to historical patterns of development and urban designs that promote the proliferation of impervious surfaces, which are known to exacerbate pluvial flood risk (Morita, 2014). The combined effects of these phenomena will likely include a worsening of existing pluvial flood zones or the creation of new ones. To evade such a future, stormwater managers must consider a range of future climate and land-cover scenarios and develop stormwater management systems that can account for a broad range of uncertainty.

From the results of this study, I identify strong arguments for the conservation and stewardship, and perhaps by extension even the construction, of wetlands in urban stormwater management systems. First, flood duration and volume increased with projected wetland loss under any climate scenario examined, for two sectors of the city and for the combined sectors. This was true even for the upper confidence intervals of precipitation, which in the case of RCP 8.5 represented a 278% increase above the mean rainfall volume of the mean 100-year, 24-hour duration storm event in 2021. Thus, I did not find an exhaustion point in these sectors for wetlands in terms of stormwater inputs, i.e., a storm producing an amount of rainfall beyond which wetlands did not significantly reduce flooding. Further, I did not observe exhaustion even at my upper bound of wetland loss under the business-as-usual scenario, indicating that the presence of urban wetlands may still provide stormwater mitigation even at high rates of wetland loss. However, the ability of wetlands to reduce flood volume through infiltration decreased as wetland loss increased, indicating that the contribution of infiltration toward flood mitigation may be lesser in small urban wetlands than the contribution of other hydrologic processes, such as flow rate reduction and surface storage (Fletcher et al., 2013; Kadykalo & Findlay, 2016). Additionally, because the construction of the EPA SWMM model allows infiltration to proceed infinitely rather than up to a certain level of saturation or depth, the reduction of flood volume found in this study may be greater than what occurs in the real world. Thus, the emphasis on the infiltration function of inland urban wetlands in the literature (Gülbaz & Kazezyilmaz-Alhan 2016; Chan et al., 2018) may mislead cities and researchers as to the major sources of the flood mitigation services of their current or proposed wetlands.

To qualify these recommendations toward wetland conservation, stewardship, and construction, I emphasize that these trends were significant for only two of the four sectors of the city. That is, I found two sectors (representing six of the 28 wetlands modeled in this study) in which wetland loss had no effect at any modeled extent. These results are consistent with the results of Sauer et al. (in revision), which examined the effects of wetland loss on flooding in the same system using a 100-year return period, 6-hour duration storm under present climate. Similar to that study, the wetlands in the two sectors that did not experience an increase in flooding did not themselves flood under any degree of wetland loss or rainfall volume. In an alternative land cover scenario where these wetlands are reduced to a size such that they themselves flood under a given rainfall event, I would then expect to see a significant correlation between wetland loss and flood volume. Additionally, Sauer et al. (in revision) identified that loss of upstream or "updrain" elements led to increased downstream or "downdrain" flooding. Thus, the spatial relationship, or configuration, of the wetlands in these two sectors may also have contributed to their flood-risk mitigation abilities in the systems to which they are connected.

Thus, I find qualified support for urban inland wetlands reducing flood risk in cities. Urban wetlands may have the ability to reduce flood risk under a variety of rainfall amounts depending on wetland dimensions and configuration within the stormwater management system. However, the difference in flooding between different wetland extents is much smaller than the difference in flooding between present-day and future 100-year, 24-hour storms. Valdivia features 22.7% wetland cover, and I estimate that 16.8% of its stormwater management system's length is wetland, which are figures that most cities that attempt to incorporate wetlands into their stormwater management system will never achieve–yet storms still generate flooding in Valdivia. It may be that cities can produce greater reductions in flooding through the strategic placement of wetlands in their network, or perhaps by engineering them (e.g., adding channels, adding gates to reduce water levels before storms) to a greater extent than does Valdivia. Urban wetlands are then perhaps best used as one form of UEI among many others to reduce the risk of flooding under a changing climate rather than as the only tool. Furthermore, urban wetlands can reduce flood risk while providing a host of other ecosystem services not afforded by other forms of UEI (Davidson et al., 2019, Millennium Assessment, 2005, Wong et al., 2017; Xu et al., 2020), such as habitat for increased biodiversity, local climate regulation, aquifer recharge, aesthetics, educational opportunities, and spiritual connection. (Correa, Blanco-Wells, Berrena, & Tacón, 2018; Millennium Assessment, 2005).

Changing patterns of precipitation will likely impact the characteristics of a city's UEI, and consequently the abilities of its inventory to provide flood-mitigation services as well as many other ecosystem services. This is particularly true for wetland UEI. In Chapter 2, the positive correlation between wetland storage conditions of surface and soil water was demonstrated. Thus, I may expect that a drier future for a city may increase the available storage of wetlands soils and surfaces through drying, and in doing so provide a reduction in flood risk, or conversely that a wetter future for a city may reduce the available storage of wetlands and increase flood risk. But of course whether or not a wetland persists as a wetland also depends on prevailing patterns of precipitation. In a drier future, a wetland may not be able to support characteristic wetland communities, its soils may compact, and it may not feature surface water for long durations. Under such conditions, wetlands may convert to another ecosystem in the views of ecologists as well as legal experts. To this point, the definition of a wetland according Chile's law number 20.283, which is used to delineate wetlands as part of the National Inventory of Wetlands, is an ecosystem associated with substrates that are temporarily or permanently saturated with water, which permit the existence and development of aquatic biota (Ministerio de Agricultura, 2011). Without aquatic plants and saturated soils, wetlands may lose their classification as wetlands in the view of the law, along with their attendant protections, such as those proposed in the Chilean law 21202 that affords elevated protections for wetlands located in urban boundaries that meet these characteristics (Ministerio del Medio Ambiente, 2020). Thus, to maintain the full suite of ecosystem services of wetlands, including their flood mitigation services, cities would benefit from wetland-management strategies and stormwater-management system design that would allow flood-mitigation services of wetlands to persist under a range of future climate conditions. This may mean the adoption of strategies that divert water to wetlands from elsewhere in the city, maintaining groundwater flows through construction restrictions, or reductions of pumping during drier seasons, among others.

Finally, I emphasize the criticality of modeling in assessing the effectiveness of urban UEI for reducing pluvial flood risk in a future characterized by climate change–for cities both with and without wetlands. Rosenzweig et al. (2021) highlighted many of the strengths and shortcomings of existing methods for pluvial flood modeling, but limited their analysis to the modeling of hydrological processes. Globally, many cities lack downscaled climate data necessary to produce general estimates of climate and intensity of extreme storm events to configure their model elements (e.g., starting values for wetland stage and soil storage) and produce design storms to input into those models. Once this preliminary work is done, cities then require robust pluvial flood models that account for several key hydrological pathways (e.g., surface flow, flow through stormwater management systems, groundwater flow, infiltration, and evapotranspiration) and account for other sources of water in the system (e.g., rivers, oceans, storm surges). Very few models in existence can even account for these pathways and water sources (Tu, Wadzuk, & Traver, 2019). Such models are rarely found in practice. However, to develop strategies to reduce the threat of flooding that future extreme events evidently pose, such models, and the data to support such models, are critical to building, maintaining, and improving resilient urban spaces.

Table 4.1. The names, host institutions, atmospheric resolutions, and oceanic resolutions of the 23 CMIP5 models used in this study to produce downscaled climate data. *lat(i,j)* and *lon(k,l)* indicate latitudes and longitudes defined by these indices that have rotated poles and cannot be reduced to values simple enough for this table.

Model name	**Host institution**	**Atmospheric resolution**	**Oceanic resolution**
ACCESS1-0	CAWCR	1.25° × 1.875°	*lat(i,j)* × *lon(k,l)*
ACCESS1-3	CAWCR	1.25° × 1.875°	*lat(i,j)* × *lon(k,l)*
BCC-CSM1-1	BCC	2.7906° × 2.8125°	0.3333° × 1°
BCC-CSM1-1-M	BCC	2.7906° × 2.8125°	0.3333° × 1°
BNU-ESM	BNU	2.7906° × 2.8125°	0.3344° × 1°
CanESM2	CCCma	2.8° × 2.8°	1° × 1°
CCSM4	NCAR	0.9424° × 1.25°	*lat(i,j)* × *lon(k,l)*
CMCC-CM	NCAR	0.75° × 0.75°	2° × 2°
CNRM-CM5	CNRM/CERFACS	1.4° × 1.4°	1° × 1°
CSIRO-MK3.6	CSIRO	1.875° × 1.875°	1.875° × 1.875°
GFDL-ESM2G	GFDL	2° × 2°	0.375° × 1°
GFDL-ESM2M	GFDL	2° × 2°	0.3344° × 1°
HadGEM2-CC	MOHC	1.25° × 1.875°	0.3396° × 1°
HadGEM2-ES	MOHC	1.25° × 1.875°	0.3396° × 1°
INM-CM4	INM	1° × 2°	0.5° × 1°
IPSL-CM5A-LR	IPSL	1.875° × 3.75°	*lat(i,j)* × *lon(k,l)*
IPSL-CM5A-MR	IPSL	1.25° × 2.5°	*lat(i,j)* × *lon(k,l)*
MIROC5	AORI/NIES/JAMEST	2.8125° × 2.8125°	1.4° × 2°
MIROC-ESM	AORI/NIES/JAMEST	2.8125° × 2.8125°	Orthogonal curvilinear coordinates
MIROC-ESM-CHEM	AORI/NIES/JAMEST	2.8125° × 2.8125°	*lat(i,j)* × *lon(k,l)*
MPI-ESM-LR	MPI-M	1.8653° × 1.875°	*lat(i,j)* × *lon(k,l)*
MPI-ESM-MR	MPI-M	1.8653° × 1.875°	*lat(i,j)* × *lon(k,l)*
Nor-ESM1	NorClim	1.8975° × 2.5°	*lat(i,j)* × *lon(k,l)*

Table 4.2. Estimated rainfall volumes from 100-year return period, 24-hour duration storms.

	Rainfall volume (mm)		
	2021	*2080, RCP 4.5*	*2080, RCP 8.5*
Lower 95% CI	147.44	155.95	160.41
Mean	226.83	262.92	297.14
Upper 95% CI	306.23	369.89	433.87

Table 4.3. Flood duration and flood volume increased significantly in two sectors of the city as well as in the combined sectors. Values represent R^2 values from linear regressions of flood duration and flood volume vs. wetland loss in each sector of the city. Presence of values indicates significant results. * indicates $p < 0.05$. ** indicates $p < 0.01$.

Flood duration

	2021			*RCP 4.5*			*RCP 8.5*		
	Lower 95%	*Mean*	*Upper 95%*	*Lower 95%*	*Mean*	*Upper 95%*	*Lower 95%*	*Mean*	*Upper 95%*
Southeast	0.83*	0.82*	0.77*	0.82*	0.78*	0.74*	0.81*	0.79*	0.73*
Barrios Bajos	0.97**	0.94**	0.87**	0.94**		0.72*	0.94**	0.88**	
Isla Teja									
Las Animas									
Combined sectors	0.85**	0.85**	0.83*	0.85**	0.75*	0.79*	0.85**	0.84*	0.77*

Flood volume

	2021			*RCP 4.5*			*RCP 8.5*		
	Lower 95%	*Mean*	*Upper 95%*	*Lower 95%*	*Mean*	*Upper 95%*	*Lower 95%*	*Mean*	*Upper 95%*
Southeast	0.85**	0.94**	0.96**	0.92**	0.95**	0.97**	0.94**	0.96**	0.98**
Barrios Bajos	0.89**	0.90**	0.91**	0.89**	0.90**	0.91**	0.89**	0.91**	0.91**
Isla Teja									
Las Animas									
Combined sectors	0.96**	0.98**	0.98**	0.98**	0.98**	0.99**	0.98**	0.98**	0.98**

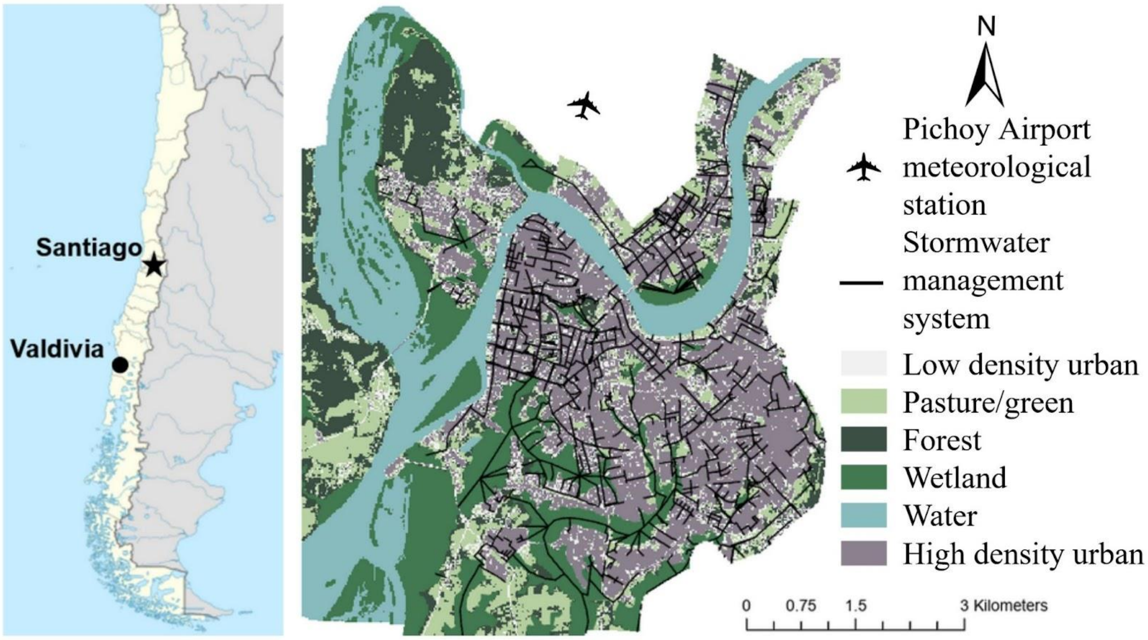

Figure 4.1. Left: Location of study site, Valdivia, Chile (39.8336 S, 73.2154 W). Right: Valdivia, and its land cover as delineated via spectral analysis of a 2010 orthophoto, drainage system, as described in 2012 by the Chilean Ministry of Public Works, and location of the Pichoy Airport meteorological station (39.656666 S, 73.087721 W; altitude: 18 m).

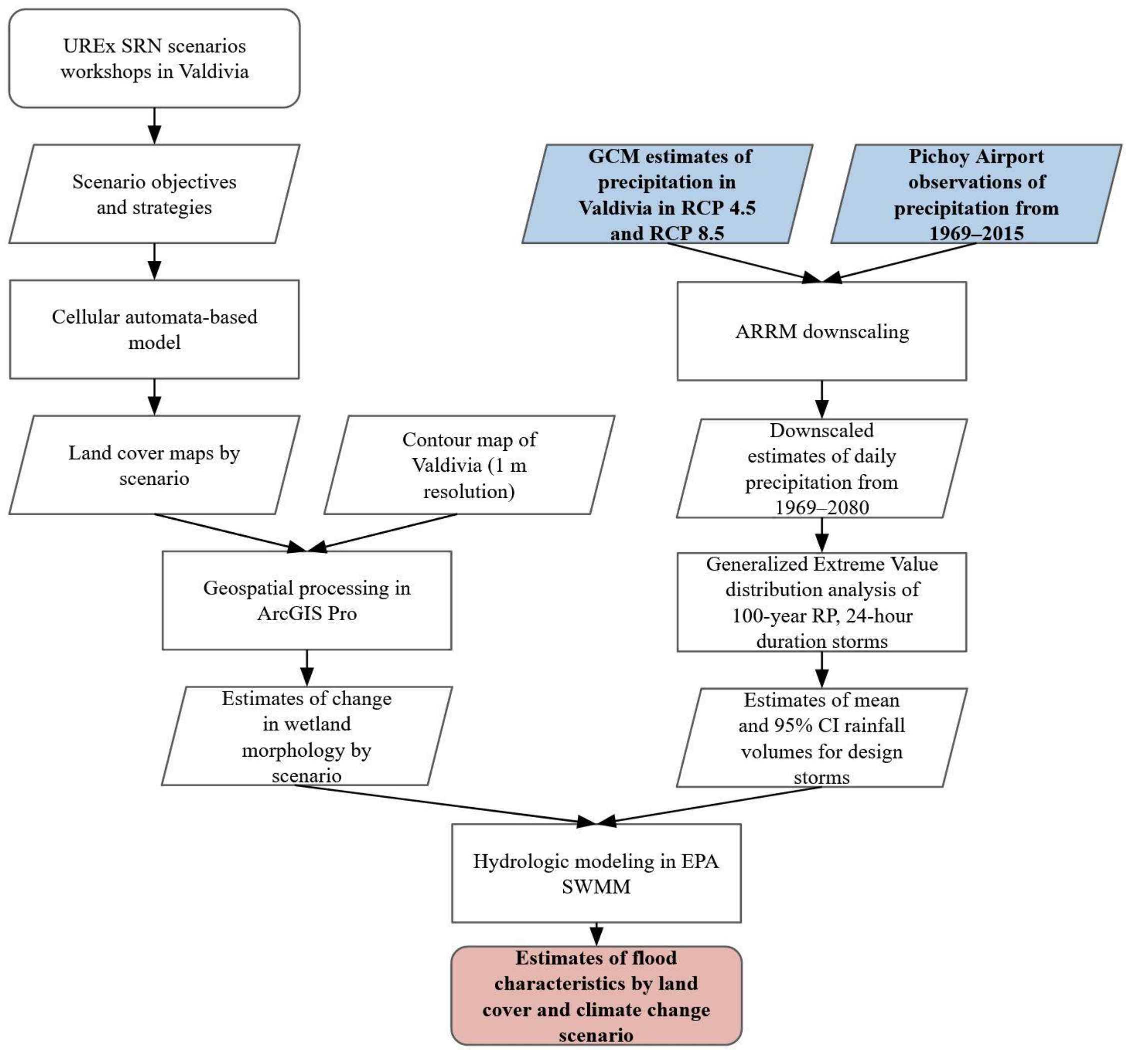

Figure 4.2. Process diagram detailing inputs and processes involved in this study. The left branch represents work that was accomplished in chapter 2, the model outputs of which were used in the present study. The right branch, bounded by the colored boxes, represents data and processes accomplished in the present study.

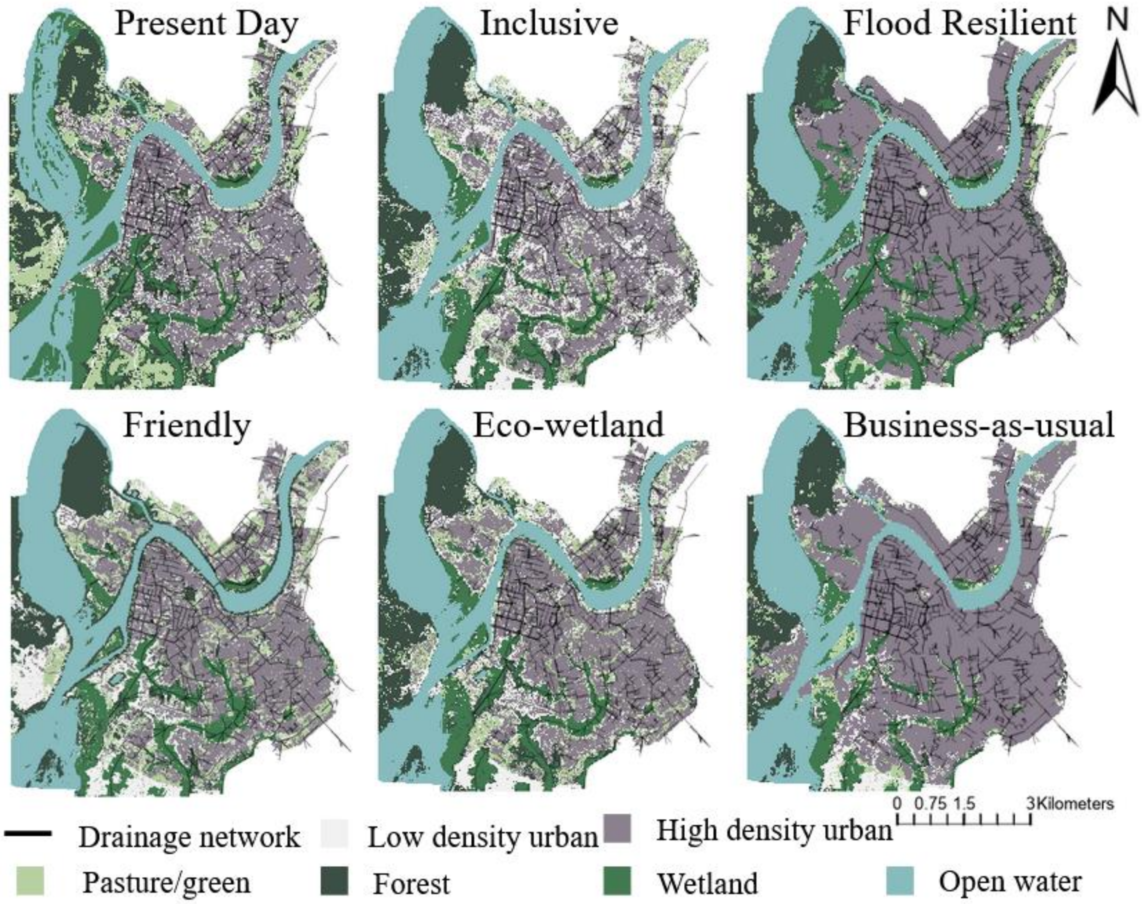

Figure 4.3. Land cover in the present day (2010) and under five scenarios of development by the year 2080. Wetland loss generally increases from left to right, and from top to bottom, compared to the present day. City-wide wetland loss for each scenario was: 9.72% in Inclusive, 13.3% in Flood Resilient, 18.3% in Friendly, 23.98% in Eco-Wetland, and 37.3% in Business-as-usual compared to present-day wetland coverage.

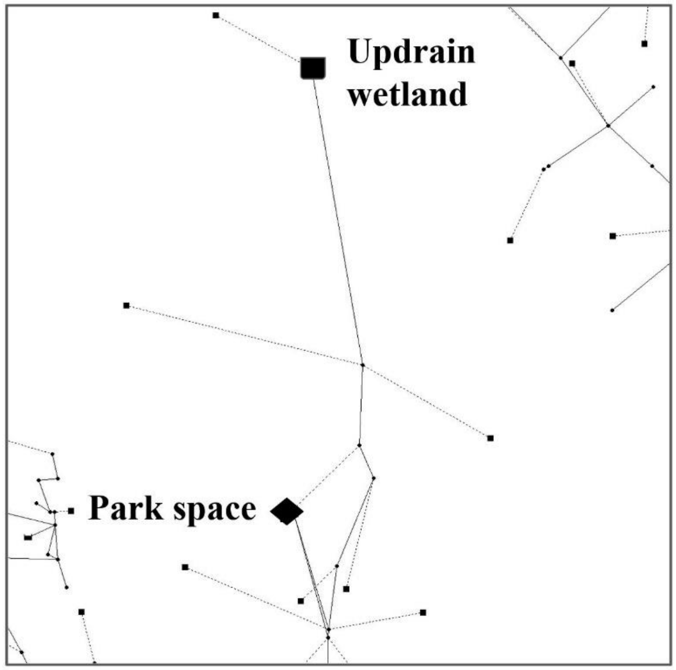

Figure 4.4. Subwatershed from the southeastern sector of Valdivia used for testing the effects of infiltration on flood volumes

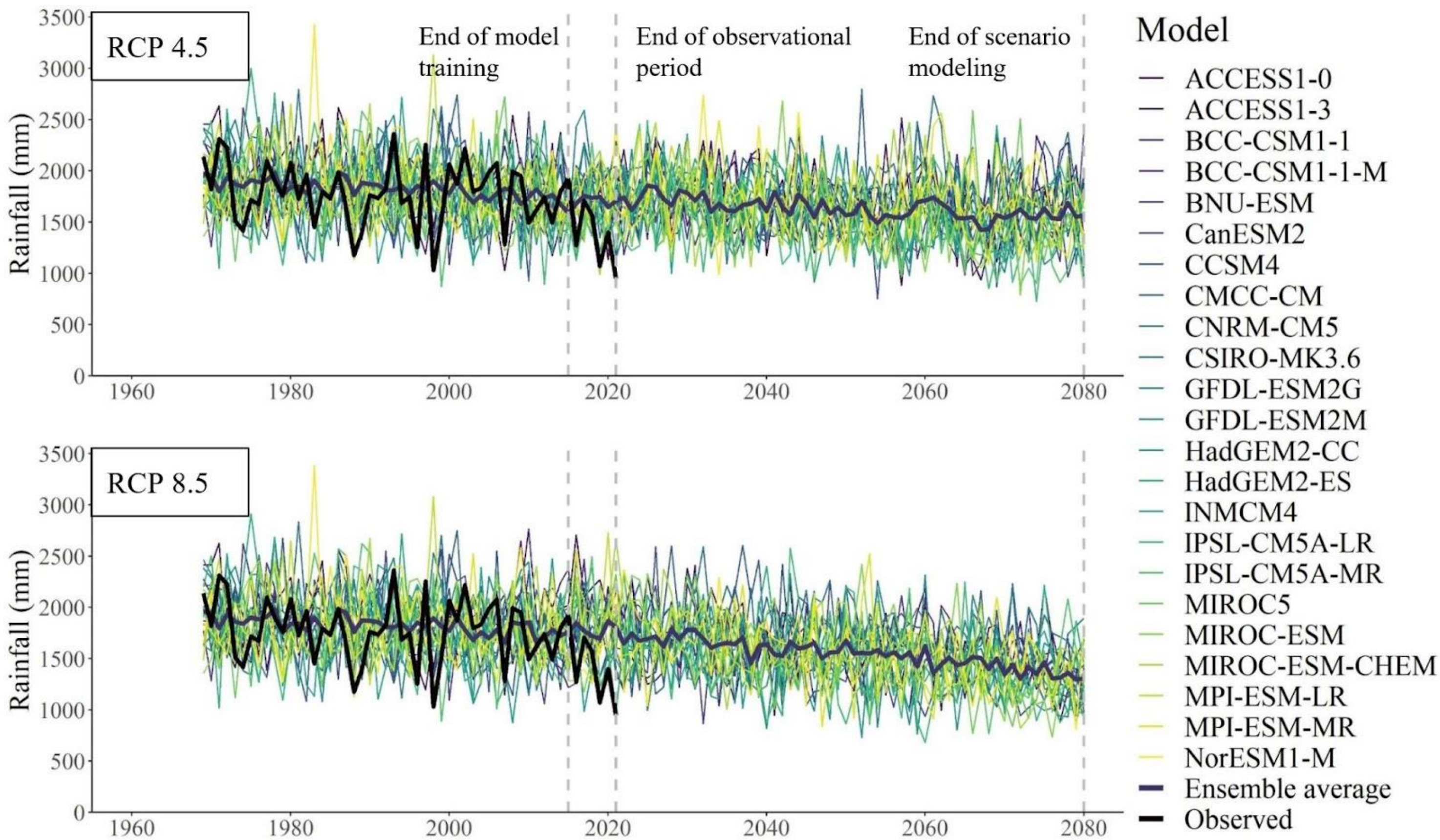

Figure 4.5. Annual rainfall volumes over time. Top: Results for the RCP 4.5 warming scenario. Bottom: Results for the RCP 8.5 warming scenario. The thick dark black line and band ending at 2021 represents the observed period data from Pichoy Airport station. The thick dark purple line and band represents the ensemble average mean and 95% CI for the models. Gray dotted lines represent starts and ends of important periods.

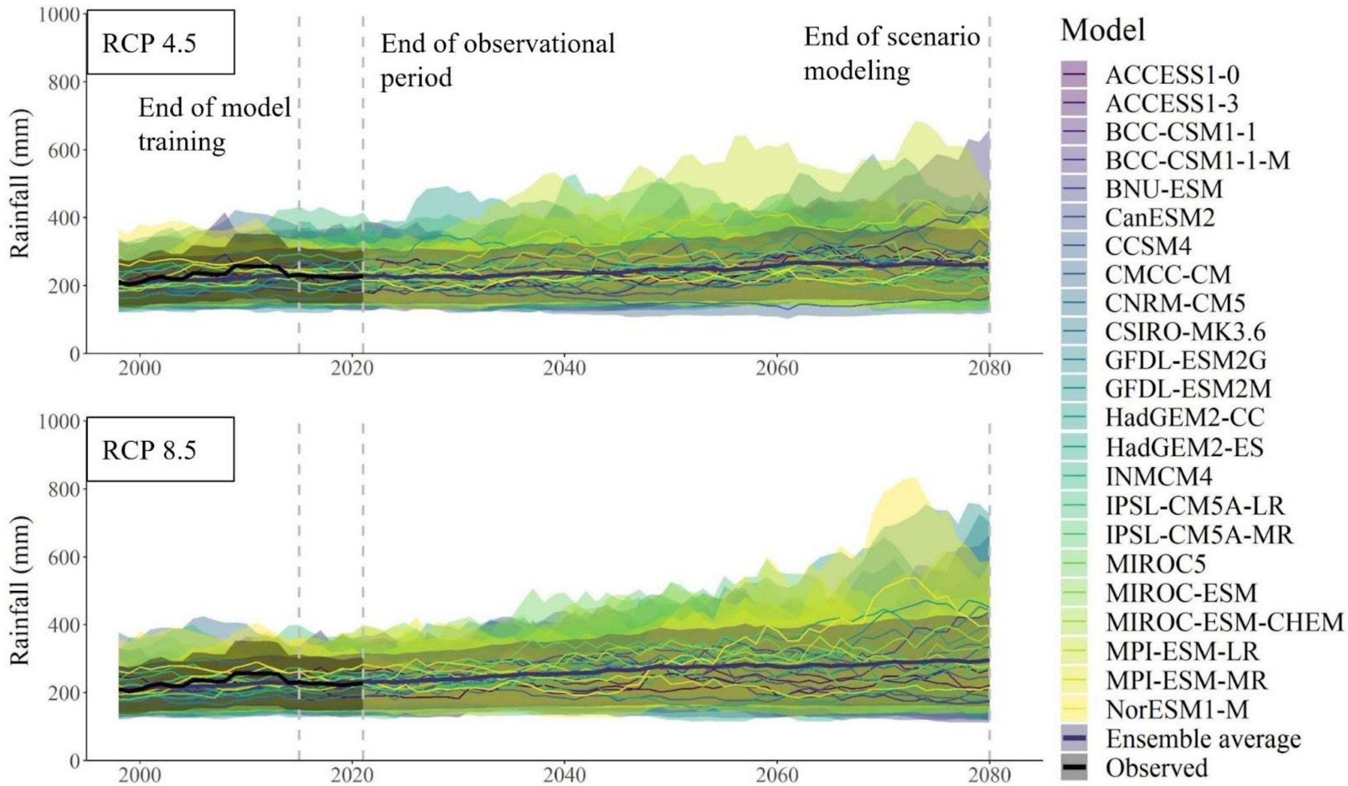

Figure 4.6. Mean (solid lines), and 95% confidence intervals (transparent bands) of 100-year return period, 24-hour duration storms over time. Top: Results for the RCP 4.5 warming scenario. Bottom: Results for the RCP 8.5 warming scenario. The thick dark black line and band ending in 2021, represents the observed period data from Pichoy Airport station. The thick dark purple line and band represents the ensemble average mean and 95% CI for the models. Gray dotted lines represent starts and ends of important periods.

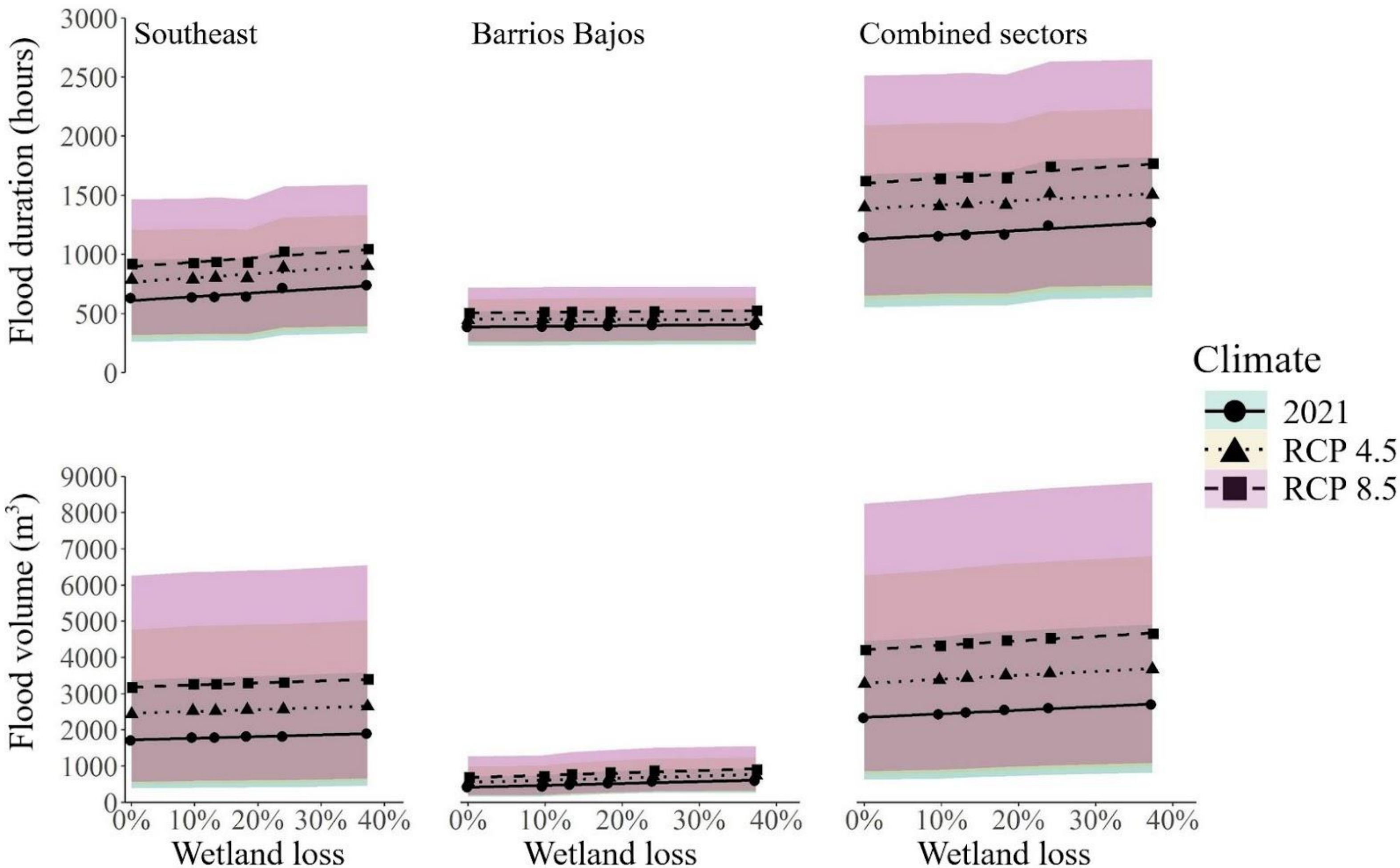

Figure 4.7. Flooding and wetland loss in different sectors of Valdivia where findings were significant. Values for the lower and upper extents of the gray bands represent flood volume for the lower and upper CI of rainfall volume in each climate scenario.

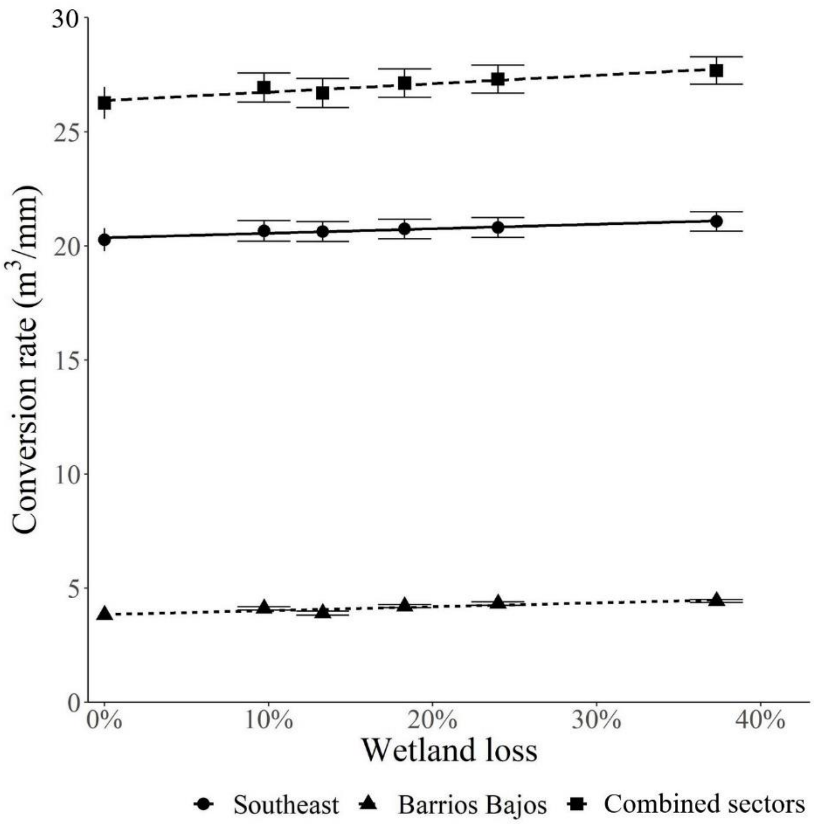

Figure 4.8. Rate of change in flood volume per unit of rainfall with wetland loss for sectors for which this trend was significant ($p < 0.05$).

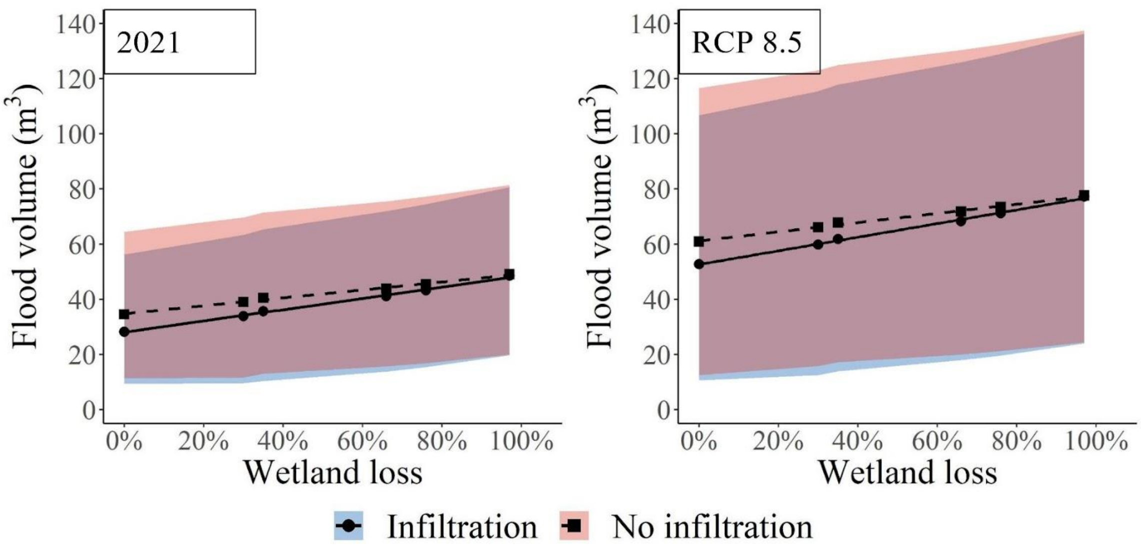

Figure 4.9. Total flood volume in the subwatershed using input rainfall volume from the lower CI, mean, and upper CI estimates of the 2021 and RCP 8.5 climate scenarios. Values for the lower and upper extents of the colored bands represent flood volume for the lower and upper CI of rainfall volume in each climate scenario.

CHAPTER 5

PLUVIAL FLOOD MODELING WITH THE ARC-MALSTRØM METHOD AND GROUND TRUTHING IN HERMOSILLO, MÉXICO

5.0. Abstract

Pluvial flooding is of increasing concern to cities contending with the dual challenges of climate change and development. Cities often estimate their flood risk using one or several relatively simple flood models that capture very few of the processes by which they might flood, such as through runoff generation or overland flow. I compared estimates of flood risk from three methods—the classic rational method in HEC-RAS, the relatively new Arc-Malstrøm method in Python, and a sketch mapping form of participatory mapping—to examine whether and how these easy-to-deploy methods might improve upon flood estimation in Hermosillo, Sonora, México. I conducted surveys on flood risk perceptions and a participatory mapping exercise with a total of 87 respondents across Hermosillo, 85% of which were at the respondent's place of employment. The survey included questions about perceptions of city-wide and survey-area flood risk, specific impacts from flooding, and flood depth and duration. This allowed examination of the relationships between responses, as well as the characteristics of salient flooding events and the potential biases of respondents during participatory mapping exercises. I found less than 4.60% average overlap between the estimated areas of flooding in all three methods, 39.0% average overlap between the rational and the sketch mapping method, 9.66% average overlap between the Arc-Malstrøm and sketch mapping methods, and 7.25% overlap between the Arc-Malstrøm and rational methods. I

also found that perceived flood risk was likely to be higher with respondents who had experienced impacts to commerce and their commute over other categories of impacts such as vehicles, structures, personal belongings and persons, and all other impacts. Though my responses in the sketch mapping exercises were biased toward reporting on street flooding, I nonetheless found it to be a useful tool to identify areas of flood risk that have an actual impact on citizens. Additionally, I found that sketch mapping is a useful method to use to foster transdisciplinarity and transformative science through its easy deployment and ability to engage traditionally excluded populations from flood resilience planning.

5.1. Introduction

The frequency of pluvial floods is expected to increase with climate change-driven increases in the intensity and frequency of rainfall in cities across the globe (O'Donnell & Thorne, 2020), and from the replacement of natural landscape features with synthetic ones as part of dominant patterns of urbanization (Lashford et al., 2019). Pluvial flooding is surface ponding or overland flow that occurs when rates of precipitation exceed the capacity of drainage systems and/or surfaces to remove it (Falconer et al., 2009; Rosenzweig et al., 2018). Pluvial flooding can lead to loss of life, damage to property, and disruption of transportation networks (Chang et al., 2010; Douglas et al., 2010; Falconer et al., 2009; Yin et al., 2016). It is also a global issue; for example, severe pluvial flooding accounts for 40% of losses from weather events in the UK (Douglas et al., 2010), and affects more than 200 cities across China (Jiang et al., 2018).

Pluvial flood modeling is critical to predicting flood locations and damages, even though model architecture and outputs have their limitations, particularly in urban areas. (Rosenzweig et al., 2020). Models in practice in urban watersheds are often two-dimensional and predict areas of elevated runoff (HEC-RAS), one-dimensional and predict flooding locations and volumes within urban drainage areas (EPA SWMM, Arc-Malstrøm (Balstrøm and Crawford, 2018)), and coupled 1D-2D models (PCSWMM) that predict flooding locations and even flood depths. Tu, Wadzuk, and Traver (2019) reviewed all of the major models currently of these types, such as GIFMod, MOUSE, and EPA SWMM, being used to simulate functions of stormwater management systems, with a special focus on those incorporating green stormwater infrastructure. They found that none of them had comprehensive abilities to model runoff generation, routing, GSI vadose zone/groundwater, evapotranspiration, and programmable control in urban settings. New, so-called "data-driven" models are of increasing interest among academics (Heonin et al., 2013; Penning-Roswell & Kordenwall, 2019; Wolfs & Willems, 2016) and offer advantages over these other forms of models insofar as they are computationally faster and not deterministically constructed (Li & Willems, 2020). The adoption of data-driven models by practitioners in cities is, in my experience, quite low, and older, more deterministic forms of flood modeling still dominate the field.

Participatory mapping has been identified as a non-modeling risk estimation approach that is crucial for successful disaster risk reduction (Mercer et al., 2009). Participatory mapping is common among development workers and researchers (Chambers, 2008; International Fund for Agricultural Development, 2009). It is also commonly used for community-based disaster risk reduction work (Twigg, 2004; Benson et al., 2007). Participatory mapping allows communities to identify hazard-prone and vulnerable areas, which may or may not be included in modeling estimates of flood risk. Cadag and Gaillard (2012) provide a review of the many forms of participatory mapping in the literature, which includes a form called “sketch mapping.” Sketch mapping involves respondents drawing on a provided map with pens or colored markers either points or shapes or both, indicating the dimensions and locations of a given disaster risk. It offers the advantages of relative ease of setup and low cost of execution, but also the disadvantages of being difficult to correct and adjust and that it is not georeferenced or scaled by the participants (Cadag & Gaillard, 2012). In urban areas, where flooding occurs among large populations of people, and where flood modeling is particularly difficult (Rosenzweig et al., 2020), sketch mapping exercises may be able to make up for the shortcomings of model estimations and to expand my understanding of the spatial characteristics of problematic flooding.

In this study, I compared three methods for flood risk estimation in a city, Hermosillo, Sonora, México, with one another: the rational method (HEC-RAS), the Arc-Malstrøm method, and the sketch mapping form of participatory mapping. Among the forms of flood risk modeling, the Arc-Malstrøm method (Balstrom and Crawford, 2018) is one of the newest, and as such its outputs have not been compared with other accepted methods for its potential overlaps and differences, nor has it been ground-truthed. I hypothesized that the rational method and the Malstrøm method overlap substantially, particularly along roadways and canals, and in large paved areas of the city like parking lots; I hypothesized that they would diverge on large, flat, and open areas of land, where infiltration might be high but which might still collect water during large storms. I also hypothesized that the participatory mapping method of flood estimation would overlap with both forms of flooding, but would have greater overlap with the Arc-Malstrøm method for its likelihood of identifying areas of pooled water.

Additionally, I wanted to examine several variables for their potential impact on the results of the sketch mapping form of participatory mapping, by pairing a participatory mapping exercise with a survey of participants regarding their flood risk perceptions and experiences. I hypothesized that people who had experienced greater numbers of impacts or more locations of flooding would rate flood risk as greater than those with fewer flood risk experiences or that had experienced flooding in fewer locations.

5.2. Methods

5.2.1. Study site

Hermosillo, population 810,000, is the capital city of the state of Sonora in México (Fig. 5.1). Hermosillo's climate is categorized as a hot desert climate according to the Köppen-Geiger climate classification system, with daily mean temperatures ranging from 24°C in winter to 32°C in the summer, with summer highs sometimes reaching 48°C. Hermosillo has a range of mean annual precipitation from 350 to 700 mm (Hallack-Alegria & Watkins, 2007), divided into two precipitation seasons: a mild rainy season in the winter, and a more extreme rainy season, driven the North American Monsoon (NAM) from the Gulf of California (Sea of Cortés) in the summer (Hallack-Alegria & Watkins, 2007). The majority of Hermosillo's annual precipitation arrives in short-duration, high-intensity events during the NAM (Vivoni et al., 2008).

The primary water source for Hermosillo is the Río Sonora, which enters the city after first being detained at a dam to the east of the municipal boundary, at about the middle latitudes of the city. The Sonoran River flows through Hermosillo via a cement canal, which serves as the end point for much of the city's stormwater management system. Stormwater is routed along the surface of the city's roads toward minor canals and ultimately to the Sonoran River canal. Toward these ends, many of the city's roadways are intended to serve as "calle canales," or street canals, and feature curbs that may exceed 1 meter in height in order to accommodate stormwater. Calle canales empty into drainage system openings that then flow through either surface canals or underground conduits to the Sonoran River. Thus, surface routing is a major form of stormwater management in Hermosillo and in most parts of the city, there is no drainage system alongside or underneath roadways to remove water draining from impervious surfaces.

5.2.2. Previous flood modeling and accounting work in Hermosillo

Hermosillo's Ministry of Public works hires contractors to assess flood risk in the city. A flood-risk map was produced by Hemek Ingeniería (Hemek), which is a corporation focused on hydrological engineering and stormwater management in Hermosillo, and is updated annually. In their assessment of Hermosillo's flood risk, Hemek primarily employed the HEC-RAS two-dimensional model, using a 100-year return period/24-hour design storm, and identified areas of high runoff (typically characterized by high slopes and impervious surfaces) and routing as areas of major flood risk. Hemek also conducts their own ground-truthing exercises, visiting sites of flooding reported to them by citizens and developers, and updates their flood risk assessment to reflect the presence of any of these areas not already included in their model outputs. Hemek provided to me their shapefile of flood-risk polygons in Hermosillo that resulted from their modeling and ground-truthing work.

5.2.3. Flood modeling with Arc-Malstrøm

To estimate areas where pluvial flooding is likely to occur during intense precipitation events, I employed the Arc-Malstrøm method developed by Balstrøm and Crawford (2018). The Arc-Malstrøm model (Balstrøm & Crawford, 2018) is a two-dimensional hydrological model that distributes precipitation along a network of topographic depressions. It is a hydrologically incomplete but simple and quick method to estimate areas of flood risk, relative to other forms of hydrodynamic modeling (Zhao, Balstrøm, Mark, & Jensen, 2021). It has been used to assess surface flooding depths and locations in areas with abundant impervious surface, such as cities, and intense rain events such as cloudbursts and monsoon storms (Balstrøm & Crawford, 2018; Hamstead & Sauer, 2021). The input value for precipitation for Hermosillo, based on estimates provided by Hemek, was 98 mm for a 100-year return period, 24-hour duration storm.

The Arc-Malstrøm method assumes that the rainfall rate exceeds rates of infiltration and evapotranspiration in the landscape, and also the rate at which any drainage infrastructure can effectively remove water from the surface. As such, the Arc-Malstrøm method is most accurate when modeling pluvial flooding that occurs as a result of very intense (e.g., of 100-year return period or more) storms, such as monsoon and cloudburst events. Arc-Malstrøm produces more accurate estimates of pluvial flooding in areas where there is very low infiltration, such as urban areas with extensive impervious surfaces, and less accurate estimates in areas with natural land-cover types or engineered land covers that promote infiltration.

The original Arc-Malstrøm model was designed for use in Python 2.7 (Python) environments and has not been maintained to function on more recent Python releases. In order to run the model on more recent Python architecture, I used the Septima fork (Septima.dk & Balstrom, 2020) of the original Arc-Malstrøm repository (Septima.dk & Balstrom, 2016), which as of June 2020 functioned without issue on Python 3.6.

5.2.4. Survey instrument and participatory mapping

Our survey instrument consisted of 10 questions (Appendix C). It began with questions about flood-risk perceptions of individuals, and whether or not they have experienced flooding in their neighborhoods (Appendix C). In the case that I was interviewing an individual in their home, they were answering questions about the neighborhood in which they live. If I were speaking to an individual in their workplace, they were answering questions about the neighborhood in which they worked. All survey respondents fell into either the “Home” or “Work” category and were not simply visiting or passing through the survey area. The location in which the survey was conducted was also recorded.

Respondents were also asked to perform sketch mapping of flooding in their locality, a form of participatory mapping that has respondents indicate flood locations (points), flood dimensions (area), or both. In the case that a respondent indicated a point of flooding, they were asked to instead outline the area of flooding, but not all respondents complied. Respondents were further asked to indicate, on a scale of 1-5, where 5 is the highest value, the level of danger they felt that flooding represented in the areas they drew. Additionally, respondents were asked to describe the kinds of impacts they experienced, and for how long the impacts they suffered lasted. Respondents were also asked if they had flood insurance. Finally, respondents were asked to provide their primary form of transportation. Respondents who indicated that they had not experienced flooding in the survey area were all assigned a “1” for the level of danger that they perceived in the area.

I selected six areas in Hermosillo to conduct these surveys (Fig. 5.1). Areas were selected for either having substantial flood risk according to rational method or Malstrøm models. Survey areas were all roughly 1 km in diameter, except for one that was extended in order to get enough interviews. Survey areas were located in areas with mixed commercial, residential, and industrial zoning. All questions and answers were originally in Spanish, and have been translated by the interviewer (the lead author) for this study. Exposure, or the time in which the respondent participated in the participatory mapping and survey exercise, averaged approximately 20 minutes. However, due to many interviewees being at work during the interview, and often the sole employee at work, this exposure may have been in non-consecutive portions of time.

Participatory mapping results were later digitized by drawing the flood areas depicted on the respondents’ maps in ArcGIS Pro (ESRI) using the Create tool. A comparison of the key features of the three flood-area estimation methods (rational, Malstrøm, and participatory mapping) are included in Table 5.1. Flood areas identified through participatory mapping were also coded according to the characteristics of the identified areas: streets, critical infrastructure, buildings, and general areas.

5.2.5. Analysis of relationships between flood risk perception, flooding impacts and characteristics, and flooding locations

In order to draw relationships between binary responses to questions (e.g., “Yes” to the question, “do you believe that flooding is a problem in this area?”) and other responses, I employed binary logistic regression analysis in R (R Core Team, 2021). Binary logistic regression is used to model the relationship between a binary target and any number of independent variables, either qualitative or quantitative. The general formula for logistic regression is:

$$logit(p) = b_0 + b_1X_1 + b_2X_2 + \ldots + b_kX_k \tag{1}$$

Where

p represents the odds ratio

b_k represents the coefficient of variable *k*

X represents the value of variable *k*.

The odds ratio (OR) represents the probability of the presence of a variable divided by the probability of the absence of a variable. After screening predictor variables to verify no violation of the assumption of the linearity of the logit, I used binary logistic regression to relate question 1 (Q1: “Do you think flooding is a problem in Hermosillo?”) to question 2 (Q2: “Do you think flooding is a problem in this area?”) and the other predictor variables:

$Q1 \sim Q2 + Impact_{Commerce} + Impact_{Commute} + Impact_{Structure} + Impact_{Vehicle} + Impact_{Persons} + Impact_{Other} + FloodDepth + FloodDuration + OrdinalRisk + TransportType + FloodLocation_{Street} + FloodLocation_{CriticalInfrastructure} + FloodLocation_{HomeBusiness} + FloodLocation_{GeneralArea}$

Similarly, I related Q2 to my variables using binary logistic regression according to the following equation:

$Q2 \sim Impact_{Commerce} + Impact_{Commute} + Impact_{Structure} + Impact_{Vehicle} + Impact_{Persons} + Impact_{Other} + FloodDepth + FloodDuration + OrdinalRisk + TransportType + FloodLocation_{Street} + FloodLocation_{CriticalInfrastructure} + FloodLocation_{HomeBusiness} + FloodLocation_{GeneralArea}$

To relate ordinal perceptions of flood risk (Question: “On a scale of 1 to 5, how dangerous do you believe flooding to be in this area?”) to my other survey variables using ordinal logistic regression in R (R Core Team, 2021). After screening predictor variables to verify no violation of the assumption of the linearity of the logit, I used ordinal logistic regression to relate ordinal perceived danger of flooding (on a scale between 1 and 5, where 5 is the most perceived danger) to my predictor variables according to the following equation:

$OrdinalRisk \sim Impact_{Commerce} + Impact_{Commute} + Impact_{Structure} + Impact_{Vehicle} + Impact_{Persons} + Impact_{Other} + FloodDepth + FloodDuration + TransportType + FloodLocation_{Street} + FloodLocation_{CriticalInfrastructure} + FloodLocation_{HomeBusiness} + FloodLocation_{GeneralArea}$

5.3. Results

5.3.1. City-wide flooding

Flood models from the rational method (the HEMEK flooding estimates) and Malstrøm methods differed substantially in terms of total area estimated to be at risk of flooding (Table 5.2) as well as the locations where that flooding was projected to occur (Fig. 5.2). HEMEK estimates 3,717.8 ha at risk of flooding (21% of Hermosillo's municipal area), whereas the Malstrøm model estimates 730 ha of flood-risk areas (4.2% of municipal area); only 209 ha of flood-risk areas between the two models overlap (1.2% of municipal area). Areas at risk of flooding according to the rational method were largely along waterways and major arterials in the city; areas at risk of flooding according to the Malstrøm model also included many major waterways, particularly the Río Sonora which runs through the middle of the city, but also along historical waterways not included in the waterways polyline file. In many locations, the Malstrøm model identified large parking lots, roofs, and intersections as areas likely to accumulate flood water.

5.3.2.1. Survey results

I successfully conducted 87 interviews across six sites. Because of local customs regarding answering doors and construction practices for residences, access to homes was difficult or impossible in most locations. Consequently, 85% (n = 74) of all surveys (N = 87) were conducted at business locations with business owners, business employees, or designated representatives for businesses. Surveys conducted at homes with residents represented 13% (n = 11) of all surveys. Surveys conducted at locations serving as both businesses and homes represented 2% of all surveys (n = 2).

5.3.2.2. Participatory mapping and comparisons with other methods of flood area estimation

The six areas surveyed comprised 573.80 ha, or roughly 3.27% of the urban area (Fig. 5.1). Within the survey areas, the rational method estimated 146.80 ha of flooding, or 25.5% of the survey areas; the Malstrøm estimated 40.30 ha of flooding, or 7.02% of the surveyed area; and participatory mapping estimated 64.40 ha of flooding, or 11.22% of the surveyed area (Table 5.2). For each site except 6, where there was no flooding according to the rational method, there was generally more overlap between the participatory-mapping flooding and the rational-method flooding than the Malstrøm-method flooding. In any given site, overlap between participatory, rational, and participatory methods was 15.5% or less of total flood areas identified by the participatory-mapping method in survey areas, and averaged across all sites was 4.98%. Overlap between flood areas estimated by the rational and Malstrøm methods was less than 15% of rational-method flood areas in survey areas, or 6.04% on average, indicating that these methods generally identified unique flood-risk areas (Table 5.2). Areas that respondents identified as areas of flooding were divisible into five categories: streets, critical infrastructure buildings, businesses, and general area (Fig. 5.2). The number of responses in each category were 85, 1, 2, and 14, respectively, with 102 total flood areas identified in the exercise.

5.3.2.3. Impacts of flooding

I determined six categories into which survey responses about impacts of flooding fell: commerce, commute, structural, vehicle, persons, and other inconvenience (Table 5.3). Impacts in the first five categories represented 97.8% of all impacts provided by respondents. One impact, "car collisions with buildings" was included in the category of "damage to structures and infrastructure" rather than "damage to vehicles" because the emphasis from the respondent was the impact to the building, and also to avoid creating multicollinearity between categories.

5.3.2.4. Relationships between survey variables and perceived flood risk

Logistic regression revealed that respondents were more likely to say that flooding was a problem in Hermosillo if they had experienced flooding in the survey area (OR = 6.10; 95% confidence interval (CI): 1.11, 46.44; $p < 0.05$). No other factor was significantly correlated with city-wide flood risk perception.

Survey respondents were more likely to report having experienced flooding in the survey area if they were impacted by flooding in the categories of commerce (OR = 25.9; 95% CI: 4.64, 543; $p < 0.05$) and commute (OR = 8.82; 95% CI: 2.19, 70.7; $p < 0.05$). Survey respondents with greater numbers of any category of impact (OR: 18.2; 95% CI: 5.69, 98.38; $p < 0.05$), and with multiple categories of impacts (OR: 87.2; 95% CI: 5.69, 98.4; $p < 0.05$) were more likely to report flooding being an issue in their survey area. The perceived danger of flooding posed to respondents in the survey area increased with the number of flooding impacts to commerce (OR = 2.61; 95% CI: 1.71, 4.10; $p < 0.05$), commute (OR = 1.77; 95% CI: 1.09, 2.95; $p < 0.05$), vehicles (OR = 3.02; 95% CI: 1.21, 7.77; $p < 0.05$), and persons (OR = 3.40; 95% CI: 1.58, 7.54; $p < 0.05$). This was also true with increasing flood depth (OR = 36.7; 95% CI: 7.89, 232; $p < 0.05$), but not with increasing duration of flooding. Perceived flood risk was not significantly related with any particular category of flood location identified in the participatory mapping exercise, or the total number of elements identified, but it was generally higher with greater numbers of flood location categories (OR = 18.5; 95% CI: 5.70, 75.8; $p < 0.05$).

Greater durations of flooding were correlated with more impacts to commerce (Spearman's Rank correlation, $\rho = 0.298$, $p < 0.05$) and to the total number of impacts ($\rho = 0.365$, $p < 0.05$), but not to location types or total locations of flooding identified by participatory mapping. Greater depth of flooding was also positively correlated with more impacts to persons ($\rho = 0.375$, $p < 0.05$) and to the total locations of flooding identified by participatory mapping ($\rho = 0.313$, $p < 0.05$).

5.4. Discussion

This study compared flooding characteristics and overlap among three methods of risk estimation. Although all three methods produced overlapping areas of flood risk, they identified unique areas at risk of flooding in the majority of cases, indicating that the use of multiple methods of flood estimation will increase the likelihood of estimating areas of flooding that impacts urban residents. Overlap between the rational and Arc-Malstrøm modeling methods of flood estimation and the participatory mapping method was highly variable in survey areas, with more overlap between the rational method and the participatory mapping method than the Arc-Malstrøm method and the participatory mapping method in each survey area and on average. Thus, in the absence of participatory mapping in a city, the use of multiple models may increase the likelihood of identifying areas that are problems for people. However, the use of either or both is unlikely to identify all or even most of the areas of flooding that pose salient risk to urban residents.

Participatory mapping and surveys may provide an incomplete picture of flood risk in a city, even in relatively small survey areas (~1 km in diameter). Since respondents were more likely to identify flooding as a problem if they were impacted in the areas of commerce or commute, it may be that participatory mapping is not effective in ground-truthing areas identified by modeling methods where people are not being directly affected by flooding—perhaps particularly in these two categories of impacts. Also, respondents over-emphasized streets as locations of flooding in the participatory mapping exercise (83% of all identified areas of flooding) relative to the number of reported impacts about disruptions in commutes (30.7% of all reported impacts). Further, while respondents were able to identify flood areas that affect access to and function of transportation infrastructure (streets and general area location categories), they did not identify flooding at the sites of any critical infrastructure that would be related to impacts like loss of electricity (7.39% of all reported impacts).

Analysis also indicated that the number of areas identified as at risk of flooding is significantly correlated with the amount of time that flood waters stay present in the area, such that areas with more transient flooding were less often identified in participatory mapping. The areas identified in participatory mapping may then be limited in scope: biased toward flooding that lasts for longer and in areas that affect people's abilities to ply their trades and move about the city. That said, mapping and surveys were done typically at businesses, and respondents in this study may be biased to answer along these lines given their locational and personal identities. Unfortunately, the sample size of surveys and mapping exercises done at locations that were not businesses was too small to test for biases within these groups with any certainty. The literature explorations is silent onthese sorts of biases in participants or responses for sketch mapping (Cadag & Gailard, 2012) or related forms of participatory mapping (Dennis, Gaulocher, Carpiano, & Brown, 2009; Schumann, Binder, & Greet, 2018), yet identification of such biases may be critical to understand the utility and scope of such methods.

In the context of desert cities such as Hermosillo where precipitation is infrequent, survey responses and participatory mapping may also be influenced by relationships between time and risk perception. I conducted my study in June and July, just before the annual monsoon season started; consequently, respondents were speaking to flood risk perceptions and remembered locations of flooding from storm events at minimum nine or ten months prior to the time of the survey. Impactful flooding areas may then have been forgotten or remembered as less severe at the time of survey, and a different response may have been received if the survey were done at the time of or just after the monsoon season. Additionally, in areas where flooding is only a potential inconvenience for a few hours of a few days in a very limited season, flood risk may not be remembered as severe, even if respondents felt that flooding was severe during the period in which it is occurring. This may be complicated further by the severity of impact, such that a lesser impact like needing to move cars from streets from flooding three years ago is not so memorable, but having your home collapse due to flooding any time in a respondent’s past is quite memorable. Indeed, previous scholarship on the cognitive aspects of risk perception revealed that the nature and severity of the damage experienced can affect flood-risk perception (Bubeck, Boltzen, & Aerts, 2012; Takao, Motoyoshi, Sato, & Fukuzono, 2004). However, I identify a clear need for more temporal considerations of risk perception–particularly of instrument-specific relationships between time and risk perception that are pertinent for exercises such as participatory modeling.

Nonetheless, there remain compelling reasons to pursue the sketch mapping form of participatory mapping in order to verify or extend upon estimated flood risk areas according to hydrological models. Participatory mapping in general may help to achieve various government agencies' goals of obtaining more active community stakeholder engagement in hazard preparedness and risk reduction efforts (Federal Emergency Management Agency, 2016; United Nations, 2015), although México's Centro Nacional de Prevención de Desastres (CENAPRED) in particular does not call for this (Salinas & Espinosa, 2004). Participatory mapping may also be considered a part of transdisciplinary work, which emphasizes the inclusion of knowledge from non-academics (Scholz, 2017) and the active inclusion of "civil society actors as an external corrective on the 'blind spots' in the science system" (Schneidewind, Singer-Brodowski, & Augenstein, 2016). Sketch mapping in particular is a method with broad potential participation by the public, including marginalized groups with low scientific training but high local experience and expertise, such as youths and the illiterate (Cadag & Gaillard, 2012). These group members participate through the use of simple tools and without complex instruction. At minimum, my work has demonstrated the utility of sketch mapping toward revealing some of these blind spots in hydrological modeling. If science is to be a tool for change in cities, methods such as sketch mapping will help resolve some of the weaknesses of other forms of flood risk management.

Sketch mapping has a much lower barrier to deployment and participation than other methods of collecting citizen knowledge of hazards, such as photovoice (Schumann, Binder, & Greer, 2018), given that materials are cheap (GIS software, pens and paper) and the time required to complete a participatory mapping exercise and survey is relatively short (~20 minutes on average for this study). For the respondents in this study, most of whom were either at their place of employment or were the employers themselves, lack of time was the most frequent barrier to participation or completion. Yet it is still less of a commitment than these other methods like photovoice (Schumann, Binder, & Greer, 2018) or workshops (Shrestha, Kröckler, Flacke, Martinez, & van Maarseveen, 2017), though these other forms may benefit from the inclusion of other processes such as social learning and knowledge co-production. Additionally, governmental organizations like FEMA recognize the important roles that the private sector can play in disaster preparation and recovery, emphasizing that the need for the business community's involvement in pre-disaster planning in order to reduce risk and retain jobs, goods and services, and a stable tax base for cities (Federal Emergency Management Agency, 2016). Sketch mapping may be a way to bring in businesses to these resilience planning processes, or to reach businesses with owners or employees who cannot participate in forms of planning that require greater time commitments.

As for the accuracy of the Arc-Malstrøm model, which is relatively new among hydrological models being deployed in cities (Balstrøm & Crawford, 2018; Hamstead & Sauer, 2021; Pallathadka, Sauer, Chang, & Grimm, 2022) and developed further in the academy (Thrysøe et al., 2021; Zhao, Balstrøm, Mark, & Jensen, 2021), much examination remains to be done. While my study revealed that the overlap between estimated flood areas of Arc-Malstrøm, the rational method, and the participatory mapping method were low, thereby indicating the potential of Arc-Malstrøm to reveal areas of flooding not readily identified by the other two methods, the veracity of these areas unique to Arc-Malstrøm being at risk of flooding was not evaluated. In an earlier version of my proposed study, I had envisioned respondents first identifying areas of flooding in the same manner as was done here, to be followed by the presentation of a map showing the Arc-Malstrøm estimated areas of flooding in the survey area. Respondents would then be asked to identify areas that they could, from personal experience, say flooded or did not flood, and to what extent, among other characteristics. However, this second map was deemed too much of a time burden for the format in which my surveys and mapping exercises were conducted and abandoned in the final form. Thus, I do not feel that my present study should be used to decide the ultimate veracity of the Arc-Malstrøm method. However, based on the overlap between methods, and from expert verification by Hermosillo’s Hemek engineering firm in some locations, Arc-Malstrøm appears to have some degree of accuracy and modeling value in this urban context.

Table 5.1. Key model features of the rational, Malstrøm, and participatory mapping methods of flood estimation. “O” indicates that a feature is optional.

	Method		
Model consideration	*Rational*	*Arc-Malstrøm*	*Participatory*
Slope	Y	Y	N
Curve number	Y	N	N
Stormwater management system flow	N	N	N
Groundwater flow	N	N	N
Catchment scale	Y	Y	N
Evapotranspiration	N	N	N
Community knowledge	O	O	Y

Table 5.2. Estimated flood areas and flood area overlaps in each of the six surveyed sites for all flood area estimation methods. R = rational method; M = Malstrøm method; P = participatory mapping method.

Site number	**R flood area (ha)**	**M flood area (ha)**	**P flood area (ha)**	**P-R-M Overlap**	**P-R Overlap**	**P-M Overlap**	**R-M Overlap**
1	24.8	16.7	3.06	15.5%	69.6%	27.6%	14.2%
2	27.2	5.45	12.8	1.43%	17.3%	11.1%	4.15%
3	21.1	1.23	1.70	1.87%	42.3%	2.95%	1.87%
4	42.7	11.6	12.5	2.68%	28.7%	7.13%	14.5%
5	30.6	0.931	9.56	1.48%	36.9%	2.36%	1.53%
6	0.00	4.56	24.8			6.87%	
Sum	146	40.3	64.4				
Average	29.3	6.72	10.7	4.60%	39.0%	9.66%	7.25%

Table 5.3. Survey responses of flood impacts and their parent categories. Number of responses indicating impact in parentheses, of 176 total responses.

Commerce (73)	**Commute (54)**	**Damage to structures and infrastructure (4)**
Interruption of custom (25)	Cannot walk (16)	Flood damage to office building (1)
Damage to merchandise (8)	Difficulty walking (3)	Canal collapse (1)
Loss of electricity (12)	Cannot drive (19)	Car collisions with buildings (1)
Damage to computers (1)	Difficulty driving (6)	Ceiling leaks (1)
Must relocate business (1)	Delay in commute (3)	
Cannot work (1)	Buses delayed (1)	
Business floods (19)	Traffic delayed (2)	
Must close business (2)	Cannot cross street (4)	
Damage to office furniture (4)		
Damage to vehicles (18)	**Damage to persons and personal property (21)**	**Other inconvenience (4)**
Cars lifted by flood water (7)	Person carried away (1)	Flash flooding (1)
Cars damaged (1)	Damage to personal clothing (1)	Emergence of rats (1)
Car collisions with cars (4)	Pay to pump water from home (1)	Emergence of roaches (1)
Car collisions with potholes (3)	Damage to home furniture (4)	Drainage backs up (1)
Must move car from street (3)	Home floods (11) Home collapsed (2)	
	Loss of electricity in home (1)	

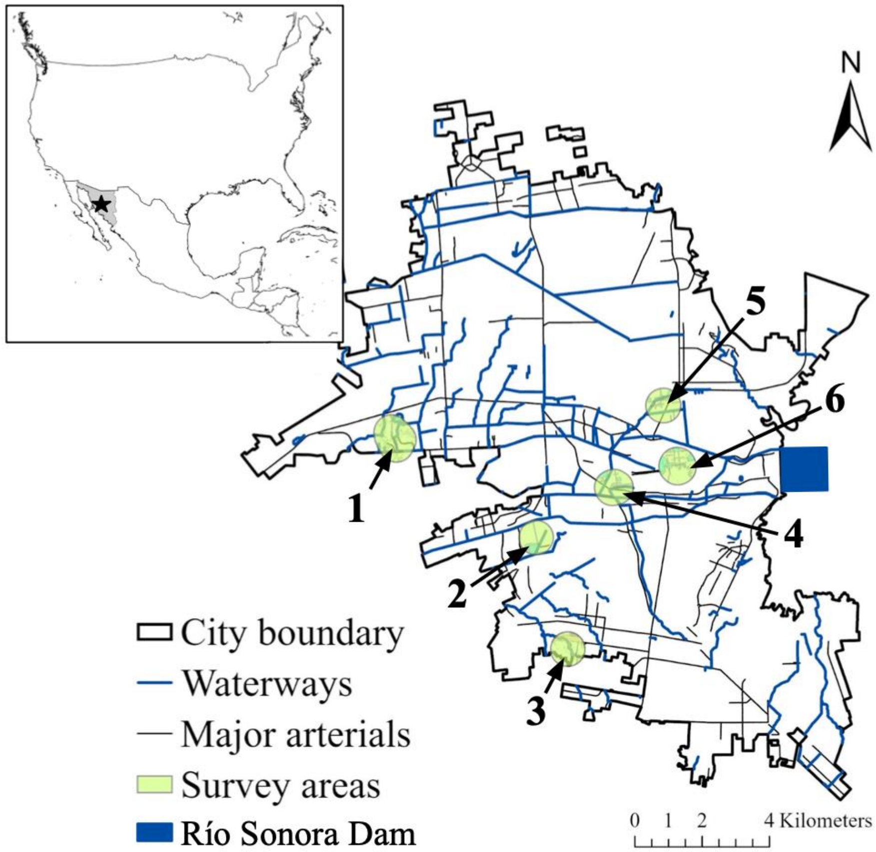

Figure 5.1. Upper left: Hermosillo in the context of North America and the Mexican state of Sonora. Lower right: City of Hermosillo overlaid with major waterways, roadways, and different methods of flood estimation.

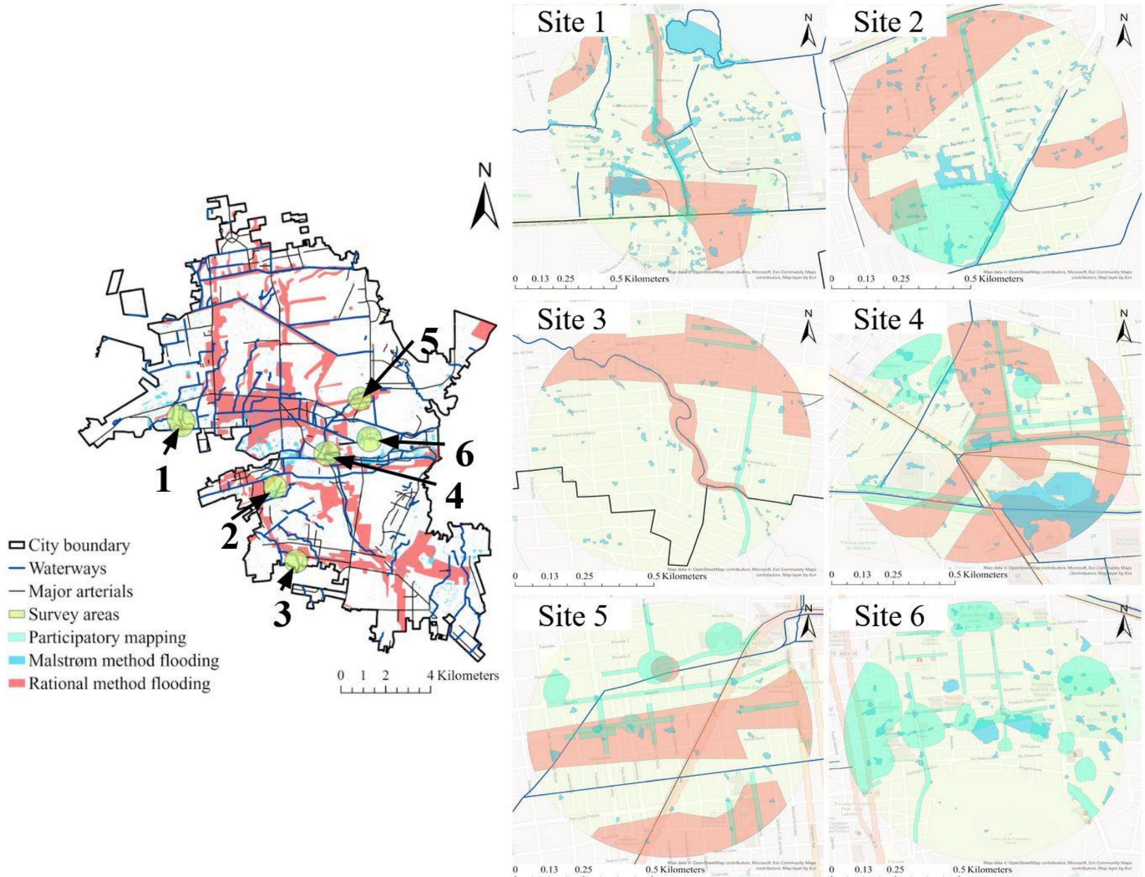

Figure 5.2. Left: Locations of survey sites in Hermosillo. Right: Detailed view of identified areas of pluvial flooding from participatory mapping, and estimated areas of pluvial flooding from the Arc-Malstrøm and rational methods.

CHAPTER 6

CONCLUSION

6.1. Summary

Pluvial floods present threats to the livelihoods and lives of urban communities across the globe, but this threat is manageable under appropriate research direction and development decisions. This dissertation highlights the interactions between internal changes in wetland UEI, stormwater management systems, changes in urban land cover, changes in climate, and the methods I use to assess pluvial flood risk. The work displayed here was interdisciplinary, employed mixed methods, and explored pluvial flood risk at multiple scales to expand the ways in which pluvial flood risk can be understood by researchers and lay people alike. In this final chapter, I will conclude with a summary of key findings, synthesize the view of this dissertation toward pluvial flood risk assessment, and offer compelling pathways forward for research.

In chapter 2, I demonstrated the effects of a range of antecedent storage conditions in six wetlands on flood duration and flood volume in Valdivia, Chile. I combined field measurements of surface and soil water storage across two campaigns, during which rainfall volumes were less than the climate average, with stormwater management models that characterized wetland storage in several ways. I found that flood duration and flood volume were positively correlated with wetland stage and soil storage, though the overall effect was small relative to total flood duration and volume. Allowing the model to account for infiltration increased the amount of flood mitigation, but this effect was also small relative to systemic flooding. Studies on the benefits of wetlands toward pluvial flood mitigation have so far not considered how the range of storage conditions that wetlands can exhibit, which are dependent on long-running climate and short-term seasonal storm conditions, will alter flood mitigation services. Daily changes in wetland storage were on the order of several centimeters, perhaps due to evapotranspiration (ET); thus, ET and storm timing may contribute to flood volume and duration in cities with wetland UEI. This research complicates broad statements about the universal benefits of wetlands in reducing flood volumes (Kadykalo & Findlay, 2016), as there appear to be other characteristics of wetlands that may reduce or increase flood risk.

In chapter 3, I examined how projected changes in wetland land cover and contributing watershed land cover affect pluvial flood risk in Valdivia. I used model estimates of future landcover in Valdivia, generated from the work of local decision-makers and leaders at UREx SRN scenarios workshops, to estimate changes in wetland and impervious cover using GIS software. These changes in land cover were then translated to changes in wetland dimensions in EPA SWMM using a novel methodology. I found that flood duration and volume increased with increasing wetland loss in some sectors of the city, such that the worst flooding occurred in scenarios that did not conserve wetlands within the city. Through modeling subsystems of the larger stormwater management system, EPA SWMM likely underestimated the effects of wetland loss on systemic flood volumes in portions of the stormwater management system. Further, in one subsystem, flood volumes decreased at locations downdrain of the wetlands rather than at the wetlands, indicating the impacts of wetland reduction may extend some distance into the network.

In chapter 4, I explored the combined effects of climate change and urban land-cover change on pluvial flooding in Valdivia. I examined downscaled climate data produced by UREx SRN researchers using the ARRM method for changes in 30-year-average annual rainfall volumes and 100-year, 24-hour storms in RCP 4.5 and RCP 8.5 warming scenarios. The data indicated diminished annual rainfall volumes but elevated rainfall volumes during 100-year, 24-hour storms. I used the ensemble average of the climate model estimates to create mean, lower and upper bounds of rainfall volumes for storms in 2080 under each warming scenario, and input these values as design storms into EPA SWMM. I combined future rainfall volumes with the EPA SWMM models used in chapter 2 to estimate the combined effects of climate change and land-cover change on flood duration and flood volume. In some sectors of the city, modeled wetland loss produced significant increases in flood duration and volume during all storm intensities. In other sectors, flood duration and volume responded significantly only to increased rainfall volume, and wetland loss had no detectable effect on flooding. In one subsystem of the city where I allowed for infiltration in the model, projected flood volume significantly increased with wetland loss under all rainfall volumes. These findings indicated that wetlands may provide significant and substantial flood mitigation services under a range of rainfall volumes, and thus can help to reduce flood risk under a range of climate conditions. However, this flood mitigation service diminished with wetland loss. Other factors likely determine whether and to what extent wetlands provide flood mitigation services in urban stormwater management systems, such as configuration of wetlands within the overall system.

In chapter 5, I explored the utility of a new form of pluvial flood risk assessment, the Arc-Malstrom method, in Hermosillo, Sonora, México, by comparing it to citizen assessments of pluvial flood risk generated through participatory mapping, as well as to existing projections of pluvial flood risk generated from rational method modeling. I conducted a survey and mapping exercise with 87 respondents to collect spatial information (e.g., location, size, depth) and impacts of pluvial flooding in the survey area. I found little overlap (maximum 16.0%; average 4.60%) of pluvial flood areas between the methods in any given survey area. Perceived flood risk was likely to be higher with respondents who had experienced impacts to commerce and their commute over other categories of impacts such as vehicles, structures, personal belongings and persons, and all other impacts. The Arc-Malstrøm and participatory mapping methods provided additional useful information on the pluvial flooding that was problematic to citizens. Perceived pluvial flood risk in the city can be managed through targeted reduction of particular areas where pluvial flooding occurs, such as in businesses and along walkways and intersections. Models do not by themselves indicate pluvial flood areas that should be of higher or lower priority to stormwater managers in cities, and so the methods in this chapter may be used by managers to prioritize intervention.

6.2. Synthesis

Pluvial flooding presents a major risk to urban communities under present climate and development practices. Whether future urban communities will contend with elevated or diminished pluvial flood risk depends on their development decisions. City development has historically been accomplished through measures that increase the impervious cover of the urban watershed and remove natural landscape features, such as wetlands, that would otherwise store and transport water away from frequently used surfaces (Greiner, Shtob, & Besek, 2020). Urban communities have also designed their stormwater management systems for storms of return periods of 2 to 15 years under historical climate conditions (ASCE/Environmental & Water Resources Institute, 2006) rather than for climate conditions they anticipate as possible in the near or distant future. Urban communities wishing to increase their resilience to pluvial floods should consider these temporal elements lest they find themselves in an avoidable future of increased pluvial flood risk. Further, in many cities across the globe, climate change is proceeding such that extreme storm events will become more common, so planning for an extreme, 100-year storm event in the present day will leave cities unprepared for extreme storms several decades before the lifespan of a typical stormwater management system element. Depending on climate change projections, strategies for reducing long-term pluvial flood risk that also prioritize the functionality of stormwater infrastructure through its full lifespan should extend design criteria to more intense storm events, such as 200- or 500-year storm events under present climate conditions.

In order to extend these design criteria to more intense storms, or to more specifically anticipate the range of events likely for a city or a region, stormwater managers will also need to contend with a considerable range on the high-end estimates of rainfall volumes under several climate pathways. In my dissertation research, I used the ensemble average of 23 downscaled models in Valdivia, Chile, under two different warming scenarios, which rendered these estimates conservative. That is, roughly half of input climate models predicted even greater rainfall volumes under both climate pathways, one estimating an upper 95% CI of nearly double the upper 95% CI of the ensemble average. If these other models end up being more accurate than the ensemble average, planned stormwater management systems may be under-designed and in need of upgrading before their lifespans are over. On the other hand, if these models end up being overestimates, then cities may overcommit scarce resources to stormwater management infrastructure.

This is a problem with no clear pathway to absolute solution, as cities are constrained by finances and space in designing their stormwater management systems and are balancing the reduction of pluvial flood risk with other priorities that require the same resources. In terms of partial solutions, the findings in this dissertation indicate the utility of two strategies: managing for pluvial flooding where it is most impactful on the lives of citizens and use of flexible stormwater management elements such as UEI. By targeting particular forms of pluvial flooding and by siting UEI appropriately, cities may achieve impactful pluvial flood risk reduction with the lowest use of resources.

Toward the first strategy, pluvial flood risk is dispersed throughout cities, but the proportion of it that is impactful on citizens is likely to be much more localized. In chapter 5 of this dissertation, I found that the perception of pluvial flood risk at scales of the city and the survey area are correlated with particular types of damage, which in turn occur in particular locations. My findings in this dissertation indicate that shielding businesses from pluvial flooding is critical, so focusing on improving retention and infiltration updrain of concentrated commercial centers may have a disproportionate impact on the perception and experience of pluvial flood risk. Similarly, cities should prioritize the reduction of pluvial flooding along major arterial roadways, as citizens correlated pluvial flood risk with disruptions to commuting. Areas necessary for the access and function of public transport should also be prioritized, both for public perception of pluvial flood risk and because economically marginalized groups may lose access to essential services during pluvial floods (de Sousa Silva, Viegas, Panagopoulos, & Bell, 2018).

Toward the second strategy, cities should expand their use of UEI such as wetlands, which may mitigate pluvial flood risk under any future rainfall volume, but they will need modeling resources to do this with any detectable effect. In this dissertation, I identified one subsystem in Valdivia where updrain wetlands mitigated downdrain pluvial flood risk at several different locations, and one subsystem where a downdrain wetland did not mitigate pluvial flooding anywhere in the system, which indicates that UEI such as wetlands provide their benefits conditionally on siting within the stormwater management system. As a general recommendation, cities may obtain the most benefit from siting UEI such as wetlands in updrain portions of their systems. However, cities should model the effects of such placement within their specific stormwater management system, and, in the case of using wetland UEI, consider a range of wetland dimensions. Wetlands of different dimensions will have different storage volumes and rates of infiltration and conveyance, so the presence and degree of the effect of wetland UEI on pluvial flooding may be detectable with some dimensions but not with others. Placement is potentially key to finding any effect of wetland UEI, but altering wetland dimensions may enable cities to ensure impactful pluvial flood risk reduction under projected future conditions. In the case that climate change models overestimate future precipitation, and cities find themselves with an “overdesigned” stormwater management system, the use of UEI such as wetlands will yield additional ecosystem services (Mitsch, W., & Gosselink, J., 2015) that are much needed in urban areas and which cannot be produced by traditional gray infrastructure (McPhearson, 2014).

6.3. Future Research Directions

Pluvial flood risk assessments would benefit greatly from being paired with other land-cover information to move beyond pluvial flood risk and into pluvial flood impact. As of now, discussions of pluvial flood risk, and most other forms of flood risk, concern issues of differential risk almost purely in areal terms (Messager, Ettinger, Murphy-Williams, & Levin, 2021; Pallathadka, Sauer, Chang, & Grimm, *in review*). That is, in deciding if different populations experience disproportionate amounts of pluvial flood risk, researchers typically compare pluvial flood area with proximity to marginalized populations, for example at the scale of the census block group. This provides an idea of broad pluvial flood exposure, but it does not communicate specific impacts or suggest measures that cities might wield to reduce the threat that pluvial floods pose to citizens. For example, in reducing pluvial flood risk in areas with high rates of poverty, it may be more impactful to these citizens to target flooding along walkways near critical infrastructure, at stops for public transportation, and in first-floor housing units than it would be to reduce flooding in nearby parks. In wealthier areas, pluvial flood-risk reduction may be more impactful along roadways where citizens feel that they are at risk of damaging themselves or their personal vehicles. Cities may find more value in pluvial flood models if researchers are able to pair basic exposure information with location and sociodemographic information to estimate impacts. Such paired information allows for more targeted, and potentially more resource-economical, interventions to reduce the threat that pluvial floods present to urban communities. Researchers may be able to identify more nuanced, and perhaps more impactful, forms of inequity between different citizen groups in cities, and make recommendations to reduce that inequity.

Of the hydrological processes in wetland UEI that can contribute to pluvial flood risk, I find no studies quantifying the effects of evapotranspiration (ET) on pluvial flood risk mitigation. Bois et al. (2017) found a large enough ET effect in a constructed wetland that it generated a sort of "biological tide," drawing water strongly during daylight hours and reducing the pull during twilight hours. In an individual wetland in a large urban area, the effect of ET may be rather small, and depending on other wetland characteristics such as configuration within the stormwater management system, may produce no notable effect on pluvial flooding. But scaled up to a city like Valdivia, where wetlands covered more than 20% of the urban area, and where much of the wetland area was vegetated, this ET effect may be quite large. I was unable to confirm an ET process as the source in my diurnal changes in wetland stage, but if it is the chief contributor of the effect, pluvial flood volumes may be reduced city-wide by a few thousand cubic meters simply by ET removing water during and after rainfall. Research into the ET effect of wetlands on pluvial flooding would be entirely novel, and would add another hydrological pathway by which wetland UEI can remove water that would not be wholly available with gray infrastructure (which would still allow for evaporation but not transpiration).

There remains a lack of studies examining differences in wetland UEI hydrology between natural, altered, and anthropogenic forms even though there are likely to be significant differences in the hydrological processes between them. In my dissertation proposal from three years ago, I referred to natural wetlands instead as “heritage” wetlands and distinguished them from anthropogenic wetlands such as constructed wetlands, restored wetlands, and "accidental" wetlands (Palta, Grimm, & Groffman, 2017). I defined heritage wetlands as natural wetlands that were not designed to perform a particular function or set of functions, although they may have been intentionally altered by humans from their spontaneous state to do so. In broader academia, heritage wetlands may be lumped into a broader category of "remnant" landscape features (Aliste & Musse, 2014). The bulk of academic literature concerned with UEI solutions to flooding focuses almost exclusively on urban stormwater GI expansion, either through engineered quasi-restoration of landscape features (Jia, Ma, & Wei. 2011) or by construction of wholly new GI projects (Belete 2018, Chan et al., 2018). There are many studies on differences between heritage wetlands and restored or constructed wetlands in terms of their other processes, such as nutrient cycling (Land et al., 2016), but none on flood mitigation. Key hydrological processes of wetlands, such as storage and infiltration, are likely different between heritage and other forms of wetlands, particularly in urban areas. Heritage wetlands for example may have different underlying soils than other forms of wetlands by virtue of being developed over longer periods, via spontaneous processes, and/or via different source materials, which will alter rates of infiltration. They may also have different dimensions than typical constructed or restored wetlands (Burgin, Franklin, & Hull, 2016), which are subject to urban land use constraints and geomorphic processes in urban areas that would not have been present under pre-urbanization formation. These differences in dimensions would alter storage and conveyance properties. Additionally, if ET does contribute substantially to water removal during and after storm events, heritage wetlands and other forms of wetlands are likely to host different plant communities (Burgin, Franklin, & Hull, 2016), which in turn will lead to differences in average rates of infiltration and ET.

There are no long-term studies of wetland hydrology in urban areas, but there are processes in urban areas that will alter wetland hydrology over time. Sedimentation in urban waterways tends to be higher than in natural waterways (Vietz, Walsh, & Fletcher, 2016), thus wetland dimensions and soil properties will be altered over time. Industrialization and other urban processes can increase transport and buildup of heavy metals in urban waterways, which can alter biodiversity of plants and in turn potentially infiltration and ET (Wu et al., 2016). Thus, the pluvial flood mitigation services of wetland UEI are likely to change, and likely diminish, over time. For the sake of understanding the true investment cost of wetland UEI, these long-term trends and needs for management should be studied.

Finally, future work on extreme events and pluvial flooding should engage more with reinsurance and post disaster payments (Hallegatte et al., 2012; Surminski, 2018) to ensure that citizens are able to recover their losses. Countries such as the United States only require flood insurance for areas designated at high risk of flooding according to FEMA, which are generally areas in fluvial and coastal flood zones but not pluvial flood zones. But given the potential of pluvial flood damages both now and in the future there is a clear need for insurance against pluvial flooding. The sharing of risk and the broad distribution of costs and compensations inherent to programs like reinsurance can potentially reduce the burden on individuals and households (Mechler et al., 2014), which may help to keep communities solvent and free of the need to mobilize when impacted by pluvial flooding.

REFERENCES

AECOM (2020). *2020 Annual Report: Municipal Separate Storm Sewer System*. Phoenix, AZ - City of Phoenix Water Services Department. Retrieved from https://www.phoenix.gov/waterservicessite/Documents/2020%20Stormwater%20Annual%20Report.pdf

Acreman, M., & Holden, J. (2013). How wetlands affect floods. *Wetlands, 33*(5), 773-786. https://doi.org/10.1007/s13157-013-0473-2

Ahern, J. (2011). From fail-safe to safe-to-fail: Sustainability and resilience in the new urban world. *Landscape and Urban Planning*, *100*(4), 341-343. https://doi.org/10.1016/j.landurbplan.2011.02.021

Alberti, M., Marzluff, J. M., Shulenberger, E., Bradley, G., Ryan, C., & Zumbrunnen, C. (2003). Integrating humans into ecology: Opportunities and challenges for studying urban ecosystems. *BioScience*, *53*(12), 1169-1179. https://doi.org/10.1641/0006-3568(2003)053[1169:IHIEOA]2.0.CO;2

Aliste, E., & Musse, A. (2014). Pensar los territorios del desarrollo: Sustentabilidad y acción pública en nombre de una ciudad imaginaria. Concepción (Chile), 1950-2010. *Revista Latinoamericana de Estudios Urbano Regionales*, *40*(120), 1.

Allmendinger, N.E., Pizzuto, J.E., Moglen, G.E., and Lewicki, M. (2007). A sediment budget for an urbanizing watershed, 1951-1996, Montgomery County, Maryland, U.S.A. *Journal of the American Water Resources Association*, *43*(6), 1483-1498.

Amigo, J., & Ramirez, C. (1998). A bioclimatic classification of Chile: Woodland communities in the temperate one. *Plant Ecology*, *136*, 9-26. https://doi.org/10.1023/A:100971420917

Anselin, L., Syabri, I., & Kho, Y. (2006). Geoda: An introduction to spatial data analysis. *Geographical Analysis*, *38*(1), 5–22. https://doi.org/10.1111/j.0016-7363.2005.00671.x

Arkema, K. K., Guannel, G., Verutes, G., Wood, S. A., Guerry, A., Ruckelshaus, M., ... & Silver, J. M. (2013). Coastal habitats shield people and property from sea-level rise and storms. *Nature Climate Change*, *3*, 913-918. https://doi.org/10.1038/nclimate1944

ASCE/Environmental & Water Resources Institute (2006). *Standard guidelines for the design of urban stormwater systems, standard guidelines for installation of urban stormwater systems, and standard guidelines for the operation and maintenance of urban stormwater systems*. Reston, VA: American Society of Civil Engineers.

Ashley, R. M., Balmforth, D. J., Saul, A. J., & Blanskby, J. D. (2005). Flooding in the future–predicting climate change, risks and responses in urban areas. *Water Science and Technology*, *52*(5), 265-273. https://doi.org/10.2166/wst.2005.0142

Babcock, R. F., & Bosselman, F. P. (1973). *Exclusionary zoning; land use regulation and housing in the 1970s*. New York, NY: Praeger.

Bahls, C. (2011). Achieving equity in health. *Health Affairs Health Policy Brief*, https://doi.org/10.1377/hpb20111006.957918. Retrieved from https://www.healthaffairs.org/do/10.1377/hpb20111006.957918/

Baker, A., Brenneman, E., Chang, H., McPhillips, L., & Matsler, M. (2019). Spatial analysis of landscape and sociodemographic factors associated with green stormwater infrastructure distribution in Baltimore, Maryland, and Portland, Oregon. *Science of the Total Environment, 664*, 461-473. https://doi.org/10.1016/j.scitotenv.2019.01.417

Balstrom, T., & Crawford, D. (2018). Arc-Malstrøm: A 1D hydrologic screening method for stormwater assessments based on geometric networks. *Computers & Geosciences*, *116*, 64-73. https://doi.org/10.106/j.cageo.2018.04.010

Barrett, B., Horne, R., & Fien, J. (2016). The ethical city: A rationale for an urgent new urban agenda. *Sustainability, 8*(11), 1197. https://doi.org/10.3390/su8111197

Belete, M. D. (2018). Ecohydrology to harmonize industrialization and ecological safety in urban environment: Case of Hawassa Lake, industries and Cheleleka wetland. *Ecohydrology & Hydrobiology*, *18*(2), 192-200.

Bell, S. A., Abir, M., Choi, H., Cooke, C., & Iwashyna, T. (2018). All-cause hospital admissions among older adults after a natural disaster. *Annals of Emergency Medicine, 71*(6). https://doi.org/10.1016/j.annemergmed.2017.06.042

Bella, N., Dridi, H., Kalla, & Kalla, M. (2020). Statistical modeling of annual precipitation in Oued El Gourzi Watershed, Algeria. *Applied Water Science*, *10*, 94. https://doi.org/10.1007/s13201-020-1175-6

Benedict, M.A., & McMahon, E.D. (2002) *Green Infrastructure: Smart Conservation for the 21st Century.* Washington D.C.: The Conservation Fund.

Bentz, A. S. (2012). Being a Tibetan refugee in India. *Refugee Survey Quarterly*, *31*(1), 80-107. https://doi.org/10.1093/rsq/hdr016

Biermann, F., & Boas, I. (2010). Preparing for a warmer world: Towards a global governance system to protect climate refugees. *Global environmental politics*, *10*(1), 60-88.

Bois, P., Childers, D. L., Courlouer, T., Laurent, J., Massicot, A., Sanchez, C. A., & Wanko, A. (2017). Confirming a plant-mediated "Biological Tide" in an aridland constructed treatment wetland. *Ecosphere*, *8*(3), e01756. https://doi.org/10.1002/ecs2.1756

Borden, K. A., Schmidtlein, M. C., Emrich, C. T., Piegorsch, W. W., & Cutter, S. L. (2007). Vulnerability of U.S. cities to environmental hazards. *Journal of Homeland Security and Emergency Management*, *4*(2), 5. https://doi.org/10.2202/1547-7355.1279

Brueckner, J. K., & Rosenthal, S. S. (2009). Gentrification and neighborhood housing cycles: Will America's future downtowns be rich? *The Review of Economics and Statistics*, *91*(4), 725-743. https://doi.org/10.1162/rest.91.4.725

Bubeck, P., Botzen, W. J., & Aerts, J. C. (2012). A review of risk perceptions and other factors that influence flood mitigation behavior. *Risk Analysis*, *32*(9), 1481-1495. https://doi.org/10.1111/j.1539-6924.2011.01783.x

Bullock, A., & Acreman, M. C. (2003). The role of wetlands in the hydrological cycle. *Hydrology and Earth System Sciences*, *7*, 358-389. https://doi.org/10.5194/hess-7-358-2003

Burgin, S., Franklin, M. J. M., & Hull, L. (2016). Wetland loss in the transition to urbanisation: A case study from Western Sydney, Australia. *Wetlands*, *36*, 985-994. https://doi.org/10.1007/s13157-016-0813-0

Cadag, J. R., & Gaillard, J. C. (2012). Integrating knowledge and actions in disaster risk reduction: The contribution of participatory mapping. *Area*, *44*(1), 100-109. https://doi.org/10.1111/j.1475-4762.2011.01065.x

Chakraborty, J., Collins, T. W., Montgomery, M. C., & Grineski, S. E. (2014). Social and spatial inequities in exposure to flood risk in Miami, Florida. *Natural Hazards Review*, *15*(3), 04014006. https://doi.org/10.1061/(asce)nh.1527-6996.0000140
Chambers, R. (2008). *Revolutions in Development Inquiry*. Earthscan, London.

Chan, F. K. S., Griffiths, J. A., Higgitt, D., Xu, S., Xhu, F., Tang, Y. T., … & Thorne, C. R. (2018). "Sponge City" in China—A breakthrough of planning and flood risk management in the urban context. *Land Use Policy*, *76*, 772-778. https://doi.org/10.1016/j.landusepol.2018.03.005

Chang, H., Lafrenz, M., Jung, I. W., Figliozzi, M., Platman, D., & Peerson, C. (2010). Potential impacts of climate change on flood-induced travel disruptions: A case study of Portland, Oregon, USA. *Annals of the Association of American Geographers*, *100*(4), 938-952. https://doi.org/10/1080/00045608.2010.497110

Chang, H., Lafrenz, M., Jung, I. W., Figliozzi, M., Platman, D., & Peerson, C. (2010). Potential impacts of climate change on flood-induced travel disruptions: A case study of Portland Oregon, USA. *Annals of the Association of American Geographers*, *100*(4), 938-952. https://doi.org/10/1080/00045608.2010.497110

Chang, H., Yu, D., Markolf, S., Hong, C., Eom, S., Song, W., & Bae, D. (2021). Understanding urban flood resilience in the Anthropocene: A social-ecological-technological systems (SETS) learning framework. *Annals of the American Association of Geographers*, *111*(3), 837-857. https://doi.org/10.1080/24694452.2020.1850230

Chang, H., Pallathadka, A., Sauer, J., Grimm, N., Zimmerman, R., Cheng, C., … & Herreros-Cantis, P. (2021) Assessment of urban flood vulnerability using the Social-Ecological-Technological Systems framework in six US cities. *Sustainable Cities and Society*, *68*, 102786. https://doi.org/10.1016/j.scs.2021.102786

Chapple, K. & Thomas, T. (2020). *Atlanta—Gentrification and Displacement*. Berkeley, CA: Urban Displacement Project. Retrieved from https://www.urbandisplacement.org/atlanta/atlanta-gentrification-and-displacement

Chattahoochee Riverkeeper. (2019). *Green Infrastructure in Practice: A Stormwater Management Case Study from Atlanta, GA*. Retrieved from https://chattahoochee.org/wp-content/uploads/2019/02/Green-Infrastructure-In-Practice_CRK-2019.pdf

Childers, D. L., Bois, P., Hartnett, H. E., McPhearson, T., Metson, G. S., & Sanchez, C. A. (2019). Urban Ecological Infrastructure: An inclusive concept for the non-built urban environment. *Elementa*, *7*(1). https://doi.org/10.1525/elementa.385

City of Portland. (2013a). *Gentrification and Displacement Studies*. Portland, OR: City of Portland Bureau of Planning and Sustainability. Retrieved from https://www.portland.gov/bps/adap/gentrification-and-displacement-studies

City of Portland. (2013b). *Gentrification Maps*. Portland, OR: City of Portland Bureau of Planning and Sustainability. Retrieved from https://www.portland.gov/sites/default/files/2020-01/appendix-a-maps.pdf

City of Portland. (2019). *Sewer and Drainage Facilities Design Manual*. Portland, OR: City of Portland Bureau of Environmental Services. Retrieved from https://www.portlandoregon.gov/bes/index.cfm?&a=360710

City of Portland (2020). *The 2020 Stormwater Management Manual*. Portland, OR: City of Portland Bureau of Environmental Services. Retrieved from https://www.portland.gov/bes/stormwater/swmm

Collins, T. W., Grineski, S. E., & Chakraborty, J. (2018). Environmental injustice and flood risk: A conceptual model and case comparison of metropolitan Miami and Houston, USA. *Regional Environmental Change*, *18*(2), 311-323. https://doi.org/10.1007/s10113-017-1121-9.

Cook, E. M., Berbés-Blázquez, M., Mannetti, L. M., Grimm, N. B., Iwaniec, D. M., & Muñoz-Erickson, T. A. (2021). Setting the state for co-production. In Hamstead, Z. A., Iwaniec, D. M., McPhearson, T., Berbés-Blázquez, M., Cook, E. M., & Muñoz-Erickson, T. A. (99-11), *Resilient Urban Futures*. New York, NY: Springer.

Correa, H., Blanco-Wells, G., Berrena, J., & Tacón, A. (2018). Self-organizing processes in urban green commons. The case of the Angachilla wetland, Valdivia-Chile. *International Journal of the Commons*, *12*(1), 573-595. https://doi.org/10.18352/ijc.856

Cutter, S. L., Boruff, B. J., & Shirley, W. L. (2003). Social vulnerability to environmental hazards. *Social Science Quarterly*, *84*(2), 242–261. https://doi.org/10.1111/1540-6237.8402002

Cutter, S. L., Emrich, C. T., Gall, M., & Reeves, R. (2018). Flash flood risk and the paradox of urban development. *Natural Hazards Review*, *19*(1), 05017005. https://doi.org/10.1061/(asce)nh.1527-6996.0000268

Dai, D. (2011). Racial/ethnic and socioeconomic disparities in urban green space accessibility: Where to intervene? *Landscape and Urban Planning*, *102*(4), 234-244. https://doi.org/10.1016/j.landurbplan.2011.05.002

Danielsson, P. (1980). Euclidean distance mapping. *Computer Graphics and Image Processing*, *14*(3), 227-248. https://doi.org/10.1016/0146-664x(80)90054-4

Davidson, N. C., Van Dam, A. A., Finlayson, C. M., & McInnes, R. J. (2019). Worth of wetlands: Revised global monetary values of coastal and inland wetland ecosystem services. *Marine and Freshwater Research*, *70*(8), 1189–1194. https://doi.org/10.1071/MF18391

Demuzere, M., Orru, K., Heidrich, O., Olazabal, E., Geneletti, D., Orru, H., ... & Faehnle, M. (2014). Mitigating and adapting to climate change: Multi-functional and multi-scale assessment of green urban infrastructure. *Journal of environmental management*, *146*, 107-115. https://doi.org/10.1016/j.jenvman.2014.07.025

Dennis, S. F., Jr., Gaulocher, S., Carpiano, R. M., & Brown, D. (2009). Participatory photo mapping (PPM): Exploring an integrated method for health and place research with young people. *Health & Place*, *15*(2), 466-473. https://doi.org/10.1016/j.healthplace.2008.08.004

De Sousa Silva, C., Viegas, I., Panagopoulos, T., & Bell, S. (2018). Environmental justice in accessibility to green infrastructure in two European cities. *Land*, *7*(4), 134. https://doi.org/10.3990/land7040134

Dettling, L., Hsu, J., Moore, K., & Thompson, J. (2017). *Recent Trends in Wealth-Holding by Race and Ethnicity: Evidence from the Survey of Consumer Finances*. Washington D.C.: Board of Governors of the Federal Reserve System. Retrieved from https://www.federalreserve.gov/econres/notes/feds-notes/recent-trends-in-wealth-holding-by-race-and-ethnicity-evidence-from-the-survey-of-consumer-finances-20170927.htm

Dewitz, J. (2019). National Land Cover Database (NLCD) 2016 Products: U.S. Geological Survey data release. https://doi.org/10.5066/P96HHBIE.

Dirección General de Aeronáutica Civil (2020). *Reporte anual de la evolución del clima en Chile*. Santiago, Chile.

Dooling, S. (2009). Ecological gentrification: A research agenda exploring justice in the city. *International Journal of Urban and Regional Research*, *33*(3), 621-639. https://doi.org/10.1111/j.1468-2427.2009.00860.x

Dong, S., Esmalian, A., Farahmand, H., & Mostafavi, A. (2020). An integrated physical-social analysis of disrupted access to critical facilities and community service-loss tolerance in urban flooding. *Computers, Environment and Urban Systems*, *80*, 101443. https://doi.org/10.1016/j.compenvurbsys.2019.101443

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*, 112-125. https://doi.org/10.1111/j.1753-318X.2010.01061.x

Eisenman, D. P., Cordasco, K. M., Asch, S., Golden, J. F., & Glik, D. (2007). Disaster planning and risk communication with vulnerable communities: Lessons from Hurricane Katrina. *American Journal of Public Health*, *97*(1). https://doi.org/10.2105/ajph.2005.084335

Elmqvist, T., Setäläl, H., Handel, S., van der Ploeg, S., Aronson, J., Blignaut, J.N., ... & de Groot, R. (2015). Benefits of restoration of ecosystem services in cities. *Current Opinion in Environmental Sustainability*, *14*, 101-108. https://doi.org/10.1016/j.cosust.2015.05.001

Ertorer, S. E. (2021). Asylum regimes and refugee experiences of precarity: The case of Syrian refugees in Turkey. *Journal of Refugee Studies*, *34*(3), 2568-2592. https://doi.org/10.1093/jrs/feaa089

ESRI. (2019). ArcGIS Desktop: Release 10.7. Redlands, CA: Environmental Systems Research Institute.

Fahy, B., Brenneman, E., Chang, H., & Shandas, V. (2019) Spatial analysis of urban floods and extreme heat potential in Portland, OR. *International Journal of Disaster Risk Reduction, 39*, 101117. https://doi.org/10.1016/j.ijdrr.2019.101117
Falconer, R. H., Cobby, D., Smyth, P., Astle, G., Dent, J., & Golding, B. (2009). Pluvial flooding: New approaches in flood warning, mapping and risk management. *Journal of Flood Risk Management*, *2*, 198-208. https://doi.org/10.1111/j.1753-318X.2009.01034.x

Federal Emergency Management Agency (2016). National disaster recovery framework. In D. O. H. Security (Ed.), (2nd ed.). Washington: Federal Emergency Management Agency.

Ferguson, A. P., & Ashley, W. S. (2017). Spatiotemporal analysis of residential flood exposure in the Atlanta, Georgia metropolitan area. *Natural Hazards*, *87*(2), 989-1016. https://doi.org/10.1007/s11069-017-2806-6

Fillo, N. K., & Bhaskar, A. S. (2021). Lawn irrigation contributions to semi-arid urban baseflow based on water-stable isotopes. *Water Resource Research*, *57*(8), e2020WR028777. https://doi.org/10.1029/2020WR028777

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. https://doi.org/10.1016/j.advwatres.2012.09.001

Fletcher, T. D., Shuster, W., Hunt, W. F., Ashley, R., Butler, D., Arthur, S., ... & Viklander, M. (2015). SUDS, LID, BMPs, WSUD and more—The evolution and application of terminology surrounding urban drainage. *Urban Water Journal*, *12*, 525-542. https://doi.org/10.1080/1573062X.2014.916314

Foster, S., Leichenko, R., Nguyen, K. H., Blake, R., Kunreuther, H., Madajewicz, M., … & Ravenborg, D. (2019). New York City panel on climate change 2019 report Chapter 6: Community-based assessments of adaptation and equity. *Annals of the New York Academy of Sciences*, *1439*(1), 126–173. https://doi.org/10.1111/nyas.14009

Fothergill, A., & Peek, L. A. (2004). Poverty and disasters in the United States: A review of recent sociological findings. *Natural Hazards*, *32*(1), 89-110. https://doi.org/10.1023/b:nhaz.0000026792.76181.d9

Groffman, P. M., Bain, D. J., Band, L. E., Belt, K. T., Brush, G. S., Grove, J. M., ... & Zipperer, W. C. (2003). Down by the riverside: Urban riparian ecology. *Frontiers in Ecology and the Environment*, *1*(6), 315-321. https://doi.org/10.1890/1540-9295(2003)001[0315:DBTRUR]2.0.CO;2

Garcés-Vargas, J., Schneider, W., Pinochet, A., Piñones, A., Olguin, F., Brieva, D., & Wan, Y. (2020). Tidally forced saltwater intrusions might impact the quality of drinking water, the Valdivia River (40 S), Chile estuary case. *Water*, *12*(9), 2387. https://doi.org/10.3390/w12092387.
Gellis, A.C., Myers, M.K., Noe, G.B., Hupp, C.R., Schenk, E.R., Myers, L., & Elsevier, B.V. (2017). Storms, channel changes, and a sediment budget for an urban-suburban stream, Difficult Run, Virginia, USA. *Geomorphology*, *278*, 128-148. https://doi.org/10.1016/j.geomorph.2016.10.031

Georgescu, M., Broadbent, A. M., Wang, M., Krayenhoff, S., & Moustaoui, M. (2021). Precipitation response to climate change and urban development over the continental United States. *Environmental Research Letters*, *16*(4), 044001. https://doi.org/10.1088/1748-9326/abd8ac

Getis, A., & Ord, J.K. (1992). The analysis of spatial association by use of distance statistics. *Geographical Analysis*, *24*, 189-206. https://doi.org/10.1111/j.1538-4632.1992.tb00261.x

Gilleland E, Katz RW (2016). extRemes 2.0: An Extreme Value Analysis Package in R. *Journal of Statistical Software*, *72*(8), 1–39. https://doi.org/10.18637/jss

Grayson, J. E., Chapman, M. G., & Underwood, A. J. (1999). The assessment of restoration of habitat in urban wetlands. *Landscape and Urban Planning*, *43*, 227-236.

Grimm, N. B., Faeth, S. H., Golubiewski, N. E., Redman, C. L., Wu, J., Bai, X., & Briggs, J. M (2008). Global change and the ecology of cities. *Science*, *319*(5864), 756-760. https://doi.org/10.1126/science.1150195

Gould, K. A., & Lewis, T. L. (2016). *Green gentrification urban sustainability and the struggle for environmental justice*. Routledge, Taylor & Francis Group.

Gülbaz, S., & Kazezyilmaz-Alhan, C. M. (2014). Investigating effects of low impact development on surface runoff and TSS with a calibrated hydrodyanamic model. *Houille Blanche-Revue Internationale De L Eau*, *3*(3), 77-84. https://doi.org/10.1051/ihb/2014031

Haaland, C., & van den Bosch, C. K. (2015). Challenges and strategies for urban green-space planning in cities undergoing densification: a review. *Urban Forestry and Urban Greening*, *14*, 760-771. https://doi.org/10.1016/j.ufug.2015.07.009

Hackworth, J. (2002). Postrecession gentrification in New York City. *Urban Affairs Review*, *37*(6), 815-843. https://doi.org/10.1177/107874037006003

Hagman, D. G. (1971). Urban planning and development—race and poverty—past, present, and future. *Utah Law Review, 46,* 46-77. Retrieved June 22, 202 from https://collections.lib.utah.edu/details?id=722886

Hajek, E., & Di Castri, F. 1975. Bioclimatografia de Chile. *Dirección de Investigación*. Rectoría Académica, Universidad Católica de Chile.
Hallack-Alegria, M., & Watkins, D. W. Jr. (2007). Annual and warm season drought intensity-duration-frequency analysis for Sonora, México. *Journal of Climate*, *20*(9), 1897-1909. https://doi.org/ 10.1175/JCLI4101.1

Hartman, C. W., & Squires, G. D. (2013). *There is no such thing as a natural disaster: Race, class, and Hurricane Katrina*. Routledge.

Hayhoe, K., Wuebbles, D.J., Easterling, D.R., Fahey, D.W., Doherty, S., Kossin, J., …& Wehner., M. (2018). Our changing climate. In D. R. Reidmiller, C.W. Avery, D.R. Easterling, K.E. Kunkel, K.L.M. Lewis, T.K. Maycock, and B.C. Stewart (Eds.), *Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment, Volume II*. Washington D.C.: U.S. Global Change Research Program. https://doi.org/10.7930/NCA4.2018.CH2

Hamstead, Z., & Sauer, J. (2021). Mapping vulnerability to weather extremes: Heat and flood assessment approaches. In Hamstead, Z. A., Iwaniec, D. M., McPhearson, T., Berbés-Blázquez, M., Cook, E. M., & Muñoz-Erickson, T. A. (Eds.) *Resilient Urban Futures* (pp. 47-66). New York, NY: Springer.

Hayward, T. (2007). Human rights versus emissions rights: Climate justice and the equitable distribution of ecological space. *Ethics & International Affairs, 21*(4), 431-450. https://doi.org/10.1111/j.1747-7093.2007.00117.x

Heonin, J., Russo, B., Mark, O., & Gourbesville, P. (2013). Real-time urban flood forecasting and modelling–A state of the art. *Journal of Hydroinformatics*, *15*(3), 717-736. https://doi.org/10.2166/hydro.2013.132

Hirabayashi, Y., Mahendran, R., Koirala, S., Konoshima, L., Yamazaki, D., Watanabe, S., ... Kanae, S. (2013). Global flood risk under climate change. *Nature Climate Change*, *3*, 816-821. https://doi.org/10.1038/nclimate1911

Hobbie, S. E., & Grimm, N. B. (2020). Nature-based approaches to managing climate change impacts in cities. Philosophical Transactions of the Royal Society B: Biological Sciences, *375*(1794), 20190124. https://doi.org/10.1098/rstb.2019.0124

Holzer, K. A. (2014). Amphibian use of constructed and remnant wetlands in an urban landscape. *Urban ecosystems*, *17*(4), 955-968. https://doi.org/10.1007/s11252-014-0373-0

Hondula, D. M., Balling, R. C., Vanos, J. K., & Georgescu, M. (2015). Rising temperatures, human health, and the role of adaptation. *Current Climate Change Reports*, *1*(3), 144-154. https://doi.org/10.1007/s40641-015-0016-4

Immergluck, D. (2009). Large redevelopment initiatives, housing values and gentrification: The case of the Atlanta beltline. *Urban Studies, 46*(8), 1723-1745. https://doi.org/10.1177/0042098009105500\

Immergluck, D., & Balan, T. (2018). Sustainable for whom? Green urban development, environmental gentrification, and the Atlanta beltline. *Urban Geography, 39*(4), 546-562. https://doi.org/10.1080/02723638.2017.1360041

Instituto Nacional de Investigación de Recursos Naturales (1978). *Estudio de suelos de la Provincia de Valdivia: Volumen 2*. Instituto Nacional de Investigación de Recursos Naturales.

Intergovernmental Panel on Climate Change (IPCC, 2021). Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Masson-Delmotte, V., Zhai, P., Pirani, A., Connors, S. L., Péan, C., Berger, S., …, & Zhou, B. (Eds.)]. Cambridge University Press.

International Fund for Agricultural Development (2009). *The IFAD Adaptive Approach to Participatory Mapping*. International Fund for Agricultural Development, Rome.

Iwaniec, D. M., Cook, E. M., Davidson, M. J., Berbés-Blázquez, M., Georgescu, M., Krayenhoff, E.S, ... & Grimm, N. B. (2020). The co-production of sustainable future scenarios. *Landscape and Urban Planning*, *197*, 103744. https://doi.org/10.1016/j.landurbplan.2020.103744

Jahn, T., Bergmann, M., & Keil, F. (2012). Transdisciplinarity: Between mainstreaming and marginalization. *Ecological economics*, *79*, 1-10. https://doi.org/10.1016/j.ecolecon.2012.04.017

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

Jerome, G. (2017). Defining community-scale green infrastructure. *Landscape Research*, 42(2), 223-229. https://doi.org/10.1080/01426397.2016.1229463

Jia, H., Ma, H., & Wei, M. (2011). Urban wetland planning: A case study in the Beijing central region. *Ecological Complexity, 8*(2), 213-221.

Jiang, Y., Zevenbergen, C., & Fu, D. (2018). Urban pluvial flooding and stormwater management: A contemporary review of China's challenges and "sponge cities" strategy. *Environmental Science & Policy*, *80*, 132-143. https://doi.org/10.1016/j.envsci.2017.11.016.

Kadykalo, A.N., & Findlay, C.S. (2016). The flow regulation services of wetlands. *Ecosystem Services*, *20*, 91-103. https://doi.org/10.1016/j.ecoser.2016.06.005

Keenan, J. M., Hill, T., & Gumber, A. (2018). Climate gentrification: from theory to empiricism in Miami-Dade County, Florida. *Environmental Research Letters*, *13*(5), 054001. https://doi.org/10.1088/1748-9326/aabb32

KewalRamani, A., Gilbertson, L., & Fox, M. A. (2007). *Status and trends in the education of racial and ethnic minorities*. Washington D.C.: U.S. Department of Education. Retrieved from https://nces.ed.gov/pubs2007/2007039.pdf

Keyvanfar, A., Shafaghat, A., Ismail, N., Mohamad, S., & Ahmad, H. (2021). Multifunctional retention pond for stormwater management: A decision-support model using Analytical Network Process (ANP) and Global Sensitivity Analysis (GSA). *Ecological Indicators, 124*, 107317. https://doi.org/10.1016/j.ecolind.2020.107317

Kim, Y., Chester, M. V., Eisenberg, D. A., & Redman, C. L. (2019). The infrastructure trolley problem: Positioning safe-to-fail infrastructure for climate change adaptation. *Earth's Future*, *7*(7), 704-717. https://doi.org/10.1029/2019EF001208

Koo, B. W., Boyd, N., Botchwey, N., & Guhathakurta, S. (2019). Environmental equity and spatiotemporal patterns of urban tree canopy in Atlanta. *Journal of Planning Education and Research*, 739456. https://doi.org/10.1177/0739456X19864149

Kumar, S., Agarwal, A., Villuri, V. G. K., Pasupuleti, S., Kumar, D., Kaushal, D. R., … & Sivakumar, B. (2021). Constructed wetland management in urban catchments for mitigating floods. *Stochastic Environmental Research and Risk Assessment*, *35*, 2105-2124. https://doi.org/10.1007/s00477-021-02004-1

Land, M., Granéli, W., Grimvall, A., Hoffman, C. C., Mitsch, W. J., Tonderski, K. S., & Verhoeven, J. T. A. (2016). How effective are created or restored freshwater wetlands for nitrogen and phosphorus removal? A systematic review. *Environmental Evidence*, *5*, 9. https://doi.org/10.1186/s13750-016-0060-0

Lang, D. J., Wiek, A., Bergmann, M., Stauffacher, M., Martens, P., Moll, P., ... & Thomas, C. J. (2012). Transdisciplinary research in sustainability science: practice, principles, and challenges. *Sustainability science*, *7*(1), 25-43. https://doi.org/10.1007/s11625-011-0149-x

La Rosa, D., & Pappalardo, V. (2020). Planning for spatial equity—A performance based approach for sustainable urban drainage systems. *Sustainable Cities and Society, 53*, 101885. https://doi.org/10.1016/j.scs.2019.101885

Lashford, C., Rubinato, M., Cai, Y., Hou, J., Abolfathi, S., Coupe, S., ... & Tait, S. (2019). SuDS & sponge cities: A comparative analysis of the implementation of pluvial flood management in the UK and China. *Sustainability*, *11*(1), 213. https://doi.org/10.3390/su1101213

Lawson, E., Thorne, C., Ahilan, S., Allen, D., Arthur, S., Everett, G., … & Wright, N. (2014). Delivering and evaluating the multiple flood risk benefits in Blue-Green cities: An interdisciplinary approach. *Flood Recovery, Innovation and Response IV*, *184*, 113-123. https://doi.org/10.2495/FRIAR140101

Lechowska, E. (2018). What determines flood risk perception? A review of factors of flood risk perception and relations between its basic elements. *Natural Hazards*, *94*, 1341-1366. https://doi.org/10.1007/s11069-018-3480-z

Levy, B. S., & Patz, J. A. (2015). Climate change, human rights, and social justice. *Annals of Global Health*, *81*(3), 310-322.

Li, X., & Willems, P. (2020). A hybrid model for fast and probabilistic urban pluvial flood prediction. *Water Resources Research*, *56*, e2019WR025128. https://doi.org/10.10129/2019WR025128.

Li, Y., Li, H. X., Huang, J., & Liu, C. (2020). An approximation method for evaluating flash flooding mitigation of sponge city strategies—A case study of Central Geelong. *Journal of Cleaner Production*, *257*, 120525. https://doi.org/10.1016/j.jclepro.2020.120525

Lin, B., Meyers, J., & Barnett, G. (2015). Understanding the potential loss and inequities of green space distribution with urban densification. *Urban Forestry & Urban Greening*, *14*(4), 952-958. https://doi.org/10.1016/j.ufug.2015.09.003

Line, D. E., Brown, R. A., Hunt, W. F., & Lord, W. G. (2012). Effectiveness of LID for commercial development in North Carolina. *Journal of Environmental Engineering*, *138*(6), 680-688. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000515

Logie, C. H., Daniel, C. A., Ahmed, U., & Lash, R. (2010). 'Life under the tent is not safe, especially for young women': understanding intersectional violence among internally displaced youth in Leogane, Haiti. *Global Health Action*, *10*(2). https://doi.org/10.1080/16549716.2017.1270816

Lundin, L. (1994). Impacts of forest drainage on flow regime. *Studia Forestalia Suecicia*, *192*, 1-22.

Maldonado, A., Collins, T. W., Grineski, S. E., & Chakraborty, J. (2016). Exposure to flood hazards in Miami and Houston: Are Hispanic immigrants at greater risk than other social groups? *International Journal of Environmental Research and Public Health*, *13*(8), 775. https://doi.org/10.3390/ijerph13080775

Maragno, D., Gaglio, M., Robbi, M., Appiotti, F., Fano, E. A., & Gissi, E. (2018). Fine-scale analysis of urban flooding reduction from green infrastructure: An ecosystem services approach for the management of water flows. *Ecological Modeling*, *386*, 1-10. https://doi.org/10.1016/j.ecolmodel.2018.08.002

McPhearson, T. (2014). Urban ecosystem services for resilience planning and management in New York City. *AMBIO*, *43*(4), 502-515. https://doi.org/10.1007/s13280-014-0509-8

McPhearson, T., Iwaniec, D. M., & Bai, X. (2016). Positive visions for guiding urban transformations toward sustainable futures. *Current Opinion in Environmental Sustainability*, *22*, 33-40. https://doi.org/10.1016/j.cosust.2017.04.004

McPhillips, L. & M. Matsler. (2018). Temporal evolution of green stormwater infrastructure strategies in three US cities. *Frontiers in Built Environment, 4*. https://doi.org/10.3389/fbuil.2018.00026

Measham, T. G., Preston, B. L., Smith, T. F., Brooke, C., Gorddard, R., Withycombe, G., & Morrison, C. (2011). Adapting to climate change through local municipal planning: barriers and challenges. *Mitigation and adaptation strategies for global change*, *16*(8), 889-909. https://doi.org/10.1007/s11027-011-9301-2

Meerow, S., Newell, J. P., & M. Stults (2016). Defining urban resilience: A review. *Landscape and Urban Planning*, *147*, 38-49. https://doi.org/10.1016/j.landurbanplan.2015.11.011

Mercer, J., Kelman, I., Suchet-Pearson, S., & Lloyd, K. (2009). Integrating indigenous and scientific knowledge bases for disaster risk reduction in Papua New Guinea. *Geografiska Annaler, B*(91), 157-183.

Messager, M. L., Ettinger, A. K., Murphy-Williams, M., & Levin, P. S. (2021). Fine-scale assessment of inequities in inland flood vulnerability. *Applied Geography*, *133*, 102492. https://doi.org/10.1016/j.apgeog.2021.102492

Millennium Ecosystem Assessment (2005). *Ecosystems and Human Well-Being: Biodiversity Synthesis*. World Resources Institute, Washington D.C. (USA).

Milly, P. C. D., Wetherald, R. T., Dunne, K. A., & Delworth, T. L. (2002). Increasing risk of great floods in a changing climate. *Nature*, *415*(6871), 514-517. https://doi.org/10.1038/415514a

Ministerio de Agricultura (2011). *Aprueba reglamento de suelos, aguas, y humedales*. [Approval of the regulation of soils, waters, and wetlands]. Retrieved from https://www.fao.org/faolex/results/details/es/c/LEX-FAOC101076

Ministerio de Obras Públicas de Chile (CMOP; 2012) *Plan maestro de evacuación y drenaje de aguas lluvias*. Ministerio de Obras Públicas, Santiago, Chile.
Ministerio del Medio Ambiente (2020). *Modifica diversos cuerpos legales con el objetivo de proteger los humedales urbanos* [Modification of diverse legal bodies with the objective of protecting urban wetlands]. Retrieved from https://www.bcn.cl/leychile/navegar?idNorma=1141461

Mitsch, W., & Gosselink, J. (2015). *Wetlands*, 5th edition. Wiley.

Moran, P. (1950). Notes on continuous stochastic phenomena. *Biometrika*, *37*(1/2), 17-23. https://doi.org/10.2307/2332142

Morita, M. (2014). Flood risk impact factor for comparatively evaluating the main causes that contribute to flood risk in urban drainage areas. *Water*, *6*(2), 253–270. https://doi.org/10.3390/w6020253

Munoz, S. M. (2021). Environmental mobility in a polarized world: Questioning the "climate refugee" label for Pacific Islanders. *Journal of International Migration and Integration*, *22*(4), 1271-1284. https://doi.org/10.1007/s12134-020-00799-6

Narayan, S., Beck, M. W., Wilson, P., Thomas, C. J., Guerrero, A., Shepard, C., ... & Trespalacios, D. (2017). The value of coastal wetlands for flood damage reduction in the Northeastern USA. *Scientific Reports*, *7*(1), 9463. https://doi.org/10.1038/s41598-017-09269

National Oceanic and Atmospheric Administration (NOAA). *NOAA Atlas 14: Precipitation-frequency atlas of the United States*. Retrieved from https://hdsc.nws.noaa.gov/hdsc/pfds/

National Community Reinvestment Coalition (NCRC, 2019). *Gentrification and cultural displacement most intense in America's largest cities, and absent from many other*s. Retrieved from https://ncrc.org/study-gentrification-and-cultural-displacement-most-intense-in-americas-largest-cities-and-absent-from-many-others/

Nelson, J. R., Grubesic, T. H., Miller, J. A., & Chamberlain, A.W. (2021). The equity of tree distribution in the most ruthlessly hot city in the United States: Phoenix, Arizona. *Urban Forestry & Urban Greening*, *59*, 127016. https://doi.org/10.1016/j.ufug.2021.127016

Nesbitt, L., Meitner, M. J., Girling, C., Sheppard, S. R., & Lu, Y. (2019). Who has access to urban vegetation? A spatial analysis of distributional green equity in 10 US cities.

Landscape and Urban Planning, *181*, 51-79. https://doi.org/10.1016/j.landurbplan.2018.08.007

Nicholls, R.J., Hoozemans, F.M.J., & Marchand, M. (1999). Increasing flood risk and wetland losses due to global sea-level rise: Regional and global analyses. *Global Environmental Change*, *9*, 69-87. https://doi.org/10.1016/S0959-3780(99)00019-9

Nowak, D. J., & Walton, J. T. (2005). Projected urban growth (2000–2050) and its estimated impact on the US forest resource. *Journal of Forestry*, *103*(8), 383-389. https://doi.org/10.1093/jof/103.8.383

Nowak, D. J., & Greenfield, E. J. (2020). The increase of impervious cover and decrease of tree cover within urban areas globally (2012–2017). *Urban Forestry & Urban Greening*, *49*, 126638. https://doi.org/10.1016/j.ufug.2020.126638

O'Donnell, E. C., & Thorne, C. R. (2020). Drivers of future urban flood risk. *Philosophical Transactions. Series A, Mathematical, Physical, and Engineering sciences*, *378*(2168), 2359-2378.

Oner, A. C., Durmaz-Drinkwater, B., & Grant, R. J. (2020). The precarity of refugees: The case of Basmane Ísmir, Turkey. *Journal of Ethnic and Migration Studies*. https://doi.org/10.1080/1369183X.2020.1732951

Ord, J. K., & Getis, A. (1995). Local spatial autocorrelation statistics: Distributional issues and an application. *Geographical Analysis*, *27*(4), 286-306. https://doi.org/10.1111/j.1538-4632.1995.tb00912.x

Ortiz, L., Mustafa, A., Rozenzweig, B., Carrero, R., & McPhearson, T. (2021). Modeling urban futures: Data-driven scenarios of climate change and vulnerability in cities. In Hamstead, Z. A., Iwaniec, D. M., McPhearson, T., Berbés-Blázquez, M., Cook, E. M., & Muñoz-Erickson, T. A. (Eds.) *Resilient Urban Futures* (pp. 47-66). New York, NY: Springer.

Pallathadka, A. K., Chang, H., & Ajibade, I. (2021) The spatial patterns of pluvial flood risk, blue-green infrastructure, and social vulnerability: A case study from two Alaskan cities. *International Journal of Geospatial and Environmental Research*, *8*(3), Article 2. Retrieved from https://dc.uwm.edu/ijger/vol8/iss3/2

Pallathadka, A., Sauer, J., Chang, H., Grimm, N. B. (2022). Urban flood risk and green infrastructure: Who is exposed to risk and who benefits from investment? A case study of three U.S. cities. *Landscape and Urban Planning*, *223*. https://doi.org/10.1016/j.landurbplan.2022.104417

Palta, M. M., Grimm, N. B., & Groffman, P. M. (2017). "Accidental" urban wetlands: Ecosystem functions in unexpected places. *Frontiers in Ecology and the Environment, 15*(5), 248-256. https://doi.org/10.1002/fee.1494

Papatzani, E., Psallidaki, T., Kandylis, G., & Micha, I. (2021). Multiple geographies of precarity: Accommodation policies for asylum seekers in metropolitan Athens, Greece. *European Urban and Regional Studies*, *29*(2), 189-203. https://doi.org/10.1177/09697764211040742

Pappalardo, V., La Rosa, D., Campisano, A., & La Greca, P. (2017). The potential of green infrastructure in urban runoff control for land use planning: A preliminary evaluation from a southern Italy case study. *Ecosystem Services*, *26*, 345-354. https://doi.org/10.1016/j.ecoser.2017.04.015

Peacock, W., & Girard, C. (1997). Ethnic and racial inequalities in hurricane damage and insurance settlements. In W.G. Peacock, B.H. Morrow, & H. Gladwin (Eds.), *Hurricane Andrew: Ethnicity, gender and the sociology of disasters* (pp. 171-190). London: Routledge.

Peacock, W. G., Dash, N., and Zhang, Y. (2006). Shelter and housing recovery following disaster. In H. Rodriguez, E. L. Quarantelli, & R. R. Dynes (Eds.), *Handbook on disaster research* (pp. 258-274). New York, New York: Springer.

Penning-Roswell, E., & Korndewal, M. (2019). The realities of managing uncertainties surrounding pluvial urban flood risk: An ex post analysis in three European cities. *Journal of Flood Risk Management*, *12*(3), e12467. https://doi.org/10.1111/jfr3.12467

Pincetl, S. (2010). From the sanitary city to the sustainable city: challenges to institutionalising biogenic (nature's services) infrastructure. *Local Environment*, *15*(1), 43-58. https://doi.org/10.1080/13549830903406065

Pincetl, S., Bunje, P., & Holmes, T. (2012). An expanded urban metabolism method: Toward a systems approach for assessing urban energy processes and causes. *Landscape and Urban Planning*, *107*(3), 193-202. https://doi.org/10.1016/j.landurbplan.2012.06.006

Python Software Foundation. *Python Language Reference*, versions 2.7 and 3.6. Retrieved June 22, 2021 from http//www.python.org.

R Core Team (2021). R: A language and environment for statistical computing (Version 4.1.1) [Software]. R Foundation for Statistical Computing, Vienna, Austria. Available from https://www.R-project.org/

Reiss, R. D., & Thomas, M. (2007). *Statistical analysis of extreme values* (3rd ed.). Basel: Birkhäuser.

Richardson, J., Mitchell, B., & Edlebi J. (2020). *Gentrification and disinvestment*. National Community Reinvestment Coalition. Retrieved from https://ncrc.org/gentrification20/

Rigolon, A., and Németh, J. (2020). Green gentrification or 'just green enough': Do park location, size and function affect whether a place gentrifies or not? *Urban Studies*, *57*(2), 402-420. https://doi.org/10.1177/0042098019849380

Rojas, C., Munizaga, J., Rojas, O., Martínez, C., & Pino, J. (2019). Urban development versus wetland loss in a coastal Latin American City: Lessons for sustainable land use planning. *Land Use Policy*, *80*, 47-56. https://doi.org/10.1016/j.landusepol.2018.09.036

Rosenzweig, B. R., Herreros Cantis, P., Kim, Y., Cohn, A., Grove, K., … & Chang, H. (2021). The value of urban flood modeling. Earth's Future, *9*(1), e2020EF001739. https://doi.org/10.1029/2020EF001739

Rufat, S., Tate, E., Burton, C. G., & Maroof, A. S. (2015). Social vulnerability to floods: Review of case studies and implications for measurement. *International Journal of Disaster Risk Reduction*, *14*, 470-486. https://doi.org/10.1016/j.ijdrr.2015.09.013

Rosenzweig, B. R., McPhillips, L., Chang, H., Cheng, C. Welty, C., Matsler, M., … & Davidson, C. I. (2018). Pluvial flood risk and opportunities for resilience. *Wiley Interdisciplinary Reviews. Water*, *5*(6). https://doi.org/10.1002/wat2.1302
Salinas, M. A. S. , & Espinosa, M. J. (2004). Serie Fascículos: Inundaciones. Retrieved from http://www.cenapred.gob.mx/es/Publicaciones/archivos/3-FASCCULOINUNDACIONES.PDF

Sauer, J., Grimm, N. B., Childers, D., & Barbosa, O. (In preparation-a). Wetland storage dynamics and variable flood risk mitigation services in Valdivia, Chile

Sauer, J., Grimm, N. B., Kunkel, K., Ballinger, A., McPhearson, T., Mustafa, A., & Barbosa, O. (In preparation-b)

Sauer, J., Grimm, N. B., Morua-Robles, A., Cook, E. M., Navarro, J. (In preparation-c). Pluvial flood modeling with the Arc-Malstrøm method and ground truthing in Hermosillo, México.

Sauer, J., Grimm, N. B., Cook, E. M., Barbosa, O., McPhearson, T., Mustafa, A. (*In revision*). Inland urban wetland loss and its impacts on urban drainage system performance in Valdivia, Chile

Scholz, R. W. (2017). The normative dimension in transdisciplinarity, transition management, and transformation sciences: New roles of science and universities in sustainable transitioning. *Sustainability*, *9*, 991. https://doi.org/10.3390/su9060991

Schumann, R. L., III, Binder, S. B., & Greer, A. Unseen potential: Photovoice methods in hazard and disaster science. *GeoJournal*, *84*, 273-289. https://doi.org/10.1007/s10708-017-9825-4

Schneidewind, U., Singer-Brodowski, M., & Augenstein, K. (2016). Transformative science for sustainability transitions. In *Handbook on sustainability transition and sustainable peace*; Bauch, C. T., Oswald Spring, U. (Eds.); Springer: Berlin, Germany, 2014.

Septima.dk & Balstrom T. (2016). Arc-Malstrøm. Retrieved July 6, 2020 from https://github.com/Kortforsyningen/malstroem

Septima.dk and Balstrom T. (2020). Septima fork of Arc-Malstrøm. Retrieved July 6, 2020 from https://github.com/Septima/malstroem

Sepúlveda, A. (2021). Escasez hídrica en Los Ríos: Valdivia sufre el año más seco desde 1853. *Meteored*. Retrieved from https://www.meteored.cl/noticias/actualidad/escasez-hidrica-en-los-rios-valdivia-sufre-el-ano-mas-seco-desde-1853.html

Shepherd, K. A., Ellis, P. A., Rivett, M. O. (2006). Integrated understanding of urban land, groundwaer, baseflow, and surface-water quality—The city of Birmingham, UK. *Science of the Total Environment*, *1*(3), 180-195. https://doi.org/10.1016/j.scitotenv.2005.08.052

Shields, C. A., Band, L. E., Law, N., Groffman, P. M., Kaushal, S. S., Savvas, K., ... & Belt, K. T. (2008). Streamflow distribution of non–point source nitrogen export from urban-rural catchments in the Chesapeake Bay watershed. *Water Resources Research*, *44*(9), 1-13.

Siders, A. R., Hino, M., & Mach, K. J. (2019). The case for strategic and managed climate retreat. *Science*, *365*(6455), 761-763. https://doi.org/10.1126/science.aax8346

Spielman, S. E., Folch, D., & Nagle, N. (2014). Patterns and causes of uncertainty in the American Community Survey. *Applied Geography*, *46*, 147–157. https://doi.org/10.1016/j.apgeog.2013.11.002

Stern, N. (2008). The economics of climate change. *American Economic Review*, *98*(2), 1-37.

Strohbach, M. W., Döring, A. O., Möck, M., Sedrez, M., Mumm, O., Schneider, A., . . . Schröder, B. (2019). The “hidden urbanization”: Trends of impervious surface in low-density housing developments and resulting impacts on the water balance. *Frontiers in Environmental Science*, *7*. https://doi.org/10.3389/fenvs.2019.00029

Stopka, T. J., Krawczyk, C., Gradziel, P., & Geraghty, E.M. (2014). Use of spatial epidemiology and hot spot analysis to target women eligible for prenatal women, infants, and children services. *American Journal of Public Health*, *104(*1), S183-S189. https://doi.org/10.2105/AJPH.2013.301769

Sussams, L., Sheate, W., & Eales, R. (2015). Green infrastructure as a climate change adaptation policy intervention: Muddying the waters or clearing a path to a more secure future? *Journal of Environmental Management*, *147*, 184-193. https://doi.org/10.1016/j.jenvman.2014.09.003

Takao, K., Motoyoshi, T., Sato, T., & Fukuzono, T. (2004). Factors determining residents' preparedness for floods in moderl megalopolises: The case of the Tokai flood disaster in Japan. *Journal of Risk Research, 7*(7-8), 775-787.

Teixeira, F. Z., Bachi, L., Blanco, J., Zimmermann, I., Welle, I., & Carvalho-Ribeiro, S. M. (2019). Perceived ecosystem services (ES) and ecosystem disservices (EDS) from trees: insights from three case studies in Brazil and France. *Landscape Ecology*, *34*(7), 1583-1600. https://doi.org/10.1007/s10980-019-00778-y

Thrysøe, C., Balstøm, T., Borup, M., Löwe, R., Jamali, B., & Arnbjerg-Nielsen, K. (2021). Floodstroem: A fast dynamic GIS-based urban flood and damage model. *Journal of Hydrology, 600,* 126521. https://doi.org/10.1016/j.jhydrol.2021.126521

Trenberth, K. (2011). Changes in precipitation with climate change. *Climate Research,* 47(1), 123-138. https://doi.org/10.3354/cr00953
Tu, M., Wadzuk, B., & Traver, R. (2019). Methodology to simulate unsaturated zone hydrology in Storm Water Management Model (SWMM) for green infrastructure design and evaluation. PLoS ONE, 15(7), e0235528. https://doi.org/10.1371/journal.pone.0235528

United Nations. (2015). Sendai framework for disaster risk reduction 2015-2030. Sendai, Japan: United Nations Office for Disaster Risk Reduction (UNISDR).

U.S. Census Bureau. (2011). *Press Release*. Retrieved from https://www.census.gov/newsroom/blogs/random-samplings/2011/07/what-are-census-blocks.html

United States Census Bureau. (2020). *Press Release*. Retrieved from https://www.census.gov/newsroom/press-releases/2020/south-west-fastest-growing.html

United States Environmental Protection Agency (2015). Stormwater Management Model (Version 5.1). Software. Retrieved from https://www.epa.gov/water-research/storm-water-management-model-swmm

Vamvakeridou-lyroudia, L. S., Chen, A. S., Khoury, M., Gibson, M. J., Kostaridis, A., Stewart, D., … & Savic, D. A. (2020). Assessing and visualizing hazard impacts to enhance the resilience of critical infrastructures to urban flooding. *Science of the Total Environment*, *707,* 136078. https://doi.org/10.1016/j.scitotenv.2019.136078

Van Coppenolle, R., & Temmerman, S. (2019). A global exploration of tidal wetland creation for nature-based flood risk mitigation in coastal cities. *Estuarine, Coastal, and Shelf Science*, *226*, 106262. https://doi.org/10.1016/j.ecss.2019.106262

Van Coppenolle, R., & Temmerman, S. (2020). Identifying ecosystem surface areas available for nature-based flood risk mitigation in coastal cities around the world. *Estuaries and Coasts*, *43*, 1335-1344. https://doi.org/10.1007/s12237-020-00718-z

Verstraeten, G., & Poesen, J. (1999). The nature of small-scale flooding, muddy floods and retention pond sedimentation in central Belgium. *Geomorphology*, *29*(3-4), 275-292. https://doi.org/10.1016/S0169-555X(99)00020-3.

Vietz, G. J., Walsh, C. J., & Fletcher, T. D. (2016). Urban hydrogeomorphology and the urban stream syndrome: Treating the symptoms and causes of geomorphic change. *Progress in Physical Geography*, *40*(3), 480-492. https://doi.org/10.1177/0309133315605048

Vivoni, E. R., Moreno, H. A., Mascaro, G., Rodríguez, J. C., Watts, C. J., Garatuza-Payán, J., ... & Scott, R. L. (2008). Observed relation between evapotranspiration and soil moisture in the North American monsoon region. *Geophysical Research Letters, 35*, l22403. https://doi.org/10.1029/2008GL036001

Voelkel, J., Hellman, D., Sakuma, R., & Shandas, V. (2018). Assessing vulnerability to extreme heat: A study of disproportionate heat exposure and access to refuge by socio-demographic status in Portland, Oregon. *International Journal of Environmental Research and Public Health, 15*, 640. https://doi.org/10.3390/ijerph15040640

Walsh, C. J., Roy, A. H., Feminella, J. W., Cottingham, P. D., Groffman, P. M., & Morgan, R. P. (2005). The urban stream syndrome: current knowledge and the search for a cure. *Journal of the North American Benthological Society*, *24*(3), 706–723. https://doi.org/10.1899/04-028.1

Wong, C. P., Jiang, B., Bohn, T. J., Lee, K. N., Lettenmaier, D. P., Ma, D., & Ouyang, Z. (2017). Lake and wetland ecosystem services measuring water storage and local climate regulation. *Water Resources Research*, *53*, 3197–3223. https://doi.org/10.1002/2016WR019445

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

Wolfs, V., & Willems, P. (2016). Modular conceptual modelling approach and software for sewer hydraulic computations. *Water Resources Management*, *31*(1), 283-298. https://doi.org/10.1007/s11269-016-1524-2

World Meteorological Organization (2021). *State of the Global Climate 2021: WMO Provisional Report*. Retrieved from https://library.wmo.int/index.php?lvl=notice_display&id=21982#.YjoTKC-B1pQ

Wu, Q., Zhou, H., Tam, N. F. Y., Tian, Y., Tan, Y., Zhou, S., ... Leung, J. Y. S. (2016). Contamination, toxicity and speciation of heavy metals in an industrialized urban river: Implications for the dispersal of heavy metals. *Marine Pollution Bulletin*, *104*(1-2), 153-161. https://doi.org/10.1016/j.marpolbul.2016.01.043.

Xu, X., Chen, M., Yang, G., Jiang, B., & Zhang, J. (2020). Wetland ecosystem services research: A critical review. *Global Ecology and Conservation*, *22*, e01027. https://doi.org/10.1016/j.gecco.2020.e01027

Yang, L., Smith, J., & Niyogi, D. (2019). Urban impacts on extreme monsoon rainfall and flooding in complex terrain. *Geophysical Research Letters*, *46*(11), 5918–5927. https://doi.org/10.1029/2019gl083363

Yin, J., Yu, D., Yin, Z., Liu, M., & He, Q. (2016). Evaluating the impact and risk of pluvial flash flood on intra-urban road network: A case study in the city center of Shanghai, China. Journal of Hydrology, *357*, 138-145. https://doi.org/10.1016/j.jhydrol.2016.03.037

Zabini, F., Grasso, V., Crisci, A., Gozzini, B. (2021). How do people perceive flood risk? Findings from a public survey in Tuscany, Italy. *Journal of Flood Risk Management*, *14*(1), 12694. https://doi.org/10.1111/jfr3.12694

Zahran, S, Brody, S. D., WG Peacock, W. G., Vedlitz, A., & Grover, H. (2008). Social vulnerability and the natural and built environment: A model of flood casualties in Texas. *Disasters*, *32*(4), 537-560. https://doi.org/10.1111/j.1467-7717.2008.01054.x

Zedler, J. B., & Kercher, S. (2005). Wetland resources: Status, trends, ecosystem services, and restorability. *Annual Review of Environment and Resources*, 30, 39-74.

Zhao, G., Balstrøm, T., Mark, O., & Jensen, M. B. (2021). Multi-scale target-specified sub-model approach for fast large-scale high-resolution 2D urban flood modelling. *Water*, *13*, 259. https://doi.org/10.3390/w13030259

Zhou, Q., Mikkelsen, P. S., Halsnæs, K., & Arnbjerg-Nielsen, K. (2012). Framework for economic pluvial flood risk assessment considering climate change effects and adaptation benefits. *Journal of Hydrology*, *414*, 539-549. https://doi.org/10.1016/j.jhydrol.2011.11.031

APPENDIX A

TABLE OF RESULTS OF LINEAR REGRESSION FOR ANNUAL RAINFALL

OVER VARIOUS TIME PERIODS

Values indicate R^2. * indicates $p < 0.05$; ** indicates $p < 0.01$.

	RCP 4.5				**RCP 8.5**			
Model	*1969–2015*	*1969–2021*	*2016–2080*	*1969–2080*	*1969–2015*	*1969–2021*	*2016–2080*	*1969–2080*
ACCESS1-0	0.165**	0.189**		0.127**			0.239**	0.240**
ACCESS1-3				0.125**			0.149**	0.150**
BCC-CSM1-1				0.039*			0.070*	0.177**
BCC-CSM1-1-M	0.088**	0.104**		0.067**	0.186**	0.101*	0.098*	0.216**
BNU-ESM				0.094**			0.073*	0.136**
CanESM2			0.123**	0.263**			0.357**	0.413**
CCSM4							0.105**	0.083**
CMCC-CM				0.14**			0.197**	0.216**
CNRM-CM5							0.180**	0.099**
CSIRO-MK3.6			0.085*	0.042*				0.056*
GFDL-ESM2G	0.174**	0.217**	0.079*	0.218**	0.083*	0.156**	0.097*	0.260**
GFDL-ESM2M	0.084*	0.087*	0.100*	0.233**			0.169**	0.319**
HadGEM2-CC			0.063*	0.065**		0.102*		0.143**
HadGEM2-ES			0.149**	0.123**			0.177**	0.217**
INMCM4	0.129*	0.104*					0.120**	0.150**
IPSL-CM5A-LR		0.122*		0.211**		0.119*	0.173**	0.299**
IPSL-CM5A-MR			0.138**	0.295**	0.084*		0.154**	0.239**
MIROC5			0.138**	0.125**			0.429**	0.286**
MIROC-ESM			0.084*	0.100**			0.283**	0.229**
MIROC-ESM-CHEM		0.091*		0.110*			0.302**	0.253**
MPI-ESM-LR				0.044*			0.238**	0.201**
MPI-ESM-MR	0.085*	0.076*		0.054*			0.192**	0.187**
NorESM1-M				0.07**			0.067*	0.190**
Ensemble average	0.434**	0.514**	0.404**	0.700**	0.344**	0.306**	0.803	0.830**

APPENDIX B

TABLE OF RESULTS OF LINEAR REGRESSION FOR 100-YEAR RETURN PERIOD, 24-HOUR DURATION STORM OVER VARIOUS TIME PERIODS

Values indicate R^2 * indicates $p < 0.05$; ** indicates $p < 0.01$.

	RCP 4.5				RCP 8.5			
	1998–2015	*1998–2021*	*2016–2080*	*1998–2080*	*1998–2015*	*1998–2021*	*2016–2080*	*1998–2080*
ACCESS1-0	0.723**	0.775**	0.718**	0.429**		0.581**	0.531**	0.069*
ACCESS1-3	0.326*	0.471**	0.177**	0.131**	0.853**	0.889**	0.205**	
BCC-CSM1-1	0.871**	0.854**	0.237**	0.180**	0.648**	0.626**	0.097*	0.423**
BCC-CSM1-1-M			0.866**	0.860**	0.296*		0.097*	
BNU-ESM	0.870**	0.648**	0.650**	0.595**	0.553**	0.388**	0.739**	0.773**
CanESM2	0.557**	0.797**	0.249**	0.625**	0.573**	0.776**	0.355**	
CCSM4	0.734**	0.795**	0.473**	0.455**	0.719**	0.745**	0.312**	0.615**
CMCC-CM	0.311*		0.647**	0.766**	0.222*	0.282**	0.420**	0.528**
CNRM-CM5	0.719**			0.140**		0.492**	0.319**	0.052*
CSIRO-MK3.6		0.351**		0.199**	0.888**	0.906**	0.664**	0.800**
GFDL-ESM2G	0.832**	0.195*	0.824**	0.776**			0.812**	0.838**
GFDL-ESM2M	0.747**	0.670**	0.304**	0.194**			0.841**	0.736**
HadGEM2-CC	0.799**	0.889**	0.335**	0.107**	0.782**	0.872**	0.813**	0.900**
HadGEM2-ES	0.314*		0.339**	0.537**		0.392**		
INMCM4	0.786*	0.707**	0.179**		0.585**		0.101*	
IPSL-CM5A-LR	0.681**	0.681**	0.092*				0.580**	0.325**
IPSL-CM5A-MR	0.788**	0.773**	0.229**		0.429**		0.389**	0.465**
MIROC5	0.845**	0.712**				0.435**		0.270**
MIROC-ESM	0.695**	0.858**	0.484**	0.653**	0.870**	0.927**	0.515**	0.754**
MIROC-ESM-CHEM	0.547**	0.686**	0.182**		0.459**	0.277**	0.806**	0.843**
MPI-ESM-LR	0.574**	0.418**	0.774**	0.874**	0.578**	0.450**	0.590**	0.779**
MPI-ESM-MR	0.313**		0.58**	0.410**	0.853**	0.781**	0.780**	0.284**
NorESM1-M		0.645**	0.282**	0.068*		0.439**	0.750**	0.493**
Ensemble average	0.944**	0.909**	0.884**	0.916**	0.914**	0.889**	0.953**	0.973**

APPENDIX C

SURVEY QUESTIONS IN SPANISH, FOLLOWED BY SURVEY QUESTIONS IN ENGLISH

Survey questions in Spanish

P1: Cree que las inundaciones son un problema en la ciudad de Hermosillo?

P2: Ha experimentado una inundación en esta área?

A los participantes se les mostrará un mapa con una zona demarcada con un círculo azul claro. El mapa tendrá los puntos de referencias locales, como gasolineras, bancos, etc.

> P3: Por favor, podría dibujar un círculo alrededor de las ubicaciones en el mapa donde ha experimentado personalmente o sabes que otras han experimentado inundaciones que se producen durante la temporada de lluvias? La idea de "experiencias" puede describir impactos personales, tal como daño a propiedad, la interrupción del uso de aceras o calles, o puede describir un lugar donde ha visto inundación.
>
> P3a: Para cada ubicación, podría describir cómo el área le impactó? Por ejemplo, el área inundada le impidió andar en algún lado como una tienda de abarrotes? Le impidió manejar? La inundación causó colapso en las tuberías en el drenaje en su casa?
>
> P3b: Para cada ubicación, podría estimar que tan profunda fue la inundación? Hasta el tobillo, la rodilla, la cintura, o más alta?
>
> P3c: Por cuanto tiempo experimentó los impactos de inundación en las áreas que proporcionó?
>
> P3d: Si experimentó daño de una inundación, como una interrupción de trabajar o daño a su propiedad, recuerda cuánto dinero tuvo que pagar para solucionar el problema causado por el evento de inundación?
>
> P3e: Podría valorar en una escala del 1a 5 cada área de inundación que ha indicado por el peligro que se presenta a ud. o a su propiedad. 1 es el menos peligroso, 5 es el más peligroso.
>
> P3f: Ha considerado comprar seguro de inundación para su casa, propiedad, o negocio? Recuerda el costo?

P4: Cuál es su principal medio de transporte?

Survey questions, in English

P1: Do you believe that flooding is a problem in Hermosillo?

P2: Have you experienced flooding in this area?

Show the participants a map with an area marked with a clear blue circle. The map will have local reference points, such as gas stations, banks, etc.

P3: Could you please draw a circle around the locations in the map where you have personally experienced flooding or where you know others have experienced flooding during the rainy season? The idea of experiences here can describe personal impacts, such as damage of property, interruption of the use of sidewalks or streets, or it could describe a location where you have seen flooding.

P3a: For each location, could you describe how the area impacted you? For example, the flooded area impeded you from walking on some side, like near a grocery store. Did it impede you from driving? Did flooding overwhelm the drainage system at your house?

P3b: For each location, could you estimate how deep was the flooding? To the ankle, knee, waist, or higher?

P3c: For how much time were you impacted by flooding in the areas you provided?

P3d: If you experienced damage from a flood, such as interruption of work or damage to property, do you remember how much money you had to pay to resolve the problem caused by the flooding event?

P3e: On a scale of 1 to 5, could you evaluate for each of the areas of flooding you indicated the danger that flooding presented to you or your property? 1 is the least dangerous, 5 is the most dangerous.Podría valorar en una escala del 1 a 5 cada área de inundación que ha indicado por el peligro que se presenta a ud. o a su propiedad. 1 es el menos peligroso, 5 es el más peligroso.

P3f: Have you considered buying flood insurance for your house, property, or business? Do you remember the cost?

P4: What is your main form of transportation?