12-2020

Water, Fish, and Fire: Interdisciplinary Research on Ecosystem Services and Climate Adaptation

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WATER, FISH, AND FIRE: INTERDISCIPLINARY RESEARCH ON

ECOSYSTEM SERVICES AND CLIMATE ADAPTATION

by

Liana Prudencio

A dissertation submitted in partial fulfillment
of the requirements for the degree

of

DOCTOR OF PHILOSOPHY

in

Watershed Science

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UTAH STATE UNIVERSITY
Logan, Utah

2020

ABSTRACT

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Department: Watershed Sciences

Ecosystem services, or benefits from the environment, have been negatively impacted due to anthropogenic activities and climate change in every region of the world. This dissertation explores multiple services, from water quality improvement to provisioning of fish and habitat, at varied scales and locations to provide a multi-faceted and interdisciplinary study of ecosystem services.

The first chapter synthesizes the literature on stormwater management and ecosystem services, finding that research at this intersection has provided many parcel-level studies and frameworks for implementing green infrastructure. I conclude with recommendations for future work including more studies that quantify services and upscale green infrastructure to a larger, watershed scale.

The second chapter uses QUAL2Kw to simulate watershed scale effects of green infrastructure on downstream ecosystem services. The study watershed is in the Salt Lake Valley, UT, USA, where urbanization has altered hydrology and water quality. Green

infrastructure alternatives in approximately 13 percent of the urban area of seven tributary watersheds to the Jordan River leads to at most a 9.3% and 9% reduction in streamflow, a 17.4% and 0.44% decrease in stream temperature, a 1.3% increase and 1% decrease in dissolved oxygen, and a 1.2% and 8.6% reduction in total phosphorus at Great Salt Lake, under winter/spring and late summer conditions respectively.

The third chapter concentrates on fire trends and adaptive management in the American Intermountain West. Climate change and human populations moving into the wildland-urban interface have increased fire frequency and area burned. The findings of this study also contribute to our understanding of the economic impacts of fire and how fire managers are adapting their actions and policies to changing conditions.

The final chapter evaluates cues for fish migrations in the Lower Mekong Basin, a region experiencing heavy and increasing fishing pressure that threatens the provisioning of fish, livelihoods, and food security for millions in the Tonle Sap system of Cambodia. Hydrologic predictors are evaluated and ranked to understand environmental cues that fish rely on for migration. Results show that changes in timing, duration, and magnitude of flows from hydropower development pose risks for many migratory fish species in this region.

(256 pages)

PUBLIC ABSTRACT

Water, Fish, and Fire: Interdisciplinary Research on

Ecosystem Services and Climate Adaptation

Liana Prudencio

Ecosystem services, or benefits from the environment, are plentiful and vary from place to place. Human activities and climate change have impacted these services in every region of the world. This dissertation explores multiple ecosystem services, from water quality improvement to provisioning of fish and habitat, in multiple and international contexts. The first chapter synthesizes the literature on stormwater management and ecosystem services, finding that research at this intersection has provided many parcel-level studies and frameworks for implementing green infrastructure. The second chapter extends the stormwater management literature by quantifying the impacts of green infrastructure on water quantity and quality at the watershed scale, showing that various amounts of green stormwater infrastructure lead to reduction in peak flow and water quality improvements via reductions in total phosphorus loadings. The third chapter contributes to our understanding of fire trends in the Intermountain West, the economic impacts of fire, and how fire managers are adapting their actions and policies. The final chapter extends this dissertation to the Lower Mekong Basin, which is experiencing heavy fishing pressure that threatens the livelihoods and food security for millions in the Tonle Sap system of Cambodia. The results show that changes in timing, duration, and magnitude of flows from hydropower development pose risks for many migratory fish in this region. With interdisciplinary approaches, these chapters have led to a multi-faceted study of ecosystem services.

ACKNOWLEDGMENTS

Completing my graduate studies and dissertation would not have been possible without the help of so many. I have had the pleasure of participating in multiple research teams that made my work possible, and I would like to thank all of my collaborators. Thank you to the members of the EPA-STAR green stormwater infrastructure team, to the Wonders of the Mekong team, and to my CAS cohort, CAS faculty, and Dr. Nancy Huntly. Thank you to past and present members of the ACWA lab for their help in preparing numerous presentations and bouncing ideas around. I want to thank my committee, Drs. Ryan Dupont, Joanna Endter-Wada, Edd Hammill, and Karin Kettenring, for their support and valuable, wide-ranging perspectives throughout my studies.

Special thanks to Dr. Dan McCool for wonderful research opportunities, and for encouraging me to run with my passion for interdisciplinary research and pursue a PhD in the physical sciences after getting degrees in journalism and the social sciences. He is a true role model in the way he advocates for people and better policy.

I would also like to thank my major advisor, Dr. Sarah E. Null, for taking me in and helping me grow as a scientist. Time and time again she shows up for me and finds the right moments to push me to be better. I am very appreciative of her example and mentorship.

Lastly, thank you to my husband, Jon, and our baby, Bayani, for their love and support as I pursue my professional aspirations. The work that I do has even more meaning with you by my side. Ikaw ang mundo ko.

Liana Prudencio

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CHAPTER 1

INTRODUCTION

Ecosystems provide humans with sustenance, livelihoods, recreation, and cultural significance. *Ecosystem services* are benefits that humans receive and rely on from ecosystems (Brauman et al., 2007). There are four types of ecosystem services: 1) provisioning services (e.g., food, energy, and water), 2) regulating services (e.g., climate regulation, water purification, and flood mitigation), 3) cultural services (e.g., aesthetics, education, and recreation), and 4) supporting services (e.g., habitat and biodiversity) (Brauman et al., 2007). Ecosystem services are produced when ecosystems are healthy and sustained by physical, chemical, and biological processes. The ecosystem services framework is increasingly used in studies across many disciplines, because it enables collaboration among specializations and pushes research beyond conventional science boundaries (Lundy and Wade, 2011).

Anthropogenic activities and climate change have altered ecosystem functions and consequently ecosystem services, and researchers and stakeholders aim to restore and maintain ecosystem services of interest (Ehrenfeld, 2000). The restoration and management of ecosystem services is a popular objective for management projects and programs due to public support for environmental benefits to humans (*Ibid*.). Effectively managing for ecosystem services sometimes leads to healthier ecosystems that are consequently more resilient and adaptive to climate change (Munang et al., 2013). This idea is the foundation of this dissertation.

The first chapter reviews research at the intersection of stormwater management and ecosystem services. The objective is to synthesize existing work and outline the

research needed to further the literature on ecosystem services related to stormwater. A systematic review of 170 articles shows that research on stormwater management and ecosystem services has increased over time. The literature so far consists largely of site-level studies and frameworks for green infrastructure implementation. Research on green stormwater infrastructure has started to move toward integrating engineering, physical science, and social science approaches to achieve sustainable and effective stormwater management, although more research contributions on this multidisciplinary path are needed.

The second chapter uses simulation modeling to answer two questions: 1) What are the effects of green stormwater infrastructure on surface water quantity and quality at reach, small watershed, and large watershed scales?; and 2) Which types of green infrastructure lead to the largest improvements in water quantity and quality at different spatial scales? By modeling alternative types of green infrastructure at different scales, I evaluate how green infrastructure can be used to manage water quality improvement, flood mitigation, and water supply.

The third chapter is an interdisciplinary study on fire management in the U.S. Intermountain West (IMW). There are three research questions that assess adaptive fire management in this region: 1) Are area burned and fire frequency increasing within the IMW?; 2) Do fires in urban or rural settings influence employment trends in local economies, and if so, how?; and 3) Do trends in fire characteristics and economic impacts of fire influence perspectives of managers and adaptive decision-making, and if so, how? Through an in-depth understanding of fire trends in this region, its impact on economies, and the challenges fire managers face in their decision-making, we can better develop

tools and policies that support adaptive fire management strategies and decisions.

Lastly, the fourth chapter statistically models environmental conditions that cue migratory fish to move in the Tonle Sap River, a major tributary to the Mekong River in Southeast Asia. Using observations of catch weight for six species over time, historical random forests ranked predictors of fish migration. The goal of this chapter is to understand the environmental conditions that support the life cycles of migratory fish and to highlight the effects of a changing climate and continued water development on fish movement.

Overall, the research presented here explores the impacts on ecosystem services from development and management, with a focus on how these services help to adapt to and alleviate climate change impacts. Interdisciplinary is another theme of this research, which is needed to address multiple social and physical facets of ecosystem services and climate adaptation. The first two chapters are contributions to a project with environmental and civil engineers and sociologists on the use of green stormwater infrastructure in the Salt Lake Valley in UT, USA. With Chapter 1, I review research by various researchers in different disciplines that is focused on ecosystem services related to stormwater management. Chapter 2 involves ecosystem services modeling alongside stakeholders and water managers in the Salt Lake Valley. Conducted with ecologists, social scientists, an applied economist, and watershed scientists, the Chapter 3 study helps understand the barriers fire managers face in their effort to adapt to changing fire trends from climate change. Chapter 4 integrates the fields of biology, ecology, and hydrology in a study that is part of a larger project on sustainable development in the Mekong River in Cambodia.

Taken as a whole, this dissertation illustrates complexity and diversity of ecosystem services and climate adaptation research, which requires approaches from multiple disciplines. It has been a privilege to conduct research that crosses disciplinary lines and creates connections among individuals with different expertise, coming together to develop science that will inform decisions for better resource management and policy. My experience with interdisciplinary research has given me skills to speak different disciplinary languages, as well as skills to readily find common ground. Now, when I am faced with a problem, I consider ways that different disciplines may approach it and alternative tools other scientists would use. I have had the opportunity to share this research with various audiences using journal publications, conference presentations, stakeholder workshops, social media posts, and blogs. My research and this dissertation provide clear examples of how to contribute research to multiple disciplines for the end goal of effective management of ecosystem services and climate adaptation.

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CHAPTER 2

STORMWATER MANAGEMENT AND ECOSYSTEM SERVICES: A REVIEW[1]

Abstract

Researchers and water managers have turned to green stormwater infrastructure, such as bioswales, retention basins, wetlands, rain gardens, and urban green spaces to reduce flooding, augment surface water supplies, recharge groundwater, and improve water quality. It is increasingly clear that green stormwater infrastructure not only controls stormwater volume and timing, but also promotes ecosystem services, which are the benefits that ecosystems provide to humans. Yet, there has been little synthesis focused on understanding how green stormwater management affects ecosystem services. The objectives of this paper are to review and synthesize published literature on ecosystem services and green stormwater infrastructure and identify gaps in research and understanding, establishing a foundation for research at the intersection of ecosystems services and green stormwater management. We reviewed 170 publications on stormwater management and ecosystem services, and summarized the state-of-the-science categorized by the four types of ecosystem services. Major findings show that: 1) most research was conducted at the parcel-scale and should expand to larger scales to more closely understand green stormwater infrastructure impacts, 2) nearly a third of papers developed frameworks for implementing green stormwater infrastructure and highlighted barriers, 3) papers discussed ecosystem services, but less than 40% quantified ecosystem services, 4) no geographic trends emerged, indicating interest in applying green stormwater infrastructure across different contexts, 5) studies increasingly integrate

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[1] Co-author: Sarah E. Null

disciplines and should fuse engineering, physical science, and social science approaches for holistic understanding, and 6) standardizing green stormwater infrastructure terminology would provide a more cohesive field of study than the diverse and often redundant terminology currently in use. We recommend that future research provide metrics and quantify ecosystem services, integrate disciplines to measure ecosystem services from green stormwater infrastructure, and better incorporate stormwater management into environmental policy. Our conclusions outline promising future research directions at the intersection of stormwater management and ecosystem services.

Introduction

Stormwater runoff provides ecosystem services, or benefits to people from the environment, including soil moisture, interflow, baseflow, groundwater recharge, and filtration of water through the environment (Barbosa et al., 2012; Burns et al., 2012; Roy et al., 2008; Walsh et al., 2016). Urbanization and increased population density alter land cover and land use, typically increasing impervious surfaces, such as asphalt, concrete, and buildings (Barbosa et al., 2012). Conventional stormwater management directly routes runoff to nearby bodies of water through storm drains, gutters, and underground systems, and is also known as gray infrastructure. Gray stormwater infrastructure reduces ecosystems services from stormwater (Roy et al., 2008) by reducing infiltration and groundwater recharge, and contaminating stormwater as runoff over impervious surfaces picks up pollutants such as heavy metals, suspended solids, nutrients, salts, oil and hydrocarbons (Tsihrintzis and Hamid, 1997).

Additionally, climate change affects stormwater and urban runoff. For example, snowfall is anticipated to shift to rainfall in mountain regions, resulting in increased

winter rainfall and runoff. Winter runoff is considered a hazard, whereas spring snowmelt runoff is considered a water resources benefit (Knowles et al., 2006). Climate change may reduce summer baseflow in rivers, despite wet winters (Null and Prudencio, 2016). Also, inter-annual variability is expected to increase with climate change (Thornton et al., 2014), leading to a re-distribution of wet and dry years (Null and Viers, 2013; Rheinheimer et al., 2016). Very wet water years are likely to increase urban runoff and present changing conditions, and opportunities, for green stormwater infrastructure.

Researchers and water managers have started to investigate the effectiveness of green stormwater infrastructure, such as bioswales, retention and detention basins, rain barrels, green spaces, wetlands, green roofs, permeable pavements, and deep infiltration wells to reduce flooding, augment surface water supplies, recharge groundwater, and improve water quality (Burns et al., 2012; Dhakal and Chevalier, 2016; Roy et al., 2008). Green stormwater infrastructure research increasingly shows that the benefits of stormwater management transcend controlling runoff volume and timing, but also provide valued ecosystem services, such as improved water quality, groundwater replenishment, recreation opportunities, and creation of diverse habitats (Dhakal and Chevalier, 2016; Vogel et al., 2015). Green stormwater infrastructure may counter impacts from urbanization while also increasing natural capacity to buffer for anticipated climate change (Barbosa et al., 2012; Hamel et al., 2013; Pyke et al., 2011; Stephens et al., 2012).

Alternative stormwater management practices have a number of terms, including best management practices, green infrastructure, low-impact development, managed aquifer recharge, and stormwater harvesting (Hoss et al., 2016; Vogel et al., 2015). In

this paper, we use the terms ‘gray stormwater infrastructure’ for engineered systems that directly route stormwater to downstream water bodies in urban or developed areas and ‘green stormwater infrastructure’ for alternative stormwater management that generates both human and ecosystem services (Keeley et al., 2013). We focus on green infrastructure implemented specifically to manage stormwater.

Ecosystem services frameworks are increasingly used in research to categorize and measure benefits that ecosystems provide to humans (Coutts and Hahn, 2015). Ecosystem services are generally categorized into four types: 1) *provisioning*, such as water supply and production of food and energy, 2) *regulating*, such as temperature regulation and water purification, 3) *cultural*, such as aesthetics and recreation, and 4) *supporting*, such as habitat for aquatic and riparian species (Burns et al., 2012; Cameron and Blanusa, 2016; Kopperoinen et al., 2014; Walsh et al., 2016). Through classifying stormwater research into an ecosystem services framework, we can understand changes to ecosystem services from urbanization and quantify benefits of shifting from gray to green stormwater infrastructure with anticipated global environmental change. Figure 2-1 shows (a) ecosystem services related to stormwater in natural environments and (b) how ecosystem services change due to urbanization coupled with climate change. As shown in the figure, ecosystem services, such as water purification, water infiltration, and groundwater storage are impaired in the urban environment from impervious surfaces, exposure to urban pollutants, and gray stormwater infrastructure.

To date, there has been no systematic review of research at the intersection of green stormwater management and ecosystem services. The objectives of this paper are to 1) review and synthesize published literature at the intersection of these topics and 2)

identify knowledge gaps that could better inform decisions and policies on green stormwater infrastructure for ecosystem services. The synthesis provided will direct future stormwater management research and aid researchers and policy-makers in managing stormwater sustainably.

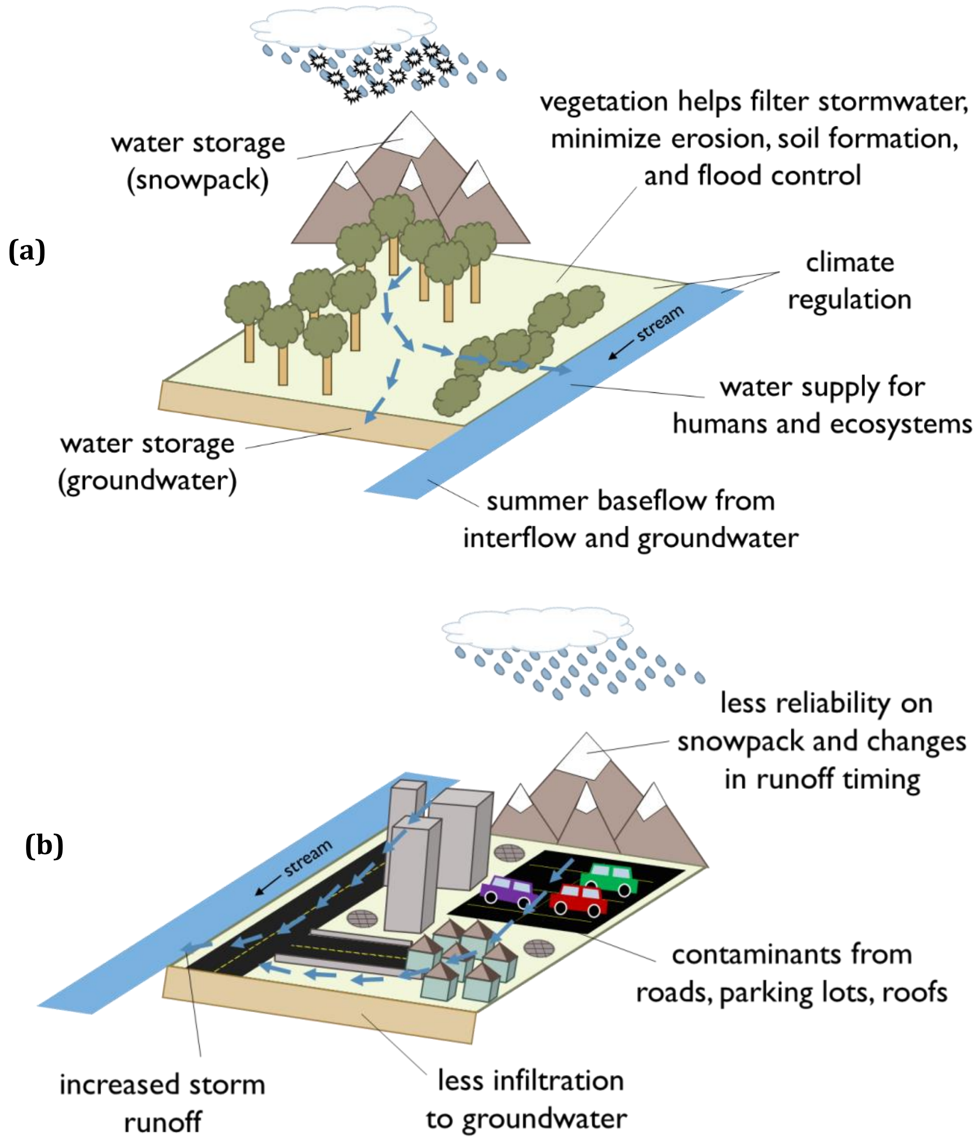

Figure 2-1. (a) Ecosystem services related to stormwater in natural environments and (b) Environmental impacts from gray stormwater infrastructure, urbanization, and climate change.

Methods/Design

We searched primary literature publications in Thomson ISI Web of Science (1975 to 2017), Water Resources Abstracts (1967 to 2017), Sustainability Science Abstracts (1995 to 2017), and Scopus (1823 to 2017) databases that included the terms "stormwater" (or "storm water") and "ecosystem services", as well as at least one green stormwater infrastructure term anywhere in the text (Table 2-1). Researchers and managers use multiple terms for green stormwater infrastructure. These include broad descriptions, such as green infrastructure and low impact development, and specific types of infrastructure such as retention basins, wetlands, and green spaces (Greenway, 2015; Klimas et al., 2016a; Kopecka et al., 2017; Pataki et al., 2011). Our search was inclusive of these terms as long as the publication focused on green stormwater management and ecosystem services-related topics. The search returned 216 results from all four databases through October 2017, with 170 papers ultimately retained that focus on green stormwater management and ecosystem services.

Following the search in the four databases, each article was reviewed and coded by the category of ecosystem services it addressed, as well as sub-categories of ecosystem services (Table 2-2). An article could address multiple ecosystem services types. We evaluated how the articles quantified and discussed each of the four categories of ecosystem services to understand benefits of green infrastructure, highlight categories that are under-represented in the literature, and identify where further ecosystem services-stormwater management research is needed.

Table 2-1. Search terms

"stormwater" ***OR*** *"storm water"* ***AND***
"ecosystem services" ***AND***
Any of the following green stormwater management-related terms: ▪ *"green infrastructure"* ▪ *"managed aquifer recharge"* ▪ *"low impact development"* ▪ *"best management practices"* ▪ *"stormwater harvesting"* ▪ *"stormwater capture"* ▪ *"green roofs"* ▪ *"basins"* ▪ *"wells"* ▪ *"rain barrels"* ▪ *"wetlands"* ▪ *"ponds"* ▪ *"permeable pavement"* ▪ *"permeable surfaces"* ▪ *"pervious pavement"* ▪ *"pervious surfaces"* ▪ *"rain gardens"* ▪ *"tree boxes"* ▪ *"swales"* ▪ *"r-tanks"* ▪ *"underground vaults"* ▪ *"green space"* ▪ *"sustainability"* ▪ *"climate adaptation"* ▪ *"management"*

Results and Synthesis

The number of stormwater management publications that discuss ecosystem services substantially increased since 2005, when the first paper on these topics was published (Figure 2-2). The number of stormwater papers on provisioning and regulating ecosystem services has been increasing faster than publications on cultural and supporting ecosystem services (Figure 2-3). Table 2-2 categorizes the number of articles that discuss the four types of ecosystem services, as well as the most prominent

subcategories of ecosystem services. We synthesize each category in the following four sections.

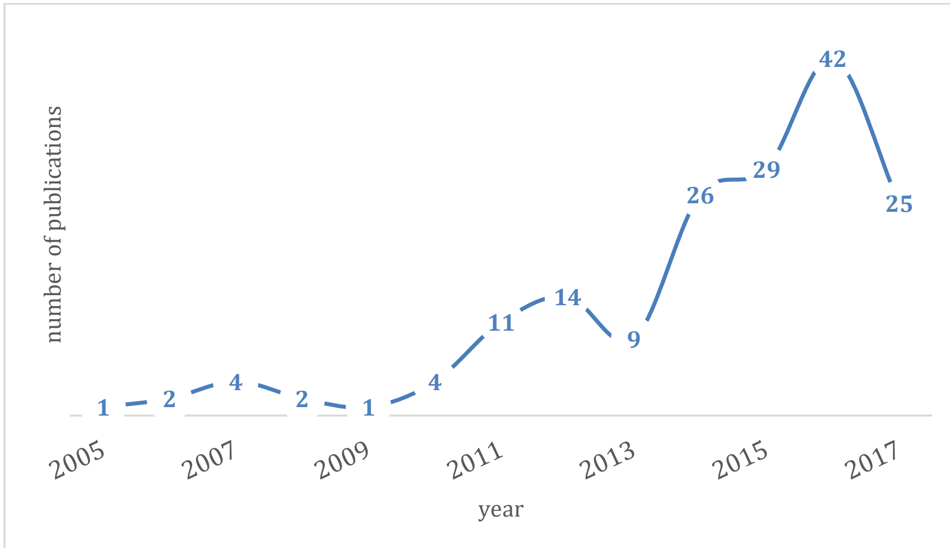

Figure 2-2. Number of stormwater-ecosystem services publications over time.

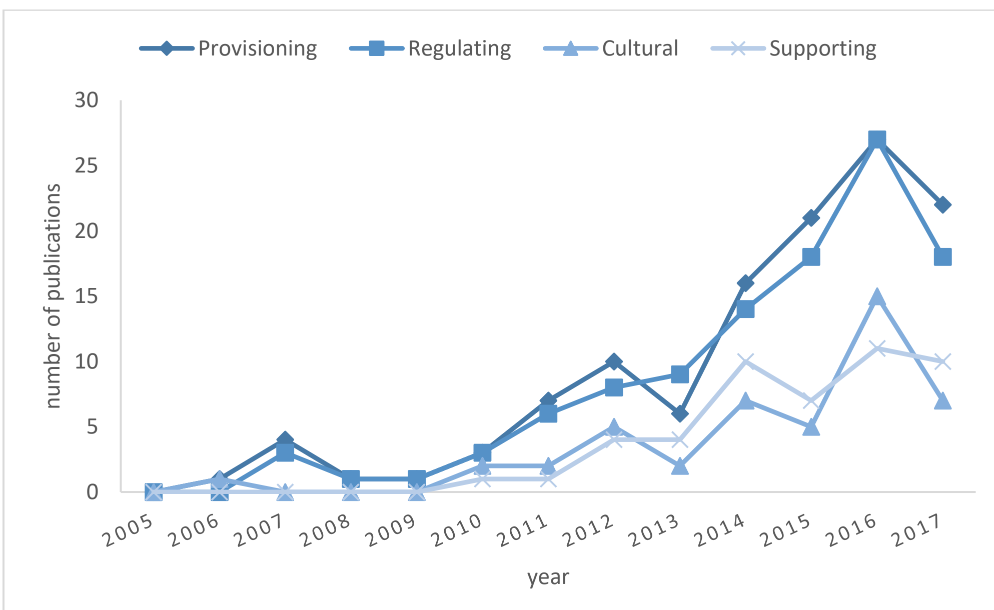

Figure 2-3. Number of stormwater-ecosystem services publications over time by ecosystem service category.

Table 2-2. Number of articles by ecosystem service category and example references by subcategory

Category	*Number of Publications*	*Subcategories*	*Example References*
Provisioning Services	119	production of vegetation/biotic material for food and energy	(Ackerman, 2012; Berland et al., 2017; Gittleman et al., 2017; Lovell and Taylor, 2013; Mayer et al., 2012; Russo et al., 2017)
		water supply and storage	(Guertin et al., 2015; Lundy and Wade, 2011; Shuster et al., 2007; Voskamp and de Ven, 2015; Xue et al., 2015)
Regulating Services	108	water purification	(Adyel et al., 2016; Bhomia et al., 2015; Dagenais et al., 2017; Heintzman et al., 2015)
		climate regulation	(Buckland-Nicks et al., 2016; Gruwald et al., 2017; Klimas et al., 2016b; Lundholm, 2015; Verbeeck et al., 2014)
		flood control	(Berland and Hopton, 2014; Doherty et al., 2014; Guertin et al., 2015; Ishimatsu et al., 2017)
		carbon sequestration	(Bouchard et al., 2013; Chen et al., 2014; Kremer et al., 2015; McPherson et al., 2011; Merriman et al., 2017)
Cultural Services	46	economic/cultural/social values	(Attwater and Derry, 2017; Garcia-Cuerva et al., 2016; Kati and Jari, 2016a; Kellogg and Matheny, 2006)
		recreation	(Ghermandi, 2016; Kandulu et al., 2014; Kremer et al., 2015; Moore and Hunt, 2012)
		education	(Hassall, 2014; Horsley et al., 2016; Larson, 2010; McDuffie et al., 2015)
Supporting Services	48	biodiversity and habitat	(Attwater and Derry, 2017; Greenway, 2015; Hassall and Anderson, 2015; Kopecka et al., 2017; Taylor and Lovell, 2014)

Provisioning Services – Provisioning ecosystem services were the most common type of ecosystem services discussed in stormwater management papers. Researchers often did not explicitly use the term “provisioning”; however, the ecosystem services they describe fall under this category. Studies on stormwater runoff and green stormwater infrastructure provisioning services focused on water supply and the production of vegetation and biomass for energy, food, and water (Ackerman, 2012; Gittleman et al., 2017; Mayer et al., 2012; Taylor and Lovell, 2014). Cities and urban areas generate water through stormwater detention (Lundy and Wade, 2011). While stormwater in cities creates flooding and pollution, it is often now viewed as a potential resource for water supply enhancement (*Ibid*.).

More specifically, researchers and stakeholders are looking to green stormwater management for climate resilient stormwater storage and supply (Shuster et al., 2007; Voskamp and de Ven, 2015). Climate change and urbanization have challenged water reliability, and planning for sustainable water supply is increasingly pertinent (Xue et al., 2015). While interest in and articles on provisioning ecosystem services have increased over the years, the studies that *quantify* provisioning services, instead of simply mentioning that they exist, are few in number. Most of the articles that examine provisioning services of green stormwater infrastructure do so with discussions of the potential of green infrastructure to enhance stormwater retention for infiltration and water supplies, as well as frameworks for implementation (Voskamp and de Ven, 2015). Some develop approaches, or identify strategies and challenges by outlining case studies (Guertin et al., 2015). For example, Guertin et al. (2015) applied a tool to simulate green infrastructure to maximize water supply on the neighborhood-scale in a semi-arid region,

identifying multiple alternatives for green infrastructure implementation.

Researchers highlighted the significant effects of vegetation and biotic production on streamflow and runoff generation (Berland et al., 2017; Starry et al., 2011; Verbeeck et al., 2014). Berland et al. (2017) outlined the role of urban trees in stormwater management, emphasizing that trees are significantly connected to urban hydrology and can increase infiltration of stormwater. Lastly, researchers studied the provisioning of food from green stormwater infrastructure (Russo et al., 2017). This research identified ecosystem services of sustainably managing stormwater, showing that water management, food security, and community development from edible urban greenery and gardens are inter-related.

Regulating Services – This category closely followed provisioning services in frequency of articles (Figure 2-3). Regulating services of stormwater are sometimes quantified for flood control, water purification, climate regulation, and carbon sequestration from green infrastructure (Berland and Hopton, 2014; Gao et al., 2015; Ishimatsu et al., 2017; McPherson et al., 2011). Researchers such as Gao et al. (2015) modeled water quality improvement and flood mitigation from green stormwater management at the city-scale and found positive results. However, the majority of studies assessed the performance of a single type of green infrastructure, such as green roofs, rain gardens, or stormwater ponds at the parcel-scale to capture and treat stormwater runoff. Smaller scale experiments provided support for nutrient attenuation, flood control, and microclimate mitigation ecosystem services of green stormwater management (Adyel et al., 2016; Wardynski et al., 2012). Multiple studies have investigated the capabilities of green infrastructure to capture and store carbon as well (Bouchard et al., 2013; Chen et

al., 2014; Kremer et al., 2015; McPherson et al., 2011; Merriman et al., 2017). These studies quantified carbon sequestration through carbon accumulation rates, carbon storage potential of vegetation and soil, and similar metrics. Overall, they support carbon sequestration from green infrastructure, with nuances from differing vegetation types and soil conditions (*Ibid*.).

Interestingly, researchers noted tradeoffs between regulating ecosystem services and provisioning services, as well as tradeoffs between different regulating services (Kuoppamaki et al., 2016; Nocco et al., 2016). Kuoppamaki et al. (2016) highlighted that green roofs reduce runoff volume but also expose runoff to more nutrients. Nocco et al. (2016) found tradeoffs between daytime evaporative cooling and nutrient reduction from rain gardens. These scholars argue that regulating services related to green stormwater infrastructure are more nuanced than provisioning services, and require attention to site-specific characteristics, like plant communities, land uses, and soil quality.

<u>Cultural Services</u> – Of the 170 articles reviewed, 46 publications discussed cultural services related to stormwater management (Figure 2-3). Several researchers conducted surveys and interviews with stakeholders, residents, officials, and decision-makers, on the perceptions and values of ecosystem services from green stormwater infrastructure (Kati and Jari, 2016a; Welsh and Mooney, 2014). Overall, the interviews provided insight into the potential strategies and obstacles of green stormwater infrastructure by user group. Kati & Jari (2016) found differences in values held by residents, managers, and politicians. For example, residents expressed attachment to a park as green infrastructure because it holds cultural value, while managers expressed negative values toward the park. They argued that research should further understand

these differences and find mutual values for future collaborative planning (*Ibid*.). Welsh & Mooney (2014) surveyed a community and interviewed experts, concluding that increasing green infrastructure implementation has potential to improve community cohesion and resiliency on top of environmental benefits of green stormwater infrastructure. The cooperation of residents toward a common goal of improving ecosystem services in their community led to this social cohesion (Welsh and Mooney, 2014). Other researchers concluded that participants' willingness to pay for green infrastructure is linked to perceived aesthetics, as well as improved hydrologic function and water quality (Londono Cadavid and Ando, 2013; MacDonald et al., 2015). Some scholars viewed perceived social values as an avenue to support and incorporate green space and infrastructure in urban areas (Attwater and Derry, 2017; Ghermandi, 2016). Property values increase from green stormwater infrastructure, particularly near green spaces installed to manage stormwater (Mazzotta et al., 2014).

Educational and recreational values from green infrastructure were discussed in the literature, with most authors asserting that green infrastructure, such as urban ponds, offer education and recreation services, and consequently improve community welfare (Hassall, 2014; Kandulu et al., 2014). Individual perceptions of these services, as well as the potential of recreation and education, were sometimes measured (Kremer et al., 2015; McDuffie et al., 2015; Wilson, 2012). An example study, conducted by Wilson (2012), found that individuals hold views that are more positive when green stormwater infrastructure includes recreation and educational opportunities.

Supporting Services – The majority of the research on supporting services of green stormwater management was centered on biodiversity and habitat provided by

green infrastructure (Greenway, 2015; Hassall and Anderson, 2015). With altered landscapes leading to habitat and biodiversity loss, the main argument was that green infrastructure preserves viable species' populations needed to support ecosystem processes, diversity, and consequently other ecosystem services (Attwater and Derry, 2017; Kopecka et al., 2017; Taylor and Lovell, 2014). However, few researchers quantified the impacts of green stormwater management on supporting services for specific habitats and species. Greenway (2015) showed that constructed stormwater wetlands provide habitat for macroinvertebrates and measured biodiversity with species richness as a metric. While studies link green space biodiversity to human well-being, researchers recognized that biodiversity preservation is more nuanced than merely implementing green infrastructure (Hassall and Anderson, 2015; Kopecka et al., 2017). They recommended more thorough examination of potential ecosystem services and limitations of green stormwater infrastructure for conservation (Dagenais et al., 2017; Mitsova et al., 2011).

Discussion

Major Findings

We identified six major findings that summarize the state of research at the intersection of green stormwater management and ecosystem services. These are discussed in turn below. *First*, most of the experiments and studies on green stormwater management were conducted at the parcel-scale (Adyel et al., 2016; Buckland-Nicks et al., 2016; Wardynski et al., 2012; Zölch et al., 2017). While implementation of green stormwater infrastructure at small scales suggests improvements to provisioning, regulating, cultural, and supporting ecosystem services, more research is warranted at the

watershed-scale to quantify regional-scale effects. Watershed-scale modeling provides an appropriate method to upscale parcel- and neighborhood-scale results (Feng et al., 2016; Garcia-Cuerva et al., 2016; McDonough et al., 2016; Wu et al., 2013).

Second, 49 of the publications (29%) included frameworks or approaches for implementing green stormwater management and highlighted barriers to implementation. Frameworks were developed for different cities and regions, and focused on facilitating decision-making and spatial planning of green stormwater management (Carter and Fowler, 2008; Chaffin et al., 2016; Dhakal and Chevalier, 2016; Hoang and Fenner, 2016; Lundy and Wade, 2011; Perales-Momparler et al., 2015; Schuch et al., 2017; Shuster and Garmestani, 2015). Authors developed frameworks based on literature reviews and case studies, and they centered their approaches on using green stormwater infrastructure to mitigate for lost ecosystem services from urbanization, adapt to climate change, or integrate multiple ecosystem services into stormwater management (*Ibid*.). Several of the frameworks emphasized barriers to implementing green stormwater infrastructure. They attributed jurisdictional overlap and insufficient incentives for partnerships between the different groups and individuals as barriers to green stormwater management (Chaffin et al., 2016; Dhakal and Chevalier, 2016; Shuster and Garmestani, 2015). Different groups also had fragmented responsibilities and interests that conflict, which in turn creates barriers for organized management (Hoang and Fenner, 2016; Perales-Momparler et al., 2015). Some authors point to inertia and lack of financial and political support as an additional barrier to green stormwater infrastructure (Carter and Fowler, 2008; Shuster and Garmestani, 2015).

Third, only 39% of publications quantified ecosystem services from green

stormwater management (Figure 2-4). Many papers summarized general relationships, or assumed relationships, between green stormwater infrastructure and ecosystem services. Regulating services were most often quantified, with diversity in the metrics used, such as carbon accumulation and phosphorus accretion (Bhomia et al., 2015; Merriman et al., 2017). The other three categories of ecosystem services were rarely quantified. Quantifying changes to ecosystem services from green stormwater infrastructure is a needed direction for the future to inform and improve green stormwater design, decision-making, planning, and implementation.

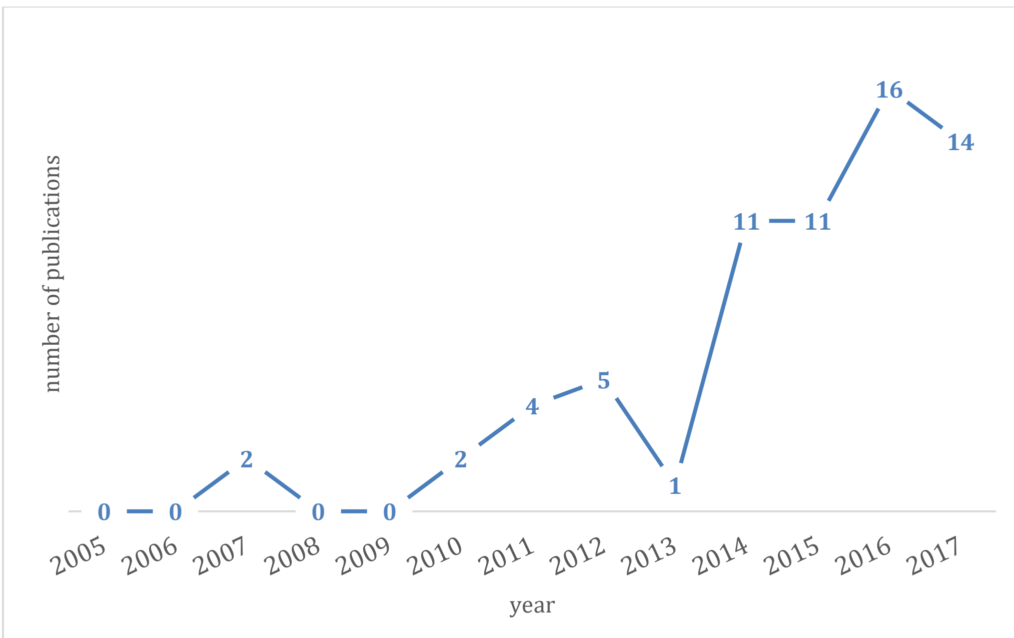

Figure 2-4. Number of publications that quantify ecosystem services related to stormwater management.

A *fourth* finding is that there were no significant global geographic patterns of research on green stormwater management and ecosystem services. Research has been conducted in a variety of places and climates, including Australia, France, the east and west coasts of the United States, and China (Bhomia et al., 2015; Gao et al., 2015;

Maillard and Imfeld, 2014; Moore and Hunt, 2012; Schuch et al., 2017; Yang et al., 2015). However, there is a lack of research at the intersection of ecosystem services and green stormwater management in developing regions and countries. This finding indicates that multiple researchers are interested in and are investigating the potential of green stormwater infrastructure to provide ecosystem services. While this is a promising finding, future research should investigate whether green stormwater infrastructure provides ecosystem services differently across cultural, socioeconomic, and sociopolitical settings.

Fifth, studies increasingly integrate engineering, physical sciences, and social sciences in their research questions. The ecosystem services approach to evaluating green stormwater management lends itself to interdisciplinary research. Nevertheless, research that incorporates all three of these disciplines are limited in number, with several of the publications coming from urban planning and landscape architecture venues (Dagenais et al., 2017; Hoang and Fenner, 2016; Horsley et al., 2016; McPherson et al., 2011; Yang et al., 2013). Further examination of multiple ecosystem services in a single study would also progress the literature. The maintenance and delivery of one ecosystem service happens in relation to other ecosystem services, and therefore, these connections among ecosystem services should be studied. In a similar vein, different combinations of green stormwater infrastructure may be more suitable than relying on one type alone. Cities likely will benefit from implementing green infrastructure throughout their watershed, which should be explored in future research.

Sixth, overlapping and redundant green stormwater infrastructure terminology is an impediment to research discovery. We searched for 25 unique terms in addition to

“stormwater” and “ecosystem services” (Table 2-1). It was necessary to search for individual types of green stormwater infrastructure, like stormwater ponds, rain gardens, or green roofs for comprehensive review (Chaffin et al., 2016; Gittleman et al., 2017; Monaghan et al., 2016; Moore and Hunt, 2011; Olguin et al., 2017; Rumble and Gange, 2017; Squier et al., 2014; Starry et al., 2011). Similarly, many terms overlap somewhat, such as green infrastructure, green space, and low impact development (Cizek, 2014; Klimas et al., 2016b; Mayer et al., 2012). While these terms are not completely redundant, they obscure search results. In addition, there is no consensus on the spelling of stormwater, with some researchers writing it as a single word, some as a hyphenated word, and some as two words. Most articles wrote stormwater as a single word and following this norm will facilitate future literature searches. We also recommend authors include a catchall term such as ‘green stormwater infrastructure’ as a search keyword for a cohesive body of literature.

Future Research Directions for Managing Ecosystems Services with Green Stormwater Infrastructure

Through organizing existing green stormwater infrastructure literature into the four categories of ecosystem services, we identified research gaps in all categories. First, many researchers referred qualitatively to the ecosystem services offered by green stormwater infrastructure, and few researchers quantified the value or impact of those benefits. Also, existing studies typically focus on one type of ecosystem service; however, utilizing an ecosystem services framework encourages multi-disciplinary research for green stormwater management (Lundy and Wade, 2011). Finally, lack of policy and institutional support for green stormwater infrastructure to provide ecosystem

services was a barrier mentioned in papers in all categories of ecosystem services. With the remainder of the discussion, we outline three main directions for future research at the intersection of stormwater management and ecosystem services: 1) quantifying ecosystem services, 2) integrating engineering, environmental, and social criteria into stormwater management, and 3) integrating stormwater management and water policy.

Quantifying ecosystem services is rarely done but is needed to better understand the extent to which green stormwater infrastructure may enhance or degrade ecosystem services. Ecosystem services are sometimes monetized (Costanza et al. 1997), but need not be economically valued to be measured. Identifying metrics to measure ecosystem services will allow researchers and stormwater managers to reduce undesirable impacts of stormwater, like erosion and water quality degradation, while enhancing ecosystem services from green stormwater infrastructure. Measuring specific ecosystem services from green stormwater infrastructure will inform decisions about stormwater management in varying climates, regions, and for different design objectives. Figure 2-5 illustrates the contribution of quantifying ecosystem services from green infrastructure to management decisions. By evaluating the quantity, location, and timing of ecosystem services from green infrastructure alternatives, decision-makers are better primed for implementing stormwater management plans to meet desired stormwater ecosystem services.

We provide example metrics to measure all categories of ecosystem services in Table 2-3. Green stormwater infrastructure research could be expanded to measure surface and groundwater supply, and the effects of urbanization and climate change on these services (Dillon et al. 2009a; Dillon et al. 2009b; Maliva 2014).

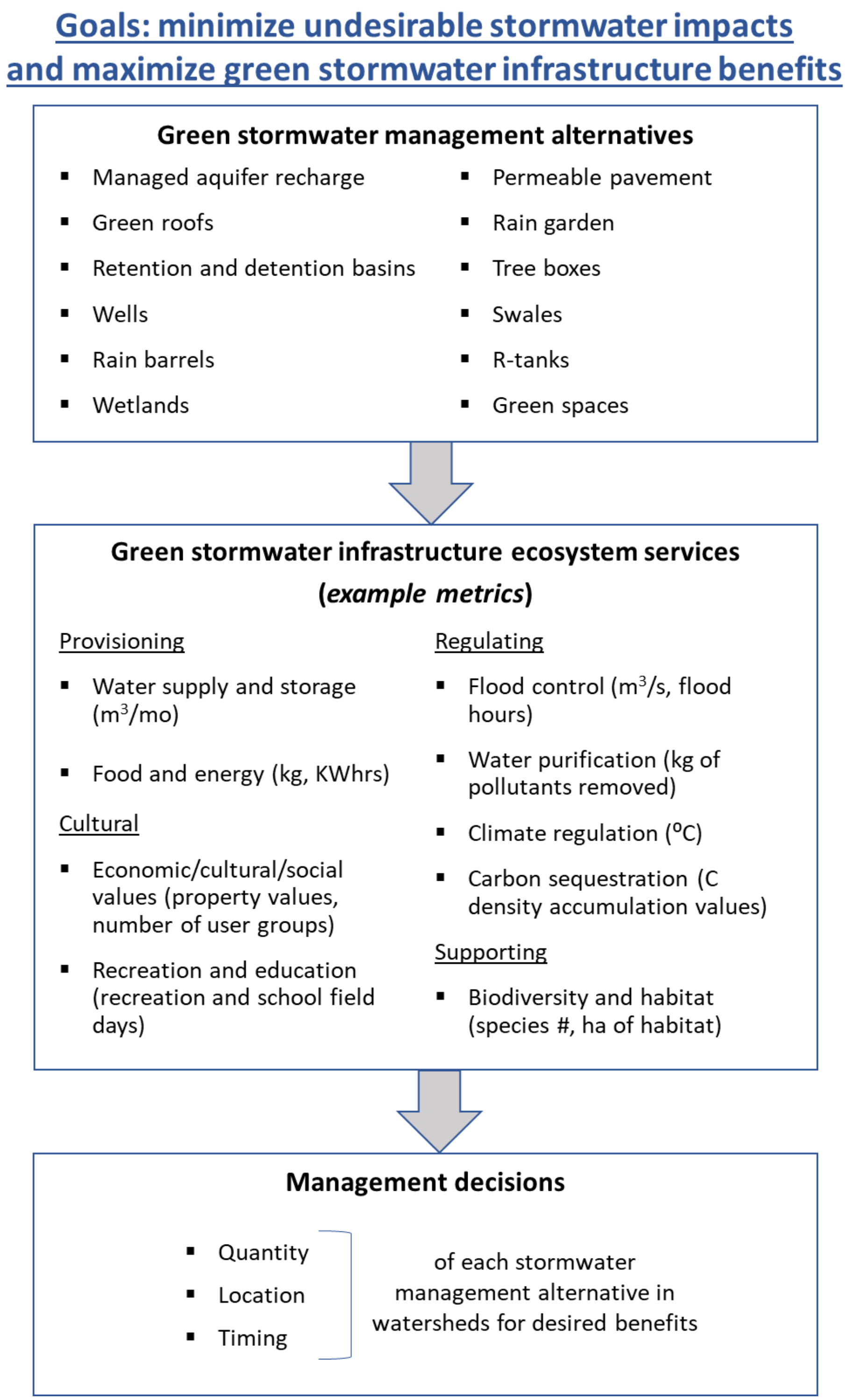

Figure 2-5. Connection between quantifying green stormwater infrastructure ecosystem services and management decisions.

Quantifying possible tradeoffs between increasing aquifer storage and introducing water quality contaminants to groundwater is a needed direction to quantify competing ecosystem services. Similarly, measuring the effects of green stormwater infrastructure design for water purification and stream temperature management is warranted, especially at the watershed- or regional-scale for spatial planning purposes. While considerable research has evaluated perceptions and values of ecosystem services from green stormwater infrastructure, cultural components of ecosystem services should be measured in future research. This could include change in property values from proximity to green stormwater projects (Mazzotta et al., 2014) or recreational metrics, such as number of boatable days in rivers (Ligare et al. 2012). Research on supporting services of stormwater management is least often studied. Green stormwater infrastructure could focus on biodiversity as an umbrella goal for resiliency of several ecosystem services in the urban setting (Connop et al., 2016).

Secondly, integrating engineering, social, and environmental criteria is needed to identify the most appropriate and effective stormwater infrastructure, and to evaluate synergies among disciplines for holistic stormwater decision-making and management (Hale et al., 2015). Engineering criteria are the bases for infrastructure and technological solutions. Environmental criteria maintain ecosystem functions of interest. Social criteria highlight economic, political, and cultural values, perceptions, and barriers to implementation. Figure 2-6 shows examples of these intersections. Our review showed that provisioning and regulating ecosystem services received more attention than other ecosystem services, but were typically evaluated one at a time (Gittleman et al., 2017; Griffin et al., 2014; Mogollon et al., 2016).

Table 2-3. Ecosystem services-stormwater management research subareas and example metrics to quantify ecosystem services from green stormwater infrastructure

Category	***Future Research Subareas***	***Example Metrics to Quantify Ecosystem Services***
Provisioning Services	Population growth and water supply reliability	Water volume, cubic meters per month (m^3/mo)
	Water storage and climate adaptation	Groundwater recharged, m^3/mo, or aquifer water level, m
Regulating Services	Water quality improvement	Temperature and contaminant change, ΔC, or dollars per pound of contaminant removed, \$/lb C
	Flood mitigation	Reduction in flood discharge magnitude, m^3/s, or reduction in flood duration (hours)
Cultural Services	Pricing strategies for cultural services	Residents' willingness to pay for aesthetics and recreational opportunities from green stormwater infrastructure, \$
	Revenue and property values	Property value change from proximity to green stormwater infrastructure, \$
Supporting Services	Biodiversity	Number of species, count
	Perceptions of resource managers and residents	Statistical analyses on managers' and residents' perceptions of species and habitats, chi-square statistic

These studies offer initial findings that support green stormwater management to maintain ecosystem services, but future research could provide a deeper investigation of green infrastructure through evaluating research questions about multiple types of ecosystem services.

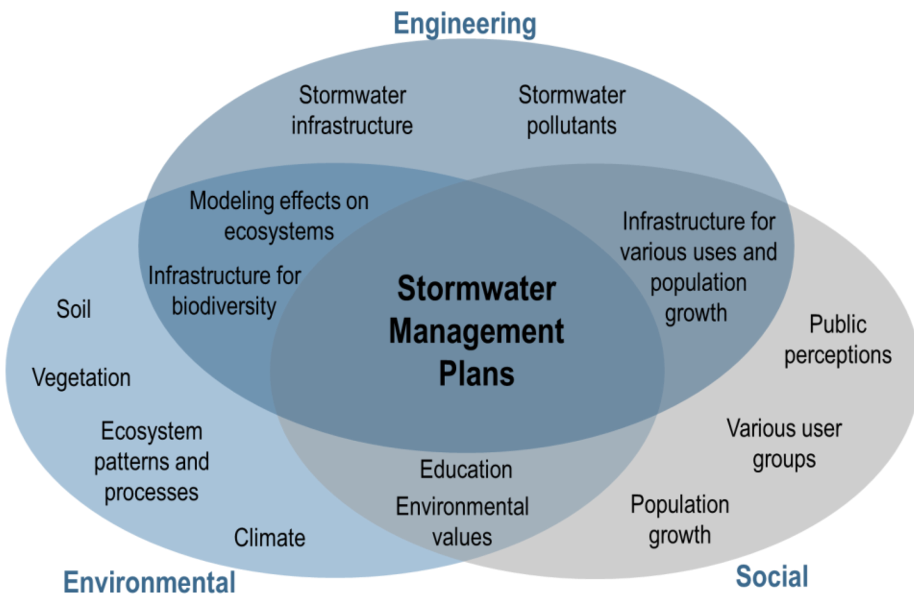

Figure 2-6. Examples of engineering, environmental, and social criteria.

Finally, we encourage scholars to quantify the social, economic, environmental, and policy benefits of green stormwater infrastructure so that green stormwater management can be integrated into environment-related policy. Stormwater governance in the U.S. is decentralized, which creates barriers from jurisdictional overlap or lack of mandate and authority in managing stormwater (Armstrong, 2015; Chaffin et al., 2016; Dhakal and Chevalier, 2016; Freeman, 2000; Shuster and Garmestani, 2015). By further integrating and explicitly addressing stormwater management research, stakeholders and decision-makers can be better informed to implement effective and resilient management practices. Here we briefly mention four policy routes that have potential to support the

investigation and implementation of sustainable stormwater practices in the US. Similar opportunities exist globally.

First, Total Maximum Daily Load (TMDL) plans, which are required for contaminated water bodies by the Clean Water Act (Elshorbagy et al., 2005), are an example method of further incorporating green stormwater management into environmental-related policy. Plans set limits on acceptable pollutant loads and outline needed changes to reduce contaminant loads. As the ecosystem services of green stormwater infrastructure for managing nutrients are measured, and as tradeoffs between enhancing water supply and water quality impacts are quantified, green stormwater infrastructure could be a direct method to attain TMDL targets. Many TMDL plans have been designated for impaired water bodies across the U.S. with recommendations for best management practices, including green stormwater infrastructure. However, little research has been conducted on the extent to which green stormwater infrastructure would need to be implemented to attain TMDL targets. Also, one component of the ESA is to address nonpoint source pollution, which is a significant part of stormwater runoff. Section 9 of the ESA requires protection of habitat for endangered fish and wildlife species. This, in turn, opens up legal possibilities to monitor and regulate nonpoint source pollution by increasing infiltration, water storage, and nutrient uptake through green stormwater infrastructure (Tzankova, 2013). Local- and state-level groundwater policy regulates and allocates groundwater. These policies may support groundwater recharge and water quality control from stormwater management (Kubasek and Silverman, 2005). Finally, researchers are increasingly studying the influence of green stormwater infrastructure on human health (Vogel et al., 2015). Current research is connecting

ecosystem services to human health and well-being in urban environments (*Ibid*.), leading to more research on the linkages between green infrastructure and ecosystem services. Public health concerns could encourage the implementation of green stormwater management (Coutts and Hahn, 2015).

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CHAPTER 3

ECOSYSTEM SERVICES FROM IMPLEMENTING GREEN STORMWATER INFRASTRUCTURE AT THE WATERSHED-SCALE

Abstract

Gray infrastructure uses pipes and culverts to collect and convey stormwater to receiving areas, typically oceans, inland lakes, and wetlands. Gray infrastructure increases stormwater runoff volume and exposure to pollutants relative to natural conditions by preventing infiltration to groundwater. Green infrastructure is increasingly proposed to reestablish natural processes and ecosystem services that are lost from urban development. This study uses QUAL2Kw to simulate streamflow, stream temperature, dissolved oxygen concentration, and total phosphorus concentration with and without implementation of grass swales, bioretention cells, and rain gardens in Utah's Red Butte Creek and Jordan River Watersheds, USA. Sixty-four model alternatives simulated streamflow and water quality during spring runoff and summer rainfall events if green infrastructure was incorporated at the reach, small watershed, and large watershed scales. The results show that the impacts of green infrastructure are only significant when green infrastructure is implemented at the large watershed scale. When green infrastructure is implemented in seven small watersheds, total phosphorus concentrations are reduced by 3-6% (18-51 μg/L, depending on the season) in reaches that total phosphorus is a pollutant of concern. Overall, modeling shows that parcel-scale green infrastructure reduce streamflow up to 9.5% in the winter/spring and 9.3% in the late summer, decrease total phosphorus up to 1.2% in the winter/spring and 8.6% in the late summer, decrease stream temperature in the winter/spring up to 17.4%, and increase dissolved oxygen

concentration up to 1.3% at downstream river outlets when green infrastructure is implemented throughout about 13% of watersheds.

Introduction

Stormwater management relies on centralized projects, such as underground conveyance, storm drains, and gutters to collect and quickly convey stormwater from urban regions (Deitch et al., 2013; Potter, 2006). However, shortcomings to conventional stormwater infrastructure exist (Burns et al., 2012; Roy et al., 2008; Tsihrintzis and Hamid, 1997). Without infiltration of stormwater into the ground, gray infrastructure transports a larger volume of runoff than would occur naturally, which creates some risk of flooding if the capacity of built infrastructure is exceeded (Figure 3-1). Reduced infiltration lowers baseflows, reducing the thermal mass of rivers and potentially raising stream temperatures (Anderson et al., 2010; Webb et al., 2008). In turn, higher stream temperatures reduce dissolved oxygen saturation concentrations, negatively impacting aquatic species and the ecosystem functions they provide (Null et al., 2017; Paul and Meyer, 2001). Finally, urbanization results in higher nutrient loadings to surface waters from sources including wastewater and fertilizers (Paul and Meyer, 2001). Increased nitrogen and phosphorus contributions to rivers promote algae growth and reduce oxygen levels, putting water bodies at risk for eutrophication (*Ibid*.).

Given the challenges from gray stormwater infrastructure and the transformation from natural to urban environments, green infrastructure may be promising for managing stormwater and ecosystem services because it reproduces natural processes and pathways of water movement over the landscape (Dhakal and Chevalier, 2017).

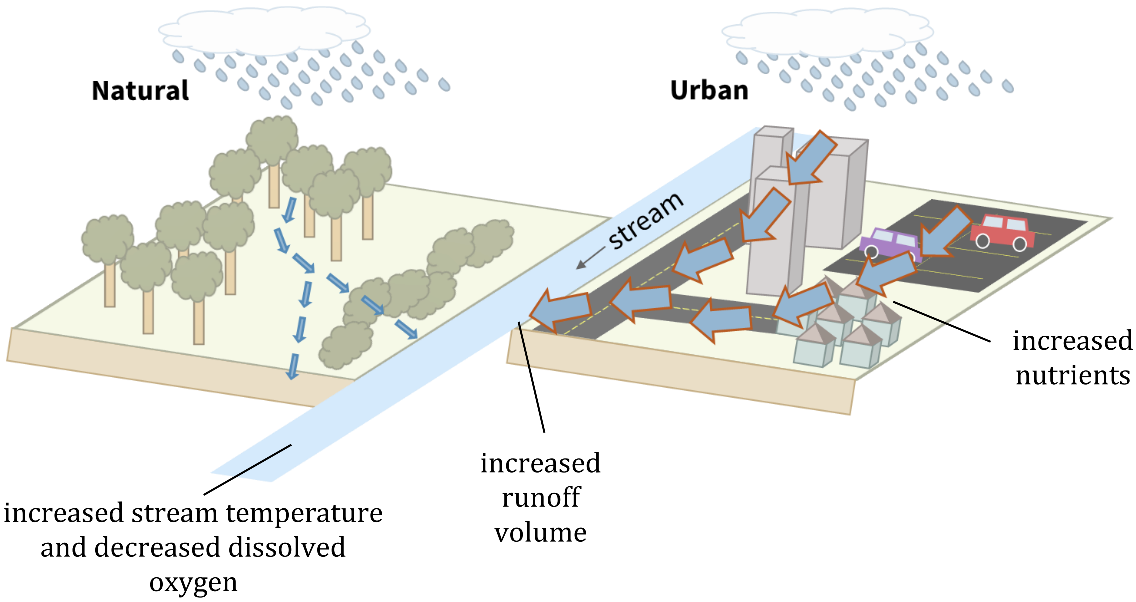


Figure 3-1. Stormwater runoff in natural and urban environments.

Ecosystem services, or environmental benefits to humans, include provisioning (e.g. water supply), regulating (e.g. water purification and flood mitigation), cultural (e.g. education and recreation), and supporting services (e.g. habitat) (Brauman et al., 2007). Through permeable areas and vegetation or media to filter runoff, green infrastructure in urban settings can improve delivery of ecosystem services that were present in a natural environment (Figure 3-2). During spring runoff conditions, reducing streamflow and flood risk is a benefit; however, during summer base case conditions, increasing streamflow is a benefit. More specifically, green infrastructure could reduce the flood peak, providing both the provisioning ecosystem service of water supply from replenished groundwater and baseflow during lower flows in the summer, and the regulating service of flood mitigation during spring runoff and higher-flow conditions (*Ibid*.).

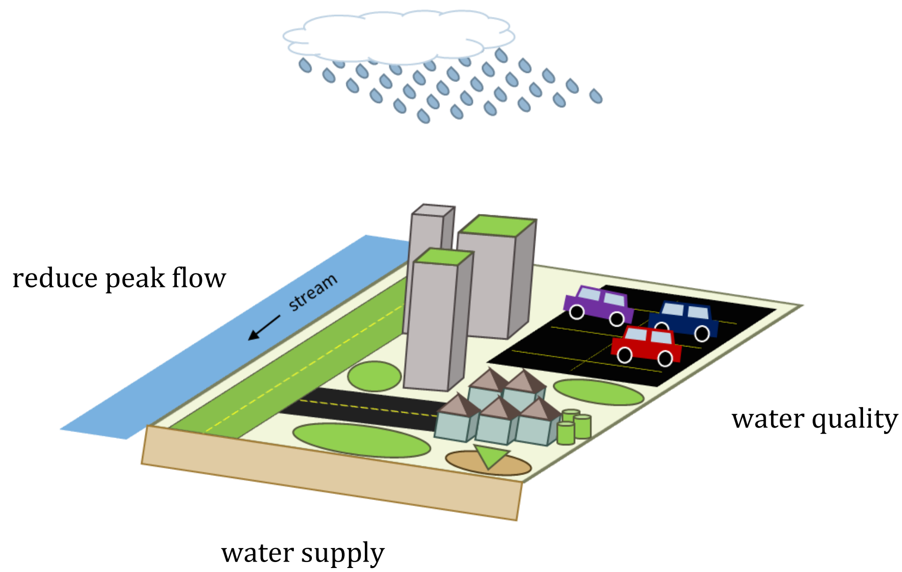


Figure 3-2. Green stormwater infrastructure in the urban environment.

Research on green stormwater infrastructure to maintain and improve ecosystem services has been increasing since the mid-2000s (Prudencio and Null, 2018). Several studies evaluate the performance of parcel-level green infrastructure on different kinds of ecosystem services, including climate regulation, water purification, flood control, and carbon sequestration (*Ibid*.). For example, Adyel et al. (2016) evaluated nutrient attenuation from a constructed wetland, and Wardynski et al. (2012) monitored a permeable parking lot to study changes in runoff volume and temperature. However, most studies use conceptual models or discuss green infrastructure and ecosystem services connections, without quantifying effects on specific ecosystem services (Kati and Jari, 2016b; Kuller et al., 2017).

Often, individuals and communities make decisions about where and how much green infrastructure to implement, and the effects of these smaller, distributed practices must be understood at larger scales (Burns et al., 2012). With most research on small,

spatially distributed green infrastructure, there is an ongoing need to understand the collective effects of these smaller projects over a watershed (Potter, 2006). As examples, Shuster et al. (2007) quantified the provisioning of groundwater recharge from rain gardens at the watershed level. York et al. (2015) simulated rainwater harvesting and bioretention at the watershed scale, finding that rainwater harvesting and bioretention for water reuse did not significantly reduce flows for downstream users. While the ecosystem services framework was not central in this study, it illustrates the provisioning services green stormwater infrastructure can provide for multiple users.

This study models the effects of green stormwater infrastructure on aquatic ecosystem services. Specifically, the objective of this study is to evaluate changes in surface water quantity and quality from implementation of different proportions and types of green infrastructure. My research questions are: 1) What are the effects of green stormwater infrastructure on surface water quantity and quality at reach, small watershed, and large watershed scales? and 2) Which types of green infrastructure lead to the largest improvements in water quantity and quality at different spatial scales? I use QUAL2Kw to simulate streamflow, stream temperature, dissolved oxygen concentration, and total phosphorus loading for spring runoff and summer conditions in Red Butte Creek and downstream Jordan River in Salt Lake Valley, UT, USA (Figure 3-3). Implementation of green infrastructure is modeled at two sites draining to Red Butte Creek, throughout Red Butte Creek, and throughout tributaries that feed Jordan River.

This study is a contribution to a larger, interdisciplinary project involving civil and environmental engineers, watershed scientists, and sociologists. Using green infrastructure data from other team members, I specifically quantify the effects of green

infrastructure on the delivery of regulating and provisioning services, specifically water quality improvement, flood mitigation, water supply, and temperature regulation. Overall, the simulation modeling in this chapter helps evaluate how much and what types of green infrastructure are needed to achieve management goals for surface water. Meetings and workshops with stakeholders and water managers in the Salt Lake Valley inform the modeling to provide useful tools to help with their stormwater management decisions.

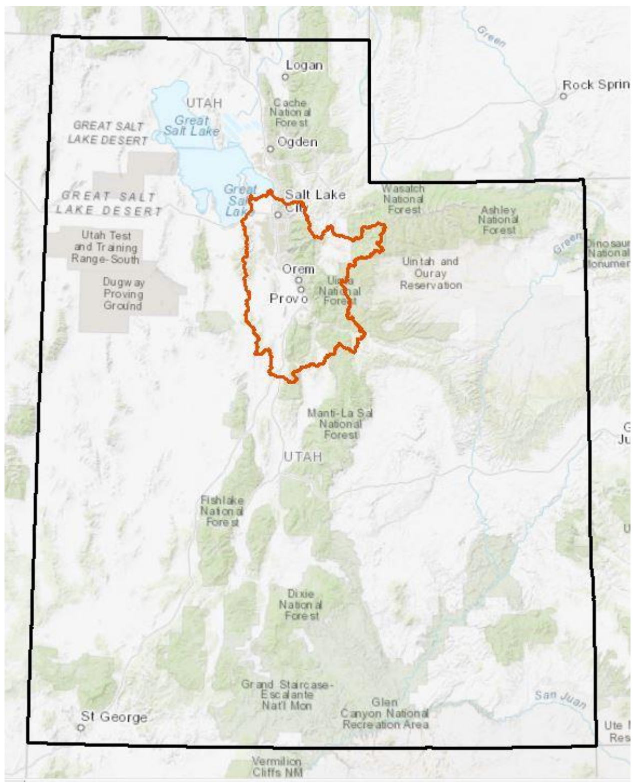


Figure 3-3. The large watershed study area is the Jordan River watershed (outlined in red) in the Salt Lake Valley of Utah.

Methods

Study Area

The Jordan River is a heavily regulated river that flows north from Utah Lake through the Salt Lake Valley about 83 km to Great Salt Lake (Figure 3-4). There are

seven canyon creeks that are tributaries of the Jordan River in the Salt Lake Valley: Little Cottonwood, Big Cottonwood, Mill Creek, Parley's, Emigration, Red Butte, and City Creeks. Other inflows to the Jordan River include groundwater flows, discharge from three wastewater treatment plants, runoff from agriculture and lawn irrigation, and urban stormwater largely through gray infrastructure (Von Stackelberg et al., 2014). There is a Total Maximum Daily Load for the Jordan River for dissolved oxygen (Adams and Arens, 2013). Total phosphorus is listed as a pollutant of concern for reaches in Salt Lake City, which is evaluated in this study. Stormwater runoff contributes approximately 4% of the total annual total phosphorus loadings to Jordan River, compared to 9% from Utah Lake (*Ibid*.).

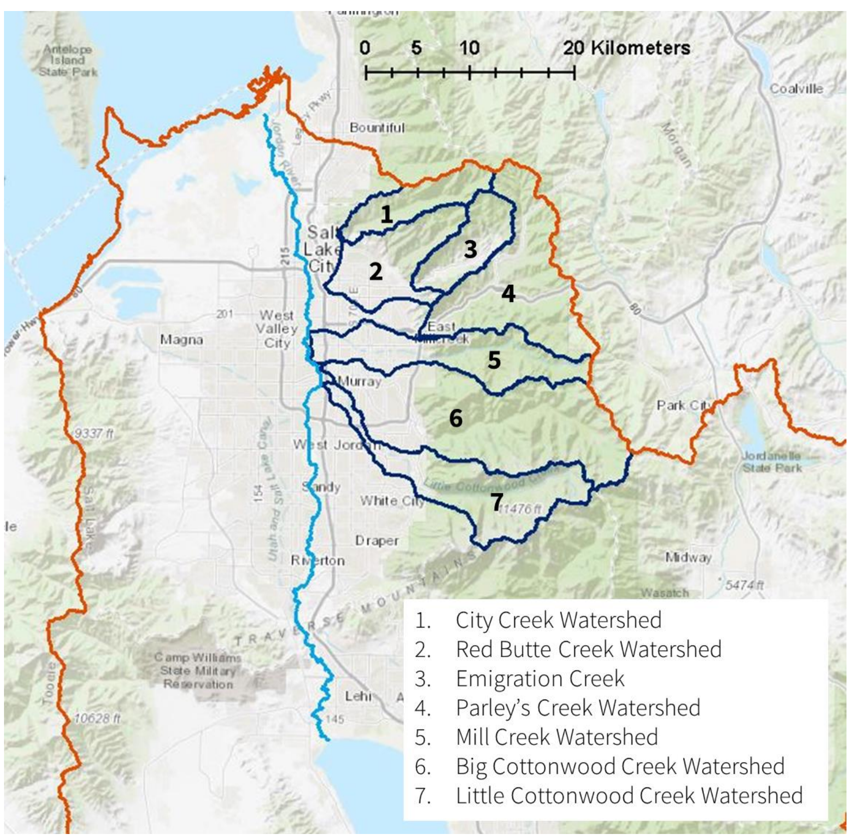


Figure 3-4. Jordan River flows from Utah Lake to Great Salt Lake. Seven tributary canyon creeks are evaluated in this study, with a focus on Red Butte Creek.

Red Butte Creek is in a snow-dominated region where stormwater infrastructure is designed for high flows during spring snowmelt, with occasional winter floods. With a basin size of 28.5 square km, Red Butte Creek is located in northeast Utah, on the west-slope Wasatch Mountains in Salt Lake County (Salt Lake County Watershed Planning & Restoration Program, 2014). Red Butte Creek's upper reaches are in a protected and designated Research Natural Area of the U.S. Forest Service. After the Red Butte Reservoir and the protected canyon, the creek abruptly changes to an urban creek through Salt Lake City to the confluence with the Jordan River about 12 km from the reservoir (Figure 3-5). It flows roughly 6 km through urban and residential areas, then flows underground though concrete culverts and emerges at Liberty Lake in a city park. Red Butte Creek then goes back underground through another concrete culvert until it joins the Jordan River. It is mostly a losing stream before it goes underground (Gabor et al., 2017). After the creek goes underground, the runoff in the city and residential areas is directed via storm drains underground as well. However, there are gaining reaches before this, where stormwater runs off into the creek through storm drains at river kilometers 2.70, 2.93, 3.22, and 3.32 (Figure 3-6).

The gaining reaches of the Red Butte Creek include 0.143 km^2 of roofs, 0.205 km^2 of parking lots, 0.237 km^2 of streets that drain into the creek (Figure 3-7). These roofs, parking lots, and streets are used in the simulation of green stormwater infrastructure alternatives on the small watershed scale. This area makes up roughly 13 percent of the modeled Red Butte watershed, when not accounting for the protected canyon where modeling green infrastructure is not relevant.

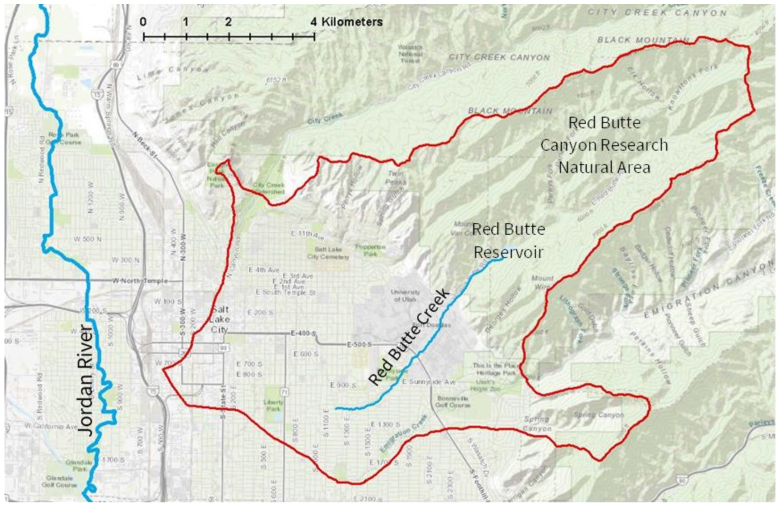


Figure 3-5. The Red Butte Creek model starts at the outlet of Red Butte Reservoir in the protected canyon. The creek abruptly changes to an urban stream after leaving the canyon.

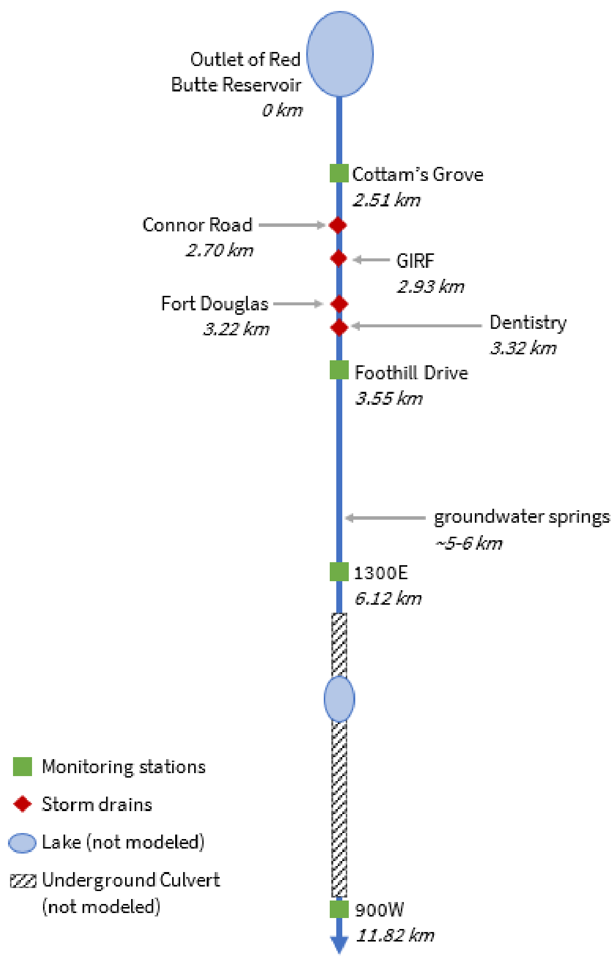


Figure 3-6. Schematic of inflows and outflows of Red Butte Creek.

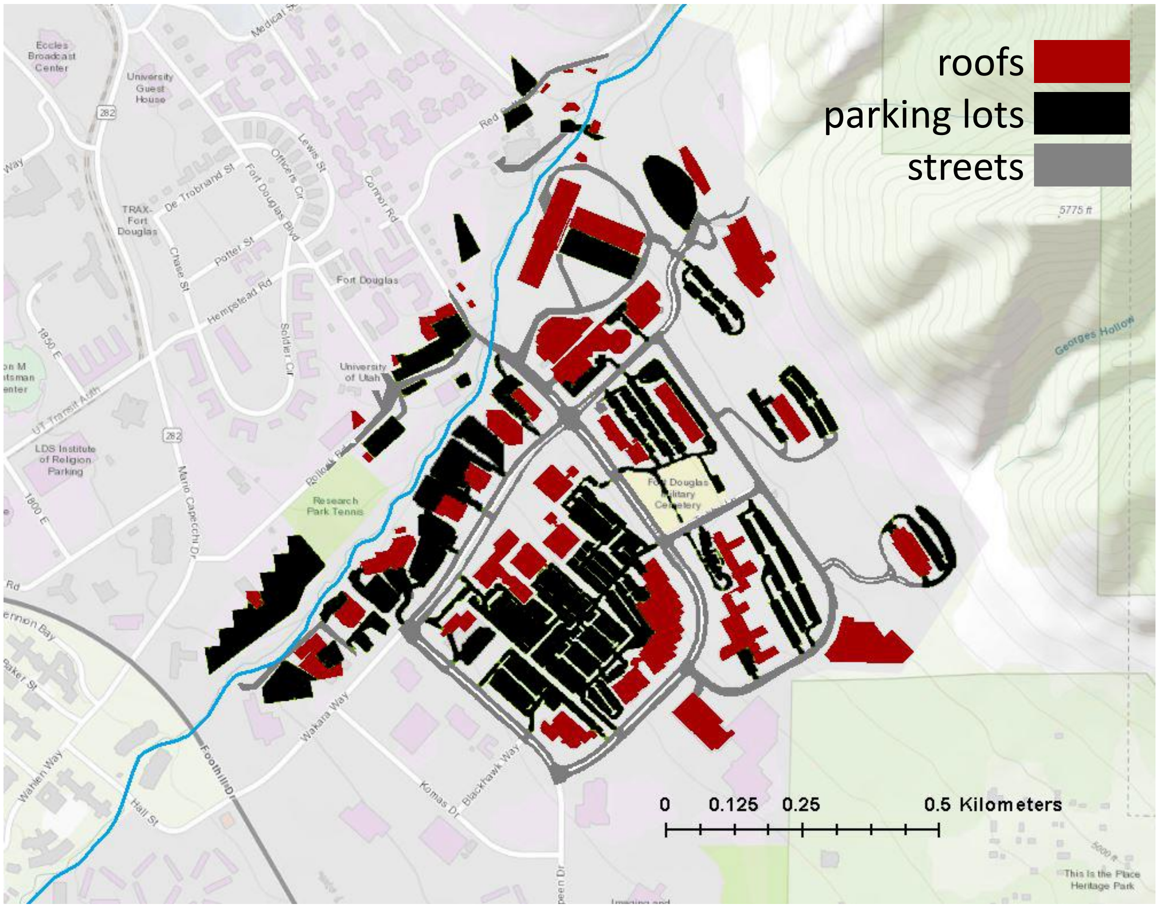


Figure 3-7. Roofs, parking lots, and streets that drain to the gaining reach of Red Butte Creek.

Model Description

QUAL2Kw version 6 was used to simulate changes in Red Butte Creek surface water quality from green stormwater infrastructure implementation alternatives. QUAL2Kw is one-dimensional, modeling streamflow and water quality change longitudinally and assuming the channel is well-mixed laterally and vertically (Pelletier and Chapra, 2008). QUAL2Kw version 6 models non-steady flows with changing water quality concentrations over time (*Ibid.*). We modeled stream temperature, dissolved oxygen, and total phosphorus because data were available for those constituents. The Jordan River QUAL2Kw version 5.1 models were developed by collaborators from the

Utah Department of Environmental Quality (DEQ), Division of Water Quality, and Utah State University (Neilson et al., 2012; Von Stackelberg et al., 2014). QUAL2Kw version 5.1 models steady flow hydraulics with repeating 24-hour boundary conditions (Pelletier and Chapra, 2008).

QUAL2Kw

QUAL2Kw simulates water quality. This study focuses on simulated changes in streamflow, stream temperature, dissolved oxygen, and total phosphorus. In version 6 of QUAL2Kw, flows can be modeled as non-steady and non-uniform with hourly-changing boundary conditions.

Stream temperature in QUAL2Kw is driven by heat fluxes from inflows and outflows, dispersion between reaches, air-water heat flux, and sediment-water heat flux.

$$\begin{aligned}\frac{dT_i}{dt} = {} & \frac{Q_{i-1}}{V_i} T_{i-1} - \frac{Q_i}{V_i} T_i - \frac{Q_{ab,i}}{V_i} T_i + \frac{E'_{i-1}}{V_i}(T_{i-1} - T_i) + \frac{E'_i}{V_i}(T_{i+1} - T_i) \\ & + \frac{W_{h,i}}{\rho_w C_{pw} V_i}\left(\frac{m^3}{10^6 cm^3}\right) + \frac{J_{h,i}}{\rho_w C_{pw} H_i}\left(\frac{m}{100cm}\right) + \frac{J_{s,i}}{\rho_w C_{pw} H_i}\left(\frac{m}{100cm}\right)\end{aligned} \quad (1)$$

where T_i is the temperature in reach i and time t, Q_i is flow in reach i, V_i is water volume in reach i, $Q_{ab,i}$ are flow abstractions in reach i, E'_i is the bulk dispersion coefficient between reaches i and $i+1$, $W_{h,i}$ is the net heat load from point sources and non-point sources into reach i, ρ_w is the density of water, C_{pw} is the specific heat of water, $J_{h,i}$ is the air-water heat flux, and $J_{s,i}$ is the sediment-water heat flux.

As stream temperatures increase, less dissolved oxygen is soluble in water, creating an inverse relationship between stream temperature and dissolved oxygen concentration (Chapra, 2008). Dissolved oxygen concentrations can also be reduced from

carbon and nutrient-rich inflows to the stream. In QUAL2Kw, dissolved oxygen also responds to plant photosynthesis, oxidation, nitrification, respiration, and reaeration.

$$S_o = r_{oa}(PhytoPhoto - PhytoResp) + r_{od}(BotAlgPhoto - BotAlgResp)\frac{A_{st,i}}{V_i} - r_{oc}FastCOxid - r_{oc}SlowCOxid - r_{on}NH4Nitr + OxReaer - CODoxid - SOD\frac{A_{st,i}}{V_i} \quad (2)$$

where S_o is dissolved oxygen, r_{oa} is the ratio of oxygen to chlorophyll, $PhytoPhoto$ is phytoplankton photosynthesis, $PhytoResp$ is phytoplankton respiration, r_{od} is the ratio of oxygen to dry weight, $BotAlgPhoto$ is bottom algae photosynthesis, $BotAlgResp$ is bottom algae respiration, $A_{st,i}$ is the area of the stream in reach i, V_i is volume at reach i, r_{oc} is the ratio of oxygen consumed per organic carbon oxidized to carbon dioxide, $FastCOxid$ is fast CBOD oxidation, $SlowCOxid$ is slow CBOD oxidation, r_{on} is the ratio of oxygen to nitrogen consumed during nitrification, $NH4Nitr$ is nitrification, $OxReaer$ is oxygen reaeration, $CODoxid$ is chemical oxygen demand oxidation, and SOD is sediment oxygen demand.

Phosphorus is often attached to soil particles and moves into surface water from runoff and soil erosion (USGS, 2020). Sources include fertilizers and organic waste in both the urban and agricultural settings. Total phosphorus (TP) is modeled in QUAL2Kw by summing organic phosphorus, inorganic phosphorus, and phytoplankton in the water column.

$$TP = p_o + p_i + r_{pa}a_p \quad (3)$$

where p_o is organic phosphorus, p_i is inorganic phosphorus, r_{pa} is ratio of phytoplankton to chlorophyll, and a_p is phytoplankton.

Model Development

Two Red Butte Creek QUAL2Kw models were developed, a spring model simulates spring runoff flow conditions for 2 days in May (May 15-16, 2016) and a summer model simulates summer conditions for 2 days in June (June 13-14, 2016) (Figure 3-8). These time periods were chosen based on data availability and to represent variable flow conditions. The upper boundary condition is the outlet of Red Butte Reservoir in the protected natural area at river kilometer 0, while the downstream boundary is roughly 5.7 km upstream of the confluence with the Jordan River where Red Butte Creek is directed underground. Data are unavailable where the creek is underground. The spring model simulates a 0.41-cm rainfall event that lasts 5 hours and occurred on May 15, 2016. The summer model simulates a 0.36-cm rainfall event that lasts 7 hours that occurred on June 13, 2016 (Appendix A, Figure 7-3). These rain events were the median for rainfall events in the months of May and June for 2016 and 2017 (Fernández Velásquez, 2018). Because QUAL2Kw does not model precipitation as a direct input, these events are represented as inflows from the four storm drains (Appendix, Figure 7-3). In both models, the river is segmented into 86 reaches. Reaches vary in length but average 140 meters. The model uses a 0.5-minute timestep, and I evaluate the model results on a 45-minute timestep output by QUAL2Kw to compare to observed data.

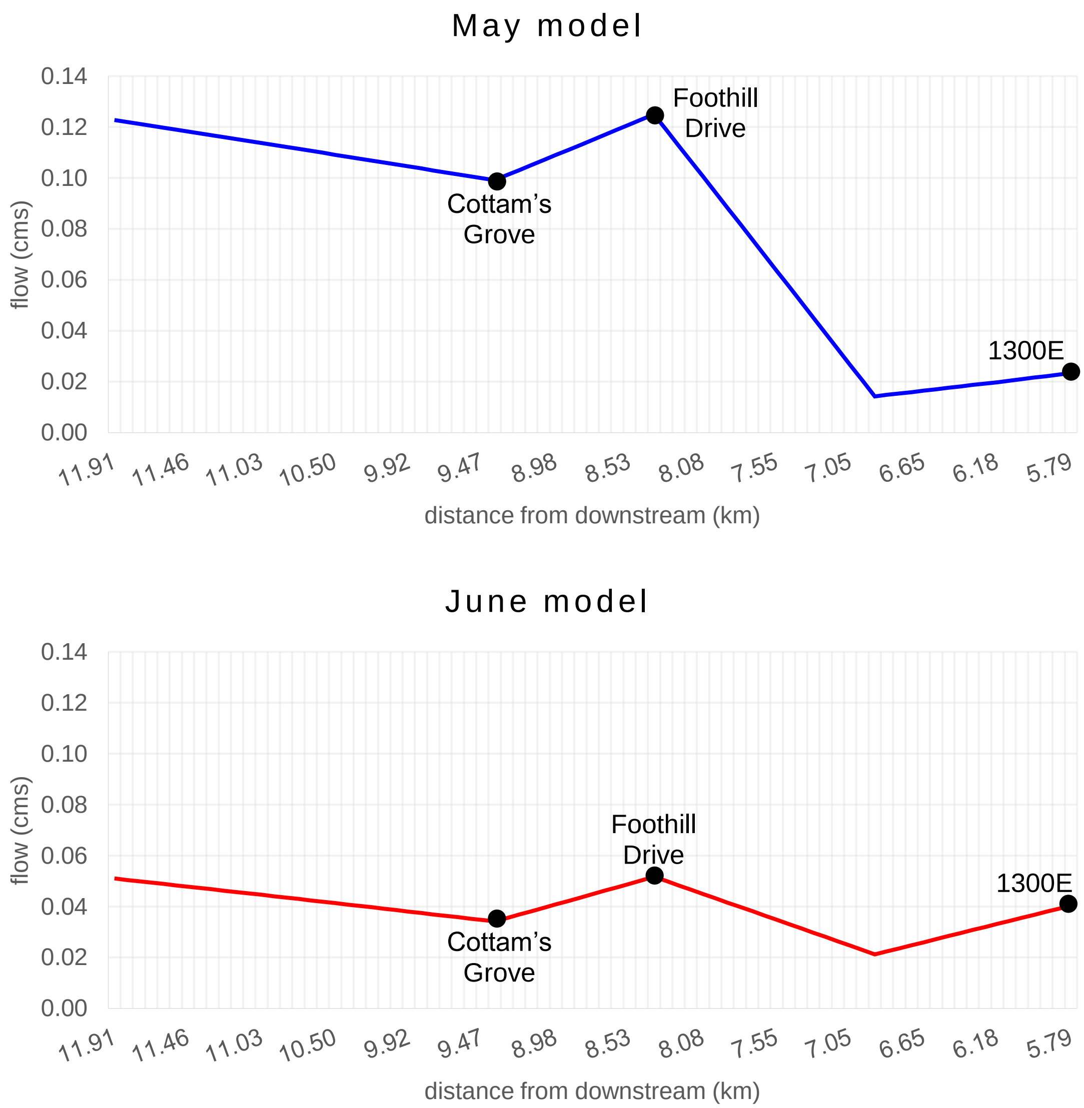


Figure 3-8. Average flow in the May 15-16, 2016 (top) and June 13-14, 2016 (bottom) Red Butte Creek QUAL2Kw model. The four storm drains and GI alternatives are modeled between Cottam's Grove and Foothill Drive. Groundwater springs upwell in the streambed starting around km 6.91 (Gabor et al., 2017).

The 82.7 km Jordan River is segmented into 166 reaches with an average length of 500 m. The developers from Utah State University and the Utah Department of Environmental Quality, Division of Water Quality modeled 6-day periods with a timestep of 11.3 minutes. In the Jordan River model, Red Butte Creek and two other tributaries combine as a single point source at river km 22.9. The developers modeled three time

periods: October 2006 during early fall/late irrigation season, February 2007 during winter/non-irrigation season, and August 2009 during summer/irrigation season. Details of the Jordan River model, including input data and model assumptions, are provided in Stantec Consulting Ltd. (2010). Outflow streamflow and water quality from the spring Red Butte Creek model are input to the February Jordan River model (referred to as the winter/spring model) to simulate high flow winter to spring conditions, and data from the summer Red Butte Creek model is input to the August Jordan River model (referred to as the late summer model) to simulate baseflow summer conditions. While there are changes in water quality after Red Butte Creek goes underground and emerges in Liberty Lake, that section of the creek is beyond the scope of this model. I assume that water quality is unchanged from 1300E to the confluence with the Jordan River. Due to the lack of data between 1300E and the confluence of Red Butte Creek with the Jordan River, I also conduct a sensitivity analysis and report what would happen in the Jordan River if flow from the Red Butte Creek is 10% and 20% more than the assumed green infrastructure alternative and 10% and 20% less than.

In the Jordan River models, the seven canyon creeks are point sources starting with Little Cottonwood Creek at river km 48.2 (Figure 3-9). The resulting percentage changes in streamflow, stream temperature, dissolved oxygen, and phosphorus for the 100% GI alternatives for the Red Butte Creek watershed are used for the seven canyon creeks to evaluate the changes at the larger Jordan River watershed scale. It should also be noted that these modeled changes in flow, stream temperature, dissolved oxygen, and total phosphorus are potentially a lower estimate for the six canyon creeks other than Red Butte. Because most of the roofs, parking lots, and streets in Red Butte Creek were in

losing reaches, the estimates from Red Butte Creek were from gaining reaches that made up a smaller amount of the watershed (roughly 13 percent when not including the protected canyon). Therefore, the estimated effects in the other six sub-watersheds would be assuming that green infrastructure is implemented in each at a similar smaller scale compared to the Red Butte Creek watershed.

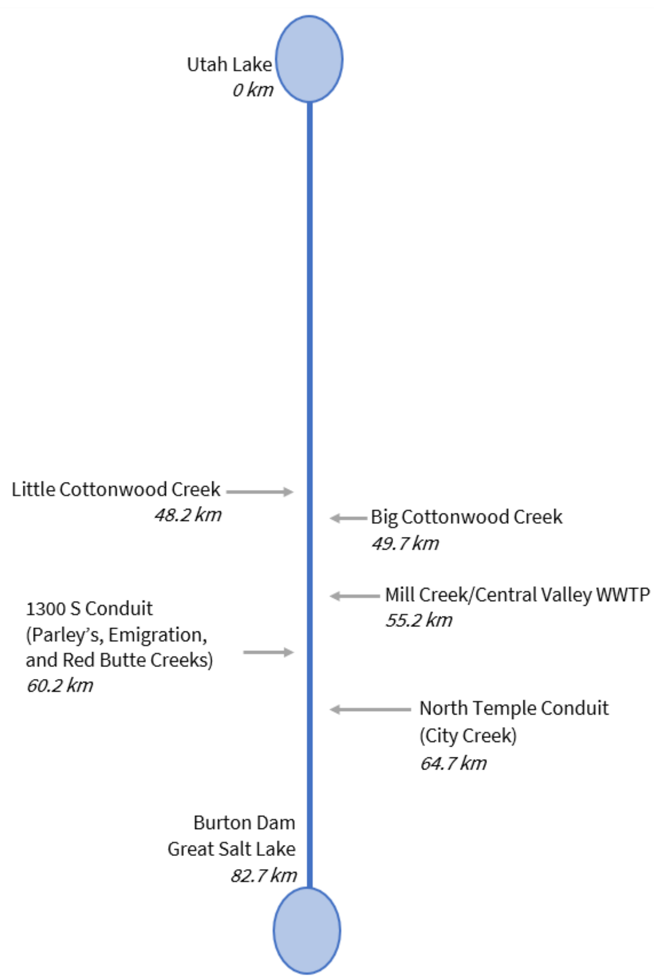


Figure 3-9. Schematic of connecting the seven canyon creeks to the Jordan River QUAL2Kw models. The seven canyon creeks are located in the lower half of the Jordan River.

Streamflow, stream temperature, dissolved oxygen, air temperature, dewpoint temperature, solar radiation, and wind speed data were collected at 15-minute intervals by

the iUTAH EPSCoR project at three sites (Appendix A, Figure 7-10). Hourly Red Butte Reservoir releases, stream temperatures and dissolved oxygen, which are the input data for the spring and summer models, are shown in Appendix A. Grab samples were also collected by the iUTAH project from 2013-2016, where total phosphorus data for these models were obtained.

Figure 3-10. iUTAH aquatic stations collected hourly flow, stream temperature, and dissolved oxygen input data. TP grab samples were collected at the same sites.

Model Calibration

Channel bottom width, percentage of coverage with bottom algae, sediment oxygen demand, hyporheic exchange, and QUAL2Kw rate parameters were calibrated for the reaches: 1) between the reservoir and Cottam's Grove, 2) between Cottam's Grove and Foothill Drive, and 3) Foothill Drive to 1300 E (Appendix A, Table 7-2 – Table 7-4).

These parameters are uniform throughout the two-day modeling period. Bottom width was estimated from visits to the iUTAH monitoring stations (Figure 3-10), and then altered to best fit measured stream temperature and dissolved oxygen concentration. Sediment oxygen demand and the growth model and rates for periphyton, macrophytes, and algae were varied within the ranges used for nine QUAL2Kw models representing other Utah rivers (Neilson et al., 2012) to best fit stream temperature and dissolved oxygen measured data. Hyporheic exchange parameter for exchange flow between the main channel and the hyporheic transient storage was adjusted based on the proportion of stormwater infiltrated during rainfall events found in published surface water-groundwater exchange in Red Butte Creek (Gabor et al., 2017).

Streamflow is estimated by closing a water balance and so is not calibrated in QUAL2Kw. The ratio of the root mean square error to the standard deviation of measured data (RSR), Nash-Sutcliffe efficiency (NSE), percent bias (PBIAS), and root mean square error (RMSE) for stream temperature and dissolved oxygen were calculated to evaluate model fit at three sites: (1) Cottam's Grove, (2) Foothill Drive, and (3) 1300E (Figure 3-10). Model evaluation guidelines have been established in the literature with recommended values for RSR (≤ 0.70), NSE (> 0.50), and PBIAS (≤ ± 25%) for stream temperature and dissolved oxygen for satisfactory model fit (Moriasi et al., 2007).

$$NSE = 1 - \frac{\sum_{i=1}^{n}(O_i - M_i)^2}{\sum_{i=1}^{n}(O_i - \bar{O})^2} \tag{4}$$

$$RSR = \frac{\sqrt{\sum_{i=1}^{n}(O_i - M_i)^2}}{\sqrt{\sum_{i=1}^{n}(O_i - \bar{M})^2}} \tag{5}$$

$$PBIAS = \frac{\sum_{i=1}^{n} O_i - M_i}{\sum_{i=1}^{n} O_i} * 100 \tag{6}$$

$$RMSE = \sqrt{\frac{1}{n}\sum_{i=1}^{n}(O_i - M_i)^2} \tag{7}$$

With observed data for total phosphorus limited to grab samples collected from 2013 to 2016, the percentage of time that the observed data fall within the maximum and minimum modeled total phosphorus loadings is reported (Null et al., 2013).

$$(TPmin_s < TP_O < TPmax_s) = \left(\frac{\Sigma W_{TPmin_{si} < TP_O < TPmax_{si}}}{n}\right) * 100 \tag{8}$$

Model Runs

The models were run with three base case conditions and 21 green stormwater infrastructure alternatives, and two model time periods (spring runoff and baseflow), for a total of 48 model runs (Table 3-1). Modeled changes in runoff volume and total phosphorus loading from implementing rain gardens that received stormwater falling on roofs, grass swales along streets, and bioretention cells in parking lots at two sites in the Red Butte Creek watershed using WINSLAMM, a stormwater quality model (Fernández Velásquez, 2018), were used as input to the water quality modeling. Fernández Velásquez (2018) modeled reductions in runoff volume from green infrastructure allowing stormwater to infiltrate for baseflow and groundwater supply. Therefore, in the green infrastructure alternative model runs, the fraction of the flow that interacts with the hyporheic storage was increased by 25 percent, which Gabor et al. (2017) found evidence

for during a storm event from river kilometer 2.51 and 3.55.

WINSLAMM models simulated 10%, 50%, and 100% implementation of green stormwater infrastructure at two sites named Connor Road and Dentistry Building (Fernández Velásquez, 2018). The WINSLAMM model estimated slightly larger changes in flow and total phosphorus for the Connor Road site (e.g. 83.2% reduction in flow and 71.6% reduction in TP with the 100% implementation alternative) compared to the Dentistry Building site (e.g. 78.6% reduction in flow and 56.3% reduction in TP for 100% implementation alternative). Therefore, this study used the Connor Road values as the upper bound estimate and the Dentistry Building values as the lower bound estimate of changes from the green infrastructure implementation.

Modeled changes to stream temperature with green infrastructure implementation are driven by streamflow changes which alter the thermal mass of the river. Modeled dissolved oxygen concentrations with green infrastructure implementation are driven by the simulated changes to stream temperature and total phosphorus loadings from green infrastructure. Model runs 2 - 13 simulate 10%, 50%, and 100% implementation of different combinations of green stormwater infrastructure at the Connor Road and Dentistry Building sites (Table 3-1). The next six alternatives simulate 1.3%, 6.5%, and 13% implementation of green stormwater infrastructure throughout the Red Butte Creek watershed. The last five model runs simulate the base case and four alternatives for the Jordan River.

Table 3-1. Model Runs

	Connor Road and Dentistry Building Storm Drain Alternatives
1	Base case for Red Butte Creek models
2	10% Grass swales
3	10% Bioretention cells
4	10% Rain gardens
5	10% Grass swales, bioretention cells, and rain gardens
6	50% Grass swales
7	50% Bioretention cells
8	50% Rain gardens
9	50% Grass swales, bioretention cells, and rain gardens
10	100% Grass swales
11	100% Bioretention cells
12	100% Rain gardens
13	100% Grass swales, bioretention cells, and rain gardens
	Red Butte Creek Watershed Alternatives
14	10% Grass swales, bioretention cells, and rain gardens using high estimate
15	10% Grass swales, bioretention cells, and rain gardens using low estimate
16	50% Grass swales, bioretention cells, and rain gardens using high estimate
17	50% Grass swales, bioretention cells, and rain gardens using low estimate
18	100% Grass swales, bioretention cells, and rain gardens using high estimate
19	100% Grass swales, bioretention cells, and rain gardens using low estimate
	Jordan River Watershed Alternatives
20	Base case for Jordan River models
21	100% GI implementation in Red Butte Watershed using high estimate
22	100% GI implementation in Red Butte Watershed using low estimate
23	100% GI implementation in 7 canyon creek watersheds using high estimate
24	100% GI implementation in 7 canyon creek watersheds using low estimate
	Sensitivity Analysis
25	100% GI implementation in Red Butte Watershed with 10% more flow from RBC
26	100% GI implementation in Red Butte Watershed with 10% less flow from RBC
27	100% GI implementation in 7 canyon creek watersheds with 10% more flow from RBC
28	100% GI implementation in 7 canyon creek watersheds with 10% less flow from RBC
29	100% GI implementation in Red Butte Watershed with 20% more flow from RBC
30	100% GI implementation in Red Butte Watershed with 20% less flow from RBC
31	100% GI implementation in 7 canyon creek watersheds with 20% more flow from RBC
32	100% GI implementation in 7 canyon creek watersheds with 20% less flow from RBC

Sensitivity Analysis – To account for the uncertainty of streamflow beyond Liberty Lake where Red Butte Creek flows underground near 1300E until it meets with the Jordan River, I conducted a sensitivity analysis on changes in streamflow. Sixteen model runs estimate the percentage change in flow, stream temperature, dissolved oxygen, and total phosphorus to Great Salt Lake when the inflow from Red Butte Creek is 10% or 20% less or more than measured flow at 1300E (the downstream boundary condition of models) (Table 3-1).

Results

Model Fit

NSE, RSR, and PBIAS values were all satisfactory or better for stream temperature and dissolved oxygen concentration at the three measured sites along Red Butte Creek (Table 3-2 – Table 3-4, Figure 3-11 – Figure 3-16) (Moriasi et al., 2007) for the base case modeling scenarios. Model fit was typically better at upstream locations of Cottam's Grove and Foothill Drive, compared to the 1300 E site. Previous research has found that groundwater influences this system, particularly in the reaches between Foothill Drive and 1300 E (Gabor et al., 2017), which may not have been captured fully with the modeling here due to data availability. For total phosphorus, the percentage of time that the observed data fell between the minimum and maximum modeled data was reported and shows more than 50 percent for all sites in both the spring and summer models (Figure 3-17 and Figure 3-18).

Table 3-2. Model fit statistics at Cottam's Grove for base case modeling scenario

spring model	**NSE**	**RSR**	**PBIAS**	**RMSE**
DO	0.53	0.59	0.74	0.08 mg/L
stream temp	0.55	0.57	3.01	0.35°C
summer model	**NSE**	**RSR**	**PBIAS**	**RMSE**
DO	0.85	0.37	0.90	0.10 mg/L
stream temp	0.93	0.26	-1.24	0.33°C

Table 3-3. Model fit statistics at Foothill Drive for base case modeling scenario

spring model	**NSE**	**RSR**	**PBIAS**	**RMSE**
DO	0.68	0.56	0.11	0.08 mg/L
stream temp	0.70	0.54	1.36	0.38°C
summer model	**NSE**	**RSR**	**PBIAS**	**RMSE**
DO	0.84	0.39	0.59	0.11 mg/L
stream temp	0.87	0.36	-0.68	0.47°C

Table 3-4. Model fit statistics for 1300 E for base case modeling scenario

spring model	**NSE**	**RSR**	**PBIAS**	**RMSE**
DO	0.51	0.69	-0.38	0.16 mg/L
stream temp	0.51	0.70	-0.09	0.21°C
summer model	**NSE**	**RSR**	**PBIAS**	**RMSE**
DO	0.51	0.69	-0.33	0.12 mg/L
stream temp	0.71	0.54	-0.43	0.42°C

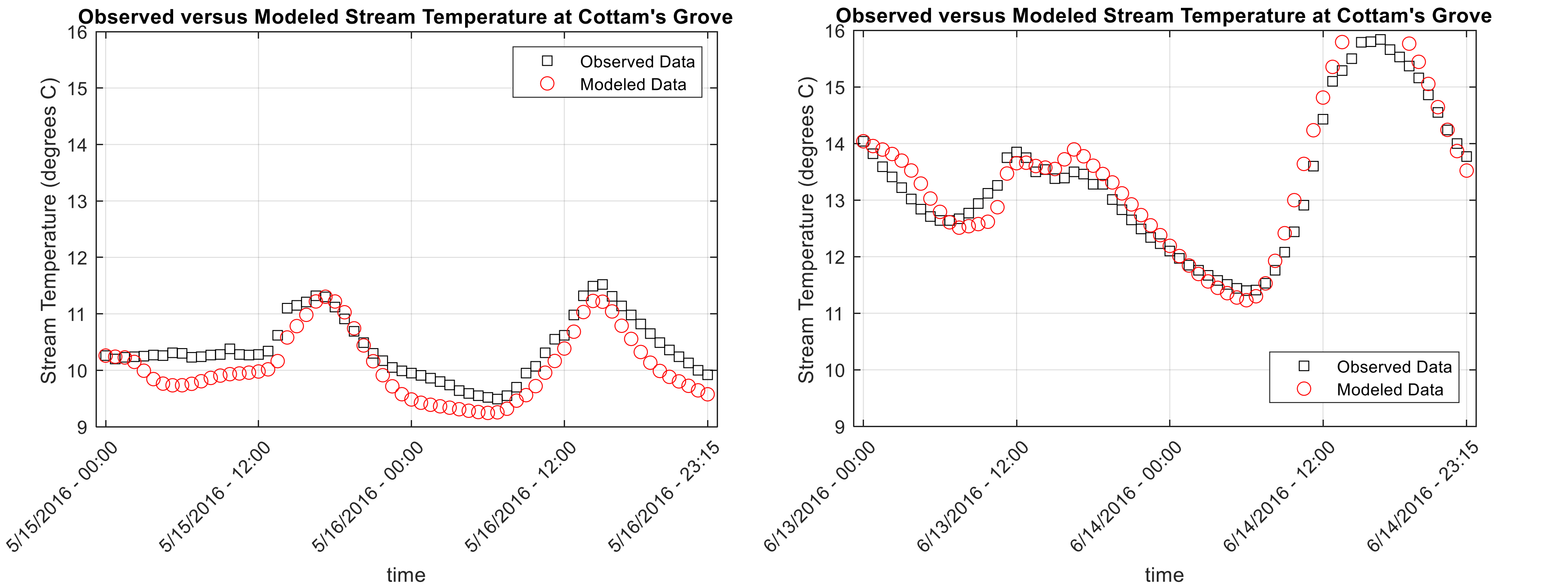


Figure 3-11. Observed versus modeled stream temperature at Cottam's Grove for the spring (left) and summer (right) models for base case modeling scenario.

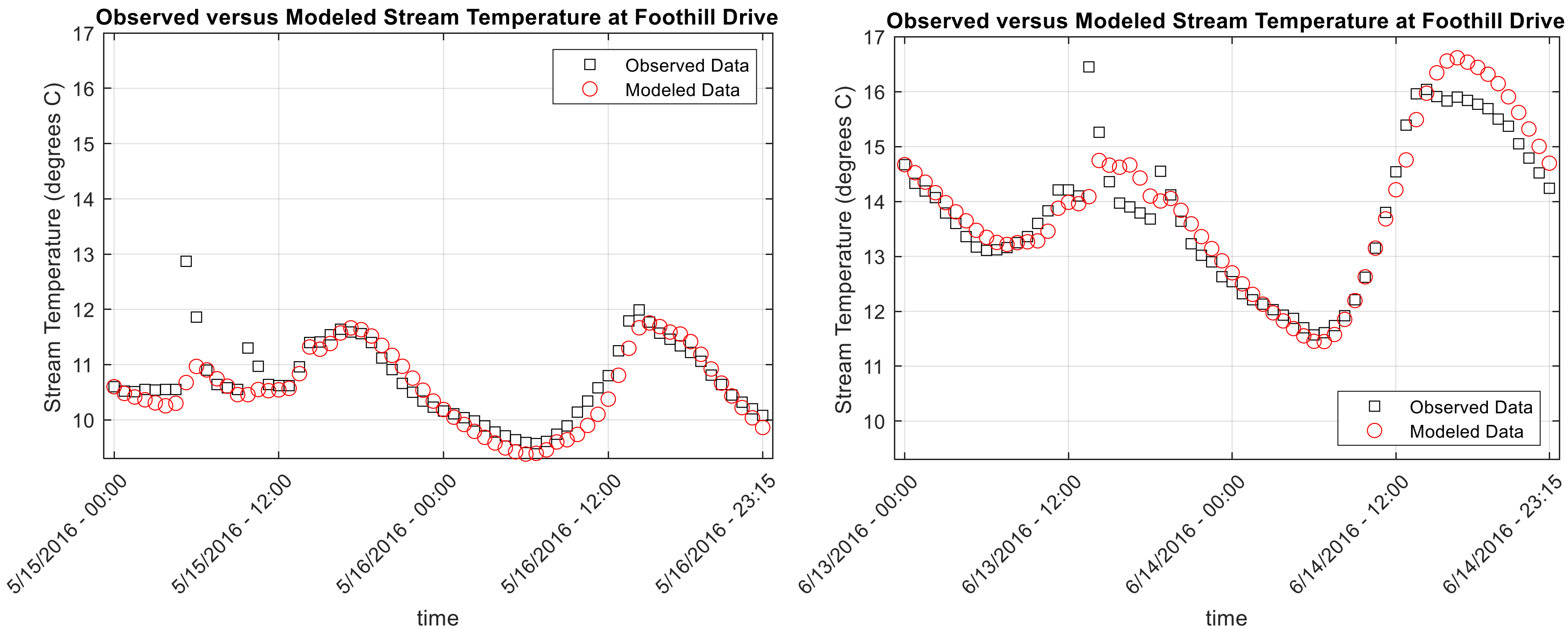


Figure 3-12. Observed versus modeled stream temperature at Foothill Drive for the spring (left) and summer (right) models for base case modeling scenario.

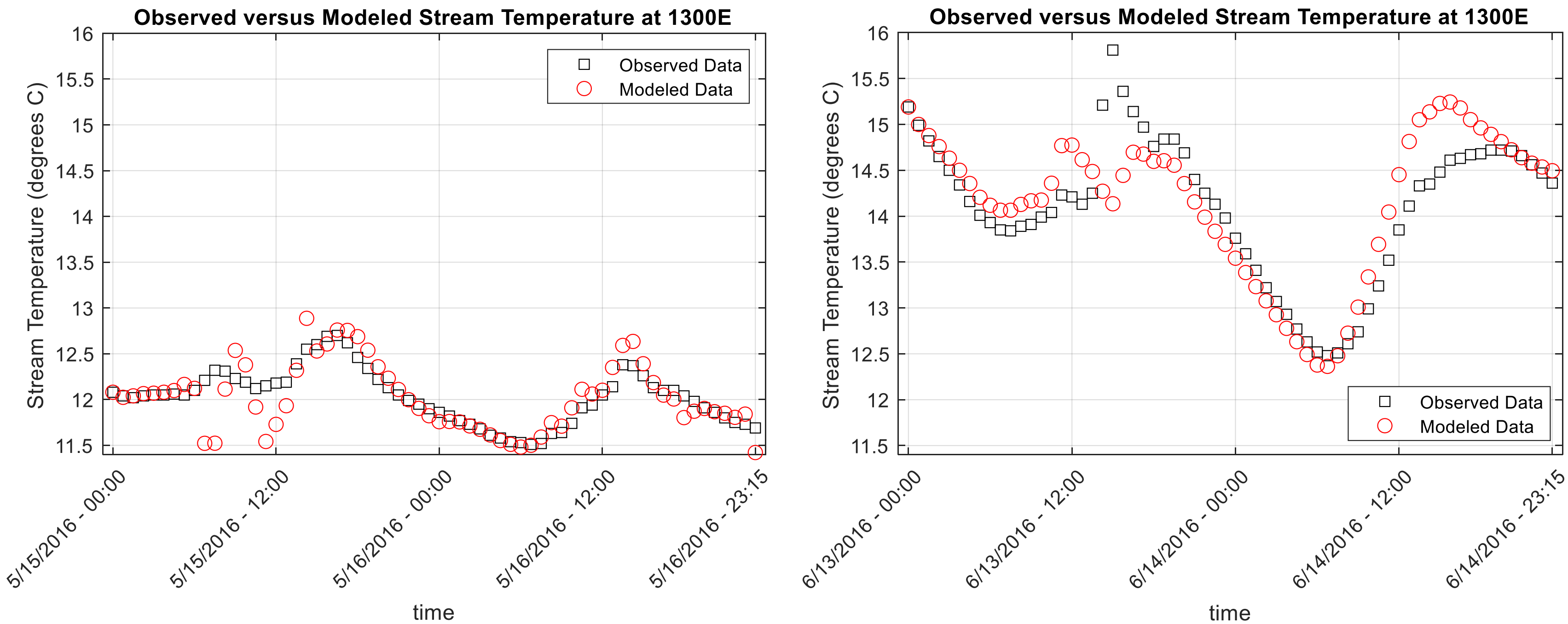


Figure 3-13. Observed versus modeled stream temperature at 1300E for the spring (left) and summer (right) models for base case modeling scenario.

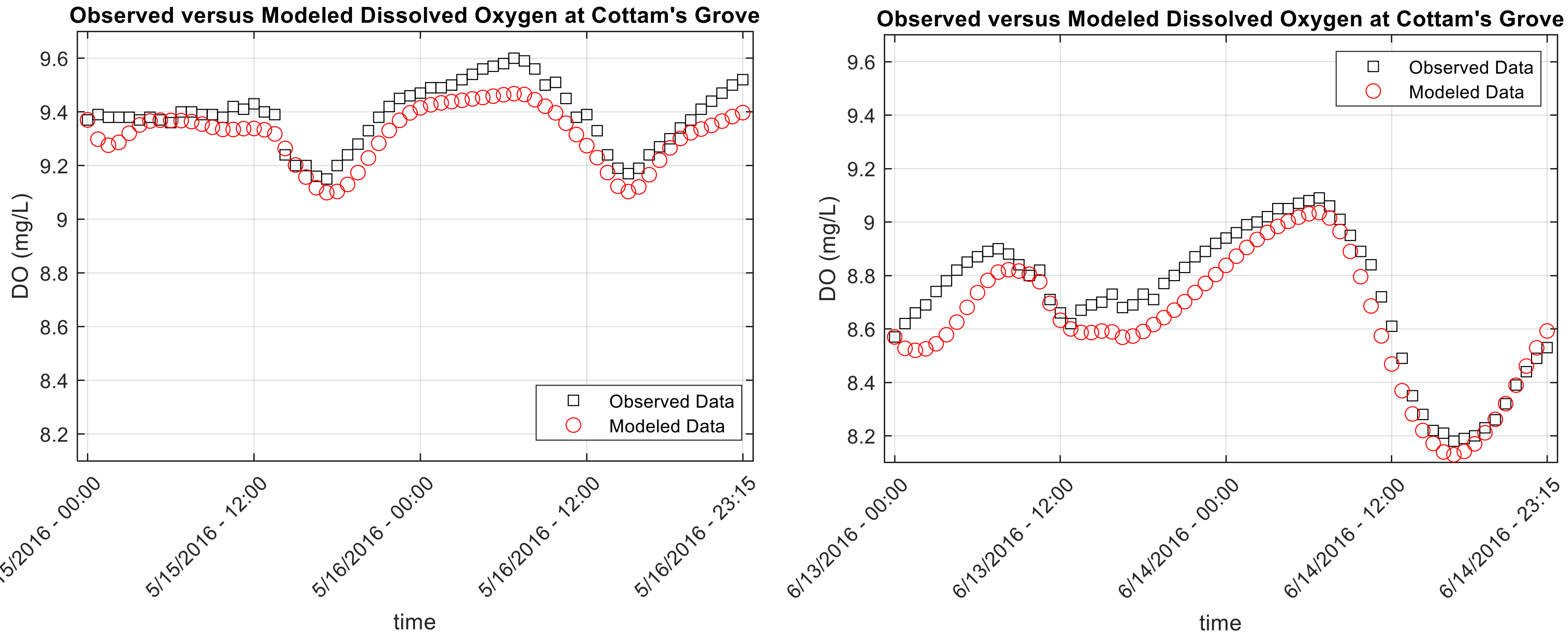


Figure 3-14. Observed versus modeled dissolved oxygen at Cottam's Grove for the spring (left) and summer (right) models for base case modeling scenario.

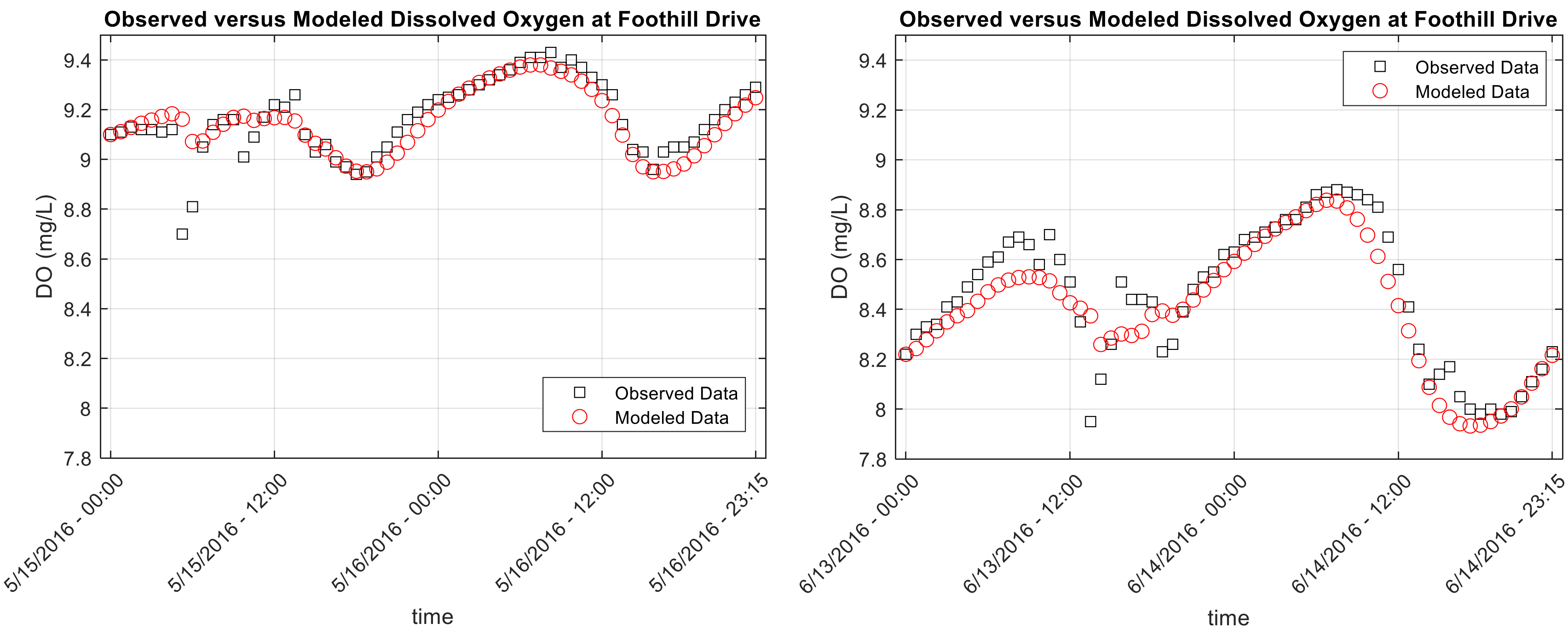


Figure 3-15. Observed versus modeled dissolved oxygen at Foothill Drive for the spring (left) and summer (right) models for base case modeling scenario.

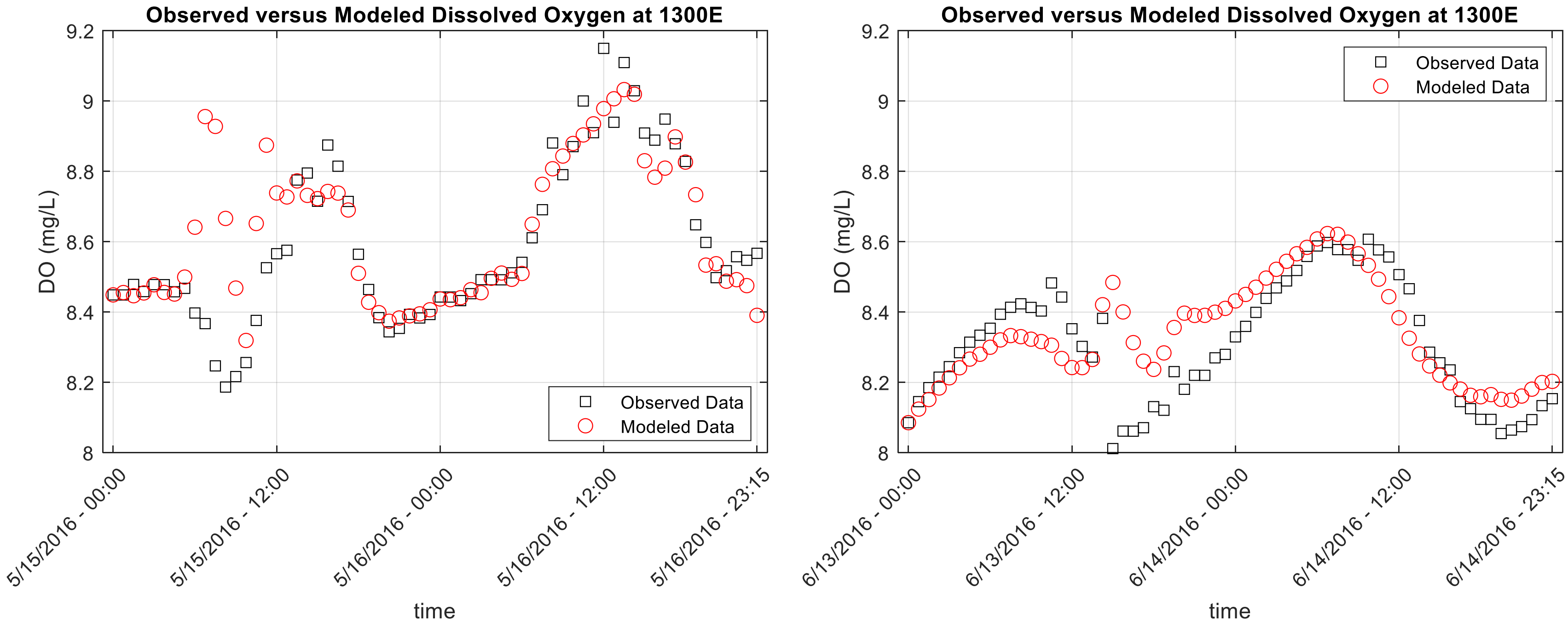


Figure 3-16. Observed versus modeled dissolved oxygen at 1300E for the spring (left) and summer (right) models for base case modeling scenario.

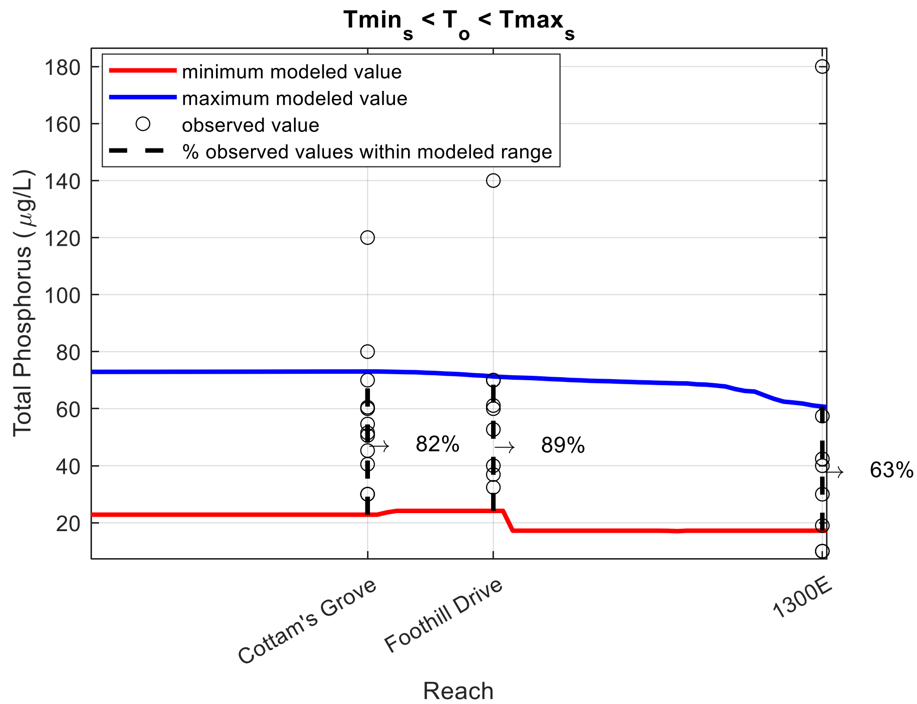


Figure 3-17. Percentage of time that observed TP values fall between the minimum and maximum modeled TP for the spring model for base case modeling scenario.

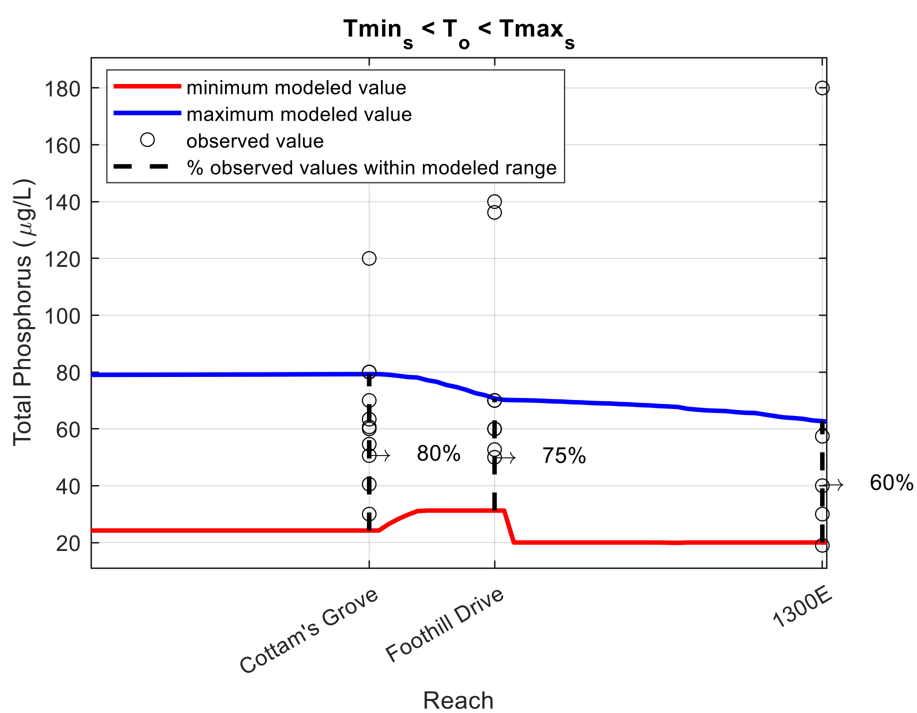


Figure 3-18. Percentage of time that observed TP values fall between the minimum and maximum modeled TP for the summer model for base case modeling scenario.

Green Infrastructure Implementation at Connor Road and Dentistry Building Sites

Twelve model runs evaluated stormwater contributions to Red Butte Creek from the green infrastructure at Connor Road and Dentistry Building storm drains. The runs predicted the impact of implementing 10%, 50%, and 100% of grass swales, bioretention cells, rain gardens, and combined types of green infrastructure that could be implemented at a given site on the reach scale. Changes to streamflow and total phosphorus concentrations from green infrastructure implementation occur downstream when green infrastructure is modeled in the reaches between Cottam's Grove and Foothill Drive where there are four storm drains at river kilometers 2.70, 2.93, 3.22, and 3.32.

Streamflow is projected to be reduced in all the Connor Road and Dentistry Building alternatives for both the spring and summer models (Figure 3-19, Figure 3-20, Figure 3-21, Table 7-5, and Table 7-6). However, these modeled changes are less than 0.1 cms, and therefore, not significant. Total phosphorus is projected to decrease by 1.1% (0.6μg/L) with implementing 50% of roof runoff areas with rain gardens, 3.6% (1.8 μg/L) with implementing 50% of streets with grass swales alongside them, and 4.4% (2.3 μg/L) with implementing 50% of parking lots with bioretention cells in the spring model (Figure 3-20 and Table 7-5). With 100% implementation of all available roofs, parking lots, and swales with green infrastructure at these two sites, total phosphorus is projected to reduce by 12.3% (6.3 μg/L). For the summer model, total phosphorus is reduced by 4.4% (2.5 μg/L) with 50% implementation of bioretention cells in parking lots and 3.6% (2.1 μg/L) with 50% implementation of grass swales along streets (Figure 3-21 and Table 7-6). Implementation of 50% of rain gardens for roof runoff shows a non-significant increase in total phosphorus. This is due to an already existing detention pond at the Dentistry Building site, which performs better than these lower levels of rain gardens for

roof runoff (Fernández Velásquez, 2018). This increase in total phosphorus turns to a reduction at the 100% implementation of rain gardens for roof runoff, when this level of implementation decreases total phosphorus more than the detention pond.

In both the spring and summer models, stream temperature is not projected to significantly be altered, as the modeled changes from the base case are less than the RMSE (Table 3-2 – Table 3-4). The non-significant stream temperature increases in these alternatives are due to reduced flows and increased atmospheric heating, as air temperature is warmer than the temperature of the water (Figure 7-5). Increases in stream temperature are not the desired impact from green infrastructure, and it is likely due to the model missing surface water-groundwater interactions. This limitation is discussed further in the next section.

Changes in dissolved oxygen concentration from green infrastructure at the reach scale is not significant, or less than the RMSE (Table 3-2 – Table 3-4). While the impacts are not significant, predicted dissolved oxygen changes differ in the spring alternatives versus the summer alternatives, highlighting the multiple interactions that impact dissolved oxygen. In the summer model, dissolved oxygen is projected to be reduced due to the higher temperatures in this modeling period. With the spring alternatives, dissolved oxygen is predicted to increase due to the reduced total phosphorus and consequently less plant growth (Figure 3-20). Additionally, there is more flow and less atmospheric heating for the spring model relative to the summer model, where all alternatives show decreases in dissolved oxygen due to increased stream temperature (Figure 3-21).

Overall, grass swales and bioretention cells reduced total phosphorus more than rain gardens. This result begins to highlight that certain types of green infrastructure help

the delivery of certain ecosystem services over others. Specifically, grass swales and bioretention cells may perform better with the service of water quality improvement. Managers will have to consider potential differences in ecosystem service delivery from different green infrastructure when making decisions regarding green infrastructure types to implement.

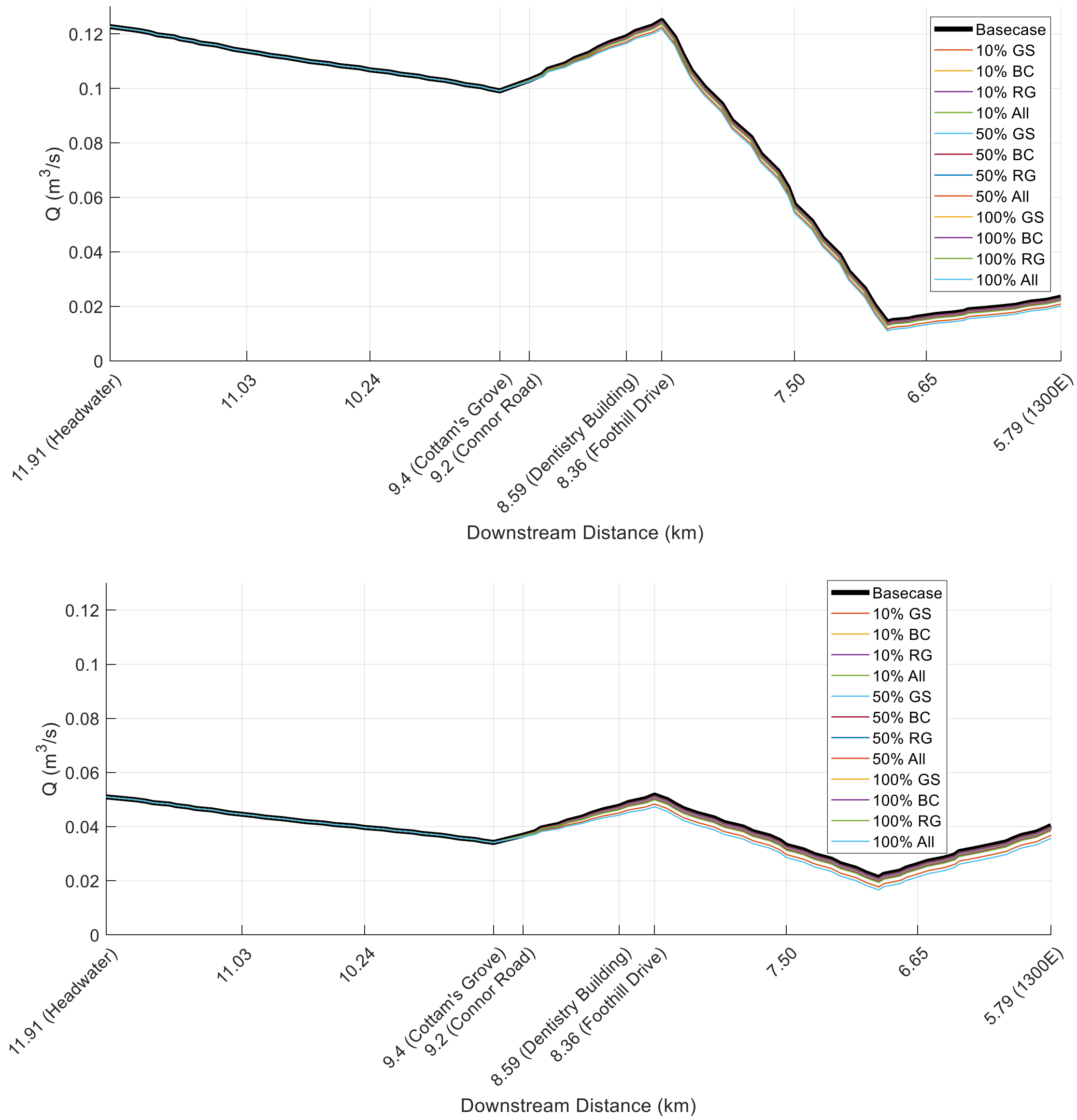


Figure 3-19. Average flow for spring (top) and summer (bottom) models for all 12 green infrastructure alternatives at Connor Road and Dentistry Building.

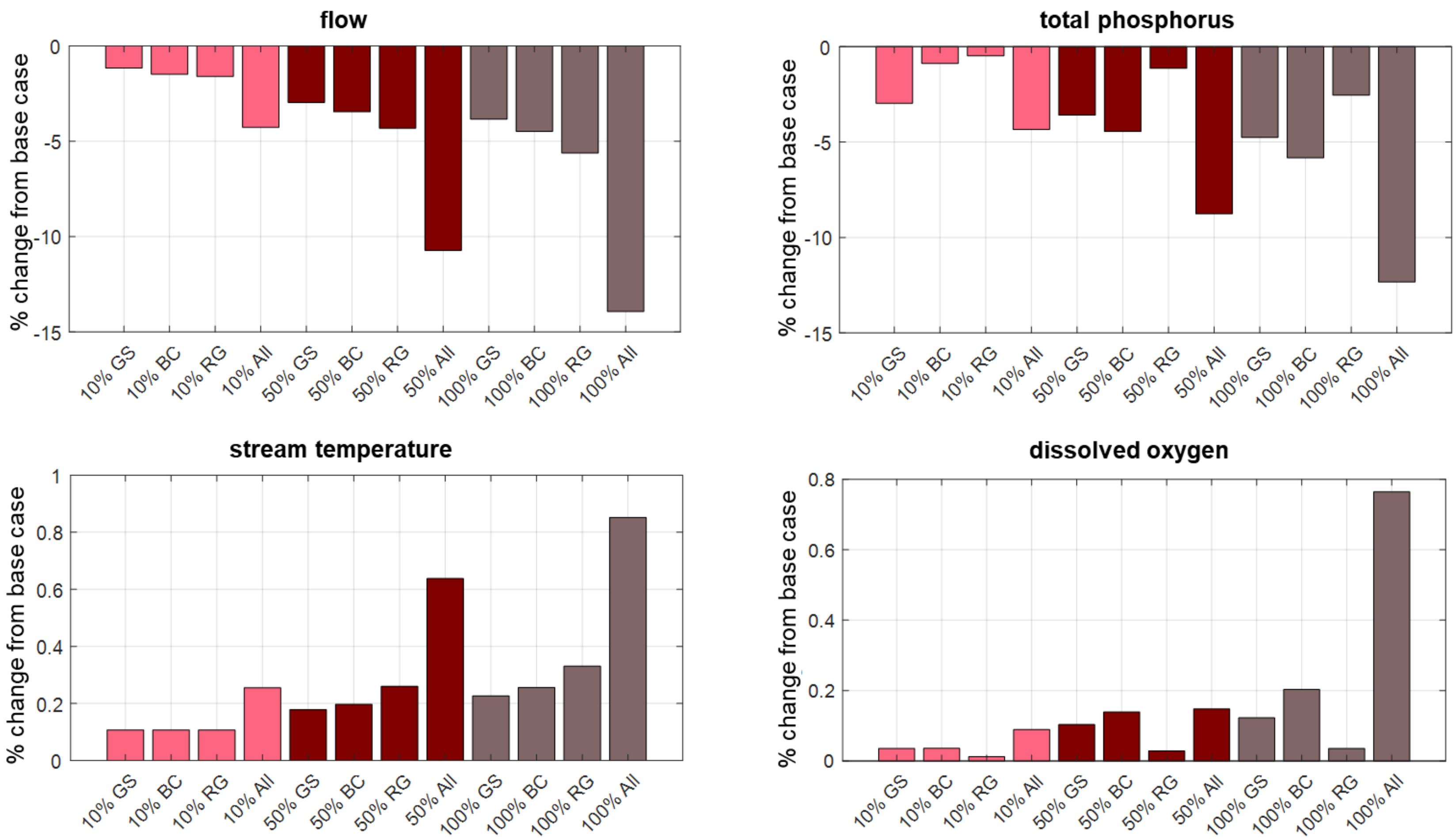


Figure 3-20. Percent changes for flow, total phosphorus, stream temperature, and dissolved oxygen between spring reach scale base case and alternatives at 1300E.

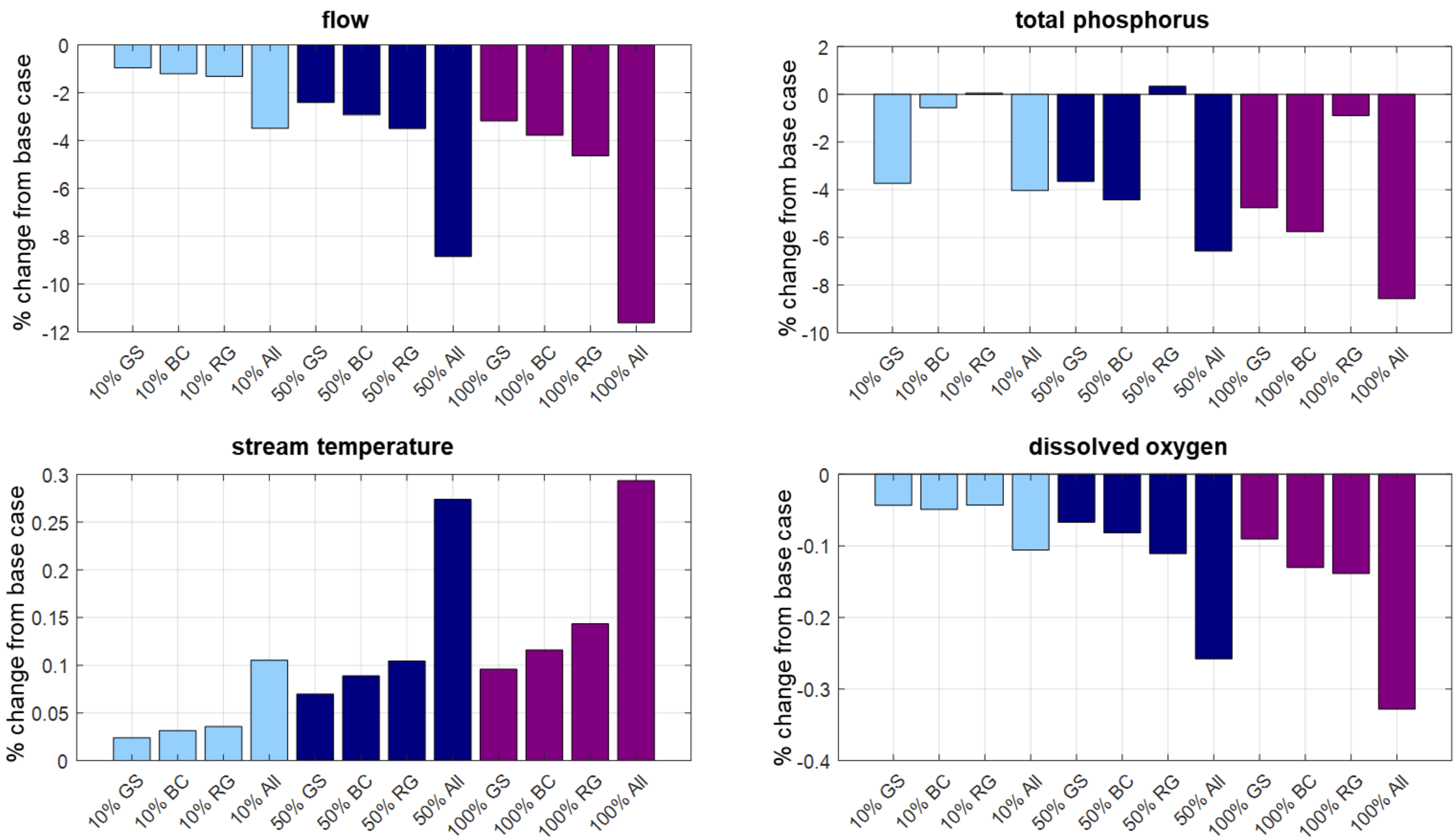


Figure 3-21. Percent changes for flow, total phosphorus, stream temperature, and dissolved oxygen between summer reach scale base case and alternatives at 1300E.

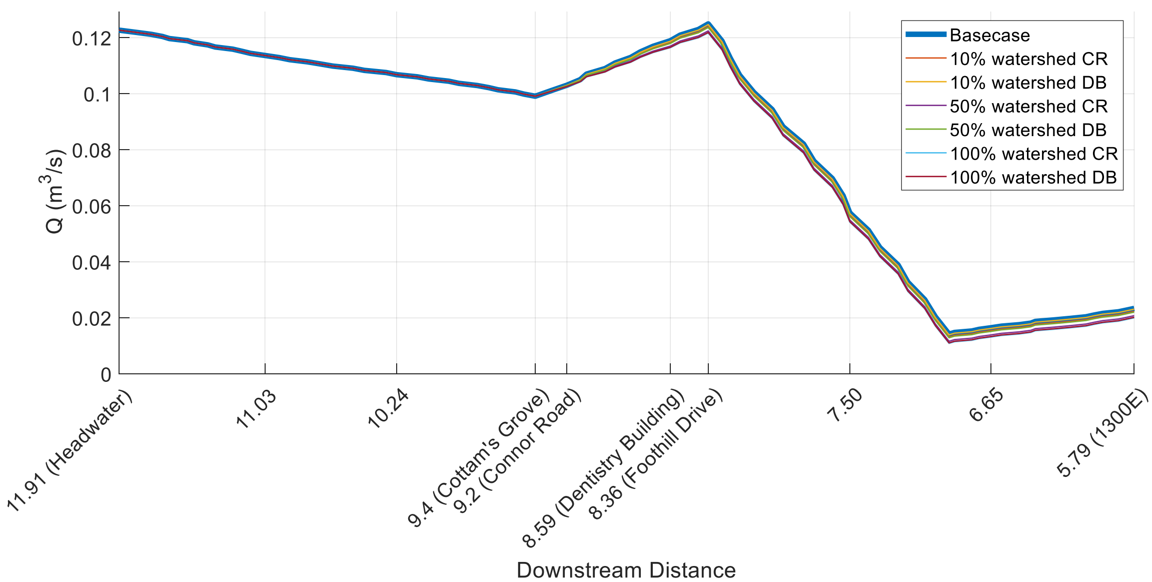
Green Infrastructure Implementation throughout Red Butte Watershed

Six model runs assess changes in streamflow, total phosphorus, stream temperature, and dissolved oxygen from 1.3%, 6.5%, and 13% green infrastructure implementation (in the gaining reaches) as watershed alternatives for Red Butte Creek watershed. Each implementation percentage or level includes estimates for grass swales, bioretention cells, rain gardens, and all three combined. There is a high and low estimate of these changes from the estimate at the Connor Road site (high estimate) compared to the estimates from the Dentistry Building site (low estimate). Overall, the trends at the Red Butte watershed scale are similar to those for the smaller scale Connor Road and Dentistry Building sites.

In the spring and summer alternatives, streamflow is not significantly decreased when green infrastructure is implemented on the small watershed scale, similar to the reach scale results (Figure 3-22, Figure 3-23, Figure 3-24, Table 7-5, and Table 7-6). The ranges for total phosphorus reductions in the spring model are 2.8% (1.4 µg/L) to 4.9% (2.5 µg/L) with 1.3% green infrastructure implementation and up to 11.4% (5.9 µg/L) to 13.3% (6.8 µg/L) when 13% of the Red Butte Creek watershed (roofs, streets, and parking lots in the gaining reach of Red Butte Creek) are treated with rain gardens, grass swales, and bioretention cells, respectively (Figure 3-23 and Table 7-5). In the summer alternatives, total phosphorus is decreased the least, by 3.4% (1.9 µg/L) to 4.7% (2.7 µg/L), for the 1.3% implementation level and the most, by 7.4% (4.2 µg/L) to 9.2% (5.2 µg/L), with 13% implementation of green infrastructure alternatives at the watershed scale (Figure 3-24 and Table 7-6).

Impacts on stream temperature and dissolved oxygen from green infrastructure at

this scale are less than the RMSE, and therefore, not significant. The non-significant trends for both constituents are the same as is shown at the reach scale in the previous section (Figure 3-23, Figure 3-24, Table 7-5 and Table 7-6).



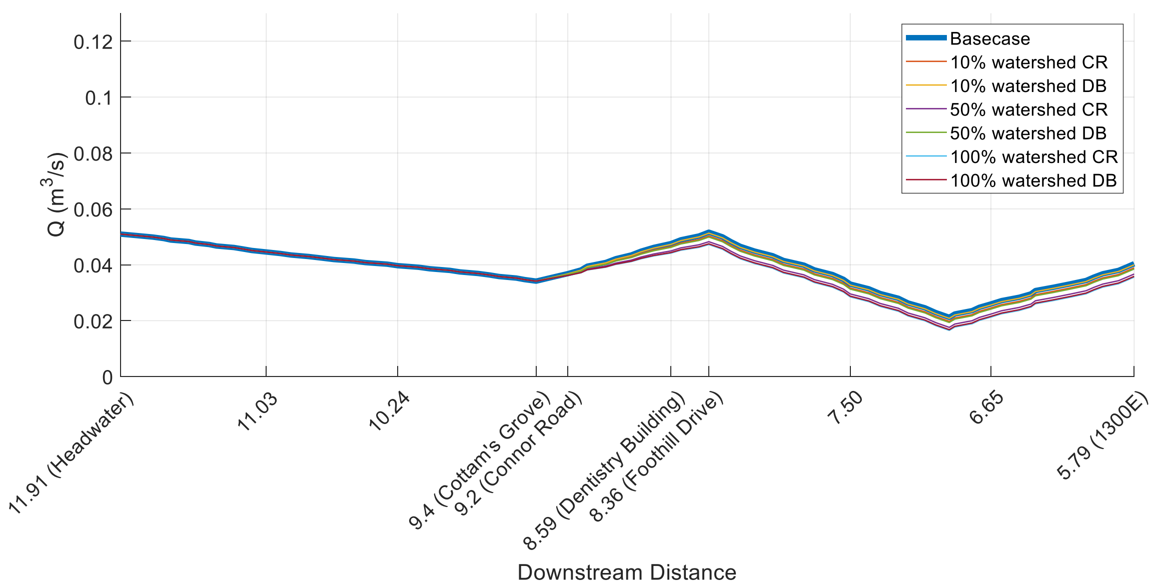


Figure 3-22. Average flow for spring (top) and summer (bottom) models for six Red Butte watershed green infrastructure alternatives.

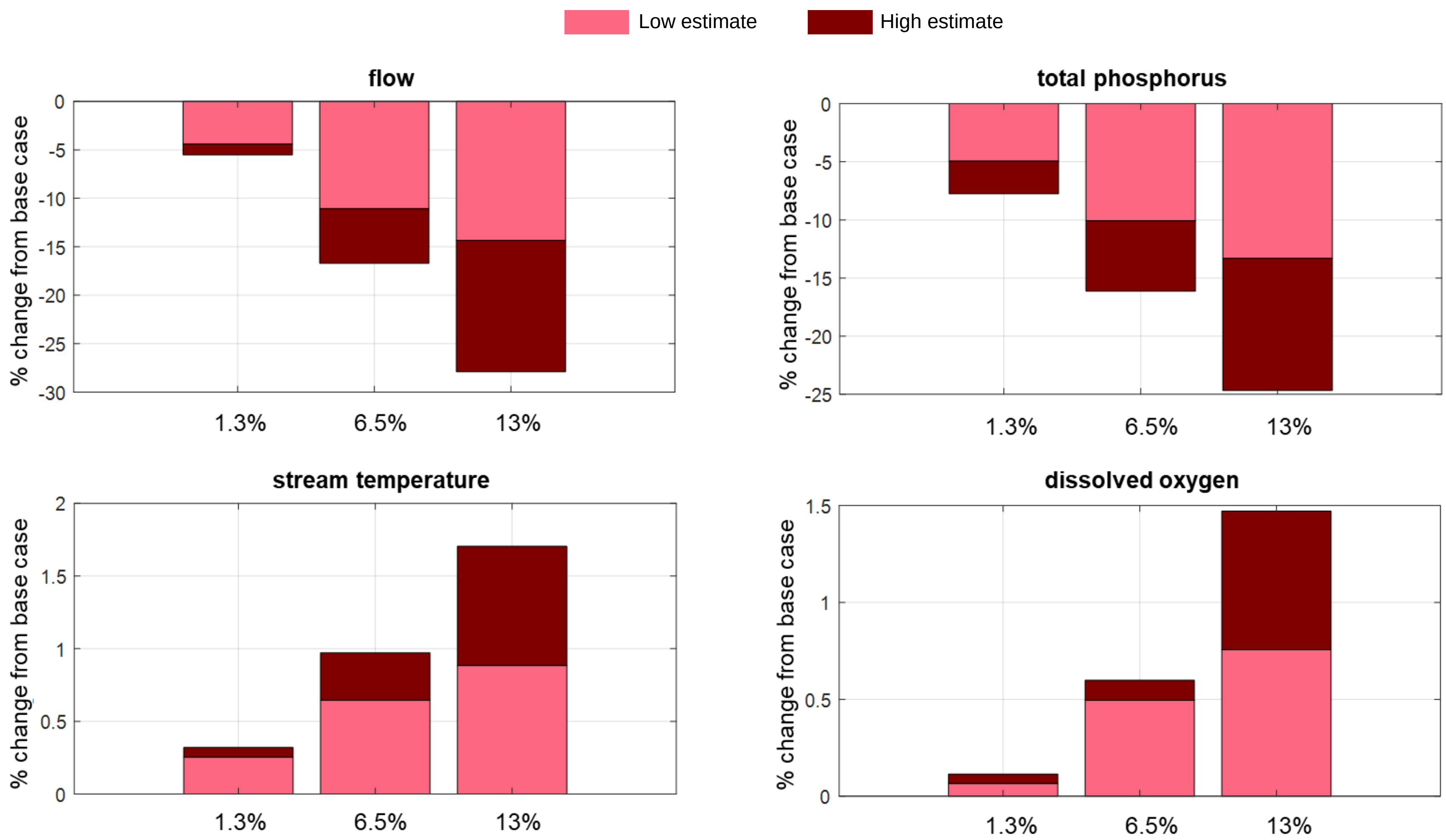


Figure 3-23. Percent changes for flow, total phosphorus, stream temperature, and dissolved oxygen between spring Red Butte watershed base case and alternatives at 1300E.

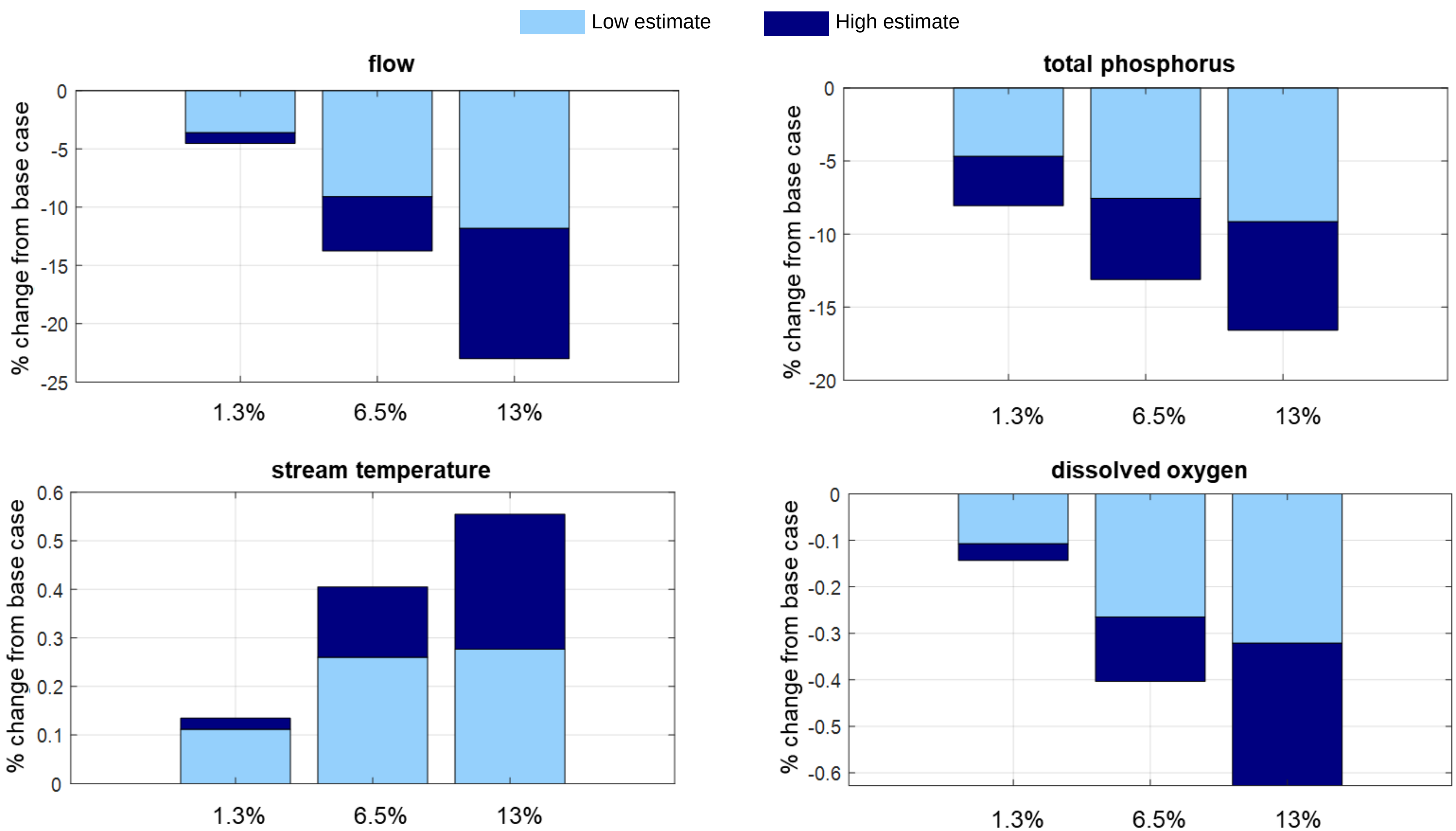


Figure 3-24. Percent changes for flow, total phosphorus, stream temperature, and dissolved oxygen between summer Red Butte watershed base case and alternatives at 1300E.

Green Infrastructure Implementation throughout Jordan River Watershed

Two model runs examine changes in streamflow and water quality in the Jordan River when green infrastructure is implemented at 13% in the small Red Butte Creek watershed. Another two model runs are used to estimate changes when green infrastructure is implemented in Red Butte Creek and six other creeks that flow into the Jordan River, using percentage change estimates from the Red Butte Creek models. Implementation of green infrastructure is modeled for seven tributaries to the Jordan River: Little Cottonwood, Big Cottonwood, Mill Creek, Parley's, Emigration, Red Butte Creek, and City Creeks. Changes in streamflow, stream temperature, dissolved oxygen, and total phosphorus occur starting where Little Cottonwood Creek meets the Jordan River 34.75 km upstream from the end of the model (Figure 3-25). The Jordan River models simulate conditions in the winter/non-irrigation season (February) and summer/irrigation season (August).

Red Butte Creek into the Jordan River – Overall, implementing green infrastructure in the Red Butte watershed alone, has negligible impacts of less than a quarter of a percent on streamflow, stream temperature, dissolved oxygen, and total phosphorus at the confluence of the Jordan River with Great Salt Lake for the winter/spring and late summer models. These are non-significant effects from treating streets, roofs, and parking lots in the gaining reaches of Red Butte Creek, which is approximately 13% of the watershed, with grass swales, rain gardens, and bioretention cells does not result in significant changes at the outlet of the Jordan River model.

Seven Canyon Creeks into the Jordan River –Implementing green infrastructure in the seven west-slope Wasatch subwatersheds reduced streamflow by 9% (0.22 cms) to

9.5% (0.23 cms) in winter/spring runoff conditions, and 8.8% (0.39 cms) to 9.3% (0.41 cms) in late summer baseflow conditions. Total phosphorus is predicted to be reduced by 1% (3.9 µg/L) to 1.2% (4.5 µg/L) in the winter/spring model and 7.7% (51.5 µg/L) to 8.6% (57.4 µg/L) in late summer.

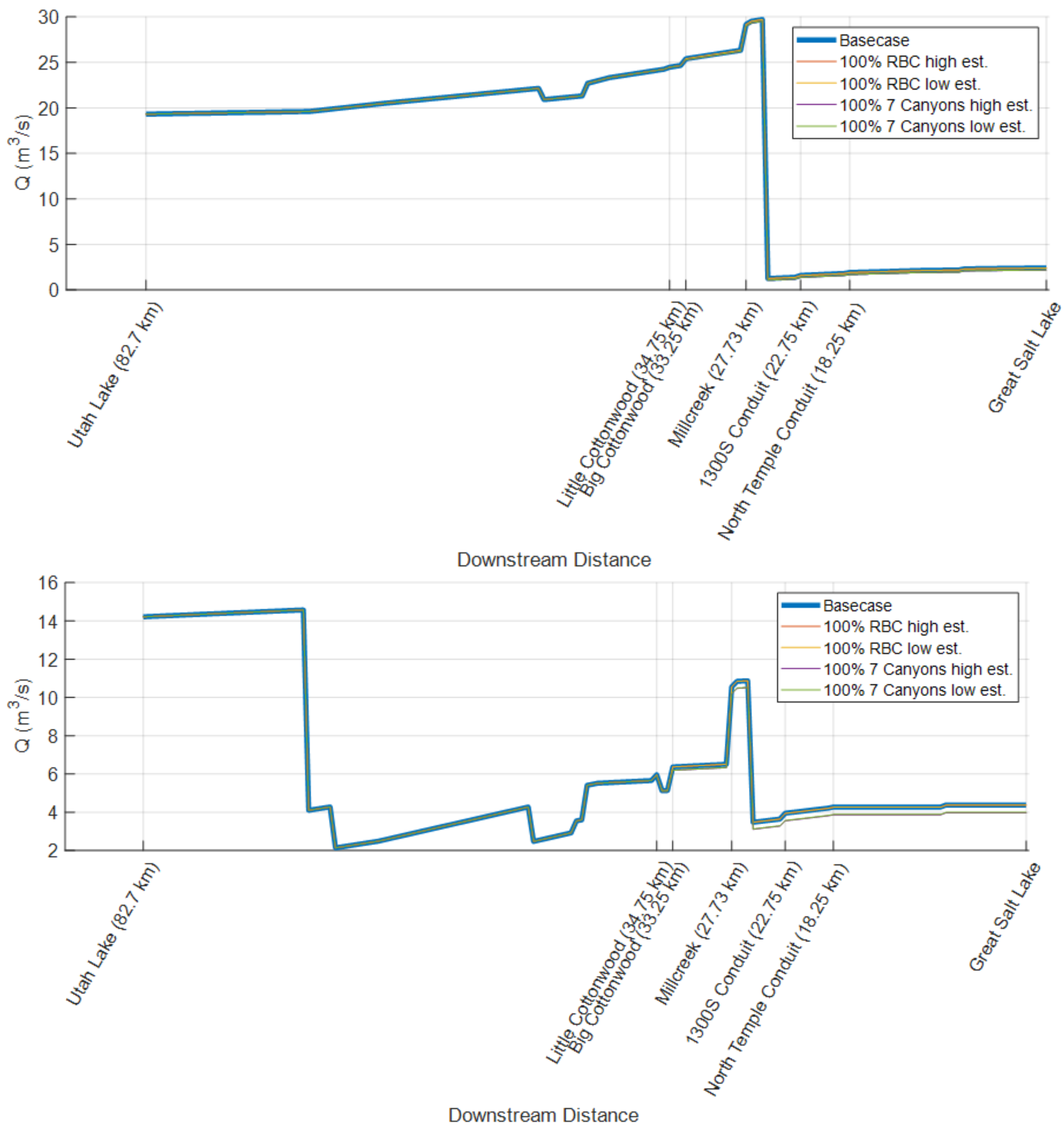


Figure 3-25. Average flow for the winter/spring (top) and late summer (bottom) models at the Jordan River watershed scale.

Stream temperature was predicted to decrease in the winter/spring model alternatives by 16.5% (0.23°C) to 17.4% (0.24°C) (Table 3-5). The decrease in the winter/spring model is a result of atmospheric conditions cooling the river, while the decrease in the late summer alternatives is due to the slight reduction of warmer inflows into the Jordan River. Lastly, dissolved oxygen is projected to increase in the winter/spring alternatives by 1.2% (0.12 mg/L) to 1.3% (0.13 mg/L) (Table 3-5).

Table 3-1. Percent changes from base case (%) for Flow, Ts, DO, and TP at the end reach of the Jordan River models

*significant/above model error

winter/spring model				
	Flow	**Stream temperature**	**Dissolved oxygen**	**Total phosphorus**
13% 7 Canyons high estimate	-9.5*	-17.4*	1.3*	-1.2*
13% 7 Canyons low estimate	-9.0*	-16.5*	1.2*	-1.0*
late summer model				
	Flow	**Stream temperature**	**Dissolved oxygen**	**Total phosphorus**
13% 7 Canyons high estimate	-9.3*	-0.44	-1.0	-8.6*
13% 7 Canyons low estimate	-8.8*	-0.41	-0.9	-7.7*

Sensitivity Analysis – Changes in flow, stream temperature, dissolved oxygen, and total phosphorus at the outlet of the Jordan River are not significant when Red Butte Creek flows in the Jordan River are -/+ 10% and -/+ 20%. In every case for the winter/spring and late summer models, the percentage change is less than 0.5% and the model error for streamflow, total phosphorus, stream temperature, and dissolved. This indicates that uncertainty in the inflows from Red Butte Creek does not result in Jordan River streamflow and water quality constituents that are measurably different from the changes in the alternatives with 13% of Red Butte Creek and 7 Canyon Creeks watersheds implemented with green infrastructure.

Discussion

Researchers and water managers are hopeful that stormwater infrastructure, like rain barrels, detention and retention basins, and green roofs, will improve ecosystem services including flood control, climate regulation, and stormwater quality (Prudencio and Null, 2018; Tzoulas et al., 2007). Overall, this study evaluates the potential of green stormwater infrastructure to enhance ecosystem services of streamflow quantity, water quality, and reduction of flood peak. The modeling in this study simulates changes in surface water volume and quality from implementing different types, amounts, and spatial scales of green stormwater infrastructure. Overall, the green infrastructure alternatives at all three scales reduce total phosphorus concentrations, therefore exhibiting the water quality improvement service from green stormwater infrastructure. Secondly, stormwater runoff is a flood risk during spring runoff in this urban system. In the spring model, green infrastructure captures runoff to mitigate flooding. Often in these alternatives, stream temperature is increased, and dissolved oxygen is decreased. The findings for changes in stream temperature and dissolved oxygen show that atmospheric heating drive these effects.

Reach scale alternatives also highlight the differences between green infrastructure types and their deliveries of different ecosystem services. Rain gardens reduce streamflow the most, which is beneficial for flood mitigation during spring runoff conditions, but not during summer baseflow conditions when more streamflow is beneficial. Grass swales reduce total phosphorus to the stream the most and decrease streamflow the least.

The Red Butte Creek models were connected to an existing Jordan River model to

provide insight on the effects of green stormwater infrastructure for aquatic habitats in the Jordan River and Great Salt Lake. Green infrastructure alternatives at the watershed-scale demonstrate that implementation of green infrastructure in multiple sub-watersheds is needed to produce a noticeable change downstream in a larger basin. Specifically, the implementation of green infrastructure across 13% of the urban area in the Red Butte Creek watershed, or 0.585 km^2 of roofs, streets, and parking lots, leads to fractions of a percentage change for streamflow, total phosphorus, stream temperature, and dissolved oxygen at the end of the Jordan River model, Great Salt Lake. These effects are less than the model error, indicating non-significance. To see measurable change, at least 13% of urban areas, or 3.51 km^2 total roofs, parking lots, and streets must be implemented with green infrastructure in the seven canyon creek watersheds. Specifically, green infrastructure implementation in the seven canyon creek watersheds lead to at most about a 9.5% (0.23 cms) reduction in streamflow, a 17.4% (0.24°C) reduction in stream temperature, a 1.3% (0.13 mg/L) increase in dissolved oxygen concentration, and a 1.2% (4.5 µg/L) reduction in total phosphorus concentration in the winter/spring Jordan River model. In the late summer Jordan River model, there is at most about a 9.3% (0.41 cms) decrease in streamflow and a 8.6% (57.4 µg/L) reduction in total phosphorus concentration.

This modeling serves as a tool to manage for Jordan River Total Maximum Daily Loads (TMDLs). The Jordan River currently has one approved TMDL for dissolved oxygen (Adams and Arens, 2013). The TMDL limits total organic matter to 3,983 kg day to achieve a desired level of 5.5 mg/L for dissolved oxygen concentration. The TMDL lists total phosphorus as a pollutant of concern in the Jordan River near Salt Lake City

(from around N. Temple to 2100S). For this reach, total phosphorus concentrations were reduced by 3-6 percent (18-51 μg/L, depending on the season) when implementing green infrastructure in the seven canyon creeks watersheds. The modeling in this study addresses the potential of using green stormwater infrastructure for the ecosystem service of water quality improvement. I show that implementing green infrastructure alone is unlikely to satisfy TMDL targets in this system, although green infrastructure implementation can be used in conjunction with other strategies, like improved wastewater treatment and reducing algae blooms in Utah Lake, to meet TMDL targets. Stormwater runoff, Utah Lake, and permitted discharge from plants account for 4%, 9%, and 81%, respectively (Adams and Arens, 2013). Therefore, strategies would need to include improvements to more than stormwater runoff to reduce total phosphorus loadings significantly.

Green infrastructure in all seven small watersheds reduces stream temperature by about 17% or 0.23°C in the winter/spring Jordan River model, which shows the potential of green infrastructure to provide regulating services. According to the TMDL for the Jordan River, stream temperature is also a pollutant of concern for segments near Utah Lake that have a Class 3A beneficial use for cold water species (Adams and Arens, 2013). In the late summer Jordan River model, some of these reaches are above the standard maximum of 20°C. However, green infrastructure impacts are simulated downstream of these reaches in violation. Future research should contribute how stream temperature is affected by green infrastructure in these parts of the upper Jordan. This, in turn, would provide more insight to the potential of green infrastructure to provide the ecosystem service of climate regulation. Additionally, streamflow is predicted to reduce

about 9% (0.22 cms in the winter/spring model and 0.4 cms in the late summer model) when green infrastructure is implemented in the seven canyon creek watersheds. In the spring, this reduction is a benefit as a flood mitigation method and way to reduce peak flow during storms. In the summer, this reduction could impact downstream users and the Great Salt Lake ecosystem. However, there are limitations in the modeling of streamflow impacts that are discussed in the next section. The findings overall highlight the potential of green infrastructure to deliver ecosystem services of water quality improvement and peak flow reduction, with room for future work to continue examining climate regulation and returning flows from green infrastructure.

Limitations

This modeling did not include surface-groundwater exchange from green infrastructure. This is of particular importance in this study system since it has complex surface-groundwater interactions (Gabor et al., 2017). QUAL2Kw is a surface water model, but with detailed understanding of green infrastructure effects on groundwater and subsurface exchange, hyporheic exchange and interactions between surface and groundwater could be better represented. This would provide better estimates for changes in stream temperature from green infrastructure. Additionally, flow and water quality data was not available for Red Butte Creek after it flows underground. The sensitivity analysis in this study accounts for the uncertainty of streamflow after it flows underground. However, water quality changes from stormwater contributions when the creek is underground may be more than the assumed conditions are at 1300E.

The models used in this study were calibrated for two modeling periods for 2-days each, meaning the models are biased to a short amount of time for specific time

periods. As the modeled rainfall events were the median size in the study period, the results may not represent what changes from green infrastructure implementation for larger or smaller rainfall events. For the Jordan River alternatives with the seven canyon creek watersheds, estimates for flow are biased to the small Red Butte Creek watershed with few gaining reaches, which may be more or less than the other watersheds.

Lastly, due to data limitations, the February Jordan River model and the May Red Butte Creek model were connected, and the August Jordan River model and June Red Butte Creek model were connected. While the environmental conditions in Jordan River modeling periods do not match the conditions of the Red Butte Creek models, the changes from Red Butte Creek and the other tributaries modeled still provide a general illustration of the impacts of implementing green infrastructure in these areas.

Conclusion

In addition to the ecosystem services evaluated in this study, harvesting stormwater via green infrastructure for additional water supply has been proposed (Dile et al., 2016). More than one-sixth of the world's population lives in snow-dominated regions and depends on snowpack for water storage and supply (Barnett et al., 2005). This population is at risk due to reduced snowpack reliability, with climate warming causing more rain and changes in runoff timing due to restructuring of the urban environment (Barnett et al., 2005; Goharian et al., 2015; Hale et al., 2015). Managed aquifer recharge is a promising alternative to water stored in snowpack (Kirk et al., 2020; Megdal and Dillon, 2015) and sometimes also benefits fisheries by increasing streamflow and decreasing stream temperatures (Kirk et al., 2020). Stormwater harvesting through green infrastructure may enable groundwater aquifer recharge by capturing stormwater

runoff and facilitating infiltration.

My research quantifies benefits to river systems and their ecosystem services from implementing green infrastructure at the watershed-scale. Furthermore, the modeling in this chapter is shared with water managers and stakeholders in the Salt Lake Valley, who have been presented updates throughout the model development process. As a whole, this work and engagement with stakeholders is an example of informing decision-making with tools for multiple ecosystem services and sustainable stormwater management.

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CHAPTER 4

THE IMPACTS OF WILDFIRE CHARACTERISTICS AND EMPLOYMENT ON THE ADAPTIVE MANAGEMENT STRATEGIES IN THE INTERMOUNTAIN WEST[2]

Abstract

Widespread development and shifts from rural to urban areas within the Wildland-Urban Interface (WUI) has increased fire risks to local populations, as well as introduced complex and long-term costs and benefits to communities. We use an interdisciplinary approach to investigate how trends in fire characteristics influence adaptive management and economies in the Intermountain Western US (IMW). Specifically, we analyze area burned and fire frequency in the IMW over time, how fires in urban or rural settings influence local economies, and whether fire trends and economic impacts influence managers' perspectives and adaptive decision-making. Our analyses showed some increasing fire trends at multiple levels. Using a non-parametric event study model, we evaluated the effects of fire events in rural and urban areas on county-level private industry employment, finding short- and long-term positive effects of fire on employment at several scales and some short-term negative effects for specific sectors. Through interviewing 20 fire managers, we found that most recognize increasing fire trends and that there are both positive and negative economic effects of fire. We also established that many of the participants are implementing adaptive fire management strategies, and we identified key challenges to mitigating increasing fire risk in the IMW.

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Introduction

Wildfires pose an increasing threat to communities and built infrastructure throughout the Western United States. Over the last four decades in the Western U.S., the total annual area burned has increased considerably with wildfires occurring at higher frequency [1, 2]. Since the mid-1980s, warmer temperatures and increased aridity have increased the fire season by ca. 78 days in this region [1, 3]. Previous research on broad regional fire trends has primarily focused on the entire Western U.S. However, the Intermountain West (IMW) – defined in this paper as consisting of Arizona, Colorado, Idaho, Montana, Nevada, New Mexico, Utah, and Wyoming – differs from the coastal parts of California, Oregon, and Washington in that the IMW states overall are largely characterized by relatively dry conditions and arid vegetation communities that make it especially vulnerable to large, high-severity fires [4–7]. This susceptibility to fire is expected to increase under warmer and more arid future climate alternatives [8]. While extensive work on fire has been conducted within this region [2, 4], a better interdisciplinary understanding of fire trends at multiple scales within this expansive, ecologically-distinct portion of the West is needed if we are to adapt human behavior for more effective fire management in the face of a changing climate.

In addition to climatic factors driving increases in wildfire, widespread development along the wildland-urban interface (WUI) – the transition zone where housing meets or is intermixed with undeveloped vegetated areas – has increased populations and values at risk [9–12]. Population in the Western U.S. has grown rapidly in recent decades [13], with substantial development and housing growth concentrated in the WUI [11, 12, 14]. With greater expansion into the WUI and increased fire frequency,

more people are exposed to property loss, especially in high density urban regions. Research also shows that closer proximity to the WUI leads to higher suppression costs [15, 16]. However, the distribution of wildfire risks and the capacity to mitigate them varies between urban and rural communities [17, 18]. Rural communities, which are more prevalent in the IMW, may be differentially affected by wildfire due to fundamental differences in socioeconomic characteristics, including a greater dependence on natural resource and recreation-based industries [17, 19, 20]. Furthermore, rural communities have limited financial resources compared to urban areas [17], although residents have been more willing to participate in suppression tactics to protect their livelihoods [20, 21].

While wildfire can physically threaten urban and rural communities, it can also have immediate and long-term consequences for local economies. The majority of short-term economic impacts of wildfire tend to be negative, such as the costs associated with firefighting, property damage, and loss of timber resources, in addition to the evacuation of local residents, impaired water and air quality, and loss of tourism, business, and recreation revenue [22]. In the long-term, wildfire may increase economic volatility or lead to unstable economic growth in the year following a fire [23]. However, wildfire may also have positive impacts in some employment sectors from increased construction of infrastructure and rebuilding of homes, restoration of forest and aquatic ecosystems, and greater opportunities for resource extraction, like salvage logging [24]. These economic costs of fire are expected to increase with changing climate conditions and greater development in wildland areas. While studies have investigated a variety of economic impacts of fire, there is still a need for a greater understanding of how

managers utilize information on these impacts to make decisions and fire mitigation policy [25]. As increased risk of fire exacerbates socioeconomic effects on communities, it is critical to understand how wildfire impacts manager perspectives and adaptive management strategies to better mitigate those risks in an uncertain future [26].

With greater development in the more fire-prone wildland and WUI areas, fire managers have been tasked with greater responsibility for the protection of private citizens in increasingly vulnerable areas. Various factors influence fire managers' decisions, including fire characteristics (e.g., fire size and frequency), expectations of affected communities and government officials, and federal fire management policy [27]. Challenges to these decisions include natural accumulation of biofuels over time, projected (if uncertain) increases in aridity in those accumulating fuels, conflicting management objectives by different resource agencies, social and political pressures to immediately suppress fire, and managing the short- and long-term cumulative impacts of fire [27–30]. Overall, the complex decision-making process for fire managers is not well understood [25]. Improving our understanding of the various influences, needs, and challenges for management decisions answers the need for increased integration of fire management into the decision-making and risk management literature [28, 31].

An interdisciplinary approach is needed to more fully understand the complex systems and consequences of wildfire in changing socio-demographic and resource management contexts [18, 32]. Responding to changes in the wildfire regime in an adaptive way requires managers to understand broader trends in wildfire characteristics over a variety of scales, understand the condition of the forest and fuels within their management domain, and also discern highly contextual information from affected

communities such as economic impacts and expectations of officials and community members. Integrating quantitative and qualitative data and analytical methods on physical and social aspects of fire advances understanding of wildfire trends and impacts.

We applied an interdisciplinary approach to investigate how recent trends in fire characteristics influence regional adaptive management in the rural and urban areas of the IMW, exploring three interrelated questions: 1) Are area burned and fire frequency increasing within the IMW?; 2) Do fires in urban or rural settings influence employment trends in local economies, and if so, how?; and 3) Do trends in fire characteristics and economic impacts of fire influence perspectives of managers and adaptive decision-making, and if so, how? We addressed these questions by quantifying fire characteristics and economic impacts and connecting them with qualitative interviews of fire managers from three regions within the IMW. Our study identifies key challenges to implementing adaptive fire and forest management strategies for both short- and long-term fire risk mitigation (Figure 4-1).

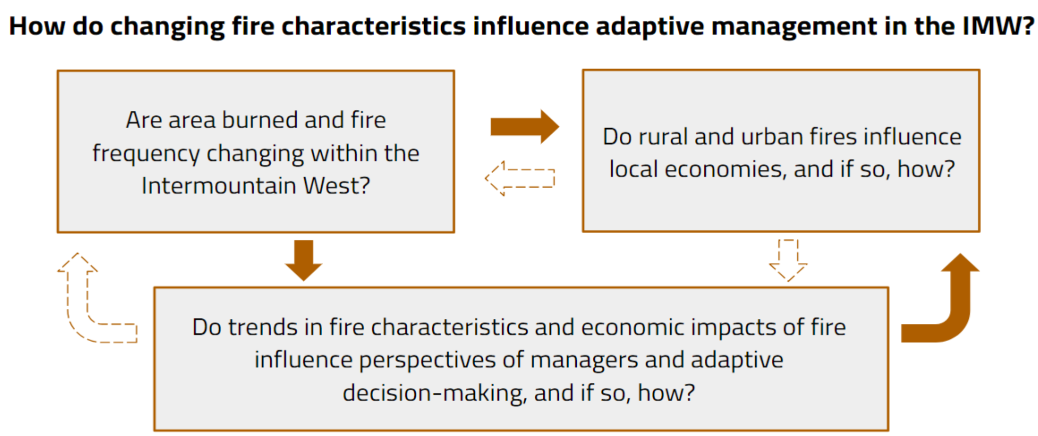


Figure 4-1. We address the overarching research question (top in bold) through investigating the sub-questions in the three boxes. The solid arrows show the connections that this interdisciplinary study addresses and are further discussed later in the paper. We acknowledge that other feedbacks exist between these questions (dashed arrows), such as managers' decisions and economies impacting fire trends.

Materials and Methods

We evaluated area burned and fire frequency for large fires across all eight IMW states. Using the 2011 National Land Cover Database and boundaries from the U.S. Census Bureau, we first quantified the amount of “burnable area” of each county (n = 281) within each state as the sum of all land cover types excluding open water, salt flats, and barren land (www.mrlc.gov) [33, 34]. We downloaded spatial data depicting the perimeters of individual fires greater than ~400 ha that burned within the region over a 32-year period (1984-2015) from the Monitoring Trends in Fire Severity (MTBS) database (www.mtbs.gov) [35]. We obtained spatial data that delineates the WUI based on housing density and wildland vegetation cover at the census block scale from the SILVIS Lab (http://silvis.forest.wisc.edu/maps/wui) [9]. Fires that occurred within 2.4 km [14, 36] of areas defined as "high housing density" (> 741.3 housing units km^{-2}) were classified as “urban fires”, while those that occurred outside of the buffer were designated as “rural fires” (Figure 4-2). In other words, “urban fires” refer to high-density WUI fires, and “rural fires” refer to low-density WUI fires. The buffer we implemented is intended to represent the distance at which urban structures are likely to become a primary concern, which may influence the vigor or strategy employed by fire suppression efforts [36].

To assess trends in area burned and fire frequency over the 32-year period at regional, state, and county levels, we calculated linear regressions in the R statistical computing environment [37]. Linear regression was used as the most conservative

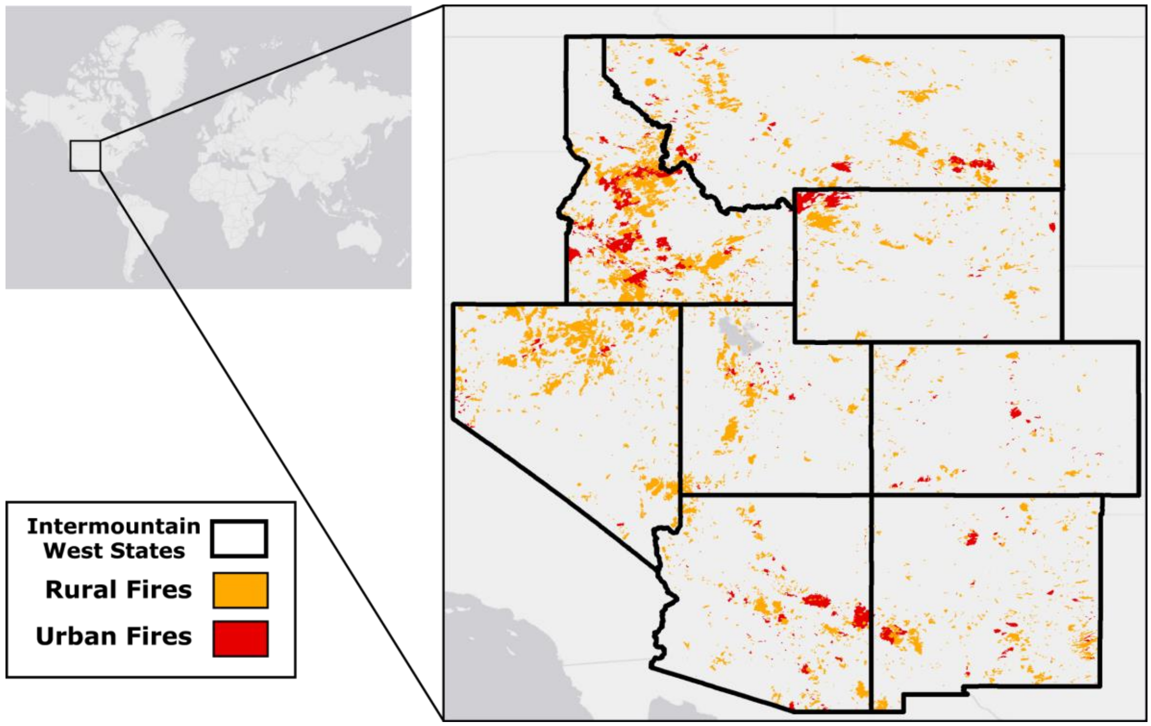


Figure 4-2. Fires over ~400 ha over a 32-year period (1984-2015), broadly classified as either "urban" (< 2.4 km from high-density census-blocks) or "rural".

approach to finding increasing or decreasing trends in the fire data shown in Appendix B (Figure 7-11 and Figure 7-12). Researchers have compared various approaches when modeling big data trends and have found linear fit to be appropriate for general overall trends [38]. For analyses of area burned, we summed the burned areas within each spatial unit (region, state, or county) by year and then normalized these values by dividing by burnable area within that unit, assessing trends in the percentage of each unit burned. For regional and state-level trends in fire frequency, we based annual fire counts on the number of fire perimeter centroids (i.e. centers) falling within each state to avoid double-counting fires that crossed state lines. For county-level frequency trends, fire counts were represented by the total number of fire perimeters intersecting each county boundary. We tested for the significance of linear trends separately for rural and urban fires at both the

regional and state-level, for both area burned and fire frequency.

To focus a portion of our economic analysis and our qualitative interviews with managers in areas that have experienced increasing trends in burned area and/or fire frequency, we identified focal counties by considering the steepness of the linear regression slopes for area burned and fire frequency in each county. Focusing on the top 5% of all regression slopes for all counties and excluding counties with increasing trends driven by outliers using a visual test, we identified 14 counties (Figure 4-3). We refer to these 14 counties as the "Increasing Focal Counties" throughout the rest of this paper. For more context on these "Increasing Focal Counties", six counties had increasing trends for burned area and twelve had increasing trends for fire frequency. This equated to a linear trend line slope greater than 7% for counties identified as our Increasing Focal Counties.

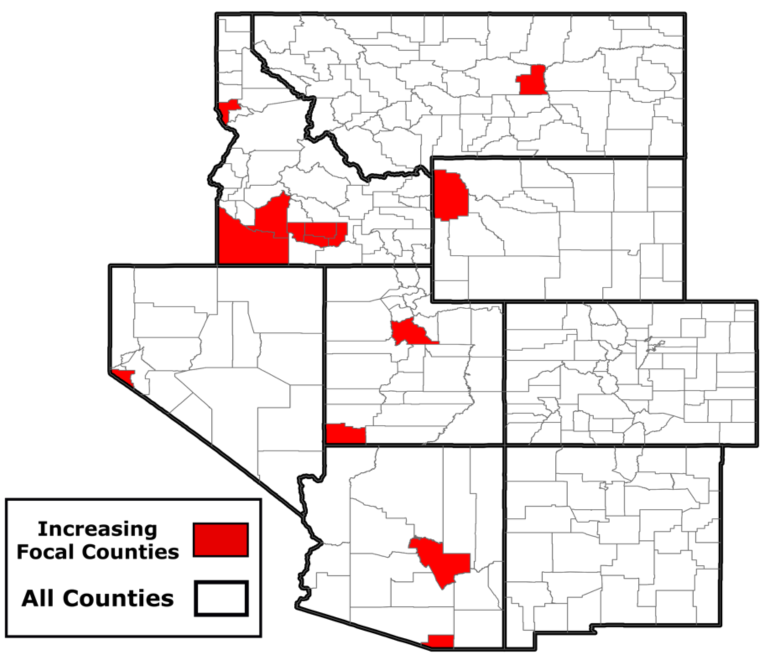


Figure 4-3. Increasing Focal Counties (Arizona [n=2], Idaho [n=7], Montana [n=1], Nevada [n=1], Utah [n=2], and Wyoming [n=1]) have experienced increasing trends for area burned, fire frequency, or both from 1984-2015. When ranking the 281 counties' regression slopes from highest to lowest, the Increasing Focal Counties are in the top 5 percent of slopes.

We estimated the impacts of urban and rural wildfires on local economies by analyzing changes in the employment rate in affected counties after each wildfire event. Our economic analysis looks at employment and fire data from 2001-2015, due to the employment data only being available from these years. We utilized monthly data on local employment rates from the Bureau of Labor Statistics (BLS) [39], retrieved online using the R package 'blsAPI' (https://CRAN.R-project.org/package=blsAPI). We then analyzed employment in relation to MTBS data on fire ignition date, fire size and location, and to our rural and urban fire classifications. We focused on five BLS employment datasets broken into three hierarchical tiers of employment specificity that range from broad to more specific sectors. The broadest category included (I) Total Employment for all IMW states (n=281 counties). The BLS divided Total Employment into two sub-categories: (1) Goods Producing, and (2) Service Providing sectors. Within each of the (1) Goods Producing and (2) Service Providing sub-categories, we further evaluated the (1a) Natural Resource and Mining, and (2a) Leisure and Hospitality sub-sectors, respectively. Each category contains monthly employment data from 2001-2015 at the county level (for a sub-sector employment example, see Figure 4-4). Graphs of employment data with the fire data used in our economic analyses can be found in Appendix B (Figure 7-6 – Figure 7-10).

We acknowledge that wildfires can have a wide range of economic impacts, including permanent loss of property or infrastructure, temporary loss of use or degradation, impacts on water, soil and forest resources, positive and negative impacts on terrestrial and aquatic wildlife, as well as costs of fire suppression and post-fire restoration. While data were not available to quantify those factors at the scale of our

analysis, we suggest that future efforts seek to compile or estimate such data for a more comprehensive analysis of economic impacts of wildfire.

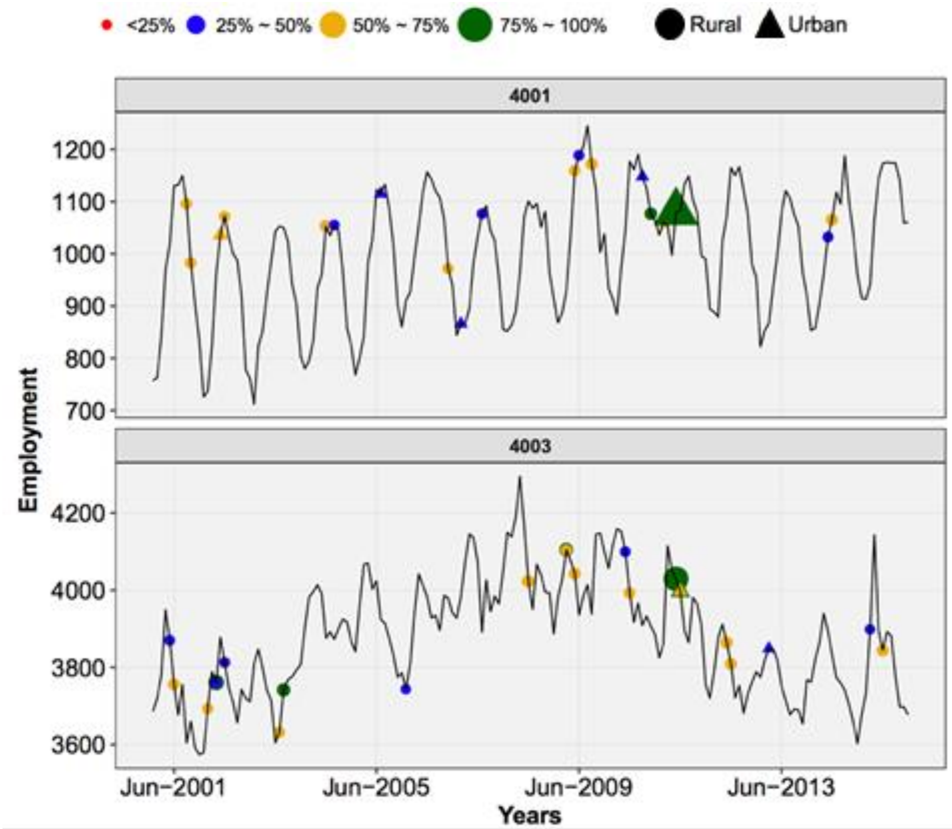


Figure 4-4. Example from two Arizona counties (Apache County - FIPS 4001; Cochise County - FIPS 4003) showing employment trends for the Leisure and Hospitality sector (2001-2015). Triangles represent urban fires, while dots represent rural fires. Different sizes of dots or triangles represent differing fire size. Fires were sorted according to size. Green dots/triangles represent the upper 25th percentile of fires, followed by the 50th-75th percentile in blue, and lower 25th percentile in red.

A central innovation of our study is the development of a new data set linking labor statistics data with MTBS fire data and the WUI classification. Nielsen-Pincus et al. (2013) studied the different impacts of urban and rural wildfire on local economies using the United States Department of Agriculture (USDA) Economic Research Service county typology to identify the rural and urban counties [23]. However, the majority of IMW

fires from the MTBS database did not cover the entire county and often crossed county and/or state lines. This creates false classifications in cases where fires occur in the urban parts of counties labeled 'rural' and vice versa. Therefore, the USDA county classifications did not have sufficient resolution for our purposes. Thus, we utilized our much higher resolution WUI urban and rural fire classification to obtain a finer spatial resolution of fire types, and used fire ignition date, location, and size from MTBS database to identify each wildfire that happened in IMW from 2001 to 2015. Our classified fire database is available as supplementary information associated with this paper.

We used an event study framework to analyze the different impacts of rural and urban fires on the employment of affected communities. Taking total employment rate for all industry as an example, the event study model gives us the change in employment rate within a county after a wildfire event,

$$log\ E_{c,t}^{total} = \sum_{j=-6}^{6} \gamma_j D_{s,t-j} + Acres + Trends + \mu_c + \mu_s + \delta_m + \varepsilon_{c,t}$$

where $logE_{c,t}^{total}$is the dependent variable, representing the percent changes in total employment rate for county c at time t. The variable $\gamma_j D_{s,t-j}$ is the fire indicator term, equal to 1 if the county is reported to have experienced wildfire in month t or 0 if not, according to the MTBS dataset. The month of wildfire ignition corresponds to (j=0). We normalized the effect in the month before the fire (j=1) to zero. $Acres$ represents the area burned (acres) in each event, to address how the size of fires can affect the local labor market. $Trends$ represents the overall linear trend of the regional logged total employment, to help account for broader economic trends of the region that may impact

employment. County fixed effects, represented by μ_c, standardize the comparison by only comparing within the same county. Variable μ_s represents the year fixed effects, thus we are only comparing impacts within the same year. Variable δ_m is the month fixed effects, while $\varepsilon_{c,t}$ shows the error term. Employment numbers can vary due to various factors, including differences in industries between counties, economic trends during different years, and changes across employment across months and seasons. These county, year, and month fixed effects help control for these changes in employment across different counties, across different years, and across different months of the year.

The model assumes that the occurrence of a fire is a random event, conditional to fire location and monthly time of year, and is uncorrelated with unknown confounding variables. We chose a 6-month event window to observe the impact of fire over time to be consistent with the seasonal trend of the BLS and fire data (Figure 4-4), both of which occur on a 6-month interval. Previous research has found longer-term lagged effects to be important when studying labor markets after fire [40, 41]. Therefore, we ran our model with a 12-month event window as well, which are also discussed briefly in the results section below. We ran the model for the five different employment sectors, defined above, and four regressions: All Fires (including all rural and urban fires within all counties), Rural Fires (including rural fires within all counties), Urban Fires (including all urban fires within all counties), and Increasing Focal Counties (rural and urban fires within the 14 counties that were classified above as experiencing increasing fire trends).

From our 14 Increasing Focal Counties (Figure 4-3), we focused our interviews in three geographic regions with clustered counties: two in Arizona, two in Utah, and six counties clustered in southwestern Idaho. We used the three regions as focused case

studies that helped qualitatively illustrate fire manager challenges. We recognize that these findings are not necessarily representative of the entire IMW region, but offer in-depth insight into regional perspectives. We used criterion and snowball sampling to conduct key informant interviews in March and April of 2018 (Utah State University Institutional Review Board Exempt Protocol #9130). We took a qualitative approach to collecting thematic interview data. While we had a small sample size of total interviews, others have utilized a similar thematic analysis [42] that identified social characteristics at the community level. Thematic analysis is an effective coding strategy that identifies common elements among participants around a specific topic and summarizes coded statements into broader themes [43].

To identify potential participants, we contacted agencies whose fire management jurisdictions were within or overlapping the specified counties in Arizona, Idaho, and Utah and sought participants whose job responsibilities included managing wildland fire through response, planning, mitigation, and prevention. To increase our sample pool, we asked potential participants for references of other key informants in their area. Using these techniques, we conducted 20 semi-structured interviews of managers from different state, tribal, and federal agencies. We primarily interviewed District Rangers, Fire Management Officers, and Fuels Specialists, all with a wide array of work history and experience. Interviews lasted between 16 and 86 min (mean = 39 min). Nineteen interviews were audio recorded with consent of the participant. One participant opted to have notes taken instead of an audio recording. This interview was fully transcribed from the notes within 24 hours. All audio recorded interviews were transcribed and then checked for accuracy by the interviewer.

While the interviews were structured in that each participant was asked the same set of questions in the same order, they were conducted in a manner to encourage free expression and explanation of participants' perspectives on: 1) local fire history and fire trends, 2) economic effects of wildfire, 3) influences on their local fire management and adaptation practices, and 4) challenges to wildfire risk mitigation (for the full interview protocol, see Table 4-1). Interviewers avoided prompting with cues to prevent priming participants responses. A thematic analysis approach was implemented, emphasizing semantic coding of explicit words used by participants to answer each question [43, 44]. Interview content was analyzed for emergent themes by the following four-step process to ensure reliable interpretations: 1) interviewers read through corresponding transcripts for accuracy; 2) interviewers read assigned transcripts and summarized the content for each interview according to key research questions; 3) a second interviewer read the transcripts and corresponding summaries to check for accuracy; and 4) interviewers and transcribers reviewed and coded summaries for major themes together while referring back to original transcripts as needed to resolve coding questions or disagreements. By this process, all transcripts were analyzed qualitatively for major themes by at least two people to increase the reliability of interpretations. During coding, the number of participants who mentioned different topics were noted for reporting major themes and corresponding responses. Managers' responses were also analyzed for possible geographic patterns as part of the thematic analysis. While participants were selected to collectively represent fire manager perspectives within the three focus areas in Idaho, Utah, and Arizona, we do not suggest they are necessarily representative of the larger Intermountain West region as a whole.

Table 4-1. Interview questions for participants regarding their perspectives on what influences their management practices and decisions.

Opening & Background Questions

- How long have you been working for _________ in a management position?
- What is the scope of your position?
- How does your work relate to fire management?
- In your opinion, has the frequency of wildfires or area burned changed in your area? If yes, how so?
- Has wildfire influenced economies in your area? If so, how?

Influences and Challenges

- Do economic impacts of fire influence your management decisions? If so, how?
- Have past fires or changes in fires over time affected your current management policies and decisions? If so, how?
- What challenges do you face in order to effectively mitigate wildland fire risk?

Community and Institutional Expectations

- What does the local community expect from your fire management decisions?
- What do government officials expect from your fire management decisions?

Local Policy Influence

- Do you have a current official fire management plan? (e.g. CWPP, CPAW) [Probe for description]
- Is this plan implemented into your routine management practices? If so, how?

Decision-Making

- Has any change in fire frequency or burned area influenced your management decisions and adaptive practices? If so, how? If not, why not?
- Do you think any future changes or events might lead to changes in fire management and policy for [your agency]? If so, what kind of changes or events might have more of an impact on fire management practices or policies?
- Do you manage fires in rural versus urban areas differently? If so, how? Would any change in fire frequency or burned area influence how you manage fires in rural versus urban areas? If so, how?
- Do economic effects of fire influence how you manage rural versus urban areas? If so, how?
- Were there any particular fires that changed your approach to or thinking about fire management?

Results

Changes in area burned and fire frequency within the IMW

Our analysis of MTBS historical fire data shows that fire characteristics have changed heterogeneously throughout the IMW. From 1984 to 2015, there were 5,569 large wildfires in the IMW, 515 of which we classified as urban and 5,054 as rural. At the regional scale, there is a significant increase in area burned by rural fires ($p < 0.1$) (Table 4-2), while focusing at the state level shows important variations in trends associated with area burned and fire frequency and are often driven by significant burn events or fire-prone areas.

Table 4-2. Regression slopes for area burned and fire frequency in both rural and urban areas. Significance is denoted at the $p < 0.1$ (*) and at the $p < 0.05$ (**) values.

	Area Burned		**Fire Frequency**	
	Rural	*Urban*	*Rural*	*Urban*
IMW	0.007*	0.002	2.834*	0.377**
AZ	0.009**	0.005	0.783**	0.032
CO	0.004**	0.001	0.209	0.046
ID	0.019	0.013*	0.590	0.040
MT	0.005	0.003	0.882*	0.022
NM	0.007	0.006**	0.240	0.069
NV	0.008	0.000	0.040	0.012
UT	0.007	0.001	0.241	0.028
WY	-0.001	-0.024*	0.349	0.038

Fire frequency has also increased in both rural ($p < 0.1$) and urban fires ($p < 0.05$) (Table 4-2). Area burned increased significantly within 28/281 counties and fire frequency increased within 22/281 counties ($p < 0.05$). When we relaxed the p-value to $p < 0.10$, 44/281 counties increased in area burned, and 42/281 counties increased in fire frequency. At the state scale, Arizona and Colorado have significantly increased in burned area for rural fires ($p < 0.05$) (Table 4-2 and Figure 4-5).

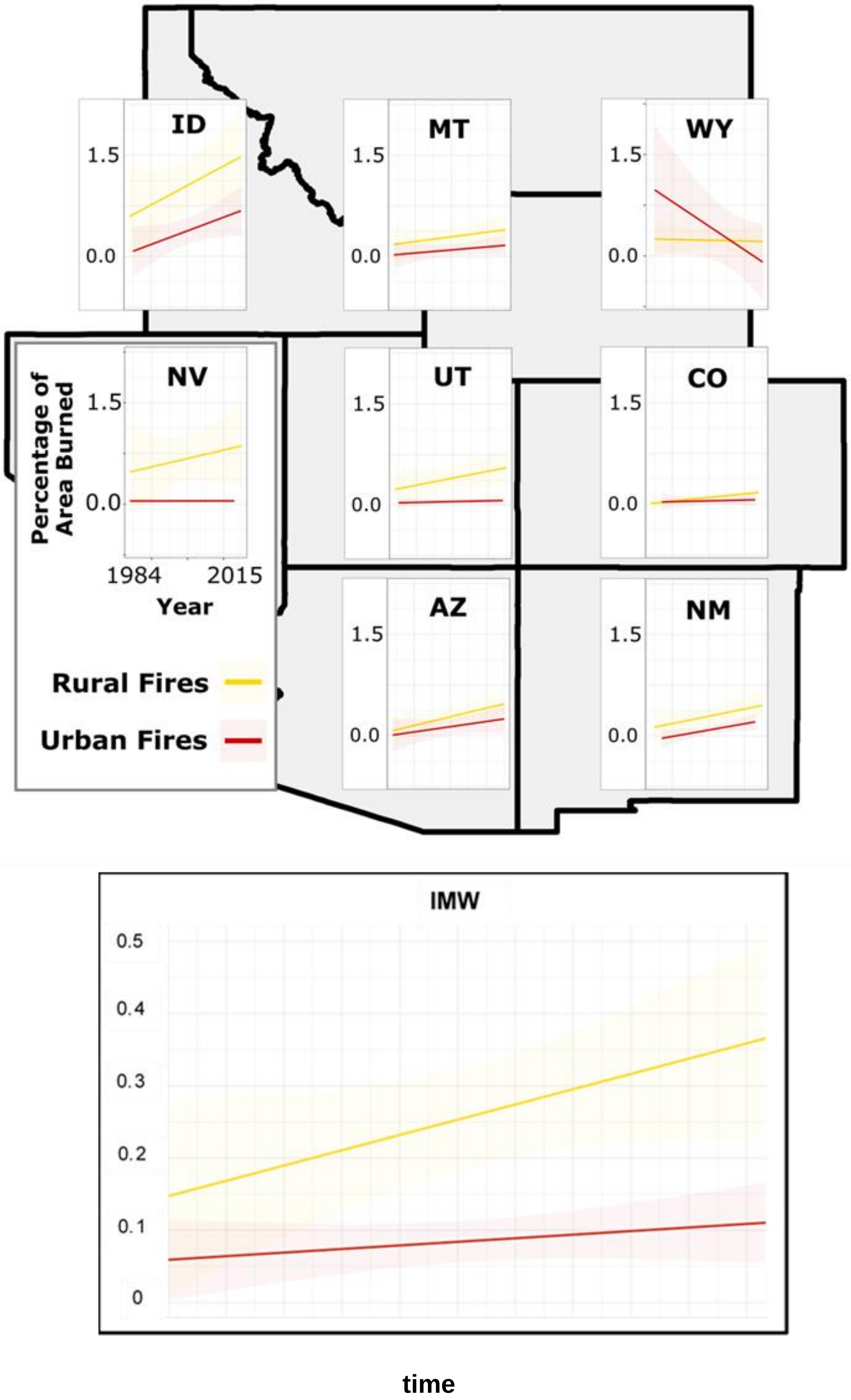
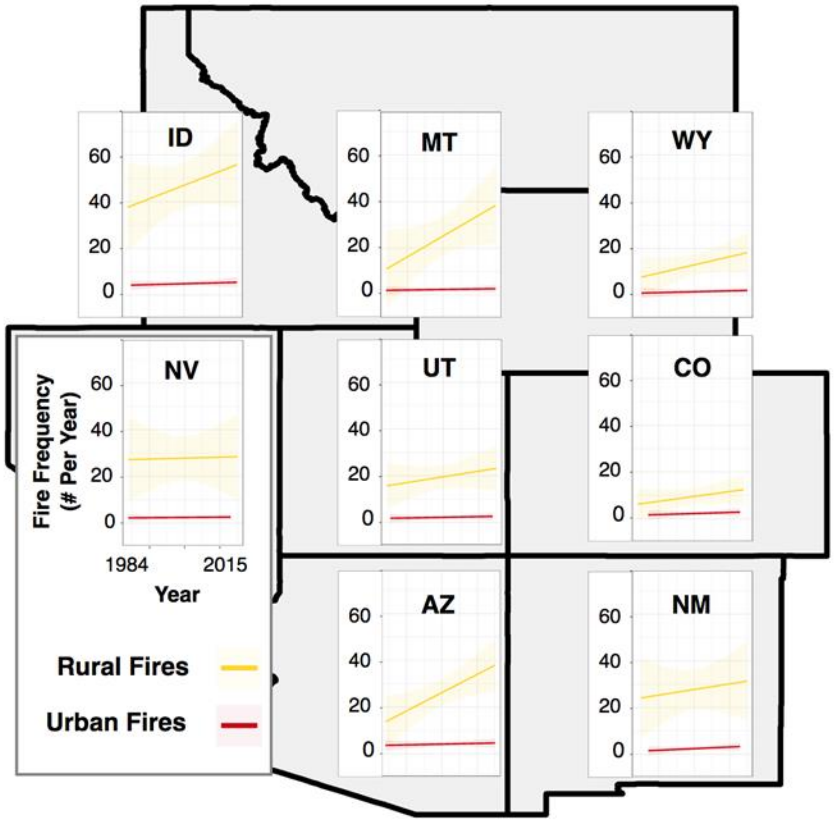


Figure 4-5. State-level linear trends in percentage of area burned for rural and urban fires between 1984 and 2015.



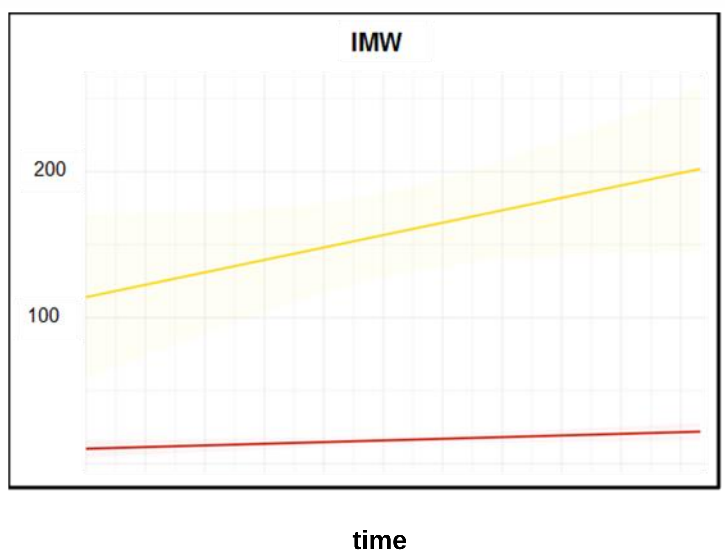


Figure 4-6. State-level linear trends in fire frequency for rural and urban fires between 1984 and 2015.

New Mexico ($p < 0.05$) and Idaho ($p < 0.1$) show significant increasing trends for area burned by urban fires (Table 4-2 and Figure 4-5). Wyoming depicts a significant decreasing trend ($p < 0.1$) in area burned by urban fires (Table 4-2 and Figure 4-5). In contrast, fire frequency has significantly increased for rural fires in Arizona ($p < 0.05$) and Montana ($p < 0.1$) (Table 4-2 and Figure 4-6). The apparent decreasing trend in area burned in Wyoming may be due to a historically large fire in Yellowstone National Park in 1988, which occurred at the beginning of our fire record and skewed the overall result. The same data, fit with the LOESS curve, are available in Appendix B (Figure 7-11 and Figure 7-12).

Economic Impacts of Fire

Fire can have a wide array of influences on local economies, including impacts on employment, property and infrastructure, air, water and soil quality, human health, costs associated with fire suppression or post-fire restoration, timber harvest, and tourism [22–24]. In this paper, we focus on employment as data are not available to quantify other impacts at the broad scale of our study. Employment data are readily available at a county scale in our time period and are evaluated monthly. As mentioned in the methods section above, we focused on a 6-month window after fires because our employment and fire data indicated a 6-month cycle (Figure 4-4). However, since other studies also find other significant effects after 6 months, we ran a 12-month model as well and included the results in Appendix B (Table 7-9 – Table 7-13). The results between the 6-month model and the 12-month model are similar, with most significant effects showing within the first 6 months after fires. There are a few positive significant effects at the end of the 12-month model, which indicates potential longer-lagged positive effects on employment.

Total Employment (I) results generally yielded positive effects of fires for all four sets of regressions: All Fires, Rural Fires, Urban Fires and Increasing Focal Counties (Table 4-3). Rural Fires and Urban Fires had differing impacts on affected county labor markets. Rural Fires had greater positive short-term impacts on affected county employment rates, and were all statistically significant at the 90% level. In contrast, Urban Fires did not have a statistically significant impact on employment at the county level. We observed statistically significant increases for 4 months after a fire event when considering both All Fires and Rural Fires. For Increasing Focal Counties that we identified as having increasing area burned and/or fire frequency, we found statistically significant positive impacts up to 2 months after fire occurrence (Table 4-3). Overall, the impacts were lower for total employment than the sub-sectors, which are discussed in depth below. However, the duration of these impacts was longer for total employment.

Table 4-3. Regression results for (I) Total Employment for the 6-month window post-fire for years 2001-2015 (*p<0.1; **p<0.05; ***p<0.01). The first column presents the results for All Fires within all 281 IMW counties (44,666 observations), the second column represents the results for Rural Fires (44,360 observations), the third column represents the results for Urban Fires (41,429 observations), and the last column represents the results for the 14 Increasing Focal Counties (2,274 observations). Effects of fires on employment are presented in percentages. The standard error for each regression is presented in parentheses.

	Dependent variable Effects of Fires on Employment (%)			
	All Fires	**Rural Fires**	**Urban Fires**	**Increasing Focal Counties**
Fire Happened	0.012***	0.013***	-0.001	0.026***
	(0.003)	(0.003)	(0.008)	(0.007)
1 Months After	0.005*	0.005**	-0.007	0.012*
	(0.003)	(0.003)	(0.007)	(0.006)
2 Months After	0.006**	0.006**	-0.001	0.012*
	(0.003)	(0.003)	(0.007)	(0.006)
3 Months After	0.005*	0.005*	0.0001	0.004
	(0.003)	(0.003)	(0.007)	(0.006)

Table 4-3. (cont.)

4 Months After	0.006**	0.005*	0.002	0.003
	(0.003)	(0.003)	(0.007)	(0.006)
5 Months After	0.002	0.002	0.006	-0.005
	(0.003)	(0.003)	(0.007)	(0.006)
6 Months After	0.002	0.001	0.007	-0.001
	(0.003)	(0.003)	(0.007)	(0.006)
Observations	44,666	44,360	41,429	2,274
R^2	0.996	0.996	0.996	0.996
Adjusted R^2	0.996	0.996	0.996	0.996
Residual Std. Error	0.115	0.115	0.116	0.101
	[df=44,345]	[df=44,039]	[df=41,109]	[df=2,220]

Fire Impacts on (1) Goods Producing & (2) Service Providing Sectors

We observed significant positive impacts for All Fires and Rural Fires for both (1) Goods Producing and (2) Service Providing sectors (Table 4-4), but the impact decreases with each subsequent month post-fire. When we compared impacts between the (1) Goods Producing and (2) Service Providing sectors, the positive impacts were greater in the Goods Producing sector immediately during and 1 month after a fire (Table 4-4 and Table 4-5). The Increasing Focal Counties with increasing fire trends had the greatest total positive impact for the (1) Goods Producing sector during the month of fire ignition. However, when these results were compared to the (I) Total Employment regression results, these positive impacts were observed for a shorter period, less than 1 month post-fire.

Table 4-4. Regression results of the (1) Goods Producing sector for the 6-month window post-fire for years 2001-2015 (*p<0.1; **p<0.05; ***p<0.01). Effects of fires on employment are presented in percentages. The standard errors are in parentheses.

	Dependent variable Effects of Fires on Employment (%)			
	All Fires	**Rural Fires**	**Urban Fires**	**Increasing Focal Counties**
Fire Happened	0.024***	0.025***	0.004	0.045***
	(0.005)	(0.005)	(0.015)	(0.012)

Table 4-4. (cont.)

1 Months After	0.009*	0.010*	-0.001	0.012
	(0.005)	(0.005)	(0.014)	(0.010)
2 Months After	0.007	0.008	0.001	0.017
	(0.005)	(0.005)	(0.014)	(0.011)
3 Months After	0.006	0.007	0.004	0.017
	(0.005)	(0.005)	(0.014)	(0.011)
4 Months After	0.009	0.010*	0.008	0.017
	(0.005)	(0.005)	(0.014)	(0.011)
5 Months After	0.006	0.007	0.010	0.005
	(0.005)	(0.005)	(0.014)	(0.011)
6 Months After	0.007	0.007	0.005	-0.003
	(0.005)	(0.005)	(0.014)	(0.010)
Observations	44,165	43,877	40,966	2,209
R^2	0.984	0.984	0.984	0.977
Adjusted R^2	0.984	0.984	0.984	0.976
Residual Std. Error	0.223	0.223	0.224	0.168
	[df=43,844]	[df=43,556]	[df=40,647]	[df=2,155]

Table 4-5. Regression results of then (2) Service Providing sector for the 6-month window post-fire for years 2001-2015 (*p<0.1; **p<0.05; ***p<0.01). Effects of fires on employment are presented in percentages. The standard error for each regression is presented in parentheses.

	Dependent variable Effects of Fires on Employment (%)			
	All Fires	**Rural Fires**	**Urban Fires**	**Increasing Focal Counties**
Fire Happened	0.006**	0.008***	-0.005	-0.002
	(0.003)	(0.003)	(0.008)	(0.007)
1 Months After	0.004*	0.005*	-0.009	0.008
	(0.003)	(0.003)	(0.007)	(0.006)
2 Months After	0.004	0.005*	-0.006	0.004
	(0.003)	(0.003)	(0.007)	(0.006)
3 Months After	0.003	0.004	-0.004	-0.002
	(0.003)	(0.003)	(0.007)	(0.006)
4 Months After	0.003	0.002	-0.002	-0.0004
	(0.003)	(0.003)	(0.007)	(0.006)
5 Months After	-0.0002	-0.00005	0.001	-0.008
	(0.003)	(0.003)	(0.007)	(0.006)
6 Months After	-0.0005	-0.001	0.004	-0.003
	(0.003)	(0.003)	(0.007)	(0.006)
Observations	44,177	43,873	40,955	2,248
R^2	0.996	0.996	0.996	0.997
Adjusted R^2	0.996	0.996	0.996	0.997
Residual Std. Error	0.116	0.115	0.117	0.095
	[df=43,856]	[df = 43,552]	[df=40,635]	[df=2,194]

Fire Impacts on (1a) Natural Resource and Mining & (2a) Leisure and Hospitality Sectors

Employment in the (1a) Natural Resource and Mining sector for All Fires, Rural Fires, and Increasing Focal Counties all had statistically significant positive labor impacts for the month when a fire was ignited (Table 4-6). The (2a) Leisure and Hospitality sector only had positive impacts 2 months after fire ignition for Rural Fires, but these impacts are not large, had a low significance level, and declined over time (Table 4-7). Negative impacts for employment in the (2a) Leisure and Hospitality sector were observed in Urban Fires 1 month after ignition. For Increasing Focal Counties, negative impacts were observed for (2a) Leisure and Hospitality sector during the month of fire ignition and 5 months post-fire.

Table 4-6. Regression results of the (1a) Good Producing: Natural Resource and Mining sector for the 6-month window post-fire for years 2001-2015 (*p<0.1; **p<0.05; ***p<0.01). Effects of fires on employment are presented in percentages. The standard error for each regression is presented in parentheses.

	Dependent variable Effects of Fires on Employment (%)			
	All Fires	**Rural Fires**	**Urban Fires**	**Increasing Focal Counties**
Fire Happened	0.013*	0.014**	-0.004	0.092***
	(0.007)	(0.007)	(0.021)	(0.018)
1 Months After	-0.001	0.002	-0.009	-0.001
	(0.007)	(0.007)	(0.020)	(0.016)
2 Months After	-0.001	0.001	-0.009	0.015
	(0.007)	(0.008)	(0.020)	(0.016)
3 Months After	-0.002	-0.002	-0.013	0.009
	(0.007)	(0.008)	(0.020)	(0.017)
4 Months After	0.005	0.004	-0.022	0.019
	(0.007)	(0.008)	(0.020)	(0.017)
5 Months After	0.0003	0.001	-0.020	0.005
	(0.007)	(0.008)	(0.020)	(0.017)
6 Months After	-0.006	-0.004	-0.027	-0.028*
	(0.007)	(0.007)	(0.020)	(0.016)

Table 4-6. (cont.)

Observations	39,406	39,112	36,346	2,181
R^2	0.953	0.954	0.953	0.949
Adjusted R^2	0.952	0.953	0.953	0.947
Residual Std. Error	0.306	0.304	0.305	0.254
	[df=39,094]	[df=38,800]	[df=36,035]	[df=2,128]

Table 4-7. Regression results of the (2a) Service Providing: Leisure and Hospitality sector for the 6-month window post-fire for years 2001-2015 (*p<0.1; **p<0.05; ***p<0.01). Effects of fires on employment are presented in percentages. The standard error for each regression is presented in parentheses.

	Dependent variable Effects of Fires on Employment (%)			
	All Fires	**Rural Fires**	**Urban Fires**	**Increasing Focal Counties**
Fire Happened	0.001	0.002	-0.017	-0.014
	(0.004)	(0.004)	(0.013)	(0.010)
1 Months After	0.005	0.005	-0.032***	0.010
	(0.004)	(0.004)	(0.012)	(0.008)
2 Months After	0.007	0.009*	-0.018	-0.003
	(0.005)	(0.005)	(0.012)	(0.009)
3 Months After	0.003	0.004	-0.011	-0.001
	(0.005)	(0.005)	(0.012)	(0.009)
4 Months After	0.001	0.001	-0.010	-0.006
	(0.005)	(0.005)	(0.012)	(0.009)
5 Months After	-0.0001	-0.001	0.001	-0.016*
	(0.005)	(0.005)	(0.012)	(0.009)
6 Months After	-0.001	-0.002	0.015	-0.005
	(0.004)	(0.004)	(0.012)	(0.008)
Observations	43,967	43,699	40,772	2,242
R^2	0.989	0.989	0.989	0.994
Adjusted R^2	0.989	0.989	0.988	0.994
Residual Std. Error	0.195	0.194	0.195	0.136
	[df=43,647]	[df=43,379]	[df=40,453]	[df=2,188]

Qualitative Interview Results

Overall, 15 participants from the three areas chosen for further investigation recognized that area burned or fire frequency increased in their jurisdictions over the last 30 years. Within the positive responses, two managers said fire frequency is increasing, five managers said area burned is increasing, and eight managers said both are increasing. Four managers responded with “It Depends” and cited the nuances of time period and

specific area, which may span different jurisdictions and counties. When managers' responses were compared with the calculated fire trends for their respective counties, seven responses matched with trends we observed in the MTBS database and seven responses had a partial match (stating either increased frequency or burned area when we identified a trend for both). Only two participant responses mismatched observed trends, either citing opposite trends from our analysis or no stated observed changes in fire trends (despite being selected for interviews because of an increasing fire trend) when a significant trend is actually observed in the data. These mismatches may be due to differences in jurisdictional boundaries from our county-level unit analysis or the fact that MTBS data includes only fires larger than 400 ha.

In general, most managers (14 participants) in the focused study areas said that changes in area burned and fire frequency influence decisions and adaptive practices in their jurisdictions, while four responded with 'No' and two with 'It Depends'.

"Repeated large fires, in general, drives where to focus our mitigation and treatments as well as threatened communities."

Adaptive strategies mentioned in response to changing fire trends are summarized in Table 4-8. Many managers mentioned increased efforts to reduce fuels and treat the landscape.

"I think how we mitigate those fuels, where we do it and how we do it has changed quite a bit throughout the years. We're putting more emphasis on mitigation work to try to get ahead of that, so that we're not spending as much money and suppression to protect [values at risk]."

For decades, the predominant fire management paradigm in the U.S. prioritized

fire suppression, with a more recent shift to longer-term planning on an ecosystem scale [25, 45]. Fire managers also mentioned repeatedly that large fires have driven policies that encourage them to more creatively minimize the size and frequency of fires. Some mentioned the need to shift firefighting tactics, including the assumption that fires will grow larger sooner.

"The long history of fire suppression has affected the fire return interval on the landscape and built up fuel loads... There is an accelerated pace to try and treat more acres annually."

Managers who said fire trends did not influence their management decisions cited the limitations of overarching fire suppression protocols that superseded the ability to enact local adaptation strategies. Overall, 18 of the participants are implementing some sort of adaptation practice regardless of fire trends. These practices include prescribed burns, mechanical fuel treatments, habitat restoration, fuel treatment experimentation, interagency cooperation, and implementation of education and outreach programs. Managers emphasized the need for adaptation and mitigation work in order to control fuels, enhance suppression efforts, and restore habitat.

"We're trying to solve the fire problem by or at the landscape health level, not just by the fire itself but with restoration because of all the invasives like cheat grass, etc. Because if you restore the landscape, then our fire frequency would go down."

While the majority of participants recognized changes in recent fire history, not everyone explicitly attributed these observed trends in fire to climate change. This result may be limited by the fact that they were not asked directly about this relationship during the interview – interviewers did not ask managers specifically if climate change

influences fire frequency or area burned. Hence, opinions of climate change's influence on these trends is not known for all participants. Regardless, whether or not managers perceived increasing trends being caused by climate change, the efforts of most managers to implement adaptation practices is helpful for climate resiliency.

Table 4-8. Adaptation strategies described by managers when asked how changes in area burned or fire frequency influenced their management decisions and adaptive practices.

Fire Trend Impacts to Adaptive Management	# of Managers
Informs/adjust fuels mitigation and calculations	8
Adjust fire response tactics	4
Affects treatments on the landscape	3
Experience informs management	3
Repeated large fires drives policy and management	3
Proactive management due to larger, frequent fires	3
Assume fires go larger sooner	2
Protect restoration investment	2
Alters grazing strategies	1

When asked if wildfire influenced economies in their area, 17 of the managers said ‘Yes’ while three were unsure. Some managers recognized the short-term positive impact that fires had on local economies, including the boost in goods and services when fire management teams patronized businesses near the fire. The influx of money and resources necessary to support a vast number of fire employees for days, weeks, or even months at a time was noticeable, especially in smaller, more rural communities. However, participants more commonly cited the negative and often longer-term impacts that fire has on communities, including the effects of smoke on health and tourism, closures to recreation areas and grazing allotments, loss of structures and property, the evacuation of residents, and the halt of commerce and e-commerce and transportation with major road and highway shut-downs. While fire did increase the immediate

opportunities for activities like salvage logging after the fire subsides, more often the negative long-term economic impacts for industries, such as sustainable timber harvesting, outweighed the short-term benefits. Managers spoke primarily about localized economic effects, but our economic analysis shows that some of these effects can be generalized to a broader region, even as broadly as the entire IMW. These generalizations are discussed later in this article.

Most managers (16 participants) said that the economic impacts of fire influenced their management decisions, while four were unsure.

“As fire managers, [the economic impacts of fire] definitely does [influence decision-making]. And from the political aspect of it, the more you impact that economy, the more political pressure I think you're going to get to resolve that situation quicker."

Most managers claimed that they tried to minimize damages to life, property, and resources on the landscape as mandated by national policy. Managers that were unsure either could not elaborate or said it “depends on values at risk.”

In light of the growing rural-urban divide in the IMW, the majority of managers (14 participants) cited differences in how they managed rural versus urban fires. Urban areas received the highest fire-fighting priority. Fires in rural areas allowed for more flexibility in management strategies, but were overall more complex in their approach due to a greater number of partnered agencies and public community involvement. A Fire Management Officer interviewed said:

"[T]he difference between rural and urban definitely comes down to where the people are, the values at risk and what resources you have to work with. . . . and what makes it a higher priority is – it's a numbers game. More people, more structures, so it

gains more [investment of resources]."

Respondents who said they do not manage rural versus urban fires differently explained that the full suppression policy for their jurisdiction compels them to be aggressive in both settings, or that they base decisions on environmental factors or values at risk regardless of whether they occur in a rural or urban setting. Most managers spoke about the urgency and constraints of fighting fire according to mandated priorities of protecting life, property, and values at risk in populated areas and the WUI, while addressing the greater flexibility to allow fires to burn in rural areas.

When asked about the primary challenges to effectively mitigate wildfire risk, the top three categories participants mentioned were limited funding and resources, bureaucracy, and human behavior and education (Table 4-9). These three challenges were all sociological-based limitations, compared to the physically-based limitations, such as changing fuel loads and future climate, which ranked fourth and sixth most mentioned, respectively. Managers said that budget cuts, limited resources, and lack of personnel made it difficult to carry out mitigation projects or accomplish restoration goals. The U.S. Forest Service spends approximately 50% of its annual budget on fire suppression and estimates an increase to 67% of its annual budget (an increase to more than $1.8 billion) by 2025 [3]; however, the need for more funding to manage increasing fire on the landscape is stressing the already limited federal budgets. Bureaucratic challenges such as project delays, paperwork, conflicting conservation management goals, and pushback from constituents, created serious limitations when working with multiple agencies or stakeholders. Some managers call for change “where the policy that's being handed down and the budgets that are being handed down are coherent and they work together so that

[fire managers] can do the work that [they] need to be doing." Other participants said that educating and changing public perceptions about resource benefits from fires, and altering human behaviors, specifically reducing human ignitions, increasing awareness and "getting private land owners to accept the responsibility of the risk" while helping mitigate along the ever-growing urban growth boundary, were the greatest challenges for fire management.

"Communities are encroaching on the National Forest. There's a lot of responsibilities that the landowners and the private landowners, private property owners, there's a lot of responsibilities that they have to accept on fire because of the location of their homes…that's the biggest thing that I've seen in the last 30 years is the occurrence of, the broadening of the Wildland Urban Interface, linear miles of it. It's increasing and that adds complexity along with the fuels that you have, and the weather that you have, the topography that you have, and adding the Urban Interface and those structures, that adds a lot of complexity."

Furthermore, while fuels mitigation was mentioned less than these socio-political challenges to adaptation, it was the most mentioned strategy impacted by fire trends (Table 4-8). This suggests that while managers acknowledge adapting fuels work to observed fire trends is an ongoing effort, such proactive measures can be constrained by the social and political challenges they face.

Table 4-9. Main challenges to wildfire risk mitigation identified by managers, summarized by categories and listed by the number of manager responses.

Identified Challenges to Management	# of Managers
Limited funding/resources	15
Bureaucracy	13
Human behavior/education	11
Changing fuel loads	7
Federal policy and administration shifts	5
Future climate	4
Competing interests/priorities	4
Development/growth	2

Participants in the three different geographic regions had different responses for some of the top cited categories. The majority of managers in Idaho had different responses compared to those in Utah and Arizona when it came to bureaucracy (ID = 10 participants; UT = 0 participants; AZ = 1 participants) and shifts in federal administration and policy (ID = 4 participants; UT = 0 participants ; AZ = 0 participants). While noting that the National Environmental Policy Act (NEPA) process is necessary for bureaucratic consent of all involved agencies, several Idaho participants mentioned it is difficult to accomplish projects in a timely manner. They further mentioned the difficulty and complexity of managing fire while also managing critical habitat and breeding area for the endangered Greater Sage Grouse (*Centrocercus urophasianus*). The conflicting management priorities of NEPA, the Clean Air & Water Acts, and special threatened and endangered species regulations restrict the window and flexibility for managers to allow fires to burn on the landscape. It creates “a big, big task getting caught up on those acres” for treatment and mitigation. While managers in Idaho cited the greatest challenges with bureaucracy and shifts in federal administration and policy for their work, there may be geographic differences in the challenges that managers face elsewhere.

Discussion

We have three primary findings regarding fire and management strategies in the IMW. First, wildfire trends are increasing in area burned and fire frequency across the IMW at the regional scale, and for some counties and states. In the past 32 years, the IMW has experienced more frequent and larger rural fires, and more frequent urban fires (Table 4-2). However, this is not to say that all parts of the IMW are experiencing increasing fire trends. While we found significant trends at the regional level and for some states, there are clearly hotspots when looking at the county level. These hotspots are also not set over time, as counties that have not burned in our data time period may now have higher fuel loads. There are many potential reasons for increasing fire trends, including changing climate, changes in fire mitigation strategies, and changes in management priorities. Across the entire Western US, recent increases in wildfire are closely associated with increases in fuel aridity and is largely driven by anthropogenic climate change [46]. Our findings align with the argument that the predominantly dry IMW region is going to continue to be vulnerable due to high soil aridity [6, 7]. Increasing burned area could be further affected by shifts in management practices away from the immediate suppression of fire, particularly in rural areas. Alternative strategies include fuels reduction (e.g., prescribed fires, mechanical treatment) and use of fires for resource benefit (e.g., allowing fires to burn where values are not at risk).

Secondly, fires have had both positive and negative effects on employment rates at the county scale over the last 15 years. The timing and magnitude of these effects varied depending on economic sector. Generally, we observed short-term positive impacts of All Fires and Rural Fires across the IMW at the county level (See Table 4-3,

Columns 1 & 2: All Fire & Rural Fire) immediately during and after a fire. These trends become weaker over time, but do not become negative. Participants referred to this as the short-term boom and long-term bust to local businesses and livelihoods, which is consistent with other research findings [23]. While we did see mostly short-term effects within the first 6 months after a fire, our study provides evidence of both short-term and long-term lagged effects with a few significant effects close to a year post-fire. When separating into the employment subsectors, fire had immediate positive impacts on the (1) Goods Producing category. Fires can increase local investment through the construction of new buildings and the rebuilding of destroyed structures, roads and utility infrastructure [24]. These positive impacts are still present at the sub-sector level of (1a) Natural Resources and Mining. We are unable to fully account for this disconnect between immediate positive effects of fire and employment in the (1a) Natural Resources and Mining sub-sector. We expect that the full impacts of fires on this sector may be better quantified by more direct data, such as suppression costs, timber sale loss, and finer scale data, such as the census block level employment data. Unfortunately, such data were not available for this study. In the (2a) Leisure and Hospitality sector, there is a negative effect on employment during the month of the fire, which is consistent with previous studies [40]. Additionally, there are delayed positive impacts of all fires and rural fires across the IMW at the county level. The BLS defined the (2a) Leisure and Hospitality category as encompassing Arts, Entertainment & Recreation and Accommodation & Food Services, and these delayed positive impacts, especially in rural areas, could be driven by the return of tourism to an area after a 1-2 month period of official restrictions or visitation avoidance after a fire [22]. However, further analysis is

needed to make this case, such as evaluating number of visitors to recreation areas. It should also be noted that there are other subsectors that may experience changes due to fire. Other studies were able to include additional subsectors of employment, such as construction and transportation, and found significant effects [41]. While we were able to find significance for the natural resource and leisure subsectors, we were unable to test effects for additional subsectors because there were insufficient data available for enough counties in other subsectors.

Third, most fire managers in the three areas in Idaho, Utah and Arizona acknowledged changing fire trends in their regions and are utilizing adaptive management strategies to mitigate changing fire patterns. They recognized some form of economic impact of fires and that these economic effects influence their management decisions. While we listed the number of participants who mentioned different topics to discuss the results of the interviews, we would like to emphasize that the more qualitative insights from the respondents should be the focus when analyzing the interviews. This third component contributes to the limited literature on understanding the decision-making process of fire managers and policy-makers [25]. The majority of managers interviewed feel the greatest challenges to fire adaptation are human factors, such as budget limitations, bureaucratic inefficiencies, and human decision-making, rather than environmental factors, such as climate change and accumulation of excessive fuel loads (Table 4-9). These human-related challenges are consistent with some of the wildfire risk literature, which calls for more landowner engagement in mitigation and adaptation [47]. Through these interviews, we also found connections to our fire trend analyses. Implementation of new fire mitigation techniques and improved firefighting efficiency,

both of which are discussed in the interviews, may serve to counteract increases in area burned and/or fire frequency. For example, thinning, prescribed burning, and the creation of fire breaks have been implemented into many management plans to help reduce the size and severity of wildfires. There was some variance in the interviews, in terms of the adaptation strategies used by managers. This could be due to the differences in local context and the lack of larger-scale policies and alternatives for climate adaptation. While there was some variation, overall, there was general consensus in what influences managers' decisions and the challenges they face. These interviews provide in-depth insight into managers' perspectives in areas that have experienced increasing fire trends. However, they are limited in generalizability to the IMW. Future research on fire management, decision-making, and policy could contribute to the literature with studies with larger sample sizes across varying fire trend contexts.

The findings for the three sub-research questions of this study inform and support one another (Figure 4-1). Our study is the first to document a positive trend in area burned and fire frequency at multiple scales for the IMW region, and furthermore, to parse those trends into urban and rural settings, and explore the effects of those wildfire trends on local economies and adaptive management practices. Notably, we find that wildfire characteristics are increasing significantly but are spatially variable throughout the IMW. While fire managers in places experiencing increasing trends are generally aware of and adapting to those trends, many are experiencing limitations in adaptive capacity, which may become increasingly problematic in the predicted warmer and drier future in the IMW. Our qualitative interviews augmented our economic analysis as participants provided information regarding costs and risks for which quantitative

economic data do not exist, including impacts on recreation and tourism. At the same time, the positive economic benefits observed several months after fires in our economic analysis (Table 4-4) were also captured in our qualitative interviews with managers who mentioned that burned areas can be logged for salvage timber. The economic analyses for the Increasing Focal Counties are in line with what managers said in interviews as well. For these focal counties, we find much larger negative impacts for (2a) Leisure and Hospitality than the other counties, indicating that more frequent or larger fires subsequently decrease tourism and recreation activity.

This study has been conducted based on available secondary data on fires and employment and the primary interview data we collected. Each dimension of the research had limitations that should be acknowledged. The fire trend analysis based on the MTBS dataset is limited to fires over 400 ha, thus overlooking smaller fires, which may be important, especially in urban settings. Economic data on fire suppression costs are not publicly available across the IMW study area, thus precluding a more direct analysis of fire-related economic impacts. Furthermore, our economic analysis of employment impacts of fire is limited to the last 15 years. Time and resource constraints limited the number of interviews with fire managers that could be conducted as well as the number of counties or areas that could be selected for this part of the investigation. Collectively, these data limitations inhibit generalization of findings across the study area and time period. Nevertheless, the insights provided here suggest trends and impacts related to fire are worthy of further investigation.

Our findings demonstrate that fires have significant economic impacts on affected communities, and that changing fire trends and economic effects influence the decision-

making and planning of fire managers. The interdisciplinary nature of this research highlights the interconnectedness of the physical, economic, and social aspects of fire, and answers the call to utilize interdisciplinary approaches to address these complex social-environmental issues [48]. Our approach provides a novel and more holistic view of fire management that is often lacking. Lastly, our research contributes valuable insights into changing fire trends, the economic impacts of fire, and perspectives of fire managers in a rapidly changing landscape.

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CHAPTER 5

PREDICTING FISH MIGRATION IN ONE OF THE WORLD'S LARGEST INLAND FISHERIES WITH HISTORICAL RANDOM FOREST MODELING

Abstract

The Mekong Basin is home to more than 800 fish species, with at least 165 documented migratory species. Overfishing, hydropower dam construction and concurrent habitat loss puts migratory fishes at risk. In Cambodia's Tonle Sap River and Lake, fish migrate from Tonle Sap Lake to the Mekong River during the dry season. Potential drivers of fish migrations are not well understood, although discharge, water level variations, changes in water quality, rainfall, and the lunar cycle are possible migration triggers. This chapter uses historical random forest models to correlate potential predictors, including streamflow (timing, magnitude, and duration), water level, precipitation, and the lunar cycle, with catch weight of six migratory species of ecological, cultural, and economic importance in the Tonle Sap River. The models confirm moon cycle, water level, and timing of flows as top environmental migration cues. These findings highlight when to limit harvesting of some mud carps, such as after the new moon or in the few days after high flows. Due to the inability to fully predict the migration of the six species in decline, these results also confirm previous conservation management strategies such as the release of immature, non-optimal length fish and mega-spawners by fisheries and locals.

Introduction

Fish provide sustenance, livelihood, and cultural significance to approximately 65 million people in the Lower Mekong Basin [1]. The bagnet fishery, or ‘Dai’ fishery as it is locally known, in Cambodia’s Tonle Sap River yields approximately 12,000 tonnes of fish every year that migrate out of Tonle Sap Lake in the early dry season to spawn upstream in the Se San, Se Kong, and Sre Pok Rivers (the ‘3S’ system) (Figure 5-1) [2, 3].The Dai fishery has been in operation for more than 140 years, and relies on the unique hydrology of the Tonle Sap system [4]. During the dry season, from November to May, the Tonle Sap River flows from Tonle Sap Lake into the Mekong River, allowing the Dai fishery to harvest fish migrating from Tonle Sap Lake. The Dai closes in the wet season when the flow of the Tonle Sap River reverses, flowing from the Mekong River into Tonle Sap Lake. This flow reversal increases Tonle Sap Lake’s area from roughly 2,600 km^2 in the dry season to 15,000 km^2 in the wet season [5].

Recent and ongoing hydropower dam construction throughout the Mekong Basin is homogenizing flows, creating drier wet seasons and wetter dry seasons [6]. Narrowing the range of flows alters the system from its natural state that fish are adapted for, potentially threatening existence of these species. Significant hydrologic alterations from dams in the Upper Mekong have already been documented (*Ibid*.). Further development in the Lower Mekong Basin will continue to magnify these effects [7].

There are 11 hydropower dams proposed on the mainstem of the Mekong River [8]. For the Tonle Sap River and Lake, homogenization of flows from dams on the Mekong River threaten the ecosystem services (i.e., the provisioning of fish) that rely on the annual flood-pulse of this system. Dams on the mainstem of the Mekong River would

also block migration routes to spawning grounds, prevent the drifting of larvae to downstream habitat, and increase mortality during juvenile life stages of fish [9, 10].

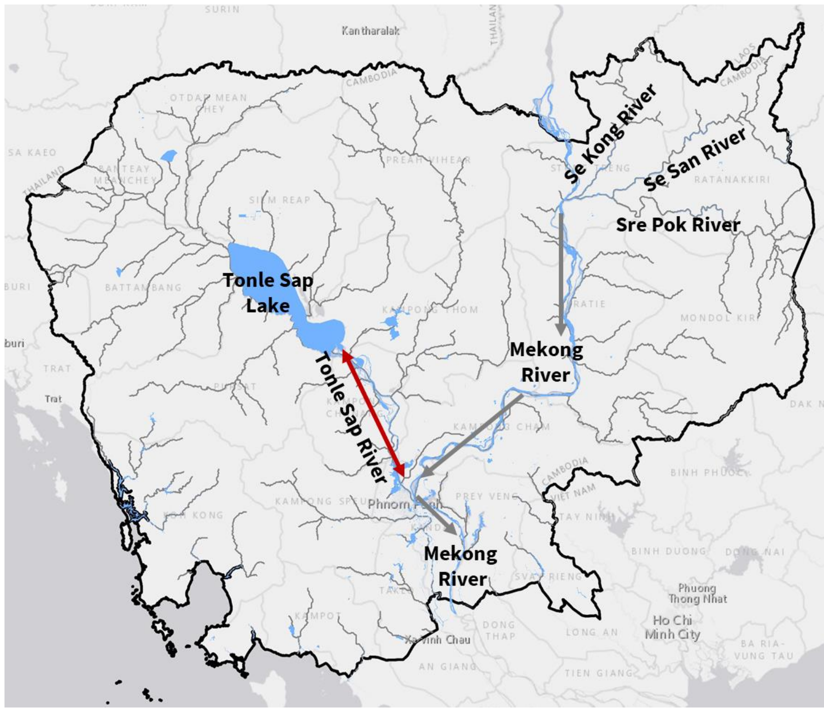

Figure 5-1. Tonle Sap River flows from Tonle Sap Lake into the Mekong River in the dry season. During flooding in the wet season, the Mekong River reverses Tonle Sap River flow into Tonle Sap Lake.

Heavy indiscriminate fishing pressure also threatens fish in the Tonle Sap system. Previous research found that the harvesting of 116 species in the Dai fishery has declined in the last 15 years [11]. While total catch remained stable over this time with lower trophic levels of fish replacing the larger, higher trophic species, there are concerns about the resiliency and sustainability of this system that is becoming dominated by smaller, fast-growing species [12]. Less diversity of fishes leads to unstable ecosystems that may not respond well to changing environmental conditions from climate change and water development (*Ibid*.).

Two families of fish (Cyprinidae and Pangasiidae) make up the largest proportion

of the total catch in the Dai fishery [11]. The six species analyzed in this study belong to those families and rely on environmental cues to initiate their migration out of Tonle Sap Lake to the Mekong (*Ibid*.). *Pangasianodon hypophthalmus* is a large river catfish, and *Cyclocheilichthys enoplos*, *Osteochilus melanopleurus*, and *Cirrhinus microlepis* are medium to large carp. *Henicorhynchus lobatus* is a keystone species, and it with *Labiobarbus lineatus*, are small species that supply abundant food.

Environmental cues that trigger fish to migrate have been studied and quantified in other systems, indicating that river discharge, lunar phase, and water temperature are important predictors of movement to spawning sites and habitats [13, 14]. In the Lower Mekong Basin, local agency fish biologists and fishermen have identified important spawning and rearing grounds and migration routes. Literature on migratory fish and migration routes in this system suggests discharge, water level variations, changes in water quality, rainfall, and the lunar cycle are possible migration triggers [2]. However, few studies quantify and systematically analyze the environmental conditions that cue migratory fish to move in the Tonle Sap River and Lake system. Identifying migration triggers will help guide fish conservation of species with significant cultural and economic value in this rapidly changing system.

The objective of this study is to identify the environmental conditions that cue six fish species to migrate from 20 potential predictors of fish migration. More specifically, historical random forest modeling will address the following research question: What hydrologic conditions cue fish to migrate for spawning in the Tonle Sap system of Cambodia? Using daily catch weight data from the Dai fishery on the Tonle Sap River, the random forest models identify hydrologic and environmental predictors important in

cueing ecologically and economically significant species to migrate from Tonle Sap Lake to the Mekong River and 3S basin in the dry season.

This chapter is a part of program for sustainable development in the Lower Mekong River Basin. The larger project involves a variety of experts, including fisheries biologists, ecologists, limnologists, hydrologists, science communicators, and local fisheries institutions in Cambodia. The focus of this study is to systematically understand the drivers that maintain ecosystem services (i.e., provisioning and cultural services) related to fisheries in the Tonle Sap system by integrating fisheries biology, ecology, and hydrology.

Methods

Historical Random Forest Modeling

Historical random forest modeling is used to quantify predictors of fish migration. Historical random forests are a relatively new form of random forest modeling and provide a systematic method of evaluating predictors of fish migration and previous values for those predictors of interest [15]. Random forests are an ensemble method of prediction, using many decision trees [16, 17]. By fitting many regression trees to a dataset and evaluating predictions from all the trees, random forests analyze the importance of several predictors on the response variable [18]. Random forests are less sensitive to collinearity than other regression methods, due to the averaging of many trees and randomization of variables selected at each split in the trees [18, 19]. Model prediction error is presented as out-of-bag mean squared errors (MSE), where predictions are tested on bootstrapped data, or random observations from the dataset. Although methods exist to select among collinear variables in random forest models, they have not

shown significant improvement in model performance and there is risk of not including enough variables, creating bias in out-of-bag errors [19].

Historical random forest models use longitudinal or time series data, which can be sampled at regular or irregular time intervals. Prediction is based on history of observations and time-varying predictor variables. Data is in the form of vectors:

$$z_{ij} = (y_{ij}, t_{ij}, x_{ij})$$

where z_{ij} is response variable, y_{ij} is the response variable for the i^{th} subject/year at the j^{th} observation time t_{ij}, and x_{ij} is the vector of predictors at time t_{ij}.

Random forest models were developed for each of the six species evaluated in this study using R package 'htree' [15]. Daily average catch weight of the species during the dry season migration from Tonle Sap Lake to the Mekong River is the response variable and hydrologic or environmental conditions were predictor variables.

Six species are included in this study. These species represent different trophic levels of the Tonle Sap system and have been declining in catch weight at the Dai fishery over the last 15 years [11] (Table 5-1). *Pangasianodon hypophthalmus* is a large river catfish, commonly with a standard length less than 80 centimeters [20]. *Cyclocheilichthys enoplos* and *Cirrhinus microlepis* are large mud carp typically with a standard length less than 60 cm in length (*Ibid*.). *Osteochilus melanopleurus* is a medium-sized mud carp that is commonly less than 35 cm in standard length (*Ibid*.). *Henicorhynchus lobatus* and *Labiobarbus lineatus* are small mud carps typically with a standard length less than 15 cm and 15.5 cm, respectively [21, 22].

Table 5-1. Maximum total length (cm) of each species in the study dataset

species	max total length (cm)
Pangasianodon hypophthalmus	158.6
Cyclocheilichthys enoplos	90.3
Cirrhinus microlepis	79.3
Osteochilus melanopleurus	73.2
Henicorhynchus lobatus	18.3
Labiobarbus lineatus	15.5

Variable importance tables rank the importance of each predictor variable included in the model. After a full historical random forest with all the predictors, each predictor is marginalized out of the model one at a time. The full model prediction error with all the predictors is then compared to the marginalized prediction errors without each predictor, providing a measure of the predictors' importance and effect on prediction performance. A z-value is produced from a paired test of the full model and marginalized model prediction errors. Higher, positive z-values indicate that prediction errors are larger if predictors are marginalized out of the model. Therefore, the predictor is useful in the model and improves model prediction performance [15]. Partial dependence plots are also produced to show the marginal effects of the top two environmental predictor variables on the response variable.

While historical random forest models have a built-in validation by testing on a bootstrapped dataset and presenting out-of-bag-error, I conducted a cross-validation process that fine-tuned the parameter (*mtry*) for number of predictors at each split in a random forest regression tree. Tuning this parameter has been shown to improve model performance [23]. The number of trees parameter in the random forest (*ntrees*) is tested

to see whether model error changes with more trees in the forest.

For cross-validation, the data were divided into eight sets, or folds, with one water year per fold. Instead of randomly sampling for the training set like in other k-fold validation methods, water years remained intact to maintain longitudinal data for the historical random forest. The first fold is held back for validation, while folds 2-8 are used for training with different *mtry* values (2, 4, 6, and 8). Specifically, I looped through different values of *mtry* with each training run, then calculated the root mean square error (RMSE) after each test. This was repeated so that each of the eight folds was held back one at a time and served as the validation or testing set. In the end, the *mtry* value that led to the lowest prediction error was used for the full model. Cross-validation error was calculated as the average of the RMSE across the validation tests.

Migratory Fish Catch Weight Per Unit Effort

The response variable for each random forest model was log-transformed daily average catch weight from the Dai fishery in the Tonle Sap River [11] (Table 5-2, Figure 7-13, Figure 7-15, Figure 7-17, Figure 7-19, Figure 7-21, and Figure 7-23). Models were developed for six species during October to March, the dry season when fish migrate, from 2002 – 2008. The Dai fishery spans 30 km in Tonle Sap River. A total of 64 Dai units are spread across 14 rows (Figure 5-2). See Ngor et al. (2018) for catch per unit effort (CPUE) calculations for the Dai fishery. The mouth of each Dai, or bagnet, is approximately 25 m, with mesh size ranging from approximately 15 cm at the mouth and 1 cm at the codend. Nets face upstream to catch migrating fish on their route to deep pools for dry season refuge [4]. It is estimated that each Dai unit captures 2.8 percent of migrating fish and that 83 percent of migrating fish have been caught by the final row of

Dais in the Tonle Sap River [4, 11]. The catch weight data over time and distributions of the catch weight and log-transformed catch weight are provided in the Appendix C (Figure 7-14 – Figure 7-17).

Table 5-2. Descriptive statistics of catch weight datasets

	Mean catch weight (kg)	**Standard deviation catch weight (kg)**	**Mean log(catch weight)**	**Standard deviation log(catch weight)**
Pangasianodon hypophthalmus	214.07	250.02	4.90	0.96
Cyclocheilichthys enoplos	61.53	98.48	3.09	1.44
Cirrhinus microlepis	108.11	125.14	4.27	0.83
Osteochilus melanopleurus	270.29	283.39	4.87	1.44
Henicorhynchus lobatus	11.39	4.55	2.35	0.41
Labiobarbus lineatus	8.96	3.63	2.11	0.41

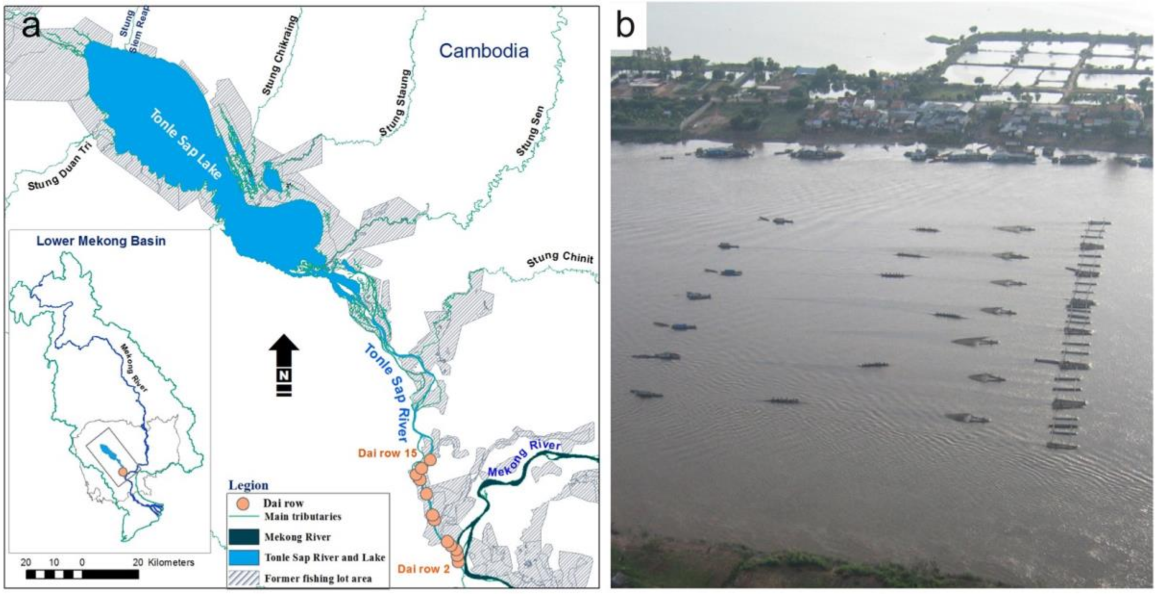

Figure 5-2. The Dai fishery on the Tonle Sap River (a) is made up of 64 units across 14 rows. Photo (b) shows seven units of one row. (Figure Source: Ngor et al., 2018).

Environmental Predictor Variables

Kummu et al. (2014) developed equations to calculate Tonle Sap River flows in and out of Tonle Sap Lake using water level data at Prek Kdam on Tonle Sap River, Phnom Penh Port at the confluence of the Mekong River and Tonle Sap River, and Kompong Luong. These are shown below:

$$Q_{TSR,in} = -15.0467 * F^2 + 859.839 * F - 782.264$$

$$Q_{TSR,out} = 8.784 * F^2 + 434.465 * F + 167.152$$

$$F = (WL_{PK})^{1.2} * (|WL_{PP} - WL_{KL}|)^{0.5}$$

where $Q_{TSR,in}$ is the Tonle Sap River flow (cms) into Tonle Sap Lake during the wet season, $Q_{TSR,out}$ is the Tonle Sap River flow (cms) out of Tonle Sap Lake during the dry season, WL_{PK} is the water level (m) at Prek Kdam on the Tonle Sap River, WL_{PP} is the water level (m) at Phnom Penh Port at the confluence of the Tonle Sap River and

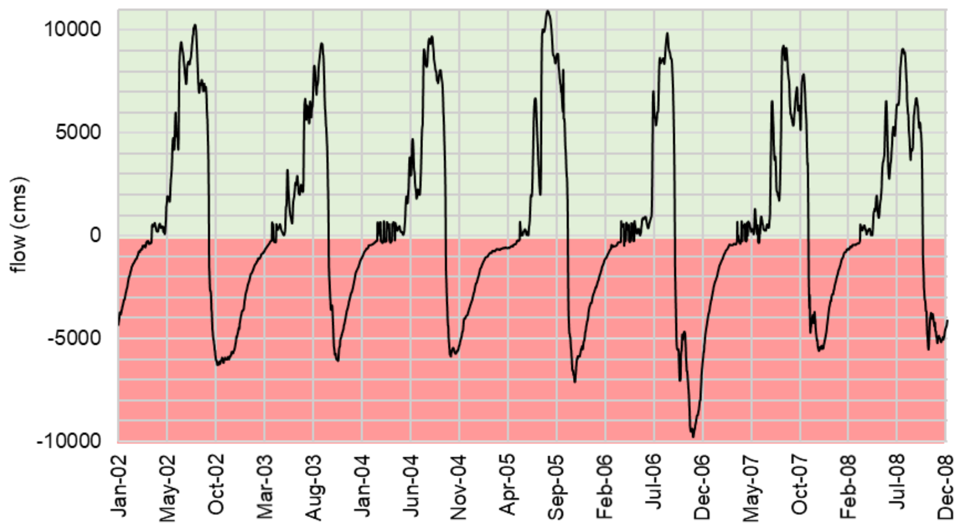

Figure 5-3. Hydrograph for Tonle Sap River for 2002-2008. The positive flow values (green) are during the wet season when the Mekong River reverses flow into Tonle Sap Lake. The negative flow values (red) are during the dry season when flow is leaving Tonle Sap Lake to the Mekong River.

Mekong River, and WL_{KL} is the water level (m) at Kompong Luong in Tonle Sap Lake.

The water balance calculation also accounts for overland flow from tributaries, precipitation to the lake, and evaporation from the lake's surface [24]. Tonle Sap River flows are calculated for 2002 to 2008 using this method and water level data from the Mekong River Commission [25]. The hydrograph for Tonle Sap River shows the dry season as negative flows leaving Tonle Sap Lake to the Mekong River (Figure 5-3). In the wet season, Tonle Sap River flows are reversed from the Mekong River pushing water back into Tonle Sap Lake.

Flow timing, duration, and magnitude define hydrologic regimes, which drive ecosystem functions and support species, habitat, and ecosystem services [26]. I calculated hydrologic metrics that represent water year (October – September) flow timing, duration, and magnitude (Table 5-3 and Appendix C, Figure 7-25). Three other

metrics of interest, precipitation (Appendix C, Figure 7-26), moon cycle, and previous catch weight, were included in the models as well. Precipitation and water level data are from the Mekong River Commission [25].

Results

Cross-Validation

Training and testing the model on separate sets, or folds, of the data in multiple runs with different parameter values leads to an average RMSE range of 0.38 to 2.1 for the six species (Table 5-4). The testing errors of the training models for two species (*Henicorhynchus lobatus* and *Labiobarbus lineatus*) are within one standard deviation in their distribution (Table 5-2). The errors for *Pangasianodon hypophthalmus*, *Cirrhinus microlepis*, and *Osteochilus melanopleurus* are just outside of one standard deviation in their distribution. This indicates that these models fit the testing folds well, relative to the variation in log(catch weight) datasets. However, these errors are close the standard deviation and the error for *Cyclocheilichthys enoplos* is beyond its standard deviation, indicating the variables in the model are not fully predicting the response variable, log(catch weight).

The cross-validation process also identifies the *mtry* value that resulted in the lowest prediction error for each model. The *ntrees* parameters is set to 100, after seeing that the out-of-bag error converges before 100 trees (Appendix C, Figure 7-33 – Figure 7-35). Increasing the number of trees more than 100 results only in an increased computation time.

Table 5-3. Twenty-one predictor variables categorized by hydrologic characteristic

Characteristic	Predictor	Description/Method of Calculation
Timing of flows (eight predictors)	Days from 1-day min/max flow in water year	Found lowest and highest 1-day flow for each water year, then calculated days from min and max for each observation
	Days from 7-day min/max flow in water year	Found lowest and highest consecutive 7-day period of flow for each water year, then calculated days from min and max for each observation
	Days from 30-day min/max flow in water year	Found lowest and highest consecutive 30-day period of flow for each water year, then calculated days from min and max for each observation
	Days from 90-day min/max flow in water year	Found lowest and highest consecutive 90-day period of flow for each water year, then calculated days from min and max for each observation
Duration of flows (four predictors)	Cumulative flow	Calculated by summing current day flow with previous days' flows in that water year
	Cumulative # of high pulse days	Summation of days above the 75^{th} percentile flow in that water year
	Cumulative # of low pulse days	Summation of days below the 25^{th} percentile flow in that water year
	Residuals from average cumulative flow	Took the average cumulative flow for the current day in each water year of the dataset, then found the difference between the current day cumulative flow from the average cumulative flow
Magnitude of flows (six predictors)	Flow	Tonle Sap River flow calculated with equations from Kummu et al. (2014)
	Above 75^{th} percentile flow	Binary variable; above 75^{th} percentile of flow for all water years in dataset (1)
	Below 25^{th} percentile flow	Binary variable; below 25^{th} percentile of flow for all water years in dataset (1)
	Water level	Water level (m) at Prek Kdam, Phnom Penh Port, and Kompong Luong
Other (three predictors)	Moon cycle	Lunar cycle rounded to 30 days; new moon (0) to full moon (30)
	Precipitation	Precipitation (mm) on Tonle Sap River in Kampong Chhang Province
	Previous catch weight	Daily catch weight data

Table 5-4. Cross-validation average RMSE and mtry tuning results

	Cross-validation average RMSE ***log(catch weight)***	***mtry* tuning results** ***# of predictors at each split***
Pangasianodon hypophthalmus	0.99	2
Cyclocheilichthys enoplos	2.1	8
Cirrhinus microlepis	0.85	8
Osteochilus melanopleurus	1.5	4
Henicorhynchus lobatus	0.41	6
Labiobarbus lineatus	0.38	8

Model Fit

The full models with the optimal *mtry* and *ntrees* parameter values determined in the cross-validation process were run. Randomly selected data, or bootstrapped data, from the dataset were then tested and produced a prediction error, or OOB MSE. These errors ranged from 0.16 to 2.3 for the six models (Table 5-5). For the full models with tuned parameters, the errors are overall smaller than the training models discussed above. The errors for four of the species (*Cyclocheilichthys enoplos*, *Cirrhinus microlepis*, *Henicorhynchus lobatus,* and *Labiobarbus lineatus*) are within one standard deviation of their distributions. Additionally, the error for *Pangasianodon hypophthalmus* is again just outside one standard deviation in its distribution. These results indicate the full model with tuned parameters fit the data well when relating the errors to the variation of the log(catch weight) datasets. R^2 statistics also show that the models explain more than 70% of the variance in the catch weight for each species (Table 5-5).

Table 5-5. Out-of-bag mean squared error for each model

	OOB MSE *log(catch weight)^2*	**R^2** *proportion*
Pangasianodon hypophthalmus	0.99	0.73
Cyclocheilichthys enoplos	1.3	0.83
Cirrhinus microlepis	0.76	0.83
Osteochilus melanopleurus	2.3	0.73
Henicorhynchus lobatus	0.19	0.71
Labiobarbus lineatus	0.16	0.73

However, the relationship between the measured and modeled data shows that the model is underestimating the high catch weight values and overestimating the lows in the measured data (Figure 5-4 – Figure 5-6). In other words, the 21 predictor variables included in this study are not explaining the high catch weight values and low catch weight values, indicating there are likely other environmental cues that impact fish migration. This is discussed further in the last section.

Variable importance tables for each of the six species rank all the predictors in the model (Appendix C, Table 7-14 – Table 4-19). Predictors were ranked by the relative percentage change in prediction error between the full model with all predictors and the marginalized prediction when a predictor variable was marginalized out of the model. The relative change in error is a measure of the sensitivity of the model to including or excluding each predictor variable. The decrease in accuracy when the top two environmental predictors are marginalized out of the model, as well as the marginal effects of the top two environmental predictors on catch weight, is presented for each of the six species.

For *Pangasianodon hypophthalmus*, the top two environmental cues are days

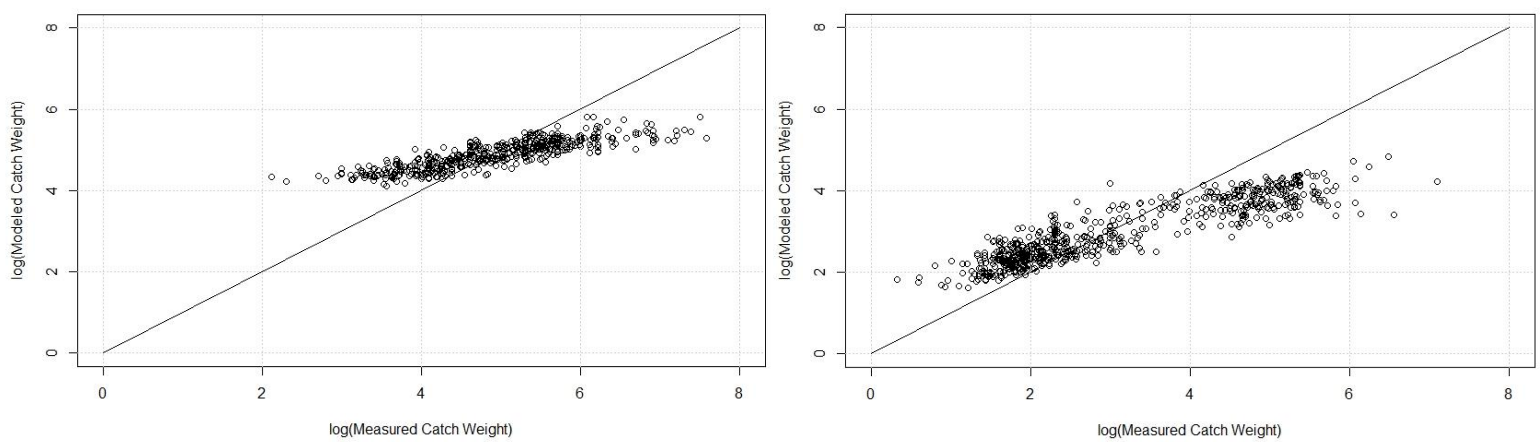

Figure 5-4. Measured versus modeled log(catch weight) for *Pangasianodon hypophthalmus* (left) and *Cyclocheilichthys enoplos* (right) with 1:1 lines.

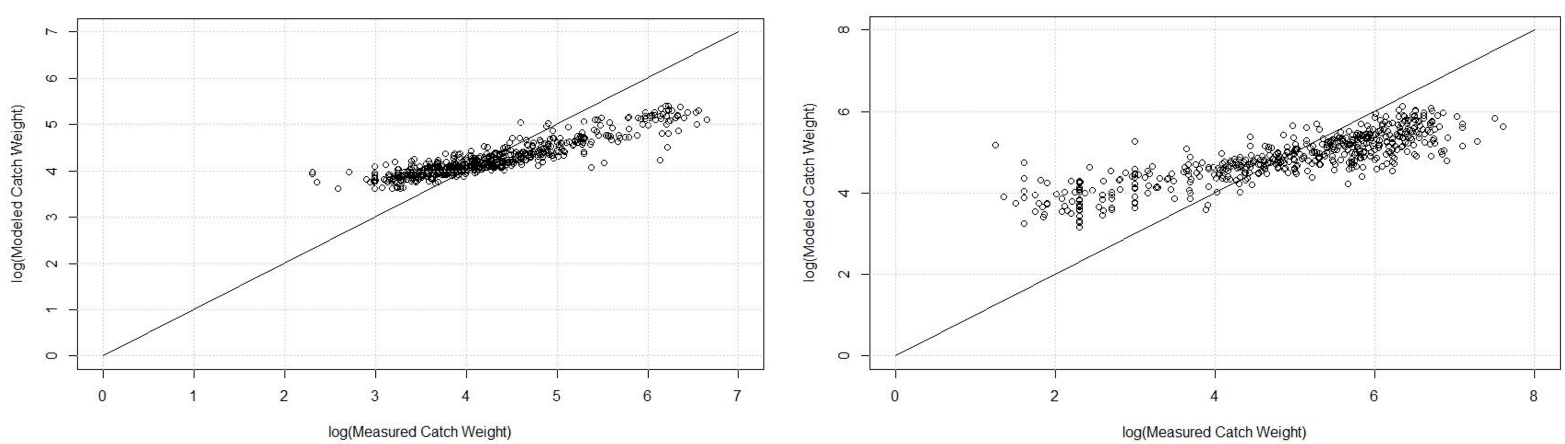

Figure 5-5. Measured versus modeled log(catch weight) for *Cirrhinus microlepis* (left) and *Osteochilus melanopleurus* (right) with 1:1 lines.

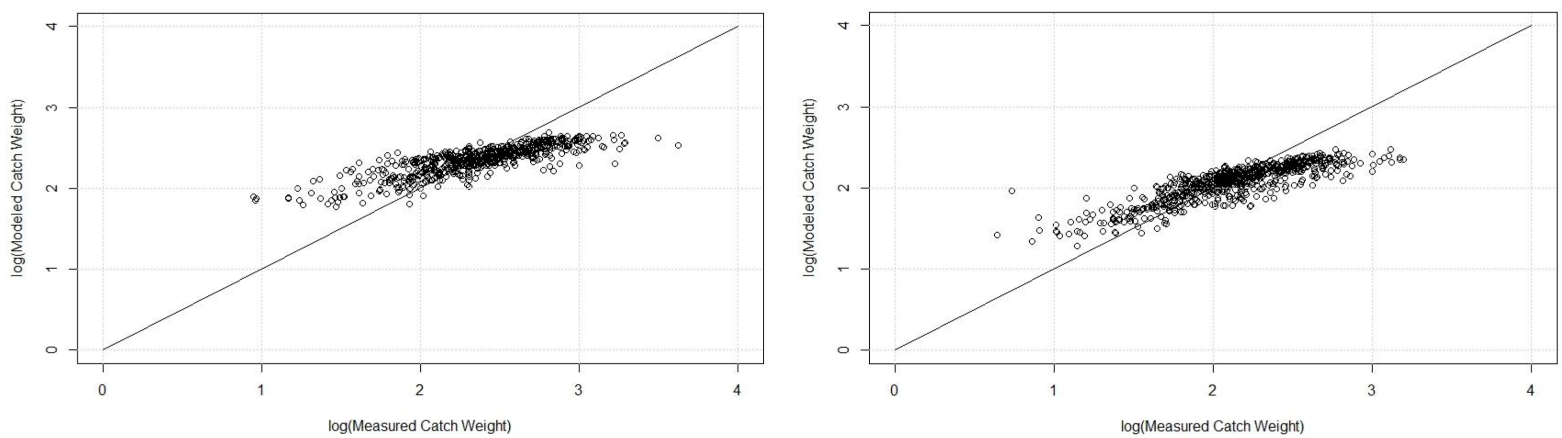

Figure 5-6. Measured versus modeled log(catch weight) for *Henicorhynchus lobatus* (left) and *Labiobarbus lineatus* (right) with 1:1 lines.

from the 90-day minimum flow for the water year and days from the 7-day maximum flow for the water year (Figure 5-7). There is also an increase in catch weight, 150 days before the 90-day minimum flow for the water year (Figure 5-8). Second, catch weight is slightly higher 0-50 days after the 7-day maximum flow period for the water year.

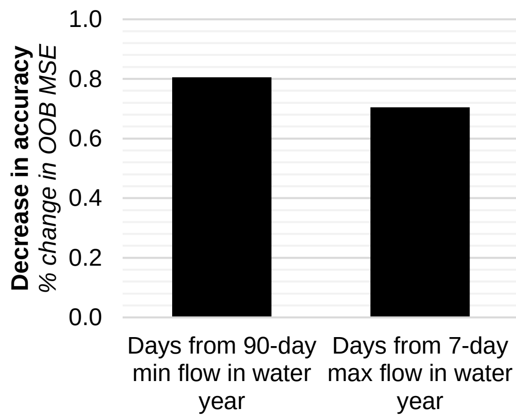

Figure 5-7. Decrease in OOB MSE when the top two environmental cues are marginalized out of model for *Pangasianodon hypophthalmus.*

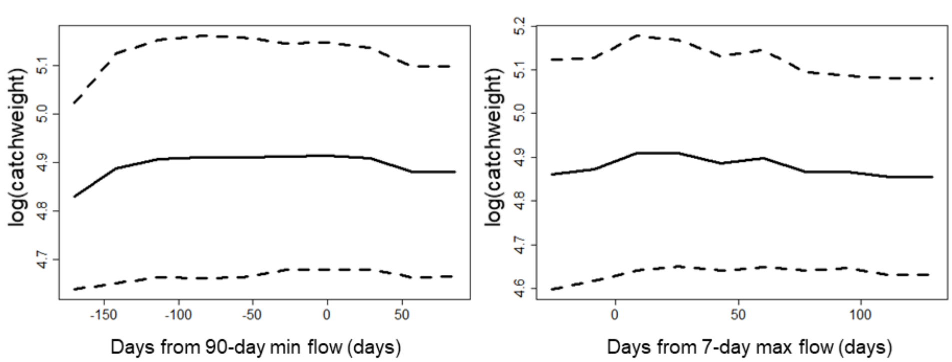

Figure 5-8. Marginal effects of the top two environmental cues on log(catch weight) with -/+ 2 standard errors (dashed lines) for *Pangasianodon hypophthalmus.*

For *Cyclocheilchthys enoplos*, the top two environmental cues are moon cycle day

and the cumulative number of low pulse days (Figure 5-9). The partial dependence plots show that catch weight is largest in the days after the new moon. Additionally, catch weight increases in the first 5 cumulative low pulse days (Figure 5-10).

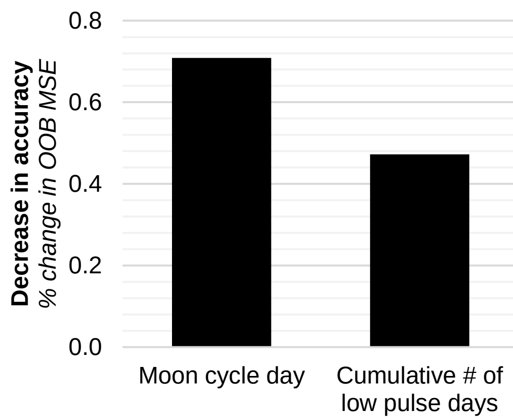

Figure 5-9. Decrease in OOB MSE when the top two environmental cues are marginalized out of model for *Cyclocheilchthys enoplos.*

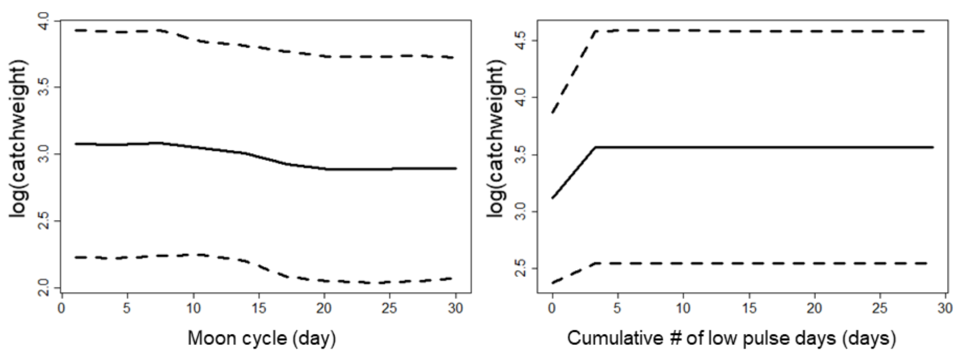

Figure 5-10. Marginal effects of the top two environmental cues on log(catch weight) with -/+ 2 standard errors (dashed lines) for *Cyclocheilchthys enoplos.*

The results for the *Cirrhinus microlepis* model show that days from the 90-day maximum flow for the water year and cumulative flow are the top two environmental predictors (Figure 5-11). Secondly, catch weight is highest approximately 50 days before the 90-day maximum flow in a water year (Figure 5-12). The duration of flow

characteristic, cumulative flow, is the second top environmental cue. Catch weight is largest closer to zero for cumulative flow, which indicates that catch weight for this species is highest at the beginning of the water year in October (Figure 5-12).

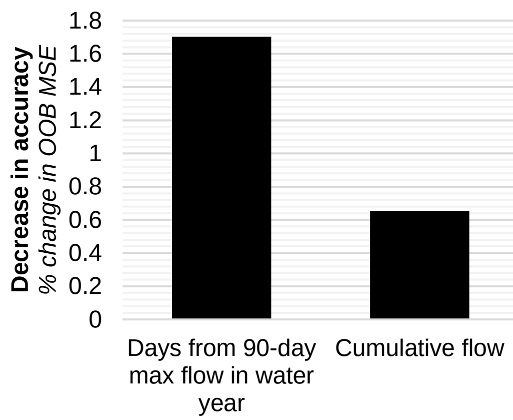

Figure 5-11. Decrease in OOB MSE when the top two environmental cues are marginalized out of model for *Cirrhinus microlepis.*

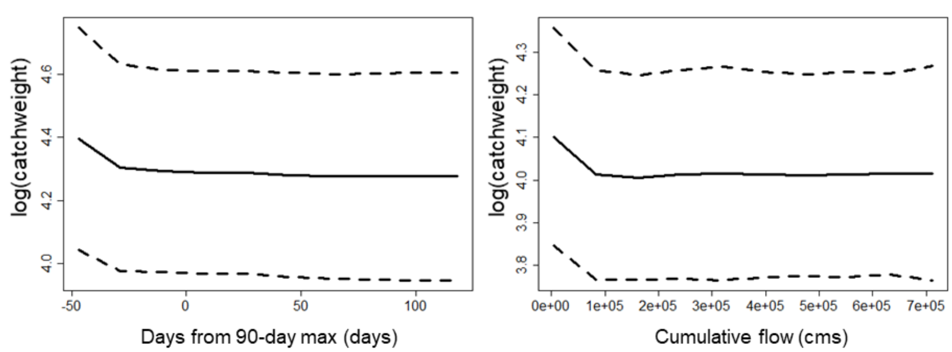

Figure 5-12. Marginal effects of the top two environmental cues on log(catch weight) with -/+ 2 standard errors (dashed lines) for *Cirrhinus microlepis.*

The *Osteochilus melanopleurus* results show that cumulative number of high pulse days and days from the 1-day minimum flow in a water year are the top two environmental predictors for this species (Figure 5-13). Specifically, an increase in the cumulative number of high pulse days leads to a decrease in catch weight (Figure 5-14).

Second, catch weight of this species is highest 150 days before the 1-day minimum flow in the water year (Figure 5-14).

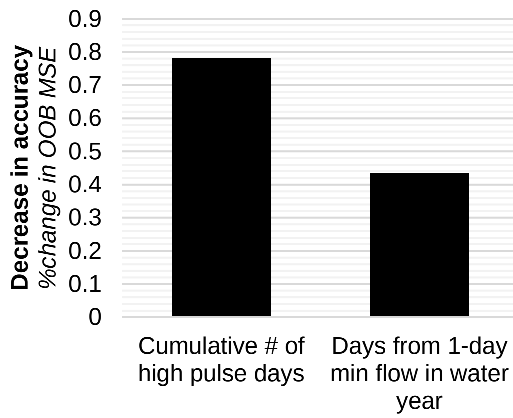

Figure 5-13. Decrease in OOB MSE when the top two environmental cues are marginalized out of model for *Osteochilus melanopleurus.*

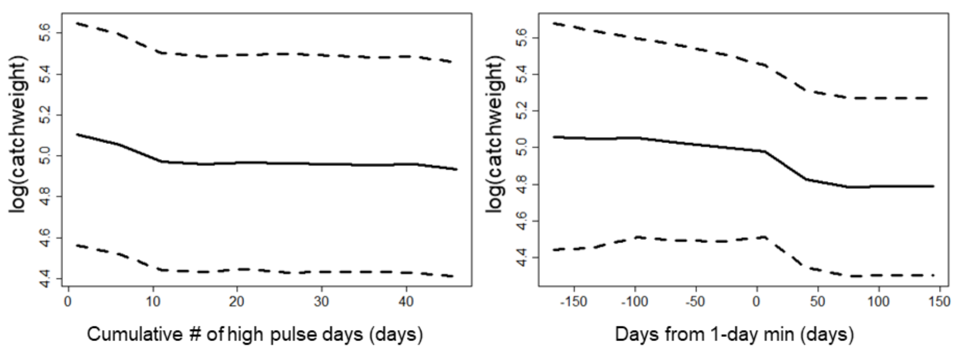

Figure 5-14. Marginal effects of the top two environmental cues on log(catch weight) with -/+ 2 standard errors (dashed lines) for *Osteochilus melanopleurus.*

For *Henicorhynchus lobatus*, days from the 1-day maximum flow for the water year and water level at the Phnom Penh Port are the top two environmental predictors (Figure 5-15). There is an increase in catch weight in the first 15 days after the 1-day maximum flow in a water year (Figure 5-16). Additionally, with higher water levels at the Phnom Penh Port, catch weight decreases. In other words, when water levels at

Phnom Penh Port increase signaling flooding, catch weight decreases due to the Mekong River reversing and the Tonle Sap River flows into Tonle Sap Lake.

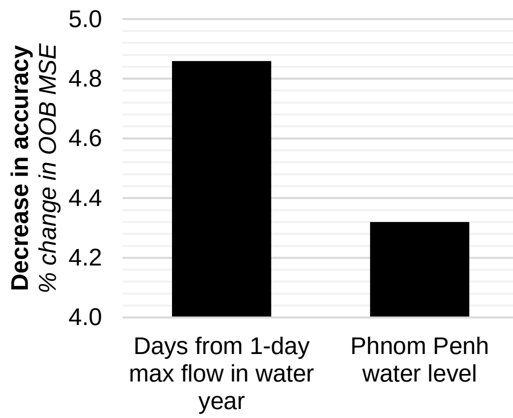

Figure 5-15. Decrease in OOB MSE when the top two environmental cues are marginalized out of model for *Henicorhynchus lobatus.*

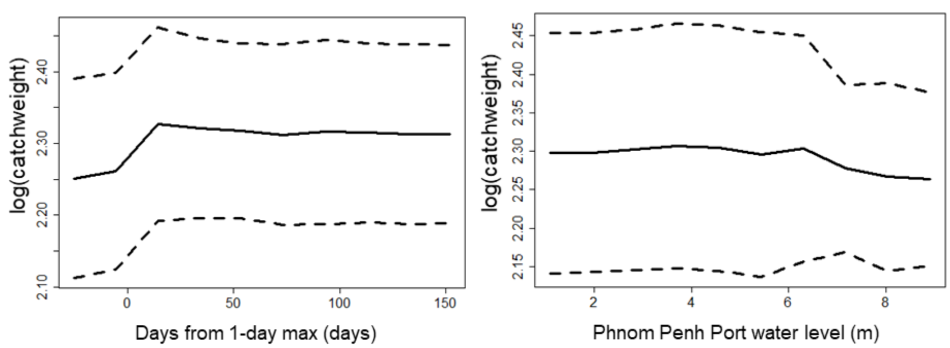

Figure 5-16. Marginal effects of the top two environmental cues on log(catch weight) with -/+ 2 standard errors (dashed lines) for *Henicorhynchus lobatus.*

Days from the 7-day minimum flow for the water year and days from the 7-day maximum flow for the water year are the top two environmental cues for *Labiobarbus lineata* (Figure 5-17). Additionally, catch weight increases 100 to 150 days before the 7-day minimum flow in a water year, as well as in the days right before the 7-day maximum flow in a water year (Figure 5-18).

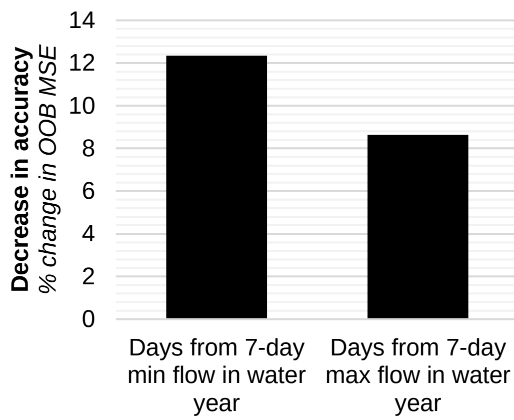

Figure 5-17. Decrease in OOB MSE when the top two environmental cues are marginalized out of model for *Labiobarbus lineata*.

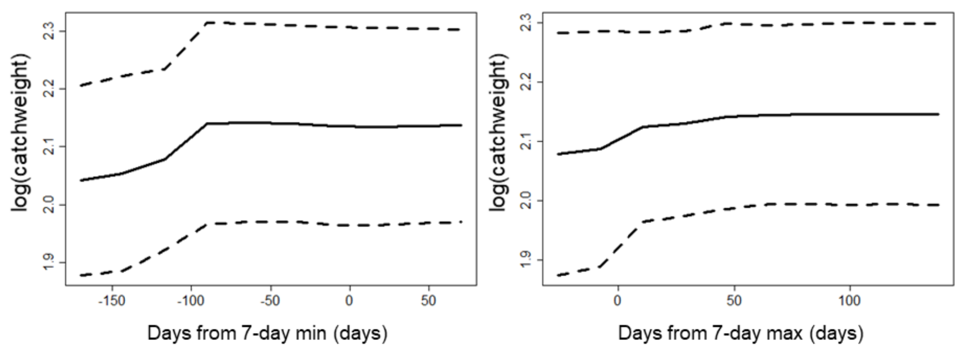

Figure 5-18. Marginal effects of the top two environmental cues on log(catch weight) with -/+ 2 standard errors (dashed lines) for *Labiobarbus lineata*.

Discussion

Using a relatively new approach to random forest modeling, environmental predictors for fish migration are identified and ranked. While the RMSEs for four of the six models are within one standard deviation of the corresponding species distributions and R^2 values for all six models are greater than 0.70, the patterns of residuals show that the models do not fully predicting the response variables. Using 21 predictors, the models are most effective at predicting the overall trends of the catch weight. However, they do

not sufficiently explain the variance of the catch weight, indicating that either: 1) there are other environmental cues needed to predict fish migration or 2) fish migration in this system is not easily predicted due to the high daily and seasonal variability of the Tonle Sap system.

For the large river catfish, *Pangasianodon hypophthalmus*, the top two environmental predictors were timing of flows metrics. The top environmental cues for the medium- to large-mud carps, *Cyclocheilichthys enoplos*, *Cirrhinus microlepis*, and *Osteochilus melanopleurus*, include moon cycle, duration of flows, and timing of flows metrics. Timing of flows and magnitude of flows metrics were the top environmental predictors for the small mud carps. As expected, previous catch weight was often a top predictor of the response variable. The partial dependence plots highlight that, with an increase of previous catch weight of *Pangasianodon hypophthalmus, Cyclocheilichthys enoplos, Cirrhinus microlepis,* and *Labiobarbus lineatus*, there is an increase in the response variable, log(catch weight). However, the focus of this study is on the environmental predictors.

From these results, all three characteristics of flows (timing, duration, and magnitude) are represented as important environmental cues for fish to migrate from Tonle Sap Lake. Timing of flow metrics are the most common top environmental predictor for catch weight, showing overlap across trophic levels. Additionally, for one of the species, model results suggest lunar phase is an important cue for migration from Tonle Sap Lake.

The dataset in this study is limited in that there are only 8 water years represented. Including more water years would allow the historical random forest algorithm to better

learn and identify patterns across more observations. Further, the prediction errors and pattern of residuals of the full models highlight that there are perhaps other predictors not included in this model that could improve model performance. Future work would benefit from including other potential predictors that may be driving fish migration in this system. Some examples of additional predictor variables include stream temperature, turbidity, and other water quality metrics.

Migratory fish species contribute many ecosystem services, from provisioning to cultural significance. The Dai fishery in Tonle Sap Lake and River contributes 60% of the annual commercial fish production in Cambodia and feeds millions [8, 12]. The Dai fishery is dependent on the unique hydrology of the Tonle Sap system that supports migratory fish. Previous research on the impacts of dams throughout the Mekong River Basin have provided evidence of the alteration of flows and fragmented habitats [6, 7]. With 11 proposed dams on the mainstem of the Mekong River, the alteration of the timing, duration, and magnitude of flows impact migratory cues, potentially disrupting the migratory fish life cycle. These findings of the top migration cues for these six species in decline, provide a better understanding of how management and conservation strategies can be developed to limit when these fish are harvested. As examples, fisheries and locals should limit their catch of: *Cyclocheilchthys enoplos* in the first five days after the new moon, *Cirrhinus microlepis* at the beginning of the dry season, *Osteochilus melanopleurus* and *Henicorhynchus lobatus* in the days after high flows.

Additionally, because the migration of these species are not easily predicted, management strategies should allow for increased protection of these fish species. Specifically, this study provides support for proposed management actions of only

harvesting mature fish that have spawned at least once before capture and that have reached a minimum length foe each species, and releasing mega-spawners or fish that are longer than the optimal length for its species by 10% or more [27]. Protecting the diversity of species and those species in decline will maintain an adaptive and more resilient ecosystem to climate change [12]. Therefore, these conservation efforts also serve as climate adaptation measures that could help with long-term sustainable development in this changing system.

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CHAPTER 6

CONCLUSION

The challenges we face as a society, such as climate change and environmental degradation, are vast and varied, and consequently our solutions should be too. Merging social and natural dimensions of environmental problems is key for improving climate adaptation science and managing for ecosystem services. The ecosystem services framework allows for discipline integration, and it encourages collaboration and communication between social and physical scientists for these complex environmental challenges.

This dissertation is an example of using interdisciplinary approaches to addressing research questions from multiple perspectives. It demonstrates multiple impacts on ecosystem services and highlights the challenges in protecting and maintaining them. With stormwater management, there are consequences from urbanization and conventional gray infrastructure on environmental quality and loss of ecosystem services. Chapter 1 provides insight to research status at the intersection of stormwater management and ecosystem services, and promising directions for the future. I highlighted research from different disciplines and illustrates how ecosystem services are relevant to many researchers and stakeholders. Chapter 2 illustrates modeling that can further our understanding of the effects of green stormwater infrastructure on downstream surface water and ecosystem services. In collaboration with environmental engineers and decision-makers, this modeling serves as a tool for adaptive stormwater management plans.

With the increasing damage to ecosystem services from fire, an important

objective for Chapter 3 is to better understand challenges and climate adaptation barriers experienced by fire managers. Our interdisciplinary team of ecologists, watershed scientists, an applied economist, and social scientists allowed for us to conduct a multifaceted approach to a broad research question, using different methods and types of data. Lastly, with increased fishing pressure in the rapidly changing Tonle Sap system posing risks to ecosystem services related to fish, Chapter 4 analyzes the environmental conditions that need to be maintained for migratory fish species. As part of a larger, multidisciplinary project, I integrated fish and hydrologic data to contribute more understanding of what environmental cues predict and impact the ecosystem services of these species.

The research area of ecosystem services and climate adaptation is wide-ranging. The chapters in this dissertation serve as an example of the diversity of work that falls under these umbrellas. I have had great opportunities to explore different environmental challenges and solutions related to water, fish, and fire, providing me with experience with various disciplinary approaches. Overall, this inclusive, multidisciplinary work offers an example of the direction that research on ecosystem services and climate adaptation should continue to move.

APPENDICES

Appendix A – Chapter 3 Supplementary Materials

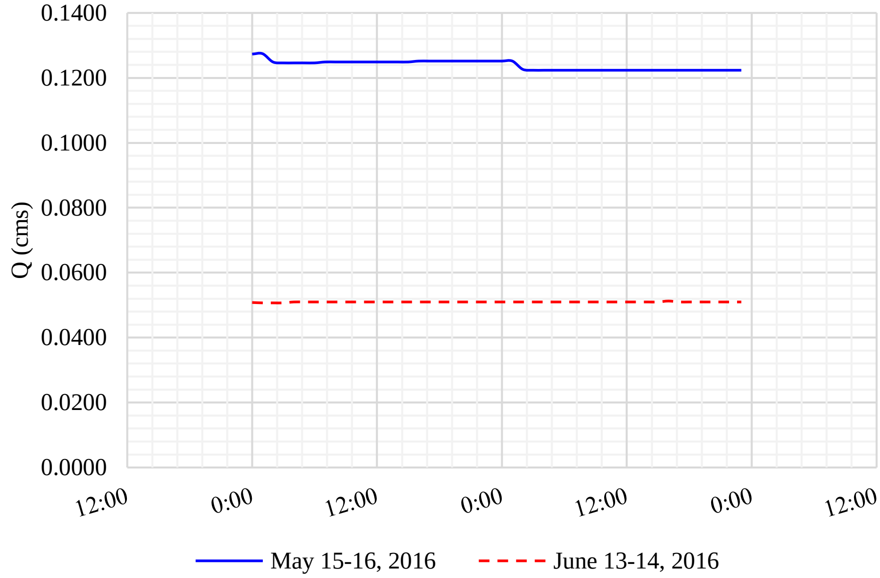
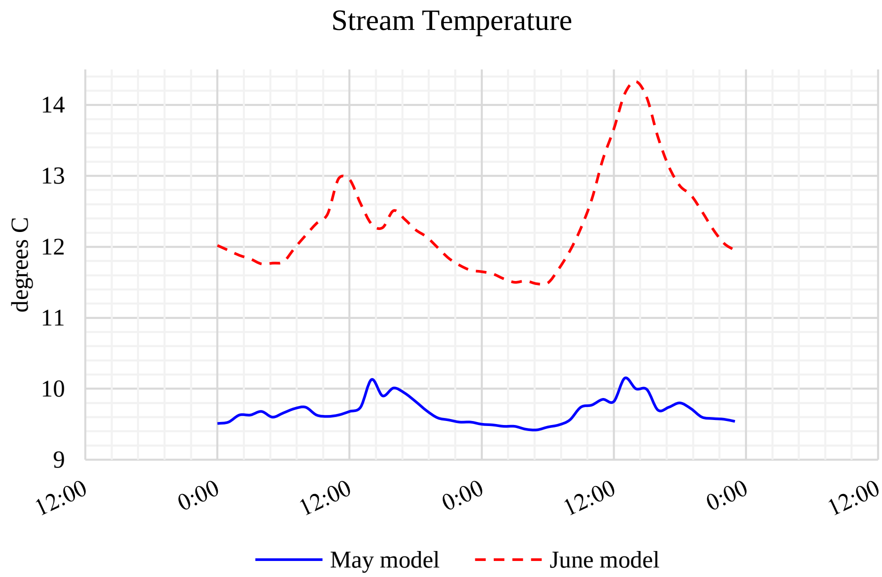


Figure 7-1. Flow at the outlet of Red Butte Reservoir for the spring and summer Red Butte Creek models.



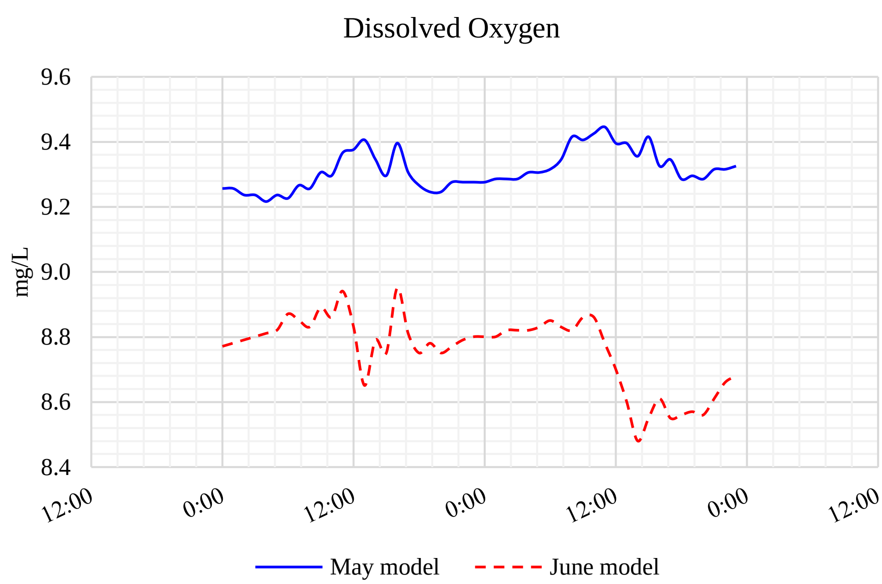
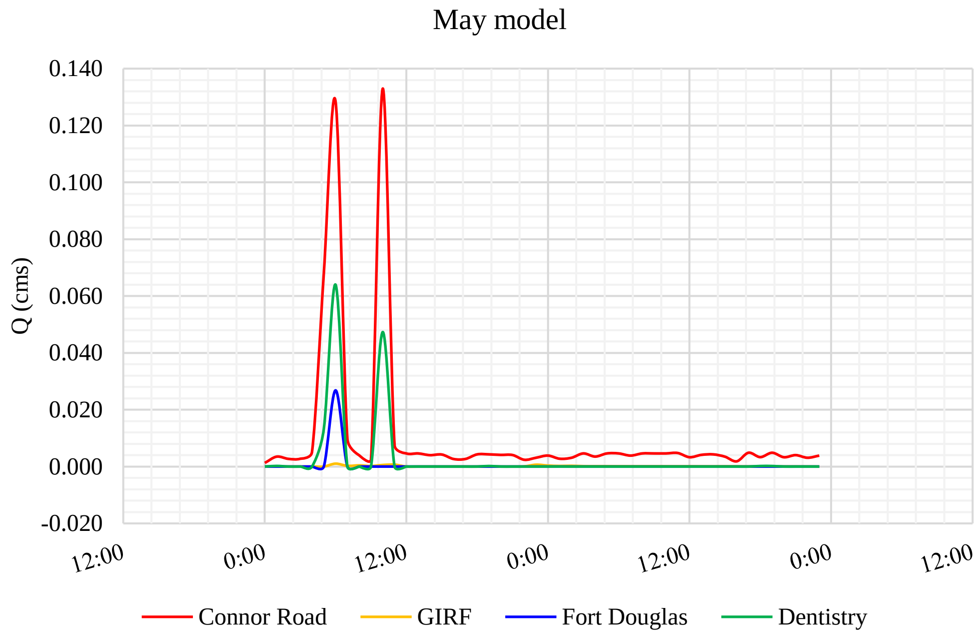


Figure 7-2. Stream temperature and dissolved oxygen at the model headwater for the spring and summer Red Butte Creek models.



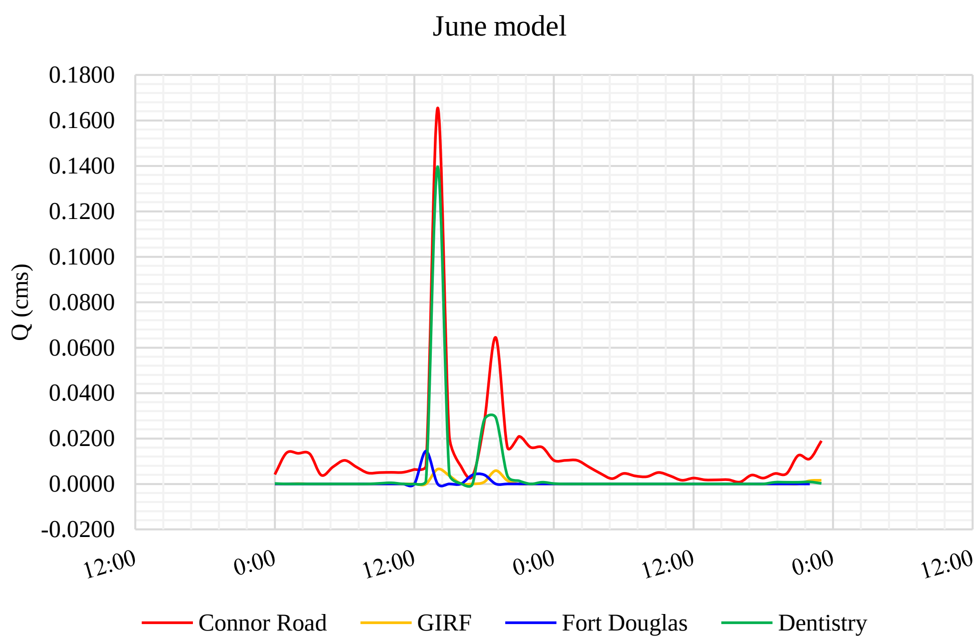


Figure 7-3. Inflows from the storm drains between Cottam's Grove and Foothill Drive for the spring and summer Red Butte Creek models.

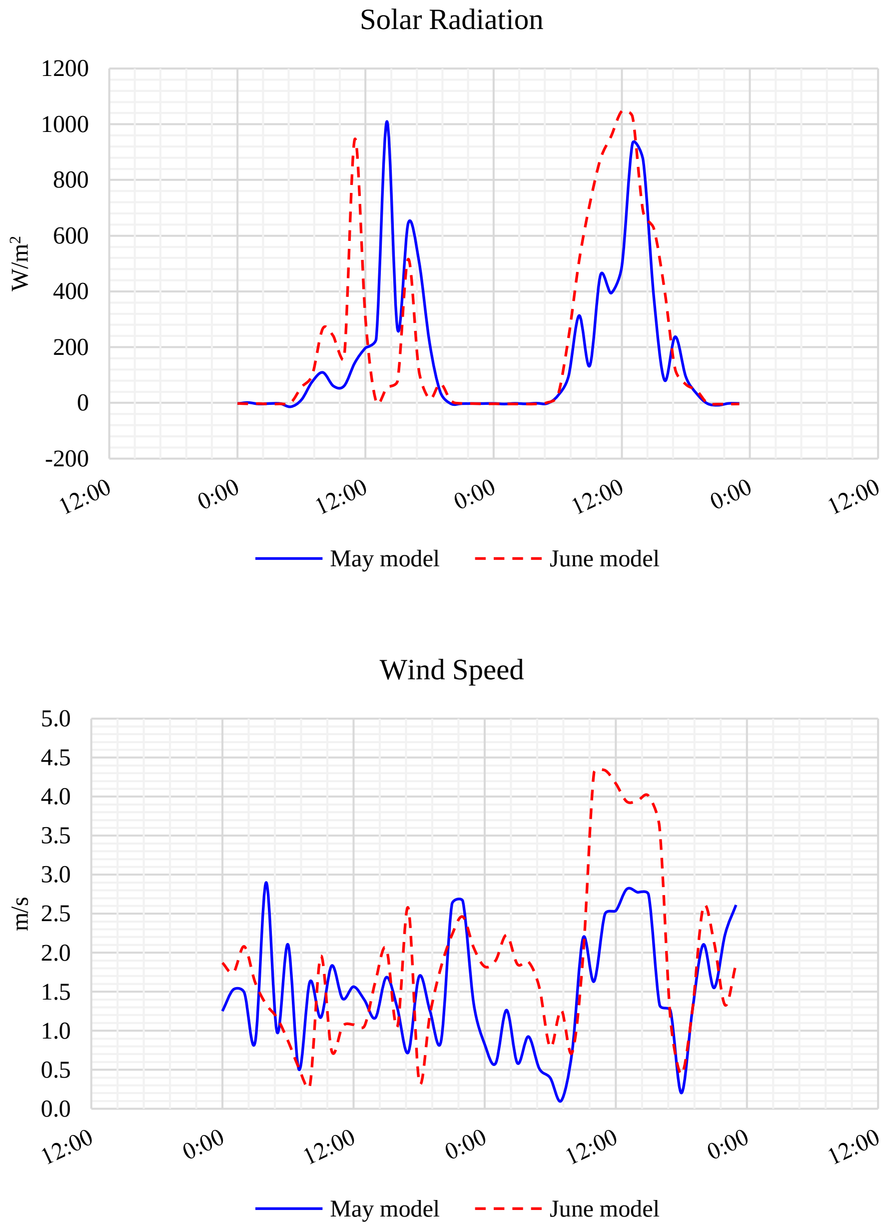
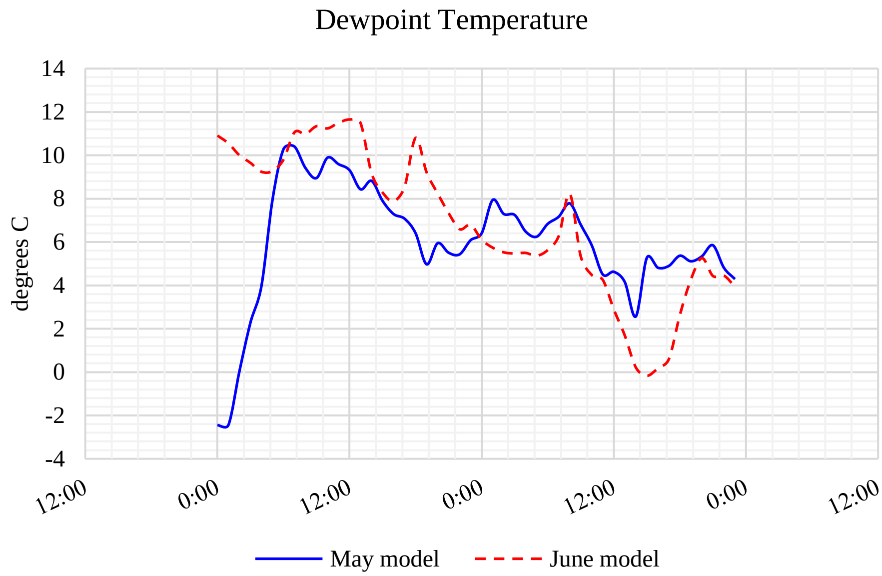


Figure 7-4. Solar radiation and wind speed during the modeled spring and summer days for the Red Butte Creek models.



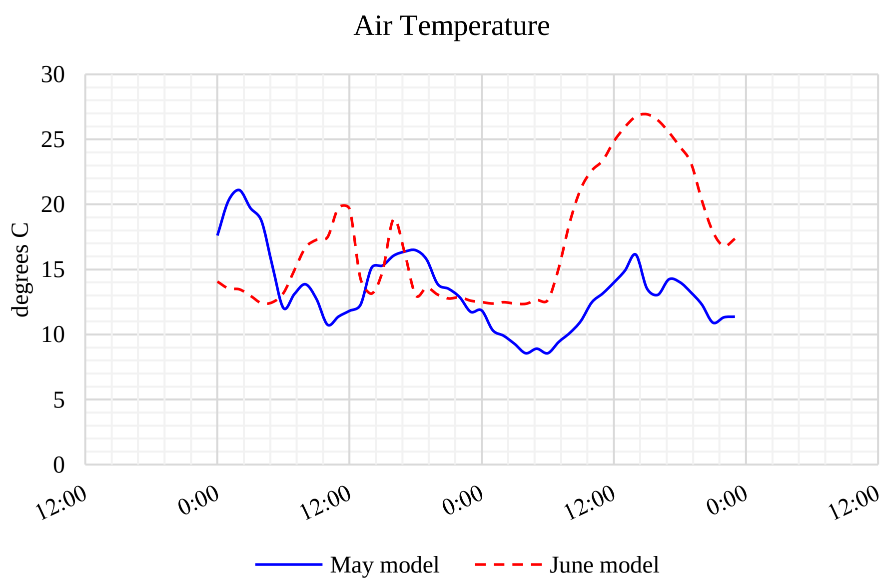


Figure 7-5. Dewpoint and air temperature during the modeled spring and summer days for the Red Butte Creek models.

Table 7-1. Average TP concentration (µg/L) grab samples collected in spring and summer from 2013-2016

spring	Headwater	RBC at Cottam's Grove	RBC at Foothill Drive	RBC at 1300E
	n=4	*n=4*	*n=3*	*n=3*
mean	50.2	61.3	60.9	38.8
min	39.8	40.5	52.7	19.0
max	80.0	80.0	70.0	57.4

summer	Headwater	RBC at Cottam's Grove	RBC at Foothill Drive	RBC at 1300E
	n=3	*n=3*	*n=2*	*n=2*
mean	53.3	76.9	105.0	105.0
min	39.8	50.6	70.0	30.0
max	70.0	120.0	140.0	180.0

Table 7-2. Channel geometry for the Red Butte Creek models

Label	Reach	Channel slope (m)	Manning's n	Spring Bottom Width (m)	Summer Bottom Width (m)
	1	0.137823	0.1	0.60	0.50
	2	0.248534	0.1	0.60	0.50
	3	0.062224	0.1	0.60	0.50
	4	0.058342	0.1	0.60	0.50
	5	0.077712	0.1	0.60	0.50
	6	0.076229	0.1	0.60	0.50
	7	0.111885	0.1	0.60	0.50
	8	0.056345	0.1	0.60	0.50
	9	0.026103	0.1	0.60	0.50
	10	0.018351	0.1	0.60	0.50
	11	0.068168	0.1	0.60	0.50
	12	0.073824	0.1	0.60	0.50
	13	0.064727	0.1	0.60	0.50
	14	0.071221	0.1	0.60	0.50
	15	0.083691	0.1	0.60	0.50
	16	0.050289	0.1	0.60	0.50
	17	0.045827	0.1	0.60	0.50
	18	0.049469	0.1	0.60	0.50
	19	0.030307	0.1	0.60	0.50
	20	0.036495	0.1	0.60	0.50
	21	0.019032	0.1	0.60	0.50
	22	0.036026	0.1	0.60	0.50
	23	0.018232	0.1	0.60	0.50
	24	0.018757	0.1	0.60	0.50
	25	0.010623	0.1	0.60	0.50
	26	0.022646	0.1	0.60	0.50
	27	0.012331	0.1	0.60	0.50
	28	0.113444	0.1	0.60	0.50
	29	0.042057	0.1	0.60	0.50
	30	0.040056	0.1	0.60	0.50
	31	0.04345	0.1	0.60	0.50
Cottam's Grove	32	0.031488	0.1	0.60	0.50
	33	0.055839	0.09	2.00	0.65
	34	0.099575	0.09	2.00	0.65
	35	0.022023	0.09	2.00	0.65
	36	0.094841	0.09	2.00	0.65
	37	0.067002	0.09	2.00	0.65
	38	0.022301	0.09	2.00	0.65
	39	0.014261	0.09	2.00	0.65
	40	0.043276	0.09	2.00	0.65
	41	0.036781	0.09	2.00	0.65
	42	0.052451	0.09	2.00	0.65
	43	0.040115	0.09	2.00	0.65
	44	0.021565	0.09	2.00	0.65
Foothill Drive	45	0.027306	0.09	2.00	0.65
	46	0.02467	0.09	1.00	0.80
	47	0.053088	0.09	1.00	0.80
	48	0.040276	0.09	1.00	0.80
	49	0.02022	0.09	1.00	0.80

Table 7-2. (cont.)

	50	0.055656	0.09	1.00	0.80
	51	0.046973	0.09	1.00	0.80
	52	0.024834	0.09	1.00	0.80
	53	0.03439	0.09	1.00	0.80
	54	0.019596	0.09	1.00	0.80
	55	0.035687	0.09	1.00	0.80
	56	0.010799	0.09	1.00	0.80
	57	0.031755	0.09	1.00	0.80
	58	0.036237	0.09	1.00	0.80
	59	0.022735	0.09	1.00	0.80
	60	0.048223	0.05	1.00	0.80
	61	0.02797	0.05	1.00	0.80
	62	0.100618	0.05	1.00	0.80
	63	0.059244	0.05	1.00	0.80
	64	0.032228	0.05	1.00	0.80
	65	0.029606	0.05	1.00	0.80
	66	0.031586	0.05	1.00	0.80
	67	0.029969	0.05	1.00	0.80
	68	0.059911	0.05	1.00	0.80
	69	0.024702	0.05	1.00	0.80
	70	0.020472	0.05	1.00	0.80
	71	0.045268	0.05	1.00	0.80
	72	0.028789	0.05	1.00	0.80
	73	0.039873	0.05	1.00	0.80
	74	0.053942	0.03	1.00	0.80
	75	0.068794	0.03	1.00	0.80
	76	0.062532	0.03	1.00	0.80
	77	0.044543	0.03	1.00	0.80
1300E	78	0.044415	0.03	1.00	0.80

Table 7-3. Sediment and hyporheic transient storage (HTS) zones inputs for the Red Butte Creek models

Spring model	Value	Units
Sediment		
Thermal diffusivity	0.0064	cm^2/sec
Zone thickness	100	cm
Hyporheic		
Flow fraction	0.4	parameter for diffusive exchange
Sediment porosity - Fraction of volume	0.4	fraction of volume
Deep sediment temperature below	13	°C

Summer model	Value	Units
Sediment		
Thermal diffusivity	0.0064	cm^2/sec
Zone thickness	100	cm
Hyporheic		
Flow fraction	0.4	parameter for diffusive exchange
Sediment porosity - Fraction of volume	0.4	fraction of volume
Deep sediment temperature below	15	°C

Table 7-4. QUAL2Kw rates used in the Red Butte Creek models and are from (Neilson et al., 2012)

Parameter	**Value**	**Units**	**Min value**	**Max value**
Stoichiometry:				
Carbon	40	gC	30	60
Nitrogen	7.2	gN	5	9
Phosphorus	1	gP	0.5	2
Dry weight	100	gD	100	100
Chlorophyll	1	gA	0.5	2
Inorganic suspended solids:				
Settling velocity	2	m/d	0	2
Oxygen:				
Reaeration model	USGS(pool-riffle)			
User reaeration model parameter A	0		3	6
User reaeration model parameter B	0		0.5	1
User reaeration model parameter C	0		-1.85	-1.5
Temp correction for reaeration	1.024			
Reaeration wind effect	None			
O2 for carbon oxidation	2.67	gO2/gC		
O2 for NH4 nitrification	4.57	gO2/gN		
Oxygen inhib model CBOD oxidation	Exponential			
Oxygen inhib parameter CBOD oxidation	0.6	L/mgO2	0.6	0.6
Oxygen inhib model nitrification	Exponential			
Oxygen inhib parameter nitrification	0.6	L/mgO2	0.6	0.6
Oxygen enhance model denitrification	Exponential			
Oxygen enhance parameter denitrification	0.6	L/mgO2	0.6	0.6
Oxygen inhib model phyto resp	Exponential			
Oxygen inhib parameter phyto resp	0.6	L/mgO2	0.6	0.6
Oxygen enhance model bot alg resp	Exponential			
Oxygen enhance parameter bot alg resp	0.6	L/mgO2	0.6	0.6
Slow CBOD:				
Hydrolysis rate	0	/d	0.05	0.25
Temp correction	1.047		1	1.07
Oxidation rate	0.103	/d	0.05	0.25
Temp correction	1.047		1	1.07
Fast CBOD:				
Oxidation rate	10	/d	0	10
Temp correction	1.047		1	1.07
Organic N:				
Hydrolysis	0.364	/d	0.05	0.3
Temp correction	1.07		1	1.07
Settling velocity	0.016	m/d	0.05	0.25
Ammonium:				
Nitrification	8.44	/d	0.05	4
Temp correction	1.07		1	1.07
Nitrate:				
Denitrification	0.27	/d	0.05	2
Temp correction	1.07		1	1.07
Sed denitrification transfer coeff	0.00242	m/d	0	1
Temp correction	1.07		1	1.07
Organic P:				
Hydrolysis	0.69	/d	0.05	0.3
Temp correction	1.07		1	1.07
Settling velocity	0.06	m/d	0.05	0.25
Inorganic P:				
Settling velocity	0.16	m/d	0	2
Sed P oxygen attenuation half sat constant	0.01	mgO2/L	0	2

Table 7-4. (cont.)

Phytoplankton:				
Max Growth rate	2.71	/d	1.5	3
Temp correction	1.07		1	1.07
Respiration rate	0.11	/d	0.05	0.5
Temp correction	1.07		1	1.07
Death rate	0.12	/d	0	1
Temp correction	1		1	1.07
Nutrient limitation model for N and P				
Nitrogen half sat constant	15	ugN/L	10	25
Phosphorus half sat constant	2	ugP/L	1	5
Inorganic carbon half sat constant	0.000013	moles/L	1.3E-06	0.00013
Phytoplankton use HCO3- as substrate	Yes			
Light model	Smith			
Light constant	57.6	langleys/d	40	110
Ammonia preference	22.7	ugN/L	15	30
Settling velocity	0.03	m/d	0.05	0.5
Include transport of phytoplankton	No			
Nitrogen uptake water column fraction	0		0	1
Phosphorus uptake water column fraction	0		0	1
Bottom Plants:				
Growth model	Zero-order			
Max Growth rate	48.4	gD/m2/d or /d	1.5	200
Temp correction	1.07		1	1.07
First-order model carrying capacity	100	gD/m2	50	200
Basal respiration rate	0.204	/d	0.02	0.2
Photo-respiration rate parameter	0.01	unitless	0	0.6
Temp correction	1.07		1	1.07
Excretion rate	0.0666	/d	0	0.5
Temp correction	1.07		1	1.07
Death rate	0.135	/d	0	5
Temp correction	1.07		1	1.07
Scour function				
Coefficient of scour function	0	/d/cms or /d/mps	0	0.1
Exponent of scour function	0		0	2
Minimal biomass after scour event	0	gD/m^2	0	10
Catastrophic scour rate during flood event	0	/d	0	100
Critical flow or vel for catastrophic scour	0	cms or m/s	0	50
External nitrogen half sat constant	172	ugN/L	100	500
External phosphorus half sat constant	25.6	ugP/L	25	100
Inorganic carbon half sat constant	0.0000379	moles/L	1.3E-06	0.00013
Bottom algae use HCO3- as substrate	Yes			
Light model	Half saturation			
Light constant	80.4	langleys/d	40	100
Ammonia preference	12.9	ugN/L	15	30
Nutrient limitation model for N and P				
Subsistence quota for nitrogen	48.5	mgN/gD	0.36	1.44
Subsistence quota for phosphorus	0.58	mgP/gD	0.05	5
Maximum uptake rate for nitrogen	817	mgN/gD/d	350	1500
Maximum uptake rate for phosphorus	11.2	mgP/gD/d	50	200
Internal nitrogen half sat ratio	4.83		1.05	5
Internal phosphorus half sat ratio	1.06		1.05	5
Nitrogen uptake water column fraction	1		0	1
Phosphorus uptake water column fraction	1		0	1
Detritus (POM):				
Dissolution rate	1.785	/d	0.05	0.5
Temp correction	1.07		1.07	1.07

Table 7-4. (cont.)

Settling velocity	0.63	m/d	0.05	0.5
Pathogens:				
Decay rate	0	/d	0	20
Temp correction	1.07		1	1.07
Settling velocity	0	m/d	0	2
alpha constant for light mortality	0	/d per ly/hr	0	1
pH:				
Partial pressure of carbon dioxide	0	ppm		
Hyporheic metabolism				
Model for biofilm oxidation of fast CBOD				
Max biofilm growth rate	0	gO2/m^2/d or /d	0	20
Temp correction	1.047		1.047	1.047
Fast CBOD half-saturation	0	mgO2/L	0	2
Oxygen inhib model				
Oxygen inhib parameter	0	mgO2/L	0.6	0.6
Respiration rate	0	/d	0.2	0.2
Temp correction	0		1.07	1.07
Death rate	0	/d	0.05	0.05
Temp correction	0		1.07	1.07
External nitrogen half sat constant	0	ugN/L	15	15
External phosphorus half sat constant	0	ugP/L	2	2
Ammonia preference	0	ugN/L	25	25
First-order model carrying capacity	0	gD/m2	100	100
Generic constituent				
Decay rate	0	/d	0	20
Temp correction	1.07		1	1.07
Settling velocity	0	m/d	0	2
Use generic constituent as COD?				
Photosynthetic quotient and respiratory quotient for phytoplankton and bottom algae				
Photosynthetic quotient for NO3 vs NH4 use	1.289719626	dimensionless	1.2	1.8
Respiratory quotient	1	dimensionless	0.85	1

Light and Heat

Parameter	**Value**	**Unit**
Photosynthetically Available Radiation	0.47	
Background light extinction	0.2	*/m*
Linear chlorophyll light extinction	0.0088	*1/m-(ugA/L)*
Nonlinear chlorophyll light extinction	0.054	*1/m-(ugA/L)$^{2/3}$*
ISS light extinction	0.052	*1/m-(mgD/L)*
Detritus light extinction	0.174	*1/m-(mgD/L)*
Macrophyte light extinction	0.015	*1/m-(gD/m^3)*
Solar shortwave radiation		
Atmospheric attenuation model for solar	Observed	
Bras solar parameter (used if Bras solar model is selected)		
atmospheric turbidity coefficient (2=clear, 5=smoggy, default=2)	2	
Ryan-Stolzenbach solar parameter (used if Ryan-Stolzenbach solar model is selected)		
atmospheric transmission coefficient (0.70-0.91, default 0.8)	0.8	
Downwelling atmospheric longwave IR radiation		
atmospheric longwave emissivity model	Brutsaert	
Brutsaert longwave emissivity parameter (used if Brutsaert longwave model is selected)		
parameter for emissivity using the Brutsaert equation	1.31	
Evaporation and air convection/conduction		
wind speed function for evaporation and air convection/conduction	Brady-Graves-Geyer	
Parameters for attenuation of solar radiation by cloud cover		
coefficient for attenuation of solar radiation by cloud cover	0.65	
exponent for attenuation of solar radiation by cloud cover	2	
Model and parameters for cloud cover adjustment of longwave radiation		

Table 7-4. (cont.)

model equation for cloudy sky adjustment of longwave radiation	Eqn 1	
coefficient for cloudy sky adjustment of longwave radiation	0.17	
exponent for cloudy sky adjustment of longwave radiation	2	
Include evaporation in flow balance		
Include evaporation in flow balance	No	

Table 7-5. Percent changes from base case for Flow, Ts, DO, and TP across alternatives for the spring Red Butte Creek model

spring model	**Percentage Change from Base Case (%)**			
Site Alternatives	Flow	Ts	DO	TP
10% GS	-1.2	0.1	0.04	-3.0
10% BC	-1.5	0.1	0.04	-0.9
10% RG	-1.6	0.1	0.01	-0.5
10% GS, BC, RG	-4.3	0.3	0.1	-4.3
50% GS	-3.0	0.2	0.0	-3.6
50% BC	-3.5	0.2	0.1	-4.4
50% RG	-4.3	0.3	0.03	-1.1
50% GS, BC, RG	-10.7	0.6	0.1	-8.8
100% GS	-3.8	0.2	0.1	-4.8
100% BC	-4.5	0.3	0.2	-5.8
100% RG	-5.6	0.3	0.04	-2.5
100% GS, BC, RG	-13.9	0.9	0.8	-12.3
Red Butte Watershed Alternatives				
10% high estimate	-4.4	0.3	0.1	-4.9
10% low estimate	-1.1	0.1	0.05	-2.8
50% high estimate	-11.1	0.6	0.5	-10.1
50% low estimate	-5.7	0.3	0.1	-6.1
100% high estimate	-14.3	0.9	0.8	-13.3
100% low estimate	-13.6	0.8	0.7	-11.4

Table 7-6. Percent changes from base case for Flow, Ts, DO, and TP across alternatives for the summer Red Butte Creek model

summer model	**Percentage Change from Base Case (%)**			
Site Alternatives	Flow	Ts	DO	TP
10% GS	-1.0	0.02	-0.04	-3.7
10% BC	-1.2	0.03	-0.05	-0.6
10% RG	-1.3	0.04	-0.04	0.04
10% GS, BC, RG	-3.5	0.1	-0.1	-4.0
50% GS	-2.4	0.07	-0.07	-3.6
50% BC	-2.9	0.09	-0.08	-4.4
50% RG	-3.5	0.1	-0.1	0.3
50% GS, BC, RG	-8.9	0.3	-0.3	-6.6
100% GS	-3.2	0.1	-0.09	-4.8
100% BC	-3.8	0.1	-0.1	-5.8
100% RG	-4.6	0.1	-0.1	-0.9
100% GS, BC, RG	-11.6	0.3	-0.3	-8.6
Red Butte Watershed Alternatives				
10% high estimate	-3.6	0.1	-0.1	-4.7
10% low estimate	-0.9	0.02	-0.04	-3.4
50% high estimate	-9.1	0.3	-0.3	-7.6
50% low estimate	-4.6	0.1	-0.1	-5.6
100% high estimate	-11.8	0.3	-0.3	-9.2
100% low estimate	-11.2	0.3	-0.3	-7.4

Table 7-7. Point Sources in the August Jordan River model starting at Little Cottonwood Creek. (Stantec Consulting Ltd., 2010)
*Bold: Seven canyon creeks evaluated in this study

Point Sources	Location (km)	Point Abstraction (cms)	Point Inflow (cms)	Temperature (mean °C)	Dissolved Oxygen (mean mg/L)	Organic P (mean ug/L)	Inorganic P (mean ug/L)	Phytoplankton (mean ug/L)
Little Cottonwood Creek	34.7	0	0.29	18.7	9.2	9	64	25.7
Brighton Canal	34.1	0.85	0	0	0	0	0	0
SW - JOR 17.07	34.08590592	0	0	0	0	0	0	0
SW - JOR 16.98	33.8766912	0	0	0	0	0	0	0
SW 4700 S Drain - JOR 16.85	33.76403712	0	0	0	0	0	0	0
SW - JOR 16.98	33.6352896	0	0	0	0	0	0	0
SW - JOR 16.54	33.3134208	0	0	0	0	0	0	0
Big Cottonwood Creek	33.2	0	1.216	18.1	10.1	41	25	22
SW - JOR 16.16	32.70187008	0	0	0	0	0	0	0
SW - JOR 15.53	31.7040768	0	0	0	0	0	0	0
SW 4100 S Drain - JOR 15.31	31.4626752	0	0	0	0	0	0	0
SW - JOR 14.56	30.22348032	0	0	0	0	0	0	0
SW - JOR 13.72	29.5314624	0	0	0	0	0	0	0
SW - JOR 13.63	29.370528	0	0	0	0	0	0	0
SW - JOR 13.49	29.2095936	0	0	0	0	0	0	0
SW - JOR 13.40	29.0486592	0	0	0	0	0	0	0
SW - JOR 12.78	28.03477248	0	0	0	0	0	0	0
SW - JOR 12.71	27.9221184	0	0	0	0	0	0	0
Mill Creek/Central Valley WWTP	27.7	0	4.033	20.75	8.3	193	2415	4.9
Placeholder - Central Valley WWTP	27.6	0	0	0	0	0	0	0
Kearns-Chesterfield Drain - JOR 12.10	27.358848	0	0.277	19	5.87	0	0	0
SW - JOR 11.92	27.0369792	0	0	0	0	0	0	0
SW - JOR 11.42	26.2323072	0	0	0	0	0	0	0

Table 7-7. (cont.)

SW - JOR 11.17 - 2100 S Drain	25.8299712	0	0	0	0	0	0	0
Surplus Canal	25.8	7.41	0	0	0	0	0	0
SW - JOR 10.70	25.07357952	0	0	0	0	0	0	0
SW - JOR 10.17	24.28500096	0	0	0	0	0	0	0
1300 S Conduit	22.9	0	0.274	15.85	8.523333333	23.82667	17.156667	9.043333333
SW - JOR 08.32	21.2433408	0	0	0	0	0	0	0
SW - JOR 08.06	20.82491136	0	0	0	0	0	0	0
800 S Drain - JOR 07.99	20.7605376	0	0	0	0	0	0	0
SW - JOR 07.68	20.21336064	0	0	0	0	0	0	0
600 S Drain - JOR 07.67	20.1972672	0	0	0	0	0	0	0
SW - JOR 07.22	19.4730624	0	0	0	0	0	0	0
SW - JOR 07.00	19.10291328	0	0	0	0	0	0	0
UP&L Diversion	18.6	0	0	0	0	0	0	0
N Temple Conduit	18.35	0	0.056	20	8.4	15	16	0.6
SW - JOR 05.46	16.60843008	0	0	0	0	0	0	0
SW - JOR 04.60	15.22439424	0	0	0	0	0	0	0
SW - JOR 03.90	14.11394688	0	0	0	0	0	0	0
South Davis South WWTP	7.8	0	0.106	22.3	8	173	1518	8.2
State Canal	2.7	0	0	0	0	0	0	0

Table 7-8. Diffuse sources in the August Jordan River model (Stantec Consulting Ltd., 2010)

Diffuse Sources	Up (km)	Down (km)	Diffuse abstraction (cms)	Diffuse inflow (cms)	Temp (°C)	Dissolved Oxygen (mg/L)	Organic P (ug/L)	Inorganic P (ug/L)	Phytoplankton (ug/L)
Segment 8	82.7	67.5	0	0.364	16	0	50	10	0
Segment 7	67.5	60.5	0	0.608	16	0	50	10	0
Segment 6	60.5	42.5	0	2.298	16	0	50	10	0
Segment 5	42.5	40	0	0.271	16	0	50	10	0
Segment 4	40	25.5	0	0.403	16	0	50	10	0
Segment 3	25.5	18.5	0	0.465	16	0	50	10	0
Segment 2	18.5	11.5	0	0	16	0	50	10	0
Segment 1	11.5	0	0	0	16	0	50	10	0
GW Exchange	41.5	31.5	0	2.5	16	0	50	10	0
GW Exchange	41.5	31.5	2.5	0	0	0	0	0	0

Appendix B – Chapter 4 Supplementary Materials

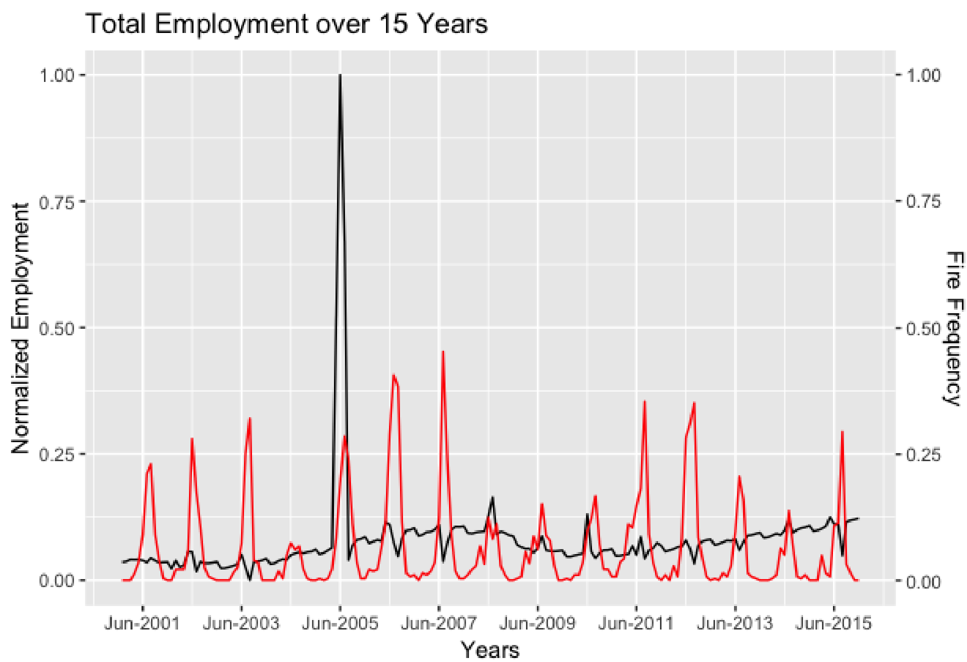


Figure 7-6. Normalized Total employment and fire frequency for the IMW from 2001-2015.

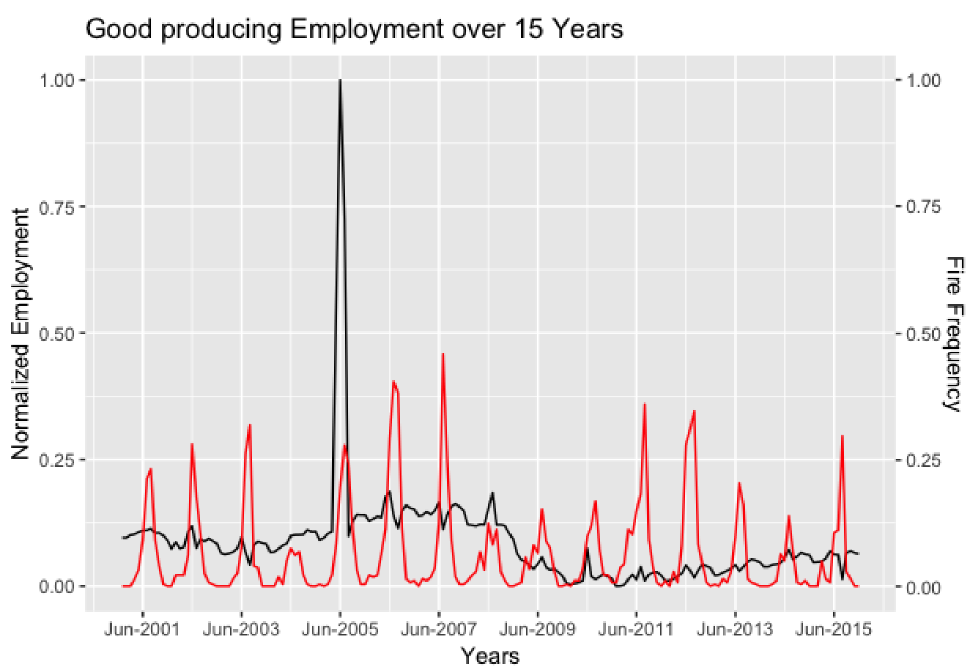


Figure 7-7. Normalized Goods-Producing employment and fire frequency for the IMW from 2001-2015.

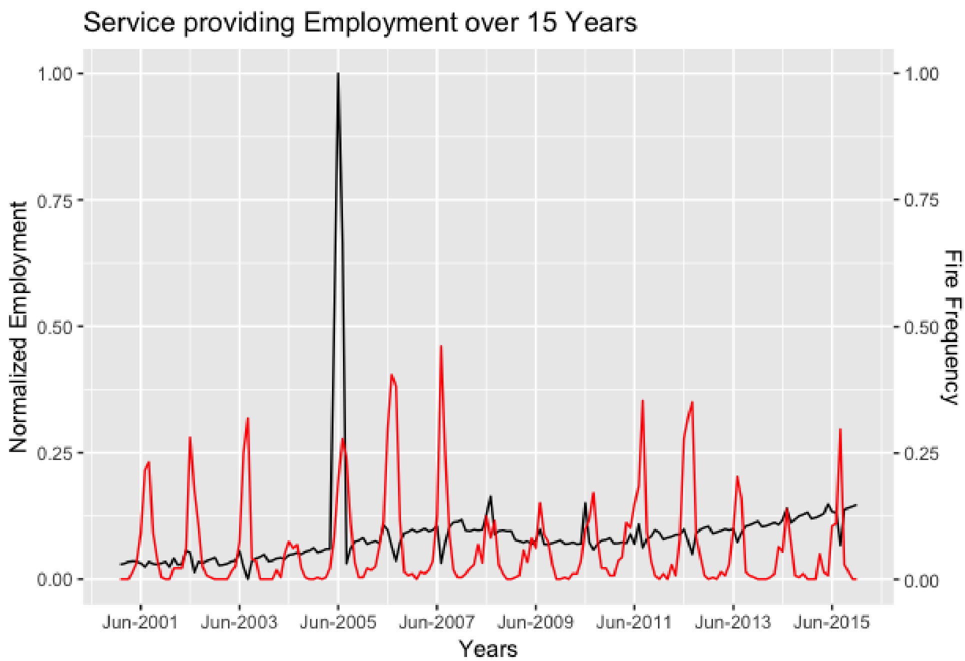


Figure 7-8. Normalized Service-Providing employment and fire frequency for the IMW from 2001-2015.

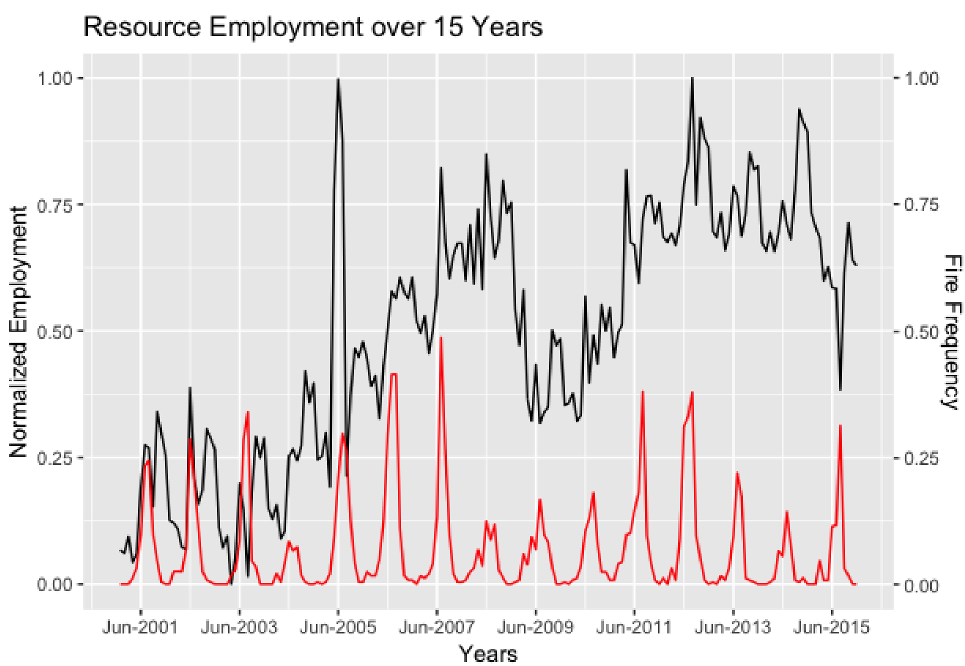


Figure 7-9. Normalized Natural Resource and Mining employment and fire frequency for the IMW from 2001-2015.

Figure 7-10. Normalized Leisure and Hospitality employment and fire frequency for the IMW from 2001-2015.

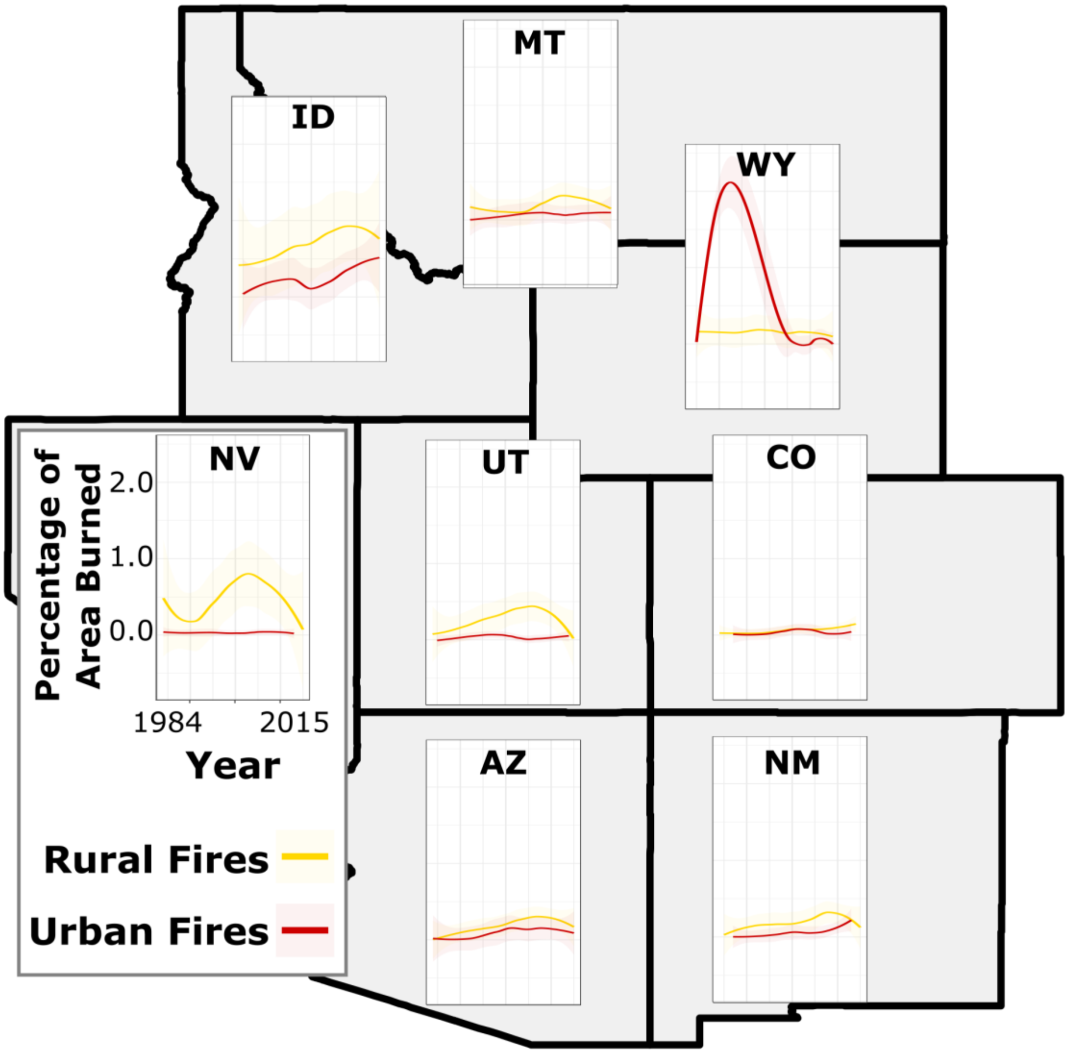


Figure 7-11. State-level LOESS curves in percentage of area burned for rural and urban fires.

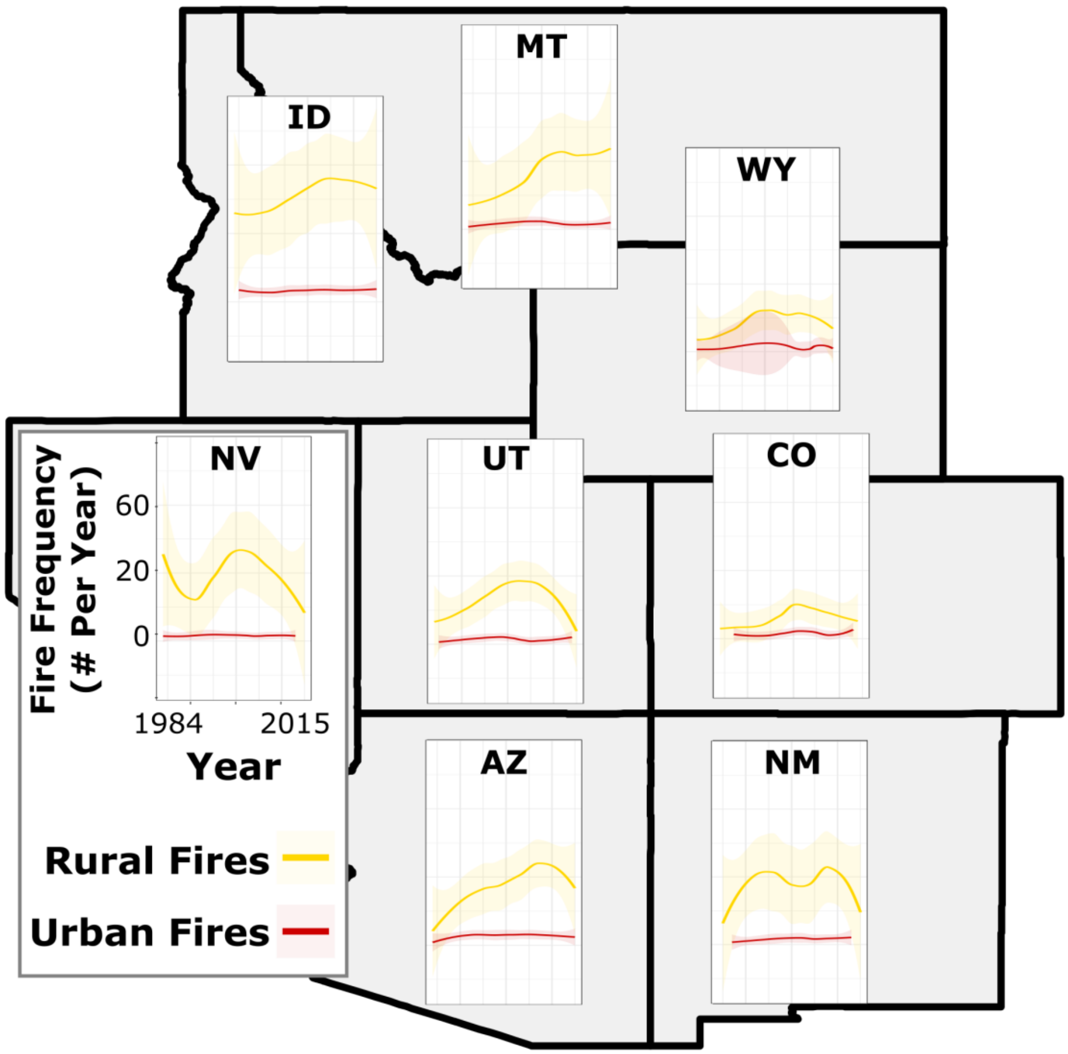


Figure 7-12. State-level LOESS curves in fire frequency for rural and urban fires.

Table 7-9. Regression results for (I) Total Employment for the 12-month window post-fire for years 2001-2015 (*p<0.1; **p<0.05; ***p<0.01). The first column presents the results for All Fires within all 281 IMW counties (44,666 observations), the second column represents the results for Rural Fires (44,360 observations), the third column represents the results for Urban Fires (41,429 observations), and the last column represents the results for the 14 Increasing Focal Counties (2,274 observations). Effects of fires on employment are presented in percentages. The standard error for each regression is presented in parentheses

	Dependent variable			
	Effects of Fires on Employment (%)			
	All Fires	Rural Fires	Urban Fires	Increasing Focal Counties
Fire Happened	0.005*	0.005	-0.001	0.020***
	(0.003)	(0.003)	(0.008)	(0.007)
1 Months After	0.005**	0.006**	-0.006	0.010
	(0.003)	(0.003)	(0.007)	(0.006)
2 Months After	0.005**	0.005*	-0.001	0.010
	(0.003)	(0.003)	(0.007)	(0.006)
3 Months After	0.004	0.004	0.0003	0.002
	(0.003)	(0.003)	(0.007)	(0.006)
4 Months After	0.005*	0.004	0.002	0.002
	(0.003)	(0.003)	(0.007)	(0.007)
5 Months After	0.002	0.002	0.005	-0.005
	(0.003)	(0.003)	(0.007)	(0.007)
6 Months After	0.001	0.001	0.006	0.0002
	(0.003)	(0.003)	(0.007)	(0.006)
7 Months After	0.001	0.001	0.003	-0.0003
	(0.003)	(0.003)	(0.007)	(0.007)
8 Months After	0.003	0.003	0.006	0.0001
	(0.003)	(0.003)	(0.007)	(0.007)
9 Months After	0.002	0.002	0.003	0.005
	(0.003)	(0.003)	(0.007)	(0.007)
10 Months After	0.002	0.003	0.0004	-0.002
	(0.003)	(0.003)	(0.007)	(0.007)
11 Months After	0.001	0.003	-0.005	-0.003
	(0.003)	(0.003)	(0.007)	(0.007)
12 Months After	0.003	0.004	-0.004	0.008
	(0.003)	(0.003)	(0.007)	(0.007)
Observations	44,666	44,360	41,429	2,274
R^2	0.996	0.996	0.996	0.996
Adjusted R^2	0.996	0.996	0.996	0.996
Residual Std. Error	0.115	0.115	0.116	0.100
	[df=44,333]	[df=44,027]	[df=41,097]	[df=2,208]

Table 7-10. Regression results of the (1) Goods Producing sector for the 12-month window post-fire for years 2001-2015 (*p<0.1; **p<0.05; ***p<0.01). Effects of fires on employment are presented in percentages. The standard error for each regression is presented in parentheses

	Dependent variable			
	Effects of Fires on Employment (%)			
	All Fires	Rural Fires	Urban Fires	Increasing Focal Counties
Fire Happened	0.007	0.009	0.003	0.032^{***}
	(0.006)	(0.006)	(0.015)	(0.012)
1 Months After	0.010^{**}	0.011^{**}	-0.00004	0.010
	(0.005)	(0.005)	(0.014)	(0.010)
2 Months After	0.007	0.008	0.002	0.013
	(0.005)	(0.005)	(0.014)	(0.011)
3 Months After	0.004	0.005	0.004	0.012
	(0.005)	(0.005)	(0.014)	(0.011)
4 Months After	0.007	0.008	0.008	0.015
	(0.005)	(0.005)	(0.014)	(0.011)
5 Months After	0.005	0.006	0.009	0.003
	(0.005)	(0.005)	(0.014)	(0.011)
6 Months After	0.005	0.005	0.005	-0.0002
	(0.005)	(0.005)	(0.014)	(0.011)
7 Months After	0.002	0.003	-0.004	-0.003
	(0.005)	(0.005)	(0.014)	(0.011)
8 Months After	0.005	0.005	0.006	-0.008
	(0.005)	(0.006)	(0.014)	(0.011)
9 Months After	0.004	0.005	0.003	0.011
	(0.005)	(0.006)	(0.014)	(0.011)
10 Months After	0.007	0.007	0.007	0.013
	(0.005)	(0.006)	(0.014)	(0.011)
11 Months After	0.007	0.008	0.005	0.008
	(0.005)	(0.006)	(0.014)	(0.011)
12 Months After	0.009^{*}	0.010^{*}	0.004	0.018^{*}
	(0.005)	(0.006)	(0.014)	(0.011)
Observations	44,165	43,877	40,966	2,209
R^2	0.984	0.984	0.984	0.977
Adjusted R^2	0.984	0.984	0.984	0.977
Residual Std. Error	0.222	0.223	0.224	0.166
	[df=43,832]	[df=43,544]	[df=40,635]	[df=2,143]

Table 7-11. Regression results of the (2) Service Providing sector for the 12-month window post-fire for years 2001-2015 (*p<0.1; **p<0.05; ***p<0.01). Effects of fires on employment are presented in percentages. The standard error for each regression is presented in parentheses

	Dependent variable			
	Effects of Fires on Employment (%)			
	All Fires	Rural Fires	Urban Fires	Increasing Focal Counties
Fire Happened	0.002	0.003	-0.004	-0.002
	(0.003)	(0.003)	(0.008)	(0.007)
1 Months After	0.004*	0.005*	-0.009	0.008
	(0.003)	(0.003)	(0.007)	(0.006)
2 Months After	0.004	0.005	-0.006	0.003
	(0.003)	(0.003)	(0.007)	(0.006)
3 Months After	0.002	0.004	-0.004	-0.002
	(0.003)	(0.003)	(0.007)	(0.006)
4 Months After	0.002	0.002	-0.002	0.00003
	(0.003)	(0.003)	(0.007)	(0.006)
5 Months After	-0.0004	-0.0002	0.001	-0.008
	(0.003)	(0.003)	(0.007)	(0.006)
6 Months After	-0.001	-0.002	0.003	-0.003
	(0.003)	(0.003)	(0.007)	(0.006)
7 Months After	0.001	0.001	0.003	0.0002
	(0.003)	(0.003)	(0.008)	(0.006)
8 Months After	0.002	0.001	0.003	-0.001
	(0.003)	(0.003)	(0.008)	(0.006)
9 Months After	0.003	0.002	0.003	0.002
	(0.003)	(0.003)	(0.008)	(0.006)
10 Months After	0.0002	0.001	-0.001	-0.005
	(0.003)	(0.003)	(0.008)	(0.006)
11 Months After	0.0002	0.001	-0.007	-0.006
	(0.003)	(0.003)	(0.008)	(0.006)
12 Months After	0.001	0.001	-0.007	-0.004
	(0.003)	(0.003)	(0.008)	(0.006)
Observations	44,177	43,873	40,955	2,248
R^2	0.996	0.996	0.996	0.997
Adjusted R^2	0.996	0.996	0.996	0.997
Residual Std. Error	0.116	0.115	0.117	0.095
	[df=43,844]	[df=43,540]	[df=40,623]	[df=2,182]

Table 7-12. Regression results of the (1a) Good Producing: Natural Resource and Mining sector for the 12-month window post-fire for years 2001-2015 (*p<0.1; **p<0.05; ***p<0.01). Effects of fires on employment are presented in percentages. The standard error for each regression is presented in parentheses

	Dependent variable			
	Effects of Fires on Employment (%)			
	All Fires	Rural Fires	Urban Fires	Increasing Focal Counties
Fire Happened	0.006	0.004	-0.006	0.072***
	(0.008)	(0.008)	(0.021)	(0.019)
1 Months After	-0.002	0.001	-0.011	-0.007
	(0.007)	(0.007)	(0.020)	(0.016)
2 Months After	-0.001	-0.0004	-0.011	0.012
	(0.007)	(0.008)	(0.020)	(0.016)
3 Months After	-0.003	-0.004	-0.015	0.003
	(0.007)	(0.008)	(0.020)	(0.017)
4 Months After	0.004	0.003	-0.024	0.014
	(0.007)	(0.008)	(0.020)	(0.017)
5 Months After	0.002	0.002	-0.022	0.006
	(0.007)	(0.008)	(0.020)	(0.017)
6 Months After	-0.001	0.002	-0.023	-0.018
	(0.007)	(0.008)	(0.020)	(0.017)
7 Months After	-0.009	-0.007	-0.032	-0.026
	(0.007)	(0.008)	(0.020)	(0.017)
8 Months After	-0.003	-0.003	-0.021	-0.011
	(0.007)	(0.008)	(0.020)	(0.017)
9 Months After	-0.004	-0.007	-0.022	0.006
	(0.007)	(0.008)	(0.020)	(0.017)
10 Months After	0.001	-0.003	-0.014	0.020
	(0.008)	(0.008)	(0.020)	(0.017)
11 Months After	0.001	0.002	-0.007	0.003
	(0.008)	(0.008)	(0.020)	(0.017)
12 Months After	0.007	0.007	-0.009	0.046***
	(0.008)	(0.008)	(0.020)	(0.017)
Observations	39,406	39,112	36,346	2,181
R^2	0.953	0.954	0.953	0.950
Adjusted R^2	0.952	0.953	0.953	0.948
Residual Std. Error	0.306	0.304	0.305	0.252
	[df=39,082]	[df=38,788]	[df=36,023]	[df=2,116]

Table 7-13. Regression results of the (2a) Service Providing: Leisure and Hospitality sector for the 12-month window post-fire for years 2001-2015 (*p<0.1; **p<0.05; ***p<0.01). Effects of fires on employment are presented in percentages. The standard error for each regression is presented in parentheses

	Dependent variable			
	Effects of Fires on Employment (%)			
	All Fires	Rural Fires	Urban Fires	Increasing Focal Counties
Fire Happened	0.00002	0.001	-0.013	-0.015
	(0.005)	(0.005)	(0.013)	(0.010)
1 Months After	0.005	0.005	-0.031**	0.009
	(0.004)	(0.004)	(0.012)	(0.008)
2 Months After	0.006	0.008	-0.017	-0.005
	(0.005)	(0.005)	(0.012)	(0.009)
3 Months After	0.003	0.003	-0.010	-0.001
	(0.005)	(0.005)	(0.012)	(0.009)
4 Months After	0.001	0.0003	-0.010	-0.006
	(0.005)	(0.005)	(0.012)	(0.009)
5 Months After	0.0001	-0.001	0.0003	-0.015*
	(0.005)	(0.005)	(0.012)	(0.009)
6 Months After	0.0001	-0.0004	0.012	-0.003
	(0.005)	(0.005)	(0.012)	(0.009)
7 Months After	0.001	0.001	0.013	-0.003
	(0.005)	(0.005)	(0.013)	(0.009)
8 Months After	0.0001	-0.001	0.002	-0.001
	(0.005)	(0.005)	(0.013)	(0.009)
9 Months After	0.0001	-0.0002	0.003	-0.0005
	(0.005)	(0.005)	(0.013)	(0.009)
10 Months After	-0.006	-0.006	-0.014	-0.005
	(0.005)	(0.005)	(0.013)	(0.009)
11 Months After	-0.006	-0.006	-0.024*	-0.010
	(0.005)	(0.005)	(0.013)	(0.009)
12 Months After	-0.002	-0.002	-0.022*	-0.006
	(0.005)	(0.005)	(0.013)	(0.009)
Observations	43,967	43,699	40,772	2,242
R^2	0.989	0.989	0.989	0.994
Adjusted R^2	0.989	0.989	0.988	0.994
Residual Std. Error	0.195	0.194	0.195	0.136
	[df=43,635]	[df=43,367]	[df=40,441]	[df=2,176]

Appendix C – Chapter 5 Supplementary Materials

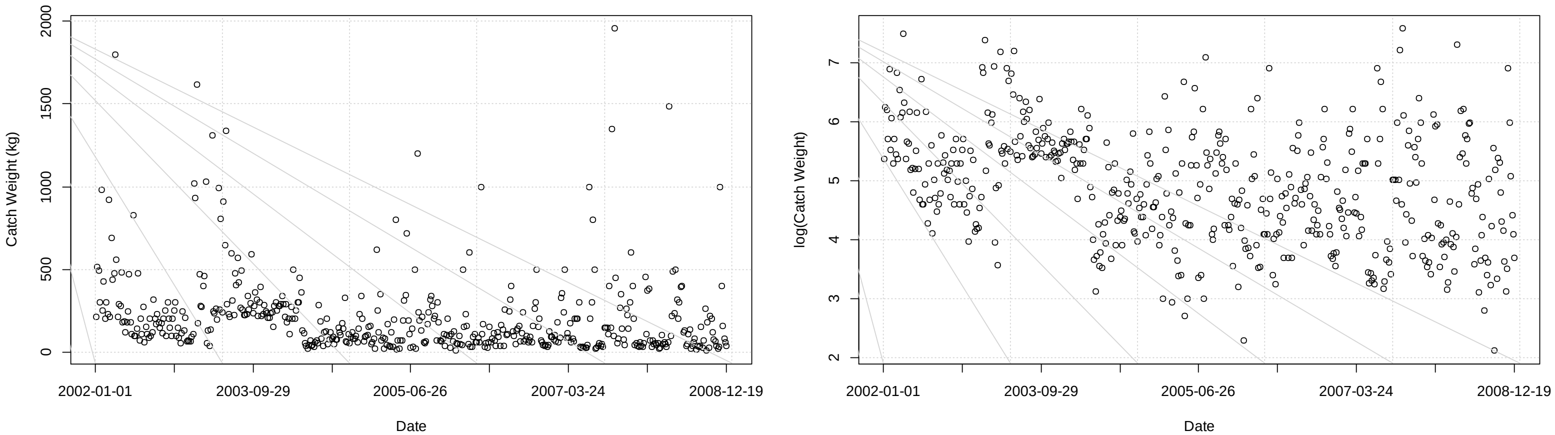


Figure 7-13. Catch weight in kg (left) and log-transformed catch weight (right) over time for *Pangasianodon hypophthalmus.*

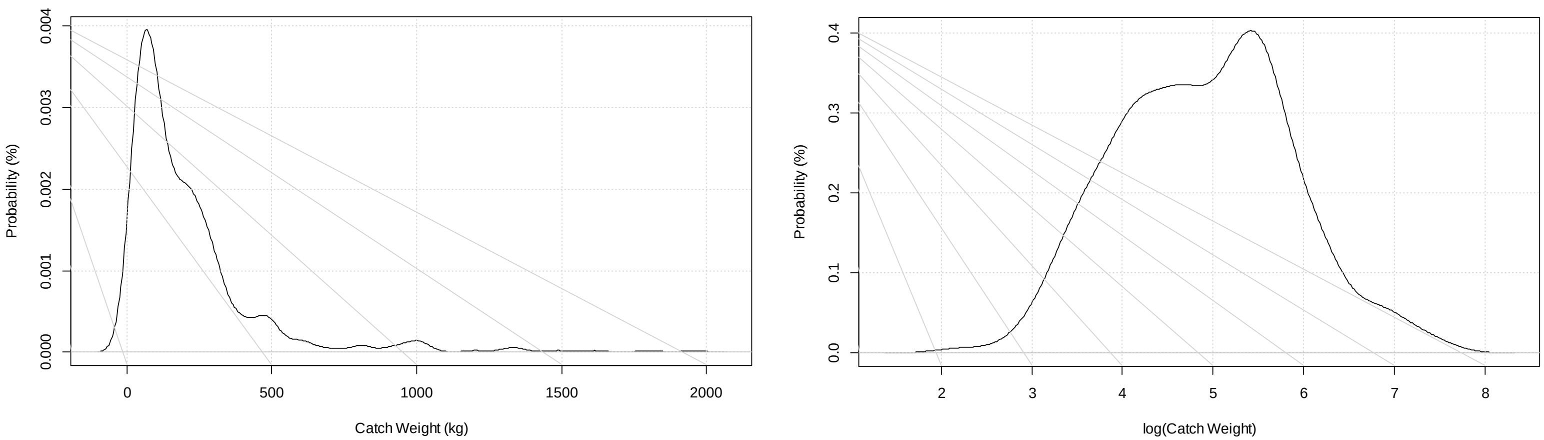


Figure 7-14. Distribution of catch weight in kg (left) and log-transformed catch weight (right) for *Pangasianodon hypophthalmus.*

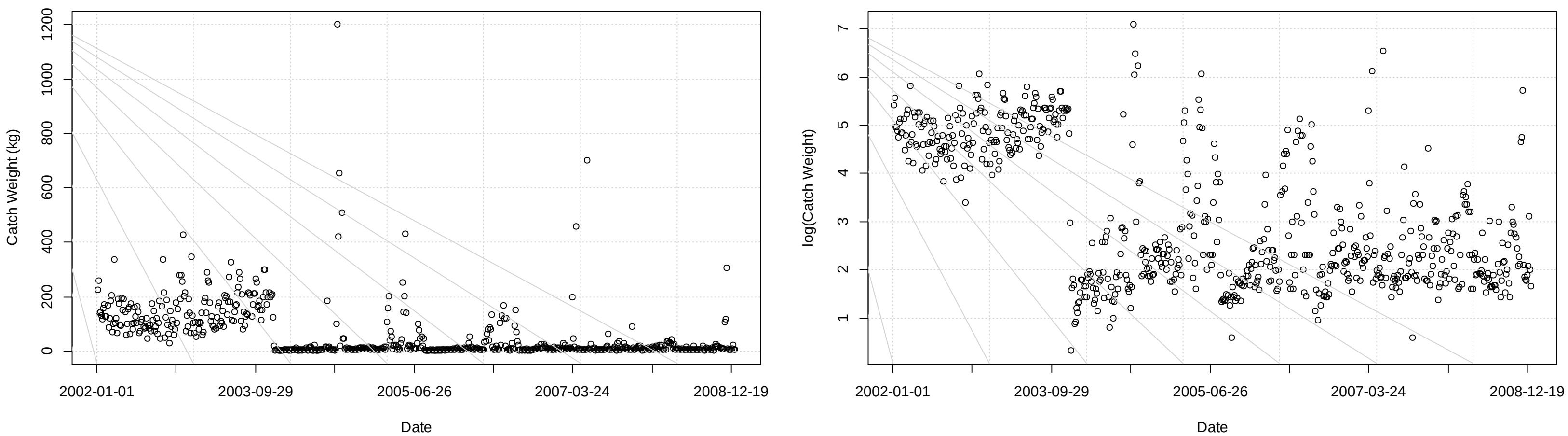


Figure 7-15. Catch weight in kg (left) and log-transformed catch weight (right) over time for *Cyclocheilchthys enoplos*.

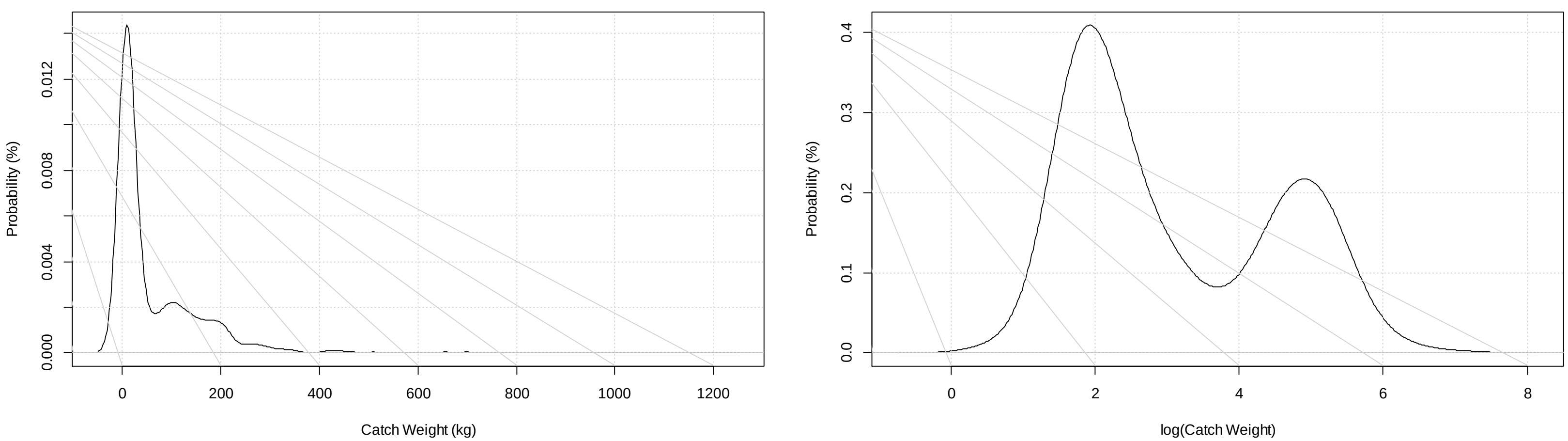


Figure 7-16. Distribution of catch weight in kg (left) and log-transformed catch weight (right) for *Cyclocheilchthys enoplos*.

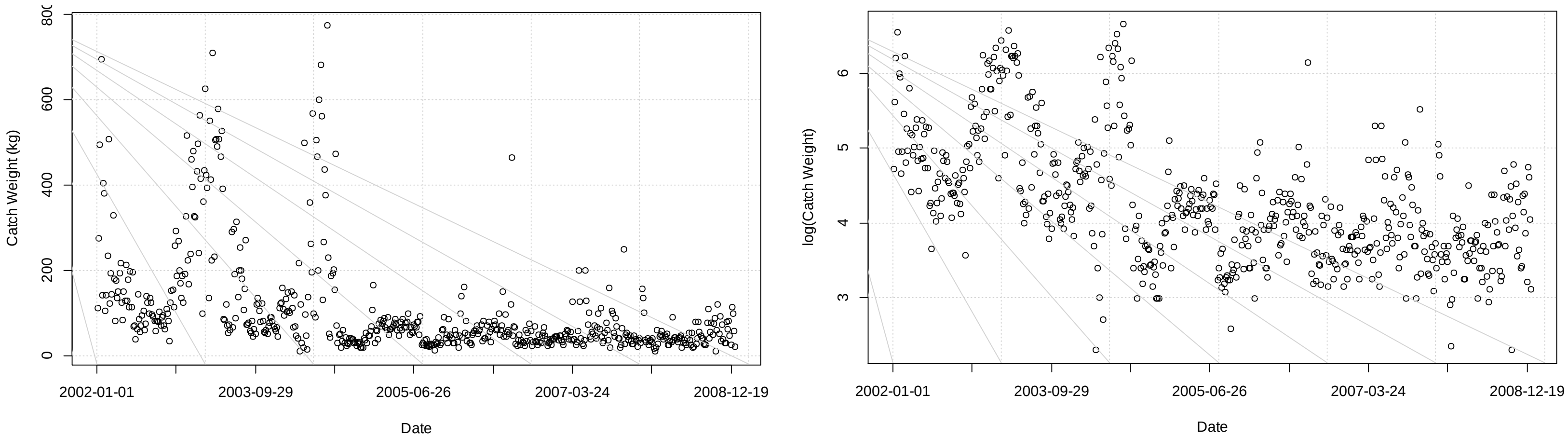


Figure 7-17. Catch weight in kg (left) and log-transformed catch weight (right) over time for *Cirrhinus microlepis*.

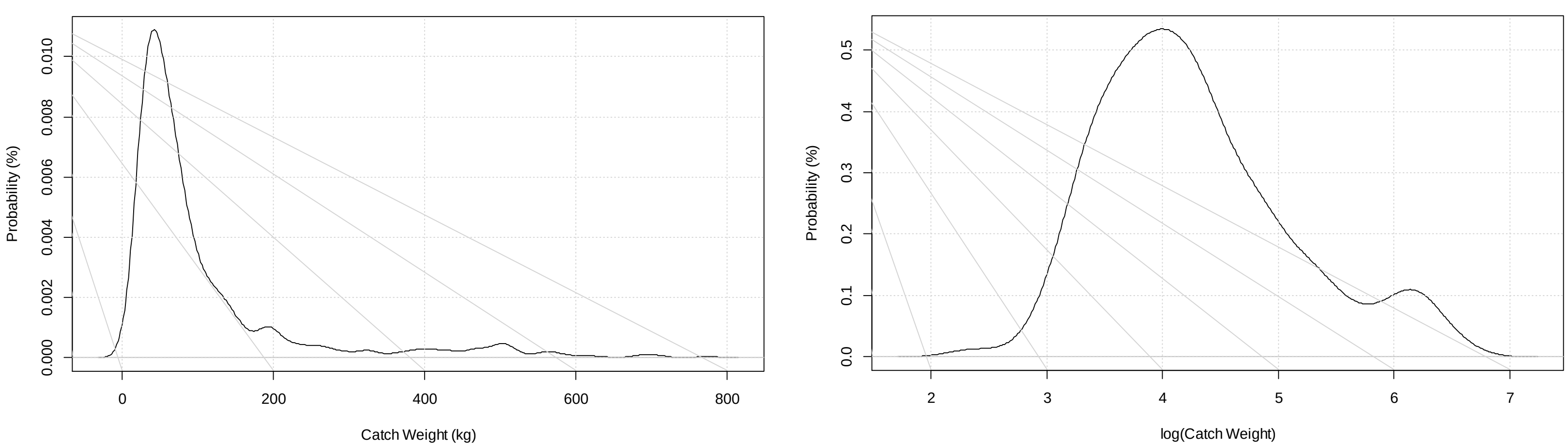


Figure 7-18. Distribution of catch weight in kg (left) and log-transformed catch weight (right) for *Cirrhinus microlepis*.

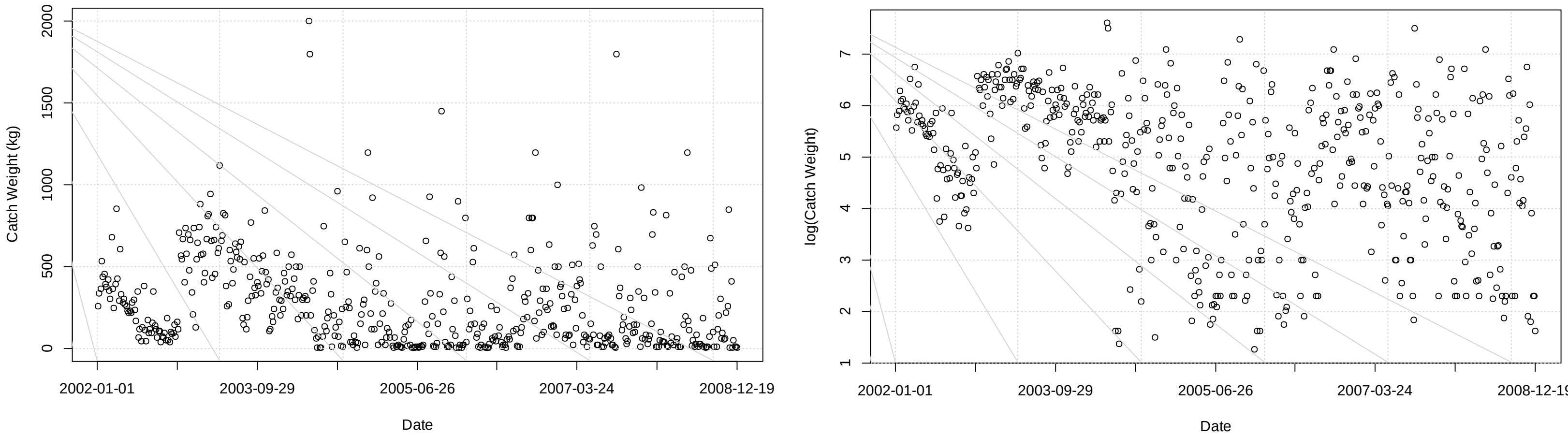


Figure 7-19. Catch weight in kg (left) and log-transformed catch weight (right) over time for *Osteochilus melanopleurus*.

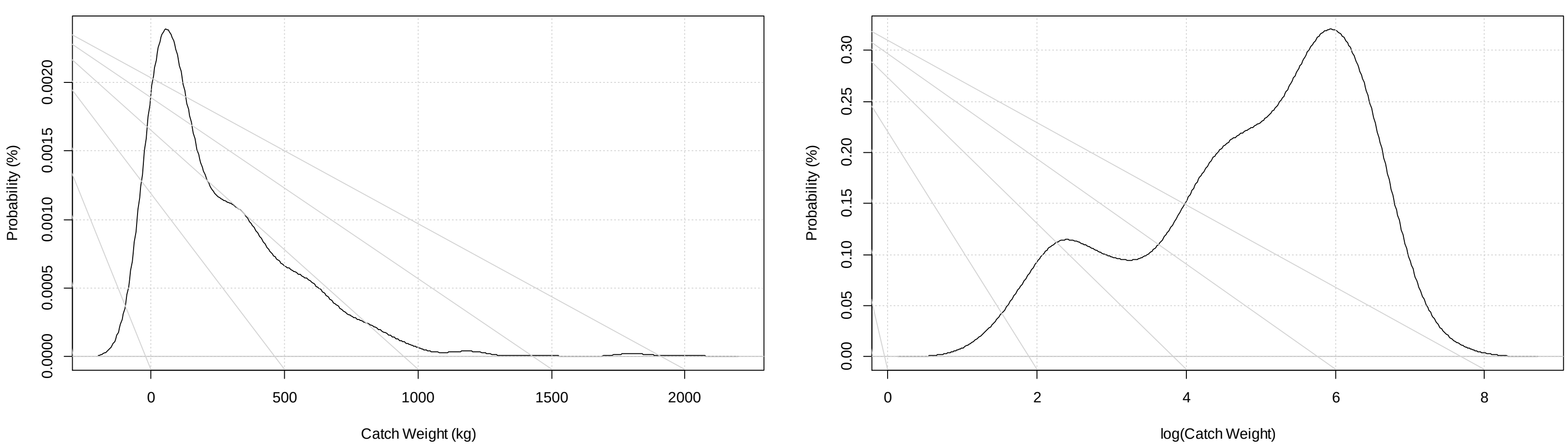


Figure 7-20. Distribution of catch weight in kg (left) and log-transformed catch weight (right) for *Osteochilus melanopleurus*.

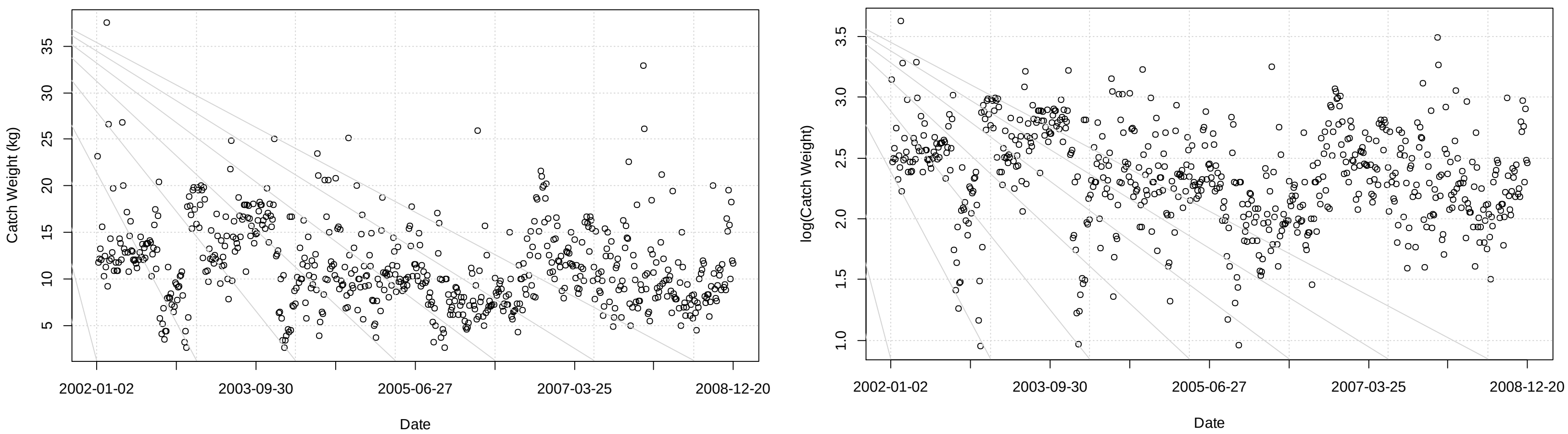


Figure 7-21. Catch weight in kg (left) and log-transformed catch weight (right) over time for *Henicorhynchus lobatus*.

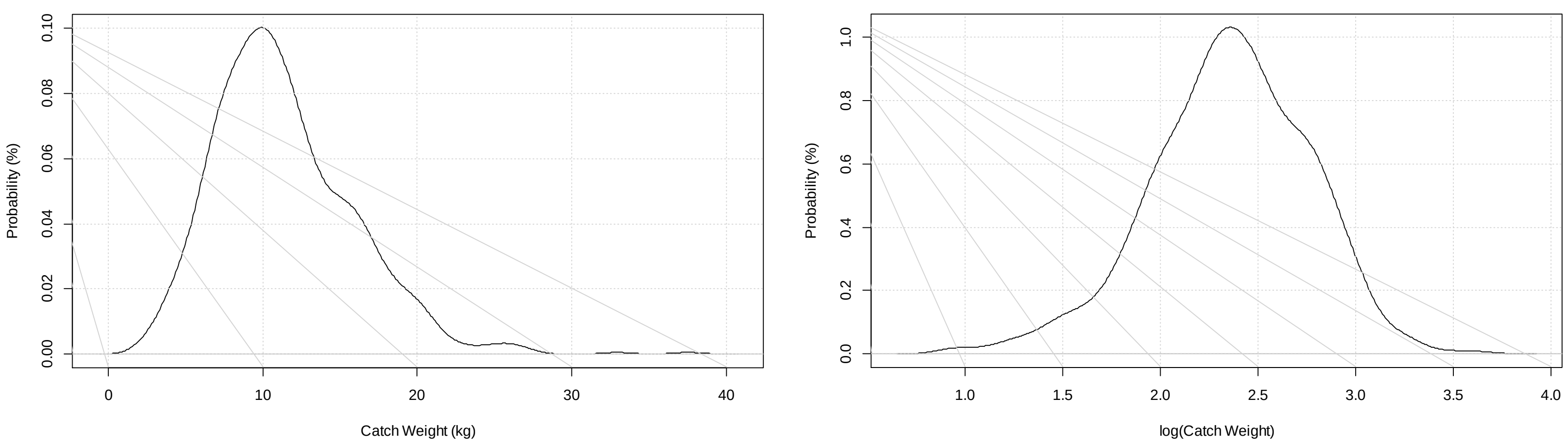


Figure 7-22. Distribution of catch weight in kg (left) and log-transformed catch weight (right) for *Henicorhynchus lobatus*.

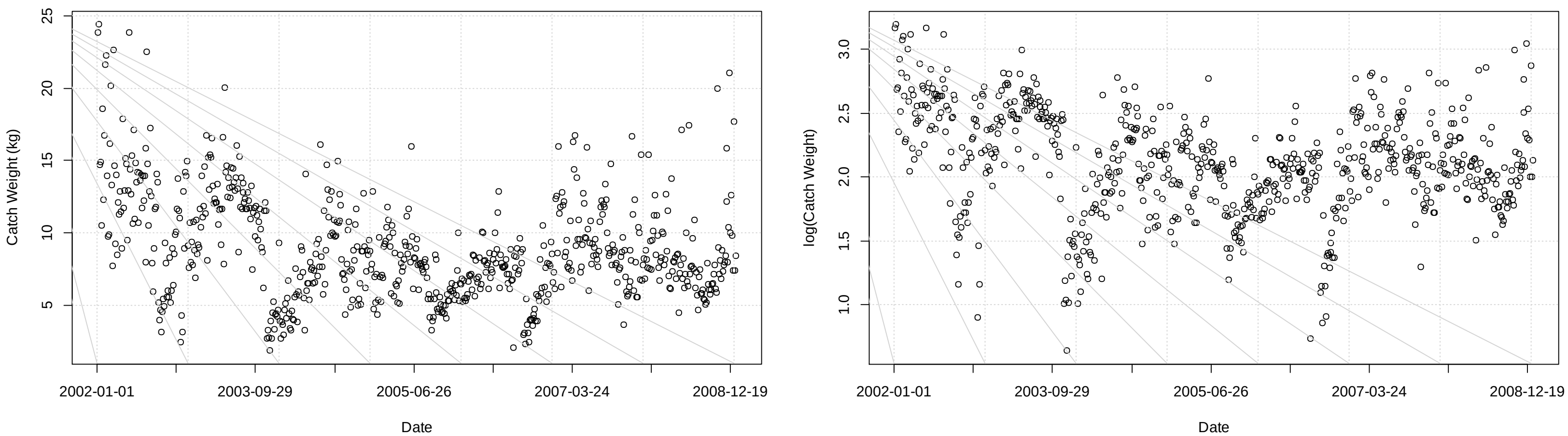


Figure 7-23. Catch weight in kg (left) and log-transformed catch weight (right) over time for *Labiobarbus lineata*.

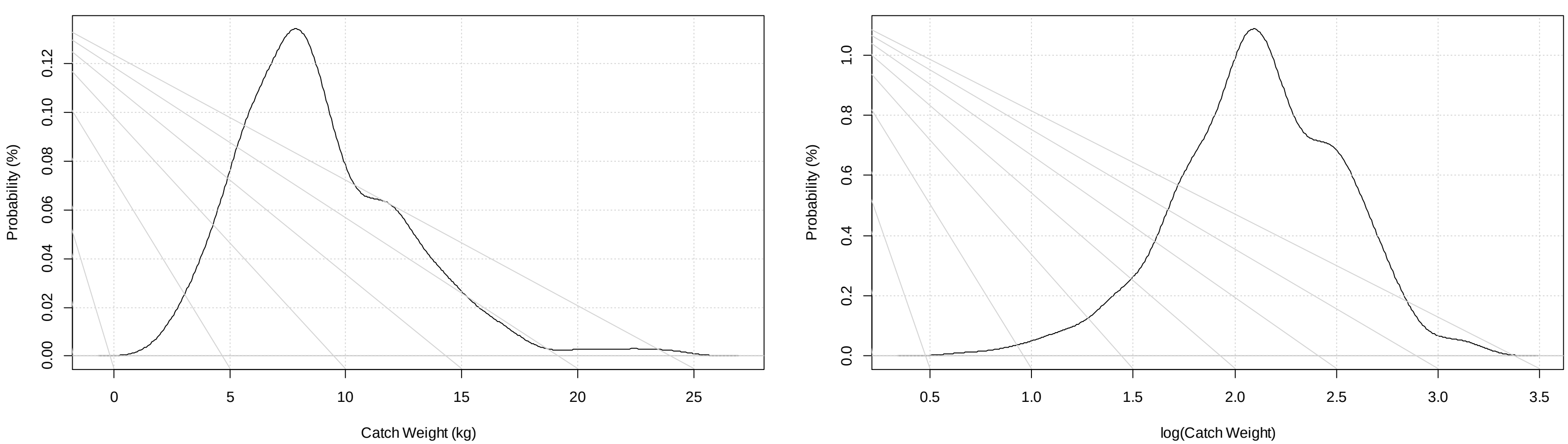


Figure 7-24. Distribution of catch weight in kg (left) and log-transformed catch weight (right) for *Labiobarbus lineata*.

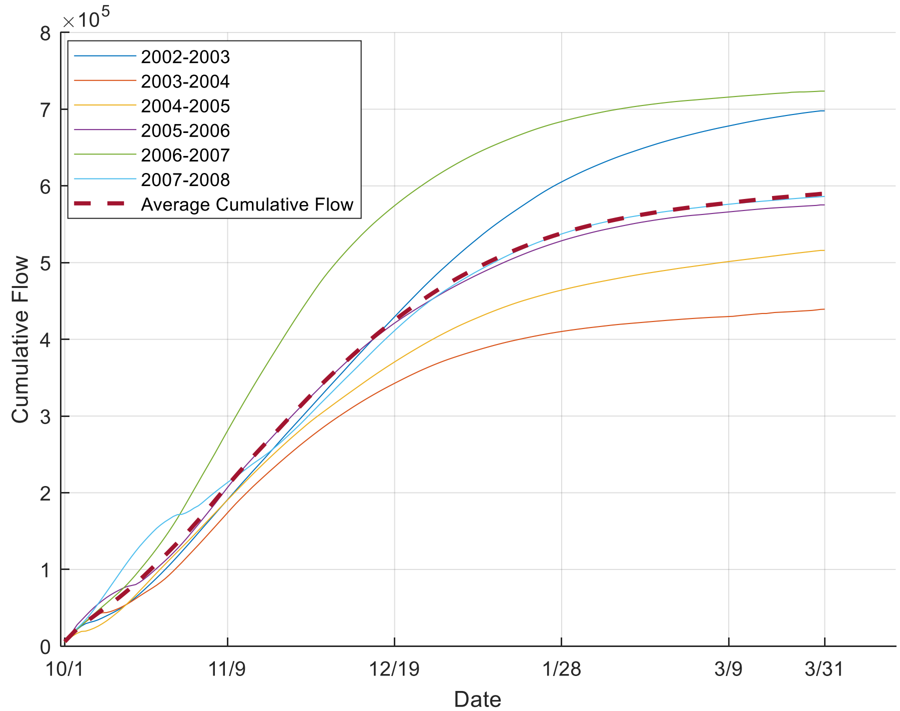


Figure 7-25. Cumulative flow for each water year.

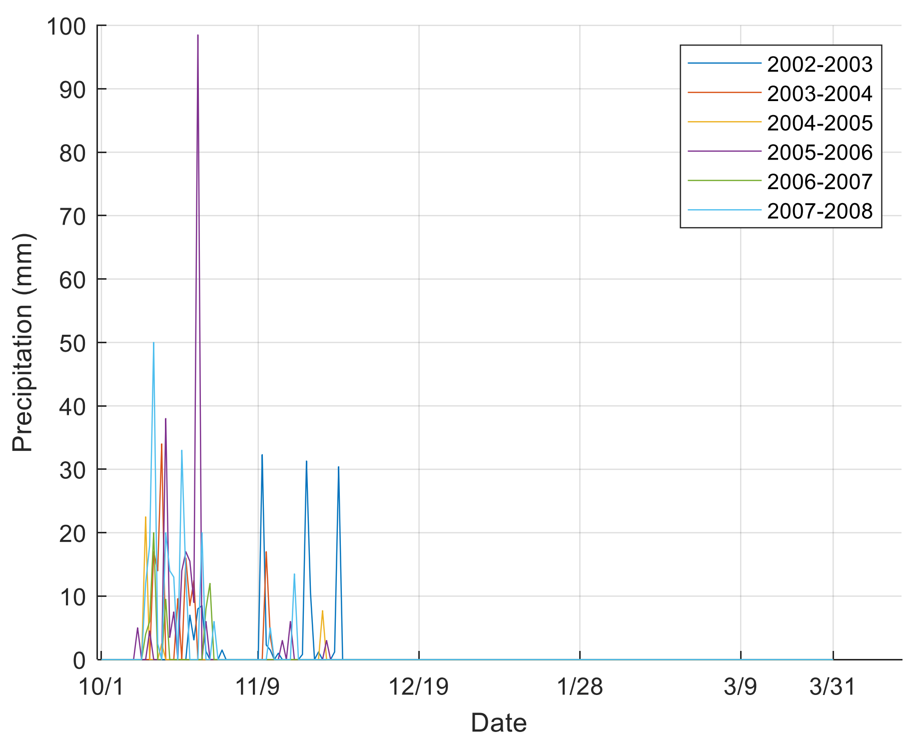


Figure 7-26. Precipitation for each water year.

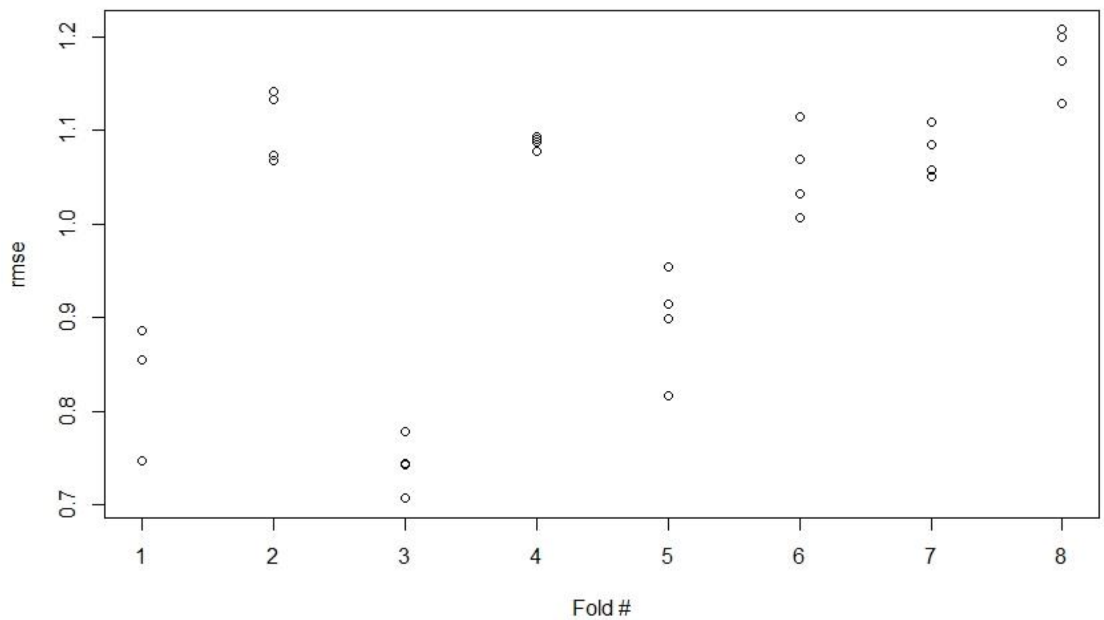


Figure 7-27. RMSE of each testing set (fold) with different mtry values for *Pangasianodon hypophthalmus.*

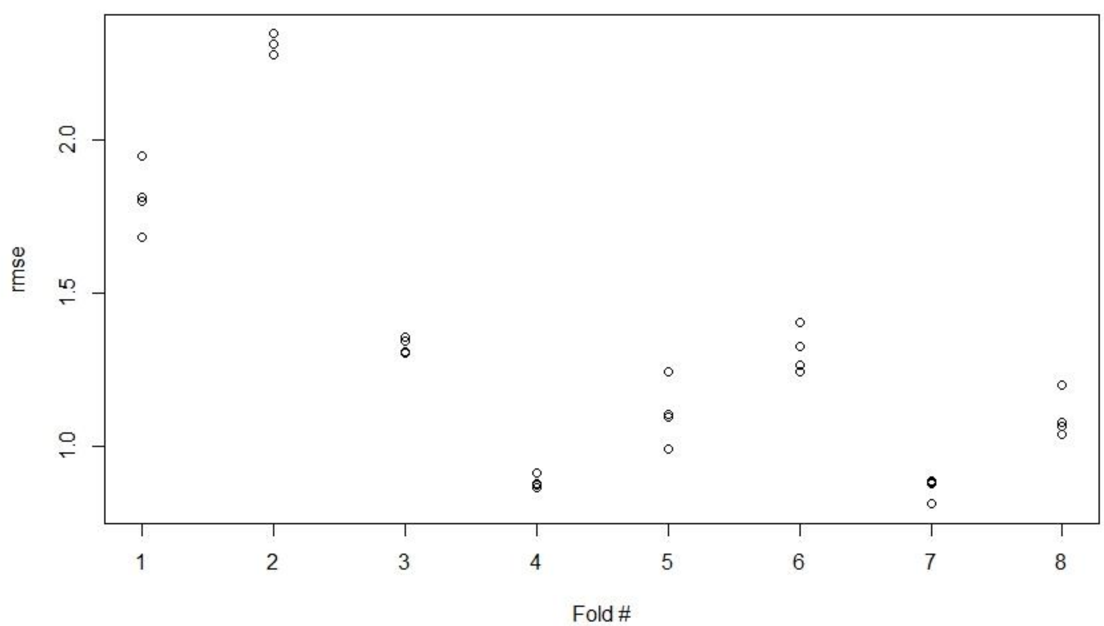


Figure 7-28. RMSE of each testing set (fold) with different mtry values for *Cyclocheilchthys enoplos.*

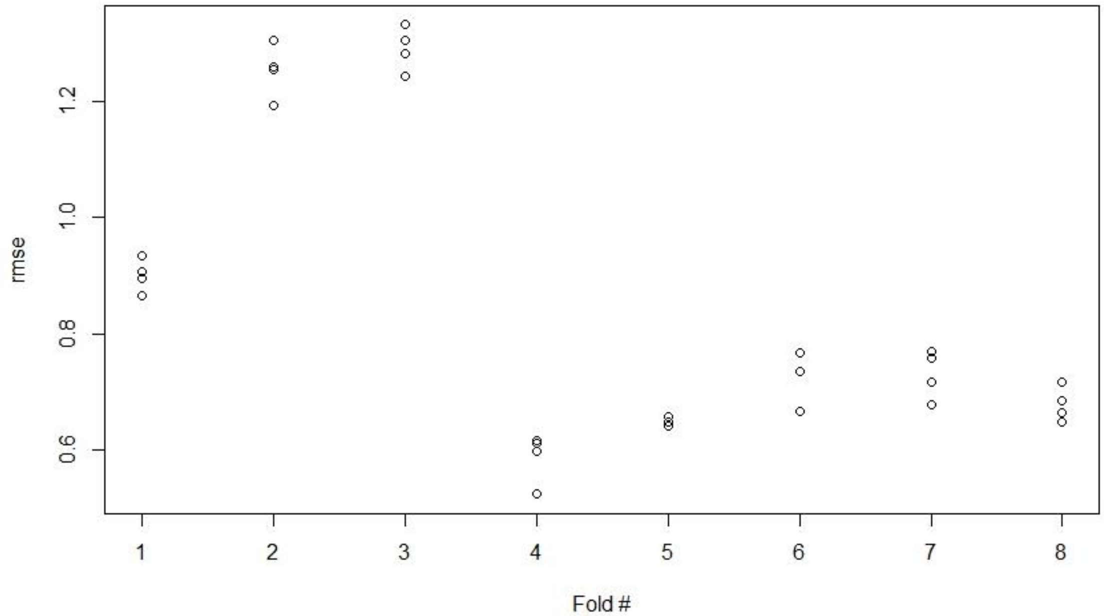


Figure 7-29. RMSE of each testing set (fold) with different mtry values for *Cirrhinus microlepis.*

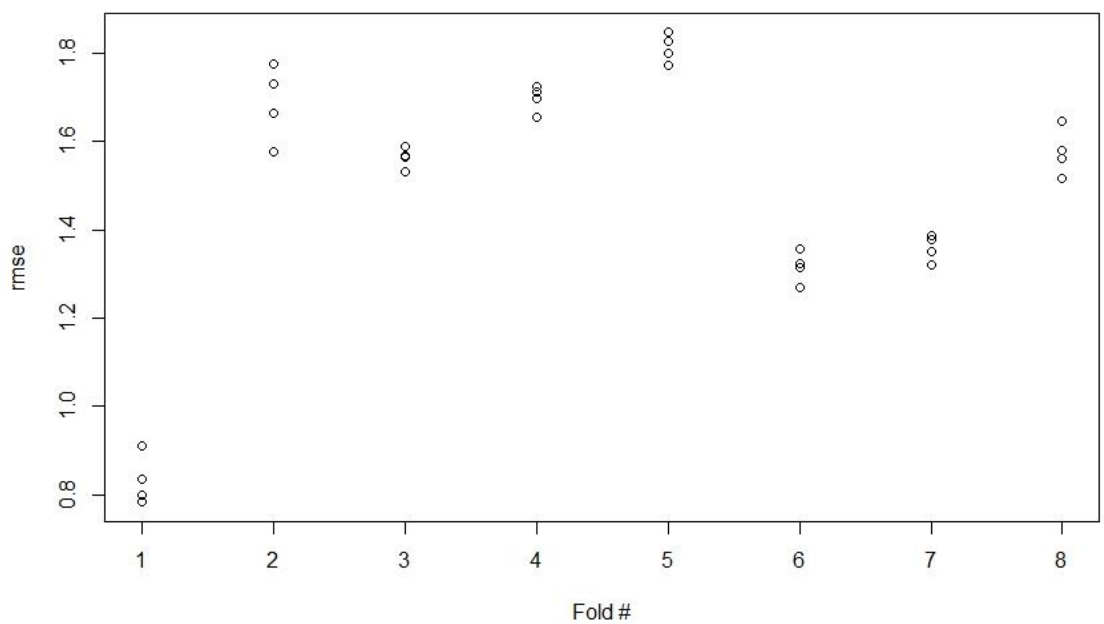


Figure 7-30. RMSE of each testing set (fold) with different mtry values for *Osteochilus melanopleurus.*

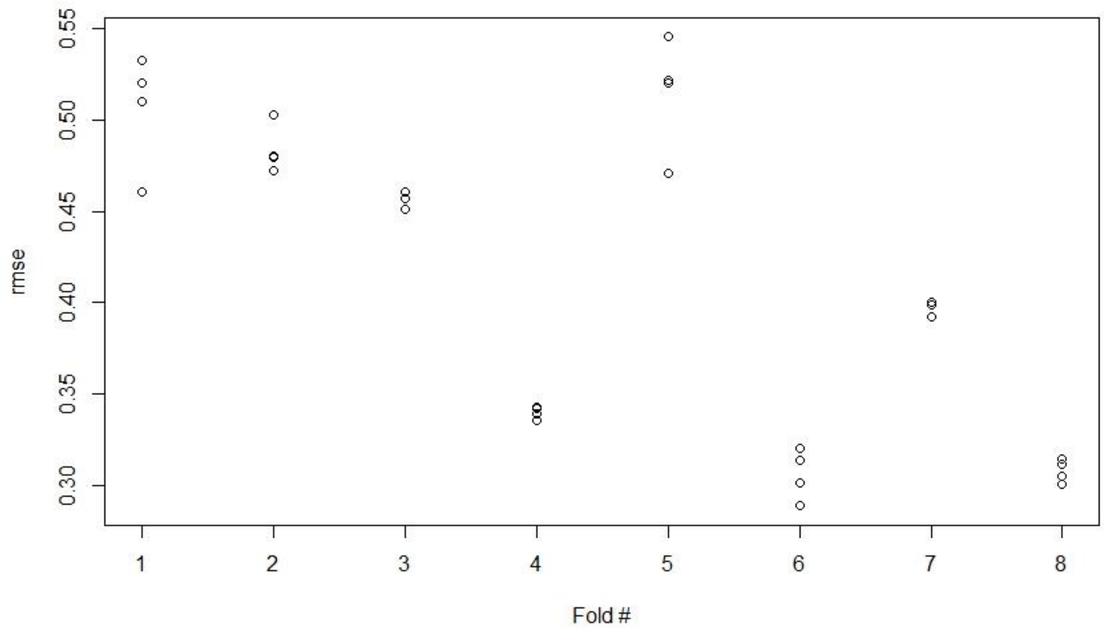


Figure 7-31. RMSE of each testing set (fold) with different mtry values for *Henicorhynchus lobatus*.

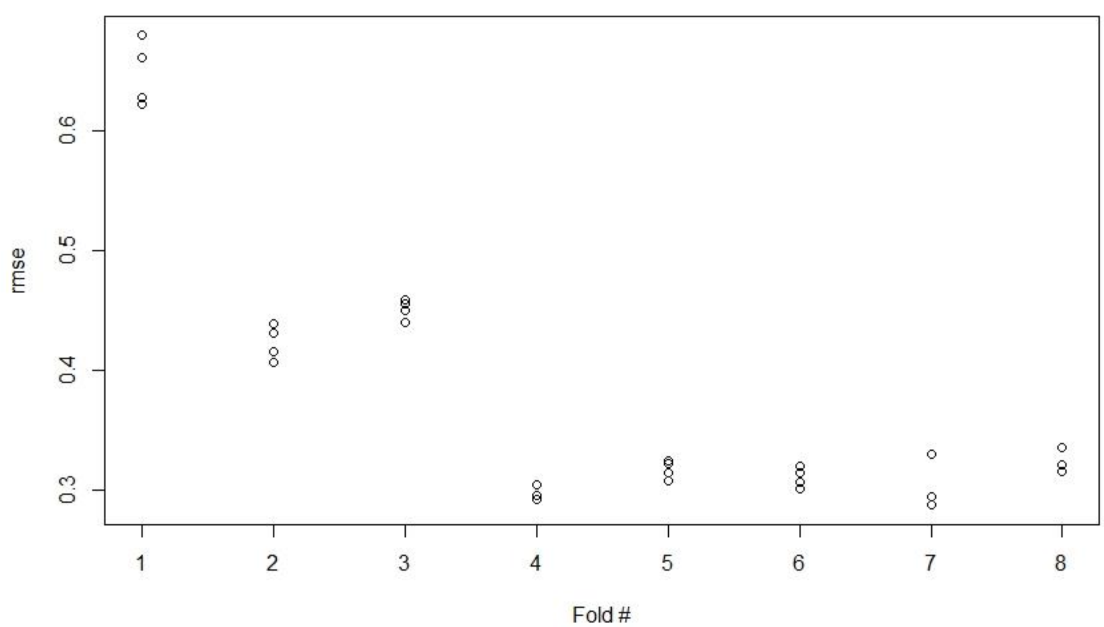


Figure 7-32. RMSE of each testing set (fold) with different mtry values for *Labiobarbus lineata*.

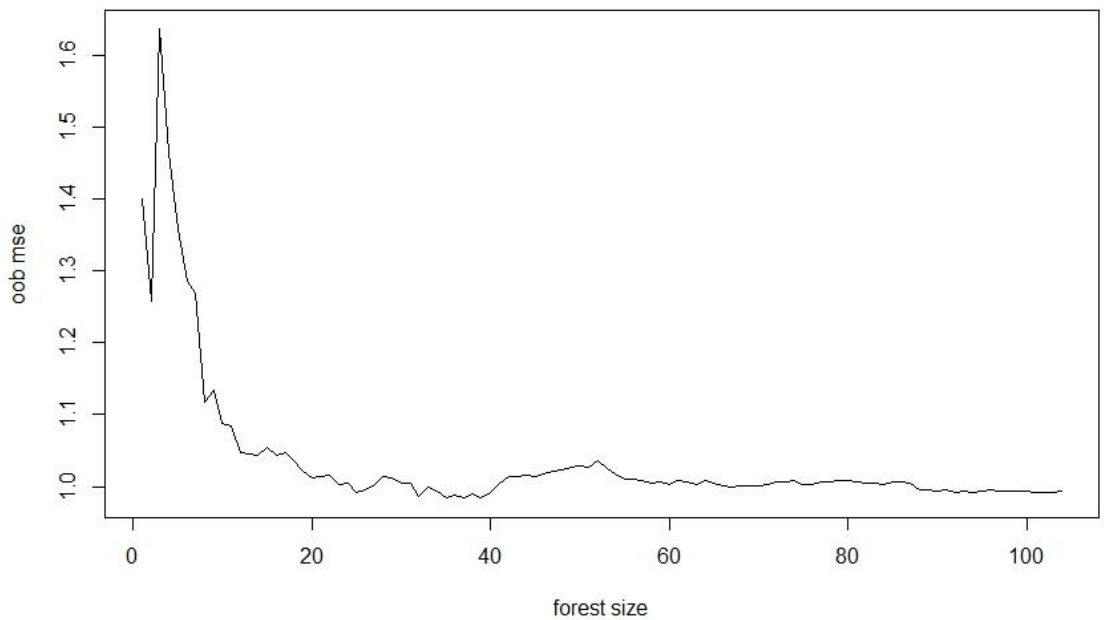


Figure 7-33. Out-of-bag error after each tree for *Pangasianodon hypophthalmus.*

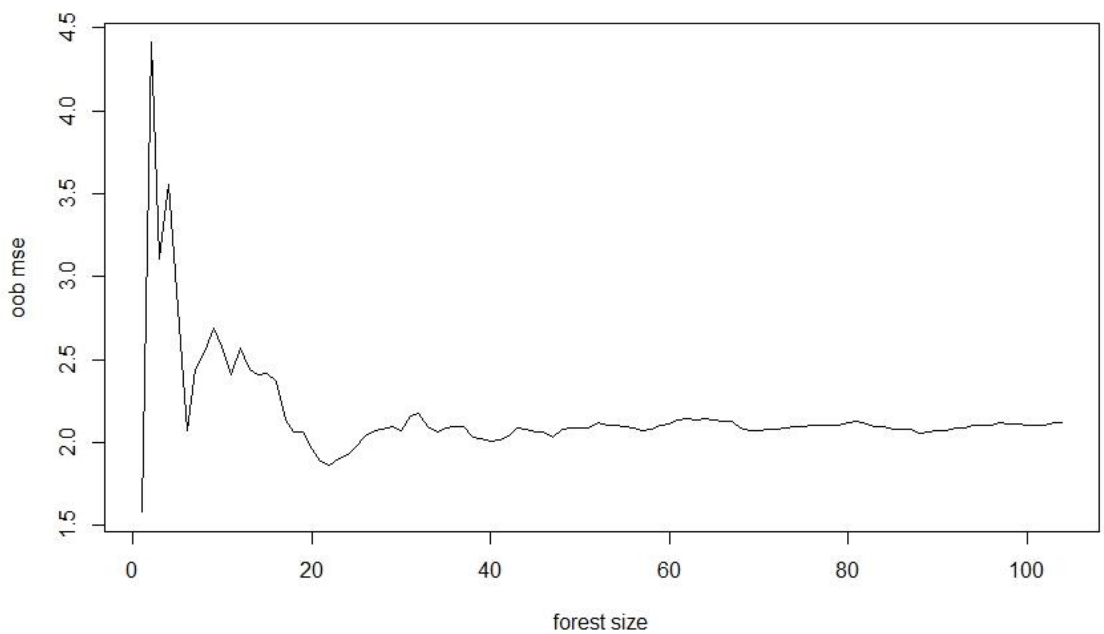


Figure 7-34. Out-of-bag error after each tree for *Cyclocheilchthys enoplos.*

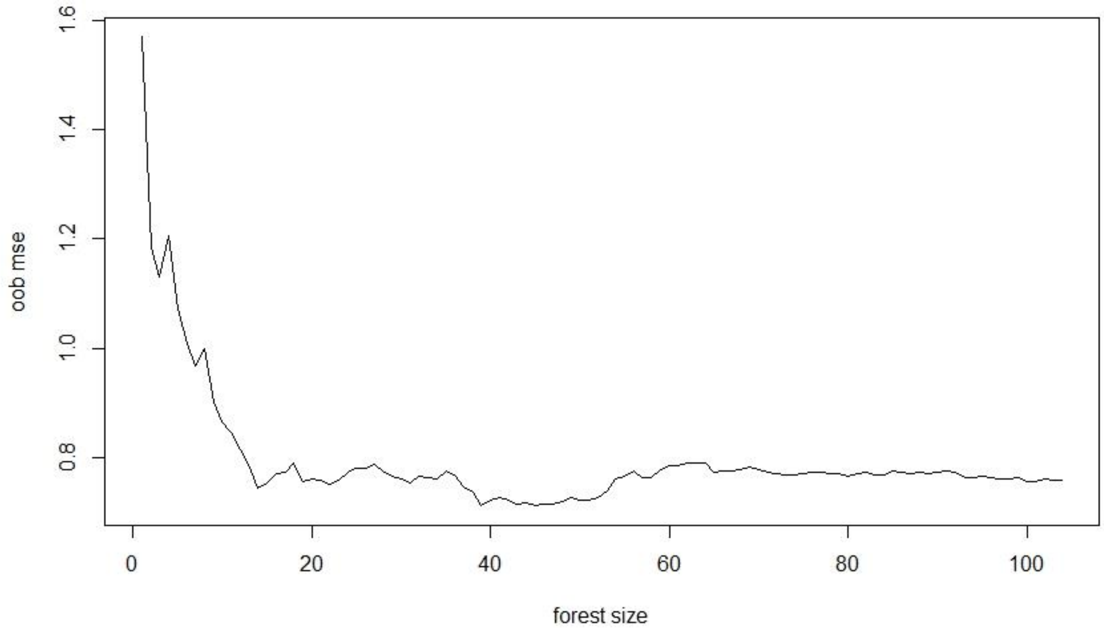


Figure 7-35. Out-of-bag error after each tree for *Cirrhinus microlepis.*

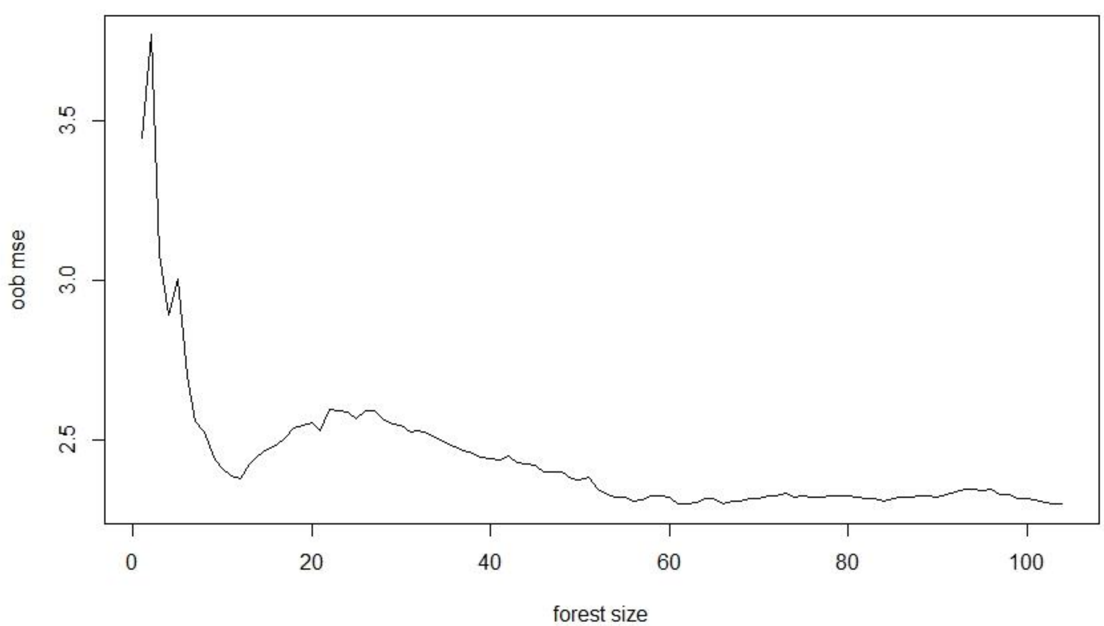


Figure 7-36. Out-of-bag error after each tree for *Osteochilus melanopleurus.*

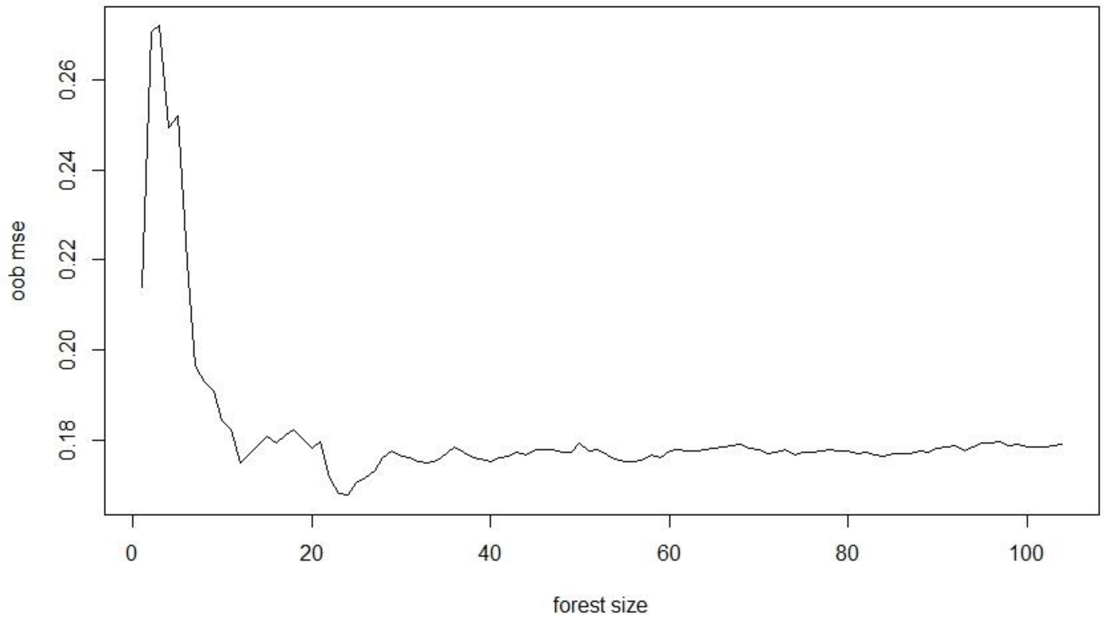


Figure 7-37. Out-of-bag error after each tree for *Henicorhynchus lobatus.*

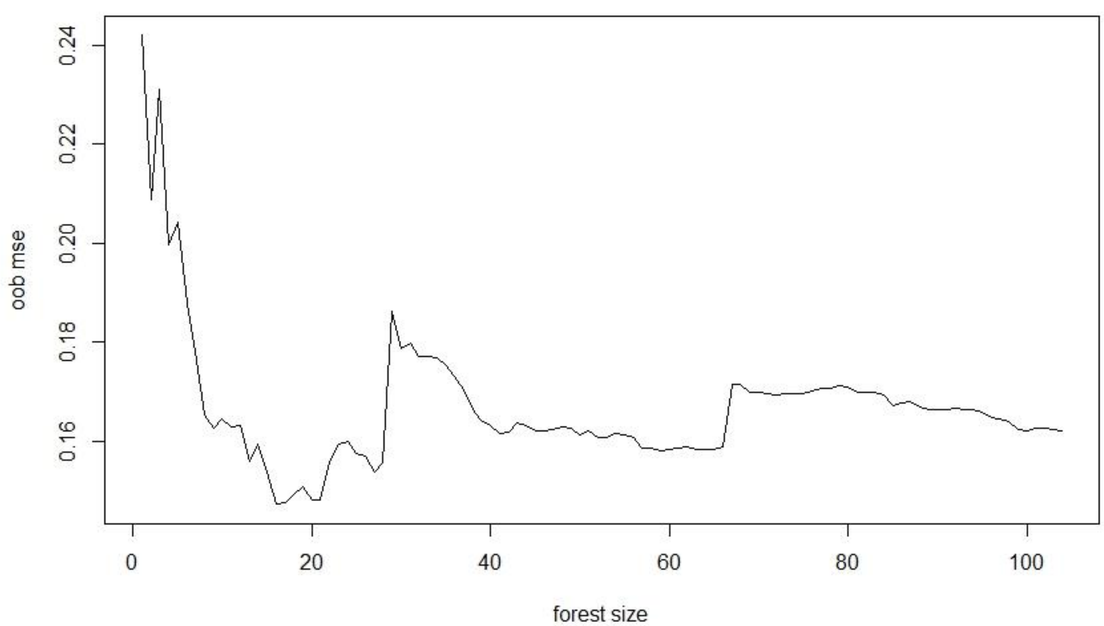


Figure 7-38. Out-of-bag error after each tree for *Labiobarbus lineata.*

Table 7-14. Variable importance table for *Pangasianodon hypophthalmus*
*Full model OOB MSE: 0.9934 in log(catch weight)^2

	Predictor	Marginalized OOB MSE	Relative Change in OOB MSE	Z-value
1	Previous catch weight	1.0055	0.012	3.157
2	Days from 90-day min flow in water year	1.0013	0.008	0.975
3	Days from 7-day max flow in water year	1.0003	0.007	1.152
4	Cumulative # of high pulse days	0.9973	0.004	0.73
5	Above 75th percentile flow	0.9967	0.003	0.436
6	Days from 30-day max flow in water year	0.9939	0	-0.033
7	Precipitation	0.9938	0	-0.125
8	Cumulative # of low pulse days	0.9934	0	0.373
9	Below 25th percentile flow	0.9934	0	-0.223
10	Days from 90-day max flow in water year	0.9929	-0.001	-0.085
11	Days from 30-day min flow in water year	0.9921	-0.001	-0.346
12	Moon cycle day	0.9908	-0.003	-0.68
13	Days from 7-day min flow in water year	0.9891	-0.004	-0.819
14	Days from 1-day max flow in water year	0.989	-0.004	-0.715
15	Cumulative flow	0.9887	-0.005	-0.698
16	Kompong Luong water level	0.987	-0.007	-0.306
17	Phnom Penh water level	0.9864	-0.007	-2.2
18	Flow	0.9863	-0.007	-1.121
19	Prek Kdam water level	0.9803	-0.013	-0.455
20	Residuals from avg. cumulative flow	0.9729	-0.021	-4.627
21	Days from 1-day min flow in water year	0.9708	-0.023	-2.056

Table 7-15. Variable importance table for *Cyclocheilichthys enoplos*
*Full model OOB MSE: 2.1178 in log(catch weight)^2

	Predictor	Marginalized OOB MSE	Relative Change in OOB MSE	Z-value
1	Previous catch weight	2.3868	0.127	2.684
2	Moon cycle day	2.1493	0.015	0.975
3	Cumulative # of low pulse days	2.1388	0.01	1.04
4	Flow	2.1285	0.005	3.19
5	Cumulative # of high pulse days	2.1267	0.004	0.555
6	Prek Kdam water level	2.1231	0.003	3.658
7	Phnom Penh water level	2.1227	0.002	1.687
8	Kompong Luong water level	2.1222	0.002	0.991
9	Days from 90-day max flow in water year	2.1207	0.001	1.165
10	Below 25th percentile flow	2.1178	0	0.984
11	Precipitation	2.1168	0	0.112
12	Above 75th percentile flow	2.1139	-0.002	-0.532
13	Cumulative flow	2.1132	-0.002	0.614
14	Days from 90-day min flow in water year	2.1102	-0.004	-0.723
15	Days from 1-day min flow in water year	2.1083	-0.004	0.587
16	Days from 1-day max flow in water year	2.1082	-0.004	1.077
17	Days from 7-day max flow in water year	2.1076	-0.005	0.877
18	Days from 30-day min flow in water year	2.1069	-0.005	0.883
19	Days from 30-day max flow in water year	2.1053	-0.006	0.227
20	Days from 7-day min flow in water year	2.0866	-0.015	-0.203
21	Residuals from avg. cumulative flow	2.0354	-0.039	0.126

Table 7-16. Variable importance table for *Cirrhinus microlepis*
*Full model OOB MSE: 0.7336 in log(catch weight)^2

	Predictor	Marginalized OOB MSE	Relative Change in OOB MSE	Z-value
1	Previous catch weight	0.8512	0.115	2.965
2	Days from 90-day max flow in water year	0.7739	0.013	1.153
3	Cumulative flow	0.7677	0.005	0.141
4	Below 25th percentile flow	0.7634	0	-1
5	Precipitation	0.7634	0	-0.902
6	Moon cycle day	0.7628	-0.001	-0.509
7	Kompong Luong water level	0.7624	-0.002	0.797
8	Days from 7-day max flow in water year	0.7619	-0.002	0.464
9	Days from 90-day min flow in water year	0.7602	-0.004	-0.587
10	Above 75th percentile flow	0.7593	-0.006	-1.926
11	Cumulative # of high pulse days	0.7584	-0.007	-1.989
12	Days from 7-day min flow in water year	0.7547	-0.012	-2.01
13	Phnom Penh water level	0.7541	-0.012	-2.834
14	Days from 30-day max flow in water year	0.7533	-0.013	-1.888
15	Flow	0.7524	-0.015	-1.295
16	Cumulative # of low pulse days	0.7524	-0.015	-0.636
17	Days from 30-day min flow in water year	0.752	-0.015	-2.744
18	Prek Kdam water level	0.7517	-0.016	-0.749
19	Residuals from avg. cumulative flow	0.7444	-0.025	-0.382
20	Days from 1-day min flow in water year	0.7416	-0.029	-2.532
21	Days from 1-day max flow in water year	0.7344	-0.038	-3.043

Table 7-17. Variable importance table for *Osteochilus melanopleurus*
*Full model OOB MSE: 2.3027 in log(catch weight)^2

	Predictor	Marginalized OOB MSE	Relative Change in OOB MSE	Z-value
1	Cumulative # of high pulse days	2.3439	0.018	1.586
2	Days from 1-day min flow in water year	2.325	0.01	0.118
3	Days from 30-day max flow in water year	2.3096	0.003	0.825
4	Precipitation	2.3055	0.001	0.769
5	Moon cycle day	2.3035	0	-0.044
6	Below 25th percentile flow	2.3027	0	0.487
7	Residuals from avg. cumulative flow	2.3017	0	0.316
8	Above 75th percentile flow	2.2973	-0.002	-0.221
9	Days from 90-day min flow in water year	2.2951	-0.003	-0.515
10	Phnom Penh water level	2.292	-0.005	-0.176
11	Flow	2.2915	-0.005	0.306
12	Previous catch weight	2.2906	-0.005	-0.232
13	Days from 90-day max flow in water year	2.29	-0.006	0.273
14	Cumulative flow	2.2894	-0.006	-0.865
15	Cumulative # of low pulse days	2.2857	-0.007	-1.85
16	Days from 30-day min flow in water year	2.285	-0.008	-0.685
17	Prek Kdam water level	2.274	-0.012	-0.364
18	Kompong Luong water level	2.2693	-0.015	-1.233
19	Days from 1-day max flow in water year	2.265	-0.016	-0.69
20	Days from 7-day max flow in water year	2.2625	-0.017	-0.819
21	Days from 7-day min flow in water year	2.2538	-0.021	-1.246

Table 7-18. Variable importance table for *Henicorhynchus lobatus*
*Full model OOB MSE: 0.1852 in log(catch weight)^2

	Predictor	Marginalized OOB MSE	Relative Change in OOB MSE	Z-value
1	Days from 1-day max flow in water year	0.1868	0.009	1.075
2	Phnom Penh water level	0.1866	0.008	0.517
3	Days from 30-day max flow in water year	0.1866	0.008	1.566
4	Moon cycle day	0.1865	0.007	0.771
5	Days from 7-day max flow in water year	0.1862	0.005	0.773
6	Flow	0.1858	0.004	-0.172
7	Above 75th percentile flow	0.1854	0.001	0.53
8	Days from 90-day max flow in water year	0.1852	0	-0.449
9	Below 25th percentile flow	0.1852	0	-0.62
10	Days from 7-day min flow in water year	0.1849	-0.002	-0.639
11	Days from 30-day min flow in water year	0.1849	-0.001	-0.421
12	Precipitation	0.1849	-0.002	-1.103
13	Days from 90-day min flow in water year	0.1846	-0.003	-1.231
14	Previous catch Weight	0.1843	-0.005	-1.272
15	Prek Kdam water level	0.1837	-0.008	-1.305
16	Cumulative # of high pulse days	0.1829	-0.012	-1.133
17	Cumulative # of low pulse days	0.1824	-0.015	-1.141
18	Kompong Luong water level	0.1822	-0.016	-2.684
19	Cumulative Q	0.1816	-0.019	-0.839
20	Days from 1-day min flow in water year	0.1813	-0.021	-1.377
21	Residuals from avg. cumulative flow	0.1802	-0.027	-1.253

Table 7-19. Variable importance table for *Labiobarbus lineatus*
*Full model OOB MSE: 0.162 in log(catch weight)^2

	Predictor	Marginalized OOB MSE	Relative Change in OOB MSE	Z-value
1	Previous catch weight	0.1652	0.02	1.084
2	Days from 7-day min flow in water year	0.1652	0.02	0.352
3	Days from 7-day max flow in water year	0.1642	0.014	0.891
4	Days from 1-day min flow in water year	0.1628	0.005	-0.011
5	Days from 30-day min flow in water year	0.1627	0.004	0.377
6	Flow	0.1624	0.003	0.958
7	Above 75th percentile flow	0.1623	0.002	1.17
8	Days from 30-day max flow in water year	0.1622	0.001	1.149
9	Cumulative # of high pulse days	0.1622	0.002	0.898
10	Days from 90-day max flow in water year	0.1621	0	0.137
11	Phnom Penh water level	0.1621	0	0.734
12	Moon cycle day	0.162	0	0.025
13	Cumulative # of low pulse days	0.162	0	-0.081
14	Below 25th percentile flow	0.162	0	-0.437
15	Days from 1-day max flow in water year	0.1619	-0.001	0.045
16	Precipitation	0.1619	0	-1.268
17	Days from 90-day min flow in water year	0.1614	-0.004	-0.43
18	Kompong Luong water level	0.1614	-0.004	-0.753
19	Prek Kdam water level	0.161	-0.006	-0.691
20	Residuals from avg. cumulative flow	0.1584	-0.022	-2.247
21	Cumulative flow	0.1557	-0.039	-1.482

Appendix D – Permission to Reprint Chapter 3 in *Fire*

June 22, 2020

Dear Ryan Choi,

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Signed: ________________________________

June 22, 2020

Dear Emily Esplin,

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Signed: ____

Liana Prudencio
Department of Watershed Sciences
Utah State University, Logan, UT, USA
lianap566@gmail.com
(469)222-2020

Education

2016- Ph.D. Watershed Science, Utah State University, Logan, UT (Anticipated 2020)
Dissertation: "Water, Fish, and Fire: Interdisciplinary Research on Ecosystem Services and Climate Adaptation"

2014 M.S. Sociology, University of Utah, Salt Lake City, UT (GPA: 3.989)
Thesis: "Cyber Concern for the Environment: A Multilevel Analysis of the Role of the Internet and the Digital Divide in Shaping Global Environmental Attitudes"

2011 B.S. Journalism and Mass Communication Minors: Sociology and Music Technology
Iowa State University, Ames, IA (Summa Cum Laude; GPA: 3.94)

Research Experience

Researcher with the USAID Wonders of the Mekong Project, Phnom Penh, Cambodia (2017-present)

- Collected hydrologic and water quality data in the Mekong and its tributaries during fieldwork trips to Cambodia
- Facilitated outreach efforts to educate local Cambodians about the project and the ecological and cultural significance of the Mekong
- Developed innovative models to evaluate fish migration triggers in the Lower Mekong Basin
- Presented preliminary findings from fish migration triggers study at the American Fisheries Society-The Wildlife Society 2019 Meeting

Research Assistantship, Dr. Sarah E. Null, Dept. of Watershed Sciences, Utah State University (2016-present)

- Collaborated with civil and environmental engineers and sociologists on an EPA-STAR study that assessed the potential of using green infrastructure for managed aquifer recharge for water storage with less reliability on snowpack
- Engaged with stakeholders and water managers in the Salt Lake Valley during bi-annual meetings
- Held workshops and demonstrated the value of the research to decision-making and policy
- Aided team members with fieldwork and research

NSF-NRT Climate Adaptation Science Fellow, Climate Adaptation Science Program, Utah State University (2017-18)

- Awarded prestigious fellowship to participate in a National Science Foundation advanced research traineeship in Climate Adaptation Science
- Led team on an interdisciplinary project studying adaptive fire management and policy in the Intermountain West
- Communicated research efforts through various media (peer-reviewed journal, conference talks, social media, blogs, NPR-UPR interview)

Private Research Assistant for Dr. Dan McCool, Dept. of Political Science, University of Utah (2014-2017)

- Helped Professor Dan McCool (expert witness in Voting Rights Act cases involving American Indians) obtain data for witness reports and other Voting Rights research efforts
- Conducted interviews and collected quantitative data to be used in reports

Survey Research Assistant for NSF-funded iUTAH, Dr. Douglas Jackson-Smith, Dept. of Sociology, Utah State University (2014)

- Collected surveys in Salt Lake and Heber Valleys on water use and opinions on water issues
- Contributed important data on household water use and behavior to decision-makers, researchers, and the public

Researcher on projects with Dr. Akiko Kamimura, Dept. of Sociology, University of Utah (2013-15)

- Led focus groups and collected surveys at the Maliheh Free Clinic to understand the experiences and needs of patients in and outside of the clinic
- Evaluated the experiences of international medical graduates through in-depth qualitative interviews

Teaching Experience

Instructor:

- Geography 1000: Intro to Physical Geography
- Sociology 1015 and 2015: Doing Sociology (online)
- Sociology 3436: Global Social Structure
- Sociology 3435: Inequality and Globalization (online)

Teaching Assistant:

- Sociology 3111: Research Methods
- Sociology 3563: Good Cop, Bad Cop: Policing in America
- Sociology 3112 Social Statistics

Skills and Expertise

Software: MS Office, ArcGIS, QUAL2Kw, R, Stata, SPSS, WEAP, GAMS
Fieldwork: surveys, use of Acoustic Doppler Current Profiler, streamflow and water quality monitoring, wilderness first aid
Science Communication
Interdisciplinary and Collaborative Work/Research
Water Policy

Publications

1. **Prudencio, L.**, Choi, R., Esplin, E. D., Ge, M., Gillard, N., Haight, J., Belmont, P., and Flint, C. G. 2018. "The Impacts of Wildfire Characteristics and Employment on the Adaptive Management Strategies in the Intermountain West." *Fire*, 1(3):46.
2. **Prudencio, L.** and Null, S.E. 2018. "Stormwater Management and Ecosystem Services: A Review." *Environmental Research Letters*. 13(2018): 033002.
3. Null, S.E. and **Prudencio, L.** 2016. "Climate change effects on water allocations with season-dependent water rights." *Science of the Total Environment*, 571:943-54.
4. Kamimura, A., Ashby, J., Trinh, H. N., **Prudencio, L.**, Mills, A., Tabler, J., Nourian. M. M., Ahmad, F., and Reel, J. J. 2016. "Uninsured free clinic patients' experience and perceptions of healthcare services and patient education." *Patient Experience Journal*, 3(2):12-21.
5. Kamimura, A., Samhouri, M., Huynh, T., Myers, K., **Prudencio, L.**, Eckhardt, J., and Al-Obaydi, S. 2016. "Physician migration: Experience of international medical graduates in the US." *Journal of International Migration and Integration*. Online publication date: March 8, 2016. Published, 03/2016.
6. Kamimura, A., Ashby, J., Jess, A., Trinh, H. N., Nourian. M. M., Finlayson, S. Y., **Prudencio, L.**, and Reel, J. J. 2015. "Impact of neighborhood environments on health consciousness, information seeking, and attitudes among US-born and non-US-born free clinic patients." *Southern Medical Journal*, 108(12): 703-709.
7. Kamimura, A., Tabler, J., Chernenko, A., Aguliera, G., Nourian, M. M., **Prudencio, L.**, and Ashby, J. 2015. "Why uninsured free clinic patients don't apply for Affordable Care Act health insurance in a non-expanding Medicaid state." *Journal of Community Health*. Online publication date: August 15, 2015. Published, 08/2015.

Papers In-Progress

1. **Prudencio, L.** and Null, S.E. "Ecosystem Services from Implementing Green Stormwater Infrastructure at the Watershed-Scale."
2. **Prudencio, L.**, Null, S.E., Ngor, P.B., Touch, B., and Chhuoy, S. "Predicting Fish Migration in One of the World's Largest Inland Fisheries with Historical Random Forest Modeling."
3. Null, SE., Farshid, A., Goodrum, G., Gray, C., Lohani, S., Morrisett, C., **Prudencio, L.** "Environmental Tradeoffs of Dams in the Sekong, Sesan, and Srepok (3S) Rivers of the Lower Mekong Basin."
4. Tromboni, F., Chandra, S., **Prudencio, L.**, Ngor, P.B., Saray, S., Hogan, Z. "The Effect of the Lower Sesan 2 Dam on the Biogeochemistry of the 3S System."
5. Campbell, T., Loury, E., Ainsley, S., Chandra, S., Dilts, T., Elliott, V., Gatke, P., Lee, D., Lohani, S., Ngor, P.B., Null, S.E., Phen, C., **Prudencio, L.**, Saray, S., Tromboni, F., Wanningen, H., Weisberg, P., Yong, D.L., Hogan, Z. "Managing Cambodia's Migratory Fish: A Vision for Success."
6. Chandra, S., Tromboni, F., **Prudencio, L.**, Sullivan, B., Ngor, P.B., Saray, S., Hogan, Z. "The Influence of the Lower Sesan River 2 Dam on Organic Carbon and Greenhouse Gas Concentrations."

Conference Presentations (as first author)

2019 "Predicting Fish Migration Triggers in the Lower Mekong Basin with Random Forest Modeling." American Fisheries Society-The Wildlife Society Conference in Reno, NV. (*Symposium Co-Organizer; Anticipated October 1, 2019*)

2018 "Assessing Fire Trends, Economic Effects, and Adaptive Management Strategies in the Intermountain West." 24th International Symposium on Society and Resource Management (ISSRM) in Salt Lake City, UT.

2017 "Calling All Collaborators: Robust Decision-Making & Climate Adaptation." Salt Lake County Watershed Symposium in West Valley City, UT

2016 "Stormwater Management Effects on Ecosystem Services: A Literature Review." American Geophysical Union Fall Meeting in San Francisco, CA.

2013 "Cyber Concern for the Environment: A Multilevel Analysis of the Role of the Internet and the Digital Divide in Shaping Global Environmental Attitudes." Sociology of Development (American Sociological Association section) Conference in Salt Lake City, UT.

Awards and Funding

2016 – 2018 NSF-NRT Fellowship, Climate Adaptation Science Program, Utah State University
2015 Phi Kappa Phi induction
2014 University of Utah Department of Sociology Conference Annual Travel Award
2014 University of Utah Graduate School Conference Annual Travel Award
2013 University of Utah Department of Sociology Conference Annual Travel Award
2011 Phi Beta Kappa induction
2011 Kappa Kau Alpha induction
2010 – 2011 Sanders Scholarship, Greenlee School of Journalism and Mass Communication
2010 Columbia Scholastic Press Association Gold Circle Awards (CM) – "December 3," Iowa State Daily, Iowa State University
2009 Columbia Scholastic Press Association Gold Circle Awards (2) – "Carnage in the Coliseum," Iowa State Daily, Iowa State University
2009 Alpha Lambda Delta induction
2009 Phi Eta Sigma induction
2009 National Society of Collegiate Scholars induction
2008 – 2012 Award for Competitive Excellence, Iowa State University

Professional References

Dr. Sarah E. Null – sarah.null@usu.edu
Associate Professor, Department of Watershed Sciences – Utah State University

Dr. Daniel Craig McCool – dan.mccool@poli-sci.utah.edu
Professor Emeritus, Department of Political Science – University of Utah

Dr. Nancy Huntly – nancy.huntly@usu.edu
Director of Ecology Center, Professor of Biology, Director of Climate Adaptation Science – Utah State University