Managing Flow Regimes and Landscapes Together:
Hydrospatial Analysis for Evaluating Spatiotemporal Floodplain Inundation Patterns with Restoration and Climate Change Implications

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Abstract

Riverine landscapes are shaped by dynamic and complex interactions between streamflow and floodplain landforms, and these physical processes drive productive and diverse freshwater ecosystems. However, human activities have fundamentally altered river-floodplain processes and degraded ecosystems. Flow regime variability has been homogenized and floodplains disconnected from rivers due to dams, diversions, levee building, and land use change. Reconciling competing demands to support ecosystems and resilience to future change is a core scientific and management challenge. This dissertation describes spatiotemporal dynamics of floodplain environments, introducing a method for flood regime classification and establishing a methodological approach for hydrospatial analysis to quantify and evaluate the response of floodplain inundation patterns and related physical habitat to restoration and flow regime change under climate change. It is motivated by the need to develop process-based and landscape-scale strategies to better manage flow regimes and landscapes together, such as coordinated levee-removal floodplain restoration and environmental flow allocations. River restoration literature is synthesized herein to examine trajectories from form-based to process-based approaches, recognize that highly modified large rivers may require coordinated physical habitat restoration and environmental flows implementation, and suggest opportunities for improved integration of restoration strategies.

A river's flood regime drives a variety of different physical and ecological functions. Characterizing different floods of a flood regime informs understanding of climate and watershed processes and the management of natural floodplain dynamics. Following cluster analysis approaches used in flow regime classification, a flood regime typology was developed for the Cosumnes River, the only major unregulated river of the west slope Sierra Nevada, California, USA. A primary contribution of this study is the establishment of flood regime classification that moves beyond typical flood frequency analysis to address a range of ecologically-relevant flood characteristics, including duration and timing.

Rehabilitating freshwater ecosystems of highly modified rivers under a changing future requires improved understanding and quantification of land-water interactions. Despite ecological implications, quantification of spatiotemporal variability is rare, particularly for management applications. An approach for evaluating spatiotemporal floodplain inundation patterns, or the hydrospatial regime, is presented in several studies. Physical inundation characteristics and associated habitat were quantified in space and time, and responses to restoration and climate change induced flow scenarios were evaluated and compared. The multi-metric approach is demonstrated for a recent levee-removal restoration site along the lower Cosumnes River.

The novel hydrospatial analytical approach developed and presented herein applies two-dimensional hydrodynamic modeling and spatial analysis to quantitatively summarize, in space and time, a range of ecologically-relevant physical metrics relating to inundation extent, depth, velocity, frequency, duration, timing, rate of change, connectivity, and heterogeneity. Comparison of metrics before and after levee-removal restoration on the Cosumnes River floodplain showed that while inundation extent greatly increased with restoration, responses varied in space and time and were different for different metrics. Changes in metrics were most substantial at intermediate flood flows. Subsequently, habitat criteria for a native floodplain fish species, Sacramento splittail (*Pogonichthys macrolepidotus*), were applied to the physical metrics. Findings suggest that restoration nearly doubled overall habitat availability, though benefits varied considerably in space and time. Flow-habitat relationships were nonlinear and not one-to-one, indicating habitat availability mediated by the physical complexity of the floodplain. Finally, floodplain responses to climate change induced streamflow scenarios were compared and the relative impacts of levee-removal restoration across the scenarios were evaluated. Results reflected the balance of increasing extreme winter flooding and declining spring flooding under future climate change scenarios. Magnitude and direction of change depended on the climate change scenario and metric. Levee removal had the general effect of dampening climate change impacts, though the relative impacts of climate change scenarios were greater than that of restoration in some cases.

This body of work presents a new methodology to analyze flow-landscape interactions, and in turn contributes to understanding of flow-ecology relationships, susceptibility to anthropogenic change, and improvements to water and land management. Several broad implications emerge from this research. It demonstrates the capacity of a riverine landscape to serve different functions at different times and supports improved management toward variable conditions. Another contribution is advancing the use of hydraulic metrics over hydrologic metrics for better connections between physical processes and ecological functions. Further, the approach allows for ecologically-relevant criteria that are spatially and temporally dependent to be evaluated explicitly (e.g., duration, connectivity, temporal sequence of flood events). Findings show that, for habitat evaluation within complex floodplain environments, habitat availability is not likely to be a simple function of flow. Floodplain hydrospatial regime responses to climate change will be mediated by flow-landscape interactions, with the potential for physical restoration activities to mitigate impacts of climate change. Despite highly modified physical processes, climate change, and freshwater diversity and productivity declines globally, there is great capacity to better balance human and ecosystem requirements. This dissertation expands scientific understanding of and informs management toward dynamic and heterogeneous riverine landscapes that support functional and resilient ecosystems.

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CHAPTER 1

INTRODUCTION: HYDROSPATIAL ANALYSIS FOR MANAGING SPATIOTEMPORAL FLOODPLAIN INUNDATION PATTERNS

Introduction: Hydrospatial analysis for managing spatiotemporal floodplain inundation patterns

Riverine landscapes are defined by variability and complexity across space and time and are extensively altered by water and land management. This dissertation advances description of these conditions to inform floodplain management and planning. Spatiotemporal floodplain inundation patterns that comprise a floodplain's hydrospatial regime are produced by the interaction of a river's flow regime – as characterized by magnitude, timing, frequency, duration, and rate of change (Poff et al., 1997) – with heterogeneous floodplain landscapes.

The shifting habitat mosaics of floodplain environments support some of the most diverse and productive ecosystems globally (Opperman et al., 2017; Tockner & Stanford, 2002; Ward et al., 2002b). However, most freshwater-dependent ecosystems are highly degraded by multiple anthropogenic stressors (Dudgeon et al., 2006; Vorosmarty et al., 2010). Large lowland alluvial rivers no longer support functional floodplains, and ecological resilience to future change has been eroded (Tockner et al., 2010; Tockner & Stanford, 2002). Floodplains are particularly vulnerable to change as they are at transition zones between terrestrial and aquatic environments, highly desirable for agricultural and urban development, and accumulate upstream impacts (Naiman et al., 2002). Natural flow variability has been homogenized, hydrologic connectivity reduced, physical gradients compressed or severed, habitat fragmented, and disturbance regimes altered (Poff et al., 2007; Ward, 1998). Dams alter the flow of over half of the world's large rivers (Nilsson et al., 2005), and diversions, groundwater abstraction, levee building, and land use change have become nearly ubiquitous in all but the most remote river basins. Changes to river morphology and sediment supply have occurred alongside flow modifications. Degradation continues as human demands for water and land resources increase and as temperatures rise and hydrology shifts with climate change (Palmer et al., 2008).

Reconciling competing demands for water and supporting ecosystem integrity and resilience within contemporary landscapes are core scientific and management challenges (Palmer et al., 2004; Poff, 2017). Reestablishing natural dynamics of land-water interaction and diverse flooding processes within the riverine landscape is fundamental to restoring floodplain ecosystems (Tockner et al., 2000; Ward et al., 2002a). Natural dynamism is particularly difficult to achieve in highly engineered and regulated rivers where opportunities are constrained and multiple intertwined factors of change must be understood and addressed simultaneously (Arthington et al., 2006; Brewer et al., 2016; Kondolf et al., 2006; Lake et al., 2007; Opperman et al., 2010). Promising process-based and landscape-scale strategies to restore floodplains include reconnecting rivers through levee removal or setbacks and adjusting flow regimes of regulated rivers (Beechie et al., 2010; Kondolf et al., 2006; Opperman et al., 2009; Palmer et al., 2008; Wohl et al., 2015b).

The science and practice of river restoration has expanded dramatically over the last several decades. Physically-based restoration measures can be categorized generally into those that alter flow regimes (i.e., environmental flows) and those that reform aspects of the physical landscape (e.g., channel redesign, levee-setbacks). Both sub-fields of restoration have followed similar trajectories, moving from form-based or static views (e.g., prescribed channel form or minimum flow releases) to a focus on activities that support physical and ecological processes (Beechie et al., 2010; Kondolf et al., 2006; Palmer et al., 2014; Yarnell et al., 2015). With this shift has come the recognition that habitat restoration or environmental flows alone may not provide substantial ecological benefits in highly modified river systems (Arthington et al., 2010; Kondolf, 2011; Wohl et al., 2015a). Landscapes and the flows with which they interact should be managed together to reestablish dynamic and heterogeneous processes driving diverse and productive ecosystems. Under hydroclimatic change and fundamentally altered novel ecosystems, such strategies are increasingly relevant (Moyle, 2013; Palmer et al., 2009; Perry et al., 2015). This transformation and the potential for improved integrated flow and physical habitat restoration planning and management are discussed in **Chapter 2**.

River-floodplain management challenges are exemplified in California's regulated rivers and highly modified landscapes, where balancing water for humans and the environment involves competing demands, a highly variable climate, as well as hydroclimatic change and water scarcity in the future. Solutions for floodplain ecosystems in California can be models for river-floodplain management globally. Enhancing hydrologic connectivity by altering flow targets and by levee removal or setbacks is central to restoration planning in the Central Valley of California. However, determining what and how physical variables of flow regimes and landscapes can be adjusted to support ecosystems has remained a fundamental challenge. A unique opportunity to pursue these concepts is presented in the lower Cosumnes River and its floodplain – the system of study for the four core chapters of this dissertation. The Cosumnes River is the only unregulated major river of the west slope Sierra Nevada, and it regularly accesses large portions of its historic floodplain. Several process-based levee-removal restoration efforts over the last several decades have promoted river-floodplain connectivity (Swenson et al., 2012). Cosumnes River research and monitoring has provided valuable insights for restoration throughout the Central Valley and globally (e.g., Ahearn et al., 2006; Florsheim & Mount, 2002; Jeffres et al., 2008; Nichols & Viers, 2017).

Variable floodplain conditions are driven by a river's flood regime. The flood regime is understood to be a primary determinant of biotic community composition and ecosystem productivity (Benke, 2001; Junk et al., 1989; Poff, 2002). Different floods serve different physical and ecological functions, whether it be extreme floods that reset the floodplain geomorphic template or long duration springtime flooding that promotes primary productivity (Opperman et al., 2010). **Chapter 3** addresses flood regime characterization, presenting a flood regime typology for the Cosumnes River (Whipple et al., 2017). The approach involved unsupervised cluster analysis of individual flood events. These flood events, identified from the 107-year (1908-2014 water years) daily flow record, were described using ecologically-relevant variables, including magnitude, timing, duration, and rate of change (Poff, 2002; Poff et al., 1997). Six flood types were identified, two of high peak flow magnitudes, one by longer duration and late season timing, another by hydrograph shape, and two low flow events separated by early and late timing. The Cosumnes flood regime was further examined for flood type inter- and intra-annual variability, association with wet or dry years, and increasing or decreasing occurrence over time. This study applies common concepts and methods used in flow regime classification to provide a more focused characterization of flood regime, valuable for understanding driving physical processes of and developing management strategies for floodplain ecosystems.

The interaction of flood regime with floodplain topography, geology, and vegetation generates complex flow paths, creates variable water depths and residence times, alters flow velocities, and affects a broad array of environmental conditions through space and time (Florsheim & Mount, 2002). The role of riverine landscape heterogeneity in shaping aquatic communities and driving ecological processes is well established (Amoros & Bornette, 2002; Power et al., 1995; Tockner et al., 2000; Ward, 1989). Enhancing floodplain ecosystem integrity and resilience within the context of many human-imposed limitations requires improved characterization and quantification of floodplain variability and complexity at spatial and temporal scales relevant to management. To extend the flood regime into the lateral dimension of the floodplain, **Chapters 4, 5 and 6** examine the Cosumnes hydrospatial regime, providing a framework for analysis of spatiotemporal floodplain inundation patterns. This follows recent research emphasizing quantification of hydraulic metrics (e.g., depth, velocity) over hydrologic metrics (e.g., peak flow magnitude) to describe floodplain inundation (Cienciala & Pasternack, 2017; Karim et al., 2012; Stone et al., 2017).

Chapter 4 formulates a hydrospatial analytical approach, informed by hydroecology and landscape ecology principles and utilizing hydrodynamic modeling and spatial analysis. The methodology was applied to a recent levee-removal site along the lower Cosumnes River to compare pre- and post-restoration configurations for the 110-year (1908-2017 water years) daily flow record. Using two-dimensional hydrodynamic modeling, spatially-resolved flow-depth and flow-velocity relationships were established, which were then combined with the daily flow time series to produce gridded time series of depth and velocity. Multiple metrics relating to inundation extent, depth, velocity, frequency, duration, timing, rate of change, connectivity, and heterogeneity were summarized across space and time. Results showed that responses to the levee removal differed depending on the metric examined, location within the floodplain, and flow magnitude. The multi-metric multi-dimensional approach presented in this chapter can provide insights into how specific conditions relevant to geomorphic and ecological floodplain processes change with restoration, offering a step to better link flow or landscape alterations to their ecological impacts.

The provision of spawning and rearing habitat for native fish is a key function of floodplains and a primary motivating factor for floodplain restoration activities (Beechie et al., 2013; Jeffres et al., 2008; Opperman et al., 2010). Building on the hydrospatial analysis approach of **Chapter 4**, **Chapter 5** quantifies and describes spatiotemporal variability of floodplain habitat. This is applied to the lower Cosumnes floodplain to quantify the habitat benefits of levee-removal restoration for Sacramento splittail (*Pogonichthys macrolepodotis*), a native floodplain fish species. Grid-based physical habitat suitability modeling was used to connect habitat suitability indices with daily gridded depth and velocity based on two-dimensional hydrodynamic modeling output. The spatially-resolved methodology allowed consideration of suitability criteria relating to duration and connectivity as well as depth and velocity. Overall, habitat availability nearly doubled with restoration, though relative benefits varied considerably in space and time. The relationship of habitat to flow was nonlinear and not one-to-one, indicating that complex floodplain environments mediate habitat availability. The novel approach presented in this chapter advances floodplain habitat quantification methods, illustrates how heterogeneous floodplains contribute to habitat across a range of flows, and can be used to evaluate management strategies and restoration designs targeting specific physical conditions in support of freshwater ecosystems.

Floodplain environments are valued for their potential to buffer against climate change impacts. **Chapter 6** addresses the need for improved understanding of the nature and degree of floodplain responses at spatial and temporal scales relevant for ecological interpretation and development of management strategies. Flow regime change – including altered runoff volume, increased extreme floods, and reduced spring snowmelt flows (Cayan et al., 2008; Das et al., 2013; Mote et al., 2018) – is a primary pathway for climate change to affect floodplains (Döll & Zhang, 2010). Applied to the lower Cosumnes River restoration site, this study uses the hydrospatial analysis approach of **Chapters 4 and 5** to quantify floodplain response to flow regime change under four climate change scenarios for a historical (1951-1980 water years) and future (2070-2099 water years) period and to compare the relative impact of these changes against those of levee-removal restoration. Physical metrics representing inundation extent, depth, velocity, duration, timing, and connectivity were evaluated. Potential changes to splittail habitat were also assessed, following the methods of **Chapter 5**. Spatiotemporal metric summaries reflected the balance between increasing extreme winter floods and declining spring floods in future flow scenarios. However, the direction and magnitude of change varied across metrics. The relative impact of levee-removal restoration was often, but not always, overwhelmed by flow regime changes. Overall, levee removal dampened changes between the historical and future periods. This study illustrates how flow-landscape interaction mediates floodplain responses to hydroclimatic change and suggests that physical restoration actions such as levee-removal may be useful for mitigating climate change impacts.

The research herein is motivated by the need to better understand the product of flow regime and landscape interaction to improve linkages to floodplain ecosystem processes and functions, to identify and describe anthropogenic change, and to develop efficient and effective water and land management strategies. As a contribution toward meeting these broad challenges, this dissertation provides literature synthesis, flood regime typology development, and the establishment and demonstration of a spatiotemporal and multi-metric approach to quantify floodplain inundation patterns under restoration and climate change scenarios. By adjusting water and land management practices, there is great potential for reestablishing dynamic and heterogeneous riverine landscapes in support of functional ecosystems that are diverse, productive, and resilient to future change.

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CHAPTER 2

RESTORING MODIFIED RIVERS BY MANAGING FLOWS AND LANDSCAPES TOGETHER: A CONFLUENCE OF IDEAS

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Restoring modified rivers by managing flows and landscapes together: A confluence of ideas

KEY POINTS

- River restoration approaches of habitat restoration and environmental flows have transformed in parallel from static to process-based goals.
- A recent confluence of ideas calls for integrated approaches to consider flow and landscape alternatives together in highly modified rivers.
- Large river restoration requires improved modeling and metrics as well as redefined goals for altered ecosystems in a changing world.

ABSTRACT

Modifications to landscapes and flow regimes of large rivers have altered the function, biodiversity and productivity of freshwater dependent ecosystems globally. Reestablishing morphological and hydrological conditions necessary to sustain ecosystems is a central challenge of restoration within these highly altered systems. There is now clear recognition that meeting this challenge requires simultaneously addressing multiple and interacting stressors. While in many cases physical habitat manipulation alone can achieve desired outcomes, most large rivers require some coupling of habitat and hydrologic manipulation. Historically, these two approaches – physical habitat restoration and environmental flow implementation – have often been implemented independently and with separate objectives. We trace the parallel transformations of different restoration approaches, from goals to reproduce static representations of form and flow regime to goals to reestablish processes. This represents a confluence of ideas for achieving cumulative benefits when habitat restoration and environmental flows are implemented together. The river restoration literature demonstrates that this coupling embodies a more holistic river restoration aimed at supporting resilient ecosystems within human dominated landscapes in a nonstationary climate. Under the larger umbrella of evolving river restoration science, this paper synthesizes current thinking to focus attention on the opportunities offered by evaluating integrated effects of landscape and flow restoration measures. It is an explicit juxtaposition of these disciplines to illustrate valuable capacity for addressing today's ecological challenges. We discuss existing examples of such integration and its context within broader process-based river restoration frameworks, and identify opportunities for further advancements in the restoration of highly modified rivers.

INTRODUCTION

Rivers and their freshwater ecosystems undergo profound and compounding change as a result of extensive human modification of streamflow, landscape, and climate. More than half of the world's large rivers are regulated by dams (Nilsson et al., 2005), which, along with long histories of leveeing, channelization, water abstraction, and land use change, have homogenized hydrologic, geomorphic, and ecological dynamics driving riverine ecosystem structure and function (Poff et al., 2007; Wohl et al., 2017). Problems multiply and intensify with biological impairments, such as invasive species, which fundamentally alter food webs and erode ecosystem resilience. Global freshwater ecosystem biodiversity is severely compromised (Meybeck, 2003; Vorosmarty et al., 2010). Increasingly, climate change amplifies threats and interacts with other factors of change, demanding proactive management strategies (Malmqvist & Rundle, 2002; Palmer et al., 2009). Degradation is particularly pronounced within large rivers and their lowland alluvial floodplain environments, which attract development and collect upstream impacts (Naiman et al., 2002). Despite being some of the most diverse and productive ecosystems, these freshwater-dependent ecosystems are also among the most impaired worldwide (Dudgeon et al., 2006; Tockner & Stanford, 2002).

Over the last several decades, recognition of the many factors affecting riverine ecosystems has led to rapid and expansive growth in the science and practice of river restoration. Applications have expanded from small-scale actions in headwater streams to new domains and challenges within large rivers (Beechie et al., 2009; Gore & Shields, 1995). Many efforts address individual problems at the site-scale, though other more comprehensive programs employ a multitude of actions at different scales. With the common goal of repairing ecosystem structure and function, typical restoration measures can be grouped generally into those that directly alter the landscape physically (e.g., channel redesign, levee removal, riparian plantings, land use restrictions) and those that modify flow regimes to encourage more natural hydrologic and geomorphic patterns and processes (Bernhardt & Palmer, 2011; Palmer et al., 2014). With the understanding that restoration of physical conditions alone – the "Field of Dreams" hypothesis of "if you build it, they will come" (Palmer et al., 1997) – does not guarantee success, there is also growing emphasis on addressing ecological limitations, such as altered food webs, through actions such as invasive species control or nutrient management (Acreman et al., 2014b; Bellmore et al., 2017; Lake et al., 2007; Naiman et al., 2012). In this discussion, we limit the scope to geomorphic and hydrologic river restoration approaches, focusing on the convergence of these underlying scientific perspectives that both seek to reestablish dynamic hydrogeomorphic processes.

In general, river restoration approaches have transformed from an early and primary focus on form or static definitions, such as creating river meanders or implementing a minimum instream flow, to a more recent focus on hydrologic, geomorphic, and ecological processes that address ecosystem functionality (Beechie et al., 2010; Kondolf et al., 2006; Palmer et al., 2014; Tharme, 2003; Wohl et al., 2015b; Yarnell et al., 2015). This shift is motivated by new scientific emphasis on management for hydrological and physical dynamics, heterogeneity, and connectivity. The flow regime is well-understood to be a master variable driving riverine and floodplain ecosystems, and the natural flow regime paradigm of Poff et al. (1997) solidified management for natural flow variability as a central tenet of river restoration (Poff & Matthews, 2013). The interaction of a variable flow and sediment regime with heterogeneous landscape patterns produces and maintains diverse conditions in space and time that support the ecological diversity and productivity for which these systems are known (Naiman et al., 2002; Ward & Stanford, 1995). The underlying basis for this understanding emerged from the flood pulse concept of Junk et al. (1989), which articulated the link between physical and biological processes, the ecological implications of a variable flood pulse interacting with complex physical structures, and the critical ecohydrological role of seasonal floodplains in large river systems. Later, Tockner et al. (2000) demonstrated the importance of river-floodplain interaction in relating hydrologic connectivity to species diversity. Hydrologic connectivity and its spatio-temporal variability is identified as a central restoration objective (Opperman et al., 2010; Ward & Stanford, 1995). However, restoring heterogeneous biophysical processes and related ecosystem functions in environments fundamentally altered by multiple pressures remains a central scientific challenge (Brewer et al., 2016; Palmer & Bernhardt, 2006).

Commensurate with the shift from form-based to process-based thinking – and its consideration of connectivity and variability in riverscapes (sensu Fausch et al., 2002) – restoration failures and unintended consequences have led to the recognition that habitat restoration actions or the implementation of environmental flows alone may be insufficient in heavily engineered and regulated rivers to achieve desired outcomes (Arthington et al., 2010; Kondolf, 2011; Palmer et al., 2014; Wohl et al., 2015a). The historical lack of integration between physical habitat restoration and environmental flows has hampered efforts to rehabilitate highly degraded riverine ecosystems, which has been noted by others (Arthington et al., 2010; Kondolf et al., 2006; Palmer et al., 2014; Wohl et al., 2015b). This paper focuses attention on this confluence of ideas with the goal of advancing the science and practice of restoring functional ecosystems in highly modified rivers, defined here as typically larger river-floodplain systems degraded by both landscape and hydrologic regime alteration (sensu Yarnell et al., 2015). Specifically, we provide a brief summary of these scientific perspectives, synthesize how each perspective enters the discourse of the other, and discuss examples of newly integrated approaches. Further, we articulate how integrated hydrogeomorphic thinking helps recognize opportunities for managing land and water together within existing frameworks to sustain dynamic processes that support diverse and productive ecosystems. We conclude by identifying promises and challenges moving forward.

EVOLUTION OF GEOMORPHIC AND HYDROLOGIC RESTORATION APPROACHES

As two subfields of river restoration, riverine habitat restoration and environmental flows literature has evolved in parallel over several decades with greater convergence recently, primarily where highly modified rivers and their floodplains are the focus. A simple keyword literature search suggests similar growth trajectories in these two fields, with around 150 citations per year each in the last decade up from around 50 in the previous decade (Figure 2-1). This analysis suggests that crossover between the two fields is increasing yet relatively low, at around 10 cross-citations per year (for the selected keywords). The separation between the fields is largely due to the historical independence of the problems each addresses, as well as their technical, economic, and socio-political motivations. As noted by Bernhardt and Palmer (2011), geographic locations where water is generally plentiful but highly urbanized and/or polluted have historically inspired habitat-based approaches to ecosystem restoration while water scarcity and subsequent infrastructure development to move and store water in arid and semi-arid environments, often in agricultural contexts, has catalyzed efforts to better allocate flows for ecosystems. For example, a suite of responses to repair streams degraded by contaminated urban runoff and loss of riparian forests would be expected to be quite different from those addressing the reduction of flood peaks due to dams. This has naturally attracted different scientific disciplines to address the variety of

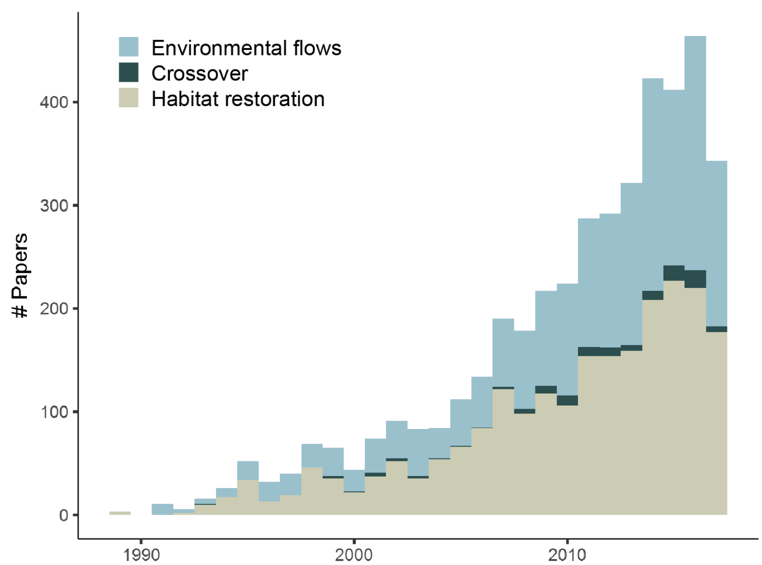

Figure 2-1. Trend over time in environmental flow and physical habitat restoration literature, including crossover between the two. Literature selected based on Web of Science ® topic searches using the following two statements to represent environmental flows and habitat restoration, respectively: 1) (“flow*” NEAR/1 “ecolog*” OR “environmental” NEAR/1 “flow*” OR “instream” NEAR/1 “flow*”) AND “river*”, 2) (“geomorph*” NEAR/1 “restoration*” AND “river*”) OR (habitat NEAR/1 restoration AND “river*”) OR “river*” NEAR/1 “restoration.” Literature meeting both of these criteria were classified as “crossover.”

context-dependent challenges. More recent crossover reflects the need for and growth in interdisciplinary approaches to river restoration (Acreman et al., 2014b; Newson et al., 2012; Palmer & Bernhardt, 2006).

Furthermore, with many physical changes originating at the reach scale, many habitat restoration responses have developed with limited geographic scope, in isolation from the larger landscape context. Conversely, flow alteration typically occurs at a much broader geographic scale such that the scope of environmental flow strategies rarely resolve reach scale needs (Arthington et al., 2010; Beechie et al., 2010). The applied side of river restoration is often disjointed, as management of land and water resources is typically conducted separately by different organizations, at different spatial scales, or by different policies (Gumiero et al., 2013; Newson et al., 2012; Poff & Matthews, 2013). Watershed management plans and restoration programs attempt to address these disconnects by providing a basin-scale context and by identifying suites of promising restoration actions, from site-specific habitat restoration actions to broad water infrastructure and policy reforms. However, a watershed plan alone does not necessarily translate to coordinated restoration efforts. In a review of California river restoration, Kondolf et al. (2007) found that although plans were often present, they often did not substantially influence project outcomes. In a separate study, Kondolf et al. (2006) analyzed restoration trajectories along axes of streamflow variability and connectivity (representing habitat restoration-type activities). They concluded that restoration actions rarely align with causes of degradation because some remedies, such as hydrologic connectivity, are easier – politically and economically – than others. In many cases, causes of degradation serve valuable economic and social purposes, such as flow alteration resulting from irrigation and hydropower, which require extensive infrastructure and investment. Thus, the relatively more costly environmental flow options may incentivize the use of physical habitat restoration as a surrogate for restoration (e.g., Rieman et al., 2015).

Geomorphic restoration transformation

Responding to growing recognition of environmental degradation, the scientific field of river restoration has expanded rapidly over the last 50 years, broadly speaking. In the United States, the environmental regulations of the late 1960s and 1970s inspired much of the early growth (Lave et al., 2010). Overall, both science and practice have typically focused on remedying local-scale conditions within smaller stream systems (Bernhardt et al., 2005; Roni et al., 2008). Traditionally, under this river restoration umbrella, the practice of habitat restoration has involved actions to reconfigure the physical structure of a river or stream to match a perceived “natural” form or to provide habitat for a particular species or, more recently, particular ecosystem services (Palmer et al., 2014; Wohl et al., 2005). Most habitat restoration activities are limited in spatial scope, tend to be prescriptive and engineering-oriented, and have been criticized as simplified applications of scientific concepts (Bernhardt et al., 2005; Kondolf, 1998; Lave et al., 2010; Palmer et al., 2014; Wilcock, 1997). These actions are often opportunistic, occurring on public land or at the request of a landowner (Clarke et al., 2003). Ideally, these physical manipulations – often focused on in-channel forms – either stabilize condition to stem further degradation or improve degraded sections of rivers toward a more desired state. However, applying static, reach-scale manipulations, without first understanding the context-specific sources of degradation may be ill-conceived (Montgomery, 2008) and counteract driving physical processes of the current system (Beechie et al., 2010), and consequently may not meet expectations (Bernhardt & Palmer, 2011). Expected benefits of riparian planting may not accrue, for example, when underlying factors of change are not addressed, such as land cover change or a declining water table (Giling et al., 2016; Stromberg et al., 2007). Generally, watershed-scale changes affecting hydrogeomorphic regimes or water quality – often a source of ecological degradation – are unrecognized, inadequately understood, or disregarded as outside the purview of the project (Kondolf et al., 2007; Palmer et al., 2014; Roni et al., 2008). Consequently, many reach-scale efforts do not address the root causes that can overwhelm such small positive actions, rendering goals unattainable (Beechie et al., 2010; Kondolf et al., 2006; Lake et al., 2007).

Clarification of these restoration limitations has motivated a shift toward process-based principles, including managing for dynamic habitats that respond to variability (Fryirs & Brierley, 2009; Kondolf et al., 2006; Palmer et al., 2005), planning at the scale of physical and biological processes (Beechie et al., 2010), accounting for local or landscape potential (Beechie et al., 2013b), and addressing causes of degradation as well as changes at the watershed level (Beechie et al., 2013b; Palmer et al., 2014; Wohl et al., 2005; Wohl et al., 2015b). A central concept is to set restoration objectives and develop actions based on processes rather than a fixed outcome (Beechie et al., 2010; Bernhardt & Palmer, 2011; Wohl et al., 2005). With a focus on how a river works, fluvial geomorphology has become an important contributor to restoration planning (Brierley et al., 2002; Gilvear, 1999). Some of the first process-based approaches were developed for salmonid habitat restoration in the Pacific Northwest (Beechie & Bolton, 1999). Recently, research in the Rocky Mountains, USA, suggests the removal of beaver historically was a primary driver of change and their reintroduction restarts important physical and biogeochemical processes (Wegener et al., 2017). Today, process-based principles underlie restoration of rivers and streams globally, at different scales and across a range of objectives.

Understanding hydrogeomorphic processes and how and why they have changed facilitates more effective and efficient restoration strategies (Beechie et al., 2013b). Process-based restoration protocols rely on watershed assessments to describe existing processes, factors of change, and potential limitations (Beechie et al., 2008; Roni et al., 2002). To develop strategic actions, reach-scale actions must be placed within a broader spatial context, from riverscape to watershed, where processes and impacts occurring at many scales are understood (Fausch et al., 2002). In highly modified rivers, such assessment helps identify how the restoration capacity of the landscape has shifted and points to restoration actions that are better aligned with the reality of these fundamentally altered systems (Beechie et al., 2013b; Clarke et al., 2003). A landscape-scale perspective is also increasingly needed to understand climate change implications and to maintain functional ecosystems within a range of possible future conditions (Beechie et al., 2013a; Williams et al., 2015).

The adoption of specific perspectives and objectives to address the multiple causes of extensive degradation in large rivers and their floodplains emerged more recently (Gore & Shields, 1995; Nilsson et al., 2007; Petts, 1989; Sparks, 1995). The complex nature of river-floodplain environments requires similarly complex restoration responses, including multidisciplinary approaches (Buijse et al., 2002). While there is general recognition that returning to some past state in these highly modified systems may not be feasible, working to restore processes can help maintain natural ecosystem functions (Opperman et al., 2010; Stanford et al., 1996). Process-oriented actions in large lowland alluvial rivers typically work to restore hydrologic connectivity and floodplain geomorphic processes (Wohl et al., 2015b). From the geomorphic restoration perspective, levee removal, for example, has been shown to restart erosional and depositional processes, reestablishing complex topography and a mosaic of dynamic habitats important for riparian forest succession and biodiversity (Florsheim & Mount, 2002).

Hydrologic restoration transformation

The scientific management of water for ecosystems, or the field of environmental flows, has also shifted from more static approaches (e.g., minimum flow, percent flow) to variable flows that support ecological and geomorphic processes and functions (Arthington et al., 2006). Arising initially as minimum flow targets for fisheries below dams in the late 1970s, the environmental flows literature now encompasses a broad array of purposes and approaches (Petts, 2009; Tharme, 2003). This shift toward more process-based thinking is represented in the widespread adoption of the natural flow regime paradigm by Poff et al. (1997), which emphasized that ecosystems are adapted to flow dynamics and variability, expressed by flow regime components of magnitude, frequency, duration, timing, and rate of change. More recent conversations suggest that climate non-stationarity and novel ecosystems require advancing the field of environmental flows beyond defining static regimes based on the natural flow regime toward dynamic flow prescriptions defined by process-based links to ecological responses (Acreman et al., 2014a; Poff, 2017).

Methods for setting flows can be categorized as hydrologic, habitat simulation, or holistic (Tharme, 2003). Hydrologic methods use the historical flow record and statistical measures of the annual hydrograph, including some of the earliest methods for establishing minimum instream flows (Tennant, 1976) as well as the more involved and widely applied Indicators of Hydrologic Alteration and Range of Variability Approach (Richter et al., 1997). As the natural flow regime is rarely achievable, particularly in highly modified rivers, the resulting compromised flow targets are often criticized for lacking a process-based grounding connected to ecological functions. Habitat simulation methods typically involve hydraulic modeling to determine flows meeting desired habitat conditions. The most common is the Instream Flow Incremental Methodology and Physical Habitat Simulation technique, which has been evolving since the early 1980s (Bovee, 1982). While these methods link directly to ecological requirements, they lack connections through fluvial geomorphic processes and are often limited to in-channel habitat for particular fish life history functions (Clarke et al., 2003; Petts, 2009). More holistic methods, such as Ecological Limits of Hydrologic Alteration (ELOHA; Poff et al., 2010), are increasingly common given their improved capacity to account for multiple ecological and geomorphic objectives and constraints in flow alterations and ecological responses. Developing appropriate and useful flow-ecology relationships for these methods is an important area of research (Arthington et al., 2010; Davies et al., 2014). Holistic methods include the "bottom-up" concept of managing for specific flows or components of a flow regime for the processes or ecological functions they serve, particularly important for prioritizing flow targets in heavily regulated systems. Such ideas are found in the early holistic Building Block Methodology (King & Louw, 1998) as well as more recent emphasis on "functional flows" (Yarnell et al., 2015). As the field of environmental flows grows, continued scientific advancements for establishing and evaluating meaningful environmental flows in highly altered rivers is an identified need (Brewer et al., 2016).

With most flow prescriptions developed based on ecological requirements directly served by flows, scientists have appealed for more attention and sophisticated methods to address transport processes and management of the natural sediment regime alongside the natural flow regime (e.g., de Jalón et al., 2016; Wohl et al., 2015a; Yarnell et al., 2015). Some prominent examples of geomorphic-focused flow objectives exist, such as the high flow experiment protocol for the Colorado River, where floods released from Glen Canyon Dam are specifically designed through modeling for their capacity to move sand and thereby meet multiple objectives, including fish habitat (Melis et al., 2015; Schmidt et al., 2001; Wright & Kaplinski, 2011). These floods are implemented alongside other species-oriented flow requirements. However, it is less common to find examples in the literature where the potential effects of geomorphic alteration options on environmental flows are considered (de Jalón et al., 2016). In general, despite calls for a focus on processes and variability of the natural flow regime to support a range of geomorphic and ecological functions, environmental flow prescriptions to date typically target individual species or geomorphic processes or mimic hydrograph shape in the absence of function or process justification (Acreman et al., 2014a; Arthington et al., 2006; Yarnell et al., 2015).

RECOGNIZING LIMITATIONS OF ISOLATED APPROACHES

There are limits to what habitat restoration or environmental flows alone can achieve within highly modified rivers. Most large rivers have undergone a variety of physical alterations (e.g., levees disconnecting floodplains) as well as modification of flow regimes (e.g., dams), which alter the availability of water in space and time, sediment and thermal regimes, and disturbance processes (Ward & Stanford, 1995). In systems such as these with interdependent causes of degradation, addressing problems individually may not achieve desired outcomes. Motivations for environmental flows tend to revolve around what combination of flows – rather than what combination of habitat restoration actions and flows – can produce desired conditions. Similarly, a habitat restoration perspective tends to produce questions focused on physical changes, sometimes in an effort to compensate for the larger watershed context including hydrologic regime change. While both scientific perspectives maintain a robust literature, the previous lack of integration has limited advances in both science and practice (Arthington et al., 2010; Kondolf et al., 2006; Palmer et al., 2014; Wohl et al., 2015b).

Historically, adverse effects of flow regime alteration have rarely been adequately assessed and accounted for in reach-scale habitat restoration projects (Tockner & Stanford, 2002). With regulated flows, simply providing room for the river to reestablish processes may not be sufficient (Kondolf, 2011) and may constrain opportunities for ecological improvement (Wohl et al., 2005). In such cases, restoration designs based on channels formed and maintained by natural flow and sediment regimes may be inappropriate, yielding unexpected results (Clarke et al., 2003; Kondolf et al., 2006; Wohl et al., 2015b). For example, levee removal in a highly regulated river may fail to achieve floodplain inundation frequency targets (Matella & Merenlender, 2015). Simply put, restoring only the morphological side of the dynamic interactions of flow and landscape is unlikely to produce the spatio-temporal variability and complexity of environmental conditions to which ecosystems are adapted. Therefore, process-based approaches require understanding of the larger-scale context, including hydrologic modifications, to encourage the diverse habitat mosaics and fluvial dynamism upon which ecosystems depend (Beechie et al., 2010; Rieman et al., 2015).

With this recognition, the role of altered flow regimes and sediment regimes in habitat restoration outcomes has become a larger part of restoration planning conversations. For example, the eco-hydrogeomorphic approach of Clarke et al. (2003) involved determining appropriate forms and strategic site selection of projects given altered landscapes and hydrologic regimes. Rohde et al. (2006) accounted for watershed-level alterations by examining physical and socio-economic factors affecting floodplain restoration, including bed load and hydrologic connectivity. Trush et al. (2000) and Downs et al. (2011), on two different rivers in California, proposed encouraging beneficial processes through downscaling river morphology to match altered flow regimes. This hydrogeomorphic concept aligns with Ward and Stanford (1995), who suggested that physical changes can be made to counter the impacts of altered flows. Far less common in the scientific literature, however, is the inclusion of flow adjustment options alongside reach-scale restoration alternatives, which allows for exploration of how managing flows in conjunction with habitat restoration actions may improve chances for success (Beechie et al., 2010).

Conversely, approaches to setting environmental flows characteristically rest on the assumption that the appropriate landscape exists with which flows can interact. This is challenged by myriad changes at multiple scales, such as sediment regime alteration, leveeing and channelization of rivers, as well as groundwater extraction, water quality degradation, and invasive species (Davies et al., 2014; Wohl et al., 2015b). Geomorphic alteration can influence the effectiveness of environmental flows (Meitzen et al., 2013). Missing geomorphic components may limit the reestablishment of dynamic physical processes, and thus make it difficult to meet ecological objectives. For example, reintroducing high-magnitude, long-duration floods of a more natural flood regime may not produce positive results if the sediment to be moved by the floods is no longer present (Arthington et al., 2010; Wohl et al., 2015a). A study by Tracy-Smith et al. (2012) on Missouri River sandbar habitat showed natural and environmental flow alternatives were less beneficial than regulated flows within the context of the channelized reach morphology of the river. Furthermore, some caution that reinstating historical flow regimes may in fact be detrimental to ecosystems (Brewer et al., 2016; Palmer et al., 2014). Spatial and temporal constraints, such as land available for floodplain processes and the timing and duration of associated flood pulses within the floodplain, are not often a part of environmental flow assessments (Lake et al., 2007; Opperman et al., 2010; Yarnell et al., 2015). As Tockner et al. (2000) articulate, variable flooding processes drive floodplain ecosystems, which depend on complex floodplain landscapes as well as flow variability.

Researchers have acknowledged these issues with recommendations to set environmental flow prescriptions based on their geomorphic context (Dunbar & Acreman, 2001). Poff et al. (2010) state that a central challenge to environmental flow advancement is including the effects of flow-landscape interactions. Others have warned that defining the natural flow regime itself is not straightforward in highly modified rivers with a long history of change that cannot be undone (Acreman et al., 2014a; Petts, 2009). Very different flow targets may result depending on whether a natural flow regime is some description of a past state or defined for its potential within the existing landscape and/or future climate. While many environmental flow approaches consider flow alternatives for their effects within the context of current geomorphic conditions to some degree or another, similar to habitat restoration applications, options to adjust the physical landscape to optimize environmental flow benefits are rarely included.

CONFLUENCE OF IDEAS

Situations where multiple causes of degradation span both hydrologic and geomorphic realms and cannot be addressed independently have become increasingly clear (Palmer et al., 2004). There is general consensus that greater ecosystem benefits are achievable when multiple problems are addressed together from a process-oriented perspective. Because hydrologic and geomorphic alterations typically do not occur in isolation and subsequently interact with one another, the literature has called for remedies that similarly address that interaction, and there are an increasing number of studies reflecting this (see Figure 2-1).

In reviewing the literature that supports integrated river restoration efforts, including research and review-type papers, we characterized the nature of integration to better understand the composition and transformation of this literature over the last several decades (Figure 2-2). Papers were classified, in a range generally grading from less to more integration, as 1) considering multiple approaches to address problems, though without evaluating the mutual benefit of integrated implementation, 2) addressing limitations in the absence of integrated approaches, 3) developing methods applicable to integrated approaches, 4) suggesting enhancing integrated efforts (though not as a focal point), 5) emphasizing the importance of integration (typically review-type papers), and 6) demonstrating integrated approaches. In addition to the rapidly growing number of papers, the degree of integration has also generally increased over time. This literature, with examples discussed in the following section, marks an important advancement within river restoration science, and provides opportunities to advance the integration of habitat restoration and environmental flows to support functional ecosystems within highly modified rivers.

The literature supports that restoration focused on meeting objectives through a combination of landscape alternatives and flows – including interactive benefits – has the greatest likelihood for success within highly modified rivers. The river restoration review of Wohl et al. (2005) is one of several publications identified that clearly express the need for such questions. In another, Beechie et al. (2010) stress that restoration of large river ecosystem functions must consider multiple restoration actions of all types for their cumulative power to achieve goals. A similar sentiment is found in Arthington et al. (2010), which challenges the river restoration community to adopt a "whole of water cycle approach" to examine the collective impacts. Palmer et al. (2014) encourage functional restoration that looks beyond the channel for watershed-level actions that address hydrogeomorphic, biogeochemical, and ecological processes. Greater effort to implement environmental flows in conjunction with floodplain restoration actions to generate ecologically beneficial flooding is a particular emphasis (e.g., Hughes & Rood, 2003; Lake, 2012). Given that processes at multiple spatial scales drive floodplain ecosystem function, Hughes and Rood (2003) highlight the value in restoration measures that reflect those scales. The "ecohydraulic trinity concept," proposed by Katopodis (2016), is an expression of this increased focus on integrated approaches by its recognition of shared principles and potential synergies for co-implementation across habitat restoration, environmental flows, and a third identified field of passage infrastructure for aquatic species. These and other publications indicate the shift toward developing river restoration measures for their collective effects on processes driving ecological functions.

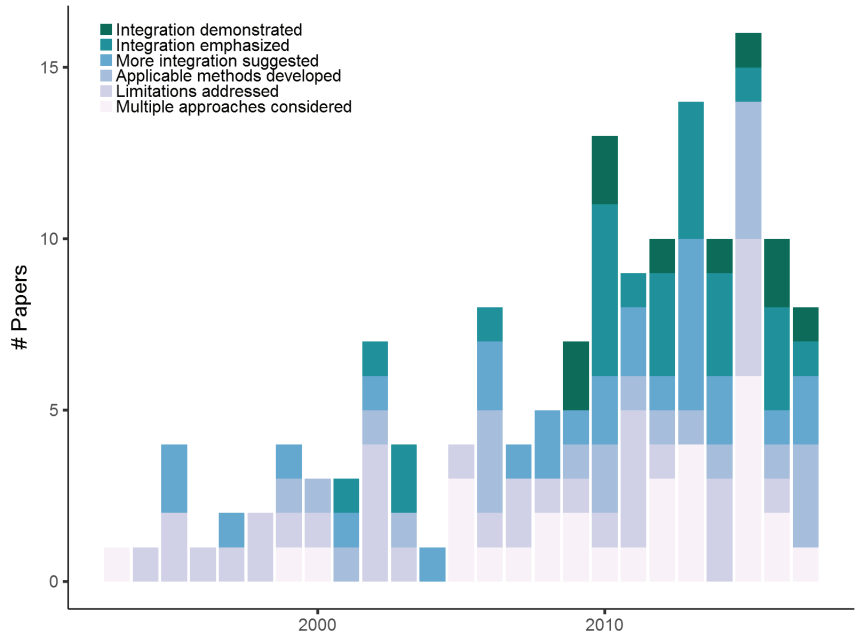

Figure 2-2. Trends in river restoration literature is shown by the number of published papers by year as a function of habitat and environmental flow integration in the rehabilitation of highly modified rivers. The earliest paper identified was from 1990.

Integrated river restoration planning requires describing driving processes of the functional ecosystem envisioned – including hydrologic, geomorphic, and ecological interactions – from which flow prescriptions and landscape elements necessary for those processes are selected. In highly modified systems, this may involve altering landscapes and/or flow regimes in ways that do not reflect historical appearances (Jackson & Pringle, 2010; Meitzen et al., 2013; Nilsson et al., 2007; Stanford et al., 1996). Integrated approaches provide a clearer understanding of degradation causes and consequences and allow selection of options with greater potential to succeed ecologically or feasibly implement. Figure 2-3 illustrates this concept by posing a conceptual example of potential benefits accrued when combining habitat restoration and environmental flow strategies. Fundamental to this integrated concept is that it is not simply the addition of environmental flow and habitat restoration prescriptions, but rather coordinated planning and implementation of actions to best achieve target outcomes.

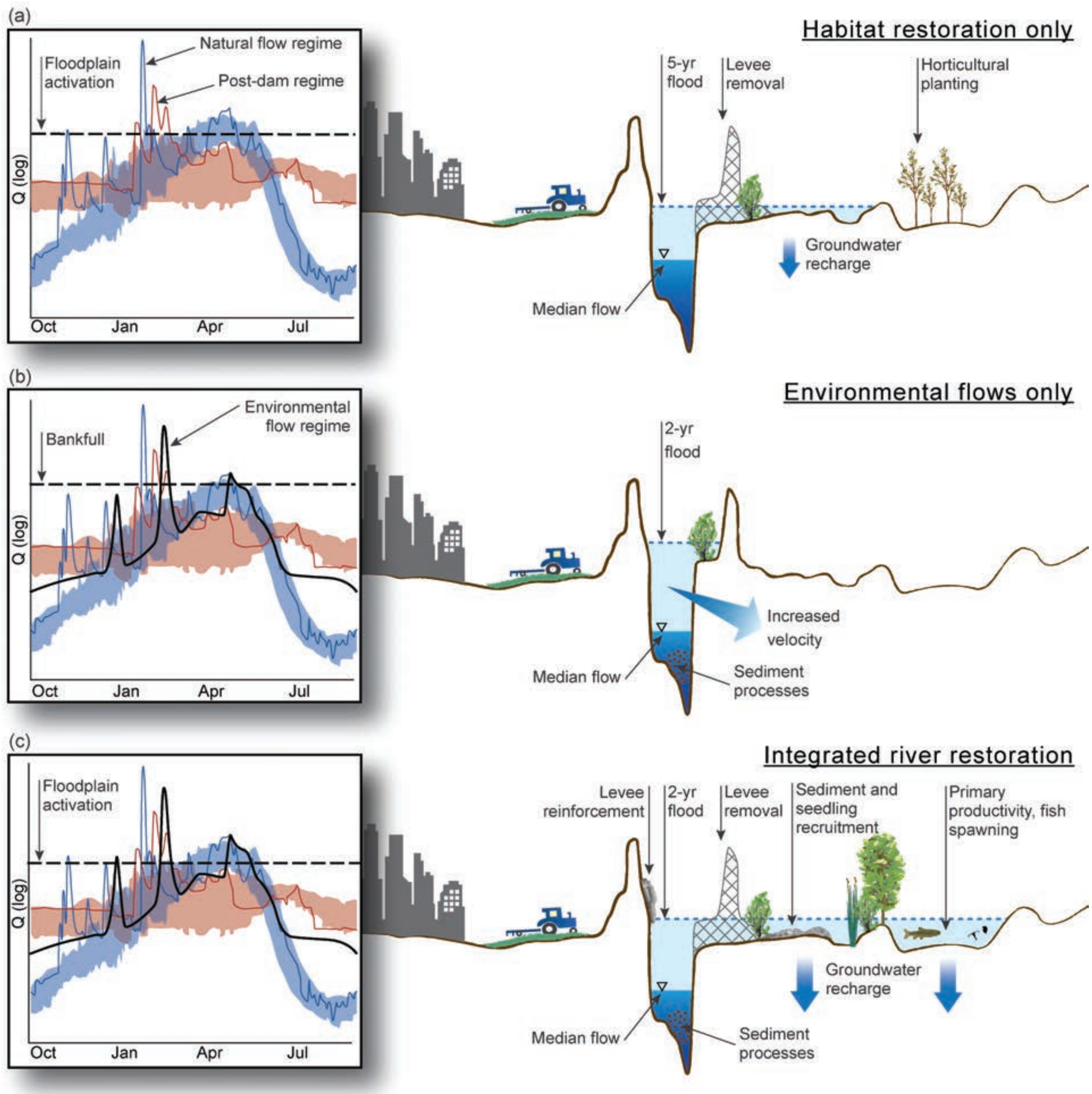

Figure 2-3. Conceptual graphic illustrating how habitat restoration (a) and environmental flows (b) can each accrue certain benefits when applied independently. The two approaches, however, if implemented together in an integrated way, can generate greater benefit than each alone (c). Inset hydrographs depict the natural flow regime (blue) and post-dam regime (red) as median daily flow with shaded 0.25-0.75 percentile range for a snowmelt dominated system in western United States. The environmental flow regime is shown in black.

Examples of integration

Recent advances within the field of river restoration provide useful examples of integrating habitat restoration and environmental flow approaches. The emphasis on comprehensive watershed-level planning and process-based restoration are common themes for articulating mutually beneficial actions for restoring hydrologic and geomorphic dynamics. Here, we identify examples of studies and applications involving integrated approaches, including new techniques afforded by modeling advances, opportunities within current environmental flow methodologies, and application within large-scale planning efforts. We focus on examples that move beyond addressing multiple stressors individually and consider interactions between or collective impact of habitat restoration and environmental flow measures.

Various studies now include the effects of different combinations of restoration measures. A recent study by Beechie et al. (2015) was expressly developed as an alternatives analysis to evaluate a set of restoration actions together, in contrast to typical approaches that consider actions individually. With the goal of improving habitat potential for salmonids along the regulated Trinity River, CA, habitat construction and channel alterations were evaluated separately and jointly based on relationships found in existing data and via Monte Carlo simulation. Although environmental flow alternatives were not included in the analysis, this scenario-based approach could easily encompass such options. A recent field experiment compared collective and individual effects of restoration measures in the Colorado River Delta of North America and suggested successful vegetation recruitment required co-implementation of environmental flows and riparian vegetation enhancement (Schlatter et al., 2017). A decision support tool developed by Stewart-Koster et al. (2010) applied Bayesian networks for several restoration case studies, demonstrating a statistical approach to address the problem of multiple stressors treated in isolation. Evaluating the interaction of multiple stressors is also useful. For example, a channel migration study for the Sacramento River, CA, examined cumulative effects of flow alteration and channelization to find some migration processes could be reintroduced with riprap removal (Fremier et al., 2014). Gilvear et al. (2002) steps through the history of change along Scottish rivers, discussing multiple interacting hydrologic and geomorphic responses to flow regulation and land use changes. This understanding was used to inform the selection of flow- and form-based approaches in support of salmonid populations.

Certain environmental flow-oriented studies have shown substantial capacity for inclusion of dynamic hydrogeomorphic interactions (Meitzen et al., 2013), and could be extended to evaluate habitat restoration options and other management alternatives as well. At a project-specific level, Katopodis (2016) presents two examples of flow releases below Canadian hydroelectric projects planned in conjunction with fish passage and habitat restoration measures. The holistic ELOHA framework includes subclassification – though often not applied (Meitzen et al., 2013) – based on local geomorphic attributes, which can address the interaction of flow with geomorphic conditions (Poff et al., 2010). Through this, physical habitat alterations could be included, particularly as options to improve ecological outcomes given flow restoration limitations (Arthington & Pusey, 2003). A recent ELOHA application in California analyzed expected benthic invertebrate community outcomes as a result of changing flow releases below a dam and applying land use measures to reduce watershed imperviousness (Stein et al., 2017). Though these two cases were evaluated separately, the methods applied in this study could easily extend to evaluating scenarios where multiple actions are simulated for their joint impact.

Advances in modeling and analysis of high resolution datasets offer new opportunities for design and evaluation. Progress in coupling hydrologic and hydrodynamic modeling support integration techniques spanning multiple scales, as discussed in the recent environmental flow publication by Brewer et al. (2016). For example, Tracy-Smith et al. (2012) concluded that restoration of both "flow and form" was necessary to improve outcomes in highly modified rivers like the Missouri River and called for a modeling framework that jointly evaluated geomorphic and hydrologic conditions. Hydrodynamic and water quality modeling for the Shasta River, CA, was used to compare improvements to quantity and quality of instream fish habitat across different combinations of restoration options, including environmental flows and riparian planting (Null et al., 2010). Though not focused on restoration applications, a recent study by Guse et al. (2015) provides a useful example of coupled hydrologic-hydrodynamic modeling to examine expected ecological responses of combined effects of climate and land use change.

Hydrodynamic modeling allows quantification of hydraulic variables such as depth, velocity and duration, and the utility of such hydraulic metrics over hydrologic metrics (i.e., flow regime metrics) for quantifying processes that link to ecological functions has been noted in a number of publications (e.g., Bond et al., 2014; Kozak et al., 2016; Turner & Stewardson, 2014). For example, Hudson et al. (2012) found indices of hydrologic connectivity captured floodplain oxbow lake dynamics more effectively than hydrologic metrics solely based on flow regime. In another example, hydraulic metrics were found effective in developing flow-ecology relationships for an environmental flows assessment for the Mara River Basin in Africa (McClain et al., 2014). In sum, metrics that better link to sediment dynamics (Wohl et al., 2015a), floodplain inundation patterns and dynamics, as well as habitat conditions, enable more effective evaluation of integrated options.

Several floodplain restoration studies provide explicit evaluation of integrated flow and habitat restoration alternatives. In one of several studies on floodplain restoration along the highly regulated and modified Missouri River, USA, Jacobson and Galat (2006) examined shallow water habitat potential under combined effects of altered flow regimes and channel forms using two-dimensional hydrodynamic modeling. In a larger-scale application involving water-surface elevation modeling, topography and soil, Jacobson et al. (2011) evaluated floodplain connectivity potential to identify local-scale habitat restoration opportunities coupled with environmental flows implementation. Further downriver within the Atchafalaya Basin, USA, Kozak et al. (2016) found that certain environmental flow prescriptions could enhance the effectiveness of small-scale habitat restoration projects. Another large river study examined floodplain connectivity for different permutations of flow and physical alteration options on the Rhône River, showing that options with flow increases along with other measures accrued the greatest benefit (Besacier-Monbertrand et al., 2014). In the Central Valley, CA, levee setbacks and other physical alterations performed in conjunction with environmental flows have been shown to promote dynamic floodplain processes that would be limited were integrated approaches not considered (Merenlender & Matella, 2013). Following this, scenario analysis for the San Joaquin River revealed tradeoffs between levee setback alternatives, floodplain grading options, and environmental flow regimes, demonstrating that both flow and physical restoration actions were needed to meet habitat goals (Matella & Merenlender, 2015).

Watershed-level planning and management is well-positioned to capture the complex nature of restoring highly modified rivers. The need for watershed-level approaches is reflected in restoration programs and institutions formed to develop and implement basin plans in many highly-developed river basins of the world. Two recent ambitious water policy reform initiatives, the European Union Water Framework Directive and Australia's National Water Initiative and its subsequent Murray-Darling Basin (MDB) Plan, include basin-planning that considers a range of potential measures and coordinates efforts to support freshwater ecosystems. These larger planning processes have the capacity to improve consideration of floodplain restoration (Gumiero et al., 2013; Hughes & Rood, 2003), the effects of land use on flow regimes (Gilvear et al., 2002), and integrated land and water management (Gonzalez Del Tanago et al., 2012). While basin plans encompass a range of actions, achieving balance can be difficult. Thus far, most planning and action in the MDB concerns the provision of environmental water despite important opportunities for additional complementary actions (Pittock & Finlayson, 2011). A large coordinated program to address hydropower impacts to fisheries has evolved for over three decades in the Columbia River Basin of the Pacific Northwest, USA. While ecological objectives have evolved from a narrow fish focus to a more comprehensive ecosystem perspective, implementing integrated ecosystem management through subbasin plans is an ongoing challenge (Leonard et al., 2015). Large-scale ecosystem-based management is also the focus within the highly engineered Sacramento-San Joaquin Basin, CA. Though many project-based restoration actions have been criticized as piecemeal, guiding documents have identified complementary restoration actions and emphasize evaluating projects for their potential to contribute to ecosystem-level impacts (Luoma et al., 2015). A common challenge to large-scale planning and management directives is following through with implementing plans and coordinating individual projects. The details of executing such plans is where methods to design and evaluate the joint impact of specific restoration measures are of critical importance.

Integrated river restoration implementation

The larger umbrella of river restoration and, within it, application of process-based thinking support the co-implementation of habitat restoration and environmental flows in highly modified rivers. Figure 2-4 illustrates how these restoration elements for such systems fit into a broadly applicable river restoration framework (Skidmore et al., 2013). In the first step, as outlined by Skidmore et al. (2013), problems and their root causes are identified, ideally through an initial watershed assessment. For highly modified rivers, this requires understanding feedbacks between multiple geomorphic and hydrologic changes from the reach- to watershed-scale. Though the initial incentive for restoration may be a single perceived problem, this step may illuminate multiple related problems and causes. Secondly, establishing the larger project context involves articulating multiple past and potential future trajectories of change, which is important to understanding how interactions between factors affecting ecosystems transforms over time. After defining goals and objectives (step 3), the fourth step includes understanding ecological responses to multiple interacting stressors occurring at different scales through measurement and/or modeling. Options for co-implementation of habitat restoration and environmental flows are explored under the fifth step, alternatives evaluation. This requires hydrologic and hydrodynamic modeling to capture the interplay between prescribed environmental flows and restoration actions, such as levee setbacks, as well as subsequent ecological responses. This may include coupling models operating at different scales. The selection of metrics that represent the outcome of hydrogeomorphic interactions relevant to ecological functions to evaluate options is particularly critical. In selecting the final restoration elements and design (step 6), alternatives best satisfying targeted processes within highly modified rivers defined by irreversible changes and constraints may involve actions that depart from restoration of a natural flow regime or reference channel morphology.

The specific elements highlighted in Figure 2-4 for highly modified rivers also align with general river restoration principles emphasizing the selection of a suite of restoration actions – geomorphic and hydrologic as well as biological, biogeochemical, or water quality restoration measures – to address multiple causes of degradation (Beechie et al., 2008). Developing effective holistic strategies rests on a process-based and landscape-scale understanding of the system that illuminates connections and interactions between physical processes, anthropogenic alteration, and ecological structure and function

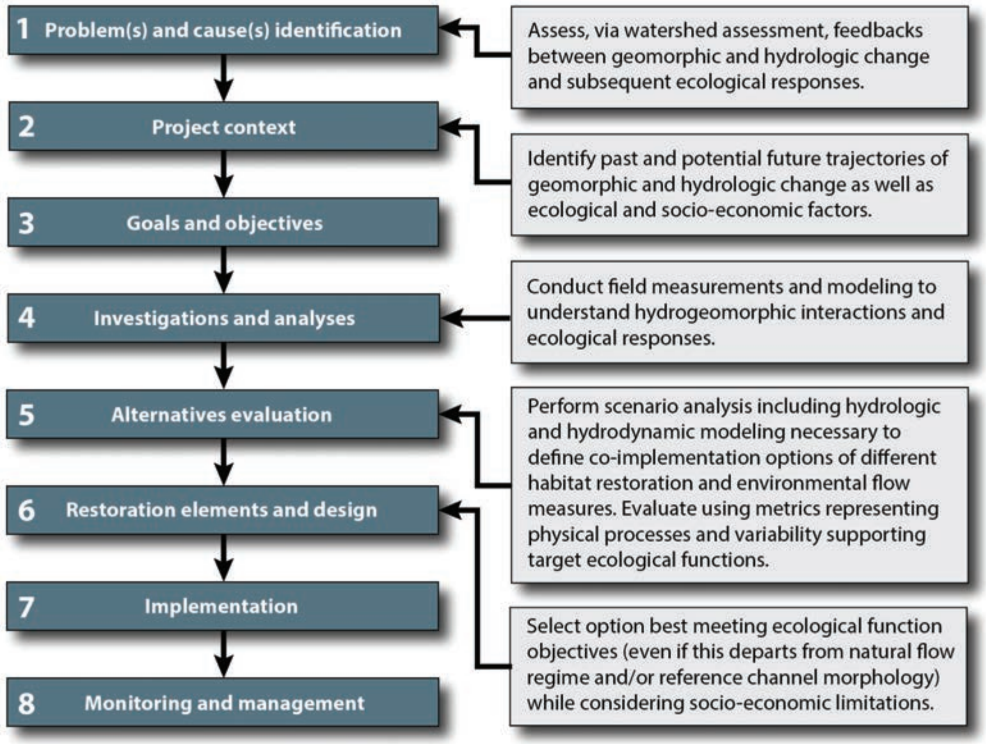

Figure 2-4. Elements specific to co-implementation of habitat restoration and environmental flow measures for engineered and regulated rivers (right column) and their fit within a general river restoration framework (on left, adapted from Skidmore et al. (2013) and Perry et al. (2015)).

(Beechie et al., 2010; Palmer et al., 2014; Wohl et al., 2005). Addressing variability and complexity across both habitat restoration and environmental flow approaches facilitates integration. In turn, habitat restoration and environmental flow actions directly affect and interact to promote and sustain variability and complexity, and thus desired ecological objectives. Aforementioned restoration frameworks emphasize the importance of understanding how multiple drivers of change interact to impact ecological functions and also encompass integrated evaluation and management of habitat restoration and environmental flow measures (and, increasingly, other measures such as food web based actions). The expectation of successfully applying these elements for highly modified rivers is the achievement of greater functional benefit than if each measure was developed independently to address different problems.

MOVING FORWARD

The understood need for and recent rise in integrated restoration approaches, particularly in highly modified rivers, indicates there are opportunities for improvement (Table 2-1). Though guiding principles and comprehensive restoration programs exist and methods for collective evaluation of restoration measures have become more sophisticated, application in practice is neither easy nor commonplace (Katopodis, 2016). For one, identifying the most promising suite of restoration actions requires improved modeling of processes at watershed scales coupled with hydraulic modeling capturing reach-scale processes (Brewer et al., 2016; Kozak et al., 2016). This includes advancing scenario-based integrated modeling tools for determining effective co-implementation of restoration measures within the limitations posed by highly modified rivers. Research addressing linkages between rivers and their floodplains is particularly promising. Modeling advances should support concurrent development of

Table 2-1. Opportunities for advancing the next phase of integrated restoration in highly modified rivers.

Learn more by doing through applied integrated restoration planning. Scientific insights should inform the management of large rivers.
Advance coupled watershed process and hydraulic modeling. Reach-scale effects of larger landscape-scale factors and response to large- and small-scale restoration actions should be evaluated in a process-based way.
Develop new metrics capturing physical process responses that reflect the interactive effect of multiple restoration actions.
Establish physical process-ecological function linkages through advancing fundamental scientific research and developing straightforward quantification methods.
Incorporate social sciences in both the planning and implementation of restoration. This includes methods to better align land and water management.
Include interactive effects due to climate change in the evaluation of restoration actions and select strategies most robust under a range of potential futures.
Develop realistic definitions for restoration targets in the context of limitations posed by human uses, fundamentally altered or novel ecosystems, and a nonstationary climate, while not losing sight of restoring natural processes that support functional and resilient ecosystems.

measureable metrics that represent hydrologic and geomorphic interactions as well as resulting ecological functions. Additionally, this work should be informed by continued fundamental scientific research into how ecological processes and functions are supported by hydrogeomorphic processes. These efforts should be pursued in conjunction with other restoration priorities, including improving internal ecosystem functioning by addressing food web dynamics, invasive species, biogeochemical processes, and water quality (Brewer et al., 2016; Naiman et al., 2012; Palmer et al., 2014; Petts, 2009; Wohl et al., 2015b). Promising directions are found in several recent publications that expand integrated restoration approaches to include food web dynamics. For example, changes to restoration outcomes as a function of interaction with habitat restoration measures is explored by Bellmore et al. (2017) and with environmental flows by Robson et al. (2017).

A fundamental challenge to implementing integrated approaches is that they are necessarily large-scale, process-based, and involve multiple interacting factors. They are also politically and fiscally cumbersome as well as scientifically complex, requiring substantial institutional, social, political, economic, and technical support (Wohl et al., 2015b). For continued momentum toward comprehensive river restoration, considerable effort will need to be devoted toward informing and engaging policy-makers and the public (Rieman et al., 2015). This includes expanding restoration science to include social science in the development, implementation, and evaluation of plans and projects (Naiman, 2013; Palmer et al., 2014). Furthermore, reforms to overcome institutional barriers and better align water and land management in both policy and institutional structure facilitate coordinated implementation of strategies (Bunn, 2017).

Restoring ecosystem functions in today's riverine landscapes must occur within the context of nonstationary climatic drivers. Climate change places additional strain on degraded riverine ecosystems, interacts with other stressors, and shifts the effectiveness or feasibility of restoration objectives and activities, potentially requiring new or adjusted measures (Palmer et al., 2009; Perry et al., 2015; Pittock

& Finlayson, 2011). In many cases, the current magnitude of alteration to large river ecosystems exceeds that expected under climate change, which encourages adaptation measures that simultaneously address continued human impacts of land use change and water regulation and diversion (Beechie et al., 2013a; Palmer et al., 2009; Rheinheimer & Viers, 2014). Scientists have begun expanding methods to assess how restoration actions and their ecological outcomes may affect and be affected by climate change. Beechie et al. (2013a) assessed a range of salmon habitat restoration actions for their potential effectiveness under climate change, finding that actions addressing processes, including floodplain reconnection and environmental flows, could better address long term resilience under change over more static instream habitat restoration measures. Similarly, Williams et al. (2015) call for projects focused on restoring processes at large scales over those focused on local conditions. A range of riparian restoration measures are evaluated by Perry et al. (2015) for their potential to accommodate climate change. Overall, studies suggest that larger-scale and more process-based approaches have greater likelihood to support functional ecosystems. And, to address climate uncertainty, a wide range of potential scenarios and restoration methods should be employed (Perry et al., 2015). These concerns amplify the need for continued development of cumulative impact assessments of restoration alternatives on physical as well as ecological processes and functions.

With riverine ecosystems increasingly impacted by human activities and climate change, the need to develop realistic definitions of restoration objectives within these constraints – including the persistence of non-native species, development within floodplains, dams regulating rivers, and nonstationary no-analog climates – is a reality faced by restoration ecology (Palmer et al., 2004). This understanding has motivated recent proposals for novel ecosystem management, reconciliation ecology, and designer or engineering approaches. The concept of reconciliation ecology as applied to freshwater ecosystem restoration seeks compromises within existing human-dominated landscapes to support biodiversity goals when more comprehensive ecosystem restoration is unattainable (Dudgeon et al., 2006; Moyle, 2013). Acreman et al. (2014a) support a designer approach to developing environmental flows under the argument that reinstating historical conditions is generally not possible. The authors' suggestions include the potential to enhance the benefits of available water through engineered channel or floodplain manipulation.

Some researchers caution that selecting desired functions or services for restoration may weaken efforts to reinstate naturally variable and dynamic processes (Higgs et al., 2014; Palmer et al., 2014). The recent paper by Hiers et al. (2016) argues that overly precise conservation and restoration targets – many of which may be unrealistic for highly altered systems – may restrict variability and erode ecosystem resilience. Along these lines, others have highlighted the concern that fine tuning or engineering conditions for only some locations or certain ecological functions may come at the expense of opportunities to support a range of processes necessary to sustain the larger ecosystem and to bolster resilience to change (Bond et al., 2014; Pittock et al., 2013). It is risky to consider engineered approaches without first assessing a broad range of options evaluated from a process-based perspective of supporting diverse ecosystem functions.

Hydroclimatic change, fundamentally altered food webs, and other human modifications suggest the need to define restoration actions by their potential to support resilience and productivity as opposed to native communities and reference conditions (Naiman et al., 2012; Poff, 2017; Rieman et al., 2015; Williams et al., 2015). Kopf et al. (2015) propose the use of Anthropocene baselines to develop more appropriate ecosystem targets when historical references are no longer relevant. Though targeting natural flows regimes or historical channel forms may no longer be appropriate in many cases, reinstating physical processes or particular functions reflective of the past while bolstering ecosystem resilience and productivity may still be possible. In the absence of these historical references, an important question, then, is how to evaluate which suite of actions may best address those objectives. The growing movement toward process-based, whole-system river restoration science enables these evaluations.

CONCLUSIONS

Rivers that have been highly impacted by flow regulation and geomorphic alteration face mounting and interacting stressors at many scales, which fundamentally alter the processes supporting diverse and productive ecosystems. Repairing the many process components and their interactions within highly altered rivers is a central challenge to riverine ecosystem management that requires integrated methods across the disciplinary fields of habitat restoration and environmental flows. Both disciplines recognize the role of land-water interactions and dynamics in driving ecological processes, and the scientific literature has converged on the underlying need to consider both hydrologic and geomorphic perturbations, and their interactive effects, for successful restoration of highly modified rivers. Complex landscapes as well as flow variability, working together, are required to maintain the variable flooding processes that drive riverine and floodplain ecosystems. Examples of integrated planning and implementation of physical habitat measures and environmental flows, some of which are discussed here, are increasing in number. We highlight examples of co-developed solutions and how these considerations fit within existing river restoration frameworks. Together, this represents a confluence of ideas that pushes both science and application to comprehensively assess the range of options available to restore functional and self-sustaining hydrogeomorphic processes driving resilient ecosystems within the many limitations posed by highly modified rivers. Moving forward, these efforts would benefit from continued improvements to coupled modeling of watershed- to reach-scale processes and new metrics (e.g., hydraulic, food-web based) by which to evaluate results. Further, addressing these more technical challenges must be met with continued fundamental research into ecological responses to changes in physical processes, as well as restoration strategies addressing other aspects of ecosystem functioning, such as food web dynamics. These developments are part of larger academic questions of establishing appropriate and meaningful restoration targets under a changing climate and in fundamentally altered ecosystems.

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CHAPTER 3

FLOOD REGIME TYPOLOGY FOR FLOODPLAIN ECOSYSTEM MANAGEMENT AS APPLIED TO THE UNREGULATED COSUMNES RIVER OF CALIFORNIA, UNITED STATES

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Flood regime typology for floodplain ecosystem management as applied to the unregulated Cosumnes River of California, United States

ABSTRACT

Floods, with their inherent spatiotemporal variability, drive floodplain physical and ecological processes. This research identifies a flood regime typology and approach for flood regime characterization, using unsupervised cluster analysis of flood events defined by ecologically meaningful metrics, including magnitude, timing, duration, and rate of change as applied to the unregulated lowland alluvial Cosumnes River of California, USA. Flood events, isolated from the 107-year daily flow record, account for approximately two-thirds of annual flow volume. Our analysis suggests six flood types best capture the range of flood event variability. Two types are distinguished primarily by high peak flows, another by later season timing and long duration, two by small magnitudes separated by timing, and the last by later peak flow within the flood event. The flood regime was also evaluated through inter- and intra-annual frequency of the identified flood types, their relationship to water year conditions, and their long-term trends. This revealed, for example, year to year variability in flood types, associations between wet years and high peak magnitude types and between dry years and the low magnitude, late season flood type, and increasing and decreasing contribution to total annual flow in the highest two peak magnitude classes, respectively. This research focuses needed attention on floodplains, flood hydrology, ecological implications, and the utility of extending flow regime classification typically used for environmental flow targets. The approach is broadly applicable and extensible to other systems, where findings can be used to understand physical processes, assess change, and improve management strategies.

INTRODUCTION

A river's flood regime, defined as the prevailing characteristics and distribution of flood pulses and variability within and across years, is controlled by geography, geology, climate, as well as human modifications, and drives physical and ecological processes within floodplain ecosystems, affecting the diversity, abundance and communities of species (Poff, 2002). Floods drive geomorphic processes, such as sediment deposition and erosion, as well as a host of biogeochemical processes, including nutrient cycling, primary and secondary productivity, and a wide range of biotic interactions. Flood pulses and their variable characteristics support a spatially and temporally heterogeneous and dynamic mosaic of habitats to which species are adapted (Junk et al., 1989; Poff et al., 1997; Tockner et al., 2000; Ward & Stanford, 1995). Flooding serves as a disturbance mechanism and generates complex hydrologic and geomorphic interactions that support ecological diversity and drive ecosystem structure and function (Resh et al., 1988; Richards et al., 2002). Different types of floods constituting a flood regime are associated with particular ecological functions (Opperman et al., 2010), extensively demonstrated in the literature, including research specific to the system of focus here, the Cosumnes River of California, USA. These include infrequent large magnitude floods causing avulsion and initiating riparian forest successional processes (Florsheim & Mount, 2002; Trush et al., 2000), snowmelt floods associated with predictable prolonged flooding and low recession rates supporting seed germination (Mahoney & Rood, 1998) and cuing reproduction of fish and amphibians (Yarnell et al., 2010), or high frequency but low magnitude spring flood pulses generating high levels of primary and secondary productivity and creating high quality fish spawning and rearing habitat (Ahearn et al., 2006; Jeffres et al., 2008; Sommer et al., 2001). The flow and flood regime components, including magnitude, timing, duration, rate of change, and frequency, that drive these and other ecological functions have been well-documented for their ecological relevance (Naiman et al., 2008;

Poff, 2002; Poff et al., 1997). Effectively characterizing the floods of a flood regime is therefore fundamental to understanding and managing processes driving floodplain functions.

Floodplains, with their flood-driven heterogeneous landscapes, support some of the most diverse and productive ecosystems globally (Naiman & Décamps, 1997; Tockner & Stanford, 2002). However, these systems are also some of the most degraded due to anthropogenic hydrologic alteration and land use change (Naiman et al., 2002; Nilsson & Dynesius, 1994). Within most large lowland alluvial rivers, fully natural flow regimes and restored landscapes are rarely achievable. Consequently, a central challenge is managing for greater function within such heavily modified riverine landscapes (Acreman et al., 2014; Palmer & Bernhardt, 2006; Sparks et al., 1998). Improving the reconciliation of human and ecosystem needs requires more precise water management (Harris et al., 2000), where water is used to provide ecological function in the most strategic manner (Poff & Schmidt, 2016; Yarnell et al., 2015). Doing so demands refined understanding of variability and the processes and functions driven by it, as well as temporally consistent features, such as snowmelt recession rates and other functional flow components (Yarnell et al., 2010).

A clear consensus on the need to improve water management for riverine ecosystems has led to numerous management strategies that typically involve flow regime characterization to set targets based on selected metrics (Petts, 2009). Over the last several decades, the natural flow regime concept of Poff et al. (1997) has encouraged the inclusion of variability in flow conditions in setting environmental flow standards (Poff et al., 2010). However, while flooding is recognized as an essential component of the natural flow regime, assessing variability in flood characteristics is often not a focus of management despite ecological outcomes. Furthermore, environmental flows science rarely considers how the surrounding landscape – often highly modified environments – can influence the ecological performance of a managed flow regime (Arthington et al., 2006; Yarnell et al., 2015). This is exemplified in Jacobson and Faust (2014), who showed that although flood frequency and duration followed expected patterns on the Missouri River, floods that should have inundated floodplains did not due to channelization and incision. As land and water management decisions are often interdependent, analysis of altered flood regimes should be examined jointly with modification of the physical landscape (Kondolf et al., 2006).

Restoring riverine ecosystem functions depends on understanding the flows that produce natural floodplain inundation patterns (Benke, 2001). While common flood frequency analysis that determines return period flows from the annual peak flow time series may be adequate in many engineering contexts, they are insufficient for interpreting ecosystem process and function. Assessment of inter- and intra-annual variability adds critical insight because while several floods may occur within a year, floods with ecologically relevant characteristics may occur far less frequently. Thus, more detailed and systematic characterization of flood regimes is needed to better target ecological needs within floodplains. Despite studies that quantify floodplain inundation dynamics (Benke, 2001), relate specific ecological functions to flood characteristics (Agostinho et al., 2004), identify thresholds to guide management (Richter & Richter, 2000), and assess climate change impacts (Hall et al., 2014), there have yet to be systematic classifications of flood characteristics into a coherent flood regime typology to inform ecological management objectives.

Classification allows simplification of flood complexity and variability for describing and interpreting the prevailing flood regime of a river and its floodplain. Classification is applied in many fields including hydrology, where it is used to generate fundamental knowledge of river form and process, assess variability at different spatial and temporal scales, provide clear and easily interpretable class definitions, and develop management guidelines (Olden et al., 2012; Tadaki et al., 2014). It can be applied at multiple scales, from the flow regime scale to the flow pulse or event scale (Olden et al., 2012). Most hydrologic classification studies group streams based on their flow regimes for regionalization and for predicting characteristics of ungaged basins (e.g., Haines et al., 1988; Toth, 2013). Flow regime classification is also used to establish connections between flow and ecology (e.g., McManamay et al., 2015), evaluate climate change impacts on flow regime characteristics of ecological relevance (e.g., Dhungel et al., 2016), and inform environmental flow standards (e.g., Kennard et al., 2010). Methods typically involve some form of unsupervised classification, or cluster analysis, which provides more objective and reproducible definitions of classes than classification using predetermined classes (e.g., Hannah et al., 2000; McManamay et al., 2014; Poff, 1996; Sanborn & Bledsoe, 2006). In one of the earliest of such studies, Burn (1989) applied cluster analysis to group stations based on watershed characteristics for purposes of regional flood frequency analysis. Both partitional (e.g., k-means) and hierarchical clustering methods are used, though hierarchical clustering with either divisive or agglomerative approaches is most common for streamflow classification (Olden et al., 2012).

Classification applied at the flood event level is far less common. In one example, Aubert (2013) classified flood events to compare clustering methods, as applied to examining relationships to water quality. Through a supervised classification approach, a recent study used fuzzy decision trees to classify floods into types to identify dominant flood processes across watersheds (Sikorska et al., 2015). Merz and Blöschl (2003) also explored flood mechanisms using pre-defined classes, or process types, with annual peak floods of Austrian catchments that were assigned using process indicators, such as timing, storm duration, and rainfall depth. For the same system studied here, Booth et al. (2006) defined flood types based on a priori classification, using pre-defined thresholds of magnitude and duration to form combinations of flood types with differential frequency. To our knowledge, the methods and objectives common to flow regime classification within the field of environmental flows has not been extended to flood type classification for floodplain management applications.

With an emphasis on ecological relevance and the use of existing data classification techniques, the objectives of this paper are to 1) establish a flood regime typology and delineation approach that captures a river's flood regime relevant for floodplain ecosystems, 2) demonstrate its effectiveness in identifying dominant flood types through application to the Cosumnes River of California, USA, and 3) relate flood types to driving mechanisms and ecological and management implications. Our flood regime typology offers a novel and systematic approach for simplifying complex information to describe a floodplain's flood regime, provides insights into climate and watershed processes, and generates needed information for water management and restoration of floodplain ecosystems.

METHODS

Overview

As a means for flood regime characterization, we establish flood types via k-means cluster analysis using individual flood events identified from the historical streamflow record and described by ecologically relevant metrics representing flood event magnitude, timing, duration, rate of change, and hydrograph shape. After clusters are assessed for stability and validated, the most distinguishing characteristics of flood types are described, as is their frequency, relationship to water year conditions, and trend. Finally, we link flood types to watershed processes and floodplain ecological functions and discuss management applications.

Study site

The Cosumnes River watershed, the case study for this analysis, is located along the west slope of the central Sierra Nevada mountain range in California, USA (Figure 3-1). It drains approximately 2,460 km^2 with elevation ranging from 2,300 m at its headwaters to near sea level at its confluence with the Mokelumne River. The Cosumnes River is the only large river of the Sierra Nevada without a major dam, and its resulting unregulated hydrograph, as well as a 107-year continuous daily streamflow record, is greatly beneficial to this study in the capacity to examine largely natural inter- and intra-annual variability in flood characteristics. Though the majority of the watershed consists of forested headwater regions, the lower watershed has been altered substantially over the last century and a half through leveeing, channelization, groundwater abstraction, and other land uses, which has profoundly altered how the still largely unregulated flood regime is expressed spatially within the floodplain. Over the last three decades, process-based restoration involving levee breaching has reconnected some of the floodplain to the river in the lower reaches, including the site used in this study. Associated scientific research and monitoring has linked this increased hydrologic connectivity to sediment deposition, increases in topographic complexity, hydrochorus dispersal of native seeds within the floodplain, riparian forest establishment and succession, primary and secondary productivity, and greater provision of spawning and rearing habitat for native fish, including juvenile Chinook salmon and the endemic minnow, Sacramento splittail (Ahearn et al., 2006; Andrews, 1999; Florsheim & Mount, 2002; Jeffres et al., 2008; Moyle et al., 2007; Swenson et al., 2003; Trowbridge, 2002).

The climate consists of cold wet winters and warm dry summers with high interannual precipitation variability due to its predominately Mediterranean-montane climate. Recent research has highlighted that river systems in California such as the Cosumnes River depend upon just a few storms to produce the majority of annual runoff, accounting for extreme interannual variability (Dettinger et

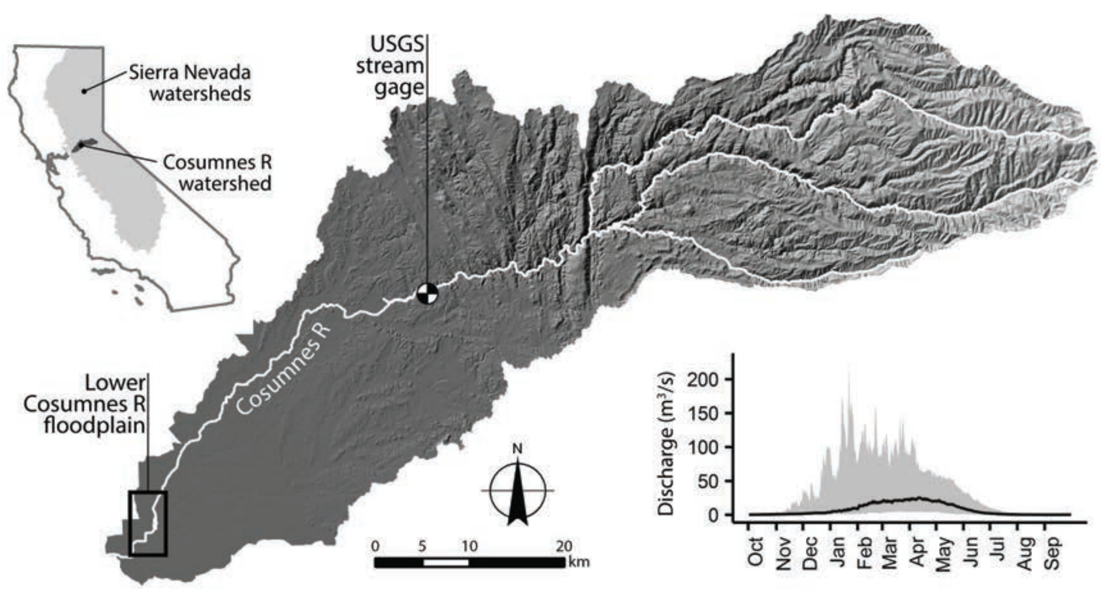

Figure 3-1. Map depicting the Cosumnes River watershed, located along the west slope of California's Sierra Nevada. The location of the USGS stream gage used in the study and the reference floodplain approximately 45 km downstream are illustrated. The inset graph shows median daily flow with shading representing the 5th-95th percentile at the gage.

al., 2011). Mean annual precipitation (1971-2000) ranges from 1,460 mm in the upper elevations to 430 mm in the lower elevations, with a spatially weighted average of 855 mm (PRISM Climate Group, 2006). Precipitation occurs primarily between the months of November and March with the majority of runoff occurring between December and May. The resulting hydrograph is rain-dominated, as much of the watershed area lies below the snow line (~90% below 1,500 m; see inset of Figure 3-1), though a spring snowmelt signature is present. Following precipitation variability, streamflow is highly variable. The flood of record in 1997 resulted in a peak daily flow of 2,630 m^3/s in contrast to the mean annual daily flow approximating 14 m^3/s. In dry years, flow ceases by the end of the summer in the lower river reaches, exacerbated by severe declines in regional groundwater levels (Fleckenstein et al., 2004).

Hydrologic data

The primary dataset used in this analysis is the daily streamflow record for the time period 1908-2014 (MHB, #11335000; U.S. Geological Survey, 2015). The river at Michigan Bar, CA drains 57% of the watershed, from which the majority of streamflow originates. Though tributary inflows and other gains and losses affect flows at the floodplain site considered here (located 45 km downstream), these are understood to be minor for purposes of examining flood characteristics (Andrews, 1999). Analyses were performed using the water year, beginning October 1. Daily precipitation data were obtained from the National Climatic Data Center COOP weather station (Fiddletown Ranch #043038) located within the upper watershed and date back to 1948 (Western Regional Climate Center, 2015). Atmospheric river and "pineapple-express" events in California have been studied and summarized by Dettinger et al. (2011) for 1948 to 2008.

Flood event identification and metrics

We identified individual flood events from the daily flow record using a previously determined floodplain inundation threshold of 23 m^3/s at MHB, flows at which lowest-lying floodplain areas connect to the river (Figure 3-2; Florsheim et al., 2006, personal observation). This flow approximates a 95% exceedance probability for the annual peak flow series (U.S. Interagency Advisory Committee on Water Data, 1982). Though floodplains are typically defined using the 1.5-year return period flow to represent bankfull (Leopold & Wolman, 1964), this is not consistent for all river reaches. Using the 1.5-year return period flow (107 m^3/s) would exclude many ecologically relevant lower-flow flood events from the analysis. The methodology applied here captures frequent, annual floods on the floodplain as well as peak annual storms. Using a flow threshold to identify flood events contrasts with common flood analyses based on the annual peak flow time series.

Flood events were isolated and numerically characterized in *R* (R Core Team, 2013). We established eight metrics derived from flow and flood regime components of magnitude, duration, timing, and rate of change, defined and described in Poff et al. (1997) and Poff (2002) as driving various ecological processes in riverine systems (Table 3-1). These factors affect both abiotic and biotic processes. The magnitude of floods affect sediment erosion and deposition, maintaining habitat mosaics and heterogeneity. For example, flood disturbance and variability along the Cosumnes River creates complex floodplain topography and initiates riparian forest successional processes (Florsheim & Mount, 2002). Floods occurring at different times of the season serve different ecological functions, whether it be winter floods that cue fish migration, or spring floods that provide rearing habitat for juvenile fish and promote primary and secondary productivity (Moyle et al., 2007). Research on the Cosumnes River has also linked flood duration (residence time) and connectivity dynamics to productivity (Ahearn et al., 2006; Grosholz

& Gallo, 2006). Rate of change, another flow and flood regime component, affects the temporal variability in habitat conditions and availability as well as seed germination and survivorship (Yarnell et al., 2010). Frequency, also identified by Poff et al. (1997), was irrelevant as a metric for summarizing the sequence of daily flows that make up single flood events, but was assessed in other stages of the analysis after floods were isolated and described. Three metrics related to the shape of the hydrograph (e.g., position of peak within the event) were included because the timing of peak flows within an event can have hydraulic, geomorphic, and ecological implications (Tockner et al., 2000). The isolated flood events were summarized on annual and monthly bases to quantitatively characterize the flood regime prior to flood type classification.

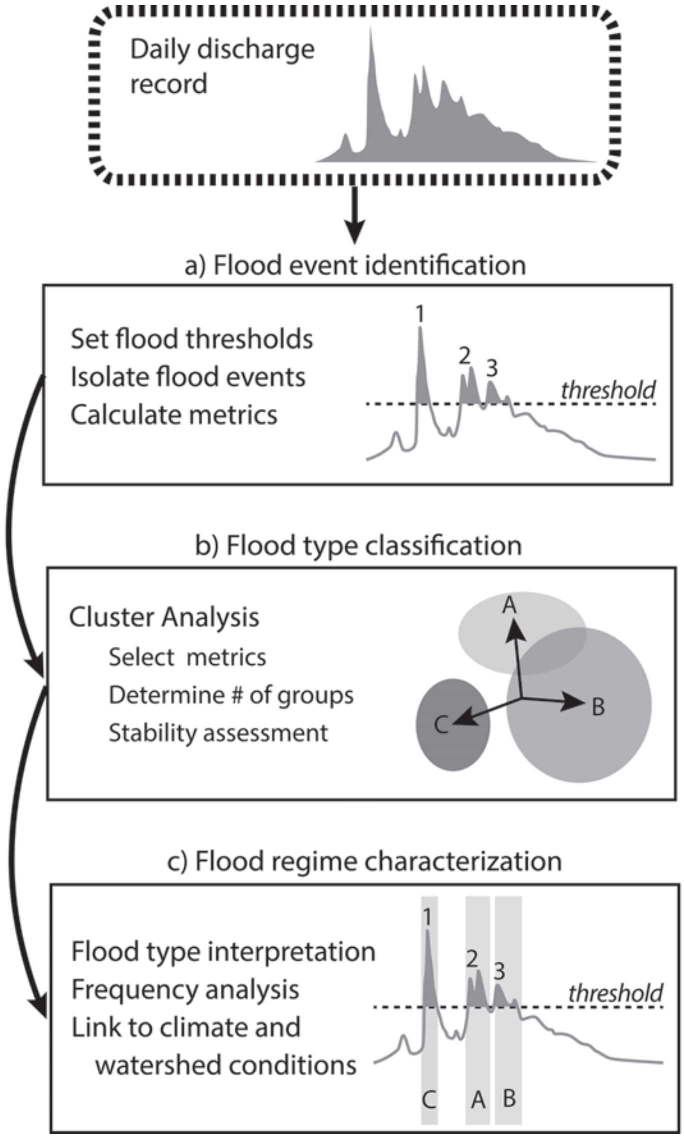

Figure 3-2. Flood typology and characterization approach. Flood events from the daily flow record input are separated using a floodplain activation threshold and characterized using selected metrics (a). Subsequently, classification is performed using cluster analysis (b). Identified flood types are then interpreted and assessed for frequency, relationship to climate factors, and trends to describe the flood regime (c).

Statistical methods for flood typing

The flood type classification methods described here addresses core classification objectives identified by Jain (2010); these include understanding data structure and developing insights into the range of conditions, as well as simplification and organization of complex multivariate data. The goal of our flood regime typology is to simplify highly variable flood events into basic types for describing essential characteristics of floods that inundate floodplains and provide information useful for managing riverine ecosystems.

We established flood type classes from the characterized flood events using k-means cluster analysis from the *R* package *fpc* (Hennig, 2014; R Core Team, 2013). K-means clustering is a common clustering method for a wide range of applications, including hydrologic classification and regionalization (e.g., Burn, 1989; Chinnayakanahalli et al., 2011; Dettinger & Diaz, 2000; Parajka et al., 2010; Poff, 1996; Sanborn & Bledsoe, 2006). As a partitional non-hierarchical clustering algorithm, it iteratively adjusts cluster centers and assigns individual points to classes based on the nearest center (Euclidian distance),

Table 3-1. Flood regime components with the metrics representing these components used for characterizing individual flood events of the Cosumnes River. Metrics annotated with (*) were used in the final cluster analysis for establishing flood types. Examples of related ecological functions and references, as discussed in Poff et al. (1997) and Poff (2002), are listed for each of the main flood regime components.

Flood regime component	Metric	Description	Ecological functions affected	References
Magnitude	Peak discharge*	Peak daily discharge (m^3/s) within flood event	Channel avulsion, sediment erosion and deposition, reset successional processes, maintenance of habitat mosaic and heterogeneity, habitat availability, reduce competition	Florsheim and Mount (2002); Opperman et al. (2010); Resh et al. (1988); Ward and Stanford (1995)
	Mean discharge	Mean daily discharge (m^3/s) across flood event		
	Volume*	Total volume of flood event (km^3)		
Timing	Start day	Water year day of the flood event beginning	Species migration cue, spawning and rearing habitat availability, primary and secondary productivity	Bailly et al. (2008); Jeffres et al. (2008); Lytle and Poff (2004); Moyle et al. (2007); Robertson et al. (2001)
	Centroid day*	Water year day of centroid volume of flood event		
	End day	Water year day of the flood event ending		
	Cumulative flow	Total water year flow volume to date of flood beginning		
Duration	Days*	Total number of flood days	primary and secondary productivity, spawning and rearing habitat availability	Bailly et al. (2008); Grosholz and Gallo (2006); Sommer et al. (1997); Sommer et al. (2004)
Rate of change	Rising rate	Maximum flow (m^3/s) difference between days on the rising limb(s) of flood event	Seed germination, habitat availability	Mahoney and Rood (1998); Stella et al. (2006); Yarnell et al. (2010)
	Recession rate*	Maximum flow (m^3/s) difference between days on the falling limb(s) of flood event		
Shape	Peak location*	Fraction of flood event duration before the day of peak flow	Nutrient cycling, primary and secondary productivity, sediment erosion and deposition patterns, export of organic and inorganic material	Florsheim and Mount (2002); Tockner et al. (2000)
	Centroid volume location	Fraction of flood event duration before the day of flood event centroid volume		
	Number of peaks	Number of hydrograph peaks within flood event		

and centers are adjusted to minimize the sum of distances between points and the associated centroid within a cluster (Jain, 2010). Though hydrologic applications often use divisive hierarchical clustering methods, such as Ward's linkage, we chose the partitional k-means approach following Hartigan and Wong (1979) because it is known to handle large datasets well, individual points are allowed to move from one cluster to another over the series of iterations, hierarchy was not relevant to interpretation, and more stable clusters were found in comparison to complementary hierarchical methods (Olden et al., 2012). All data were normalized (subtracting the mean and dividing by the standard deviation) prior to analysis.

To perform k-means clustering, we first specified the distinguishing variables and number of classes. Because of their ecological relevance (Poff, 2002), we included at least one metric from each of the flood regime components as variables in the analysis (see Table 3-1). We conducted principal components analysis (PCA) to examine redundancy and the relative strength of different metrics in explaining the variance in the data. Based on this analysis, metrics with higher explanatory power were prioritized for inclusion. For final metric selection, we used clustering strength and stability, as discussed later. In addition to metric selection, the location of cluster centers and choice of the number of classes can impact clustering results. To address potential subjectivity, we used randomized cluster seed locations and several common statistical criteria, including within cluster sum of squared errors and silhouette width (Olden et al., 2012; Rousseeuw, 1987), to determine the optimal number of classes.

Stability of resulting flood types was assessed via the *clusterboot* function in the *fpc* package for *R* (Hennig, 2014). We used this function to apply 1000 sampling runs using a nonparametric bootstrap scheme, where new flood datasets were sampled with replacement from the original set of floods (Hennig, 2007). Such stability assessments have been used in previous hydrologic classification applications (Mackay et al., 2014; McManamay et al., 2014). The more stable clusters are those that maintain cluster membership despite minor changes to the original dataset in each resample. To measure cluster stability, the Jaccard stability index (i.e., the proportion of the intersection and union of two sets) is calculated between each resampled cluster and the most similar cluster in the original set, which are then averaged to produce a stability measure for each cluster (Hennig, 2007). Clusters with indices above 0.75 are thought to form valid stable clusters, while those below 0.5 are indicative of dissolved clusters (Hennig, 2014). The Jaccard similarity index was also used to determine which set of metrics and number (i.e., k-value) of clusters produced the most stable clusters. Instead of comparing the highest average score across all clusters from each combination of metrics and number of clusters, we selected those with the highest minimum cluster score (i.e., comparing the lowest scoring cluster of each set). Since stability alone cannot guarantee valid clusters, we complemented this with visual validation of the cluster separation to assess how well classes were distinguished. In this analysis, highly isolated flood types are not expected since floods result from many interacting environmental variables and processes, causing many floods to lie between the predominant flood types.

Flood regime characterization

Post-classification, the identified flood types were assessed and compared, and then examined with regard to frequency, relationship to water year conditions (e.g., wet versus dry), and trend. Where applicable, analysis was performed for both the number of events and the number of days for a given flood type. Frequency, a natural flow regime component, was calculated empirically for the flood types both inter- and intra-annually. Flood types were also examined for their association with other types within years. We compared flood types in relation to the water year conditions (defined by annual flow quantiles), which revealed clear distinctions between flood types, but also provided an independent measure of the strength of the classification, as floods in wet years are expected to have different characteristics from those in dry years. We explored whether a change in the frequency or dominance of different flood types had occurred over the period of record with trend analysis on the number of events, number of days, and volume of each flood type using 5 and 10 water year block averages of each variable. Block averages helped to provide data independence and address the fact that some years had no events of particular types. We estimated and tested trends by fitting a generalized least squares (GLS) model with the method of maximum likelihood using the *nlme* package in *R* (Chatfield, 1989; Dahlke et al., 2012; Pinheiro et al., 2013; R Core Team, 2013). To address autocorrelation, autoregressive moving average (ARMA) correlation structures were fit to the residuals (Fox, 2002). Finally, we linked flood types to relevant climatic and watershed processes and to ecological functions, and discussed floodplain management implications.

RESULTS

Flood event summary

Using a flow threshold of 23 m^3/s, we identified a total of 532 individual floods from the 107-year record spanning water years 1908-2014. Flood event summary statistics revealed that the number of flood events ranged from 0 to 13 per year, with a median of 5 events and 68 days of flooding (Table 3-2). Event volumes summed within a water year accounted for a median of over two-thirds (66.9%) of the annual flow volume. More flood events occurred in January through March, while March and April had the highest number of flood days. A median 46.3 m^3/s discharge was recorded for the peak daily flow. Total flood volumes were most variable, with a median of 0.010 km^3. The median flood duration was 3 days. For flood timing, mean date of flood center of mass (Stewart et al., 2004) was February 18, ranging from October 14 to June 30. Recession rate, quantified as the maximum decline over a day within a flood event, had a median of -18 m^3/s. Most flood peaks occurred toward the beginning of the flood event. All flood event metrics were highly variable, requiring subsequent classification to distinguish characteristic flood types.

Classification of flood types

METRIC SELECTION AND CLUSTER STABILITY

Examination of statistical redundancy aided the selection of metrics within each flood regime component, since we decided to use at least one metric related to each component in the cluster analysis for purposes of ecological relevance. As expected, most metrics within each component were highly correlated. Magnitude, duration, and rate of change metrics were also highly correlated. PCA revealed that over 95% of the variance in the data was explained by the first five principal components. The peak flow, centroid date, peak location, and flood event volume had the highest absolute loadings associated with these components. Duration and recession rate – each included for their previously discussed ecological relevance – were associated with higher loadings in the first principal component (along with peak flow).

Final metric selection and cluster number was based on the cluster stability results (Jaccard similarity index) from multiple bootstrapped (B = 1,000) cluster analysis runs using permutations of cluster numbers (within cluster sum of squared error (SSE) supported using six to eight clusters) and the sets of metrics meeting criteria. In comparing the lowest scoring cluster of each combination, the highest consisted of six classes and the six metrics summarized above (peak flow, flood event volume, duration (log transformed), centroid timing, recession rate, and peak location). The average cluster stability for this combination was 0.80, and all clusters had values above or close to the suggested threshold of 0.75

for stable clusters (Hennig, 2014). The flood types are primarily separated by magnitude (peak flow and flood volume) as well as duration and recession rate along the first component axis, while timing dominates the second axis (Figure 3-3). A single possible outlier is the flood of record (1997), but its removal did not substantially affect cluster results, so it was retained in the analysis.

FLOOD TYPE DESCRIPTION

The six classified flood types are easily discerned from the metrics (Figure 3-4). Referred to here as *Very Large* events, the floods with the highest peak flows (median = 598 m^3/s) were clustered together and also associated with high volumes (median = 0.69 km^3), long durations (median = 90 days), and steep recessions (median = -319 m^3/s per day). Only 17 of the total 532 events (3.2%) were classified as this type, including the largest flood on record. The second highest peak flow magnitude class is distinguished both for its high peak flows and volumes (median = 300 m^3/s, 0.26 km^3) and long duration (median = 35 days), referred to as *Large and Long* events. This class included 46 flood events (8.6%). The long duration of these two classes is attributed to the high peak flows but also the multiple storms that occur over the period of the flood, which maintains flow above the floodplain inundation threshold. The flood type with the latest seasonal timing centroid, the *Long and Late* type, also has the third longest duration.

Table 3-2. Summary of 532 flood events identified from the 107 years of the historical daily flow record. Annual and monthly summary statistics, including median, mean, standard deviation (SD), and coefficient of variation (CV), are provided along with the metrics used in the cluster analysis.

Type	Parameter	Median	Mean	SD	CV
Annual	#events/yr	5	5.0	2.6	52%
	#days/yr	65	63.9	51.5	81%
	total % of annual volume	66.9	54.9	30.9	56%
Monthly	#events October	0	0.0	0.2	628%
	#events November	0	0.2	0.6	289%
	#events December	0	0.7	1.1	148%
	#events January	1	1.1	1.2	107%
	#events February	1	1.0	1.0	93%
	#events March	1	0.9	1.0	104%
	#events April	0	0.6	0.8	145%
	#events May	0	0.3	0.5	195%
	#events June	0	0.1	0.3	434%
	#days October	0	0.0	0.3	615%
	#days November	0	0.7	2.4	360%
	#days December	0	3.2	5.8	184%
	#days January	3	7.9	9.6	122%
	#days February	8	10.9	10.1	92%
	#days March	10	14.5	12.2	84%
	#days April	17	15.2	13.1	86%
	#days May	4	9.7	11.8	122%
	#days June	0	1.8	5.1	279%
Metrics	Peak flow (m^3/s)	46.3	101.5	149.8	148%
	Volume (km^3)	0.010	0.064	0.1	223%
	Duration (days)	3	12.8	23.9	186%
	Centroid day (water year day)	141	141.1	50.0	35%
	Recession rate (m^3/s)	18.0	47.4	90.0	190%
	Peak location	0.02	0.2	0.2	130%

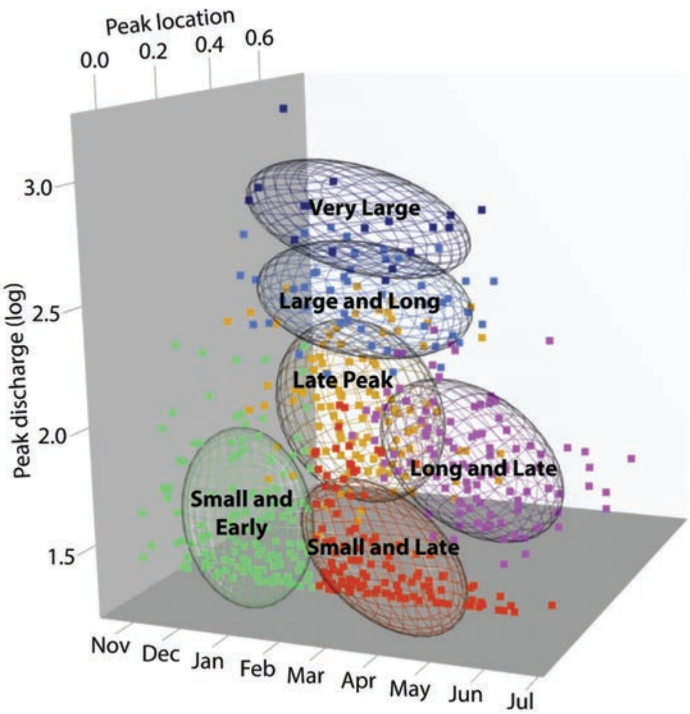

Figure 3-3. Clustered flood events along three metrics used in analysis: magnitude of peak discharge (log transformed), centroid date, and location of hydrograph peak (fraction of flood event duration before the day of peak flow). Ellipses cover the 50% confidence region for each type. The color scheme is consistent across figures.

One flood type is predominantly characterized by its late peak within the flood hydrograph; referred to as the *Late Peak* flood type. Its other metrics are mid-range compared to other flood types. Two types are distinguished by their very low magnitudes and short durations (often only a day or two long). They are separated by timing, with the *Small and Late* events occurring late in the season and the *Small and Early* occurring earliest of all the types. These last four types are larger classes, with membership ranging from 101-139 events (19-26%). Substantial differences in the range of metric values within each of the flood types can be found. For example, the *Very Large* and *Large and Long* classes have the greatest magnitude range. Both *Small and Late* and *Small and Early* types are fairly tight across the metrics, except for centroid day, which is the primary metric distinguishing the two classes.

Flood type frequency

Understanding the inter- and intra-annual frequency of different flood types allows for improved interpretation of ecological implications of flood events. As shown in Table 3-3, interannual frequency of the *Very Large* type is just 15%, and in only one year did two of these events occur. The *Late Peak* type is the most frequent, occurring in 64% of the years. However, *Long and Late*, *Small and Late* and *Small and Early* types have a greater percentage of years with two or more events, and *Small and Late* and *Small and Early* types each have over 5% of years with four or more events. The *Small and Early* type has the greatest mean intra-annual frequency of 2.2 events per year (assessed only for those years containing the flood type). Although *Very Large* events are usually over 90 days long, *Large and Long* and *Long and Late* events are also long (means of 47 and 19 days, respectively), but occur much more frequently. Understanding this difference is useful for evaluating the relative importance of the flood types in the provision of flooded habitat, for example.

Given that multiple events of different flood types usually occur within a given flood season, knowing flood type associations is also valuable for understanding flood type frequency (Table 3-4). We found that years with *Very Large* events were often associated with *Small and Early* or *Late Peak* events, while years with *Long and Late* or *Small and Late* events rarely had a *Very Large* event. Years with *Large and Long* events were associated with a wider range of flood types and occurred with *Small and Early* events over 80% of the time. All other percentages of association within years were below 70%. This analysis also showed that *Small and Late*, *Small and Early*, and *Late Peak* events all occurred with each

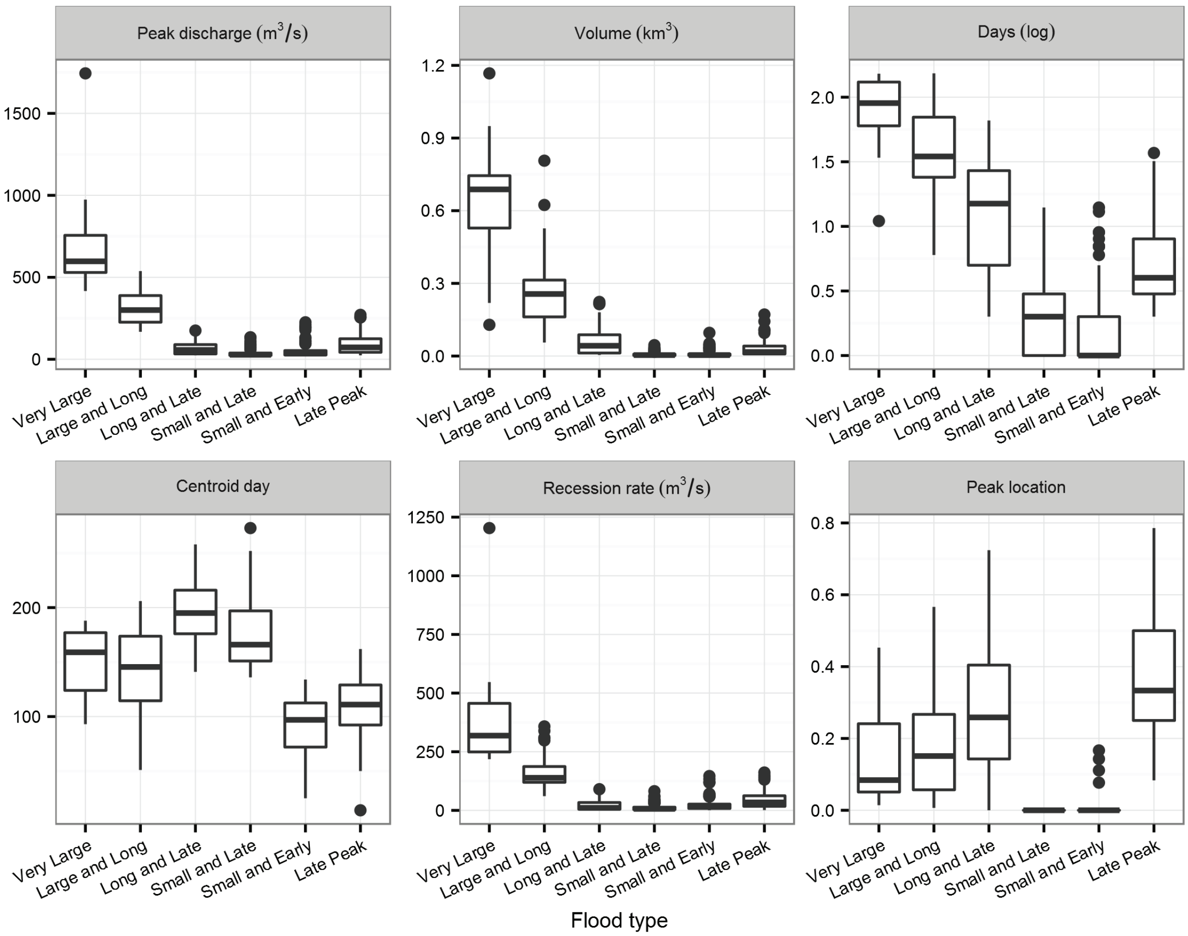

Figure 3-4. Box plots for each metric used in the analysis by flood type. Unscaled metrics are shown for purposes of interpretation. Median is shown with the first and third quartiles. Whiskers extend to the highest and lowest values within 150% of the inter-quartile range. For metric descriptions, refer to Table 3-1.

other over 50% of the time. Lastly, Figure 3-5 provides a visual depiction of the occurrence, duration, frequency, as well as high degree of inter- and intra-annual variability, of flood types across the streamflow record. Most years begin with short events of usually *Small and Early* or *Late Peak* flood types. Years with *Very Large* events clearly lack *Large and Long* or *Long and Late* events. In years where few long duration events occur, *Small and Late* events are more prevalent.

Relationship to water year

Although climate-related metrics were not used for classification purposes, we found distinct relationships between flood types or sets of flood types and the water year conditions. We defined five water year types using flow quantiles: very wet (0.8-1 quantile), wet (0.6-0.8 quantile), normal (0.4-0.6 quantile), dry (0.2-0.4 quantile), and critically dry (0-0.2 quantile). The clearest association between flood types and water year type was between *Very Large* events and very wet years, where all except for one *Very Large* event occur in this water year type (Figure 3-6). Similarly, *Large and Long* floods are also associated with wetter year types. No *Large and Long* events occur in dry and critically dry water years. Other flood

Table 3-3. Frequencies of events for each flood type showing a) interannual and, b) intra-annual empirical frequency. Interannual includes frequencies for at least 1, 2, 3 and 4 events within a year for each flood type. Both number of events as well as number of days is assessed for intra-annual frequencies. Intra-annual frequencies were calculated only for those years containing flood types.

	a) Interannual				b) Intra-annual			
	≥1 event	≥2 events	≥3 events	≥4 events	# events (mean)	# events (sd)	# days (mean)	# days (sd)
Very Large	15%	1%	0%	0%	1.1	0.3	91.2	41.7
Large and Long	36%	7%	1%	0%	1.2	0.5	47.2	33.4
Long and Late	53%	31%	8%	1%	1.8	0.8	18.6	16.0
Small and Late	60%	35%	14%	6%	1.9	1.0	2.2	2.0
Small and Early	60%	40%	18%	7%	2.2	1.2	2.2	2.1
Late Peak	64%	24%	8%	2%	1.5	0.8	6.2	5.8

Table 3-4. Associations of flood types within flood seasons. The diagonal shows the percent of years with that flood type. Co-occurrence values are calculated as the percent of years including the flood types in the rows with those in the columns (e.g., 25% of the years that have *Very Large* also have *Large and Long* and 11% of the years that have *Large and Long* also have *Very Large*).

	Very Large	Large and Long	Long and Late	Small and Late	Small and Early	Late Peak
Very Large	15	25	25	19	69	63
Large and Long	11	36	63	63	82	68
Long and Late	7	42	53	68	56	67
Small and Late	5	38	61	60	53	61
Small and Early	17	48	50	53	60	69
Late Peak	14	38	55	57	64	65

types are more evenly spread across the different water year types, although the *Long and Late* and *Small and Late* types are least associated with wetter years.

In examining the flood type composition of different water year types we found that only 2.1 events and a total of 4.6 days of flooding occurred on average in critically dry years, predominantly composed of *Small and Late* events. Comparing events and days illustrates the relative substantial contribution of *Late Peak* events to flood days. For the dry water year type, the *Small and Late* type had the greatest percentage of events within a year, while the *Long and Late* type had the greatest percent of days. In normal water years, the number of events was fairly well distributed across the flood types (except for the lack of *Very Large* events), with *Large and Long* and *Long and Late* types contributing disproportionately to the percent of flood days. The wet water year class had the highest number of events on average (>7) and was similar to the normal water year save for the larger contribution of *Large and Long* and *Late Peak* events. For very wet years, the average number of events dropped below five, but they had the highest average number of flood days (130). The flood type distribution shows that the *Small and*

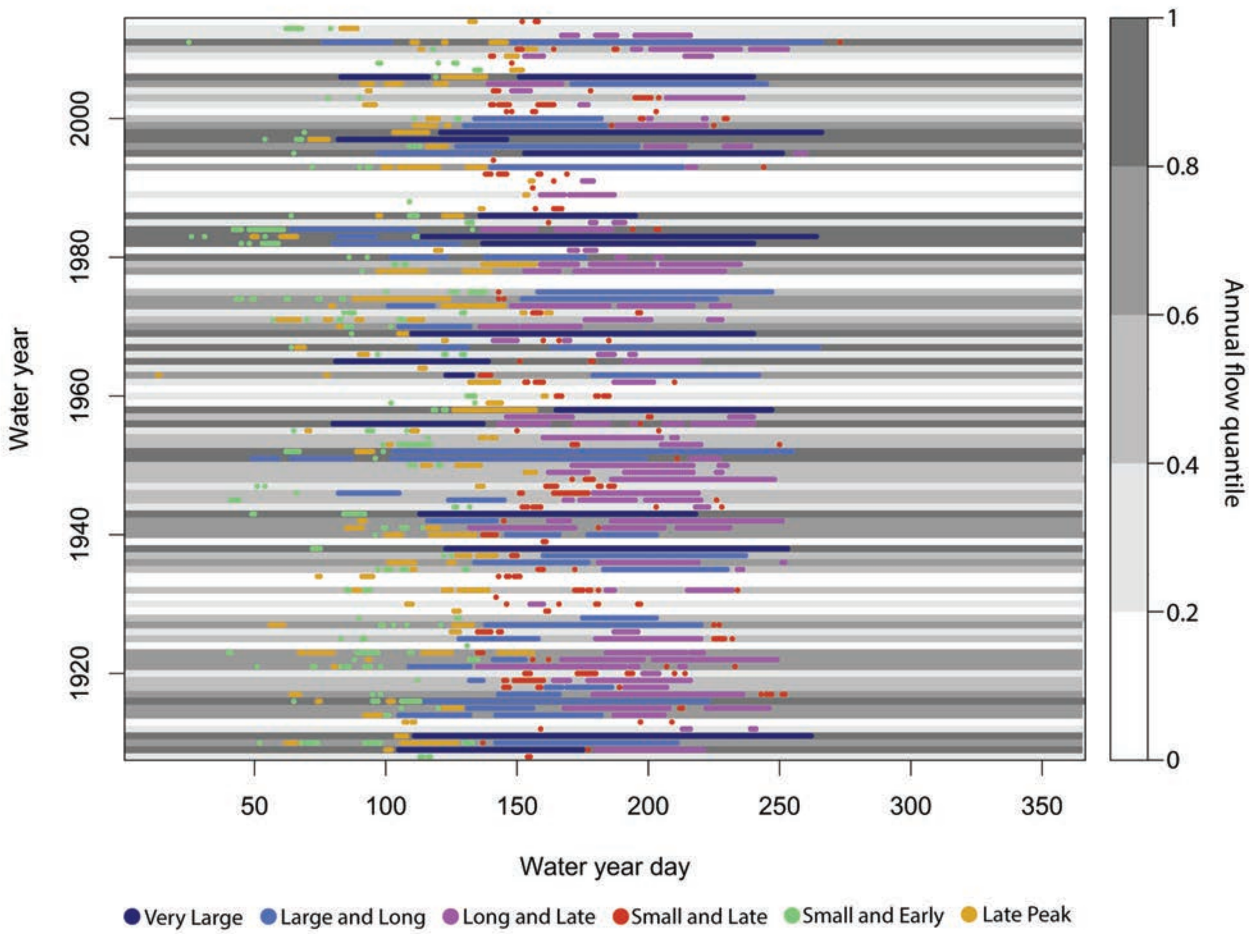

Figure 3-5. Each day of the 532 individual flood events is shown over the period of record, colored by their flood type classification. To relate the events and their frequencies to climate conditions, the water year types based on annual flow quantiles are shown in the right part of the plot (wetter years are the darker shades).

Early type tended to have the greatest number of events, while by far the greatest proportion of flooding days was attributable to *Very Large* and *Large and Long* events.

Trend analysis

Two flood types show evidence of statistical trends, analysis for which was conducted using 5-yr block averages of each flood type's contribution to the annual flow volume starting in 1910 through 2014 (Figure 3-7). All years during this period were used, including those periods where no floods occurred. Based on AIC values and plots of the autocorrelation function and partial autocorrelation function, appropriate generalized least squares (GLS) models were selected and fit to the data. A significant ($p < 0.01$) increasing trend in the percent of annual volume of *Very Large* events was found. Based on the regression analysis, this trend amounted to an increase of 14% in the percent volume that *Very Large* events contribute to the annual total flow volume. The trend for the *Large and Long* type was also significant ($p < 0.01$), suggesting decreases of 10% in the percent volume *Large and Long* events contribute to annual volume. Thus, the flood type associated with the very largest magnitude floods show an increasing dominance within the annual total flow, while the second largest magnitude type is declining

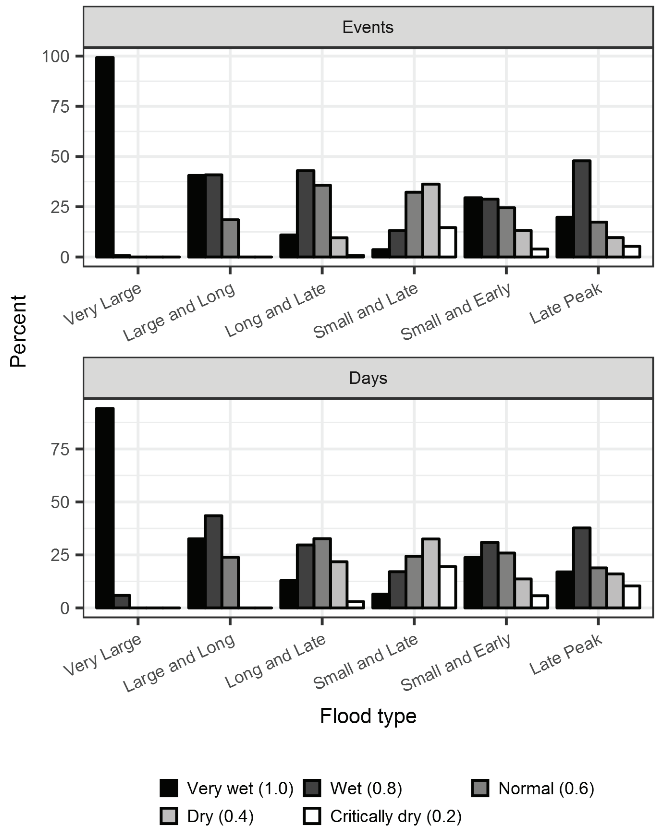

Figure 3-6. The percent of flood types associated with each water year type for the percent of events (a) and days (b). Water year types are defined by annual flow quantile (in parentheses) from the daily flow record at the Cosumnes River gage. Darker shading is associated with wetter year types. Each flood type grouping sums to 100%.

in dominance. Though not significant, the other four flood types also showed declining trends.

DISCUSSION

Flow regime classification has been used extensively over the last several decades to better understand and manage riverine ecosystems (Kennard et al., 2010; Olden et al., 2012; Poff & Ward, 1989). The flood event classification proposed here presents an extension of these methods by providing more systematic and higher resolution characterization of the range of flood types and their inter- and intra-annual variability within flood regimes that generate the dynamic, yet predictable habitat conditions to which species are adapted. The typology elucidates driving mechanisms and related ecological functions of floods, thereby improving our ability to understand and manage riverine ecosystems.

Flood regime typology

The primary metrics used to characterize flood events are derived from flow and flood regime components of magnitude, duration, timing, and rate of change, which are well established in the literature for their ecological significance (e.g., Lytle & Poff, 2004; Poff, 2002; Poff et al., 1997; Rood et al., 2005; Sparks et al., 1998) and are commonly applied in classification studies linking hydrology and ecology (e.g., Belmar et al., 2011; Kennard et al., 2010; Mackay et al., 2014). The metrics identified for the cluster analysis were associated with the principal components explaining the majority of the variance across the identified flood events and resulted in the most stable clusters. Metrics related to magnitude and timing appear to be the best for classifying floods given their correlation with the first two principal components, and visual separation between the identified flood types. These findings are similar to the principal flow regime elements of magnitude and temporal variability identified by Belmar et al. (2011).

Our flood regime typology approach established six flood types from historical daily streamflow data, reducing the highly variable Cosumnes River flood regime into manageable elements. The *Very Large* flood type with the highest peak flows occurred in only 15% of the years on record, but this type dominates the flood season when it occurs. The second highest magnitude class (*Large and Long*) is more common (36% of years). The volume centroids for both of these high magnitude classes define the peak of the flood season in late February and early March, with the events typically beginning in late January or early February. The *Long and Late* flood type is present in over half of the years and occurs most frequently with *Small and Late* and *Late Peak* events. The *Small and Early* and *Small and Late* events usually occur twice within the same season and in roughly two-thirds of the years on record, with both types in about half of the years. The *Late Peak* type is the most common: all other flood types are associated with this type in roughly two-thirds of the years in which they occur. The variability shown in the frequency and co-occurrence of the different flood types illustrates the complexity of the river's flood regime. At the annual scale, no single set of flood types or number of floods defines the flood regime, reflecting the highly variable regional climate. However, general expectations for a given year's composition of flood types based on the water year type can be made.

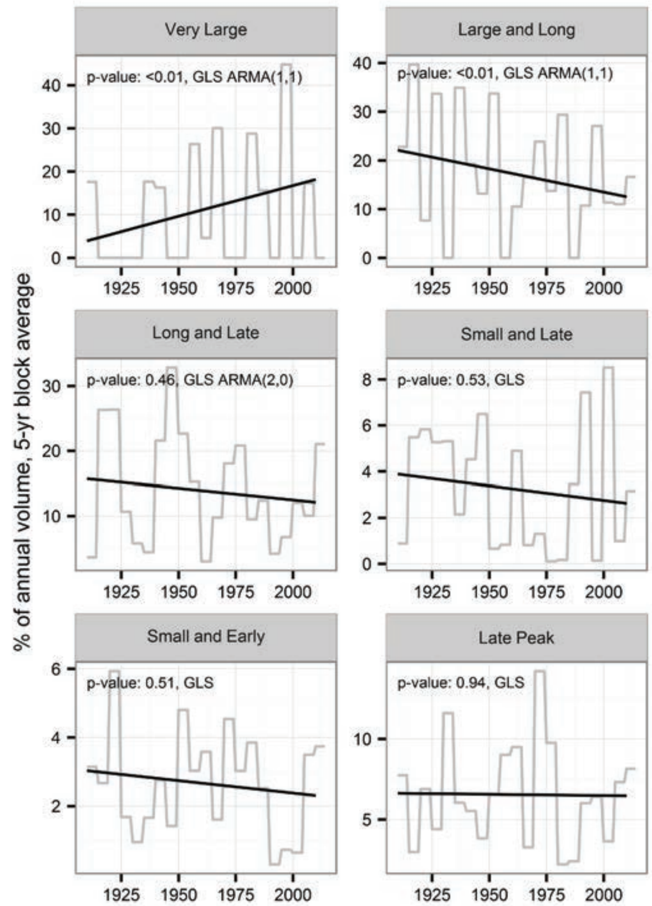

Figure 3-7. Time series of 5-year block averages of the percent of annual volume for each flood type over the Cosumnes River gage period beginning in 1910. The linear regression fits are shown as lines, with the slope, *p*-value, and GLS model used included as text within each plot.

Previous research on the Cosumnes River established 10 flood types based on pre-defined class boundaries for flood peak flow and duration (Booth et al., 2006). Using a similar floodplain connectivity threshold (25 m^3/s) to isolate flood events, three flood magnitude and four flood duration classes were used to classify flood events. Booth et al. (2006) found that their long (21-70 days) and small to medium magnitude (<100 m^3/s) flood type (L1) was associated with early spring timing, similar to this study's longer and later flood type (*Long and Late*). While the 10 types of Booth et al. (2006) offer more classes defined by peak magnitude and duration (selected to capture large differences applicable for management), these may be less meaningful than the class distinctions in this analysis defined by a wider array of metrics. For example, their short and low magnitude class (S1) included events classified into three different types identified here (*Small and Early*, *Small and Late*, and *Late Peak*). This comparison suggests that the six types defined by this study distill flood event variability into fewer classes while also accounting for a larger number of distinguishing and ecologically relevant characteristics.

There are potential limitations to the approach presented here. The daily flow record should cover a sufficiently long period to capture climatic variability. Given the dependence upon the underlying time series, awareness of the potential of nonstationarity to affect results is also necessary. In addition, effectively separating floods from the daily flow record requires that the flow at which the floodplain of interested is inundated be known, which can be difficult to determine particularly in highly modified systems. A selected discharge value that is lower than the floodplain inundation threshold will include events of minimal ecological relevance, as these floods would not activate the floodplain (sensu Williams et al., 2009). Similarly, a selected discharge threshold that is too high will omit some floods from classification that inundate the floodplain and affect ecosystem processes and functions. Relatively small variation in the threshold may alter the metric criteria and number and stability of clusters, but the basic flood type characteristics would likely persist. To explore this idea, we determined mean metric values for flood thresholds representing exceedance probabilities between 90 and 99%, and found they deviated from mean metrics of the selected threshold by less than 25% (except for the 99% exceedance probability flood volume metric, which deviated by 40%). As the threshold selection affects the lower flood flow days, flood volume and low peak magnitudes are expected to respond the most, which may either expand membership of the small magnitude flood classes or potentially even cause the elimination of these classes if the threshold is set much higher. The addition of new years of data also may affect flood types, potentially as a result of non-stationarity in longer term trends (Null & Viers, 2013), though these are likely to be relatively small changes to the class membership. In addition, though there may be a strong ecological rationale for the selected metrics for aiding interpretation, it must be balanced by statistical measures for determining cluster analysis parameters (Mackay et al., 2014). Finally, the benefits and drawbacks to available clustering techniques and algorithms should be taken into consideration when applying this typological approach to other systems.

The flood type classification approach as developed here can be applied to other river floodplain systems where flood conditions vary inter- and intra-annually. While hydrologic regimes and associated flood regimes vary widely across the globe, from highly seasonal tropical systems (Junk et al., 1989) to sporadic arid systems (Hughes & James, 1989), metrics relating to magnitude, duration and timing are expected to be universally applicable for interpreting ecological function (Agostinho et al., 2004; Hughes, 1990; Poff et al., 1997). Since climatic and watershed drivers vary widely across systems, flood types are expected to be quite different from those established in this study.

Relating flood types to watershed conditions

The flood types identified in this study reflect the physical state of the watershed, including climate and antecedent conditions. Other flood classification studies such as Merz and Bloschl (2003) have focused on such processes to define classes *a priori* (e.g., long-rainfall, short-rainfall, rain-on-snow, etc.). Hydrologic responses to climatic forcing vary across watersheds due to interacting factors including topography and geology (Wagener et al., 2007), making it useful to explore watershed-specific relationships to typical storm types while seeking to understand commonalities across watersheds. Examining the flood types of events known to be associated with particular conditions, which here include rain-on-snow, multiple storm events, atmospheric river events, snowmelt recession, first flood, as well as water year type, can help connect these types – not pre-defined by processes – to possible driving physical mechanisms of different flood types. These relationships are illustrated conceptually in Figure 3-8, where for each watershed or climate process, an arrow was drawn across the 90% ellipse to intersect the centroid of the associated floods identified from various existing datasets, described in the following text.

First, the largest floods on record for California's Central Valley are rain-on-snow events (Kattelmann et al., 1991). We found that, of 15 such events documented (Fissekis, 2008; Kattelmann et al., 1991; Leavesley, 1997), 10 aligned with the *Very Large* flood type and 3 with *Large and Long* events. Second, *Very Large*, *Large and Long*, and *Long and Late* events were associated with multiple storm events occurring close together (identified as >1 set of continuous days of precipitation over the course of a flood;

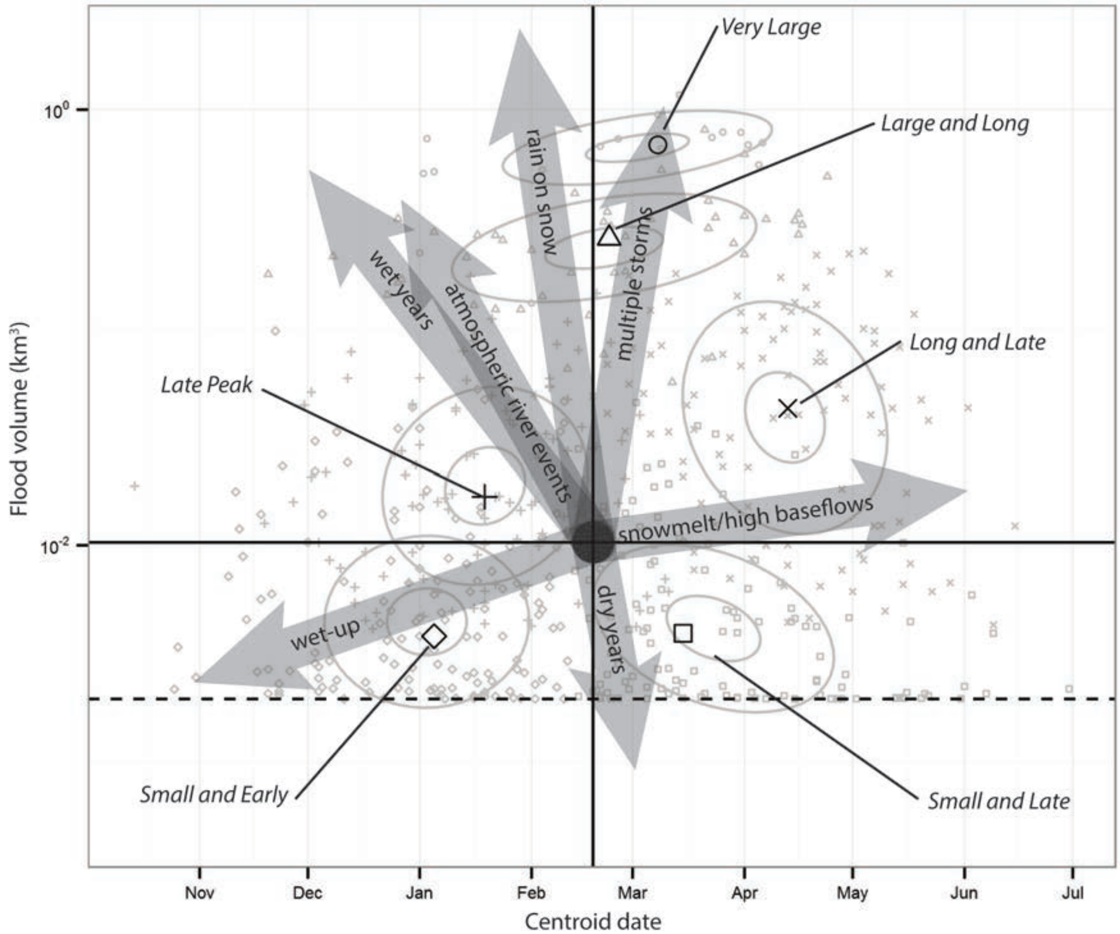

Figure 3-8. Association of flood types with climate and watershed conditions. Flood events of known conditions are represented by the arrows (drawn across the 90% ellipse to intersect the centroid), which overlay flood events (gray points) grouped by flood types (gray ellipses for 10% and 50% of the data).

Western Regional Climate Center, 2015). Third, using atmospheric river events summarized in Dettinger et al. (2011), we found that all of the *Very Large* events were associated with such storms within the period of available data (post-1948). The remaining events were classified as *Large and Long*, *Small and Early*, or *Late Peak*. Fourth, the flood events with volume centroids after the start of the spring snowmelt recession on the Cosumnes River – defined as April 13 by Epke (2011) – are shown to overlap predominantly with *Long and Late* and *Small and Late* events. Fifth, nearly all of the first floods of the season are either *Small and Early* or *Late Peak* events. This is likely associated with antecedent moisture conditions, where the watershed is still dry, producing short, but relatively higher magnitude events. Finally, events associated with very wet years span a range of flood types, but *Very Large* events occurred almost exclusively in these years. The dispersed nature of these events suggests that very wet years include a range of precipitation and watershed conditions that allow for this diversity of flood types. In contrast, events occurring in critically dry years are much more concentrated within the domain of *Small and Late* events, which occur at times of the year when larger events would otherwise be occurring. These associations suggest a physical basis to the identified flood types which supports the validity of the types, but also demonstrate that the types are not completely explained or separated by the watershed and climate factors explored here and thus support the classification performed.

Flood type interpretation for ecosystem functions

Aligning the characteristics of the resulting flood types from this analysis with ecological processes and functions facilitates the interpretation of the flood types for their ecological relevance (Figure 3-9). Floods of different magnitude serve different physical and ecological functions and affect species differently. Infrequent high peak magnitude flows – associated with the representative *Very Large* and *Large and Long* flood types shown in Figure 3-9 – are associated with sediment erosion and deposition

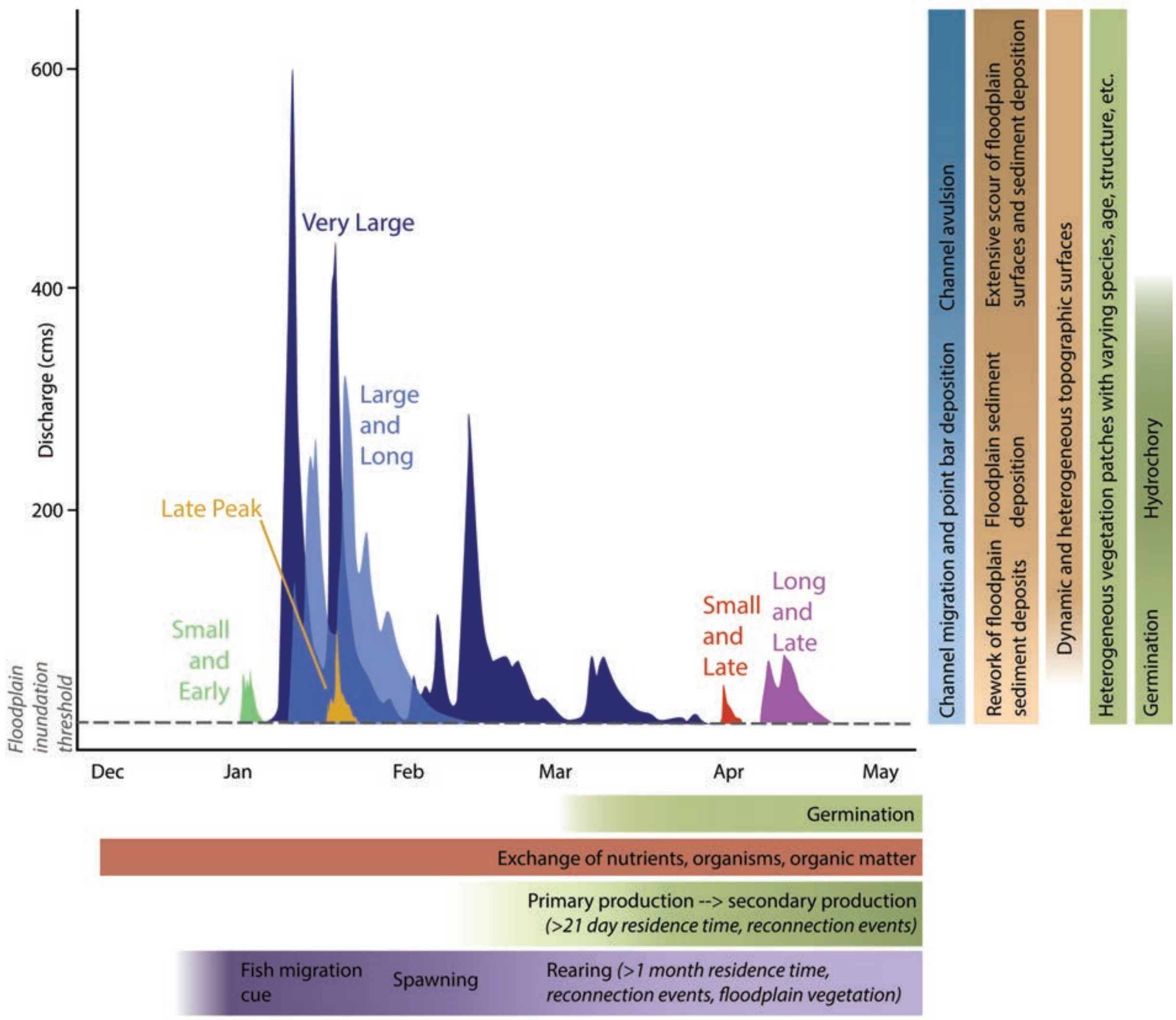

Figure 3-9. Floodplain physical and ecological processes and functions in California's Central Valley derived from available literature connected to the timing and magnitude of flood types (Andrews, 1999; Florsheim & Mount, 2002; Crain et al., 2004; Ahearn et al., 2006; Grosholz & Gallo, 2006; Stella et al., 2006; Jeffres et al., 2008; Opperman et al., 2010). Six flood events from the historical record, each most representative of the identified flood type, are shown. The selected processes and functions are shown along the axis (or axes) representing the driver(s) of relevance, with shading representing shifts in the processes. Characteristics of specific events, their antecedent conditions, spatial attributes of the floodplain, and other abiotic and biotic conditions will also affect ecological outcomes.

producing high levels of disturbance that supports heterogeneous habitat mosaics, resets successional processes, and reorganizes ecosystem structure (Florsheim & Mount, 2002; Resh et al., 1988; Ward et al., 2002). On the Cosumnes River, two levee breaching events in the 1980s and 1990s reconnected the floodplain to flood disturbance processes, resulting in sediment deposition and recruitment of large wood and initiating riparian forest successional processes (Andrews, 1999). Linking floods to their associated disturbance mechanisms at this site, Florsheim and Mount (2002) studied the sand-splay complex formation and evolution, which generated local physical variability that affected the patterns of riparian vegetation establishment. Their conceptual model of sand-splay generation links lower magnitude flood flows to reworking of sediment within the floodplain, while high magnitude events transport sediment onto the floodplain, creating new formations. The extent to which the sediment is moved within the floodplain is also affected by the event duration. Therefore, while the flood types with high peak flows (*Very Large*, *Large and Long*) serve critical functions of creating new floodplain landforms, lower magnitude flood types of sufficient duration that occur frequently (*Long and Late* and *Late Peak*) provide the regular addition of new substrate material and reworking of sediment to shift the habitat mosaic without resetting the landscape (Opperman et al., 2010).

Floodplain vegetation community composition is also affected by floods through hydrochory, for which the magnitude (via processes similar to sediment deposition) and timing (which is species-dependent) of flood events are governing factors (Nilsson et al., 2010). Subsequent successful recruitment of the dispersed seeds is dependent upon flood timing and recession rates, as addressed by the "recruitment box model" of Mahoney and Rood (1998), establishing a recession rate of 2.5 cm/day during the spring growth period for cottonwood seedlings. For the snowmelt dominated Tuolumne River in California, cottonwood seed dispersal aligned with the seasons peak flow period while willows were more associated with the later spring snowmelt flows (Stella et al., 2006). Mapping Cosumnes River flood types on these functions, the earlier season high flows of *Very Large* and *Large and Long* events can be expected to serve seed dispersal functions through hydrochory for cottonwood, while *Long and Late* and *Late Peak* events may also serve dispersal functions for willow species. Later season flooding that provides long-duration lower-recession rate receding hydrograph limbs with the capacity to promote seed germination and growth align with *Long and Late* as well as the end of *Very Large* and *Large and Long* events. Multiple flood types therefore support different riparian forest successional processes.

Regular floodplain connectivity via frequent lower peak flows create dynamic and heterogeneous habitat conditions in space and time (Junk et al., 1989; Tockner et al., 2000). Such flood events can substantially reconfigure floodplain habitat mosaics spatially without necessarily changing the overall composition (Ward et al., 2002). The frequent low magnitude pulses promote nutrient exchange and the movement and transformation of organic matter (Robertson et al., 1999), as well as serve species' life history requirements such as providing fish spawning and rearing habitat (Welcomme, 1979). Inundation timing, duration, and connectivity control fish habitat conditions as well as primary and secondary productivity, which generates needed food for rearing juvenile fish and for export downstream (Sommer et al., 2004). For example, fish with different reproductive strategies in the Upper Pantanal, Brazil have been shown to be highly correlated with the duration, timing and magnitude of flows (Bailly et al., 2008). Baranyi et al. (2002) found zooplankton productivity and community composition within a Danube River floodplain to be particularly correlated with water age, representing hydrologic and connectivity conditions. Within a floodplain restoration site along the Cosumnes River, Ahearn et al. (2006) demonstrated periods of disconnection and reconnection could maximize primary productivity and export, and Grosholz and Gallo (2006) found that zooplankton biomass peaked with residence times of two to three weeks. Of the established flood types, intra-annually frequent *Small and Late*, *Long and Late*, and *Late Peak* flood types offer the shorter reconnection events that support these floodplain functions. The *Small and Late* and *Long and Late* types as well as later season *Late Peak* events occur during the time of the year when juvenile fish, such as Chinook salmon or California native and obligate floodplain spawning Sacramento splittail, use floodplain habitats, including those along the Cosumnes River, for rearing (Jeffres et al., 2008; Sommer et al., 1997). Research has shown that native fish populations on the Cosumnes River are supported over alien fishes by early spring flooding, followed by disconnection (Crain et al., 2004). Also, the temporal and spatial complexity of habitat produced by variable flooding conditions allows fish to locate optimal habitat conditions (Jeffres et al., 2008). While the *Very Large* and *Large and Long* events may also serve such functions if they continue into the spring months, their low frequency and long periods of connection suggests that these events alone would be inadequate to sustain viable fish populations.

Management implications

The flood typology presented here offers characterization both within and across years of a variety of flood conditions within floodplains that could be used to achieve greater variability reflective of more natural conditions in managed riverine systems. As such, this flood typology can provide an important basis for hydrodynamic modeling, flood type forecasting, and ecological studies linking flood types to specific functions to inform management decisions. Overall, this approach supports efforts to maintain natural variability, a core principle in river restoration (Naiman et al., 2002; Petts, 2009; Poff et al., 1997; Ward et al., 2001). Managing toward a more natural flood regime, with the flood typing methods presented here helping define spatial and temporal variability of flood characteristics driving floodplain habitat diversity, is expected to promote ecosystem diversity and productivity (Ward et al., 1999).

For the largely unregulated Cosumnes River, the primary management variables are landscape modifications, which could be made to best take advantage of the identified flood types for supporting a suite of ecological functions and processes within the floodplain. At the Cosumnes River floodplain restoration site of focus here, managers can use these flood types in a variety of applications to refine expectations for the type and extent of physical habitat provided within and across years. Previous research on the Cosumnes River has influenced the development of setback levees in other river systems of California's Central Valley (Andrews, 1999; Stofleth et al., 2007), and the established flood types could be used to evaluate how these restoration projects have changed floodplain inundation patterns. Additionally, future projects can use these flood types to evaluate the potential effectiveness of different restoration scenarios through characterizing the varying response to different flood types. For example, hydrodynamic modeling of these flood types could improve understanding of floodplain inundation spatiotemporal variability, and specific flood types could be targeted through floodplain restoration for the physical habitat they would be expected to provide.

By characterizing variable conditions, a flood regime typology of unimpaired hydrology also offers detailed information that can be applied, for example, to environmental flow targets of regulated rivers to better prioritize the range of conditions to which species are adapted (Nislow et al., 2002). Specifically, it can provide a simplified set of flood types with associated characteristics and frequencies to target in given years. Further, a flood typology for a regulated system can be used to compare to unimpaired conditions and more explicitly characterize regulation impacts and potential ecological consequences. Having multiple metrics and flood types also allows for finer resolution of particular aspects of change. For example, land use change may affect the rising rate of spring floods or cause new summer floods (Sparks et al., 1998), flow regulation may increase the frequency of floods during particular seasons (Robertson et al., 2001), dams may only affect high magnitude flows, or climate change may increase the frequency of high-magnitude winter floods while changing the timing or existence of floods related to snowmelt (Safeeq et al., 2015; Stewart et al., 2004).

Tools to not only identify, quantify, and classify such dynamics, but also to manage for that variability are essential to better manage highly modified rivers for ecological functions. The flood types identified here capture the predominant flood characteristics from a complex flood record while offering greater detail that can be used to link floods to their ecological implications than a typical flood frequency analysis. In practice, ecosystem management goals can be more clearly articulated with flood type inter- and intra-annual frequency, and with association to water year type and other climatic and watershed conditions. More refined flood flow targets, in terms of annual variability, timing within a season, as well as magnitude and duration, may be possible knowing that only some years or certain flood types provide particular habitat conditions. For example, a better understanding of flood types and their variability could refine general conservation objectives concerning floodplain activation flows and connectivity to floodplain habitat in current Central Valley flood management plans (DWR 2015). The clustering methods presented here identify flood types of a flood regime, drawing attention to this critical component of flow regimes. Analysis of flood type inter- and intra-annual variability offers greater opportunity to examine as well as manage for variability of a range of flood characteristics.

Furthermore, focusing on the flood regime and its inherent variability, as opposed to satisfying particular species requirements, can encourage process-based management approaches that support overall ecological integrity and resilience (Beechie et al., 2010; Tockner et al., 2003; Wohl et al., 2005). Application of this research can also inform the functional flows approach proposed by Yarnell et al. (2015), where established flood types could be included as targeted components of the hydrograph for their relationship to geomorphic and ecological functions. Overall, the flood regime typology established here provides a useful tool for understanding the lateral dimension of the natural flow regime and managing riverine and floodplain environments.

CONCLUSIONS

Sustaining freshwater ecosystem functions and improving resilience to future anthropogenic change requires not only a better understanding of floodplain inundation patterns, but also readily applicable techniques to classify and quantify such patterns and their inherent variability. This study presents a flood typology to inform characterization of a river's flood regime, applied to a lowland floodplain site on the unregulated Cosumnes River of California, USA. Traditional hydrologic classification techniques are utilized, including k-means cluster analysis, to establish a systematic description of a river's flood regime relevant to floodplain ecosystems.

We show that flood event characterization, using ecologically meaningful metrics such as magnitude, timing, duration, rate of change, and hydrograph shape, is useful for classifying the highly inter- and intra-annually variable Cosumnes River flood regime. A total of six flood types were distinguished: 1) very high peak magnitude floods, 2) high peak and long duration floods, 3) later season and longer duration floods, 4) low magnitude and late season floods, 5) low magnitude and early season floods, 6) and floods where peak flows occurred later in the hydrograph. Assessing inter- and intra-annual frequencies of the flood types revealed a much higher resolution of flood variability than available with typical flood frequency analysis. High magnitude and long duration flood types were predominantly associated with wet years and the similarly-timed small event type was associated with dry years; we also found that the small and early type was less associated with the type of year and instead typically marked the first floods of the season. Trend analysis showed that very large floods have significantly increased while the flood type with the second highest magnitude has decreased over the period of record.

While different typologies and associations to specific ecological functions are expected in other systems, the broadly applicable approach presented here is shown to systematically identify flood types and quantify them as a means to describe a river's flood regime. Application of these methods provides a refined understanding of floodplain hydrological conditions. Characterizing flood types using variables universally understood for their ecological significance, such as magnitude, timing, and duration, facilitates the ecological interpretation component of this approach. To the extent that ecological information is available, flood type interpretation can link types to the functions served (or identify types with little ecological relevance) and quantify characteristics and variability related to ecosystem functions. The identified flood types can also be used to improve understanding of climatic and watershed processes driving flood variability. We propose that this approach can be used to improve environmental flow targets for floodplain systems, by managing toward more natural flood regimes characterized by aspects, such as magnitude, timing, and duration, universally well-established as ecologically important. And, where possible, this approach can leverage existing information to better understand the ecological implications of flood types. In sum, this research offers new ways to establish needed information to support more functional floodplain inundation regimes within our current and future riverine landscapes.

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CHAPTER 4

HYDROSPATIAL ANALYSIS TO QUANTIFY SPATIOTEMPORAL PATTERNS OF INUNDATION ON A RESTORED CENTRAL VALLEY FLOODPLAIN, CALIFORNIA

Alison A. Whipple

Hydrospatial analysis to quantify spatiotemporal patterns of inundation on a restored Central Valley floodplain, California

ABSTRACT

Rivers and their floodplains interact dynamically in space and time, supporting diverse and productive ecosystems. Floodplain processes have been intensely disrupted by human modifications, which continue with changing demands for water and land uses, as well as climate change. New strategies are needed to improve ecosystem functions and resilience. Quantification of the hydrospatial regime, defined as spatiotemporal floodplain inundation patterns, advances understanding of how river and floodplain management actions, such as levee setbacks or environmental flow regulations, affect land-water interactions and variability. This research establishes a framework for analyzing the hydrospatial regime and demonstrates its utility in describing impacts of a levee-removal floodplain restoration along the lower Cosumnes River, California. This captures the lateral dimension of the flow regime, including connectivity and heterogeneity. Based on two-dimensional hydrodynamic modeling, this multi-metric and multi-dimensional approach was developed to quantify conditions in space and time over a period of record. Results for the Cosumnes River floodplain restoration show different responses to restoration depending on flow, floodplain position, and specific metric, illustrating the utility of spatial and temporal quantitative metrics. Although accumulated inundated area over time and velocity increased with restoration, variables such as duration and depth decreased or changed only slightly. Spatially, changes were concentrated near the primary levee-removal site. The most substantial gains in extent and duration of inundation occurred with more frequent intermediate flood flows, whereas the highest flood flows declined in maximum inundated area with restoration. Hydrospatial analysis refines restoration expectations and informs management strategies, and is broadly applicable to large rivers and their floodplains.

INTRODUCTION

Riverine landscapes (sensu Thorp et al., 2006; Ward et al., 2002a) are defined by their variability and complexity in space and time, particularly in lowland alluvial river-floodplain environments. Shifting mosaics of terrestrial and aquatic habitat patches and diverse and interconnected ecosystem processes, operating across a range of spatiotemporal scales, support high levels of biodiversity and productivity on floodplains (Amoros & Bornette, 2002; Opperman et al., 2017; Tockner & Stanford, 2002; Ward et al., 2002b; Ward et al., 1999). The interaction between flood regime and floodplain morphology is captured in characteristic spatiotemporal inundation patterns, defined here as a floodplain's hydrospatial regime.

Human activities have deeply disrupted the processes and functions of riverine landscapes, resulting in severely degraded freshwater-dependent ecosystems globally (Dudgeon et al., 2006; Tockner et al., 2010). Low-lying floodplains are particularly vulnerable due to their high desirability for agriculture and urban development and their accumulation of upstream impacts (Naiman et al., 2002; Tockner & Stanford, 2002). Sustaining viable populations and supporting functional ecosystems must be done within bounds placed by human uses and the realities of fundamentally altered, or novel ecosystems (Acreman et al., 2014; Palmer et al., 2004; Poff, 2017). This requires improved understanding and management of complex physical processes, and their variability, to which freshwater ecosystems are adapted.

Streamflow fundamentally shapes aquatic and riparian ecosystem structure and function, which is a central concept in riverine ecology (Naiman & Décamps, 1997; Poff et al., 1997; Power et al., 1995). Flood pulses – with characteristic magnitudes, frequencies, durations, timing, and rates of change (Poff et al., 1997) – shape and maintain geomorphic patterns and processes and also control many environmental variables affecting biotic community composition and ecosystem productivity (Junk et al., 1989; Tockner et al., 2000). Hydrologic connectivity – the exchange of materials, energy, and organisms through water (Pringle, 2001) – is primarily a product of flood regime interacting with the landscape. It is manifest in the heterogeneous and dynamic conditions of riverine landscapes at multiple scales and varies across longitudinal, lateral, vertical and temporal dimensions (Ward, 1989). The ecological significance of spatiotemporal heterogeneity within riverine landscapes is well recognized (Power et al., 1995; Ward, 1989). Differences in hydrologic connectivity and related environmental variables help shape macroinvertebrate and fish assemblages and biodiversity (e.g., Andrews et al., 2014; Arthington & Balcombe, 2011; Davidson et al., 2012; Gallardo et al., 2009; Simões et al., 2013; Tockner et al., 2000). An important implication of physical heterogeneity interacting with variable hydrology is that different locations support different ecological functions at different times, and this diversity of conditions increases the chances that species can meet life history requirements and exploit favorable conditions (Amoros & Bornette, 2002; Sparks, 1995; Sparks et al., 1998).

Restoration of river-floodplain ecosystems would benefit from better quantification of spatiotemporal floodplain inundation patterns to define relationships between hydrologic regime and landscape heterogeneity and to develop appropriate flow- and form-based actions (Bond et al., 2014; Tockner et al., 2000). Relationships between flow, floodplain geomorphology, and floodplain inundation characteristics are not necessarily aligned or linear (Hudson et al., 2013; Karim et al., 2016; Turner & Stewardson, 2014). Measures previously applied to characterize floodplain inundation relate to flow regime elements of magnitude (e.g., flooding extent, depth, velocity), frequency, timing, duration, and rate of change, as well as spatial variables representing connectivity and heterogeneity (e.g., Cienciala & Pasternack, 2017; Murray-Hudson, 2009; Powell et al., 2008; Stone et al., 2017). Assessing variability in these characteristics allows for more detailed and accurate understanding of ecosystem processes and functions and potential response to restoration actions (Chen et al., 2015; Karim et al., 2013; Poff, 2002; Ward et al., 1999).

Early hydraulic assessments include the widely applied instream flow incremental methodology (IFIM) and associated physical habitat simulation (PHABSIM; Bovee, 1982). However, such assessment methods typically focus on in-channel, single-species flow-habitat evaluations, and are not typically concerned with quantifying spatiotemporal variability and complexity. Recent research has shown improvements in quantifying floodplain inundation patterns from the reach- to basin-scale, where more local studies are often informed by hydrodynamic modeling while large-scale assessments are performed using satellite imagery. Experiencing rapid advancements (Teng et al., 2017), hydrodynamic modeling for floodplain inundation, now often done in two-dimensions (2D, depth-average velocity), has benefited from improved technology and computing power, making more physically-based models at higher resolution possible. Increasingly applied for restoration (e.g., Clilverd et al., 2016; Gergel et al., 2002; Jacobson & Galat, 2006; Matella & Merenlender, 2015; Wen et al., 2013), hydrodynamic modeling allows for efficient and effective description of floodplain inundation and hydrologic connectivity. Many studies have a specific focus, such as availability of shallow water habitat with different flow and channel alternatives (Jacobson & Galat, 2006), inundated area and flood duration impacts of dams (Nislow et al., 2002), and large scale inundation patterns for environmental flow determination (Thomas et al., 2015). Matella and Jagt (2014) extended beyond typical evaluations to establish area-duration-frequency curves and quantify expected annual habitat. Stone et al. (2017) examined the implications of flow regime alternatives on floodplain factors including inundation frequency and flux. Hermoso et al. (2012) included a "water residency time" metric (in units of area-days) to determine floodplain inundation extent and duration.

The need to support functional floodplain ecosystems with modified and changing hydrologic regimes and riverine landscapes motivates this hydrospatial analytical approach. It is informed by existing flow ecology and landscape ecology principles, but formalizes a method to characterize the floodplain hydrospatial regime by quantifying spatial and temporal variability of inundation patterns. Its utility is demonstrated in comparing pre- and post-restoration conditions. This hydrospatial analysis uses 2D hydrodynamic modeling to establish spatially-resolved daily depth and velocity estimates for a daily flow time series, and subsequently summarizes conditions for ecologically-relevant metrics in space and time and at daily, event, and water year scales. Hydrospatial analysis is applied to a recent levee-removal restoration site along the lower Cosumnes River floodplain within the Central Valley, California. This chapter describes the approach developed for hydrospatial analysis, discusses metrics applied, examines the effects of restoration over spatial and temporal scales, explores relationships to hydrologic conditions represented by water year type and flood type, and discusses management implications. Hydrospatial analysis advances understanding of spatiotemporal land-water interactions, and thus allows for more explicit links between water and land management options as well as ecological functions.

METHODS

Overview

The general process established for hydrospatial analysis consists of several components (Figure 4-1). A 2D hydrodynamic model was developed for the Cosumnes River case study area for both pre- and post-restoration topographic conditions and then used to determine spatially-resolved flow-depth and flow-velocity relationships. Using the historic record daily flow time series, flow depth and velocity was computed across the 2.1 km^2 site. Analysis of output data involved calculation of ecologically-relevant spatiotemporal metrics derived from flood regime elements of magnitude, timing, duration, rate of change, and inundation frequency, as well as spatial characteristics relating to connectivity and heterogeneity. A set of metrics were chosen to describe these primary characteristics and considered for their ecological relevance, multi-collinearity and capacity to explain the variance in the data, ease of interpretation, use in the literature, as well as computational feasibility. Selected metrics were applied to the hydrodynamic modeling-derived output data and conditions summarized spatially and temporally at the flood event and water year scale. Pre- and post-restoration conditions were then compared.

Study area

This study was performed for a floodplain restoration site along the lower Cosumnes River within the Central Valley of California (Figure 4-2). The Cosumnes River drains an approximately 2,460 km^2 watershed on the west slope of the central Sierra Nevada mountain range. It reaches a maximum elevation of 2,300 m and flows into the Sacramento-San Joaquin Delta at sea level where it then joins the Mokelumne River. No large dam regulates its flow, partly attributable to its smaller area and lower average elevation (lack of snowpack) in comparison to other major watersheds of the Sierra Nevada. While land use change, including grazing, has affected the upper watershed, this part of the watershed remains largely undeveloped. However, the foothills and lower alluvial reaches are highly impacted by multiple pressures, including levees, channel incision, gravel mining, and groundwater abstraction associated with agriculture

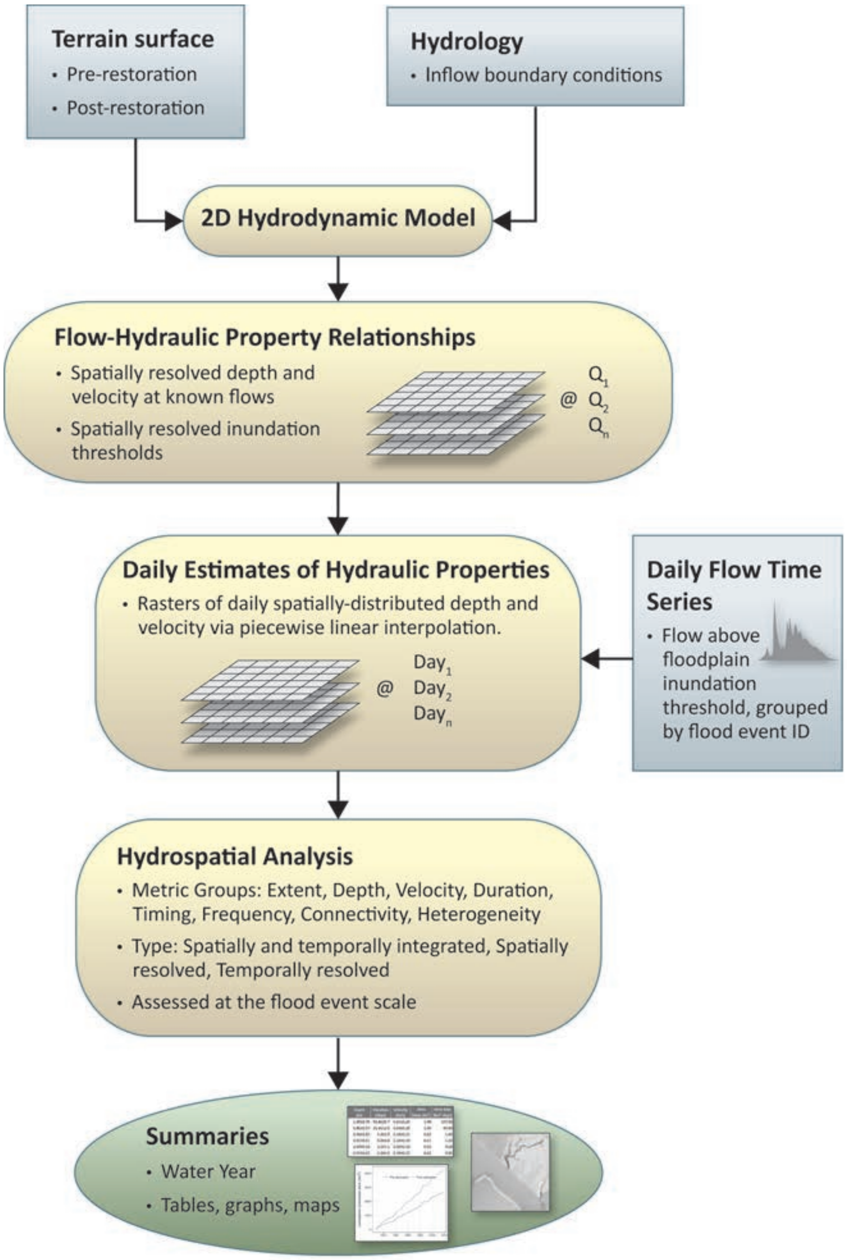

Figure 4-1. Process for preparing and conducting hydrospatial analysis, characterizing floodplain inundation patterns in space and time.

and urban development (Constantine, 2001; Fleckenstein et al., 2004). The river's sediment regime was also heavily altered beginning in the mid- to late-1800s by Sierra Nevada gold mining.

The climate is Mediterranean-montane, characterized by cool, wet winters and hot, dry summers along with high inter-annual variability. Spatially-weighted mean annual precipitation (1971-2000) is approximately 855 mm (PRISM Climate Group, 2006). Mean annual daily flow in the Cosumnes is 14 m^3/s over the 110-year flow record, with an instantaneous peak flow of 2,630 m^3/s (1997). In dry years, flow ceases by the end of summer in the lower reaches, exacerbated by groundwater pumping (Fleckenstein et al., 2004). The notably high variability of water resources of this region is attributed to the fact that the majority of precipitation and runoff typically occurs in just a few brief winter storms (Dettinger et al., 2011). Flood flows, defined by the observed floodplain inundation threshold of 23 m^3/s, account for two-thirds of annual flow (see Chapter 3; Whipple et al., 2017). The majority of runoff occurs in the wet season between December and May. The hydrograph is mixed rain-snow, with high winter storm peaks and a spring snowmelt recession (Yarnell et al., 2010). As only about 10% of the Cosumnes watershed typically receives snow (>1,500 m elevation), snowmelt is a less significant component of the hydrograph than in other higher-elevation watersheds. Climate change impacts to the Sierra Nevada are associated with declining snowpack and earlier melt (Cayan et al., 2008), and increased precipitation and flood extremes (Berg & Hall, 2015; Das et al., 2013; Pierce et al., 2013) and drought risk (Diffenbaugh et al.,

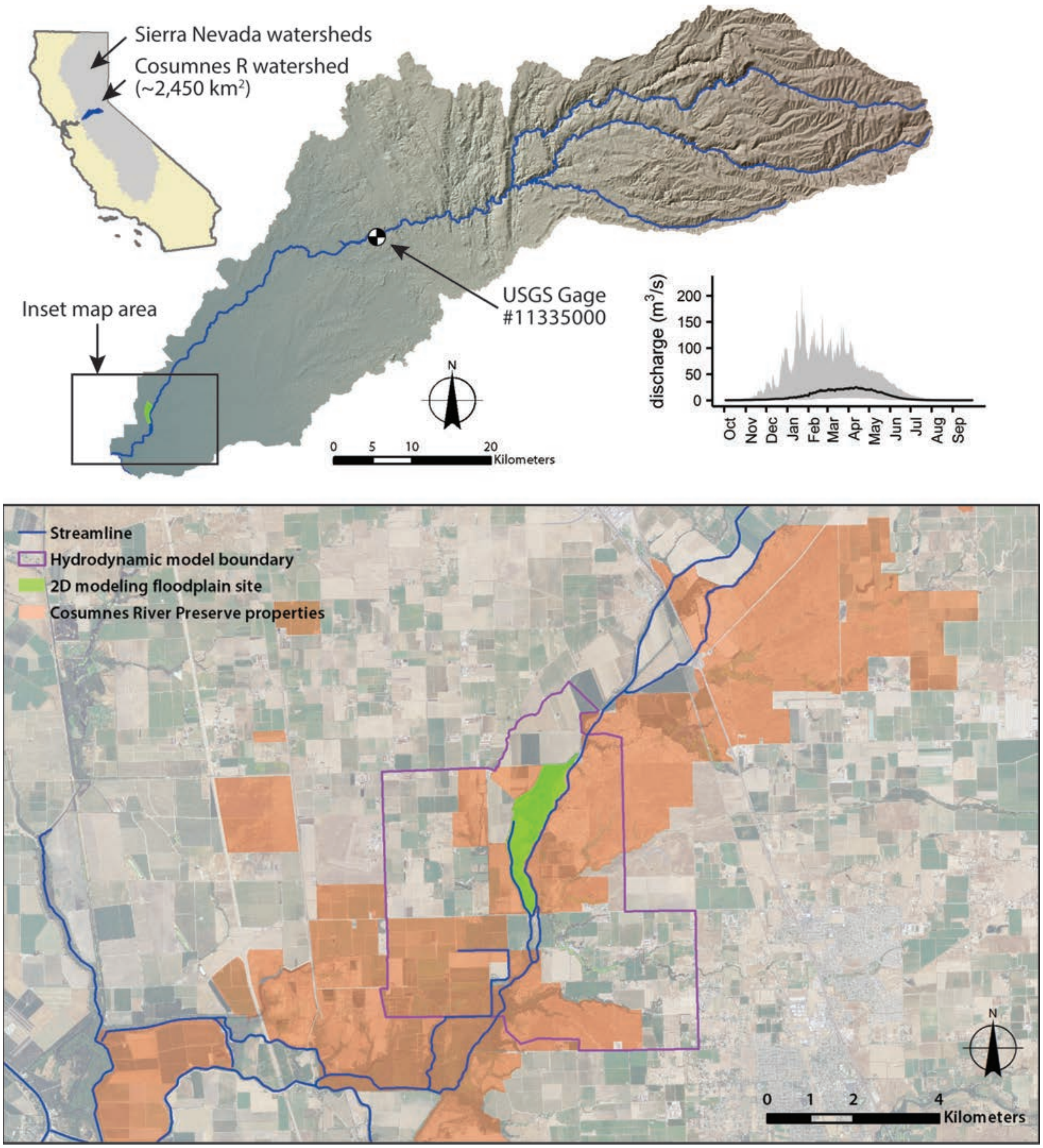

Figure 4-2. Map depicting the Cosumnes River watershed, located along the west slope of California's Sierra Nevada. The location of the USGS stream gage used in the study and the reference floodplain approximately 45 km downstream are illustrated. The inset graph shows median daily flow with shading representing the 5th-95th percentile at the gage. (Aerial imagery: USDA 2016)

2015), which is supported by observed changes in snowmelt and earlier peak runoff over recent decades (Mote et al., 2005; Mote et al., 2018). Recent research also suggests increases in extreme high floods and declining trends in spring snowmelt floods on the Cosumnes River (see Chapter 3; Whipple et al., 2017).

The lower floodplain receives on average five flood events per year. The flood typology of Whipple et al. (2017) described six flood types characterizing the Cosumnes flood regime, including an infrequent very high and multi-peaked, long-duration flood type, a second high flow type, a later spring snowmelt-related flood type, a type distinguished by earlier peaks within the flood event, and two low magnitude and short duration types, separated by early and late season timing. The four lower magnitude types were found to occur in the majority of years.

Unlike the many regulated and leveed rivers in California's Central Valley, the Cosumnes River regularly accesses large areas of its historic floodplain, though many levees still limit connectivity. Over the last three decades, process-based levee removal restoration experiments have been conducted within the Cosumnes River Preserve (managed by The Nature Conservancy and a consortium of agencies), generating landscape-scale transformation. Research has linked the increased hydrologic connectivity to sediment deposition patterns (Florsheim & Mount, 2002; Nichols & Viers, 2017), riparian forest successional processes (Trowbridge, 2007), primary and secondary productivity (Ahearn et al., 2006), and native fish spawning and rearing (Jeffres et al., 2008). This research has provided useful insights that inform restoration elsewhere in California and globally (Opperman et al., 2009). Although the lower Cosumnes River is highly modified, some floodplain areas interact with a largely natural flow regime. For these reasons, the Cosumnes River floodplain is a unique opportunity to study spatial and temporal variability of inundation and effects of restoration.

The particular restoration site here is in a rural agricultural landscape. Prior to substantial anthropogenic change beginning in the mid-1800s, the area was a complex riverine landscape, referred to as the "Cosumnes Sink" with patches of perennial and seasonal wetlands intersected by anastomosing and distributary channels and natural levees occupied by valley oak riparian forests (Whipple et al., 2012). The river has since been confined to a single incised channel (Constantine, 2001), and the landscape is both drier and homogenized topographically. However, during larger floods, the floodplain site, neighboring restoration areas, and adjacent agricultural fields, become a vast inundated area, connecting with the low-lying Delta downstream, reminiscent of the "inland sea" created historically by floods in the Central Valley (Kelley, 1989).

The restoration site is about 1.1 km^2, bound on its eastern side by the Cosumnes River, to the north by riparian forest, to the west by levees protecting agriculture and farmhouses, and coming to a point at the downstream end (Figure 4-3). Aside from the forested area in the north and trees lining higher elevation levees, the majority of the site is covered with herbaceous vegetation. While the site is fairly level, the riparian forest north of the main levee breach is more topographically complex and has several side channels that convey substantial flow onto the restoration site during floods. Due to this connectivity, the <1 km^2 forested area was also included in the modeling and analysis (for a modeled area of 2.1 km^2). A remnant side channel also runs along the west boundary on the east side of the levee. This side channel is connected to the river during floods at the site's downstream end. An earlier unplanned levee breach introduced some topographic complexity, developing a sand splay at the center of the site prior to restoration. The main element of restoration project construction, implemented in the fall of 2014, was the removal of approximately 250 m of river-side levee at the upstream end of the site along with a large graded area. A section of levee (~75 m) also was removed and a swale (~2 m deep) was excavated at the downstream end of the site. In addition, three sections of an interior "ring" levee were removed. The restoration significantly increased flood extent and frequency for the site, and subsequent floods

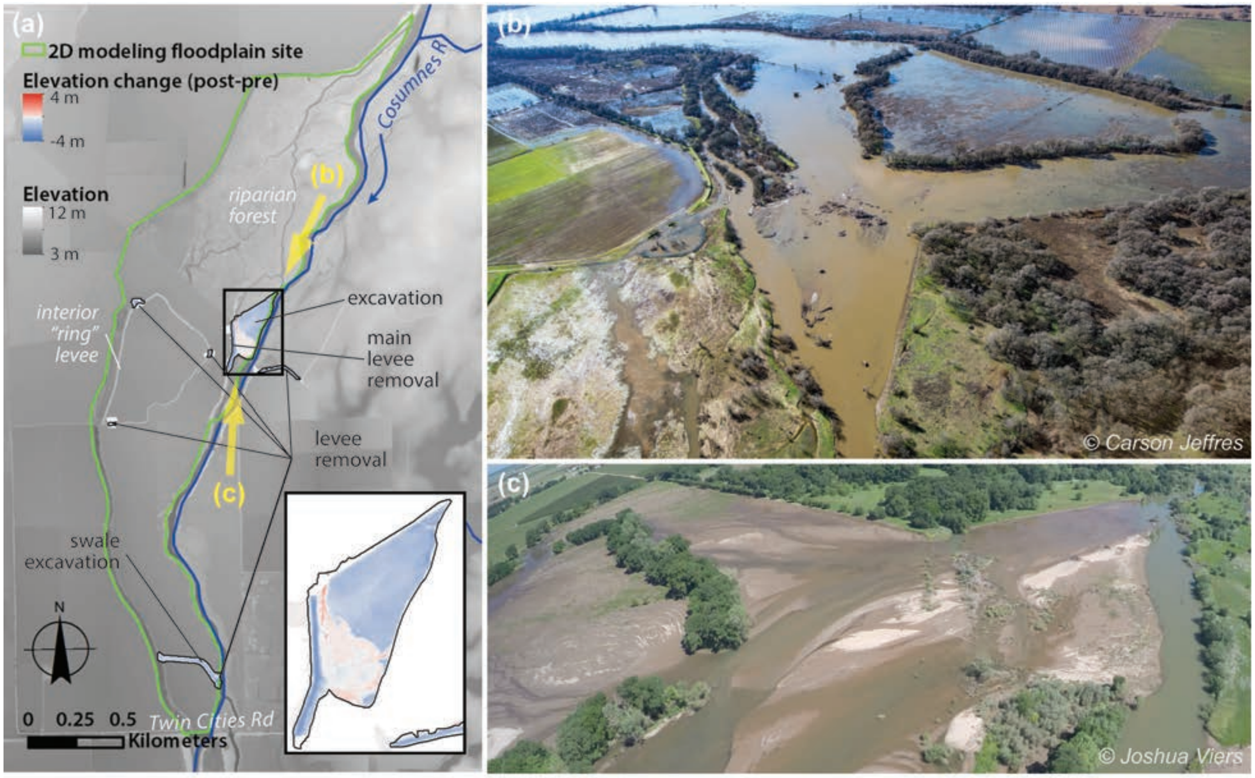

Figure 4-3. Cosumnes River floodplain restoration site with excavation and levee removal topographic alterations (a) and post-restoration aerial photographs looking downstream (b) and looking upstream at the main levee breach location (c). In (a), areas where topographic alteration occurred due to restoration are shown in color, with blue indicating elevation loss post-restoration and red indicating elevation gain. The inset map shows elevation change in the area of the main levee removal. (LiDAR: CDWR 2010)

have initiated a range of hydrogeomorphic responses, causing spatially variable sediment deposition and erosion and recruiting riparian vegetation (Nichols & Viers, 2017).

Hydrologic and hydraulic data

The primary hydrologic data used in this study was from the long-term continuous streamgage on the Cosumnes River at Michigan Bar (MHB, #11335000; U.S. Geological Survey, 2017), with mean daily flows available back to October 1907 (or the 1908 water year, WY). For developing and calibrating the hydrodynamic model, 15-minute flow time series data from the MHB streamgage was used, which begins in WY1984. The MHB streamgage is approximately 45 km upstream of the floodplain site, capturing most of the watershed area (57%) and the area generating most of the runoff. For modeling purposes, however, all flows were transformed to represent flows at the site, accounting for time lag and tributary inputs (Appendix A). This process was informed by previous hydrologic modeling of the Cosumnes River and adjacent watersheds to develop flood frequency flows for a regional hydrodynamic model (David Ford Consulting Engineers, 2004). Correlations between MHB and tributary flows were established, and travel time lags were computed by comparing the arrival time of peak flow at MHB and the floodplain site. The daily flow record of a USGS streamgage just upstream of the site (MCC, #11336000), operating between 1943 and 1982, was used to assess the effectiveness of the transformation, using criteria established by Moriasi et al. (2007). A satisfactory fit was determined based on a Nash-Sutcliffe Efficiency (NSE) of 0.69, percent bias (PBIAS) of 9.8%, and a ratio of root mean square error to observation standard deviation

(RSR) of 0.55. Water surface elevation (WSE) data were used for hydrodynamic model calibration. Field-based monitoring at several in-channel and floodplain locations was performed using Solinst® Levelogger® water level data loggers. Data were collected at 15-minute intervals over the three years leading up to the restoration (WY2012-2014) and the two years after restoration (WY2015-2016).

Once hydrodynamic modeling was complete, all subsequent analyses used the MHB 110-year mean daily flow record (WY1908-2017). The record was processed by first extracting only days having floodplain inundation, using a previously-determined conservative flood threshold of 23 m^3/s established by floodplain connectivity observations over the two decades prior to restoration in 2014 (see Chapter 3; Florsheim et al., 2006; Whipple et al., 2017). Though some floodplain areas inundate at lower thresholds post-restoration (e.g., the excavated swale), these areas are small and would not substantially affect the overall results. Continuous days above the flood threshold were assigned individual flood event and type identification numbers, following Whipple et al. (2017). Days were also characterized by whether they were part of a rising or falling limb of a hydrograph and assigned the highest flow within the last seven days.

Hydrodynamic modeling

A coupled one- and two-dimensional hydrodynamic model was established for the lower Cosumnes River and its floodplain to obtain spatially-resolved estimates of depth and depth-averaged velocity at various flows within the floodplain study area. The modeling used the U.S. Army Corps of Engineers' Hydrologic Engineering Center River Analysis System (HEC-RAS 5.0) software (Brunner, 2016). The software can be used for one-dimensional (1D) or 2D modeling or in combination as, in this case, a coupled 1D river channel and 2D floodplain area. Models can be run using the Diffusion Wave equations or the full Saint Venant depth-averaged shallow water equations for conservation of mass and momentum. The Saint Venant method was applied in this study. The model employs an implicit finite volume solution algorithm to solve the equations, allowing for longer computational time steps, enhanced stability, and more efficient wetting and drying (conditions common in floodplain environments). The computational grid was unstructured, providing greater flexibility in cell size and shape (from three to eight sides). Another model feature valuable to this application is the use of subgrid terrain to inform detailed hydraulic property tables of the computation grid cells. This feature allows larger computational grid cells, while retaining the resolution of underlying topographic data in model output. As with many hydrodynamic models, this model does not account for some hydrologic processes, such as evapotranspiration and surface water-groundwater interactions. Also, data requirements and computation time can be substantial, potentially restricting the extent and/or resolution of the model.

The model uses inputs of topography, hydrologic boundary conditions, and spatially variable land surface roughness. Separate models were constructed for the pre- and post-restoration floodplain configurations (before and after WY2015). For pre-restoration conditions, topography was derived from 2007 LiDAR (CDWR 2010; Hegedus & Simmons, 2011). Areas that were topographically altered by restoration activities were subsequently surveyed in the field using Real Time Kinematic (RTK) Global Positioning System (GPS) equipment (Topcon HiperLite+ and HiperV) in the fall of 2014. These newer survey data were combined with the 2007 LiDAR to generate a continuous surface representing post-restoration conditions. An interpolated surface based on cross-sectional surveys (spaced approximately every 200 m) was used for in-channel geometry, which was held constant for both restoration conditions. The interpolated channel surface was developed because the dense riparian forest canopy adjacent to the channel obscured some of the LiDAR data. All digital elevation data used as input to the model was of 1 m resolution.

The hydrodynamic modeling followed previous studies in the region, including 2D modeling of the site done for permitting (Robertson-Bryan Inc., 2011) and earlier regional 1D modeling (Blake, 2001; Hammersmark et al., 2005). However, the data, model configurations, and output resolution were not directly applicable for this research. For this study, the model configuration included approximately 10 km of channel, the 2.1 km^2 gridded floodplain study site of focus for 2D modeling, 19 storage areas representing other nearby floodplain areas in 1D, as well as multiple structures connecting model elements. The model grid representing the floodplain restoration site consisted of 1,250-1,400 computational grid cells (the number varied somewhat between pre- and post-restoration configurations) at an average cell size of ~1,600 m^2 (with a range of approximately 110-4,200 m^2). Output was generated at the 1 m resolution of the topographic input provided by the subgrid capability of the model. In developing the computational mesh, cell boundaries were aligned with topography. The total area of the model domain was ~42 km^2, allowing for adequate distance between the floodplain of focus and model boundaries to minimize boundary effects. Spatial land cover data, produced by the Geographic Information Center for the California Department of Water Resources, were used to set spatially variable roughness estimates, based on reported values (Chow, 1959; GIC 2012). Upstream boundary input was at 15-minute intervals, based on the transformed 15-minute MHB USGS flows, and a normal depth assumption was applied to the downstream boundary, as no rating curve was available for flood flows. Model computations used a 10 second time step.

Models were iteratively calibrated by comparing observed and simulated WSE over the course of a flood selected pre- and post-restoration at multiple locations. Two other flood events, one for each restoration configuration, were used for model testing. Aerial photographs acquired during floods also were used for visual calibration of flooding extent. Calibrated parameters were allowed to differ between the two configurations. The primary model parameter adjusted in the calibration process was surface roughness (Manning's *n*), both in the channel as well as within the floodplain (and spatially varying). Other parameters used in model calibration included the downstream friction slope and weir coefficients. Model performance was evaluated using common measures, principally NSE, PBIAS, and RSR (Moriasi et al., 2007). The model performed well at the floodplain restoration site of focus, with NSE >0.7 in-channel, and averaging 0.58 and 0.85 pre- and post-restoration, respectively, for within-floodplain sites (see Appendix A). Though velocity data useful for calibration were unavailable, the modeling produced velocity values generally under 1 m/s which yielded the satisfactory results in WSE and extent. While additional efforts may improve the models, primary limitations are likely related to data availability. The interaction of in-channel flow and floodplain topography as represented in floodplain inundation extent, depth, and velocity were deemed sufficient for the purposes of this research, which is most concerned with relative differences between the pre- and post-restoration scenarios. The focus is on new methods to quantify floodplain inundation patterns based on hydrodynamic model results, as opposed to the model process and utility. Appendix A contains further discussion of the development and calibration of the pre- and post-restoration hydrodynamic models used here.

Unsteady flow simulations for pre- and post-restoration models were done for a stepped hydrograph that spanned from below floodplain inundation flows (10 m^3/s) to the record peak daily flow (1,745 m^3/s). Each step consisted of constant flow for three days, which provided adequate time for changing inflows to move through the floodplain. Conditions on the floodplain at the end of each of these steps represented the in-channel flow. The magnitude and intervals between each step were selected for use in setting spatially resolved flow-depth and flow-velocity relationships. At lower flows, where floodplain inundation was very sensitive to flow, increments were smaller (e.g., every 10-20 m^3/s). At high flows, increments were larger (e.g., every 100 m^3/s). The stepped hydrograph procedure captured the ponding that occurs with the recession of a hydrograph, which would not have been possible with steady-state modeling. Output consisted of gridded (1 m resolution) depth and velocity for pre- and post-restoration configurations at known flows on both the rising and falling limb of a hydrograph.

Metric selection

Metrics were selected for their capacity to quantify a range of spatial and temporal floodplain inundation conditions based on daily spatially-resolved depth and velocity data derived from hydrodynamic modeling. A total of 47 individual metrics were initially computed in the analysis (Table 4-1). Together, the metrics describe a range of physical characteristics representing floodplain environmental conditions and processes relevant to species life history requirements and, more broadly, ecosystem structure and function (Table 4-2). The use of multiple physically-based metrics relating to ecologically important hydrologic conditions is common in the ecohydrological and environmental flows literature (e.g., Clausen & Biggs, 2000; Kennard et al., 2010; Olden & Poff, 2003; Poff, 1996; Richter et al., 1996). Of these approaches, The Nature Conservancy's Indicators of Hydrologic Alteration (IHA) and Range of Variability (RVA) approach is the most common in practice (Richter et al., 1996; The Nature Conservancy, 2009). A key distinction between such applications and the research here is that this study examines metrics for a spatially-resolved time series instead of simply a streamflow time series. Studies that address the spatial component of flows typically evaluate individual metrics as a function of flow, in line with IFIM and PHABSIM methods (Bovee, 1982; Stalnaker, 1979) and more sophisticated efforts toward quantifying habitat area (e.g., Erwin et al., 2017; Tomsic et al., 2007). Spatially-resolved metrics, or metrics that require evaluation of spatially-resolved data though time, are less frequently used. In line with this research, some recent studies are using more complex spatiotemporal inundation metrics (e.g., Cienciala & Pasternack, 2017; Coleman et al., 2015; Stone et al., 2017).

Metrics were developed for each of seven metric groups relating to flow regime components of magnitude (divided into extent, depth, and velocity groups; $n = 14$), timing ($n = 4$), duration ($n = 2$), rate of change ($n = 4$), and frequency ($n = 4$), as well as two additional groups capturing spatial aspects of floodplain inundation, connectivity ($n = 6$) and spatial heterogeneity ($n = 15$). Different metrics address the multiple dimensions over which conditions can be evaluated, namely 1) integrated across space and time as a single tabular value, 2) temporally integrated but spatially resolved (geographic), and 3) spatially integrated but temporally resolved (time series). This cross-tabulation means that metrics are similar, but different in their dimension of analysis (e.g., spatially-resolved duration and spatial mean duration). The following paragraphs briefly describe the metrics within their groups. Key aspects of ecological relevance are outlined in the following paragraphs, but more comprehensive assessments are cited in Table 4-2.

EXTENT, DEPTH, AND VELOCITY

Extent of inundation is a primary descriptor of floodplain inundation and a fundamental determinant of the quantity of available aquatic floodplain habitat. Flood extent is also useful for indicating the potential for vegetation change, erosion and deposition, and groundwater recharge. Eight extent metrics were computed, including spatially and temporally integrated metrics of maximum area (*MaxA*), maximum percent of total area inundated (*PMaxA*), mean area (*MA*), maximum volume

Table 4-1. Metrics used for hydrospatial analysis, classified by metric group, related to flow regime components, and identified as metrics that are spatially and temporally integrated, spatially resolved, and temporally resolved.

Metric Group	Flow Regime Components	Summary Type	Metric	Description
Extent	Magnitude	Space and time integrated	MaxA	Area maximum
			PMaxA	Maximum percent of total area inundated
			MA	Area mean
			MaxVol	Volume maximum
	Magnitude; Duration		ADay	Area-days
	Magnitude	Temporally resolved	A	Area
			PA	Percent area
			Vol	Volume
Depth	Magnitude	Space and time integrated	MaxDm	Spatial mean of max depth
		Temporally resolved	Dm	Spatial mean depth
		Spatially resolved	MaxD	Max depth
Velocity		Space and time integrated	MVm	Spatial mean of mean velocity
		Temporally resolved	Vm	Spatial mean velocity
		Spatially resolved	MV	Mean velocity
Duration	Duration	Space and time integrated	Durm	Duration mean
		Spatially resolved	Dur	Duration
Timing	Timing	Space and time integrated	WYD-MaxA	Water year day of max area
			WYD-MaxVol	Water year day of max volume
			WYDCn-Vol	Water year day of centroid volume
Rate of change	Rate of change	Space and time integrated	MaxRRm	Mean of max rising rate
			MaxFRm	Mean of max falling rate
		Spatially resolved	MRR	Max rising rate
			MFR	Max falling rate
Frequency	Frequency	Space and time integrated	IFNm	Mean number of times inundated
		Spatially resolved	IFN	Number of times inundated
Frequency; Duration	Frequency; Duration	Space and time integrated	IFPm	Mean percent of time inundated
		Spatially resolved	IFP	Percent of time inundated
Connectivity	Variability; Magnitude	Space and time integrated	CADay	Connected area-days
			DCADay	Disconnected area-days
		Temporally resolved	CA	Connected area
			DCA	Disconnected area
	Variability; Duration	Spatially resolved	C	Percent of time connected
			DC	Percent of time disconnected
Heterogeneity	Variability	Space and time integrated	MPN	Mean patch number
			MPSm	Mean patch size

Metric Group *(cont.)*	Flow Regime Components	Summary Type	Metric	Description
Heterogeneity *(cont.)*	Variability *(cont.)*	Space and time integrated *(cont.)*	MPSCV	Patch size coefficient of variation
			MIMaxD	Moran's I of max depth
			MIDur	Moran's I of duration
			MaxDCV	Max depth spatial coefficient of variation
			MVCV	Mean velocity spatial coefficient of variation
			DurCV	Duration spatial coefficient of variation
			MTE	Mean total edge length
			TPEm	Patch edge length mean
		Temporally resolved	LPS	Area of largest patch
			PSm	Mean patch size over time
			PN	Patch number
			TE	Total edge length

(*MaxVol*), and area-days (*ADay*), and temporally resolved metrics of daily inundated area (*A*), percent area (*PA*), and volume (*Vol*). The *ADay* metric is unique in that it combines both extent and duration in a single value representing the sum of inundated area over a period of time. This metric has been used by others to indicate available floodplain habitat and restoration potential (e.g., Coleman et al., 2015; Hermoso et al., 2012). The use of multiple descriptors can be helpful, however, as the same *ADay* value can have large areas of short durations and small areas with long durations. For depth and velocity, spatially and temporally integrated metrics of spatial mean of maximum depth (*MaxDm*) and spatial mean of mean velocity (*MVm*) were used. Spatial mean depth and velocity (*Dm*, *Vm*) are two temporally resolved metrics, and maximum depth (*MaxD*) and mean velocity (*MV*) are the related two spatially resolved metrics. Depth and velocity are the most common physical measures used to quantify habitat suitability, in PHABSIM and similar studies (Erwin et al., 2017; Jung & Choi, 2015; Matella & Merenlender, 2015; Tomsic et al., 2007). Not only do species require particular depths and velocities for survival (e.g, minimum depths for fish passage, depth preference for foraging by juvenile fish, maximum swimmable velocities), but most aquatic ecosystem processes are affected as well. These processes include primary and secondary production, vegetation community composition, aspects of water quality, such as temperature, and sediment deposition/erosion.

DURATION, TIMING, RATE OF CHANGE, AND FREQUENCY

Duration of inundation varies spatially across the floodplain and was characterized by a spatially and temporally integrated metric, spatial mean duration of inundated areas (*Durm*), and a spatially resolved duration (*Dur*) metric. Inundation duration is an indicator of ecological functions of floodplains, including primary and secondary productivity (Ahearn et al., 2006; Grosholz & Gallo, 2006) and incubation and rearing of juvenile fish (Jeffres et al., 2008; Moyle et al., 2004). Timing of inundation is also critical for these functions. For example, fish may not be present early in the season and warmer temperatures of later spring flooding enhances productivity. Timing metrics were based on water year day (beginning on October 1) and included water year day of maximum inundated area (*WYDMaxA*),

Table 4-2. Different ecohydrogeomorphic functions of riverine landscapes related to metric groups, with selected references that discuss aspects of these relationships.

Ecohydrogeomorphic processes and functions	Metric group(s) involved	Selected references
Habitat availability for freshwater-dependent species	Extent; Depth; Velocity; Duration; Timing; Patch characteristics	Bayley (1991); Feyrer et al. (2006); Gasith and Resh (1999); Simões et al. (2013); Sommer et al. (1997); Ward and Stanford (1995); Welcomme (1979)
Habitat predictability for freshwater-dependent species	Timing; Frequency of events; Frequency of inundation	Biggs et al. (2005); Lytle and Poff (2004); Poff et al. (1997)
Habitat accessibility for freshwater-dependent species	Connectivity	Amoros and Bornette (2002); Tockner et al. (2000)
Diversity of available habitat conditions	Depth; Velocity; Connectivity; Patch characteristics; Spatial heterogeneity	Bellmore et al. (2012); Bornette et al. (1998); Jeffres et al. (2008); Naiman and Décamps (1997); Opperman et al. (2010); Tockner et al. (2000); Ward et al. (2002b); Ward et al. (1999)
Species reproductive cues	Timing; Rate of change	Bailly et al. (2008); Lytle and Poff (2004); Poff et al. (1997); Stella et al. (2006); Yarnell et al. (2010)
Fish growth and mortality/ yield	Extent; Duration; Spatial heterogeneity	Agostinho et al. (2004); Bailly et al. (2008); Gorski et al. (2011); Jeffres et al. (2008); Moyle et al. (2004); Sommer et al. (1997); Welcomme (1979)
Stranding potential	Rate of change; Connectivity	Jeffres et al. (2008); Yarnell et al. (2010)
Hydrochorus seed dispersal	Extent; Timing	Nilsson et al. (2010)
Seed release, germination, and seedling survival	Extent; Duration; Timing; Rate of change	Stella et al. (2006); Stromberg et al. (2005); Ward and Stanford (1995)
Riparian forest succession	Extent; Velocity; Duration; Rate of change; Frequency of inundation; Spatial heterogeneity	Mahoney and Rood (1998); Richter and Richter (2000); Rood et al. (2003); Ward and Stanford (1995)
Vegetation community composition	Extent; Depth; Duration; Frequency of inundation	Auble et al. (1994); Mahoney and Rood (1998); Poff et al. (1997); Richter and Richter (2000); Rood et al. (2003)
Vegetation community distribution	Extent; Patch characteristics; Spatial heterogeneity	Dahm et al. (1995)
Soil moisture availability for riparian vegetation	Extent; Depth; Duration; Frequency of inundation	Mahoney and Rood (1998)
Primary and secondary productivity	Extent; Duration; Timing; Frequency of events; Frequency of inundation; Connectivity	Ahearn et al. (2006); Amoros and Bornette (2002); Gallardo et al. (2009); Grosholz and Gallo (2006); Junk et al. (1989); Opperman et al. (2010); Robertson et al. (2001); Sommer et al. (2004); Zilli (2013)

Ecohydrogeomorphic processes and functions *(cont.)*	**Metric group(s) involved**	**Selected references**
nutrient exchange	Extent; Duration; Frequency of events; Frequency of inundation; Connectivity	Junk et al. (1989); Poff et al. (1997); Sparks (1995); Tockner et al. (2000); Ward and Stanford (1995); Welcomme (1979)
aerobic/anaerobic conditions and decomposition rates	Extent; Duration; Timing; Frequency of inundation	Dahm et al. (1995); Tockner et al. (2000)
water quality (e.g., temperature, turbidity)	Depth; Velocity; Duration; Timing; Connectivity	Ahearn et al. (2006); Dahm et al. (1995); Tockner et al. (2000)
sediment erosion/deposition	Velocity; Duration; Frequency of inundation	Florsheim and Mount (2002); Opperman et al. (2010); Richter and Richter (2000)
channel avulsion	Velocity; Frequency of inundation	Florsheim and Mount (2002); Opperman et al. (2010)
groundwater infiltration	Extent; Depth; Duration; Frequency of inundation	Stromberg et al. (1996)

water year day of maximum volume (*WYDMaxVol*), and water year day of volume centroid (*WYDCnVol*). The volume centroid is defined as the point at which half of the total volume has occupied the floodplain. Rate of change metrics centered on changes in depth, using spatially and temporally integrated metrics of spatial mean of maximum daily depth increases on the rising limb of the hydrograph (*MaxRRm*) and decreases on the falling limb of the hydrograph (*MaxFRm*). The related spatially resolved metrics used were maximum daily depth increases on the rising limb of the hydrograph (*MaxRR*) and decreases on the falling limb of the hydrograph (*MaxFR*). These metrics provide a sense of how quickly conditions change and how responsive different areas of a floodplain might be to flow changes in the channel. Rate of change has particular implications for species stranding and seedling germination and survival (Rood et al., 2005; Yarnell et al., 2010). Metrics of inundation frequency addressed two separate but similar aspects: the number of times inundated and the percent of time inundated. The metrics include spatially and temporally integrated (*IFNm*, *IFPm*) and spatially resolved (*IFN*, *IFP*) forms. Inundation frequency and its variability affect the predictability of habitat for freshwater-dependent species. Frequency influences ecological processes such as nutrient exchange, export of primary and secondary production, and vegetation communities and successional processes (Ahearn et al., 2006; Opperman et al., 2010; Poff et al., 1997).

CONNECTIVITY

Hydrologic connectivity is a fundamental driver of river-floodplain ecosystems. It is highly variable within floodplain environments, and is a driver of overall floodplain spatiotemporal variability and complexity (Amoros & Bornette, 2002). Connectivity is required for species to access floodplain habitat, and different degrees of connectivity affect species assemblages (Tockner et al., 2000), with important implications for species diversity. Connectivity, related to residence time, also affects productivity and biogeochemical processing (Grosholz & Gallo, 2006). Highly connected floodplain patches will likely be more similar to the river than more disconnected areas (Hudson et al., 2012). The connectivity metrics applied here address whether areas are associated with surface water connections to the river channel (basically whether an area is ponded or not). Both connected and disconnected areas were quantified: 1) as connected or disconnected area-days (*CADay*, *DCADay*) in a spatially and temporally integrated form, 2) as connected or disconnected area (*CA*, *DCA*) over time, and 3) percent of time connected or disconnected (*C*, *DC*) over space.

HETEROGENEITY

Landscape ecology variables used to describe complex landscapes informed the selection of heterogeneity metrics, with metrics relating to the characterization of inundated patches, spatial autocorrelation, spatial variability, and edge. Landscape ecology theory has been advocated for its capacity to describe riverine landscapes (e.g., Carbonneau et al., 2012; Fausch et al., 2002; Wiens, 2002). While a wide range of landscape ecology metrics exist in the literature, some question the capacity to link these metrics to ecological processes and functions. However, spatial heterogeneity and how it changes over time affects ecosystem processes, species assemblages, and population dynamics (Ward et al., 2002a). Patch dynamics, spatial autocorrelation, and edge characteristics have all been used to assess spatial aspects of riverine landscape heterogeneity and its ecological implications (Bowen et al., 2003; Cooper et al., 1997; Datry et al., 2016; Le Pichon et al., 2009). For example, patch size and density indicates landscape fragmentation, and ability of species to interact and utilize inundated areas. Spatial autocorrelation, the tendency for places geographically closer together to be more similar than those farther apart, of landscape features or environmental variables also affects the distribution of organisms and, consequently, biotic interactions. The amount of edge or ecotonal conditions within a landscape (e.g, patch metrics) can indicate greater complexity in environmental conditions, which may, for example, affect the balance of abiotic and biotic controls in ecosystem structure and function. Species and processes are differentially affected by edge conditions. The metrics applied here were selected for their capacity to evaluate patterns of inundation, their ecological relevance, as well as practical considerations such as computation time. To quantify patch dynamics, spatially and temporally integrated metrics of mean patch number (*MPN*), mean of mean patch size (*MPSm*), and the coefficient of variation of mean patch size (*MPScv*) were used. Temporally resolved metrics included area of largest patch (*LPS*), mean patch size (*PSm*), and number of patches (*PN*). For spatial autocorrelation, the common global Moran's I was used (Cooper et al., 1997; Legendre & Legendre, 2012), applied at a resolution of 9 m to maximum depth (*MIMaxD*) and duration (*MIDur*). More involved analysis to determine appropriate spatial autocorrelation measures (e.g., semivariogram analysis, Moran's I at a range of spatial lags, or local Moran's I) was found to be too computationally intensive. As overall measures of spatial variability, spatial coefficients of variation were evaluated for maximum depth (*MaxDcv*), mean velocity (*MVcv*) and duration (*Durcv*). Edge metrics included two spatially and temporally integrated metrics, mean total edge length (*MTE*) and mean of patch edge length (*TPEm*), and one temporally resolved metric, total edge length (*TE*).

Hydrospatial analysis

Calculation of the spatiotemporal floodplain inundation metrics used spatially-distributed (i.e., gridded, raster formatted) daily estimates of depth and velocity (1 m resolution) at known flows on hydrograph rising and falling limbs generated by hydrodynamic modeling. Raster data for each flood day in the 110-year time series were created by spatially-distributed (at-a-cell) piecewise linear interpolation using the modeling output. For days on the hydrograph falling limb, a raster representing spatially-resolved inundation thresholds was required to determine whether a ponded cell should be wet or not. This was needed because the ponding produced in the hydrodynamic model was a result of record high flood flows. For smaller peak floods, less area would be ponded and the inundation threshold raster provided an approximation. For each day, only cells with inundation thresholds below the peak flow within the last seven days were allowed to be wet. It was assumed that after seven days, ponded area would no longer reflect peak flows due to infiltration and evaporation. This was informed by field observations, though actual conditions vary substantially with flood magnitude, depth, timing and other local factors. Time series of spatially-resolved depth and velocity rasters were generated for both pre- and post-restoration scenarios. The general procedure to use 2D hydrodynamic modeling at known flows to determine spatially-resolved values for a flow time series is similar to the recent study of Stone et al. (2017). To improve computational speed, the spatial resolution was resampled from 1 m to 3 m, which was visually assessed to confirm that spatial detail was still adequately captured. All processing was performed in *R* (R Core Team, 2016) using the *raster* package (Hijmans, 2015).

Hydrospatial analysis is defined by four-dimensional data, the planar surface, time, and the metric in question (e.g., depth). Numerous possibilities exist for summarizing conditions in space and time, both statistically and visually. Raster grid processing within *R* was conducted one water year at a time (to limit the number of rasters processed at a time), with most metrics computed at the flood event scale. As an example, to compute duration metrics, the number of days each grid cell was inundated for a flood event was stored in a new raster representing that event, which was analyzed along with the other rasters representing other floods within the same water year. From the event-based rasters, spatial mean duration for each event was calculated. The hydrospatial analysis output consisted of 1) a data table of computed spatially and temporally integrated metrics for each flood event for the entire time series, 2) a data table with computed temporally resolved metrics for each day of flow for the entire time series examined, and 3) rasters representing the spatially resolved metrics for each flood event. Given large datasets, the multi-core processing capability within the *raster* package was utilized to reduce computation time. These three types of output – spatially and temporally integrated, temporally resolved, and spatially resolved results – were then used to statistically summarize metrics (e.g., mean, coefficient of variation), summarize metrics at the event and water year scale, and develop graphical (e.g., exceedance probability, time series) and geospatial summaries of the metrics.

Hydrospatial analysis was done for both pre- and post-restoration conditions and subsequently compared statistically, graphically, and geospatially. For spatially and temporally integrated metrics, water year statistics of mean, median, and coefficients of dispersion (CD) were compared for the two conditions, including the use of a deviation factor (DF), following IHA methods (Caruso, 2013; The Nature Conservancy, 2009). The CD was calculated as the difference between the 75th and 25th quantile divided by the median and the DF as the difference between post- and pre-restoration divided by pre-restoration. To examine statistical significance of differences (for spatially and temporally integrated metrics), the non-parametric Wilcoxon signed-rank test was applied, with the null hypothesis of no difference between the paired conditions (Hollander et al., 2013). Testing statistical significance here is used to indicate whether the magnitude of difference was greater than the variance within the underlying data, as the direction is informative on its own. Relationships also were explored between hydrospatial metrics and water year types (annual daily flow volume quantiles of very wet (0.8-1), wet (0.6-0.8), normal (0.4-0.6), dry (0.2-0.4) and critically dry (0-0.2)) as well as flood types established by Whipple et al. (2017). Comparisons were supported by pairwise Wilcoxon rank sum tests.

Identification of priority metrics

Upon completing analysis for all metrics, principal component analysis (PCA) was used to determine which hydrospatial metrics were most useful in describing the primary sources of variance in the data, reducing the number of metrics to minimize redundancy while still describing the main sources of variation (Olden & Poff, 2003). Due to complexity in applying PCA to spatially or temporally resolved data, analysis was limited to the set of spatially and temporally integrated metrics (i.e., summarizing conditions across the floodplain and at the flood event or water year temporal scale; see Table 4-1). A total of 24 metrics were examined for multicollinearity, with three metrics (*MaxA*, *CADay*, and *WYDMaxVol*) eliminated prior to the analysis based on high correlations ($r > 0.9$) with other metrics in the same metric group. Analysis was done in *R* using the *prcomp* function (R Core Team, 2016). Given that metrics were calculated at event and water year scales, each were subjected to PCA separately, though a common set of priority metrics was sought (i.e., metrics were prioritized for their joint utility in explaining variance in both datasets). Both the pre- and post-restoration values were included in the analysis together because a single set of metrics best describing both conditions was desired. As a correlation matrix analysis, all data were centered and scaled so metrics had a common variance and contributed equally to the PCA. The broken-stick rule was used to evaluate the statistical significance of the PCs (Jackson, 1993). PCA was implemented in *R* (R Core Team, 2016) through the *BiodiversityR* package (Kindt, 2007).

Priority metrics for use in hydrospatial analysis were identified by selecting metrics with highest eigenvector loadings contributing to the PCs explaining most of the internal variance. The number selected for each PC was roughly in proportion to the proportion of variance explained by the PC. In addition, a criterion that at least one metric from each metric grouping be included was applied (*ADay* was considered as an extent as well as duration metric), by reasoning that each grouping describes ecologically meaningful aspects of floodplain inundation. This approach follows others that select PCA metrics for their ecological importance and capacity for interpretation (Chinnayakanahalli et al., 2011; Monk et al., 2007). This additional criterion meant that the PCA was used to identify the most useful metrics within each grouping for representing the data. For purposes of interpretation, metrics with higher loadings were also used to explain the dominant patterns of variation within the data. Results and discussion are focused on metrics selected from this process, but also report some results from other metrics for added comparison and interpretation. It should be noted that, depending on the application, other metrics may be of interest. As long as metrics can be calculated from daily spatial and temporally resolved depth and/or velocity estimates, the hydrospatial analysis approach presented here can be easily applied.

RESULTS

Selected priority metrics

Each of the metric groups – extent, depth, velocity, duration, timing, rate of change, frequency, connectivity, and heterogeneity (see Table 4-1) – were useful in explaining variance in the data, and some metrics in those groups were more useful than others. The PCA indicated redundancy across metrics, and within metric groups. The first two PCs were significant based on the broken-stick method for water-year-based summaries, while the first three PCs were significant for event-based summaries. The first four PCs explained 79% and 85% of variation among water years and events, respectively, formed by the set of 24 hydrospatial metrics. To consider variables contributing to the explanation of this greater percentage of total variation, metrics with high loadings for the first four PCs were examined.

The groupings of metrics with higher loadings on each of the primary PCs were similar between the water-year- and event-based summary datasets, but not identical. Loadings for the metrics in the first two PCs are all relatively low (<0.4), suggesting that no single metric has a very strong influence on these components. Metrics relating to extent, depth, velocity, duration, and rate of change had the highest loadings on the first PC. The priority metrics selected include percent maximum area (*PMaxA*), mean area (*MA*), area-days (*ADay*), mean of maximum depth (*MaxDm*), velocity mean (*MVm*), and mean falling rate (*MaxFRm*). As *ADay* is a composite metric representing both extent and duration, this metric was set a priority metric over *Durm*. Spatial heterogeneity metrics dominated the second PC, with mean of patch edge length (*TPEm*) and mean patch number (*MPNm*) selected as priority metrics. Frequency and connectivity metrics had the highest loadings for the third PC in the water year summary dataset, and the number of times inundated (*IFNm*) and disconnected area-days (*DCADay*) were selected as priority. Spatial heterogeneity characteristics dominated in the event-based summary dataset, and maximum depth coefficient of variation (*MaxDcv*) was selected as priority. Metrics related to timing clearly stood out with the highest loadings on the fourth PC for both datasets, and water year day of centroid volume (*WYDCnVol*) was selected. Summarizing the PCA results, Table 4-3 shows the selected priority metrics having the highest loadings across both datasets for each of the nine metric groups. Reporting and visualization of results focus on these selected priority metrics, though other metrics are also discussed.

Spatially and temporally integrated summaries

EXTENT OF INUNDATION

Floodplain inundation extent increased in most cases in response to the levee removal restoration action (Table 4-4; Figure 4-4). On average, the maximum area of inundation within the floodplain in a given year (*MaxA, PMaxA*) under pre-restoration conditions was 1.4 km^2 (66% of total area) and 1.5 km^2 (71% of total area) under post-restoration conditions. Water year cumulative area inundated (*ADay*) increased from an average of 32.1 km^2-day to 45.7 km^2-day. This overall increase in flood extent is primarily attributable to fewer events of minimal inundated area under post-restoration conditions. For flood events at the upper end of areal distribution, however, the maximum inundated area decreased in response to restoration. This result is reflected in the increase in mean water year summarized *PMaxA* (66% to 71%), but decrease in the median percentile (82% to 77%). All other extent metrics (those that are not just considering the maximum water year inundated area) were all consistently higher post-restoration, having deviation factors of >0.14 at the water year level and >0.80 at the event scale. The differences between pre- and post-restoration inundation extent metric were all significant at the event scale, according to the Wilcoxon signed-rank test ($p < 0.05$), but only significant for mean area (*MA*) and *ADay* metrics at the water year scale.

More detailed differences are found in the distributions of the metrics in Figure 4-5. For example, the range of 50-75% *PMaxA* increased the most over pre-restoration conditions. The distribution at the event level for this metric shifted toward a bimodal distribution in response to restoration. For *MA*, the distribution shift was more even (at both the water year and event scale), and toward higher values. The *ADay* metric at the event level is highly skewed due to a few very wet years, and differences in between pre- and post-restoration occur mostly in the lower range of the values. For water year summaries, exceedance probability plots indicate likelihood of different conditions (Figure 4-6). Of the selected extent metrics, *PMaxA* was found to be notably higher post-restoration in the range of about 60-85% exceedance probability. For example, 50% water year maximum inundated area was exceeded approximately 65%

Table 4-3. Loadings on the first four principal components (PCs) for the water year and event summaries of the 24 spatially and temporally integrated hydrospatial metrics. Loadings greater than the absolute value of 0.25 are in bold. The twelve priority metrics selected are noted with an asterisk.

Metric group	Metric	Water year				Event			
		PC1	PC2	PC3	PC4	PC1	PC2	PC3	PC4
Extent	PMaxA*	**0.25**	0.00	0.21	0.11	**-0.27**	0.09	0.03	0.13
	Aday*	**0.25**	-0.09	0.19	-0.09	-0.22	-0.22	-0.17	-0.23
	MA*	**0.27**	0.14	-0.08	0.00	**-0.26**	0.08	0.05	0.06
	MaxVol	**0.26**	-0.16	0.07	0.07	**-0.26**	-0.07	0.13	0.01
Depth	MaxDm*	**0.25**	-0.21	-0.18	-0.12	-0.23	-0.19	0.25	0.02
Velocity	MVm*	**0.25**	0.11	-0.18	-0.02	**-0.26**	0.07	0.18	0.15
Duration	Durm	0.22	-0.14	-0.04	-0.17	-0.18	**-0.28**	-0.23	-0.22
Timing	WYDMaxA	-0.05	0.16	0.09	**-0.63**	0.05	**-0.26**	**-0.25**	**0.50**
	WYDCnVol*	0.00	0.18	0.09	**-0.66**	0.04	-0.28	**-0.27**	**0.47**
Rate of change	MaxRRm	0.23	-0.14	-0.08	-0.08	-0.23	-0.10	0.04	0.00
	MaxFRm*	**-0.26**	0.02	0.21	0.06	0.24	0.00	-0.23	-0.04
Frequency	IFNm*	0.13	-0.09	**0.55**	0.05	-0.20	-0.20	-0.23	-0.23
	IFPm	0.19	-0.16	**0.45**	-0.01	-0.18	**-0.28**	-0.23	-0.22
Connectivity	DCADay*	0.23	-0.07	**0.30**	-0.01	-0.24	-0.13	**-0.25**	-0.13
Spatial heterogeneity	MPN*	0.22	**0.27**	0.02	0.08	-0.20	**0.29**	-0.01	0.12
	MPSm	0.15	-0.24	-0.21	0.02	-0.11	-0.19	**0.31**	0.00
	MPScv	0.21	-0.18	-0.08	0.06	-0.22	-0.08	0.17	-0.05
	MIMaxD	0.18	**0.30**	0.02	0.16	-0.19	**0.25**	-0.22	0.02
	MIDur	0.22	**0.27**	-0.06	0.05	-0.22	0.19	-0.17	0.09
	MaxDcv*	-0.12	**0.29**	0.19	0.21	0.13	0.23	**-0.33**	-0.16
	MVcv	-0.22	-0.15	0.11	0.03	0.24	-0.12	-0.03	-0.21
	Durcv	0.14	**0.31**	-0.17	0.03	-0.15	0.21	-0.21	0.14
	MTE	0.24	**0.27**	-0.05	0.03	-0.24	0.22	-0.04	0.08
	TPEm*	0.05	**-0.38**	-0.21	-0.04	0.11	**-0.33**	**0.26**	0.08

of the time under pre-restoration conditions, but approximately 84% of the time under post-restoration conditions. The concave shape means that values change slowly at lower exceedance values. For *MA*, post-restoration conditions were consistently higher, except for the lowest and highest exceedance probabilities. The shape of the *MA* curves are flatter, with a consistent slope across most of the probabilities, except for the extremes. The percentage shifts in *ADay* were consistent across the range of exceedance probabilities, and the shapes of both curves are convex, indicating larger changes at lower exceedance probabilities. For example, at 75% exceedance probability, *ADay* increased from about 7 to 10 km^2-day, while at 10% exceedance probability, *ADay* increased from about 76 to 108 km^2-day. Both represent about a 40% increase.

Variability across water years and across events for the extent metrics decreased overall with restoration, indicated by the coefficient of dispersion (CD), shown in Table 4-4. This can be caused by the decrease in extremes with restoration. Both the direction and magnitude of the deviation factors (DF) of

Table 4-4. Summary statistics for hydrospatial metrics at the water year scale (e.g., water year maximum daily inundated area) and at the flood event scale (e.g., maximum inundated area per flood event). The bootstrapped 95% confidence intervals (CI) for the median values are shown, as well as the coefficients of dispersion (CD; (75th%ile-25th%ile)/median). Deviation factors are computed for the medians and CDs of the pre- and post-restoration conditions. The Wilcoxon signed-rank test is also shown for purposes of comparing conditions, with those associated with a *p*-value < 0.05 bolded (where the null hypothesis of no significant difference is rejected).

Summary scale	Metric Group	Metric	Metric units	Pre-restoration				Post-restoration				Deviation factors		Wilcoxon signed-rank p-value
				Mean	Median	CI	CD	Mean	Median	CI	CD	Median	CD	
Water year	Extent	MaxA	km^2	1.38	1.71	(1.58-1.86)	0.74	1.48	1.61	(1.46-1.72)	0.44	-0.06	-0.40	0.262
		PMaxA	%	66.35	82.04	(75.99-89.34)	0.74	71.14	77.49	(70.25-82.68)	0.44	-0.06	-0.40	0.262
		MA	km^2	0.23	0.23	(0.21-0.26)	0.74	0.33	0.31	(0.27-0.35)	0.66	0.39	-0.12	<0.001
		MaxVol	m^3	1.48E+06	1.22E+06	(7.8E+05-1.6E+06)	1.86	1.51E+06	1.38E+06	(1.1E+06-1.7E+06)	1.38	0.14	-0.26	0.136
		ADay	km^2-day	32.06	24.29	(15.04-34.53)	1.77	45.74	34.53	(19.88-48.61)	1.67	0.42	-0.05	<0.001
	Depth	MaxDm	m	0.52	0.48	(0.42-0.5)	0.36	0.51	0.49	(0.46-0.52)	0.39	0.03	0.08	0.103
	Velocity	MVm	m/s	0.03	0.03	(0.02-0.03)	0.71	0.05	0.05	(0.04-0.05)	0.65	0.89	-0.09	<0.001
	Duration	Durm	day	7.37	6.47	(6.03-6.99)	0.54	8.14	6.28	(5.33-6.76)	0.66	-0.03	0.24	0.005
	Timing	WYDMaxA	wy day	133	138	(128-144)	0.34	133	137	(129-143)	0.33	0.00	-0.02	1.000
		WYDMaxVol	wy day	133	138	(128-143)	0.34	133	138	(128-144)	0.34	0.00	0.01	0.197
		WYDCnVol	wy day	140	146	(140-152)	0.24	143	149	(143-154)	0.24	0.02	-0.01	<0.001
	Rate of change	MaxRRm	m/day	0.24	0.20	(0.15-0.23)	0.96	0.19	0.17	(0.13-0.21)	1.28	-0.16	0.33	<0.001
		MaxFRm	m/day	-0.26	-0.23	(-0.26--0.2)	-0.64	-0.26	-0.24	(-0.26--0.22)	-0.45	0.04	-0.31	0.564
	Frequency	IFNm	count	6.17	6.29	(5.7-6.62)	0.63	6.42	6.47	(5.86-6.93)	0.72	0.03	0.15	<0.001
		IFPm	%	9.92	10.10	(8.91-11.53)	0.81	10.90	11.38	(9.77-14.47)	0.93	0.13	0.14	<0.001
	Connectivity	CADay	km^2-day	24.45	16.54	(8.49-23.89)	2.01	38.97	27.91	(15.18-40.18)	1.83	0.69	-0.09	<0.001
		DCADay	km^2-day	7.43	6.99	(5.37-9.22)	1.21	6.51	7.19	(6.62-9.24)	0.86	0.03	-0.29	0.001
	Heterogeneity	MPN	count	359	383	(350-409)	0.51	528	572	(543.78-607.23)	0.35	0.49	-0.31	<0.001
		MPSm	m^2	4.11E+03	9.89E+02	(9.0E+02-1.1E+03)	2.07	1.97E+03	1.20E+03	(1.1E+03-1.3E+03)	0.80	0.21	-0.62	0.071
		MPScv	%	255.21	91.05	(78.94-105.09)	4.50	206.62	110.27	(97.68-116.13)	2.49	0.21	-0.45	0.200
		MIMaxD	Moran's I	0.80	0.84	(0.83-0.86)	0.08	0.80	0.84	(0.83-0.86)	0.09	0.00	0.15	0.161
		MIDur	Moran's I	0.62	0.64	(0.6-0.66)	0.23	0.64	0.66	(0.62-0.67)	0.22	0.03	-0.07	<0.001
		MaxDcv	%	94.52	99.87	(96.81-104.65)	0.19	95.01	96.94	(92.62-98.92)	0.25	-0.03	0.32	0.506
		MVcv	%	153.89	159.94	(155.51-166.63)	0.17	120.91	120.75	(112.62-126.14)	0.35	-0.25	1.07	<0.001
		Durcv	%	62.40	64.73	(62.04-68.93)	0.31	63.04	64.22	(63.27-67.01)	0.23	-0.01	-0.25	0.440
		MTE	m	7.58E+03	7.65E+03	(7.17E+03-8.33E+03)	0.49	9.99E+03	1.02E+04	(9.6E+03-1.1E+04)	0.42	0.33	-0.15	<0.001
		TPEm	m	46.68	41.93	(39.49-43.51)	0.28	41.76	40.23	(35.84-43.04)	0.46	-0.04	0.63	0.021

Summary scale *(cont.)*	Metric Group	Metric	Metric units	Pre-restoration				Post-restoration				Deviation factors		Wilcoxon signed-rank p-value
				Mean	Median	CI	CD	Mean	Median	CI	CD	Median	CD	
Event	Extent	MaxA	km^2	0.63	0.26	(0.21-0.32)	3.64	0.81	0.65	(0.51-0.86)	1.78	1.52	-0.51	<0.001
		PMaxA	%	30.22	12.52	(9.94-15.42)	3.64	38.82	31.51	(24.46-41.5)	1.78	1.52	-0.51	<0.001
		MA	km^2	0.23	0.14	(0.11-0.15)	2.01	0.33	0.26	(0.24-0.29)	1.34	0.95	-0.33	<0.001
		MaxVol	m^3	5.04E+05	9.68E+04	(7.3E+04-1.2E+05)	4.56	5.84E+05	1.74E+05	(1.3E+05-2.2E+05)	4.24	0.80	-0.07	<0.001
		ADay	km^2-day	6.47	1.29	(0.92-1.55)	4.09	9.23	2.42	(1.93-2.89)	2.71	0.88	-0.34	<0.001
	Depth	MaxDm	m	0.52	0.41	(0.4-0.42)	0.52	0.52	0.47	(0.45-0.52)	0.69	0.14	0.31	0.015
	Velocity	MVm	m/s	0.03	0.02	(0.01-0.02)	1.44	0.05	0.03	(0.02-0.04)	2.61	0.87	0.82	<0.001
	Duration	Durm	day	7.31	5.56	(5.02-5.97)	0.66	8.03	5.29	(4.9-5.72)	0.80	-0.05	0.22	0.140
	Timing	WYDMaxA	wy day	139	137	(131-143)	0.54	139	137	(131-143)	0.54	0.00	0.00	0.533
		WYDMaxVol	wy day	139	137	(131-142)	0.53	139	137	(131-143)	0.54	0.00	0.01	<0.001
		WYDCnVol	wy day	142	140	(134-147)	0.54	141	140	(135-147)	0.54	0.00	0.00	<0.001
	Rate of change	MaxRRm	m/day	0.24	0.18	(0.15-0.21)	1.72	0.19	0.06	(0.03-0.07)	5.84	-0.67	2.39	<0.001
		MaxFRm	m/day	-0.26	-0.21	(-0.23--0.2)	-0.91	-0.26	-0.18	(-0.2--0.16)	-1.26	-0.11	0.39	0.826
	Frequency	IFNm	count	1.25	1.01	(1.01-1.01)	0.09	1.29	1.01	(1.01-1.01)	0.15	0.00	0.58	<0.001
		IFPm	%	0.02	0.02	(0.01-0.02)	0.66	0.02	0.01	(0.01-0.02)	0.80	-0.05	0.22	0.140
	Connectivity	CADay	km^2-day	4.94	0.74	(0.48-0.93)	4.33	7.86	1.49	(1.04-1.88)	3.53	1.00	-0.18	<0.001
		DCADay	km^2-day	1.50	0.49	(0.39-0.54)	3.95	1.31	0.88	(0.77-0.96)	1.58	0.81	-0.60	0.326
	Heterogeneity	MPN	count	332	174	(119-217)	2.71	502	576	(539-657)	1.02	2.30	-0.62	<0.001
		MPSm	m^2	2.48E+03	8.99E+02	(8.7E+02-9.5E+02)	0.49	1.58E+03	1.02E+03	(9.6E+02-1.1E+03)	0.80	0.13	0.64	0.094
		MPScv	%	93.93	57.39	(53.77-60.92)	0.88	103.90	85.86	(80.09-91.56)	0.80	0.50	-0.09	<0.001
		MIMaxD	Moran's I	0.83	0.86	(0.86-0.86)	0.09	0.83	0.87	(0.86-0.87)	0.08	0.01	-0.10	0.137
		MIDur	Moran's I	0.64	0.74	(0.73-0.76)	0.53	0.66	0.75	(0.74-0.77)	0.44	0.01	-0.17	<0.001
		MaxDcv	%	99.09	110.94	(109.7-112.71)	0.41	99.94	85.31	(74.51-90.24)	0.73	-0.23	0.76	0.035
		MVcv	%	160.15	166.22	(163.72-169.83)	0.26	126.81	106.09	(74.6-116.54)	0.99	-0.36	2.78	<0.001
		Durcv	%	63.79	63.77	(60.07-71.34)	0.72	63.68	61.78	(58.52-65.13)	0.72	-0.03	0.00	0.248
		MTE	m	7.76E+03	5.68E+03	(4.9E+03-6.2E+03)	1.38	1.03E+04	1.07E+04	(1.0E+04-1.1E+04)	0.73	0.88	-0.47	<0.001
		TPEm	m	45.81	43.05	(41.86-44.65)	0.49	47.54	32.70	(31.01-34.12)	1.12	-0.24	1.29	0.014

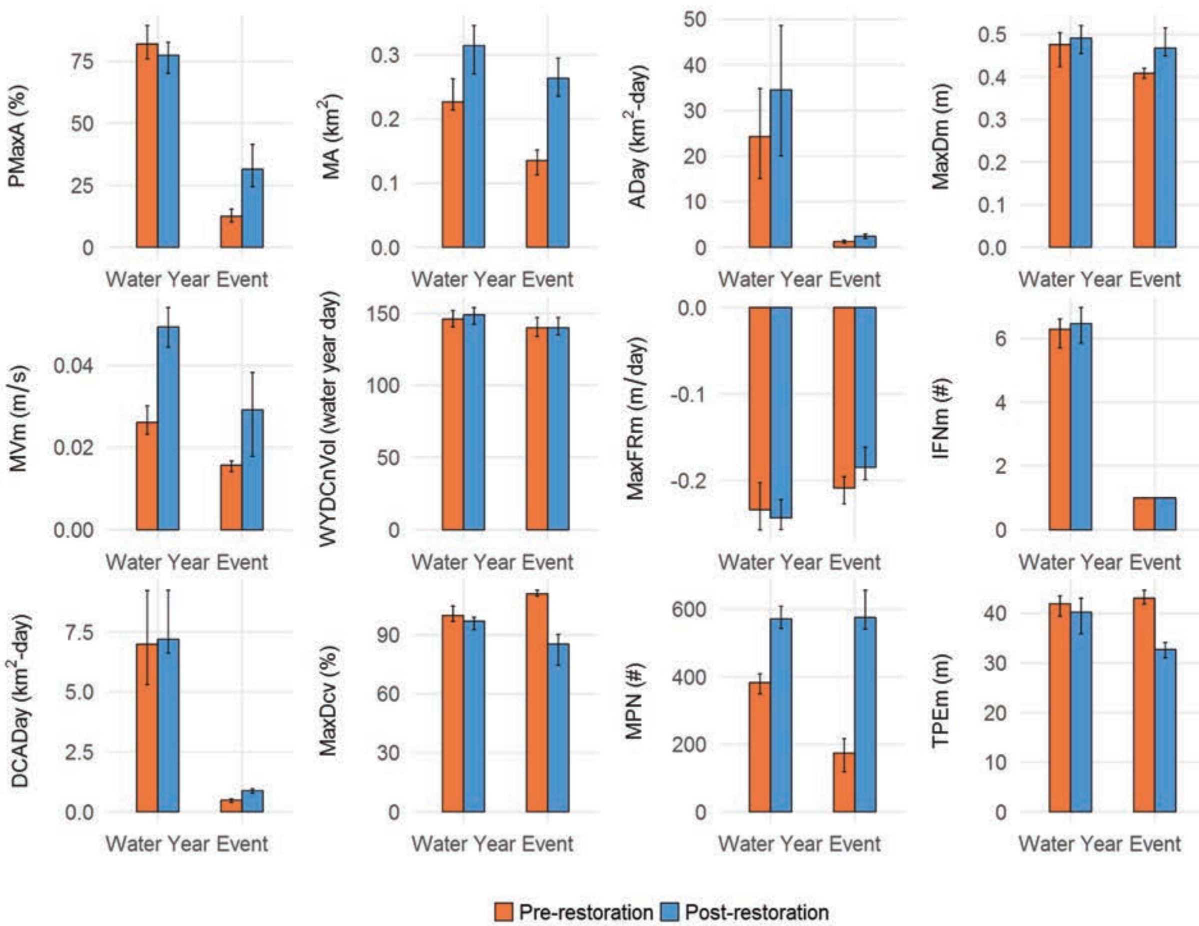

Figure 4-4. Median conditions for selected hydrospatial metrics that summarize conditions across both space and time. These show differences between pre- (orange) and post-restoration (blue) conditions as well as between metrics summarized at the flood event versus water year scale. Error bars represent the bootstrapped 95% confidence intervals for the median values shown.

CD were not particularly tied to those of median values. The range of DF values were within similar ranges for median and CD at the water year scale, but larger in absolute value for median over CD at the event scale. Variability was consistently greater across events than across water years.

DEPTH AND VELOCITY

The spatial mean of maximum depth (*MaxDm*) did not change substantially with restoration. On average, pre-restoration *MaxDm* was 0.52 m across water years and across events, and post-restoration *MaxDm* was just slightly lower at 0.51 m at the water year scale and 0.52 m at the event scale. The change in *MaxDm* was minimal (see Table 4-4, Figure 4-4), with median conditions increasing slightly by about 1 cm and 6 cm at the water year and event scale, respectively. In contrast, the spatial mean of mean velocity (*MVm*) did change significantly with restoration, from 0.03 to 0.05 m/s on average (at water year and event scales). The associated DFs were >0.85. The density distribution for *MVm* illustrates this shift and also shows the overall broader distribution of the metric (see Figure 4-5). Both *MaxDm* and *MVm* appear to develop more bimodal distributions post-restoration for event scale summaries. No discernable differences

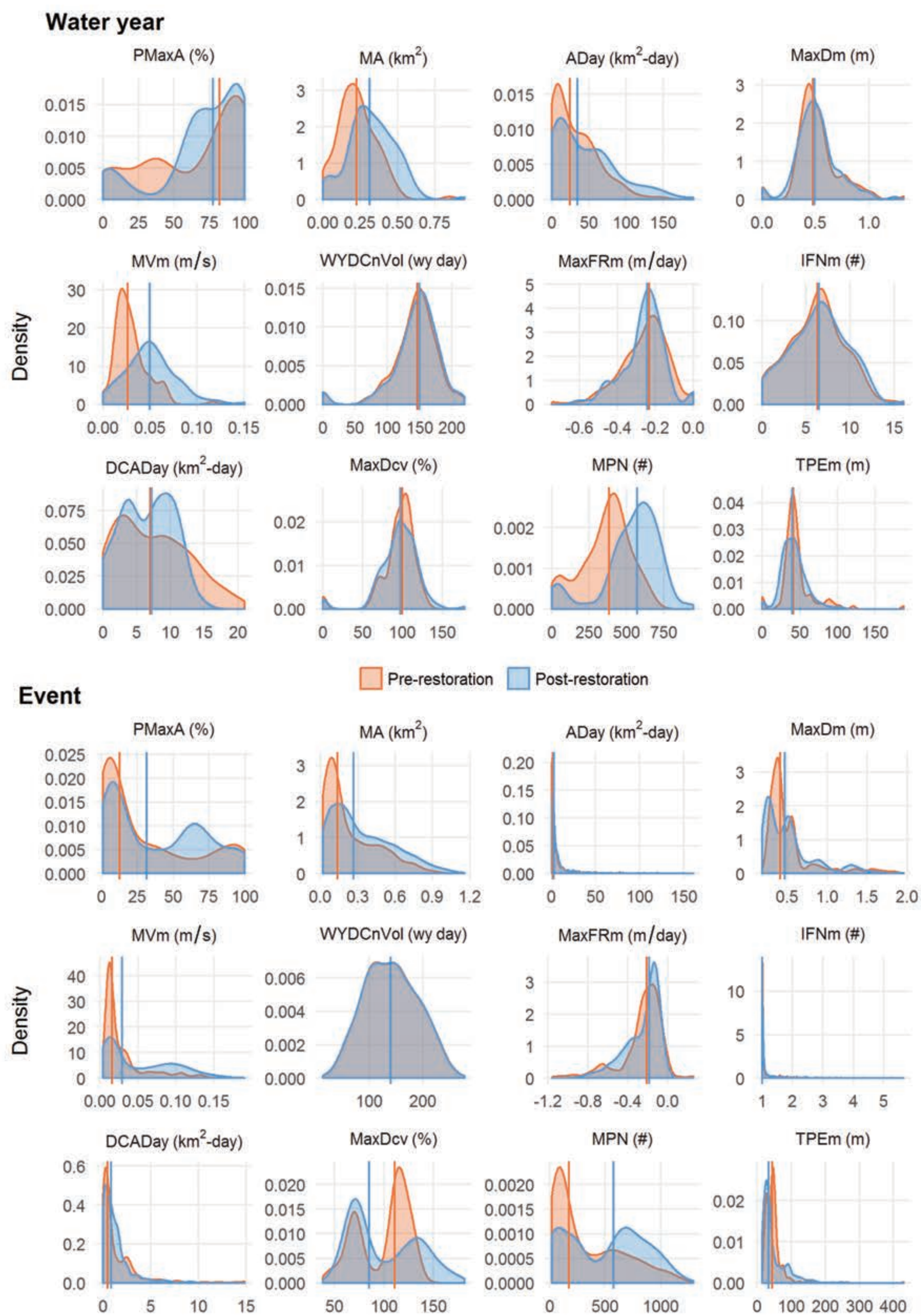

Figure 4-5. Density plots for each of the twelve selected hydrospatial metrics. The top sets of plots show metrics summarized at the water year scale, while the bottom show metrics summarized at the flood event scale. Vertical lines indicate median values. See Table 4-4 for x-axis units for each metric.

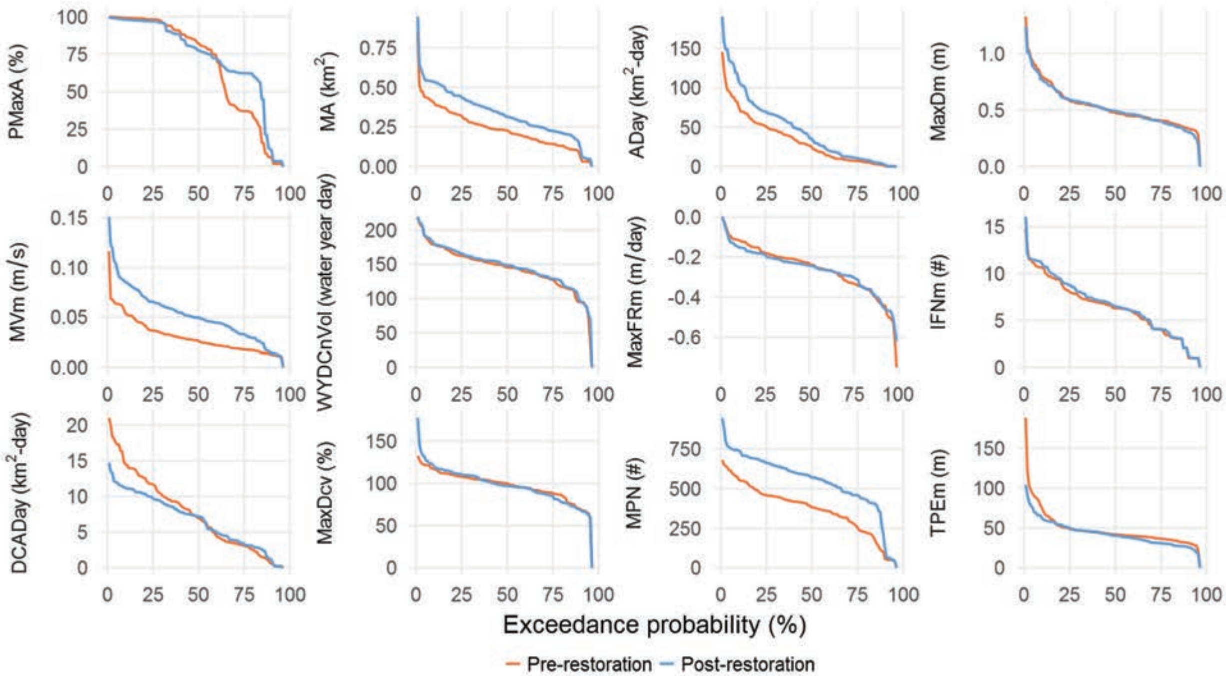

Figure 4-6. Empirical exceedance probability plots for each of the twelve selected hydrospatial metrics, summarized at the water year scale. These plots are similar to exceedance probability plots of water year maximum daily flow, or flow duration curve.

in exceedance probability are seen for *MaxDm* (see Figure 4-6). The shape of this curve is similar to a normal distribution. For *MVm*, 0.05 m/s is expected to be exceeded about 14% of the time under pre-restoration conditions and about 50% of the time under post-restoration conditions, for example. Variability in *MVm* also increased after restoration at the event scale, with a DF of 0.82, similar to that of median conditions. At the water year scale, *MVm* variability decreased slightly, with a DF of -0.09.

DURATION

Across water years, the mean and median of the spatial mean duration (*Durm*) increased and decreased, respectively, with restoration (see Table 4-4). On average, inundation duration was 7.4 days (median = 6.5 days) pre-restoration and 8.1 days (median = 6.3 days) post-restoration. Trends for summary at the event scale were similar. Deviation factors for median conditions were small (-0.03 and -0.05). The changes were significant at the water year scale, but not at the event scale. Variability in *Durm* increased post-restoration, with DFs of 0.24 and 0.22 for water year and event-scale summaries, respectively.

TIMING

Overall, maximum flooding occurred in mid- to late-February both pre- and post-restoration. The timing of maximum inundated area (*WYDMaxA*), maximum volume (*WYDMaxVol*), or centroid volume (*WYDCnVol*) did not change substantially in response to restoration, particularly at the event scale. However, significant increases in *WYDCnVol* at the water year scale and *WYDMaxVol* and *WYDCnVol* at the event scale were found (see Table 4-4, Figure 4-4). The *WYDCnVol* occurred three days later under post-restoration conditions at the water year scale, but were less than a day on average at the event scale.

Given that the timing of flows used to estimate pre- and post-restoration floodplain inundation patterns did not change for the two scenarios, little change would be expected. The shape of the exceedance probability graph shows that values change most rapidly at the highest exceedance probabilities (as the centroid volume rarely occurs early in the season; see Figure 4-6). Variability was greater across events than across years, and changed minimally with restoration.

RATE OF CHANGE

The rate of flow increases on the rising limb of the hydrograph (*MaxRRm*) decreased significantly with restoration, from an average of about 24 cm/day to about 19 cm/day (see Table 4-4). Minimal change was detected for the falling limb of the hydrograph (*MaxFRm*), except for median conditions at the event scale, which decreased from a falling rate of 21 cm/day to 18 cm/day. The distribution of the selected priority metric, *MaxFRm*, is left skewed, with similar distributions for the water year and event level summaries (see Figure 4-5). From the exceedance probability graph (see Figure 4-6), only small differences are discernable. At the lower exceedance probabilities, post-restoration conditions have higher falling rates (more negative), and at the high exceedance probabilities, post-restoration conditions have lower falling rates (less negative). Variability increased for both metrics post-restoration.

FREQUENCY

Analysis suggests that inundation occurred on average 6.2 times a year or a total of 10% of the time in a year under pre-restoration conditions, and on average 6.4 times a year or 11% of the year under post-restoration conditions (see Table 4-4, Figure 4-4). Within a flood event, inundation occurred on average about 1.3 times both pre- and post-restoration. The median percent time inundated at the event scale decreased slightly (at the event scale, mean percent of time (*IFPm*) is basically duration expressed as a percent of a year). Deviation factors were relatively low, but differences were significant, except for *IFPm* at the event scale. For distribution, the mean number of times inundated (*IFNm*) under pre-restoration conditions was more concentrated around the median than under post-restoration conditions (see Figure 4-5). Variability increased under post-restoration conditions for both metrics, with DFs greater than those for median conditions.

CONNECTIVITY

Inundated area connected to the river (CADay) increased with restoration, following the overall increase in *ADay* (see Table 4-4, Figure 4-4). Mean disconnected area (*DCADay*) declined, though the median increased. Most of the inundated area stayed connected, with an water year average of 24 km^2-day and 39 km^2-day pre- and post-restoration compared to disconnected area of 7.4 km^2-day and 6.5 km^2-day pre- and post-restoration. Median conditions were significantly different, except for *DCADay* summarized at the event level, and consistently lower than the mean (see Figure 4-5). The distribution for *DCADay* at the water year scale shows a substantial decrease in the upper extremes of water year *DCADay* and becomes bimodal under restoration conditions. This difference in trends at the lower and upper extremes is evidenced by the exceedance probability plot for *DCADay* (see Figure 4-6). At lower exceedance probabilities, there is less disconnected area post-restoration, but at the higher exceedance probabilities there is more. Exceedance probability slopes are fairly constant for this metric. Variability decreased post-restoration, indicating decreased extremes in connected and disconnected days.

HETEROGENEITY

A total of ten metrics related to spatial heterogeneity were summarized, two of which were selected as high priority (see Table 4-4; Figure 4-4). Mean patch number (*MPN*) increased under restoration conditions from about 360 to 530. Correspondingly, the length of water year mean total patch edge (*TPEm*) decreased from about 47 m to about 42 m. A distinctly bimodal distribution developed post-restoration for *MPN*, summarized at the event scale (see Figure 4-5). The distribution of *TPEm* has a pronounced right skew. In examining exceedance probabilities based on the data, the large increases in post-restoration *MPN* can be seen at all exceedance probabilities, except for the highest probabilities (see Figure 4-6). The slope is gradual, with the tails changing sharply. For *TPEm*, values are higher pre-restoration for most probabilities, except for probabilities in the range of 25-50%, where the lines converge. The slope is quite flat with values not changing rapidly for most exceedance probabilities, only changing sharply at the extremes. Variability decreased for *MPN*, but increased for *TPEm*, with the magnitude of DFs larger for median and CD, but the opposite for *TPEm*.

For the other metrics, several changes in response to restoration are of note. Along with *MPN*, several patch related metrics follow similar increases post-restoration, including *MPSm*, *MPScv*, and *MTE*. The two measures of spatial autocorrelation (*MIMaxD*, *MIDur*) slightly increased under post-restoration conditions, which can be interpreted as inundated locations within the floodplain were more likely to be adjacent to others in post-restoration conditions. Spatial variability, as represented by the coefficient of variation of values across the floodplain (*MaxDcv*, *MVcv*, *Durcv*), decreased post-restoration.

Hydrospatial conditions over time

EXTENT OF INUNDATION

In comparing differences in inundated area (*A*) over time, cumulative distributions for the period of record show stepped increases, associated with wetter periods (Figure 4-7a). Summarized over the water year, cumulative median inundation extent begins to increase after the beginning of January, rises over the months of February-April, and levels off in late April (Figure 4-7b). Overall, post-restoration cumulative inundated area was about 43% greater than pre-restoration conditions. Over the period of record, a total of 5,031 km^2 accumulated with the post-restoration configuration compared to 3,527 km^2 with the pre-restoration configuration. For summary at the scale of the water year, median cumulative inundated area was 34.5 km^2 post-restoration over 24.3 km^2 pre-restoration, and the 25th-75th percentile interval was also wider (by 35%) under post-restoration conditions.

Temporally resolved percent inundation area (*PA*) provides additional detail concerning when during the season increased inundation occurs. For both pre- and post-restoration, daily median *PA* values did not rise above zero until after the beginning of February and returned to zero in mid-May. The water year distribution shows median post-restoration values nearly twice that of pre-restoration conditions fairly consistently (generally around 5% and 8% pre- and post-restoration). However, the 75th percentile was highest for post-restoration in the spring months (Figure 4-8). Pronounced gains in inundated area later in the season are visible in Figure 4-9. The maximum difference in *PA* was a gain of approximately 30%. Relatedly, longer periods of larger inundated areas also appear as a result of the post-restoration configuration. Only for brief periods did inundated area decline with restoration; most days (~90%) gained inundated area.

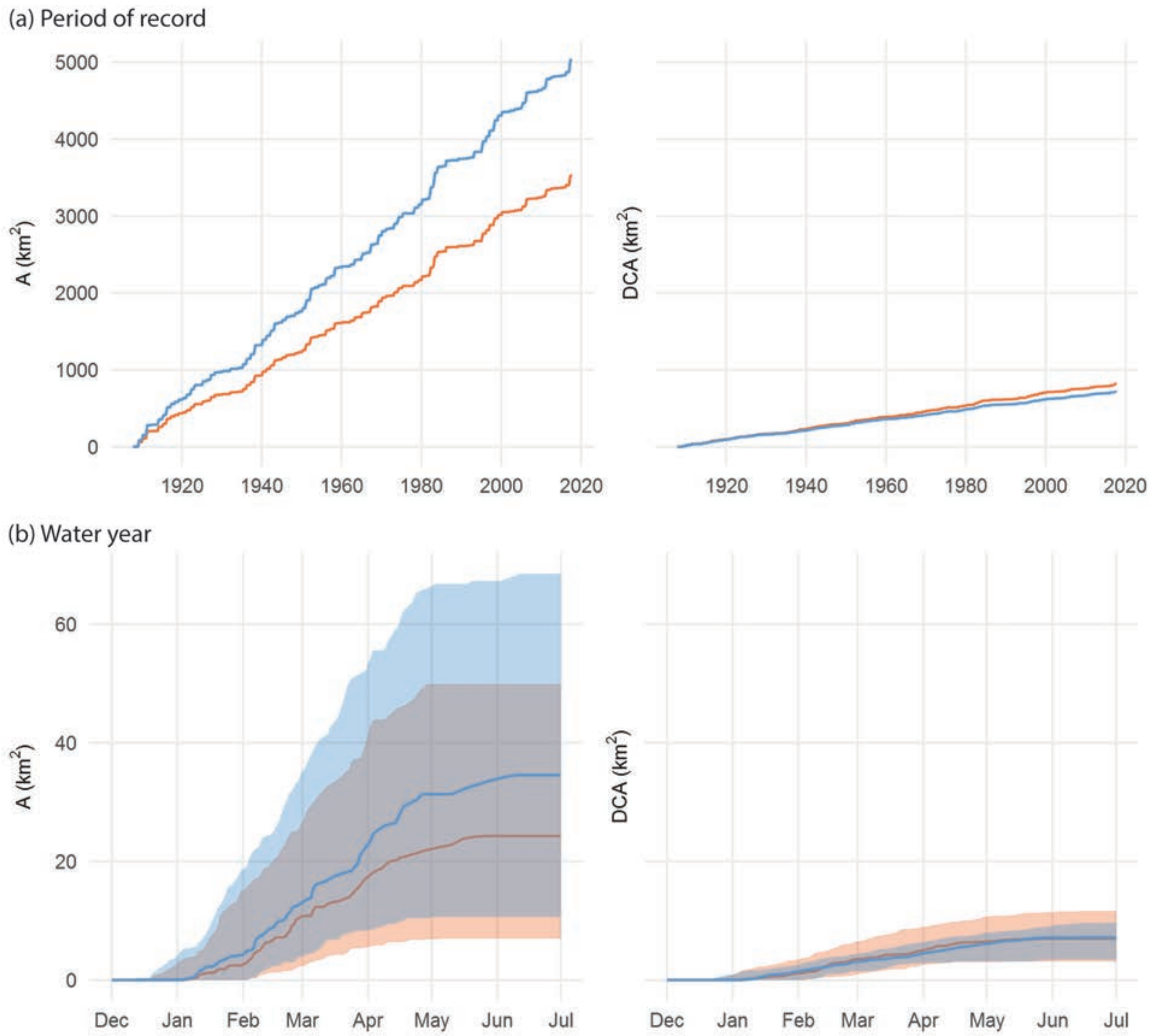

Figure 4-7. Cumulative distributions of pre- (orange) and post-restoration (blue) conditions for the period of record (a) and at the water year scale (b) for extent metrics of inundated area (*A*) and disconnected area (*DCA*). Shading represents the 25th to 75th percentile.

DEPTH AND VELOCITY

For the spatial mean of daily depths (*Dm*), the pre-restoration median and 75th percentile were consistently above post-restoration (see Figure 4-8). However, the daily time series illustrates the substantial variability in the differences in *Dm* (Figure 4-10). Figure 4-10 also illustrates the rarity of extreme depths. An important consideration in comparing the spatial mean values is that only inundated grid cells were included in the computation. So, for example, a day where a small area was inundated pre-restoration may cause deeper average depth than that of the same day under post-restoration conditions where a much larger area was inundated, but with shallower depths overall.

In comparing spatial mean of velocity (*Vm*), the water year distribution shows median conditions slightly lower, but the upper percentiles much higher post-restoration (see Figure 4-8). Over the period of the wet season, *Vm* does not vary greatly. Longer periods of higher velocities under post-restoration conditions are particularly apparent in Figure 4-11. Absolute daily differences reached maximums of around 0.12 m/s.

CONNECTIVITY

Of the total inundated area, area disconnected from the river (*DCA*; i.e., ponded) decreased about 12% in the post-restoration configuration, overall. This made up a relatively low fraction of the overall inundated area, about 23% and 14% for pre- and post-restoration, respectively. While median water year cumulative disconnected area was quite similar (7.0 km^2 versus 7.2 km^2 pre- and post-restoration), the 75th percentile was considerably higher for the pre-restoration configuration, suggesting greater extremes (see Figure 4-7).

The water year time series for disconnected area (*DCA*) shows relatively little difference between pre- and post-restoration conditions earlier in the season and increases post-restoration later in the season, similar to *PA* (see Figure 4-8). The 75th percentile peaks earlier in the season relative to overall inundated area. The greater extremes with pre-restoration conditions is particularly apparent in the daily time series in Figure 4-12. The seasonal changes in differences are also visible, with the preponderance of increased *DCA* post-restoration in the spring periods in contrast to mostly decreased *DCA* post-restoration toward the beginning of the season. As connected area (*CA*) basically tracks total inundated area (*A*; equal to *CA*+*DCA*), *CA* is not discussed.

HETEROGENEITY

The water year distributions for the number of patches within the floodplain (*PN*) show greater median and 75th percentile values under post-restoration conditions (see Figure 4-8). Median *PN* was about two times greater for most of the inundated period. The magnitude of difference varied particularly for the 75th percentile, which increased later in the season. Between the median and 75th percentile, *PN* more than doubled under both floodplain configurations. The extreme high values for *PN*

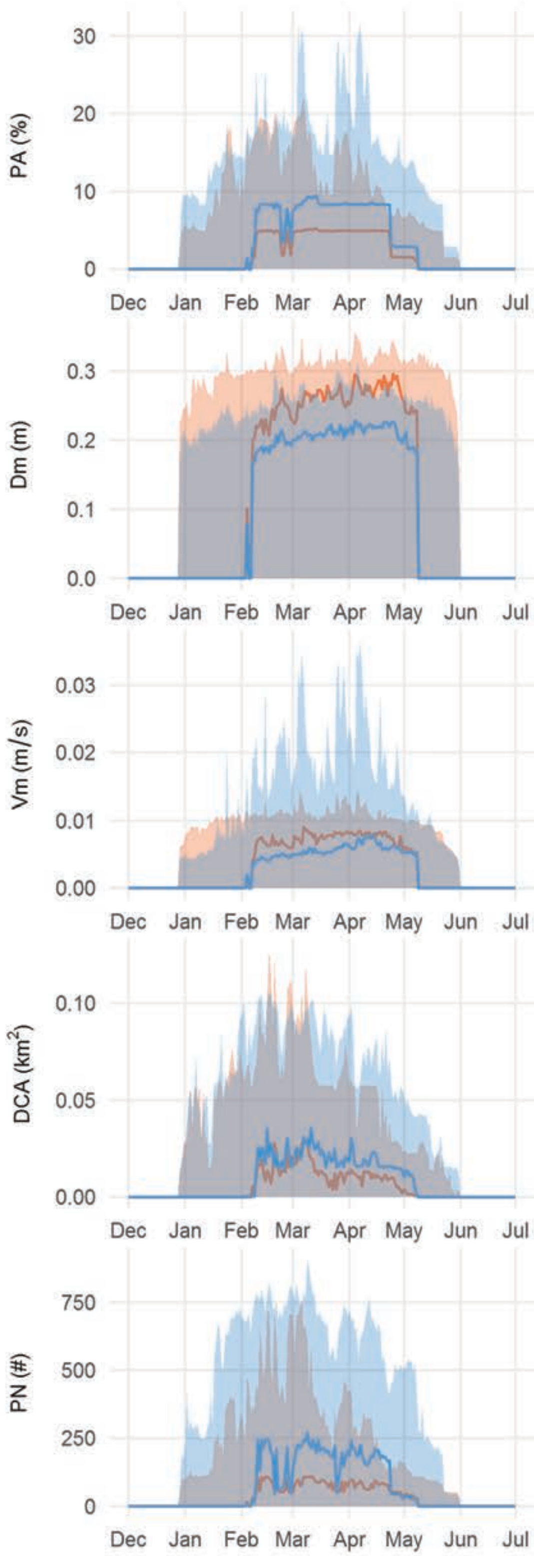

Figure 4-8. Annual distribution of daily median values for selected temporally resolved hydrospatial metrics to compare pre- (orange) and post-restoration (blue) conditions. Shading represents the 25th to 75th percentile.

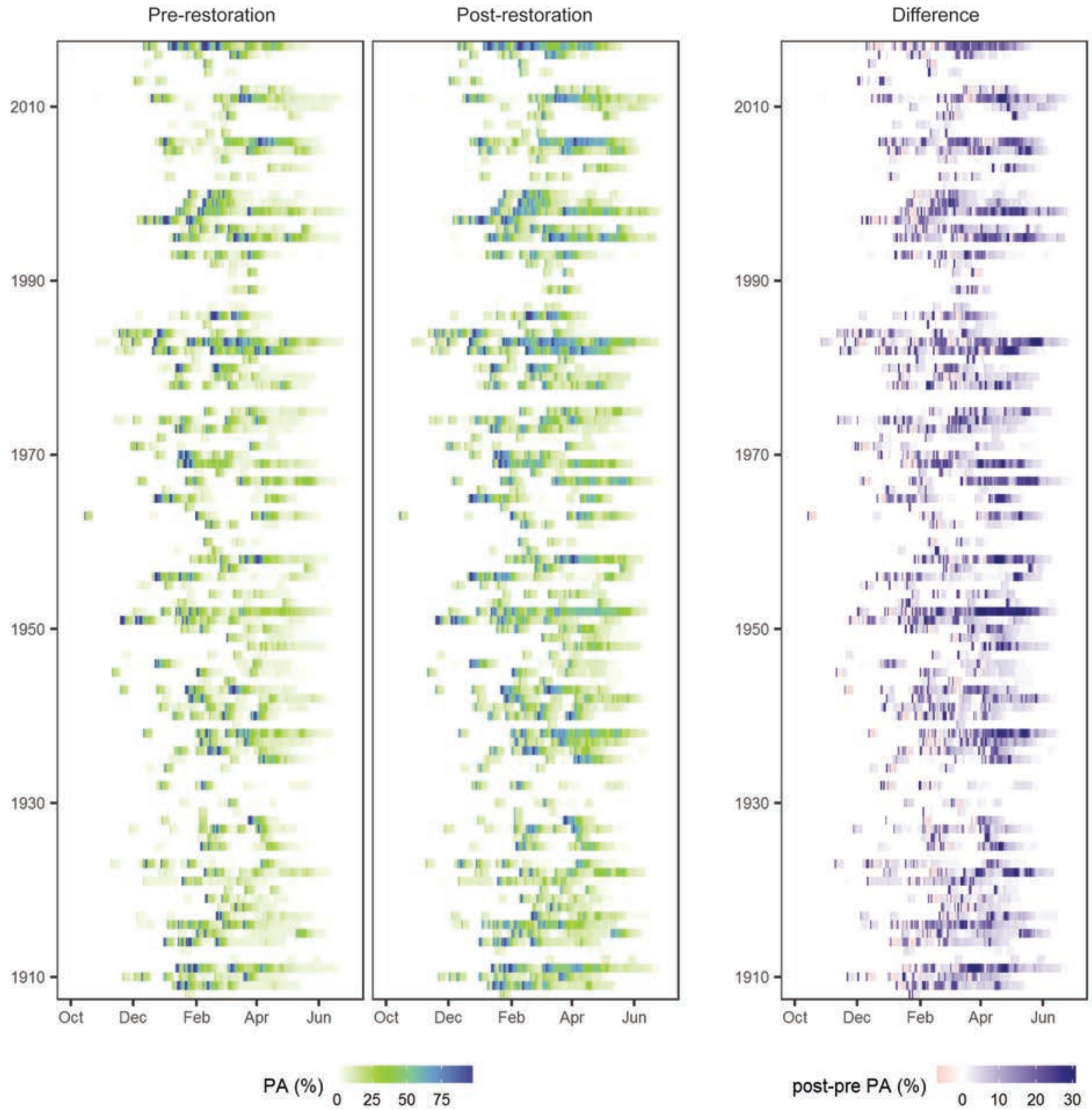

Figure 4-9. Daily percent inundated area (*PA*) for the period of record for pre- (1st panel) and post-restoration (2nd panel) and for the difference (3rd panel). For differencing, red indicates decreases post-restoration and blue indicates increases post-restoration.

occurred only for brief periods (Figure 4-13). However, consistent patterns within the daily difference were difficult to discern.

Spatially resolved summary

DEPTH AND VELOCITY

Spatial patterns of maximum flood event depths (*MaxD*) largely followed floodplain topography (Figure 4-14). Figure 4-15 shows that most floodplain surface (~75%) was <1 m for water year maximum conditions and <0.5 m for the maximum of all events. Maximum depths were greater under pre-

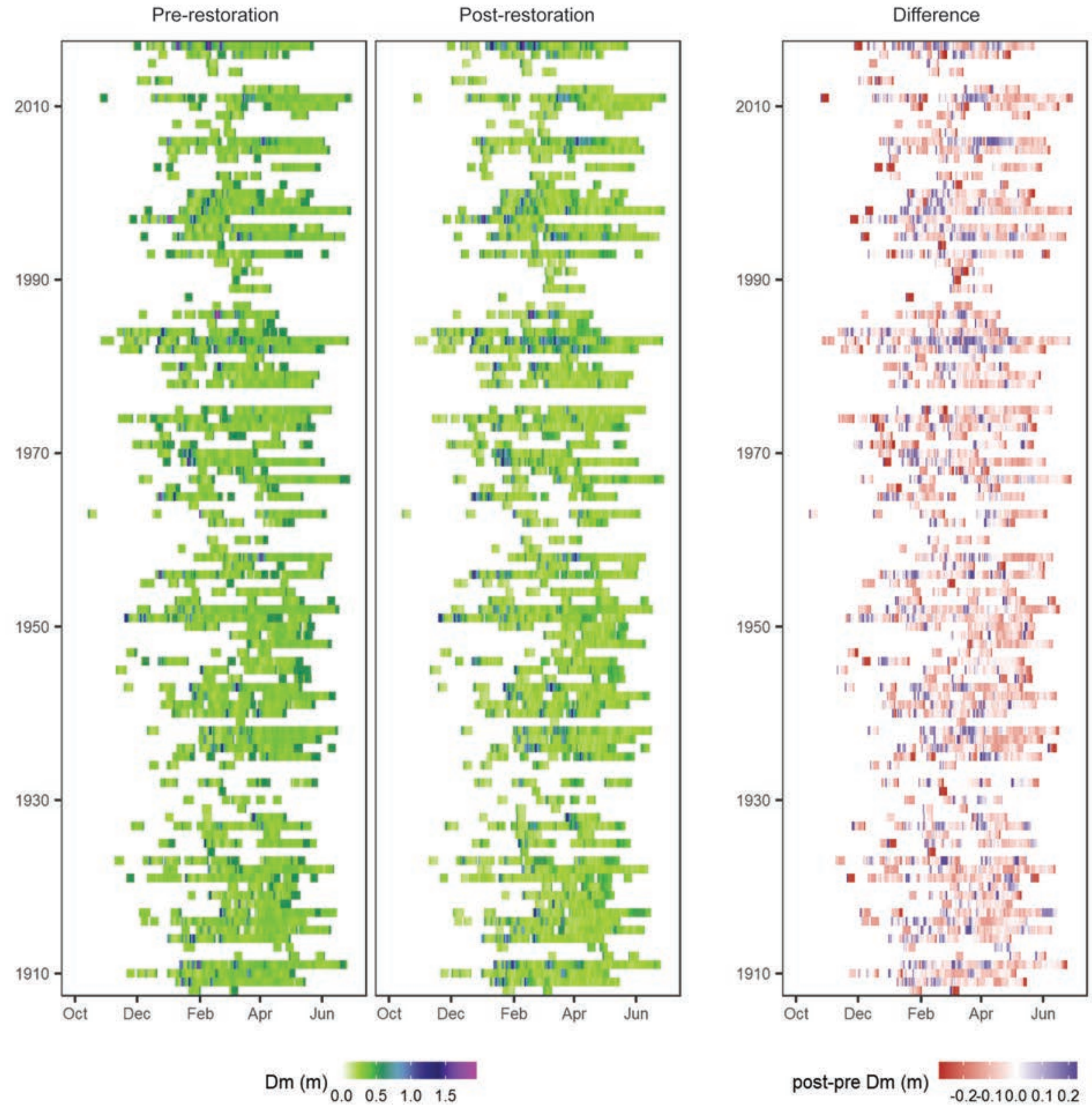

Figure 4-10. Daily spatial mean depth (*Dm*) for the period of record for pre- (1st panel) and post-restoration (2nd panel) and for the difference (3rd panel). For differencing, red indicates decreases post-restoration and blue indicates increases post-restoration.

restoration conditions near the main levee removal, in the floodplain area between the river and the levee. With the levee removed in the post-restoration configuration, maximum depths were lower (except for the floodplain surface that was graded to lower elevation). Modest depth increases occurred post-restoration in the central part of the floodplain and in the excavated swale at the downstream end of the floodplain. The restoration action also decreased maximum depth in the riparian forest area just north (upstream) of the primary levee removal site. Depth differences were >1 m only in small areas.

Mean velocity (*MV*) patterns were spatially heterogeneous across the floodplain and related to topographic variation and adjacency to the river (Figure 4-16). Nearly all of the area (>90%) both pre-

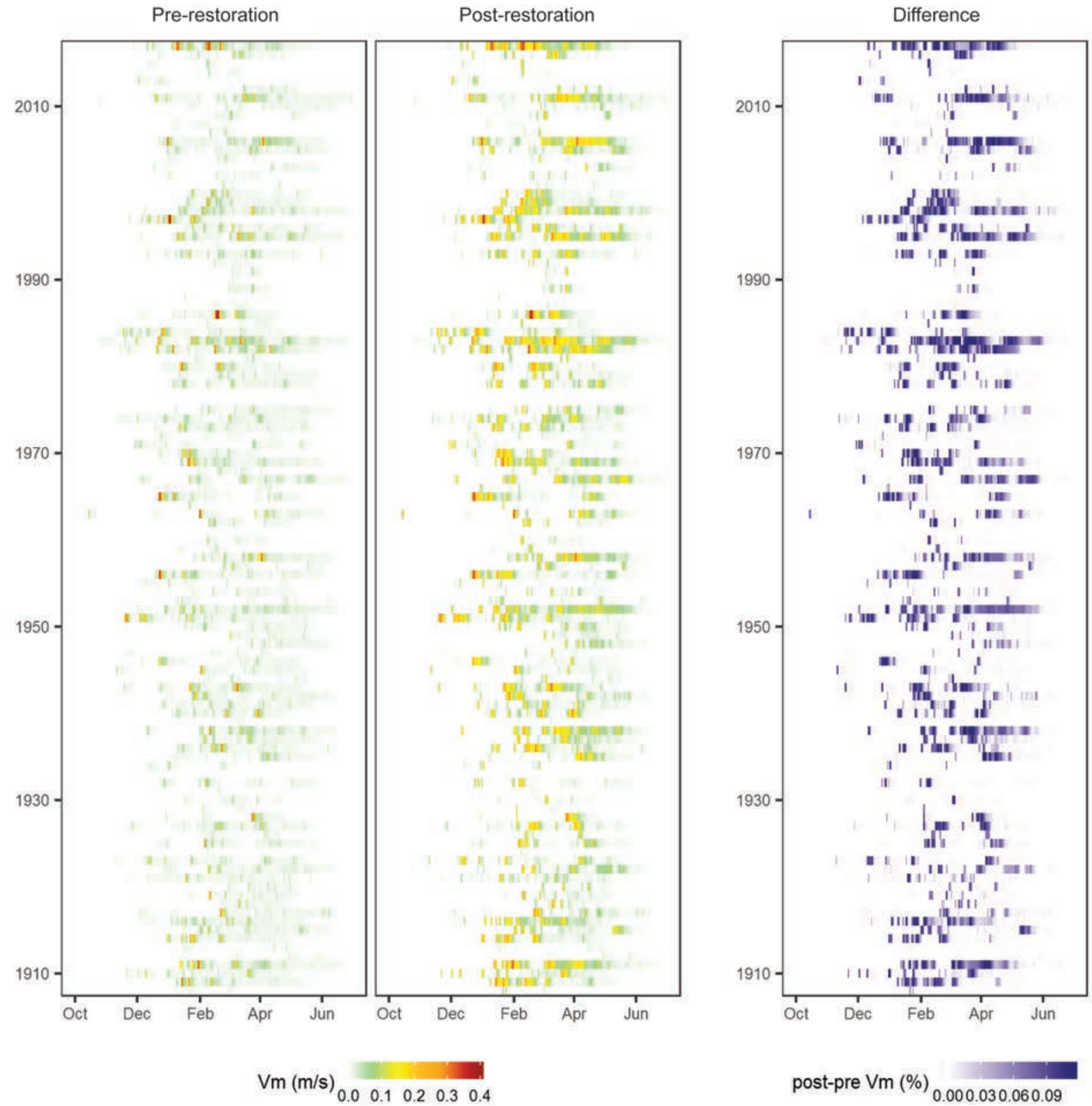

Figure 4-11. Daily spatial mean velocity (*Vm*) for the period of record for pre- (1st panel) and post-restoration (2nd panel) and for the difference (3rd panel). For differencing, red indicates decreases post-restoration and blue indicates increases post-restoration.

and post-restoration had low velocities (<0.25 m/s; see Figure 4-15). The areas of highest velocities were along the river at natural points of floodplain connectivity and levee breaches. Post-restoration, additional areas of higher velocities were at the interior ring levee breaches. The complexity of topography within the upper floodplain forest area also stands out. Mean velocities were higher overall post-restoration (<0.5 m/s difference) for most of the restoration site. Areas of reduced velocities were smaller but differences were larger in magnitude for some spots (up to 1 m/s).

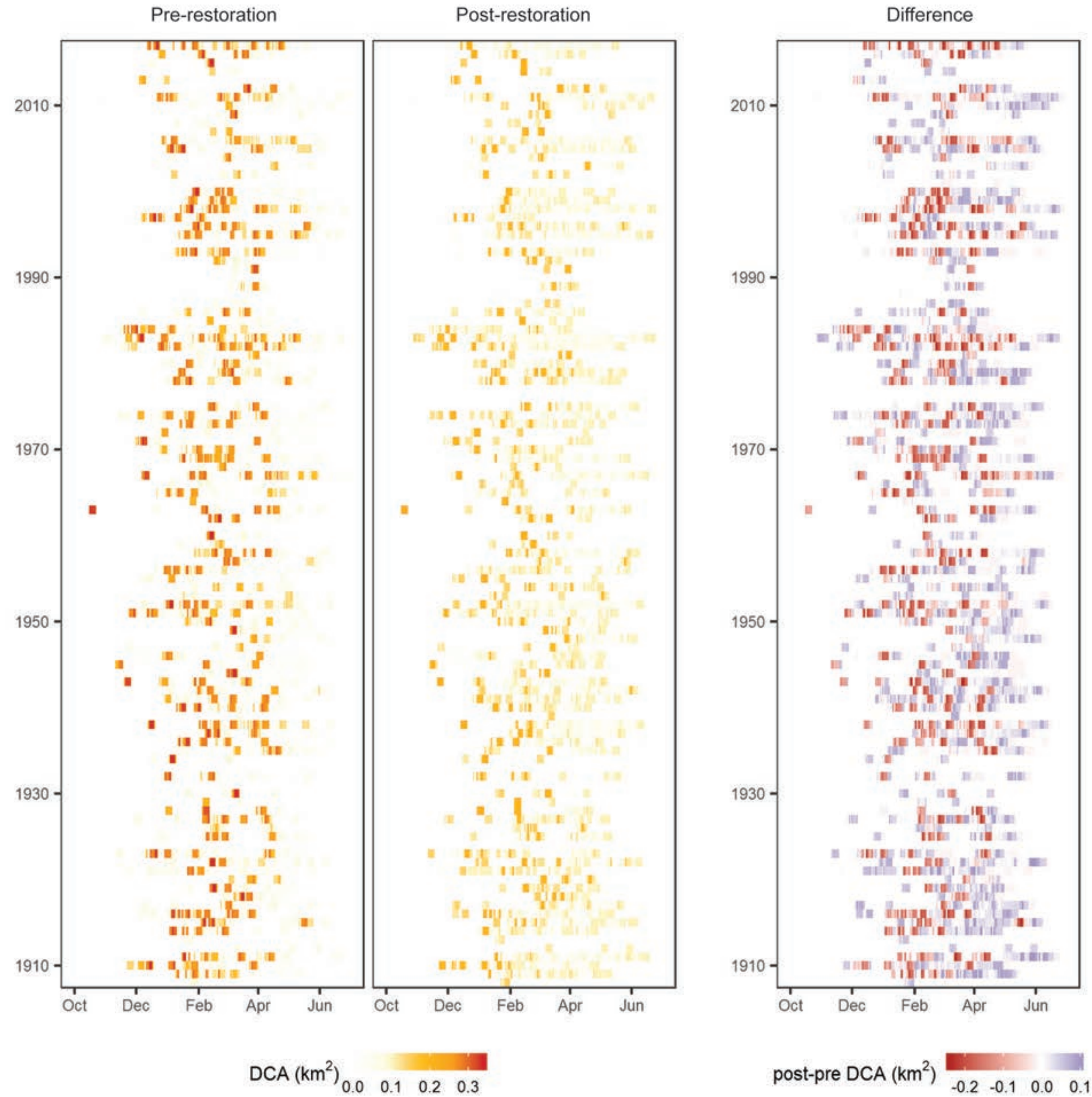

Figure 4-12. Daily disconnected (ponded) inundated area (*DCA*) for the period of record for pre- (1st panel) and post-restoration (2nd panel) and for the difference (3rd panel). For differencing, red indicates decreases post-restoration and blue indicates increases post-restoration.

DURATION

Inundation duration at the water year and event scale was found to vary spatially across the floodplain between 0-49 days and 0-18 days, respectively (Figure 4-17). The majority of the floodplain was inundated for 6 versus 14 days per water year (pre- and post-restoration) and 1 versus 3 days on an event scale (see Figure 4-15). Differences were most pronounced in the central portion of the floodplain restoration site. At the downstream end of the site, inundation duration increased for the excavated swale, but decreased along the edge of an area that was often ponded under pre-restoration conditions. Relatively small differences were found in duration upstream of the restoration site within the riparian forest. Spatial patterns are quite similar between the water year and event summaries.

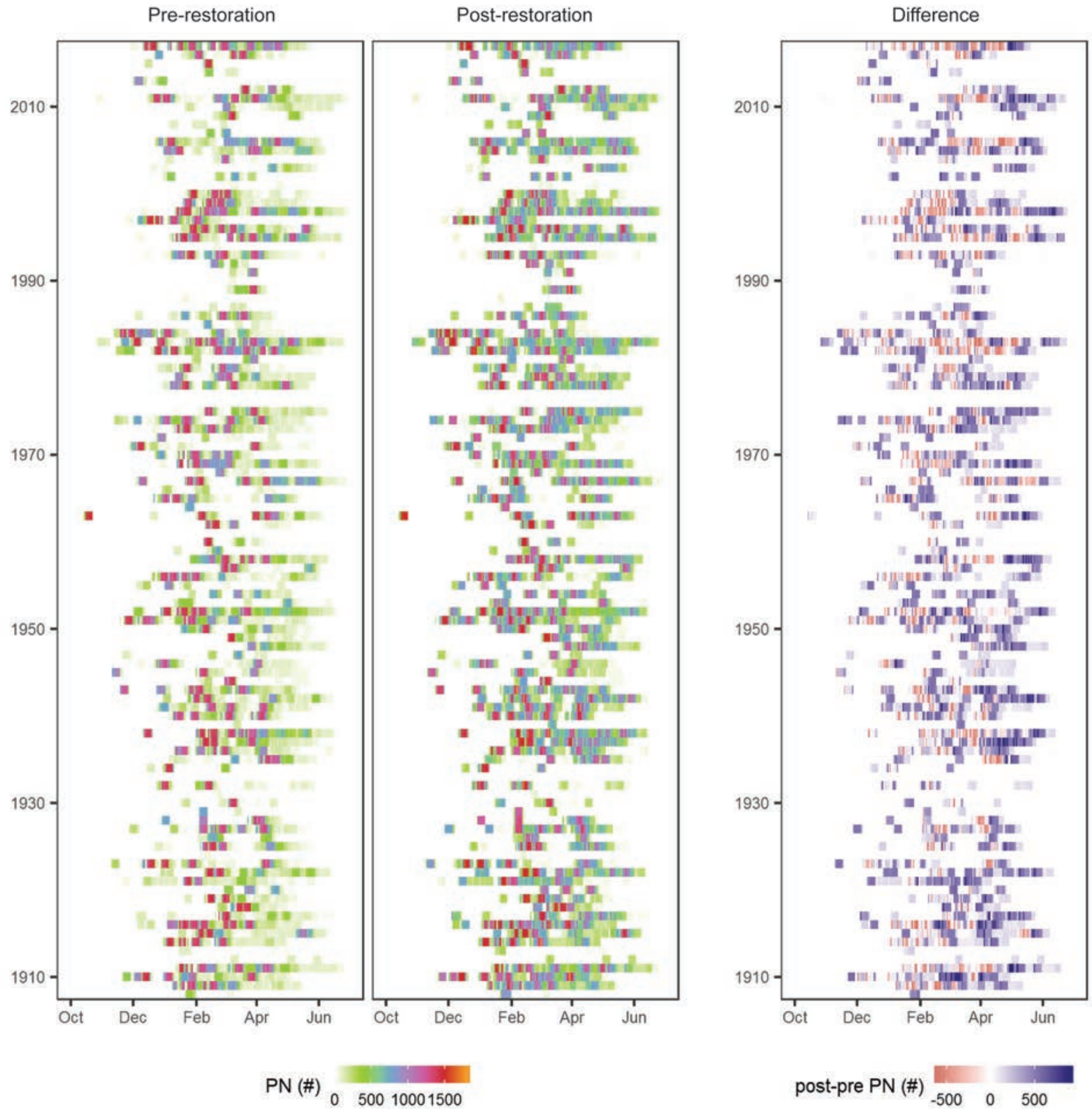

Figure 4-13. Daily count of inundated patches (*PN*) for the period of record for pre- (1st panel) and post-restoration (2nd panel) and for the difference (3rd panel). For differencing, red indicates decreases post-restoration and blue indicates increases post-restoration.

Two points should be considered when comparing these conditions. First, inundated areas disconnected from the river (i.e., ponded) are assumed to stay wet for seven days (to allow for infiltration and evaporation). Second, locations not inundated during a flood (in the case of event summaries) or season (in the case of water year summaries), were included in the calculation of mean. For example, an area inundated for 10 days in one event but 0 days for two other events within a water year would be assigned 10 days for the water year maximum, but the event summary for that year would be 3.3 days.

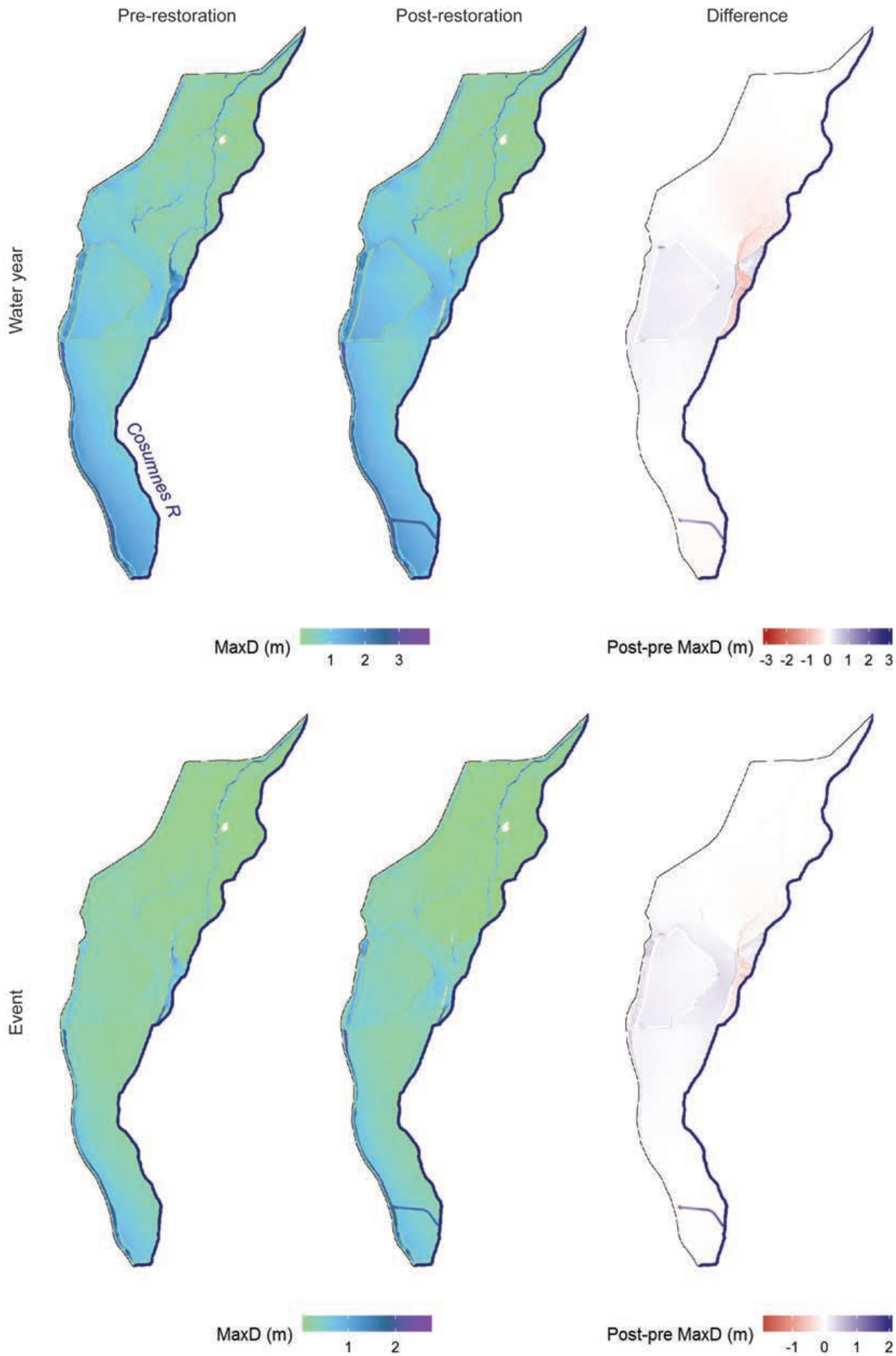

Figure 4-14. Spatial distribution of mean of maximum depth (*MaxD*) for water year maximum (top row) and all flood events (bottom row). For differencing, decreases post-restoration are red and increases are blue.

RATE OF CHANGE

For maximum falling rate (*MaxFR*), distribution across the floodplain was fairly even, with nearly all of the floodplain associated with declines of <1 m/day (Figure 4-18, see Figure 4-15). Pre-restoration, the areas of greatest rates of change occurred between the levee and river at the main breach site as well as along the side channel along the western boundary of the floodplain. The rates in these areas were substantially reduced (>0.5 m/day) under post-restoration conditions, which was exchanged for small areas of increased falling rates within the interior of the floodplain restoration site and within the excavated swale at the downstream end of the site.

FREQUENCY

Under pre-restoration conditions, frequency of inundation varied across the floodplain from less than once a year (close to 0 times per event) on average within the upper floodplain forest to around five times a year (around 1 per event) in the lower portion of the floodplain (Figure 4-19). While the maximum frequency did not change substantially pre- and post-restoration, about 50% of the floodplain inundated 1-2 more times a year (see Figure 4-15), particularly the area that was protected by the interior ring levee pre-restoration. While frequency increased across much of the floodplain with restoration, frequency decreased in the floodplain forest area just upstream of the main levee removal site. Very little change occurred within the upstream and downstream parts of the floodplain. As with duration, an area that does not inundate during an event is counted as a zero in the summary.

CONNECTIVITY

Other than small patches that were disconnected for brief periods of time, pre-restoration conditions showed substantial disconnectivity (ponding for >10% of the

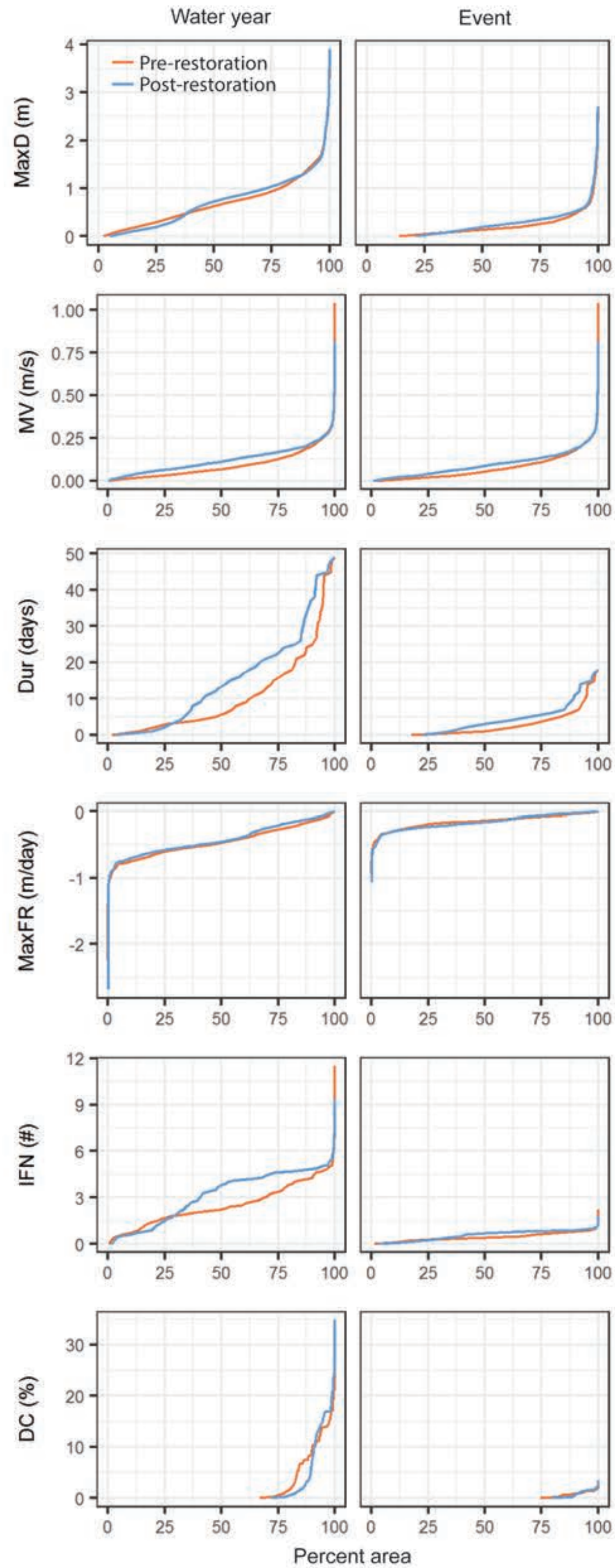

Figure 4-15. Cumulative spatial distribution for pre- and post-restoration configurations of mean conditions for selected spatially resolved metrics, summarized by water year (first column) and across all events (second column).

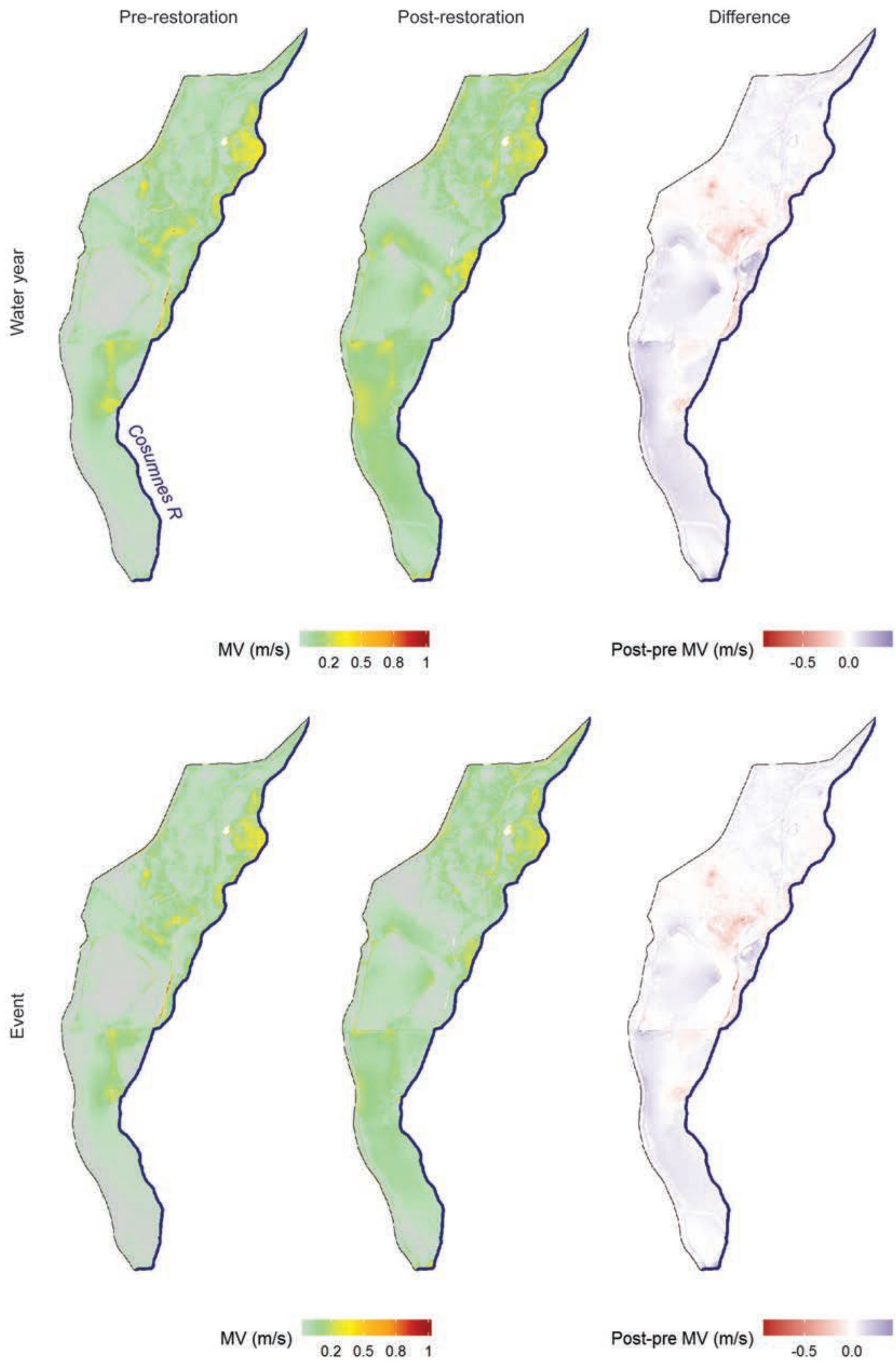

Figure 4-16. Spatial distribution of mean of mean velocity (*MV*) for water year maximum (top row) and all flood events (bottom row). For differencing, decreases post-restoration are red and increases are blue.

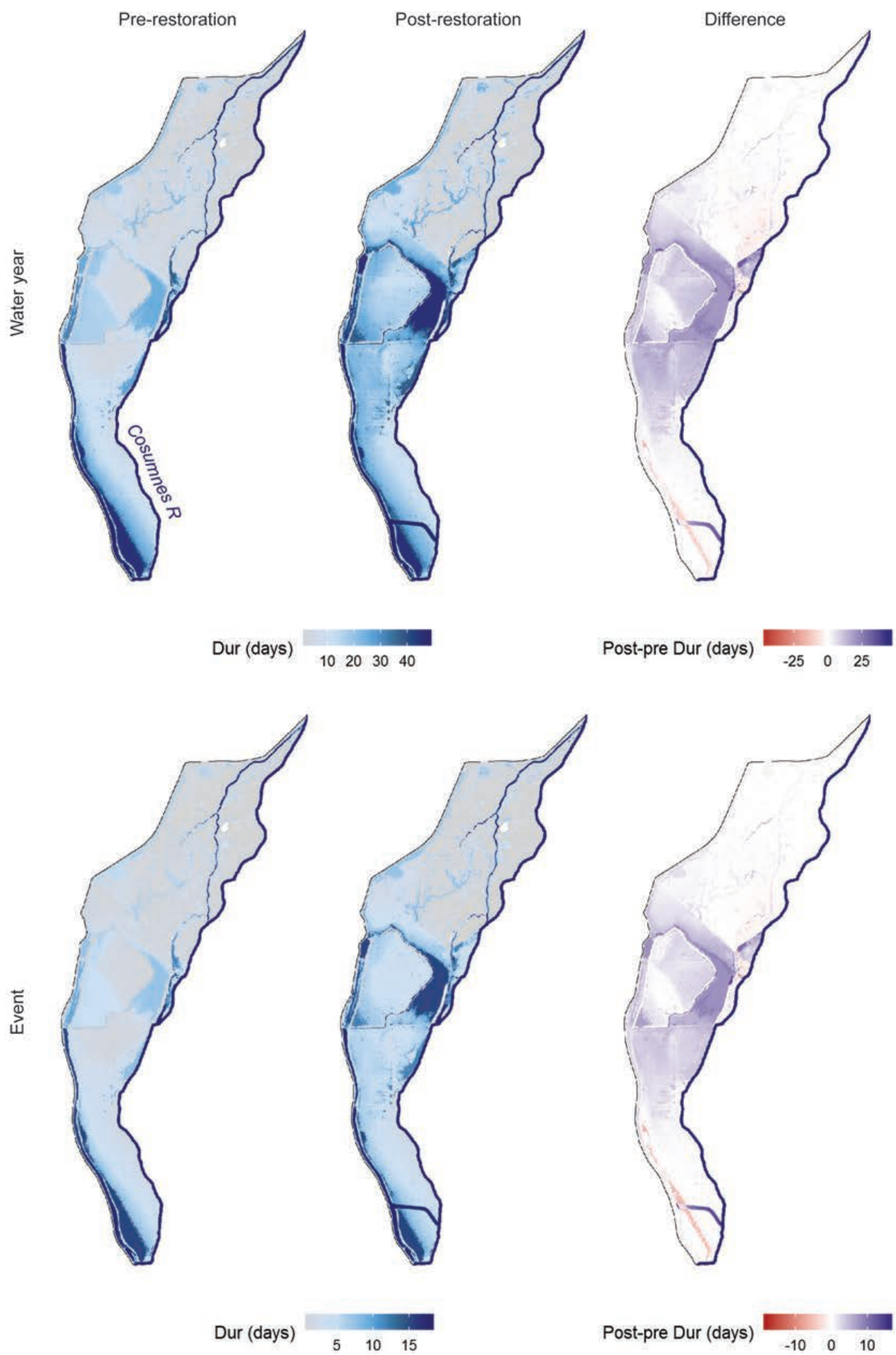

Figure 4-17. Spatial distribution of mean inundation duration (*Dur*) for water year maximum (top row) and all flood events (bottom row). For differencing, decreases post-restoration are red and increases are blue.

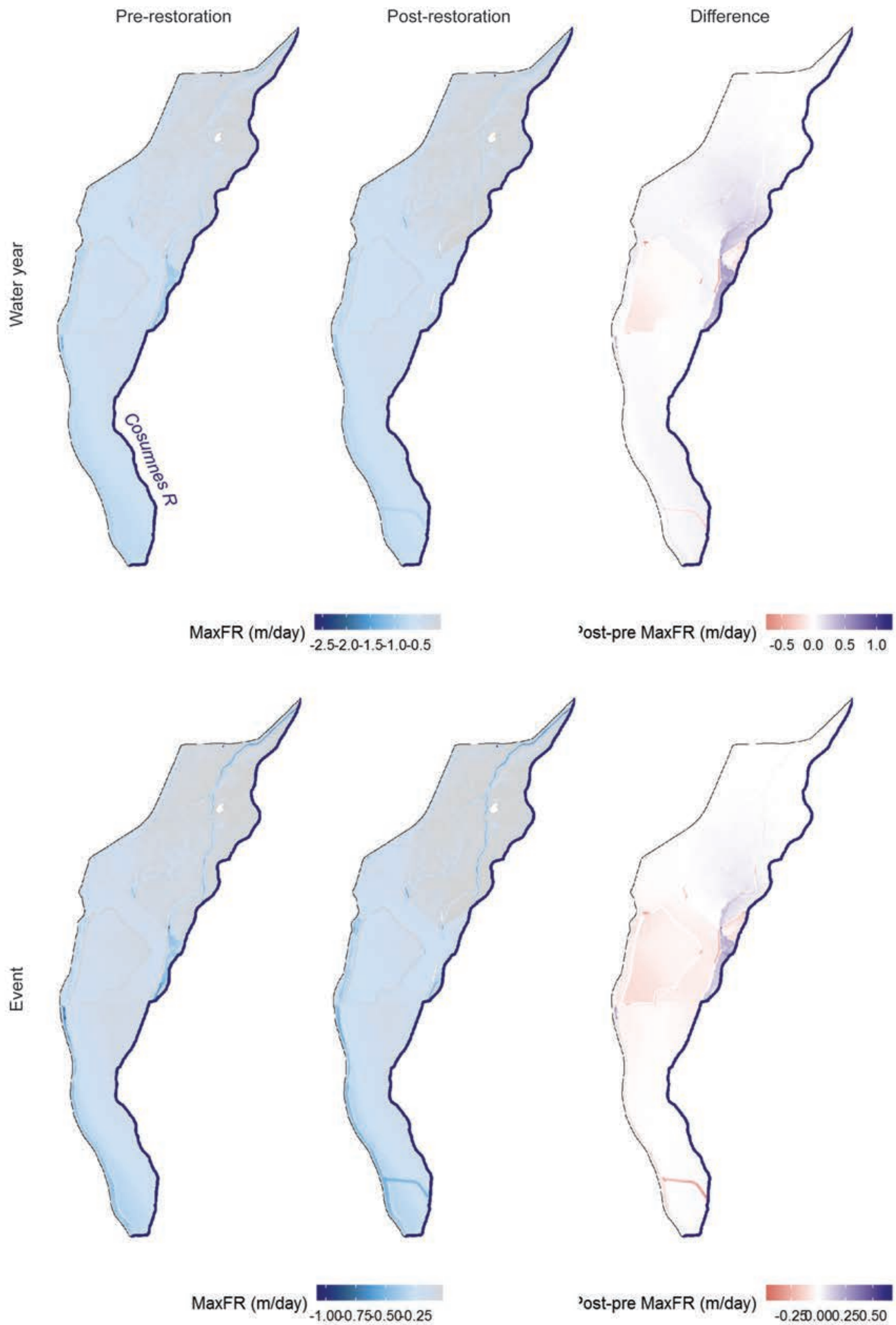

Figure 4-18. Spatial distribution of mean of maximum falling (negative) rate (*MaxFR*) for water year maximum (top row) and all flood events (bottom row). For differencing, higher magnitude falling rates post-restoration are red and lower magnitude rates are blue.

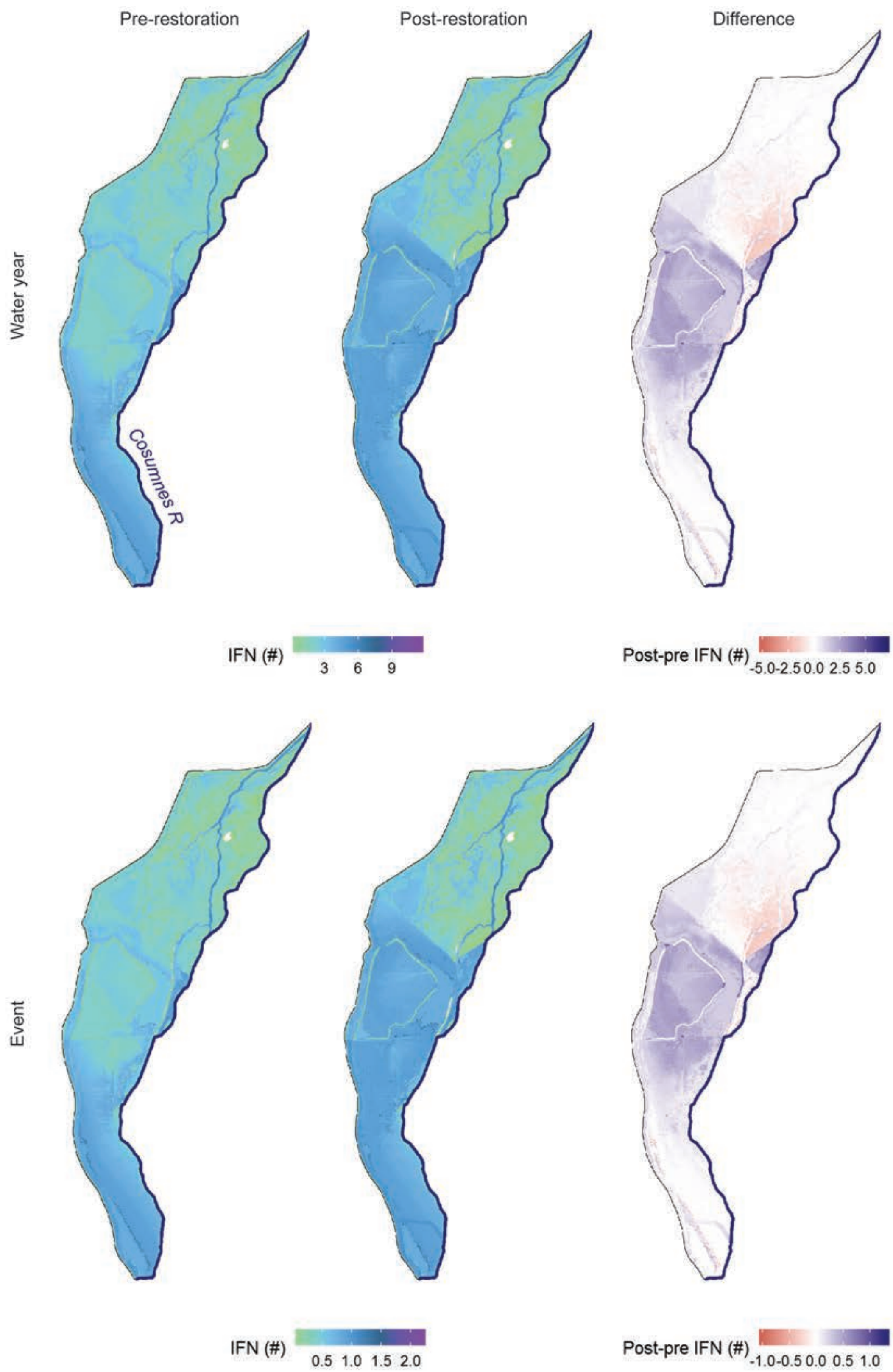

Figure 4-19. Spatial distribution of mean inundation frequency (*IFN*) summarized for water year total (top row) and all flood events (bottom row). For differencing, decreases post-restoration are red and increases are blue.

water year total or >1% of the water year across all events) on the west side of the levee at the site of the main levee removal, within the interior ring levee, and within the side channel along the western boundary of the floodplain (Figure 4-20). For post-restoration, the area of disconnectivity to the west of the main levee removal remained, increasing in magnitude over pre-restoration conditions. The areas of substantially decreased disconnectivity were within the interior ring levee and along the western boundary side channel. This finding suggests a tradeoff, while breaching of the interior ring levee and the swale construction increased connectivity in some areas, other areas became more prone to ponding post-restoration. Overall, more of the ponded area was disconnected for a greater portion of the year under post-restoration conditions (see Figure 4-15).

Relationship to flow regime

Exploration of the relationship between hydrospatial metrics and flow regime demonstrates that floodplain inundation response to flow is mediated by landscape topography and configuration. Some metrics were sensitive to water year type (defined as annual flow volume quantiles), while others were not (Figure 4-21). Also, spread in data distributions were highly variable. Metrics related to extent (*PMaxA*, *ADay*), duration (also *ADay*), connectivity (*DCADay*), and frequency of inundation (*IFNm*) were most affected by water year type, higher in wet years and lower in dry years. For very wet years (0.8-1 quantile), *IFNm* did not follow this trend and was lower overall than during wet years, suggesting that so much of the floodplain inundated for long periods in very wet years that the number of rewetting occurrences decreased. Most differences between the water year types for these metrics were significant (at the water year scale), according to pairwise Wilcoxon rank sum tests ($p < 0.05$). This general relationship was also followed by depth (*MaxDm*), velocity (*MVm*), and patch number (*MPN*). The opposite occurred for falling rate (*MaxFRm*) and spatial variability of depth (*MaxDcv*), which were higher in drier years than wetter years. As expected, the summary across all events showed greater spread and thus fewer significant differences due to the many types of flood events that occur in a given year. Relationships to water year type did not differ substantially between pre- and post-restoration conditions. However, increased *PMaxA* post-restoration was found to be most pronounced for dry (0.2-0.4 quantile) and critically dry (0-0.2 quantile) years. The least sensitive metrics summarized at the water year scale were water year day centroid volume (*WYDCnVol*) and mean total patch edge (*TPEm*). Pairwise Wilcoxon rank sum tests suggested no significant differences between water year types for these two metrics, except for pre-restoration *TPEm* in very wet years (Table 4-5).

Floods were also classified by flood type, following Whipple et al. (2017), and summarized hydrospatial metrics for each of those flood type groups compared (Table 4-6; Figure 4-22). As flood types are distinguished by magnitude of flows as well as flow regime characteristics of timing, duration, and hydrograph shape, these characteristics are reflected in the hydrospatial metrics. Most metrics for the two large-magnitude flood types stood out from the other flood types, either in magnitudes (e.g., *PMaxA*, *MaxDm*) or spread of the data (e.g., *PMaxA*, *MaxDcv*, *MPN*). For example, events with the two large-magnitude flood types were all inundated for a maximum of >75% of the floodplain area (*PMaxA*), whereas the other flood types spanned all but the highest percentages. Depth and velocity also were higher for the two large magnitude flood types. The two later-season flood types and two earlier-season types, as would be expected, had later and earlier *WYDCnVol*. Similar to water year type relationships, metrics relating to spatial heterogeneity (e.g., *TPEm*) were less associated with particular flood types. However, the large magnitude flood types had a higher distribution of patch numbers (*MPN*) and lower depth variability

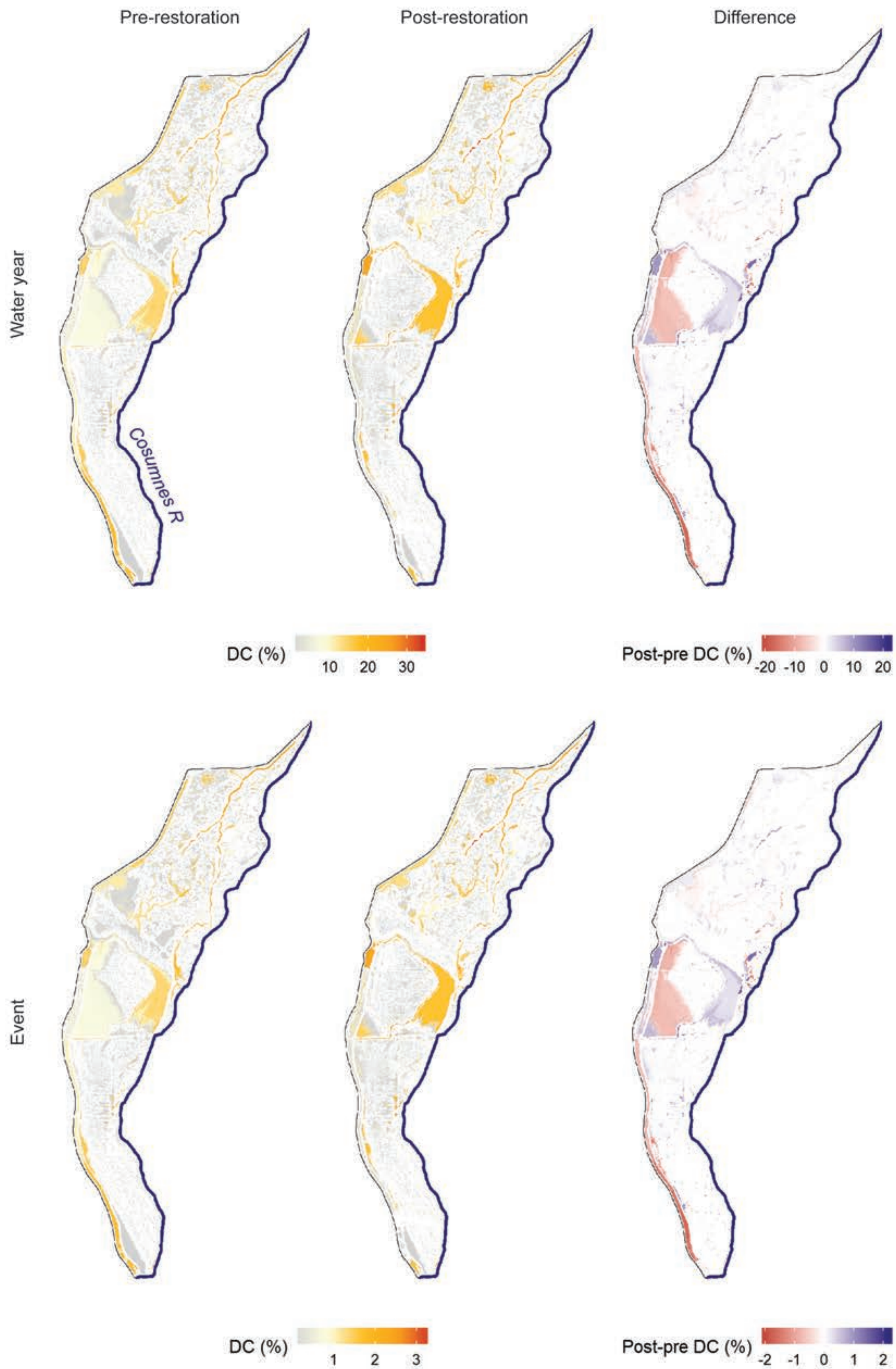

Figure 4-20. Spatial distribution of mean percent of year disconnected (*DC*) for the water year (top row) and all flood events (bottom row). For differencing, decreases post-restoration are red and increases are blue.

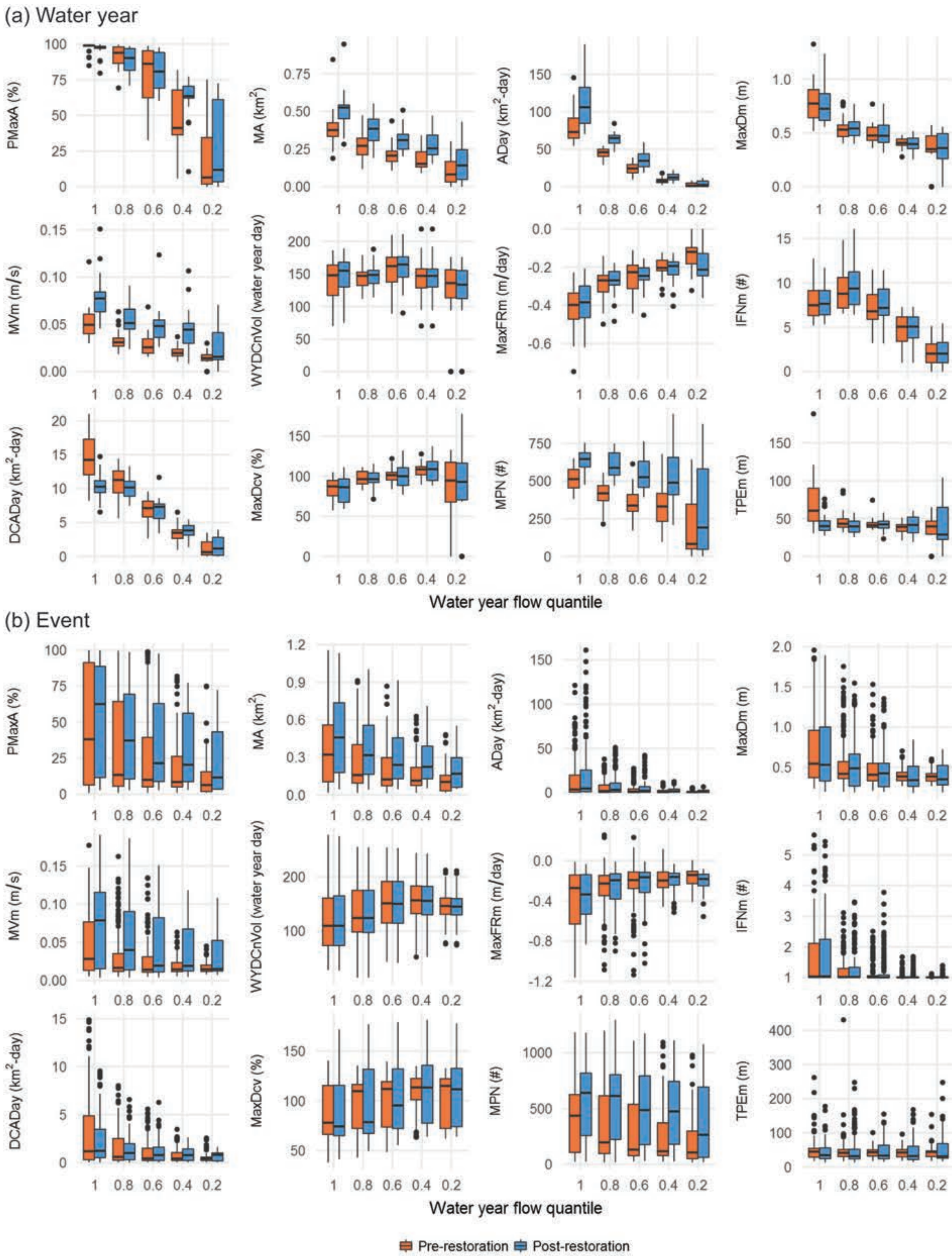

Figure 4-21. Summary statistics for each of five water year types (established as annual flow volume quantiles) for pre- and post-restoration conditions for each of the 12 selected priority hydrospatial metrics, at the water year (a) and event scale (b).

Table 4-5. Summary of non-parametric pairwise Wilcoxon rank sum tests to compare water year type group levels (annual flow quantiles) of water year and flood events for each selected metric under pre- and post-restoration conditions. Text is bolded for paired groups where the null hypothesis of no effect is rejected ($p < 0.05$).

		Water year								Event							
		Pre-restoration				Post-restoration				Pre-restoration				Post-restoration			
Metric	**WY type**	**1.0**	**0.8**	**0.6**	**0.4**	**1.0**	**0.8**	**0.6**	**0.4**	**1.0**	**0.8**	**0.6**	**0.4**	**1.0**	**0.8**	**0.6**	**0.4**
PMaxA	**0.8**	**0.001**	NA	NA	NA	**0.001**	NA	NA	NA	**0.032**	NA	NA	NA	**0.035**	NA	NA	NA
	0.6	**0.001**	0.074	NA	NA	**0.001**	0.074	NA	NA	**0.002**	0.366	NA	NA	**0.001**	0.357	NA	NA
	0.4	**<0.001**	**<0.001**	**0.003**	NA	**<0.001**	**<0.001**	**0.001**	NA	**<0.001**	**0.031**	0.366	NA	**<0.001**	**0.038**	0.357	NA
	0.2	**<0.001**	**<0.001**	**<0.001**	**0.001**	**<0.001**	**<0.001**	**<0.001**	**0.001**	**<0.001**	**0.005**	0.084	0.366	**<0.001**	**0.009**	0.097	0.357
MA	**0.8**	**0.004**	NA	NA	NA	**0.001**	NA	NA	NA	**0.037**	NA	NA	NA	**0.026**	NA	NA	NA
	0.6	**<0.001**	0.137	NA	NA	**<0.001**	0.137	NA	NA	**0.002**	0.300	NA	NA	**0.001**	0.207	NA	NA
	0.4	**<0.001**	**0.019**	0.137	NA	**<0.001**	**0.038**	0.176	NA	**<0.001**	**0.050**	0.304	NA	**<0.001**	**0.026**	0.308	NA
	0.2	**<0.001**	**<0.001**	**0.006**	**0.022**	**<0.001**	**<0.001**	**0.005**	**0.013**	**<0.001**	**0.004**	0.074	0.300	**<0.001**	**0.001**	**0.036**	0.207
ADay	**0.8**	**<0.001**	NA	NA	NA	**<0.001**	NA	NA	NA	0.061	NA	NA	NA	0.090	NA	NA	NA
	0.6	**<0.001**	**<0.001**	NA	NA	**<0.001**	**<0.001**	NA	NA	**0.002**	0.258	NA	NA	**0.003**	0.229	NA	NA
	0.4	**<0.001**	**<0.001**	**<0.001**	NA	**<0.001**	**<0.001**	**<0.001**	NA	**<0.001**	**0.019**	0.273	NA	**<0.001**	**0.018**	0.229	NA
	0.2	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**0.007**	0.061	0.273	**<0.001**	**0.004**	0.053	0.229
MaxDm	**0.8**	**<0.001**	NA	NA	NA	**<0.001**	NA	NA	NA	0.035	NA	NA	NA	**0.044**	NA	NA	NA
	0.6	**<0.001**	0.258	NA	NA	**<0.001**	0.127	NA	NA	**0.001**	0.339	NA	NA	**0.001**	0.346	NA	NA
	0.4	**<0.001**	**<0.001**	**0.011**	NA	**<0.001**	**<0.001**	0.052	NA	**<0.001**	**0.006**	0.339	NA	**<0.001**	**0.012**	0.408	NA
	0.2	**<0.001**	**<0.001**	**0.011**	0.353	**<0.001**	**<0.001**	**0.013**	0.406	**<0.001**	**0.015**	0.339	0.593	**0.002**	0.161	0.865	0.865
MVm	**0.8**	**0.001**	NA	NA	NA	**0.008**	NA	NA	NA	0.068	NA	NA	NA	**0.049**	NA	NA	NA
	0.6	**0.001**	0.105	NA	NA	**<0.001**	0.150	NA	NA	**0.001**	0.352	NA	NA	**0.001**	0.305	NA	NA
	0.4	**<0.001**	**0.002**	0.066	NA	**0.001**	0.150	0.524	NA	**<0.001**	0.189	1.000	NA	**<0.001**	0.170	1.000	NA
	0.2	**<0.001**	**<0.001**	**0.001**	**0.038**	**<0.001**	**<0.001**	**0.014**	**0.037**	**0.008**	0.481	1.000	1.000	**0.001**	0.088	1.000	1.000
WYDCnVol	**0.8**	1.000	NA	NA	NA	1.000	NA	NA	NA	0.069	NA	NA	NA	0.095	NA	NA	NA
	0.6	0.326	0.266	NA	NA	0.449	0.397	NA	NA	**<0.001**	**0.012**	NA	NA	**<0.001**	**0.015**	NA	NA
	0.4	1.000	1.000	0.781	NA	1.000	1.000	0.449	NA	**<0.001**	**0.008**	0.813	NA	**<0.001**	**0.009**	0.857	NA
	0.2	1.000	0.812	0.059	0.812	0.449	0.427	**0.020**	0.449	**0.009**	0.177	0.682	0.347	**0.015**	0.208	0.646	0.334
MaxFRm	**0.8**	**<0.001**	NA	NA	NA	**0.007**	NA	NA	NA	0.144	NA	NA	NA	**0.029**	NA	NA	NA
	0.6	**<0.001**	0.159	NA	NA	**<0.001**	0.256	NA	NA	**0.002**	0.100	NA	NA	**<0.001**	0.395	NA	NA
	0.4	**<0.001**	**0.015**	0.159	NA	**<0.001**	0.052	0.293	NA	**0.001**	**0.049**	0.832	NA	**<0.001**	0.156	1.000	NA

Metric *(cont.)*	WY type	Water year								Event							
		Pre-restoration				Post-restoration				Pre-restoration				Post-restoration			
		1.0	**0.8**	**0.6**	**0.4**	**1.0**	**0.8**	**0.6**	**0.4**	**1.0**	**0.8**	**0.6**	**0.4**	**1.0**	**0.8**	**0.6**	**0.4**
	0.2	**<0.001**	**<0.001**	**0.003**	**0.034**	**<0.001**	**0.040**	0.110	0.483	**<0.001**	**0.002**	0.195	0.201	**0.010**	1.000	1.000	1.000
IFNm	**0.8**	0.159	NA	NA	NA	**0.050**	NA	NA	NA	0.202	NA	NA	NA	0.262	NA	NA	NA
	0.6	0.321	**0.022**	NA	NA	0.633	**0.022**	NA	NA	**0.011**	0.157	NA	NA	**0.016**	0.262	NA	NA
	0.4	**<0.001**	**<0.001**	**0.005**	NA	**<0.001**	**<0.001**	**<0.001**	NA	**0.003**	**0.014**	1.000	NA	**0.007**	0.055	1.000	NA
	0.2	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**0.023**	**0.026**	1.000	1.000	0.051	0.137	1.000	1.000
DCADay	**0.8**	**0.002**	NA	NA	NA	0.799	NA	NA	NA	0.064	NA	NA	NA	0.227	NA	NA	NA
	0.6	**<0.001**	**<0.001**	NA	NA	**<0.001**	**<0.001**	NA	NA	**0.001**	0.337	NA	NA	**0.011**	0.535	NA	NA
	0.4	**<0.001**	**<0.001**	**<0.001**	NA	**<0.001**	**<0.001**	**<0.001**	NA	**<0.001**	0.052	0.556	NA	**0.001**	**0.049**	0.560	NA
	0.2	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**0.001**	**0.023**	0.337	0.556	**0.004**	**0.049**	0.535	0.560
MaxDcv	**0.8**	**0.005**	NA	NA	NA	0.193	NA	NA	NA	0.065	NA	NA	NA	0.097	NA	NA	NA
	0.6	**0.002**	0.707	NA	NA	**0.033**	1.000	NA	NA	**0.006**	0.649	NA	NA	**0.002**	0.357	NA	NA
	0.4	**<0.001**	**0.029**	0.125	NA	**0.001**	0.102	0.443	NA	**<0.001**	**0.028**	0.649	NA	**<0.001**	**0.007**	0.357	NA
	0.2	0.707	0.630	0.624	0.342	1.000	1.000	1.000	0.443	**0.016**	0.371	0.774	0.899	**0.011**	0.357	0.945	0.945
MPN	**0.8**	**0.005**	NA	NA	NA	0.496	NA	NA	NA	0.324	NA	NA	NA	1.000	NA	NA	NA
	0.6	**<0.001**	0.117	NA	NA	**0.012**	0.217	NA	NA	0.053	0.889	NA	NA	0.355	1.000	NA	NA
	0.4	**<0.001**	**0.042**	0.424	NA	0.052	0.440	0.726	NA	**0.004**	0.324	0.889	NA	0.257	1.000	1.000	NA
	0.2	**<0.001**	**0.001**	**0.010**	**0.037**	**0.005**	**0.005**	0.062	0.062	**0.003**	0.073	0.400	0.889	0.075	0.393	1.000	1.000
TPEm	**0.8**	**0.029**	NA	NA	NA	1.000	NA	NA	NA	1.000	NA	NA	NA	1.000	NA	NA	NA
	0.6	**0.020**	1.000	NA	NA	1.000	1.000	NA	NA	1.000	1.000	NA	NA	1.000	1.000	NA	NA
	0.4	**<0.001**	**0.029**	0.274	NA	1.000	1.000	1.000	NA	1.000	1.000	1.000	NA	1.000	1.000	1.000	NA
	0.2	**0.003**	0.195	0.440	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000

Table 4-6. Summary of non-parametric pairwise Wilcoxon rank sum tests to compare flood type group levels of flood events for each selected metric under pre- and post-restoration conditions. Cells are highlighted for paired groups where the null hypothesis of no effect is rejected (p <0.05).

		Event									
		Pre-restoration					Post-restoration				
Metric	Flood type	Very large	Large and long	Long and late	Small and late	Small and early	Very large	Large and long	Long and late	Small and late	Small and early
PMaxA	Large and long	**<0.001**	NA	NA	NA	NA	**<0.001**	NA	NA	NA	NA
	Long and late	**<0.001**	**<0.001**	NA	NA	NA	**<0.001**	**<0.001**	NA	NA	NA
	Small and late	**<0.001**	**<0.001**	**<0.001**	NA	NA	**<0.001**	**<0.001**	**<0.001**	NA	NA
	Small and early	**<0.001**	**<0.001**	**0.002**	**<0.001**	NA	**<0.001**	**<0.001**	**0.001**	**<0.001**	NA
	Late peak	**<0.001**	**<0.001**	**0.003**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**0.003**	**<0.001**	**<0.001**
MA	Large and long	**0.009**	NA	NA	NA	NA	**<0.001**	NA	NA	NA	NA
	Long and late	**<0.001**	**<0.001**	NA	NA	NA	**<0.001**	**<0.001**	NA	NA	NA
	Small and late	**<0.001**	**<0.001**	**<0.001**	NA	NA	**<0.001**	**<0.001**	**<0.001**	NA	NA
	Small and early	**<0.001**	**<0.001**	0.244	**<0.001**	NA	**<0.001**	**<0.001**	**0.023**	**<0.001**	NA
	Late peak	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
ADay	Large and long	**<0.001**	NA	NA	NA	NA	**<0.001**	NA	NA	NA	NA
	Long and late	**<0.001**	**<0.001**	NA	NA	NA	**<0.001**	**<0.001**	NA	NA	NA
	Small and late	**<0.001**	**<0.001**	**<0.001**	NA	NA	**<0.001**	**<0.001**	**<0.001**	NA	NA
	Small and early	**<0.001**	**<0.001**	**<0.001**	**0.007**	NA	**<0.001**	**<0.001**	**<0.001**	**0.019**	NA
	Late peak	**<0.001**	**<0.001**	0.207	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**0.018**	**<0.001**	**<0.001**
MaxDm	Large and long	**<0.001**	NA	NA	NA	NA	**<0.001**	NA	NA	NA	NA
	Long and late	**<0.001**	**<0.001**	NA	NA	NA	**<0.001**	**<0.001**	NA	NA	NA
	Small and late	**<0.001**	**<0.001**	0.723	NA	NA	**<0.001**	**<0.001**	0.469	NA	NA
	Small and early	**<0.001**	**<0.001**	0.090	0.166	NA	**<0.001**	**<0.001**	**0.008**	**0.015**	NA
	Late peak	**<0.001**	**<0.001**	0.723	0.723	**0.010**	**<0.001**	**<0.001**	0.469	0.239	**0.001**
MVm	Large and long	**0.017**	NA	NA	NA	NA	**0.005**	NA	NA	NA	NA
	Long and late	**<0.001**	**<0.001**	NA	NA	NA	**<0.001**	**<0.001**	NA	NA	NA
	Small and late	**<0.001**	**<0.001**	**0.007**	NA	NA	**<0.001**	**<0.001**	**<0.001**	NA	NA
	Small and early	**<0.001**	**<0.001**	0.150	0.196	NA	**<0.001**	**<0.001**	**0.039**	**0.039**	NA
	Late peak	**<0.001**	**<0.001**	**0.007**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
WYDCnVol	Large and long	0.889	NA	NA	NA	NA	0.798	NA	NA	NA	NA
	Long and late	**<0.001**	**<0.001**	NA	NA	NA	**<0.001**	**<0.001**	NA	NA	NA
	Small and late	**0.002**	**<0.001**	**<0.001**	NA	NA	**0.004**	**<0.001**	**<0.001**	NA	NA
	Small and early	**<0.001**	**<0.001**	**<0.001**	**<0.001**	NA	**<0.001**	**<0.001**	**<0.001**	**<0.001**	NA
	Late peak	**0.002**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
MaxFRm	Large and long	**0.003**	NA	NA	NA	NA	**0.012**	NA	NA	NA	NA
	Long and late	**<0.001**	**<0.001**	NA	NA	NA	**<0.001**	**<0.001**	NA	NA	NA
	Small and late	**<0.001**	**<0.001**	**<0.001**	NA	NA	**<0.001**	**<0.001**	**0.047**	NA	NA
	Small and early	**<0.001**	**<0.001**	**0.003**	0.106	NA	**<0.001**	**<0.001**	0.615	**0.013**	NA
	Late peak	**<0.001**	**<0.001**	**0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
IFNm	Large and long	**<0.001**	NA	NA	NA	NA	**<0.001**	NA	NA	NA	NA
	Long and late	**<0.001**	**<0.001**	NA	NA	NA	**<0.001**	**<0.001**	NA	NA	NA
	Small and late	**<0.001**	**<0.001**	**<0.001**	NA	NA	**<0.001**	**<0.001**	**<0.001**	NA	NA
	Small and early	**<0.001**	**<0.001**	**<0.001**	0.085	NA	**<0.001**	**<0.001**	**<0.001**	0.057	NA

		Event									
		Pre-restoration					Post-restoration				
Metric *(cont.)*	Flood type	Very large	Large and long	Long and late	Small and late	Small and early	Very large	Large and long	Long and late	Small and late	Small and early
	Late peak	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**0.012**
DCADay	**Large and long**	**<0.001**	NA	NA	NA	NA	**0.001**	NA	NA	NA	NA
	Long and late	**<0.001**	**<0.001**	NA	NA	NA	**<0.001**	**<0.001**	NA	NA	NA
	Small and late	**<0.001**	**<0.001**	**<0.001**	NA	NA	**<0.001**	**<0.001**	**<0.001**	NA	NA
	Small and early	**<0.001**	**<0.001**	**<0.001**	0.053	NA	**<0.001**	**<0.001**	**<0.001**	**<0.001**	NA
	Late peak	**<0.001**	**<0.001**	0.874	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**0.001**	**<0.001**	**<0.001**
MaxDcv	**Large and long**	**<0.001**	NA	NA	NA	NA	**<0.001**	NA	NA	NA	NA
	Long and late	**<0.001**	**<0.001**	NA	NA	NA	**<0.001**	**<0.001**	NA	NA	NA
	Small and late	**<0.001**	**<0.001**	0.090	NA	NA	**<0.001**	**<0.001**	0.655	NA	NA
	Small and early	**<0.001**	**<0.001**	0.771	**0.011**	NA	**<0.001**	**<0.001**	0.143	**0.002**	NA
	Late peak	**<0.001**	**<0.001**	0.771	0.090	0.771	**<0.001**	**<0.001**	0.724	0.741	**0.032**
MPN	**Large and long**	1.000	NA	NA	NA	NA	0.598	NA	NA	NA	NA
	Long and late	**<0.001**	**<0.001**	NA	NA	NA	**0.002**	**<0.001**	NA	NA	NA
	Small and late	**<0.001**	**<0.001**	**<0.001**	NA	NA	**<0.001**	**<0.001**	**<0.001**	NA	NA
	Small and early	**<0.001**	**<0.001**	1.000	**<0.001**	NA	0.598	**0.024**	0.222	**<0.001**	NA
	Late peak	1.000	0.832	**<0.001**	**<0.001**	**<0.001**	0.598	0.598	**<0.001**	**<0.001**	**0.001**
TPEm	**Large and long**	**<0.001**	NA	NA	NA	NA	**0.047**	NA	NA	NA	NA
	Long and late	**<0.001**	1.000	NA	NA	NA	0.206	1.000	NA	NA	NA
	Small and late	**<0.001**	1.000	0.869	NA	NA	1.000	0.056	0.149	NA	NA
	Small and early	**<0.001**	**0.018**	**0.004**	**<0.001**	NA	**<0.001**	**0.001**	**<0.001**	**<0.001**	NA
	Late peak	**<0.001**	**<0.001**	**<0.001**	**<0.001**	0.394	**<0.001**	**<0.001**	**<0.001**	**<0.001**	0.888

(*MaxDcv*), suggesting flow thresholds for minimum numbers of patches and more similar depths in the areas of the floodplain that are inundated at high flows.

Hydrospatial metrics resolved at the daily scale were also compared to daily flows (Figure 4-23). Metrics responded differently to given daily flows, and metrics varied non-linearly with flow. The graph for percent area (*PA*) is concave, with the floodplain inundating nearly completely at about 300 m^3/s, which is consistent for both restoration conditions. Several metrics (e.g., *PA*, *Dm*, and *Vm*) rise more rapidly at lower flows under post-restoration conditions. Pre-restoration spatial mean depth (*Dm*) surpasses post-restoration conditions at higher flood flows (>400 m^3/s). Two of the metrics (*DCA* and *PN*) return to zero around the threshold flow of ~300 m^3/s. At low flood flows, when the floodplain is just wetted, the spread in metric values tends to be wider, which is likely related to disconnected areas (ponding) on the falling limb of the hydrograph at the lower flows.

DISCUSSION

The research presented here responds to the needs to better link changes affecting physical processes to their ecological implications and to develop appropriate river restoration strategies for improving ecosystem integrity within human-dominated and changing landscapes (Palmer et al., 2014; Poff, 2017). It applies an interdisciplinary hydrologic and landscape ecology approach to include the landscape context in the evaluation of flood regime. The hydrospatial analysis developed specifically

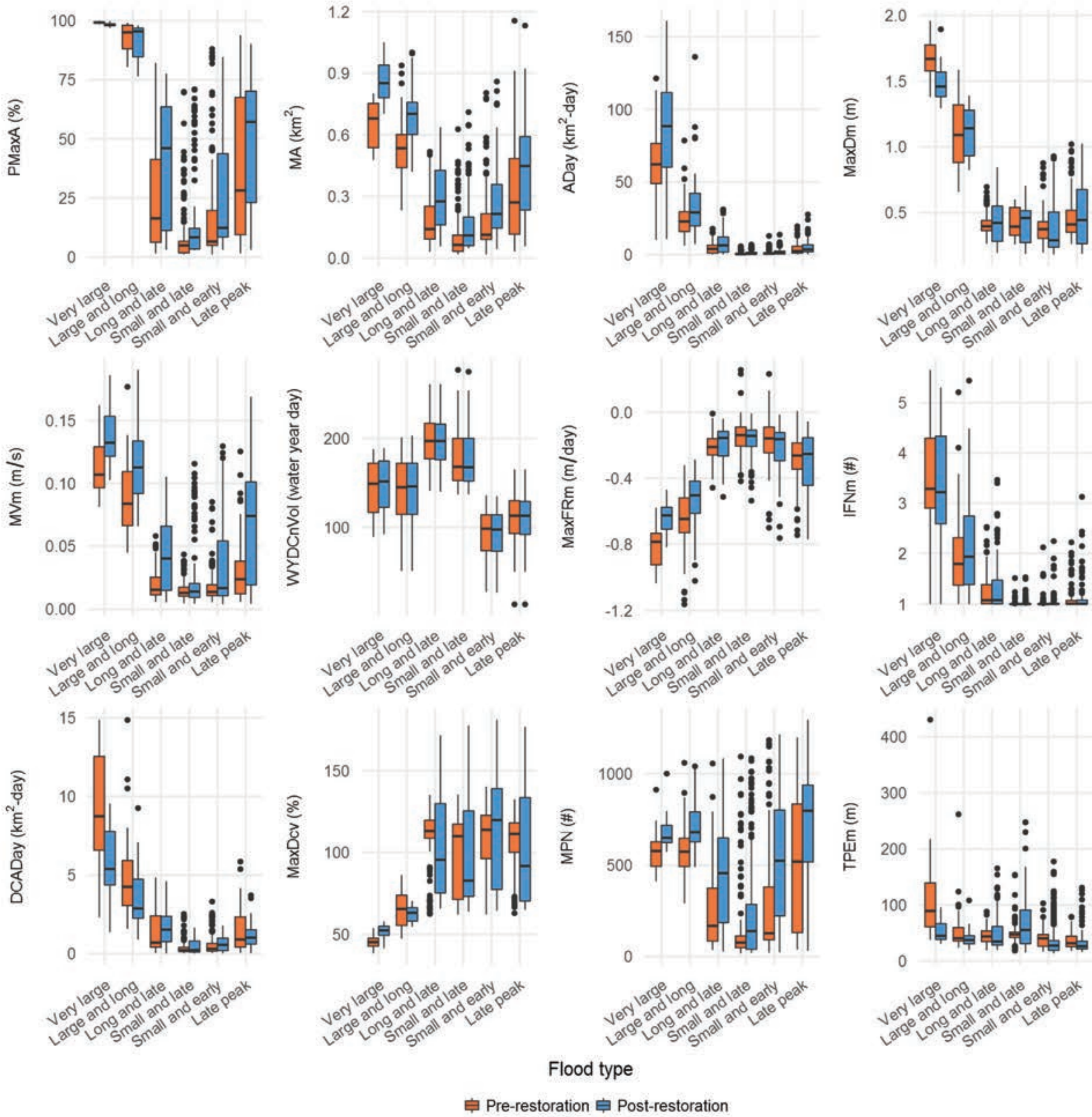

Figure 4-22. Summary statistics at the event scale for each of six Cosumnes River flood types (flood types established by Whipple et al. (2017)) for each of the 12 selected priority hydrospatial metrics, for pre- and post-restoration conditions.

addresses recent calls for advancing the use of hydraulic metrics in planning and evaluating river restoration alongside commonly applied hydrologic metrics (Bond et al., 2014; Brewer et al., 2016; Kozak et al., 2016). This research presents the evaluation of a range of physical metrics associated with different ecological implications. Metrics were selected based on their use in hydrology and landscape ecology, and for their ability to represent lateral extensions of natural flow regime components. Flow regime components of magnitude, duration, timing, frequency and rate of change were quantified spatially as well as temporally. This included evaluation of depth and velocity, measures available through hydrodynamic modeling. Analysis of the spatial dimension also allowed consideration of connectivity and heterogeneity,

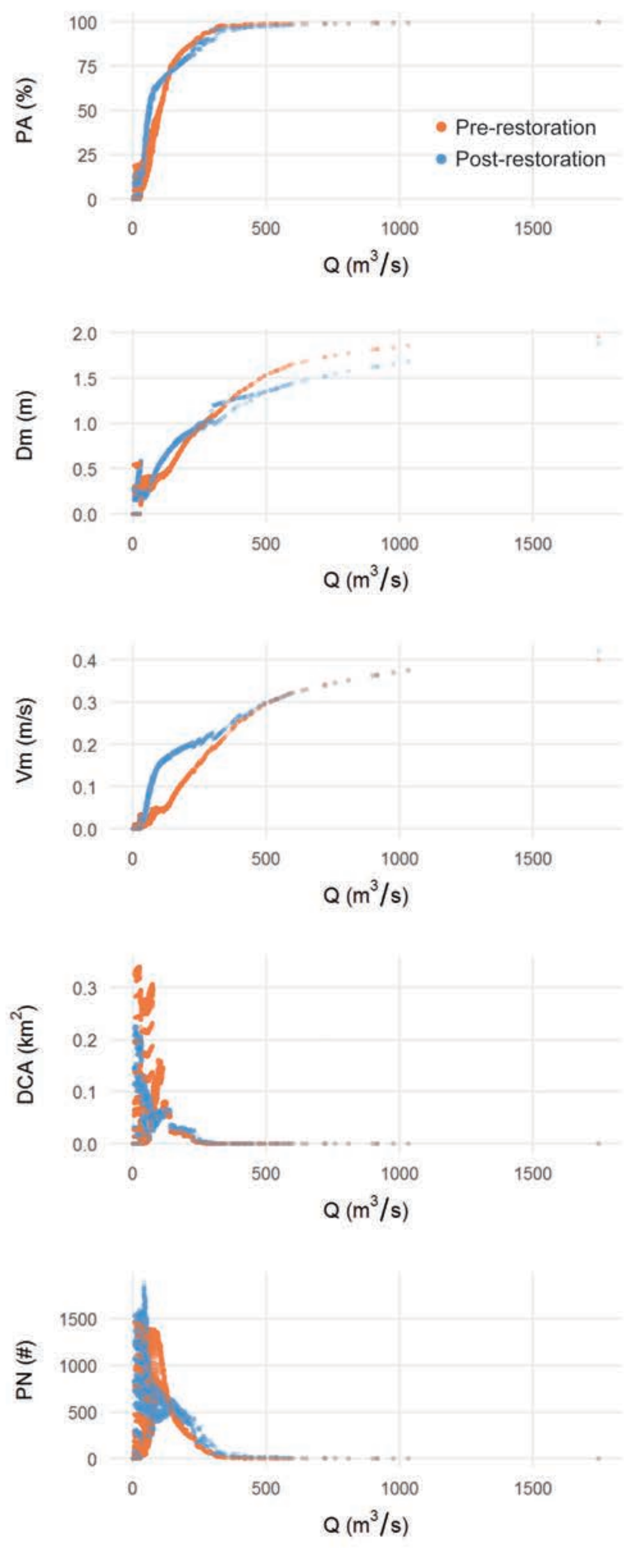

Figure 4-23. Relationship of five hydrospatial metrics with daily flow for pre- and post-restoration conditions.

which are well-established drivers of riverine and floodplain processes and functions (Amoros & Bornette, 2002; Junk et al., 1989; Tockner et al., 2002; Ward, 1989). Improved characterization of the natural spatiotemporal variability of floodplain inundation patterns – the goal of this research – facilitates the determination of ecological responses to human modifications of flows and the landscapes with which they interact.

Restoration responses

Examination of floodplain inundation patterns showed that responses to levee removal restoration varied in magnitude and direction of change over space and time and differed depending on the metric considered (Table 4-7). For example, area inundated summed over time (*ADay*), velocity (*MVm*), and patch number (*MPN*) showed strong positive responses to the levee-removal restoration, duration (*Durm*) and patch edge (*TPEm*) negatively responded, and depth (*MaxDm*) and several heterogeneity metrics (e.g., *MPSm*, *MIMaxD*) showed relatively little response. While many metrics were highly related to flow magnitude, relationships tended to be non-linear, not 1:1, and/or involve thresholds (see Figure 4-23). Results are similar to multivariate modeling results of Turner and Stewardson (2014), who found weak relationships to flow for some hydraulic measures and concluded that hydrologic measures were insufficient for predicting important ecologically-relevant conditions, such as depth or duration. Spatiotemporal inundation patterns are a manifestation of flow regime mediated by the floodplain landscape, and failure to consider these interactions when planning and evaluating restoration actions may result in missed opportunities or unintended consequences. The hydrospatial analysis approach provides some quantitative measures to tackle this challenge.

Table 4-7. General summary of Cosumnes River floodplain responses to levee removal.

Inundation patterns are a manifestation of flow regime mediated by floodplain landscape.
Magnitude and direction of change differs across metrics.
Magnitude and direction of change differs in space and time.
Inundation extent and frequency increases overall, primarily related to increases at intermediate flood flows most common in the spring months.
Impact of extremes (high flood flows, dry years) is dampened overall.
Centroid volume of flood inundation occurs earlier in the season.
Metrics of spatial heterogeneity generally increase.
Changes are most substantial closest to the main levee removal.
Tradeoffs occur for metrics such as disconnectivity and duration in the spatial distribution of change.
Interannual variability declines for some metrics (e.g., inundated area), but increases for others (e.g., connectivity).
Relationships to flow magnitude are non-linear, express thresholds, and are not 1:1.
At extreme high flood flows, inundated area declines.

Levee removal restoration produced the intended and expected consequence of increasing the frequency and extent of floodplain inundation. Research findings provided new information concerning the nature of the increased floodplain connectivity. Overall, the most substantial gains in extent and duration of inundation occurred with the more frequent, intermediate flood flows. At high flows (approximately 135-300 m^3/s, 1.7-3.2 year recurrence interval), the percentage of floodplain area inundated pre-restoration exceeded that of post-restoration conditions. Greater gains at intermediate flows are also related to the fact that extremely high flows already inundated most of the floodplain prior to restoration. Results indicate that magnitude-related extremes lessened in response to restoration, both in terms of extreme lows (dry years and small events) and highs (very wet years or large floods), as suggested by negative deviation factors for coefficients of dispersion (see Table 4-4). For example, maximum area inundated (*PMaxA*) declines somewhat with levee removal, and spatial mean depth (*Dm*) is lower at high flows. This is likely related to a number of factors including lower overall water surface elevation with larger areas of the floodplain available to inundate, the expansion of more shallowly inundated areas with levee-removal (bringing down spatially-averaged depth), as well as less area inundated at high flows in the northern floodplain forest (see Figure 4-4, Figure 4-6, Figure 4-14, and Figure 4-23). At the other end of the spectrum, the number of years with extremely low area-days (*ADay*) declined post-restoration (see Figure 4-4 and Figure 4-7). Results show that the seasonal timing of the centroid volume of flows shifted slightly later (by several days), suggesting that changes to landscape configuration can change temporal aspects of inundation.

Comparison to other research

The research presented here builds on other recent efforts to quantify spatial and temporal aspects of flood regimes. Stone et al. (2017) similarly used 2D hydrodynamic modeling to estimate spatially resolved hydraulic variables for a multi-decadal daily flow time series. This study follows Stone et al. (2017)

in taking advantage of 2D hydrodynamic modeling benefits to determine spatially-resolved hydraulic properties, without requiring modeling the full period of record. Modeling by Stone et al. (2017) was in steady state, as opposed to the unsteady modeling used here to address ponded areas.

Several studies have established relationships between flow and a particular metric of interest, such as connectivity or area meeting particular criteria (e.g., habitat suitability), using hydrodynamic modeling. A particularly applicable series of studies examined variable hydrologic connectivity across different floodplain features over time using results from 1D and 2D hydrodynamic modeling (Karim et al., 2015; Karim et al., 2012; Karim et al., 2013; Karim et al., 2016). They showed that degree and variability in connectivity is related to landscape position and topography. In another study, floodplain inundation regimes were defined using inundation extent and duration to examine the influence of morphology, climate, and flow alteration (Cienciala & Pasternack, 2017). Studies comparing the regulated Missouri River and unregulated Yellowstone River used 2D hydrodynamic modeling to quantify several physical metrics related to habitat and evaluate changes over time. Bowen et al. (2003) examined patch dynamics of shallow water and low velocity areas to determine their implications for riparian vegetation. And, a spatiotemporal analysis of Erwin et al. (2017) compared area meeting velocity and depth criteria for pallid sturgeon under altered and unaltered flow regimes in channelized and unchannelized reaches.

Few studies have quantified aspects of spatial heterogeneity and connectivity in riverine landscapes using metrics derived from landscape ecology. Two decades ago, Cooper et al. (1997) highlighted the need for multiple measures of spatial heterogeneity applied to riverine environments. Studies since have involved examinations of patch dynamics including quantification of differences in areas with slow current between regulated and unregulated systems (Bowen et al., 2003), comparison of habitat availability between natural and artificial river reaches (Le Pichon et al., 2009), the dynamics of shallow water habitat (Jacobson & Galat, 2006), and characterization of spatiotemporal variability of flowing and dry patches in intermittent rivers (Datry et al., 2016). The patch metrics applied in this study, including patch size, patch number, and edge length, could be used similarly to these studies to evaluate specific conditions of ecological relevance. The evaluation of connected and disconnected inundated area was preceded by several studies that measured wetland-river connectivity. Hudson et al. (2012) showed that connectivity characteristics, or "connectivity signatures," varied across a floodplain, with some features, such as distant oxbow lakes, poorly represented by hydrologic measures of in-channel flow. A series of studies on Australian floodplain wetlands illustrate the importance of accounting for spatially- as well as temporally-variable connectivity (Karim et al., 2015; Karim et al., 2012; Karim et al., 2016). Carbonneau et al. (2012) applied remote sensing methods to link hydraulic properties to connectivity and heterogeneity across large areas with relatively high resolution. Similar to the multi-metric approach here, a spatiotemporally-resolved examination of a variety of landscape metrics was developed as a GIS-based decision support tool for restoration planning in the lower Columbia River estuary (Coleman et al., 2015). These studies generally illustrate the important role of the floodplain landscape (i.e., topography) in mediating floodplain conditions, an overall outcome of this analysis. Overall, this area of research supports the conclusion that the spatial context of flood regimes should be accounted for when assessing physical and ecological impacts of restoration.

Limitations

Limitations to this research fall into several categories. Concerning hydrodynamic modeling, uncertainties arise from factors such as representations of model inflow and outflow boundary conditions, computational grid and DEM resolution, cross-sectional representation of channel geometry, and the handling of infiltration and evaporation (or general lack thereof, in the case of this research). The issues of infiltration and evaporation are a particular area of uncertainty in the analysis here because ponded (disconnected) patches were assumed to go dry (due to infiltration and evaporation) after seven days, though this varies substantially depending on the time of year, depth of inundation, antecedent moisture conditions, etc. Typical hydrodynamic modeling also does not address sediment dynamics and the landscape evolution that results as a flood regime interacts with its floodplain. While different topographic scenarios can be modeled to represent an evolving landscape through time, these changes are not typically included in dynamic modeling. Similarly, changing vegetation patterns (and therefore roughness) resulting from floodplain sediment dynamics are not included in the model. Thus, the research here should be viewed as a scenario analysis examining effects of changing only the topographic elements of levee-setback restoration, holding other factors constant.

Though unsteady flow simulation allowed for estimation of ponded area, some advantages of unsteady hydrodynamic modeling are lost. Namely, flood wave propagation across the floodplain during events are not accounted for as this would require complete modeling of the time series to be analyzed, which is computationally prohibitive in most cases. Hydrodynamic modeling of depth and velocity at known flows are used in piece-wise linear interpolation to evaluate the full time series, which means that the flows selected for modeling and the intervals between them influence the outcome. For example, if a particularly large area of a floodplain inundates at a flow that falls in the middle of a large interval between modeled flows, that inundation threshold will not be captured (i.e., depth for a cell that is dry at one flow and 1 m deep at another flow will be linearly interpolated for the flows in between). Also, estimations of spatially-resolved depth and velocity are made using the daily flow time series, which means elimination of sub-daily flow fluctuations, including instantaneous peak flows. A better understanding of uncertainty in this estimation could be made by comparing metrics for paired floods using daily and sub-daily time steps.

The selected metrics presented here represent a wide range of ecologically relevant conditions. However, many metrics are highly correlated (Olden & Poff, 2003). Further efforts to reduce redundancy could lead to a more limited or different set of metrics to represent a hydrospatial regime. However, ecological relevance and interpretability are important in metric selection. Along these lines, additional metrics could be developed and calculated, and some may be more appropriate for some applications. For instance, metrics for antecedent conditions may be useful in relation to soil moisture affecting flood duration and extent (Powell et al., 2008). Potentially useful metrics applied in other studies include mass flux (Stone et al., 2017), Froude number (Turner & Stewardson, 2014), and proportion of flow (Chen et al., 2015). Computational limitations constrained selection of metrics in this analysis, such as metrics of spatially-resolved flowpath distance and spatial autocorrelation at a range of spatial resolutions. With improvements of computational speeds and handling of large datasets, complex metrics (e.g., metrics requiring spatial neighborhood analysis on daily time step) will become more feasible.

While this approach improves the spatiotemporal resolution of important physical characteristics that affect the form and function of floodplain ecosystems, many other factors driving ecological functions are not considered. Specifically, the approach could be extended to consider water quality metrics relating to temperature and turbidity, as long as adequate field data or modeling output were available. Factors such a biotic controls and internal food web dynamics are less readily applied to this approach, but should be considered alongside hydrospatial analysis for restoration planning and evaluation. To advance the ecological utility of the hydrospatial analysis approach presented here, metrics and results should be validated through ecological monitoring.

Application to management

Quantifying physical responses of riverine-floodplain systems to anthropogenic modifications and to restoration actions offers information critical to understanding implications for freshwater-dependent ecosystems. Earlier research has been devoted to such quantification, typically a form of flow regime classification and change analysis or flow-habitat relationships for particular target species. Recognizing that landscape-mediated flood regimes drive floodplain ecosystems, this research provides spatiotemporal evaluation of floodplain inundation patterns relating to ecologically-relevant factors including extent, depth, velocity, duration, timing, connectivity, and heterogeneity. Following the natural flow regime paradigm, managing for the natural hydrospatial regime is expected to support river-floodplain ecosystems. This approach improves understanding of floodplain dynamics resulting from the interaction of flow regime and landscape configuration and topography. As articulated by Amoros and Bornette (2002), floodplain restoration should reinstate hydro-geomorphic dynamics to encourage spatiotemporal variability. The hydrospatial analysis approach helps manage for heterogeneity, especially important in an era of global change, where overly prescriptive approaches may limit options (Hiers et al., 2016; Schindler & Hilborn, 2015). In this approach, a range of metrics selected to represent the hydrospatial regime are evaluated to support management strategies for spatially and temporally variable physical conditions that together drive ecosystem processes and functions.

Readily applied to scenario analysis, the hydrospatial analysis approach can be used as a 'state-of-the-system' analysis step. Results can then be applied in the process of evaluating effects of environmental flow alternatives (or changes to flow regime under climate change), or physical habitat restoration (as in the levee-removal restoration evaluated here), or both together. Although the reliance on 2D hydrodynamic modeling may limit applications depending on scale, data availability, and cost, restoration planning increasingly involves hydrodynamic modeling to evaluate potential project impacts. Once hydrodynamic modeling is complete, subsequent application of the hydrospatial analysis approach is relatively straightforward, providing useful and rarely-available spatially and temporally resolved input for decision-making tools that could evaluate specific pre-determined targets. Subsequent analyses also are possible, such as estimating nutrient exchange and carbon production or quantifying physical habitat requirements of native floodplain fishes. Hydrospatial methods could be adapted for use at larger spatial scales where 2D hydrodynamic modeling is computationally prohibitive, such as quantifying environmental flow impacts at the river scale (i.e., 10s of kilometers). Such adaptations could build on other forms of spatial analysis of floodplain inundation (e.g., Hermoso et al., 2012; Jacobson et al., 2011; Peake et al., 2011). While quantification would necessarily be at a coarser spatial scale and certain metrics, such as those involving velocity, would not be possible, insights would likely be gained through multi-metric spatiotemporal floodplain inundation pattern analysis. Importantly, scaling up would allow for evaluation of how larger riverine landscapes can serve different functions in different locations depending on the flood and landscape configuration.

The approach presented here is intended for managing highly modified flow regimes and landscapes common to most large lowland rivers. Achieving ecological goals may not be possible through environmental flows or habitat restoration alone (see Chapter 2; Arthington et al., 2010; Kondolf, 2011; Mount et al., 2017; Palmer et al., 2014; Wohl et al., 2015). Rather than restoration actions targeting elements of a natural flow regime or physically altering channel and/or floodplain to mimic reference conditions (which may or may not produce needed physical processes and dynamics within the floodplain), this analytical approach supports tailoring of restoration actions to influence the hydrospatial regime. Developing strategies in this way should better support natural flooding processes and dynamics and thus be more directly tied to ecosystem processes and functions. In highly modified systems, this may mean that flows and/or landscapes most conducive for target hydrospatial conditions may not resemble prior conditions, but nevertheless support natural functions and ecological diversity and productivity. The analytical methods presented here also can help define potential unintended consequences of actions, such as introducing infrastructure to engineer floodplain inundation in the absence of natural connectivity (Bond et al., 2014) or environmental flow rules that do not consider changes within floodplains (Stone et al., 2017). A hydrospatial analysis approach can therefore inform needed compromises in managing water resources and riverine landscapes, encompassed by the concept of reconciliation ecology where ecosystem resilience is encouraged while also meeting human demands (Dudgeon et al., 2006; Moyle, 2013). This research provides a tool to address questions on what restoration actions may be most beneficial given limitations from human interference (e.g., upstream dams, channelization, and climate change).

This study directly informs current restoration efforts along the lower Cosumnes River, California. Overall, research findings refine restoration expectations, including quantifying spatial and temporal gains in inundation extent and duration, defining changes in frequencies of particular conditions, highlighting the role of intermediate flood flows in affecting restoration outcomes, identifying tradeoffs in where ponding occurs, and revealing opposing shifts in depth, duration, and inundation frequency depending on landscape position. The levee removal caused variable responses across the floodplain and over time, suggesting that the configuration and location of physical restoration can help target dynamic and heterogeneous floodplain conditions to support an overall restoration outcome.

Geomorphic and ecological implications can be inferred from these results. Greater sediment movement and transport further into the floodplain is expected with the higher average velocities, along with scour at the main levee breach location. Following geomorphic change, regeneration of riparian vegetation is expected particularly in the proximity of breach locations due to coarse sediment deposition and hydrochorous dispersal. Despite declines in water year maximum inundated area, increases in mean inundated area and area-day metrics and disproportionate gains in late winter and spring suggest greater capacity for in situ primary and secondary production (Ahearn et al., 2006; Grosholz & Gallo, 2006). In turn, more productivity increases food availability for higher trophic organisms and cascading downstream benefits from nutrient export. The overall slight increase in the number of inundation events annually indicates greater potential productivity and export downstream (Grosholz & Gallo, 2006). Increased floodplain inundation increases habitat availability for native fish populations (Jeffres et al., 2008; Junk et al., 1989; Welcomme, 1979). Hydrospatial results also show fewer years of extremely low extent and duration of inundation, lessening the severity of drought years on floodplain inundation-dependent processes. Particular locations in the floodplain where increased inundation is likely are around the main levee breach and interior ring levee. Inundation patterns are also generally more heterogeneous after restoration, which could indicate divergent properties across different patches leading to more diverse biological communities (Tockner et al., 2000; Wiens, 2002).

CONCLUSIONS

Flood regime and floodplain morphology together generate the dynamics driving diverse and productive floodplain ecosystems. Humans have fundamentally altered both flows and landscapes, causing extensive degradation of freshwater-dependent ecosystems. Stemming the impacts of human-induced change requires better support for productive and resilient ecosystems through reinstating dynamic physical processes that promote spatially and temporally variable conditions. Understanding spatiotemporal patterns of inundation is fundamental to developing such strategies. The multi-metric and multi-dimensional hydrospatial analysis approach presented here quantifies how altered flows and/or landscapes affect physical conditions of floodplain inundation. Using output from 2D hydrodynamic modeling as the basis for hydrospatial analysis, inundation metrics relating to extent, depth, flow velocity, timing, duration, rate of change, and frequency, as well as spatial elements of connectivity and heterogeneity are evaluated. This approach, demonstrated by comparing pre- and post-restoration conditions along the Cosumnes River floodplain, California, is an important step linking restoration actions to their ecological impacts. The largely unregulated flow regime of the Cosumnes River combined with efforts over the last several decades to reconnect the river and its floodplain through levee-breach restoration offers a unique opportunityli to quantify river-floodplain interaction and effects of restoration on spatiotemporal floodplain inundation patterns. Hydrospatial analysis shows different responses to levee-removal restoration depending on flow, location within the floodplain, and metric evaluated, demonstrating the utility of this new quantitative method for evaluating flow dynamics over space and time. This study informs efforts to reestablish dynamic and heterogeneous riverine landscapes to support functional ecosystems along the lower Cosumnes River, elsewhere in California's Central Valley, and in other large highly modified river systems globally.

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CHAPTER 5

QUANTIFYING SPATIOTEMPORAL HABITAT BENEFITS OF FLOODPLAIN RESTORATION: APPLICATION TO SACRAMENTO SPLITTAIL OF THE COSUMNES RIVER, CALIFORNIA

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Quantifying spatiotemporal habitat benefits of floodplain restoration: Application to Sacramento splittail of the Cosumnes River, California

ABSTRACT

Large lowland rivers are heavily regulated and modified, muting spatial and temporal dynamics necessary to maintain riverine-floodplain ecosystem processes and functions. Effective and efficient ecosystem restoration requires improved understanding of spatiotemporal floodplain habitat patterns reflecting the interaction of flow regime and landscape, or the hydrospatial regime. This study presents a flexible approach for floodplain habitat quantification and applies it to a recent levee-removal floodplain restoration project along the unregulated but highly-modified lower Cosumnes River of California's Central Valley. Two-dimensional hydrodynamic modeling output and habitat suitability indices linked physical conditions and ecological response. The hydrospatial approach retains spatial and temporal resolution throughout the analysis, allowing for the consideration of complex floodplain interactions, such as inundation duration and connectivity, in determining habitat quality. Daily grid-based habitat suitability was calculated over the 110-year Cosumnes daily flow record for a native floodplain fish species, the Sacramento splittail (*Pogonichthys macrolepodotis*), which was then summarized as habitat area over space and time. Results show that levee-removal restoration nearly doubled overall habitat availability, though available habitat varies considerably in space and time. Results also indicate that flows alone may not adequately predict floodplain habitat availability, suggesting that commonly-applied flow-habitat relationships may be insufficient for complex floodplains. Broadly, this study advances floodplain habitat quantification methods, improves understanding of spatiotemporal variability and flow-landscape interaction that drives physical and ecological processes, and informs the development of management strategies to better support functional and resilient floodplain ecosystems into the future.

INTRODUCTION

Human modifications of large rivers and their floodplains have led to profound degradation of some of the world's most productive and biodiverse ecosystems (Dudgeon et al., 2006; Tockner & Stanford, 2002). Floodplains are naturally variable and complex environments, generating high productivity that fuels food webs (Junk et al., 1989; Ward et al., 2002). Reducing or eliminating that variability is often a focus of water management and engineering activities (Arthington & Balcombe, 2011; Poff et al., 2007). Flow regime change due to dams and diversions as well as land use change, including levee-building that severs rivers from their floodplains, have eliminated vital habitats and fundamentally altered the processes that create and maintain the dynamic and complex environments to which species are adapted (Lytle & Poff, 2004; Opperman et al., 2010; Poff et al., 1997). This recognition has led to efforts to reconnect floodplains to their adjacent rivers to support freshwater-dependent species and overall ecosystem resilience in an era of anthropogenic change (Beechie et al., 2013; Opperman et al., 2009).

Functional floodplains are defined by their provision of essential spawning and rearing habitat for fish, maintenance of riparian vegetation and successional processes, and overall ecological productivity and diversity (Opperman et al., 2010). Following gradients in hydrologic connectivity and residence time, community composition and diversity of floodplain food webs vary spatially and temporally (Simões et al., 2013; Tockner et al., 2000). Physical habitat diversity creates a range of ecological niches, which structures fish distributions and translates to high biodiversity (Górski et al., 2013; Kobayashi et al., 2014). In what has been described as a freshwater productivity pump (Ahearn et al., 2006), floodplain inundation prompts an initial expansion of primary productivity (phytoplankton), followed by growth in primary consumers (zooplankton and macroinvertebrates) that provide valuable food resources for higher trophic levels, such as fish (Bellmore et al., 2012; Gallardo et al., 2009; Grosholz & Gallo, 2006; Sommer et al., 2004). These cycles are reset by subsequent flood pulses causing reconnection that serves to export productivity not utilized within the floodplain to ecosystems downstream (Bunn et al., 2006; Lehman et al., 2008). Spatial patterns of connectivity created as flood pulses interact with floodplain topography therefore influence this productivity cycle and support higher trophic levels.

The ecological significance of floodplains within the Central Valley of California is increasingly recognized, with floodplain reconnection a primary goal of large-scale restoration efforts (Davis et al., 2017; Delta Stewardship Council, 2013). The once vast mosaics of floodplain and wetland habitats have been transformed almost completely – a nearly 95% loss – over the last 150 years into one of the most productive agricultural regions globally (The Bay Institute, 1998; Whipple et al., 2012). An immense water management infrastructure of dams, diversions, aqueducts, and levees as well as groundwater pumping supports the agricultural economy and nearly two thirds of California's population, but has drastically reduced the amount and altered the timing of water available for ecosystems (Hanak et al., 2011). One response to these and other changes has been a precipitous decline in native fish. Over 80% of California's native freshwater fishes are threatened or extinct (Moyle et al., 2011). The most notable are the iconic anadromous Chinook salmon (*Oncorhynchus tshawytscha*) and steelhead (*Oncorhynchus mykiss*), which use floodplains as juvenile rearing habitat. Species once common in the slow moving waters of floodplains and dynamic estuarine environments are particularly threatened (Moyle et al., 2010), marked by the extinction of the endemic thicktail chub (*Gila crassicauda*), extirpation of Sacramento perch (*Archoplites interruptus*), and precipitous population declines of endangered Delta smelt (*Hypomesus transpacificus*). The Sacramento splittail (*Pogonichthys macrolepidotus*), a native floodplain fish species, is similarly at risk due to changes in habitat availability and was formally listed as threatened under the Endangered Species Act (Sommer et al., 2007). The aquatic ecosystems of the San Francisco Bay-Delta estuary and the Sacramento and San Joaquin Rivers that flow into it are now considered to be novel by some (Moyle, 2013), yet opportunities exist to encourage natural processes that better support native species and ecological resilience to future change. Recent research has demonstrated potential benefits of process-based restoration through levee setbacks as well as managed floodplain inundation of agricultural land (Florsheim & Mount, 2002; Jeffres et al., 2008; Katz et al., 2017; Sommer et al., 2001). Improvements hinge on the successful co-implementation of changes to the physical landscape as well as environmental flows to achieve habitat availability and variability important for native species (see Chapter 2; Moyle et al., 2010; Opperman et al., 2010).

Habitat loss, along with water quality impairment and altered food webs, is a primary driver of ecosystem degradation (Malmqvist & Rundle, 2002), and habitat quantification helps link hydrologic and physical landscape alteration and ecosystem response. Physical habitat simulation has long been a cornerstone of stream and river restoration, particularly as it relates to establishing instream or environmental flow requirements to support salmonid fisheries (Dunbar et al., 2012; Huckstorf et al., 2008; Petts, 2009). The conventional approach is to establish relationships between in-channel habitat for key species and flow (i.e., habitat as a function of discharge), where suitability indices are used to connect physical conditions to ecological requirements. Threatened and endangered fish species are usually the focus, as they tend to drive restoration and conservation efforts (Dunbar et al., 2012). The oldest and most commonly-applied methods are those of the instream flow incremental methodology (IFIM) and associated physical habitat simulation (PHABSIM), which assign suitability indices based on hydraulic properties that are then used to compute a weighted usable area (WUA) and hydraulic habitat suitability (HHS) associated with different flows (Bovee, 1982; Maddock, 1999; Stalnaker, 1979). Methods have primarily been applied using one-dimensional (1D) hydrodynamic models, with two-dimensional (2D) hydrodynamic modeling increasingly common for its capacity to better account for spatial complexities (Leclerc et al., 1995; Pasternack et al., 2004). Studies evaluating floodplain habitat for restoration and conservation are still rare compared to in-channel assessments (though see Erwin et al., 2017; Matella & Jagt, 2014; van de Wolfshaar et al., 2010), despite the important functions floodplains serve in large-river ecosystems.

Within floodplain landscapes, flood characteristics such as magnitude and duration are mediated by floodplain landforms, which is reflected in the spatiotemporal availability and connectivity of habitat (Moyle et al., 2010; Tockner et al., 2000; Ward et al., 2002). Available floodplain habitat depends on how hydraulic variables change over space and time in relationship to the flood hydrograph; a relationship which is often non-linear and spatially variable (Dyer & Thoms, 2006; Guse et al., 2015). Moving beyond in-channel assessments of habitat therefore requires knowledge of connectivity patterns and spatial variability in flood dynamics (Stone et al., 2017). However, flow and physical habitat restoration objectives are often set without consideration of spatial variability or a clear understanding of how proposed changes may affect conditions across the floodplain environment, limiting the capacity to link restoration actions to specific ecological benefits.

Understanding the ecological ramifications of altered riverscapes (sensu Fausch et al., 2002) and mechanisms of change is essential to developing strategies for rehabilitating diverse and productive ecosystems, particularly within the context of highly managed and changing environments. In most highly altered systems, reinstating natural flow regime and river morphology is not possible, yet substantial capacity exists to adjust components of the hydrograph and the landscape with which it interacts to encourage dynamic processes that increase habitat availability, productivity, and ecological resilience to change (see Chapter 2; Kondolf et al., 2012; Stanford et al., 1996; Yarnell et al., 2015). For these systems, knowledge of hydrologic alteration or historical landscape configuration may be insufficient for developing effective restoration strategies. Determining beneficial restoration actions requires more comprehensive assessments of land-water interactions in space and time, and ecologically-relevant conditions that result (Jacobson & Galat, 2006).

The approach taken here addresses these issues by expanding physical habitat quantification to spatial and temporal aspects important for floodplains, including hydrologic connectivity, inundation duration, and sequences of flood events. The objectives of this study were to quantify floodplain habitat response to restoration and to describe and compare spatial and temporal habitat variability using hydrospatial analysis. This is demonstrated for splittail at a restoration site along the lower Cosumnes River floodplain, California. This chapter presents the application of a new approach for using spatiotemporal quantification of floodplain inundation patterns to evaluate habitat availability within riverscapes. In contrast to typical habitat simulation methods, available habitat is evaluated using spatially distributed depth and velocity from daily streamflow records. This accounts for relationships between flow and habitat not being one-to-one and allows explicit inclusion of spatially-resolved temporal patterns of hydrologic connectivity and inundation duration. Specifically, output from 2D hydrodynamic modeling is combined with a daily streamflow record to establish a time series of gridded floodplain inundation depth and velocity, to which habitat suitability criteria relating to depth, velocity, connectivity, duration, and timing are applied. Study outcomes include time series of habitat availability and duration curves, but are distinguished from output of typical physical habitat quantification methods by their derivation from spatially- and temporally-resolved analysis (i.e., analysis of habitat suitability at a given location for each day). Furthermore, the spatial aspects of the analysis allow for additional visualizations of spatiotemporal distribution of habitat suitability. Research implications are discussed in terms of specific restoration benefits as well as improved understanding of ecological responses to floodplain hydrospatial regime alteration. Overall, this research seeks to advance evaluation of ecological implications of floodplain hydrologic and morphological modifications, addressing needs to reestablish dynamic processes within highly altered and managed landscapes of lowland alluvial rivers.

METHODS

Study area

Of the major rivers flowing into California's Central Valley, the Cosumnes River is unique for its largely unregulated hydrograph and connectivity with much of its former floodplain. The approximately 2,460 km^2 watershed extends from headwater elevations of 2,300 m to its confluence with the Mokelumne River at sea level (Figure 5-1). Hydrology reflects the Mediterranean-montane climate, with the majority of runoff occurring between December and May and distinguished by high inter- and intra-annual variability (Whipple et al., 2017). The peak flow of record is 2,630 m^3/s , while mean annual daily flow is 14 m^3/s. Historically, winter floods emerged from the foothills and entered a maze of distributary and anastomosing channels, inundating large expanses of wetlands and riparian forests for several months of the year (Whipple et al., 2012). Some backwater aquatic habitat persisted through the dry summer months, supported by historically high groundwater. Like other lowland Central Valley rivers, the Cosumnes River now flows in a highly incised and leveed channel with only larger floods inundating substantial portions of the former floodplain (Figure 5-2).

Along the river's most downstream reach within the Cosumnes River Preserve – which is managed by The Nature Conservancy and a consortium of agencies – natural and intentional levee breaches over the last three decades have encouraged river-floodplain reconnection and the reestablishment of floodplain processes (Florsheim & Mount, 2002; Swenson et al., 2012). This has recruited riparian vegetation, increased primary and secondary productivity, and provided valuable habitat for fish, including the native Sacramento splittail (Moyle et al., 2007; Ribeiro et al., 2004). The research presented here focuses on quantifying splittail habitat benefits for the most recent levee-removal restoration project (implemented in the fall of 2014), which involved two primary levee breaches, breaches within an interior "ring" levee, and the excavation of a swale to promote downstream drainage on a 2.1 km^2 floodplain site (see Figure 5-1; Nichols & Viers, 2017).

Hydrodynamic modeling

To predict spatially-resolved depth and velocity at a range of flows, 2D hydrodynamic modeling was performed for both pre- and post-restoration topographic configurations. Models were developed using the U.S. Army Corps of Engineers' Hydrologic Engineering Center River Analysis System (HEC-RAS 5.0; Brunner, 2016). For the unstructured computational mesh, topographic LiDAR data were combined with local field surveys of channel cross sections and areas altered by restoration activities (CDWR 2010). Input hydrology was derived from the USGS Michigan Bar streamflow gage, which has a continuous daily flow record dating back to the 1908 water year (110 years) and is located approximately

45 km upstream of the restoration site (MHB, #11335000; U.S. Geological Survey, 2017). The HEC-RAS 5.0 subgrid capability allowed for computational grid cells at a resolution of 110-4,200 m^2 with output at the resolution of the underlying topography (1 m^2). Model calibration involved iteratively adjusting channel and floodplain surface roughness (Manning's *n*) as well as weir coefficient parameters. Adjustments were made using observed water surface elevation at several in-channel and floodplain locations (see Chapter 4 and Appendix A for further information).

Unsteady flow simulations for pre- and post-restoration were used to generate estimates of inundation depth and flow velocity across the floodplain at selected flows on the rising and falling limb of a hydrograph. Specifically, simulations were performed for a long stepped hydrograph that spanned flows below floodplain inundation (10 m^3/s) to above the record peak daily flow (1,745 m^3/s). The steps consisted of the selected flows, set at increments of 10-100 m^3/s, which were extended for several days. The flow increments between steps were smaller where floodplain inundation extent changed rapidly as a function of flow. Modeling of both the rising and falling limb of the hydrograph allowed for inclusion of isolated inundation (i.e., ponding) that occurs after flood peaks, important for understanding spatiotemporal variability of connectivity and inundation duration.

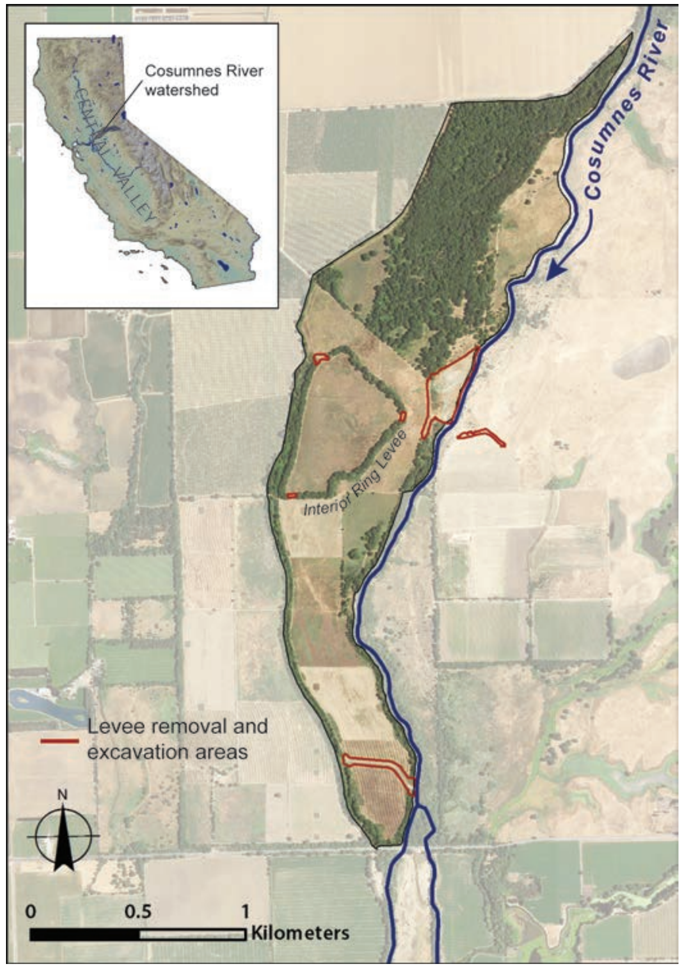

Figure 5-1. The lower Cosumnes River floodplain restoration site and primary levee removal and excavation areas (outlined in red). The location of the Cosumnes River watershed within the Central Valley of California is shown in the inset map. (Aerial imagery: USDA 2016)

Habitat suitability criteria

The Sacramento splittail was selected for this study because of its dependence on floodplains for spawning and rearing. Floodplain habitat loss is primarily responsible for dramatic declines in splittail populations (Moyle et al., 2007; Sommer et al., 2007), making the species a focus for conservation and restoration efforts and particularly suitable for evaluating floodplain restoration benefits (e.g., Cloern et al., 2011; Matella & Merenlender, 2015). Research has demonstrated promising potential for improved management of floodplains to support native fish through enhanced river-floodplain connectivity in response to flood pulses (Sommer et al., 2001; Sommer et al., 2014). Restoring floodplain inundation processes and patterns beneficial to splittail is expected to support overall function and resilience of floodplain ecosystems.

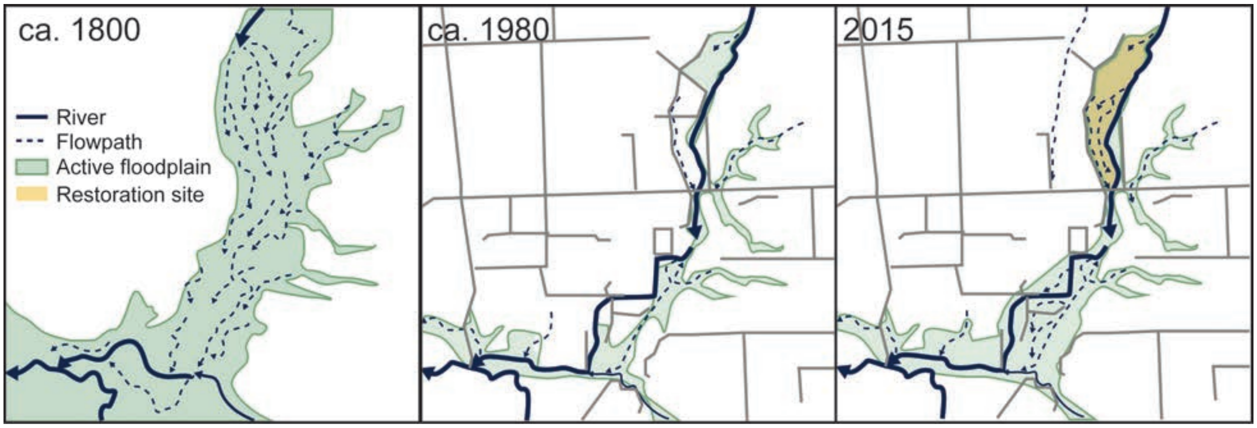

Figure 5-2. Conceptual illustration of transformation of lower Cosumnes River floodplain. The floodplain restoration site (yellow) was once part of a larger wetland landscape intersected by distributary and anastomosing channels, formerly known as the "Cosumnes Sink." Channelization, leveeing, farming activities, and groundwater pumping reduced the extent of regularly inundated floodplain. The restoration project of focus was implemented in the fall of 2014, representing the most recent of a series of process-based floodplain reconnection projects.

A cyprinid native to California, splittail migrate upstream from the San Francisco Bay-Delta estuary beginning in January to spawn in seasonally inundated floodplains along the Sacramento and San Joaquin Rivers and their tributaries, including the lower Cosumnes floodplain (Moyle, 2002; Moyle et al., 2004). Eggs adhere to submerged vegetation and require up to ten days of incubation, after which newly hatched fish remain in the floodplain for several weeks, preferring shallow depths (<2 m). Splittail are also affected by temperature, salinity, and dissolved oxygen, but generally tolerate the ranges typical in floodplains (Moyle et al., 2004). Longer periods of inundation meet life history requirements for spawning and rearing and also promote the production of food (zooplankton) for young fish (Grosholz & Gallo, 2006). These benefits have been shown by research finding superior health for juvenile splittail raised on floodplains compared to those raised in riverine environments (Ribeiro et al., 2004). Splittail longevity and fecundity suggest that populations are adapted to the highly variable climate and can persist through several years of drought with limited floodplain access (Moyle et al., 2004).

Splittail habitat requirements and preferences for depth, velocity, connectivity, duration, and timing were translated into habitat suitability indices (HSIs) based on published literature and expert opinion (Figure 5-3; see Appendix B; Moyle, 2017; Sommer, 2017; Suddeth, 2014). Separate relationships to depth and velocity were established for spawning and juvenile rearing conditions, as spawning splittail prefer overall greater depth and have lower preference for very low velocities in comparison to juvenile splittail (see Figure 5-3). Habitat suitability, assigned as an index between 0 (no value) to 1 (ideal conditions), is typically used in hydraulic habitat quantification studies to connect environmental conditions and ecological response (e.g., Benjankar et al., 2015; Dunbar et al., 2012; Guse et al., 2015). Depth and velocity are the most frequently used habitat parameters. Substrate and temperature are also used, but were not included in this analysis because, 1) information for establishing robust vegetation relationships was lacking and vegetation cover on the floodplain was fairly uniform in quality and, 2) water temperature is not a limiting factor for the season of interest. Connectivity, duration, and timing are not commonly used in habitat suitability analyses, but are included here for their spatial variability in floodplain environments and importance to floodplain ecological functions. Connectivity requirements were included in determining inundation duration rather than establishing separate suitability criteria, where inundated areas were considered potential habitat where a surface water connection to the river channel existed or where ponded areas reconnected to the river within seven days (an assumed period of infiltration and evapotranspiration, which was not accounted for in the hydrodynamic modeling). This effectively eliminates ponded areas that do not reconnect to the river.

Hydrospatial habitat analysis

Methods of this study build on the framework for hydrospatial analysis introduced in Chapter 4 and apply methods of grid-based habitat suitability modeling (e.g., Benjankar et al., 2015; Carnie et al., 2016; Guse et al., 2015; van de Wolfshaar et al., 2010; Yao et al., 2015). The habitat suitability criteria in Figure 5-3 were used to assign suitability index scores to spatially-resolved (gridded) daily estimates of depth and velocity derived from the 2D hydrodynamic modeling of the floodplain restoration site. Analysis was done for the 110-year MHB streamflow gage period of record (1908-2017 water years). Only days where flow exceeded the assigned floodplain inundation threshold of 23 m^3/s were included (Whipple et al., 2017), a total of 9,867 flood days across 545 flood events. The number of days exceeding this threshold per water year ranged from 0 (which occurred in 1933, 1961, 1976 and 1977) to 221 days (in 1983). Spatially-resolved piece-wise linear interpolation from the modeling output of depth and velocity at known flows was used to establish daily gridded (9 m^2, resampled from 1 m^2) estimates of depth and velocity for each flood day for both pre- and post-restoration topographic configurations (see Chapter 4). Suitability indices (values ranging from 0-1) for each habitat parameter (i.e., depth, velocity, connectivity, duration, and timing) were then assigned to each cell and for

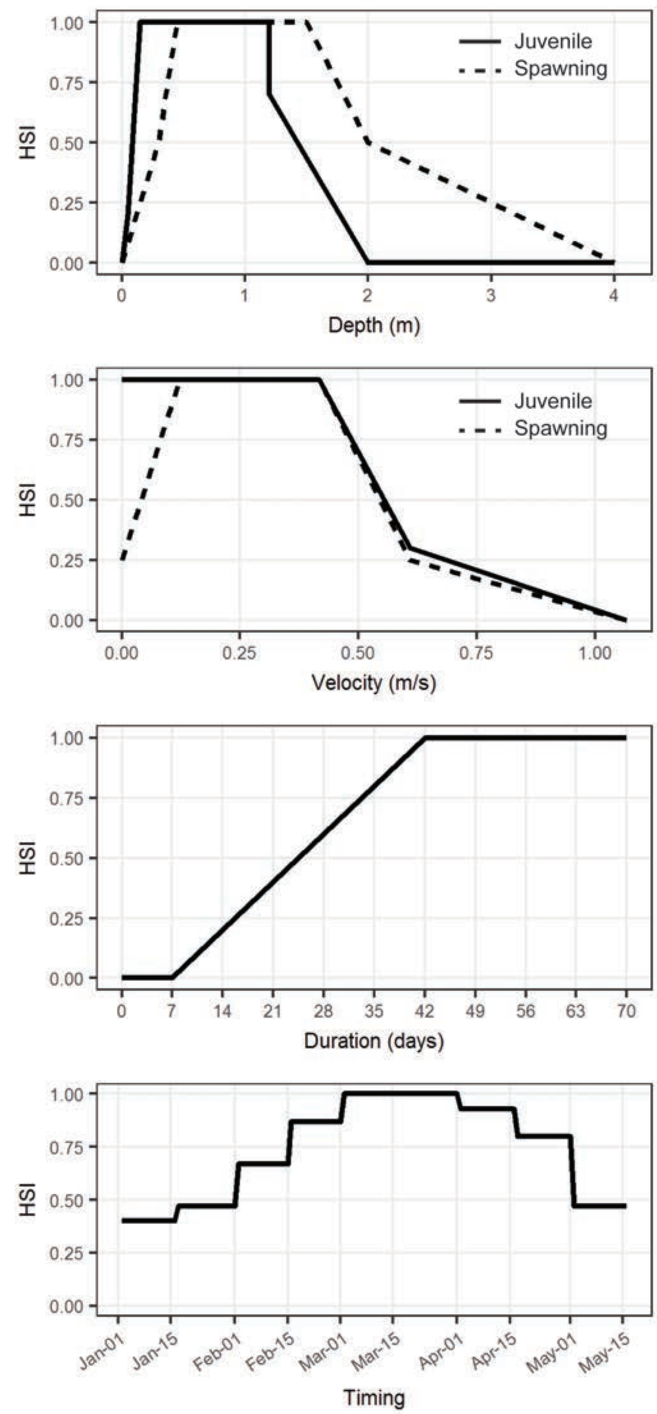

Figure 5-3. Sacramento splittail habitat suitability indices for physical conditions of depth, velocity, inundation duration, and seasonal timing. Depth and velocity are separated by life stage: spawning (dashed line) and juvenile (solid line), while the two life stages are combined for duration and timing. Indices were derived from existing literature and expert opinion (see Appendix B for further explanation).

each day. The steps used to develop daily distributions of overall habitat suitability, weighted usable area (WUA), and hydraulic habitat suitability (HHS) estimates are described in the following text and presented conceptually in Figure 5-4.

First, for each day in the analysis, binary grids indicating inundated cells connected to the river (1), disconnected cells (0), and dry cells (NA) were established from the depth dataset and patch analysis. Cell-by-cell and day-by-day, the total duration of inundation associated with each cell was determined, and then each cell was given the habitat suitability index (HSI) corresponding with that duration. This step also excluded ponded cells that did not reconnect to the river. For example, a cell that was inundated and connected to the river for four days and then ponded for two days before going dry would be assigned a duration value of four for the first four days only (i.e., the ponded days were not usable habitat due to disconnection). The daily gridded sets of depth and velocity were then assigned their corresponding HSIs (cell-by-cell and day-by-day). Timing was not spatially resolved, and therefore a single HSI for the whole floodplain area was assigned based on Julian day (see Figure 5-3).

The assigned daily gridded HSIs and single-value time series of timing HSI were then combined into cell suitability indices (CSIs) using the geometric mean. The geometric mean is often used in habitat suitability applications because a zero suitability for any individual HSI results in a zero combined HSI (e.g., Hanrahan et al., 2004; Tomsic et al., 2007). The CSI is defined in Benjankar et al. (2015) as:

$$CSI_i = (\Pi_j SI_{i,j})^{1/m} \qquad \text{(Eq. 1)}$$

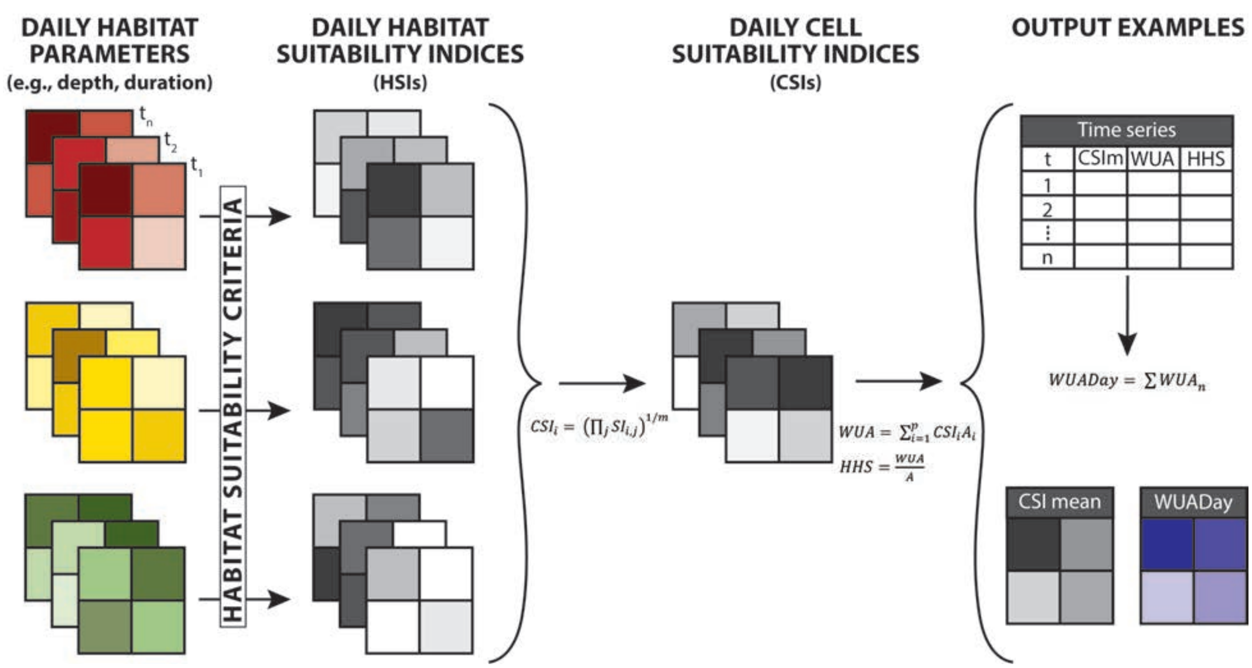

Figure 5-4. Conceptual illustration of the process to combine habitat suitability criteria with physical parameters into cell suitability indices (CSIs). The daily CSIs are used to determine output, including time series of spatial mean CSI (CSIm), weighted usable area (WUA), and hydraulic habitat suitability (HHS), as well spatial summaries of CSIs and WUADay (WUA summed over time).

where i represents each cell, j represents each suitability parameter, and m represents the total number of parameters. Subsequently, calculation of daily WUA and HHS were from commonly-applied equations (Benjankar et al., 2015; Guse et al., 2015; Stalnaker et al., 1995):

$$WUA = \sum_{i=1}^{p} CSI_i A_i \quad \text{(Eq. 2)}$$

and

$$HHS = WUA/A \quad \text{(Eq. 3)}$$

where, for each daily calculation, p represents the total number of inundated cells, A_i is the area of the cell, and A is the total inundated area. An additional metric, WUADay, was calculated, defined as the sum of daily WUA for a period of interest (e.g., water year or flood event; Hermoso et al., 2012).

These daily grids of cell suitability were then summarized in space and time following hydrospatial analysis summaries (see Chapter 4). Namely, output included daily time series (110 years, water years 1908-2017) of spatially-averaged habitat variables (spatial mean CSI, WUA, WUADay, and HHS), summaries at the flood event and water year scale, and spatially-resolved and temporally-resolved computations of habitat variables. This was performed for both the pre-restoration and post-restoration topographic configurations. For summary at the event scale, all events for the period of record were evaluated together, whereas summary at the water year scale involved evaluating conditions for each water year first before summarizing. Calculations were performed separately for splittail spawning and juvenile rearing requirements. Comparison between the pre- and post-restoration spawning and juvenile splittail habitat was conducted statistically and graphically. In addition to standard summary statistics, non-parametric measures including the coefficient of dispersion (CD), deviation factor (DF), and Wilcoxon rank sum test were used to analyze the data (Hollander et al., 2013; The Nature Conservancy, 2009). The CD is the difference between the 75th and 25th quantile divided by the median, and the DF is the difference between post- and pre-restoration habitat values divided by pre-restoration values. The relationship between habitat availability and flow regime was explored via comparisons of daily output as well as by grouping output by water year and flood types (see Chapter 3; Whipple et al., 2017). These were examined graphically and supported by the Kendall's rank correlation test and the pairwise Wilcoxon signed-rank test (Hollander et al., 2013). All scripts for this analysis used *R* (R Core Team, 2016), with the *raster* package providing the functions to handle and process the large spatial datasets (Hijmans, 2015).

RESULTS

Restoration habitat benefits to Sacramento splittail

The Cosumnes River floodplain levee removal restoration approximately doubled available Sacramento splittail habitat, as defined by suitability requirements for depth, velocity, duration, timing, and connectivity of inundation (Table 5-1). Deviation factors (DFs) – indicating habitat availability change between pre- and post-restoration conditions – for juvenile and spawning weighted useable area (WUA) and WUADay summarized at the water year and event scale ranged from 0.75 to 1.38 (where 1 indicates a doubling over pre-restoration conditions). The overall availability of splittail spawning habitat was 5-25% lower than that of rearing juveniles. On average, 8.8 km^2-d of juvenile habitat accumulated in a season under pre-restoration conditions, which increased to 17.2 km^2-d post-restoration (7.0-14.7 km^2-d for spawning splittail). Over the period of record, the maximum WUADay for a water year was 57 and 97

Table 5-1. Summary statistics for hydrospatial metrics at the water year scale (e.g., water year maximum weighted usable area, WUA) and at the flood event scale (e.g., maximum WUA per flood event). Metrics include spatial mean cell suitability index (CSIm), weighted usable area (WUA), WUA summed over time (WUADay), and hydraulic habitat suitability (HHS). The bootstrapped 95% confidence intervals (CI) for the median values are shown, as well as the coefficients of dispersion (CD; interquartile range divided by median). Deviation factors are computed for the medians and CDs of the pre- and post-restoration conditions. Results from the Wilcoxon signed-rank test is also shown (values significant at a 95% confidence level are bolded).

Summary scale	Life stage	Metric	Statistic	Pre-restoration				Post-restoration				Deviation factors		Wilcoxon signed-rank p-value
				Mean	Median	CI	CD	Mean	Median	CI	CD	Median	CD	
Water year	Juvenile	CSIm	Mean	0.085	0.081	(0.07-0.09)	0.68	0.118	0.120	(0.1-0.14)	0.77	0.47	0.14	<0.001
			Max	0.499	0.630	(0.59-0.74)	0.86	0.570	0.671	(0.65-0.73)	0.50	0.07	-0.42	<0.001
		WUA	Mean	0.039	0.033	(0.02-0.04)	1.22	0.071	0.059	(0.05-0.07)	1.35	0.80	0.11	<0.001
			Max	0.232	0.147	(0.1-0.2)	2.26	0.405	0.280	(0.1-0.39)	2.38	0.90	0.05	<0.001
		WUADay	Sum	8.811	5.315	(2.62-8.25)	2.24	17.150	9.294	(3.83-14.44)	2.48	0.75	0.11	<0.001
		HHS	Mean	0.159	0.150	(0.13-0.17)	0.85	0.210	0.224	(0.21-0.24)	0.68	0.49	-0.20	<0.001
			Max	0.533	0.688	(0.64-0.8)	0.84	0.589	0.699	(0.66-0.75)	0.50	0.02	-0.40	<0.001
	Spawning	CSIm	Mean	0.058	0.056	(0.05-0.06)	0.70	0.087	0.092	(0.08-0.11)	0.82	0.63	0.17	<0.001
			Max	0.327	0.413	(0.39-0.49)	0.87	0.425	0.467	(0.41-0.51)	0.67	0.13	-0.23	<0.001
		WUA	Mean	0.030	0.023	(0.02-0.03)	1.35	0.059	0.046	(0.04-0.06)	1.45	1.03	0.07	<0.001
			Max	0.221	0.115	(0.06-0.14)	2.50	0.403	0.273	(0.08-0.39)	2.43	1.38	-0.03	<0.001
		WUADay	Sum	6.979	3.820	(1.94-5.96)	2.55	14.679	6.886	(1.63-10.59)	2.85	0.80	0.12	<0.001
		HHS	Mean	0.105	0.097	(0.08-0.11)	0.92	0.147	0.159	(0.14-0.18)	0.67	0.64	-0.27	<0.001
			Max	0.348	0.450	(0.43-0.52)	0.87	0.436	0.481	(0.42-0.51)	0.64	0.07	-0.26	<0.001
Event	Juvenile	CSIm	Mean	0.092	0.069	(0.06-0.08)	1.50	0.133	0.082	(0.07-0.09)	2.21	0.18	0.47	<0.001
			Max	0.272	0.193	(0.17-0.21)	1.95	0.342	0.302	(0.27-0.33)	1.58	0.56	-0.19	<0.001
		WUA	Mean	0.040	0.018	(0.02-0.02)	2.85	0.073	0.036	(0.03-0.04)	2.17	0.98	-0.24	<0.001
			Max	0.080	0.024	(0.02-0.03)	3.66	0.138	0.045	(0.04-0.05)	2.81	0.92	-0.23	<0.001
		WUADay	Sum	1.778	0.139	(0.11-0.17)	4.70	3.462	0.247	(0.17-0.29)	3.96	0.78	-0.16	<0.001
		HHS	Mean	0.170	0.133	(0.12-0.15)	1.73	0.233	0.212	(0.19-0.23)	1.32	0.59	-0.24	<0.001
			Max	0.293	0.209	(0.19-0.24)	2.08	0.360	0.327	(0.29-0.35)	1.49	0.57	-0.28	<0.001
	Spawning	CSIm	Mean	0.062	0.047	(0.04-0.05)	1.48	0.097	0.060	(0.05-0.07)	2.12	0.29	0.43	<0.001
			Max	0.177	0.118	(0.1-0.13)	2.12	0.239	0.202	(0.18-0.22)	1.66	0.71	-0.22	<0.001
		WUA	Mean	0.030	0.011	(0.01-0.01)	3.30	0.059	0.025	(0.02-0.03)	2.46	1.22	-0.26	<0.001
			Max	0.071	0.016	(0.01-0.02)	4.41	0.132	0.033	(0.03-0.04)	3.79	1.03	-0.14	<0.001
		WUADay	Sum	1.409	0.086	(0.07-0.1)	5.80	2.963	0.164	(0.11-0.2)	4.87	0.91	-0.16	<0.001
		HHS	Mean	0.112	0.083	(0.07-0.09)	1.86	0.162	0.145	(0.13-0.16)	1.40	0.74	-0.25	<0.001
			Max	0.191	0.125	(0.11-0.14)	2.30	0.250	0.220	(0.2-0.24)	1.58	0.76	-0.31	<0.001

km^2-d pre- and post-restoration, respectively (1983 water year; juvenile splittail). The Wilcoxon signed-rank test indicated that the increases in post-restoration habitat availability were significant ($p < 0.05$) for all habitat metrics. The bootstrapped 95% confidence intervals (CIs) for median values at the water year scale suggested significant differences in mean, but not in maximum spatial mean of cell suitability index

(CSIm) and WUA. At the event scale, all CIs suggested significant differences between the pre- and post-restoration conditions, except for mean juvenile splittail CSIm.

As a measure of overall habitat quality within the inundated floodplain, the average water-year maximum of daily juvenile rearing hydraulic habitat suitability (HHS) increased from 0.53 to 0.59 and at the event scale increased from 0.29 to 0.36 (shifts in spawning conditions are similar, but lower). The summaries of CSI and HHS suggest that average quality was boosted by restoration, while maximum habitat quality (at the water year scale) was not substantially enhanced. In addition to the increased habitat quality, a substantial portion of the increase in splittail habitat WUA is attributable to the overall increased extent of inundation and therefore overall greater potential area for suitable habitat.

Examination of the habitat metric distributions showed that, overall, peaks in densities either shifted toward higher values (e.g., water year mean HHS) or broadened with greater densities at higher values (e.g., event max CSI or water year mean WUA; Figure 5-5). Also, extreme high values increased in most cases post-restoration. High frequencies of low WUA values are apparent, with extreme right skew. Of the 110-year record, the number of water years associated with >50 km^2-d of juvenile splittail habitat increased from just once pre-restoration to 12 times post-restoration. Conversely, the number of years with <2 km^2-d of juvenile splittail habitat decreased from 41 to 37 post-restoration. Many distributions exhibit ceiling thresholds, pointing to a maximum CSI achievable given the multiple suitability criteria. Maximum CSI and HHS are both distinctly bimodal at the water year scale, showing that higher maximum suitability occurred with greatest frequency (medians higher than means). Distributions are quite different between summaries at the water year scale and the event scale. Extreme mean values were higher at the event level because individual extreme events were included that were otherwise smoothed in water year means. For maximum summaries, distributions shifted toward higher values for water years over events because events of low maximum values were not included in water year maximums. Though habitat availability was lower overall, the shapes of distributions were similar for spawning splittail.

Spatiotemporal habitat variability in response to restoration

TEMPORAL VARIABILITY

Splittail habitat was highly variable interannually, shown by coefficients of deviation (CD, interquartile range divided by the median) ranging from 1.22 to 2.55 (see Table 5-1). Habitat variability increased slightly with restoration. The time series of estimated annual and cumulative WUADay shows that most years of above average WUADay were followed by several years of low habitat availability, producing pronounced steps in the cumulative distribution (Figure 5-6). Extended low habitat periods are notable in the 1930s, 1950s, and late 1980s to early 1990s, with only four years of no habitat.

Exceedance probability plots show the frequency of different levels of WUADay and changes in response to restoration. They show an approximate doubling of habitat at most probabilities with restoration, consistent with overall summaries (Figure 5-7). For example, under pre-restoration conditions, a WUADay of about 13 km^2-d was exceeded 23% of the time, which corresponds with a 45% exceedance probability under post-restoration conditions (and a 23% probability corresponds with 26 km^2-d). Reflected in the concave shape of the curves, at low exceedance probabilities the extreme highs of WUADay changed rapidly with changing probability, and at low WUADay the probability of exceedance increased rapidly. Differences were greater between pre- and post-restoration than they were between habitat provided for juvenile and spawning life stages. To examine the factors contributing to WUADay, the likelihoods of exceeding different total numbers of days within a season that exceed particular WUA

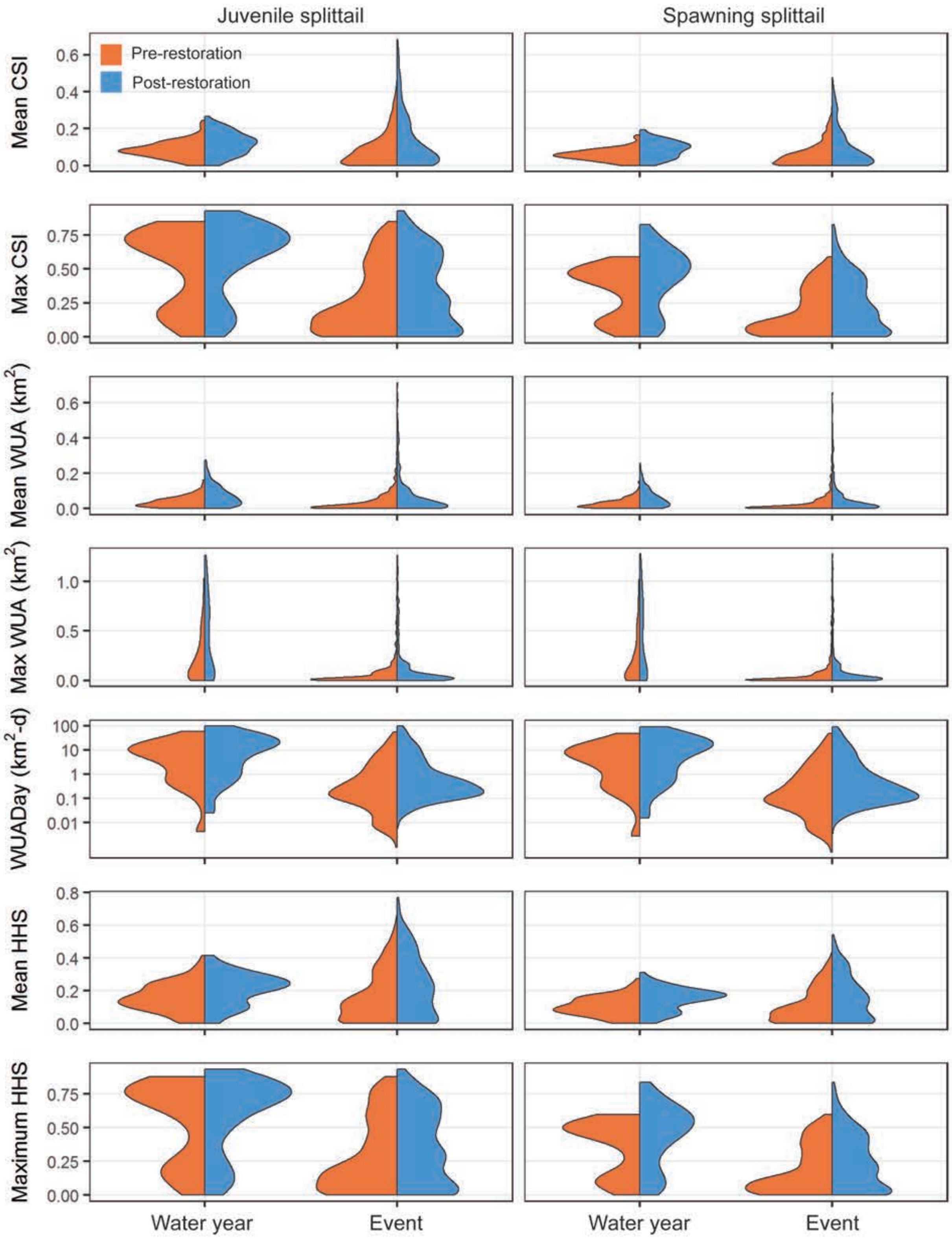

Figure 5-5. Violin plots of the distribution of Sacramento splittail habitat metrics, for pre- (orange) and post-restoration (blue) conditions, summarized at the water year and event scale. Note the log-scaling on the y-axis for WUADay.

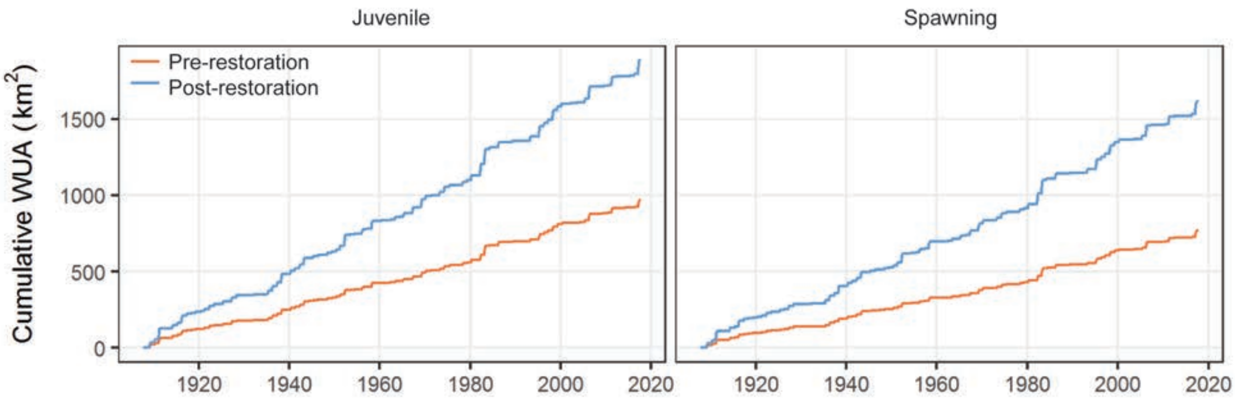

Figure 5-6. Period of record accumulation of habitat availability for juvenile and spawning Sacramento splittail estimated from suitability criteria.

thresholds are shown in Figure 5-8. Here, response to restoration is more variable. Overall, the number of days in a year meeting higher WUA thresholds (e.g., >50 ha) had greater increases in exceedance probability post-restoration in comparison to lower WUA thresholds (e.g., >1 ha). For example, achieving 50 days of >1 ha of WUA for juvenile splittail increased in exceedance probability from 57% to 63% with restoration, while 51 days of >50 ha of WUA increased in exceedance probability from 1% to 9%. At low exceedance probabilities, the number of days changed more rapidly for high WUA thresholds than they did for low thresholds, pointing to rare occurrence of high WUA for long periods of time.

Seasonal patterns show median habitat availability began to increase above zero in mid-February under both restoration configurations (Figure 5-9). Under median conditions, habitat availability maintained fairly evenly through the end of March, at which point availability tended to increase until the beginning of April before falling in late April to zero availability by mid-May (the cut-off date set through the established habitat suitability criteria). The April increase seems more consistent under post-restoration conditions. The most noticeable difference with restoration is the much higher median conditions in late April to early May, which effectively pushed positive median habitat availability out by several weeks. The upper quartile expands substantially post-restoration, suggesting that median conditions increased in large part due to this increased variability.

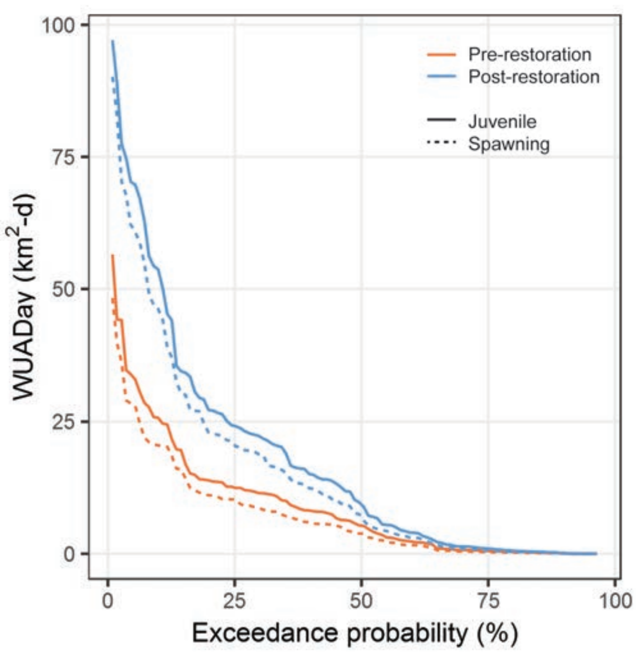

Figure 5-7. Empirical exceedance probability distributions for pre- (orange) and post-restoration (blue) annual habitat (WUADay) for juvenile (solid lines) and spawning (dashed lines) Sacramento splittail.

SPATIAL PATTERNS

Spatial summary of habitat availability shows patterns following topography as well

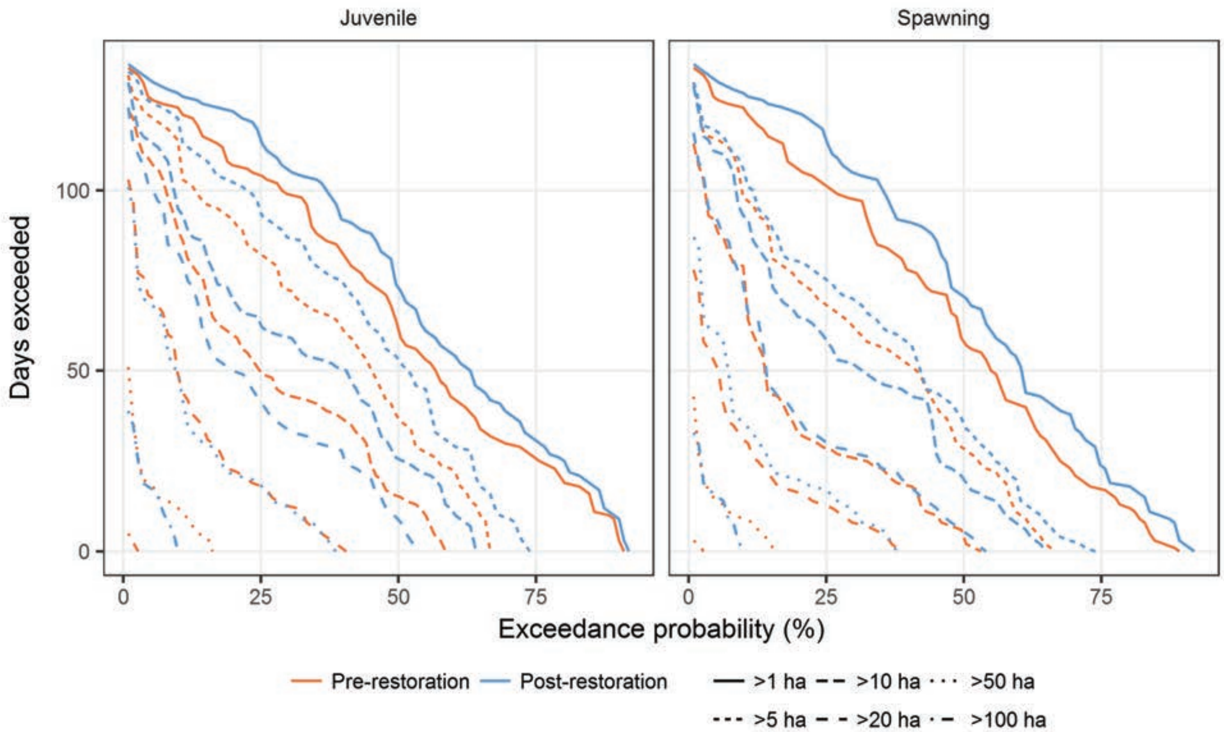

Figure 5-8. Empirical exceedance probability distributions for the number of days in a water year exceeding different levels of available daily habitat (WUA; ha) for pre- (orange) and post-restoration (blue) juvenile and spawning Sacramento splittail.

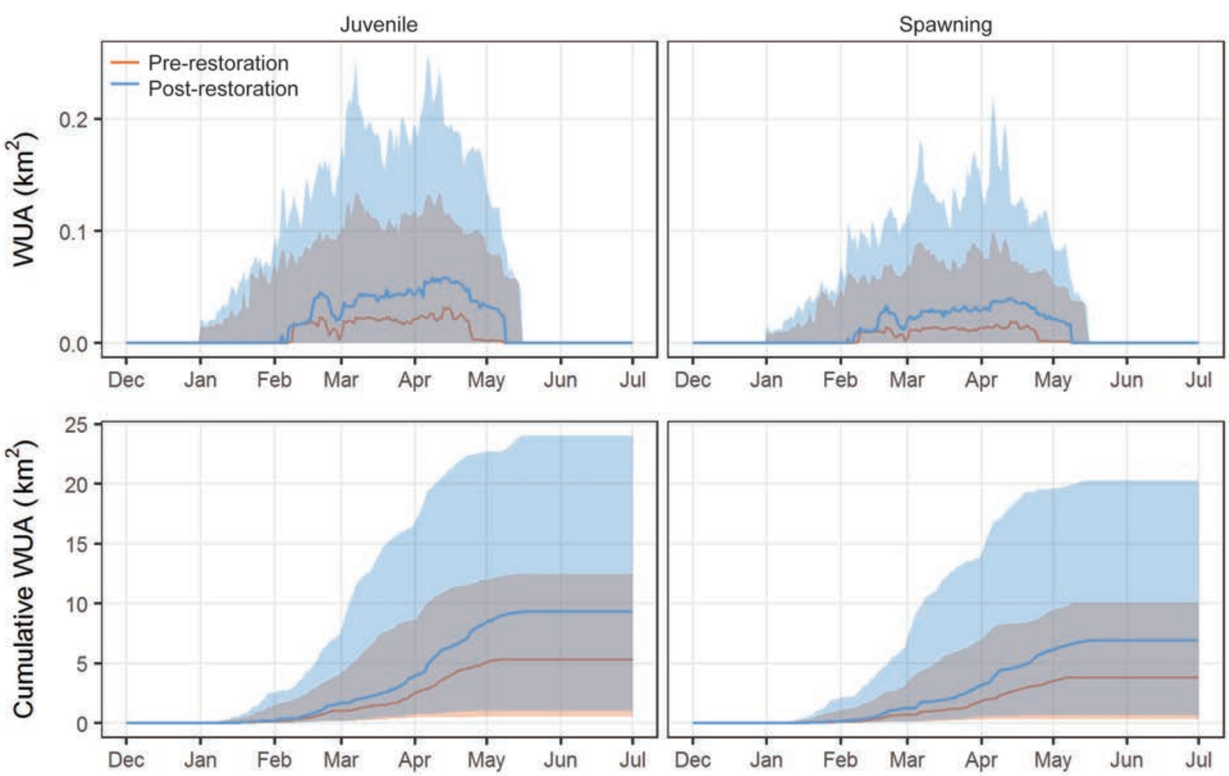

Figure 5-9. Seasonal distributions of available habitat (WUA) pre- (orange) and post-restoration (blue) for juvenile and spawning Sacramento splittail. Shading illustrates the interquartile range.

as proximity and connectivity to the river channel (Figure 5-10). High levels of habitat availability were concentrated in particular floodplain areas, which expand with restoration. Under both configurations, for example, the low-elevation area at the downstream end of the floodplain provided substantial habitat. While maximum annual WUADay was higher for juvenile splittail than spawning splittail in this area, it appeared to increase the most for spawning splittail. Resulting from the requirement that inundation duration be at least seven days, the highest-elevation areas, concentrated in the upstream forested area, produced no habitat. With restoration, the area of no habitat expanded. Overall, most of the floodplain area showed at least modest increases in habitat with restoration (Figure 5-11). Most of the habitat benefits from restoration were concentrated in the central part of the floodplain, associated with the primary river-side levee removal as well as the three small breaches performed along the interior ring levee. Additional benefit was provided in smaller areas comprising the excavated swale and connected western side-channel at the downstream end of the floodplain. Patterns for juvenile and spawning splittail habitat were similar overall, though the spatial extent of maximum annual spawning habitat increases were greater than for juvenile habitat. Small parts of the floodplain actually declined in habitat availability with restoration, primarily just upstream from the main levee-breach area and a topographically-aligned patch in the downstream portion of the floodplain.

The spatial variability in habitat tracks patterns of CSI (Figure 5-12). It is of note, however, that a large area of lower post-restoration maximum CSI (just upstream from the main levee removal) does not amount to substantial losses in habitat availability in the same area, likely because the area is inundated for such brief periods of time with large floods. In addition to greater extent of higher quality habitat post restoration, maximum CSI increases also were greater than maximum CSI decreases. The most a grid cell increased in maximum CSI was 1.0 while the most a cell decreased was 0.7.

Habitat relationship to flow regime

As a fundamental driver of floodplain ecosystems, the flow regime of the Cosumnes River was shown to be highly related to splittail habitat availability. Significant correlations ($p < 0.05$) were found between total water year availability (WUADay, km^2-d) and water year flow volume (juvenile splittail: pre-restoration *tau* = 0.84, post-restoration *tau* = 0.85; Figure 5-13). Habitat availability was also statistically significantly different for different water year types (annual flow quantiles; Figure 5-14a), as supported by the pairwise Wilcoxon signed-rank test. During critically dry water years (0.0-0.2 quantile) gains in habitat were less (a 42% increase in mean habitat availability) than during other water year types (increases from 78-99%). However, while water year flow volume is an important predictor of habitat, results also show considerable spread in the data, particularly for the wet water year types (0.6-0.8 and 0.8-1.0 quantiles). For example, total juvenile splittail habitat availability during wet years (0.8-1.0 quantile) ranged from 12 to 97 km^2-d post-restoration (6-57 km^2-d pre-restoration), and reached only 0 to 1.1 km^2-d post-restoration (0-0.9 km^2-d pre-restoration) during critically dry years. Further, when classifying the individual flood events by the flood types of Whipple et al. (2017), the pairwise Wilcoxon signed-rank test suggested that different flood types translated to significant differences in habitat availability ($p < 0.05$), though considerable spread in the data for each group was observed (Figure 5-14b).

In relating daily habitat availability (WUA) to daily flow, considerably more variability was observed than with relationships at the water year scale, attributable to habitat criteria relating to timing and duration. However, correlations between these variables were still statistically significant (juvenile splittail: pre- and post-restoration Kendall's *tau* = 0.55, $p < 0.05$). As shown in Figure 5-15, the maximum WUA for any given flow produced a fairly clean and smooth curve at higher flows, but WUA was found

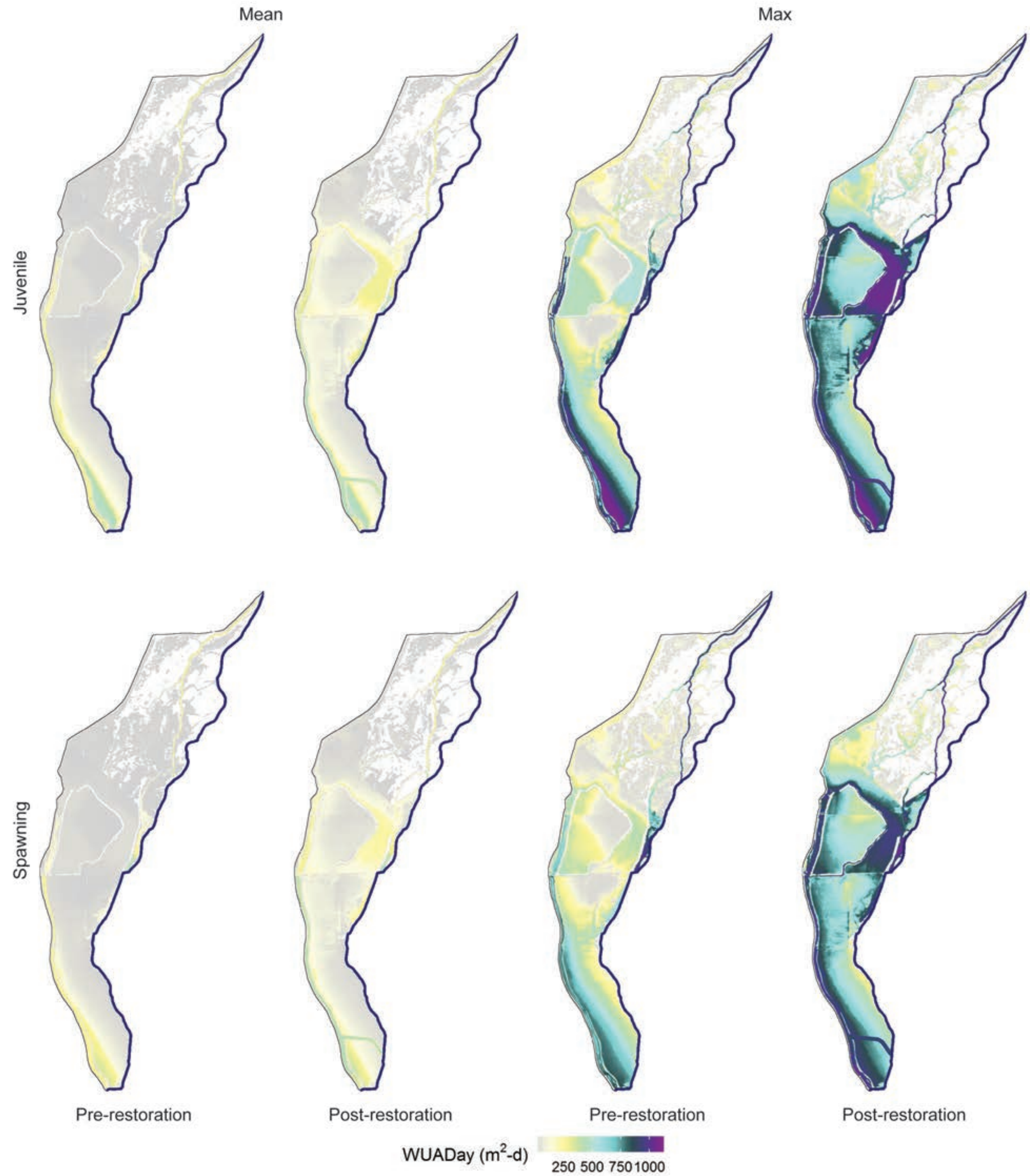

Figure 5-10. Spatial summary (mean and max) of annual habitat availability (cell-based WUADay) for juvenile and spawning Sacramento splittail. Note that, as a cell-based summary, WUADay units are in m^2-day.

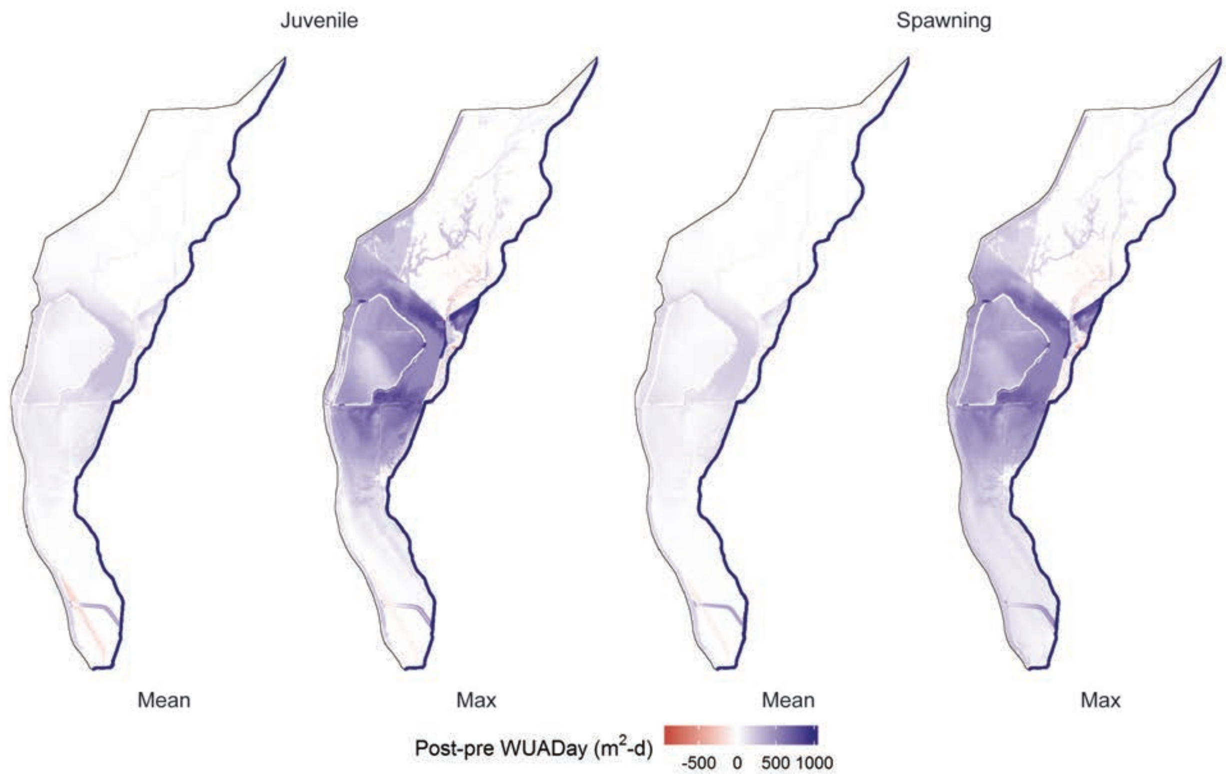

Figure 5-11. Spatial summary (mean and max) of differences in annual habitat availability (cell-based WUADay; m^2-day) for juvenile and spawning Sacramento splittail.

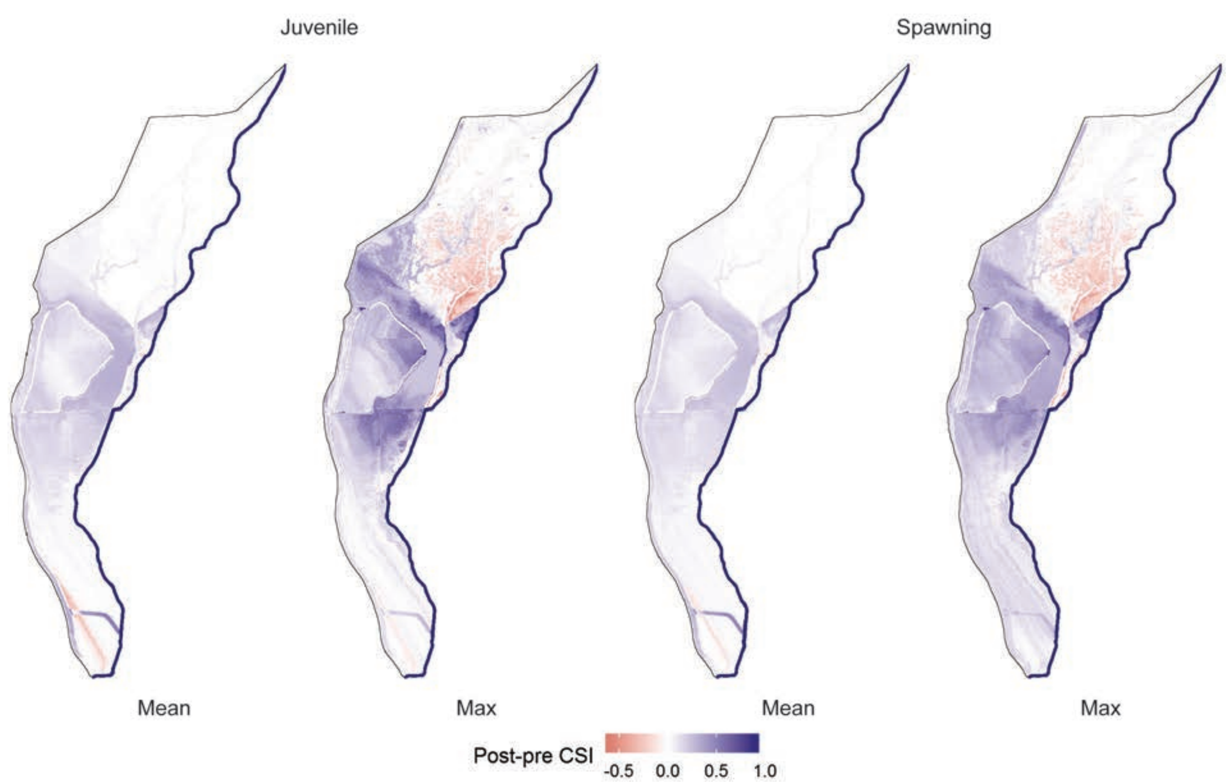

Figure 5-12. Spatial summary (mean and max) of differences in habitat suitability (cell suitability index, CSI) for juvenile and spawning Sacramento splittail (based on annual time series of maximum event-average CSI).

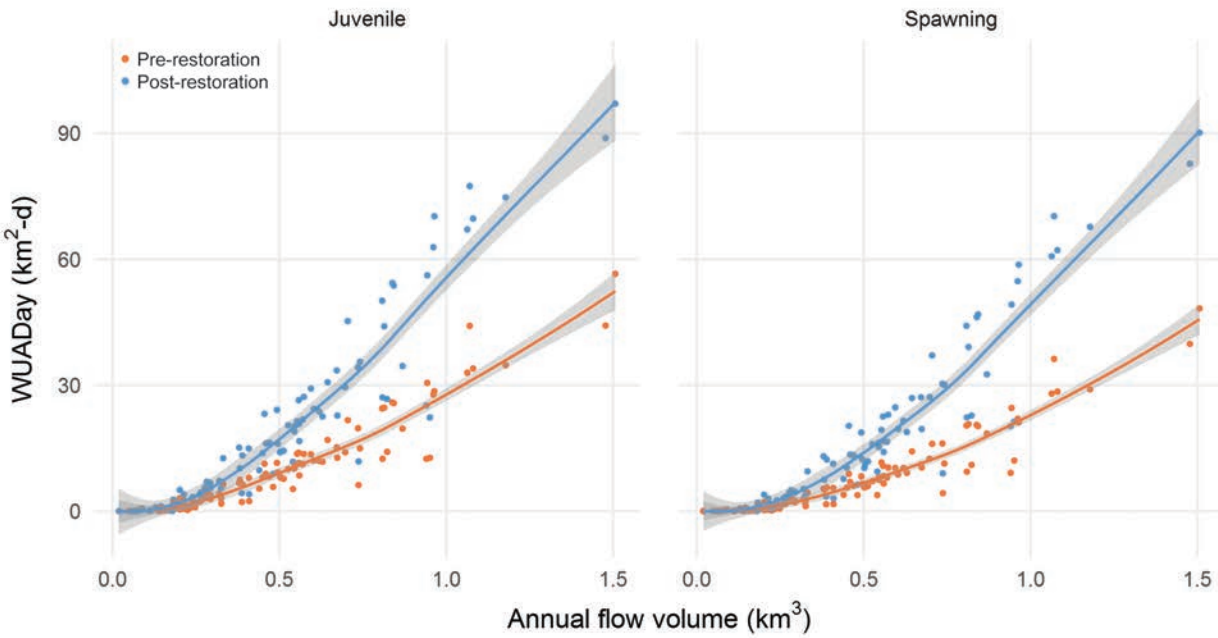

Figure 5-13. Scatterplot of juvenile and spawning Sacramento splittail habitat (WUADay) versus Cosumnes River annual flow volume (km^3) for pre- (orange) and post-restoration (blue) conditions, with a loess smoothing line (confidence intervals determined via standard errors, assuming normal distribution).

at many values between 0 and the maximum WUA for each flow. The plot also shows a sharp increase in maximum daily WUA for post-restoration conditions starting at around 38 m^3/s. After rapidly rising with flow, maximum juvenile splittail habitat availability tapers off around 150 m^3/s for both pre- and post-restoration conditions, followed by a pronounced, but less-sharp decline. Spawning splittail habitat availability appears to maximize around 200 m^3/s and declines less rapidly than for juvenile splittail with continued increases in flow. Figure 5-16 shows the relationship between the difference (post- minus pre-restoration) in daily WUA and flow over time to better illustrate how the multiple habitat criteria that make up the determination of WUA are manifested across the flow regime and at what points in the flow regime the restoration makes the most difference. This suggests that, within the appropriate seasonal time period, the longer duration recession limbs of medium to high flood peaks produced the greatest increases in habitat.

DISCUSSION

Spatiotemporal response to restoration

This research demonstrates the value of quantifying habitat availability through spatiotemporal analysis of the hydrospatial regime, which is particularly relevant for highly dynamic floodplain environments. Flood hydrograph characteristics (including duration and timing), landscape patterns, as well as the interaction between the two, affect habitat outcomes. Results from this analysis show that overall habitat availability (weighted usable area; WUA) for Sacramento splittail increased post-restoration with levee-removal and reconnection of the floodplain with the Cosumnes River. However, habitat availability and the habitat changes due to restoration varied in space and time.

Restoration benefits accrued due to increased habitat quality (i.e., mean cell suitability index, CSI) as well as increased potential area of available habitat (i.e., inundation extent and duration). Post-restoration improvements were found for average habitat availability as well as for upper percentiles. Habitat was variable inter-annually, with just a few years with extremely high habitat availability. Long-term average annual habitat (WUADay) was reached about once every three years (for post-restoration juvenile splittail). Seasonally, pronounced benefits from restoration did not begin until late February and median habitat availability increased the most in the late spring, including the extension of positive median availability by several weeks. Results show that some spatial patterns of habitat availability were sustained between pre- and post-restoration, while restoration habitat benefits concentrated in the central part of the floodplain in the vicinity of the main levee-breach and contributed most to the added habitat benefits. The excavated swale provided additional habitat at the downstream end of the site, which was particularly pronounced during lower flood flows. Some tradeoff occurred, where small areas saw reduced habitat suitability (CSI) and WUA in areas that experienced faster drainage due to nearby restoration elements (e.g., the excavated swale). In general, spawning splittail habitat was found to be somewhat less available post-restoration than juvenile splittail habitat, but were otherwise quite similar in spatial and temporal patterns. Finally, while daily and annually summarized WUA and flow regime characteristics are highly correlated, results indicate that the relationship between flow and habitat is not 1:1 and that higher flows do not necessarily translate to greater habitat availability. This suggests that where temporally- and spatially-dependent factors such as duration

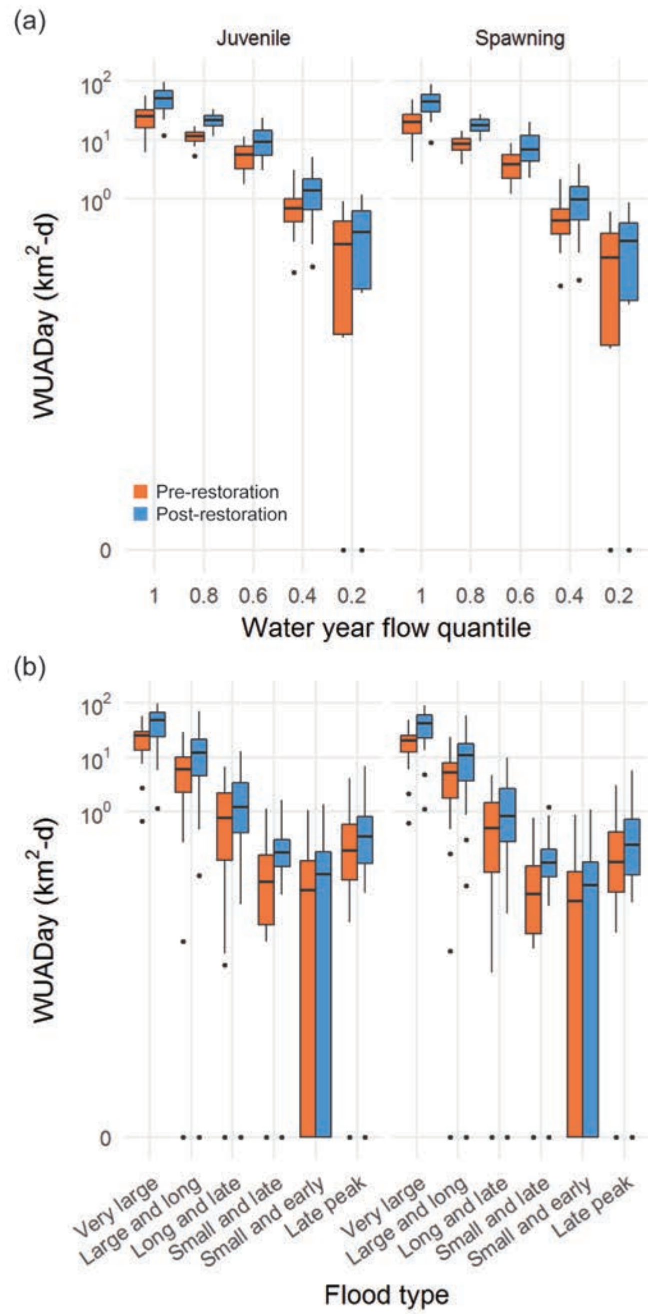

Figure 5-14. Comparisons for water year flow quantiles (a) and flood types (b) of juvenile and spawning Sacramento splittail habitat (WUADay), pre- (orange) and post-restoration (blue). Flood types follow Whipple et al. (2017). They are associated with different frequencies across the 110-year historical record: Very large (n = 18), Large and long (n = 47), Long and late (n = 101), Small and late (n = 126), Small and early (n = 144), Late peak (n = 109). Note the log-scale on the y-axis (zero values are represented by 1e10^{-6}).

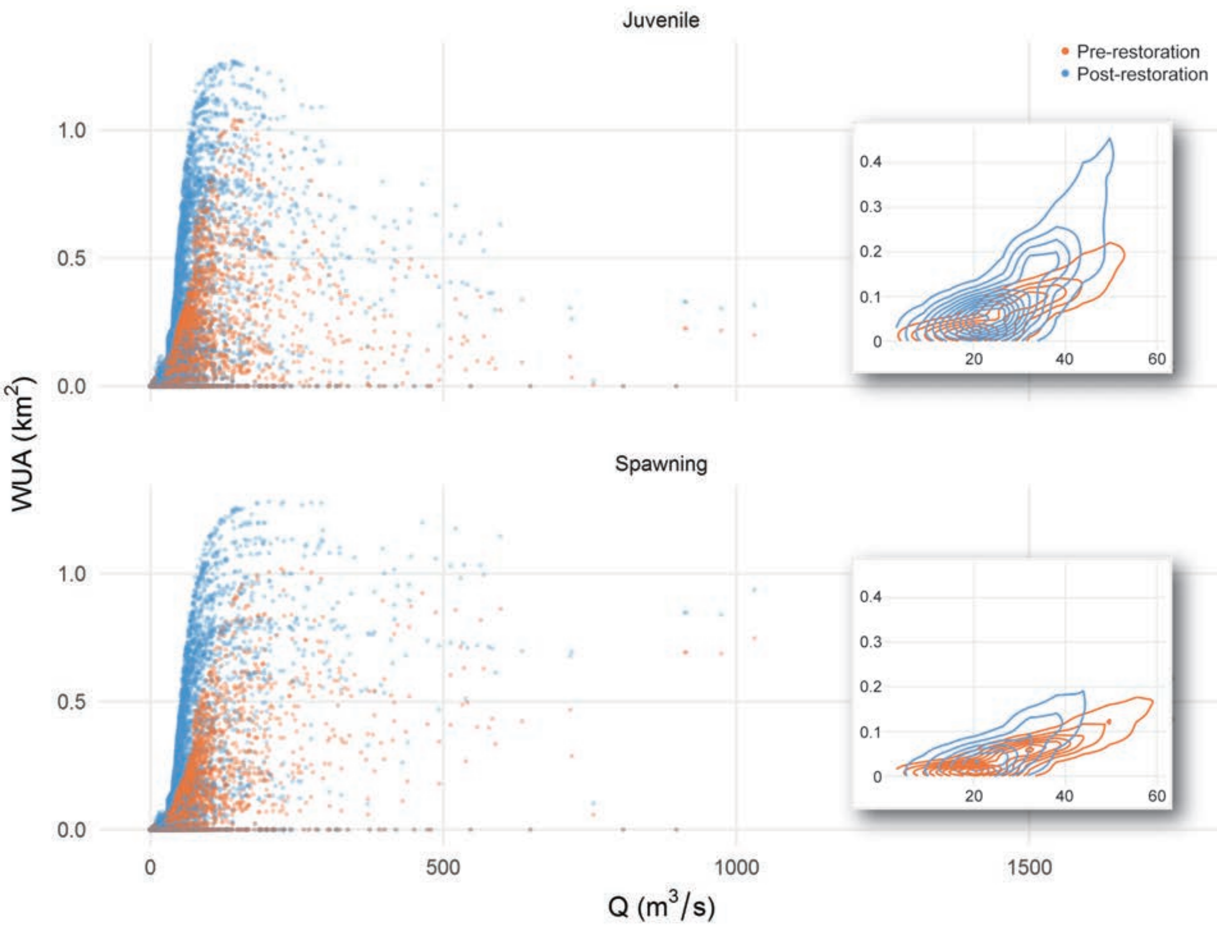

Figure 5-15. Daily flow-habitat relationships. Scatterplot and kernel density estimation plots (insets, note the limited range) of daily juvenile and spawning Sacramento splittail habitat (WUA) versus Cosumnes River daily discharge (m^3/s) for pre- (orange) and post-restoration (blue) conditions.

and connectivity matter, such as in floodplain environments, simple flow-WUA relationships may misrepresent overall habitat availability.

Advancing physical habitat quantification

This study aligns with discussions in the riverine habitat quantification literature over the last several decades that increasingly recognize the importance of accounting for spatiotemporally variable habitat (e.g., Carbonneau et al., 2012; Jacobson & Galat, 2006; Maddock, 1999; Merenlender & Matella, 2013). Most applications relate to effects of flow regulation, though some consider landscape or channel changes. Particularly relevant insights have come from research on habitat changes in the regulated Missouri River, including regulation of habitat response to altered flow regime by different channel configurations (Jacobson & Galat, 2006), homogenization of shallow-water habitat patch dynamics (Bowen et al., 2003), and effects of channelization on the availability of floodplain habitat (Erwin et al., 2017). For the San Joaquin River in the Central Valley, Matella and Merenlender (2015), using area-duration-frequency analysis, determined that implementing environmental flows in addition to planned levee-removal/setback options was needed to achieve desired habitat benefits. Other studies have demonstrated promising directions in describing habitat response to flows mediated by geomorphic change (Escobar-

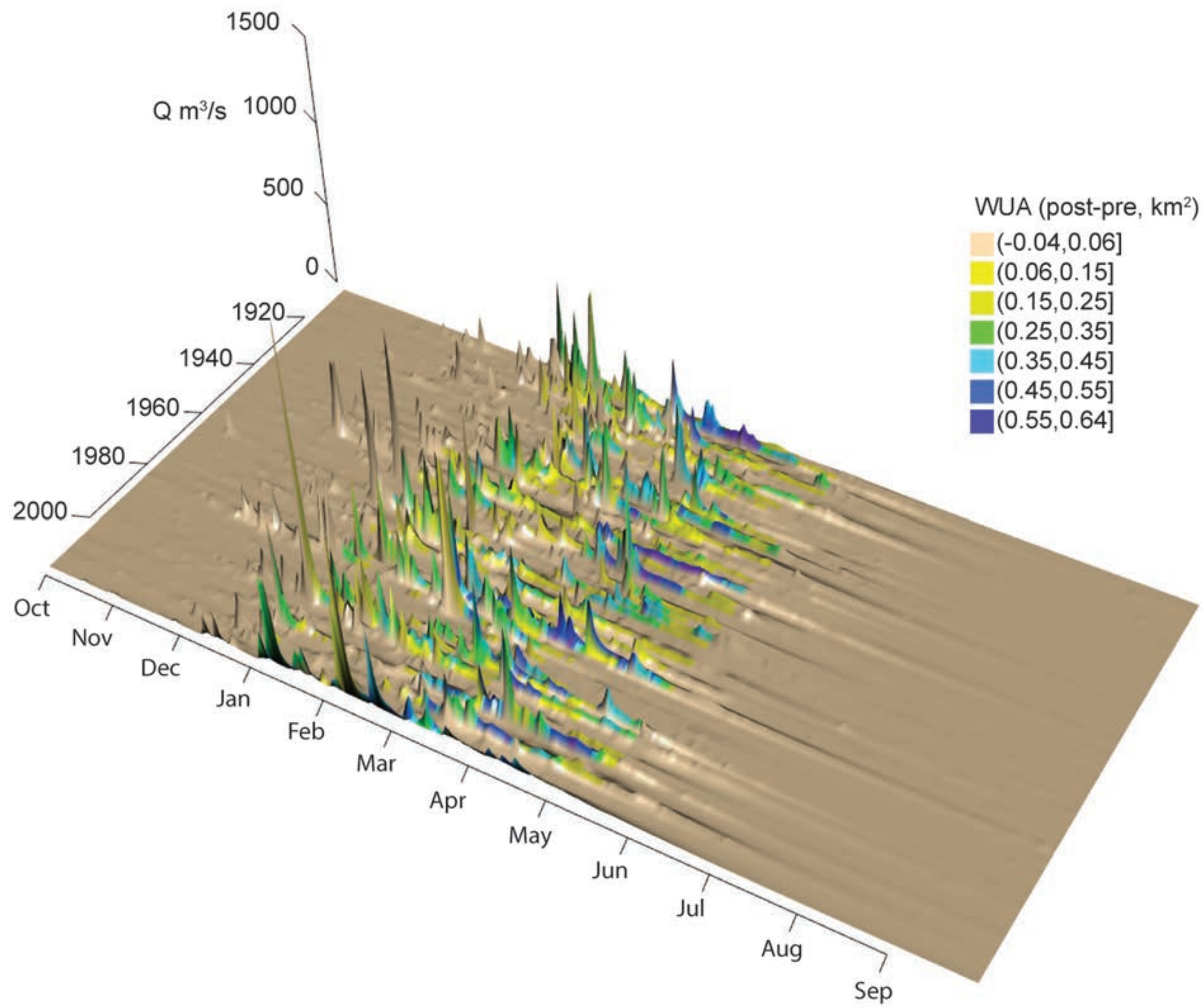

Figure 5-16. Habitat availability differences over time. Multi-dimensional plot of the historical flow record of the Cosumnes River showing the difference between pre- and post-restoration conditions for daily juvenile Sacramento splittail habitat availability (WUA) in color (low values in tan to high values in blue).

Arias & Pasternack, 2011; Yarnell et al., 2012), modeling fish access to floodplain habitat (Meitzen et al., 2017), addressing habitat patchiness (Dyer & Thoms, 2006; Li et al., 2016), defining fish passage flows (Grantham, 2013; Holmes et al., 2016), scaling habitat quantification to the basin- or population-scale (Wheaton et al., 2018), and quantifying patterns at multiple scales using wavelet analysis (Carbonneau et al., 2012).

The habitat quantification methods applied here advance more common techniques by analyzing conditions at high spatial resolution for a daily time series, which allows the application of habitat criteria that relate to temporal flood sequence and spatial proximity, and also enables spatially- and temporally-resolved output. This distinction means that aspects of spatial configuration (e.g., connectivity) and temporal sequence (e.g., duration), factors critical in determining floodplain habitat quality and availability, can be included at the basic unit of analysis (daily, individual grid cell). Options for evaluating habitat suitability and WUA are therefore expanded beyond their typical scale of analysis. This expansion includes the duration, timing and connectivity criteria applied here, as well as other criteria such as minimum patch size (Carnie et al., 2016).

By retaining spatial and temporal resolution throughout the analysis, this research addresses two important considerations in physical habitat quantification. First, this spatial, hydrodynamic analysis demonstrates that flow alone may be inadequate for assessing floodplain habitat. That is, quantifying habitat for individual flows to subsequently extrapolate across a time series may not adequately address the substantial spatial variability present in floodplain environments and could misrepresent overall habitat availability. Inundated area that loses connection to the river (i.e., ponding) on the receding limb of a hydrograph is another factor that changes the relationship between flow and duration, which these methods address. Second, WUA can be summed over time (WUADay) spatially for individual flood events or seasonally, building from methods presented in Chapter 4 and other recent research quantifying floodplain physical properties over time (Stone et al., 2017). Previous efforts have related flow duration curves and flow-habitat relationships to predict the probability of exceeding various levels of habitat availability (e.g., Chan et al., 2012; Matella & Jagt, 2014; Stamou et al., 2018). However, while these provide valuable insights into expected habitat, other measures such as seasonal cumulative totals, inter-annual variability, and spatial patterns are not easily determined. For example, application of the peak flood time series informs annual maximum habitat availability, but years with similar peak flows may be associated with starkly different overall habitat availability if the number of flood days between the two years are substantially different. Thus, though the habitat exceedance probabilities produced in this analysis are similar in form, the construction is fundamentally different, building from spatiotemporal computation of WUA.

In addition to the commonly-applied CSI and WUA metrics, WUADay proved to be an inclusive metric that accounted for time over which habitat is provided. This concept is similar to the water residency time metric introduced by Hermoso et al. (2012). One important limitation to its utility is that by integrating space and time, a small area inundated for a long period of time could receive the same value as a large area for just a brief period. However, from a habitat availability context, this issue can be mitigated through habitat suitability criteria relating to area and duration that are used in the WUA calculation (e.g., if a short duration flood or small patch size means lower quality habitat, then the resulting WUADay should be correspondingly lower than the same area without those limitations). To aid interpretation, it is recommended that WUADay be examined alongside spatially and temporally variable CSI and WUA and be assessed spatially as well (i.e., the sum of each grid cell WUA for the time period in question).

Limitations and next steps

Limitations of this approach should be considered in interpreting results and in future applications. The limitations and uncertainties of 2D hydrodynamic modeling can affect results. For example, if suitability criteria are highly sensitive to small changes in velocity, then potential error in spatially-distributed velocity estimates could substantially affect habitat estimates. Also, modeling results are applied to daily mean flow, which do not account for instantaneous peak flood flows. As with most studies of this nature, the spatially-resolved depth and velocity modeling outputs are used as static representations of given flows, which means that temporal and spatial patterns of flood waves propagating through a complex floodplain dynamically in time are not included. A limitation common to hydrodynamic modeling studies, the influence of flow pulses (below overland flood connection), groundwater interaction, antecedent conditions, infiltration, evaporation, and precipitation are also not directly considered.

As a physical habitat quantification study, a fundamental issue is that physical habitat is not the only determinant of ecological response; biotic controls and other non-modeled factors can interfere with the use of habitat by species, and biological and physical interactive effects affect suitability (Boavida et al., 2013; Dunbar et al., 2012; Petts, 2009). For Sacramento splittail, habitat suitability also depends on temperature, salinity, dissolved oxygen, time of day, presence of submerged terrestrial vegetation, and availability of food (Moyle et al., 2004; Sommer et al., 2008). As physical habitat requirements of other species or whole-ecosystem measures are not included in this analysis, inferring overall ecosystem benefits of the restoration is more challenging – a common problem for physical habitat quantification studies. However, it is understood that the criteria applied for splittail are likely to translate well to other important floodplain functions, such as primary and secondary productivity (Grosholz & Gallo, 2006). While developing HSIs via expert opinion is often the best option, suitability assignments have also been shown to be highly variable across experts (Radinger et al., 2017). Many studies have proposed more sophisticated methods for developing and combining HSIs, which could be incorporated into the hydrospatial approach employed here (e.g., Dunbar et al., 2012; Jung & Choi, 2015; Li et al., 2016; Lin et al., 2015; Yi et al., 2014).

Identifying limitations can also point to opportunities for refining study results and expanding the capacity of this approach. Several recent studies couple hydrodynamic modeling with temperature monitoring data or modeling (Wheaton et al., 2018; Yao et al., 2015). Inclusion of factors such as minimum patch size, vegetation cover, and antecedent conditions also could refine this approach. Recent research by Bellmore et al. (2017) offers potential ways for examining food web interactions and implications. A systematic assessment of the spatiotemporal sensitivity of results to changes in HSIs could help identify potential thresholds (e.g., does changing the minimum acceptable inundation duration to six days substantially affect habitat availability?). Combined with an examination of uncertainty produced by the hydrodynamic modeling, this could be used to generate uncertainty estimates for outcomes of WUA. Additionally, results can be verified using empirical monitoring studies, with attention to defining mutually-appropriate spatial and temporal scales.

Restoration management implications

Increasing the extent and frequency of inundation through levee removal and excavation nearly doubled overall habitat availability for Sacramento splittail, a priority fish for conservation and restoration. These findings align with other research indicating restoration of river-floodplain connectivity is promising for enhancing freshwater ecosystems (e.g., Beechie et al., 2013; Erwin et al., 2017; Jeffres et al., 2008; Matella & Merenlender, 2015; Schindler et al., 2016). Restoration also increased the frequency of years with large areas of available habitat and extended the timing of availability later into the spring. Results also suggested where the greatest habitat benefits can be expected: a core area near the main levee breach as well as the excavated swale. Though the site is small compared to the >10 km^2 of Cosumnes River floodplain (and >200 km^2 floodplain along the Sacramento River in the nearby Yolo Bypass), its importance as floodplain habitat may be disproportionately large given that it inundates at relatively low flows and therefore provides habitat when other floodplains are dry.

This study offers insights for restoration efforts elsewhere in the Central Valley and river restoration globally. Results imply that similar floodplain reconnection projects focused on increasing inundation extent and duration at intermediate flood flows as opposed to high flood flows may be most beneficial for increasing floodplain habitat opportunities for spawning and juvenile rearing splittail, as well as rearing Chinook salmon (which have similar habitat requirements to rearing splittail). Findings suggest that levee removal or setback restoration actions coupled with a dynamic flow regime both improve habitat quality and overall quantity, even within the highly modified landscape of the Central Valley. Benefits can be expected in all but the driest years, with greater benefits in wetter years with longer-duration flooding. Seasonally, more habitat was produced in the later spring, which could help mitigate effects of earlier flow timing from climate change (Knowles et al., 2018; Stewart et al., 2005). Additionally, results indicate that restoration elements that either promote long-duration inundation (e.g., the swale excavation) or ponding with regular reconnection (e.g., main levee breach) contributed the most to added habitat. More generally, this research illustrates that floodplain habitat varies in space and time, which points to the importance of landscape heterogeneity and a variable flow regime in planning and management. Managing for heterogeneity and variability means that the right combinations of flow and topography will more likely occur at least somewhere within a floodplain to support floodplain functions.

The spatiotemporal approach applied here can be used to help restoration and water resource managers develop more sophisticated actions and objectives – whether they be for physical restoration measures, environmental flow applications, or both – that link physical changes to ecological consequences. The approach here is particularly relevant to highly dynamic floodplain environments, where factors such as spatially-distributed inundation duration and connectivity have profound implications for ecological functions. The need for more sophisticated approaches in floodplain habitat evaluations for restoration and conservation is supported by the finding that WUA could not be represented well through flow-WUA relationships, which are commonly used to evaluate in-channel habitat. As a scenario analysis tool, this approach produces information to understand relative benefits. Importantly, output is produced at scales used in management and planning, which can be lumped or split spatially or temporally depending on the application (Dunbar et al., 2012). By preserving the resolution of the input (daily, gridded), there is greater potential for exploring factors that produce overall outcomes, evaluating the sensitivity to or addition of habitat suitability criteria, and visualizing outcomes in multiple dimensions. Results can also be used to examine trade-offs that might occur when restoration actions in one part of a floodplain site generate detrimental or unexpected effects in another.

While this approach quantifies restoration outcomes, it also describes variability and identifies benefits and ways to manage for variability and heterogeneity. The habitat quantification methods are also flexible. Established with the application to a suite of species needs or ecosystem functions in mind, these methods can help investigate the multiple functions of floodplains through their shifting mosaics of habitats. Applications could include the development of restoration designs that target different conditions in different parts of the floodplain for different environmental flows, with the goal of meeting multiple ecosystem objectives. Potential to build from recent efforts includes the application of the multi-species HSI used by Zhao et al. (2017). Because floodplains are dynamic, not all ecological functions are served everywhere at the same time. Conceptually, as flows and landscapes interact, optimal foraging habitat for juvenile fish may shift in space and time, while patterns of optimal primary production may change in a different way. Hydrospatial habitat analysis allows for investigation and more specific quantification of that variability. It can be used to determine combinations of flood regime and landscape configurations that best support a set of ecological functions and to optimize different parts of floodplains for different functions.

CONCLUSIONS

The analysis presented here quantifies the relative habitat benefit of levee-removal restoration promoting floodplain hydrologic connectivity. Overall, spawning and juvenile rearing habitat for the native floodplain fish, Sacramento splittail, was estimated to nearly double. Results also show that high flood flows were not necessarily better and that habitat was maximized at more intermediate flood flows, but only with longer duration inundation. Spatial analysis also showed that not all parts of the floodplain provided better habitat. Analysis of conditions over time indicated that restoration extended habitat availability by several weeks, a potential hedge against hydroclimatic alteration to flow timing. The interaction of flows and landscape configuration produces habitat benefits in different places at different times, indicating the value of managing for heterogeneous conditions such that habitat benefits can be realized across a range of flows and climatic conditions. An important implication of this research is that where spatially-resolved duration and connectivity matter, such as in floodplain environments, establishing flow-WUA only relationships may be inadequate to fully evaluate habitat suitability and benefit.

As emphasis on considering ecological needs in the management of heavily-modified river-floodplain environments increases, new approaches are needed to better balance the often seemingly conflicting demands placed on these systems. Floodplain ecosystems depend on the very dynamics that traditional river management often seeks to eliminate. Integrating spatiotemporal dynamics into existing management requires an understanding of how to support flow and landscape interaction that best meets ecosystem needs. With many highly-managed large rivers entering the realm of novel ecosystems, where achieving some historical reference condition is not possible, attention needs to be paid toward managing landscapes and flow regimes together to activate the physical processes that will support more productive, diverse, and resilient ecosystems into the future.

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CHAPTER 6

RESPONSES OF A RESTORED FLOODPLAIN TO CLIMATE CHANGE USING HYDROSPATIAL ANALYSIS

Alison A. Whipple

Responses of a restored floodplain to climate change using hydrospatial analysis

ABSTRACT

Climate change induced changes in flow regime can have direct and interacting effects on floodplains and their ecosystems, often complicated by past human activities. Developing management strategies requires improved understanding of these changes and of how management alternatives might influence outcomes. This study applies hydrospatial analysis – a multi-metric spatiotemporal analysis – to quantify floodplain responses to climate change driven flow changes and to evaluate the influence of restoration. Two-dimensional hydrodynamic modeling is used to assess a levee-removal restoration site on the lower floodplain of the unregulated Cosumnes River, a central Sierra Nevada river flowing into California's Central Valley. Floodplain inundation patterns are compared for four climate change scenarios, a historical and a future time period, and before and after floodplain restoration configurations (a total of 16 30-year periods). To evaluate floodplain responses, metrics of inundation extent, depth, velocity, duration, timing, connectivity, and habitat availability for a native fish, Sacramento splittail (*Pogonichthys macrolepodotis*), were used. Changes in floodplain metrics related to opposing trends of declining spring flooding and increasing extreme winter flooding. However, the magnitude and direction of change were not the same across space and time and varied with the metric and climate change scenario. In addition, relationships between flow regime change and hydrospatial metric change were not consistent, implying that predicting floodplain response to climate change based on flows alone may be inappropriate. Though variable, effects of levee-removal restoration lessened overall between the historical and future periods. Restoration also dampened climate change effects in some cases, including reducing maximum depths and decreasing habitat losses. This research shows that the interaction of flows and the floodplain landscape mediates spatiotemporal floodplain responses to hydroclimatic change and suggests floodplain restoration can mitigate some climate change impacts. The quantified multi-metric spatiotemporal approach presented here is useful for assessing habitat restoration and environmental flow alternatives that together best meet objectives under climate change.

INTRODUCTION

A changing climate brings an uncertain future to freshwater ecosystems, complicated by myriad other human factors of change (Ficke et al., 2007; Meyer et al., 1999; Moyle et al., 2013). Most large rivers of the world have been severely degraded by more than a century of intense human activity (Dudgeon et al., 2006; Malmqvist & Rundle, 2002; Nilsson et al., 2005), which has reduced natural environmental variability and complexity that support diverse and productive river-floodplain ecosystems (Bunn & Arthington, 2002; Poff et al., 2007). Dams, diversions, groundwater abstraction, channelization and levees, and land use change have altered flow regimes and disconnected floodplain landscapes (Tockner & Stanford, 2002). Ecosystems are also affected by poor water quality, invasive species, and overharvest. Degraded riverine ecosystems are now less resilient to climate change, which also amplifies various ecological stressors (Capon et al., 2013; Palmer et al., 2009). Flow regime change is a primary way climate change affects riverine ecosystems (Döll & Zhang, 2010). It occurs as altered annual runoff volume, shifts in flow timing due to earlier snowmelt, dry season scarcity, more extreme floods and droughts, and human responses to climate change (Berg & Hall, 2015; Cayan et al., 2008; Das et al., 2013; Li et al., 2017; Mote et al., 2018; Swain et al., 2018). Climate altered flow timing may disrupt species life history cues, spring and summer water scarcity may reduce riparian recruitment and expose fish at vulnerable life stages, and more extreme floods may alter geomorphic processes, eliminate important riparian habitat, and reduce fish survival through increased sediment scouring (e.g., Death et al., 2015; Perry et al., 2012; Rivaes et al., 2012; Wade et al., 2013). Overall, highly modified river systems are expected to be more severely affected by climate change relative to unregulated rivers, prompting calls for proactive management that addresses these new challenges (Palmer et al., 2008).

River-floodplain environments are a target for restoration due to their ecological significance, degraded states, high vulnerability, and potential to buffer against climate change. Ecological processes, species life history requirements, and geomorphic processes depend upon natural spatiotemporal patterns – relating to magnitude, frequency, duration, and timing – of floodplain inundation (Junk et al., 1989; Poff, 2002; Poff et al., 1997; Ward et al., 2002). Restoring floodplains by reconnecting them to rivers through levee removal or setback, in addition to adjusting flow regimes to better support ecological processes, is a promising strategy to reestablish dynamic processes and bolster ecological resilience to change (Beechie et al., 2013; Opperman et al., 2009; Palmer et al., 2008; Wohl et al., 2015). However, restoration planning is often done without consideration of climate change (Donley et al., 2012). Determining likely ecological responses and identifying effective restoration strategies requires improved understanding of how hydroclimatic change may affect spatiotemporal variability of inundation patterns and how it may be manifested within highly modified riverine landscapes.

Quantitative methods involving hydrologic and hydrodynamic modeling are increasingly used to explore potential flow-mediated effects of climate change. In particular, hydrogeomorphic habitat modeling and scenario analysis provide needed information at scales appropriate for inferring ecological responses to change and for exploring management alternatives (Poff, 2002). Much of the progress has been enabled by improved capacity to process large datasets and by advances in two-dimensional (2D) hydrodynamic modeling (Teng et al., 2017). A common approach is to use downscaled global climate model (GCM) projections as input to a hydrologic model, where output is subsequently evaluated or applied in subsequent hydrodynamic, water operations, or ecological modeling (Xu et al., 2005). For example, climate change projections have been used to assess changes in floodplain wetland connectivity (Karim et al., 2016), large-scale floodplain inundation patterns (Langerwisch et al., 2013), riparian extent and composition (Moradkhani et al., 2010; Rivaes et al., 2012), availability of habitat for fish and their food sources (Guse et al., 2015; Matella & Merenlender, 2015), and fish migration (Boughton & Pike, 2013). The many levels of uncertainty in this top-down approach makes multi-model ensembles and evaluations of relative sensitivity to change especially important. Studies often focus on particular variables (e.g., annual maximum inundated area, spatial heterogeneity, or physical habitat at specific flows), whereas more comprehensive assessments of floodplain inundation regimes are rare.

The main objective of this study was to quantify changes in spatiotemporal floodplain inundation patterns and fish habitat availability in response to different climate change induced flow scenarios using the methodological approach for hydrospatial analysis presented in Chapters 4 and 5. In addition, this study aimed to evaluate the relative effects of levee-removal floodplain restoration. Analysis was done for a floodplain restoration site along the lower Cosumnes River in the Central Valley of California. Inundation patterns were represented through physical metrics describing the extent, depth, velocity, duration, timing, frequency, and connectivity of the floodplain to the Cosumnes River. Physical habitat suitability criteria were applied to quantify habitat for the Sacramento splittail (*Pogonichthys macrolepidotus*), a native floodplain fish species. As one of few floodplains within the highly altered Central Valley with frequent river-floodplain connectivity, this system is a unique opportunity to study the implications of climate change on floodplain restoration outcomes. This study uses available hydrologically modeled daily flow projections based on downscaled output from four GCMs (Knowles et al., 2018). These are applied to previously-established spatially-resolved flow-depth and flow-velocity relationships (see Chapter 4), generating gridded daily output for analysis. This study offers insight into potential changes to floodplain inundation patterns and related habitat under climate change within the Central Valley of California, as well as provides broader insights into the response of floodplain dynamics to multiple scenarios of change.

METHODS

Overview

The analysis presented here to evaluate floodplain response to climate change was based upon two primary inputs from prior research (Figure 6-1). The first consisted of daily climate change streamflow projections established by Knowles et al. (2018) using an approach common to top-down climate change impact analyses. Their basic process involved 1) transformation of output from multiple GCMs to downscaled and bias-corrected temperature and precipitation, 2) hydrologic modeling driven by these scenarios to produce daily runoff values, and 3) runoff routing to generate daily streamflow projections. The second input, established in Chapter 4, was developed using 2D hydrodynamic modeling for pre- and post-restoration floodplain topographic configurations to establish spatially-resolved flow-depth and flow-velocity relationships. In this study, the hydrodynamic modeling output was combined with bias-corrected daily climate change streamflow scenarios using spatially-resolved piece-wise linear interpolation. This process produced gridded daily estimates of depth and velocity for a total of eight scenarios (four climate

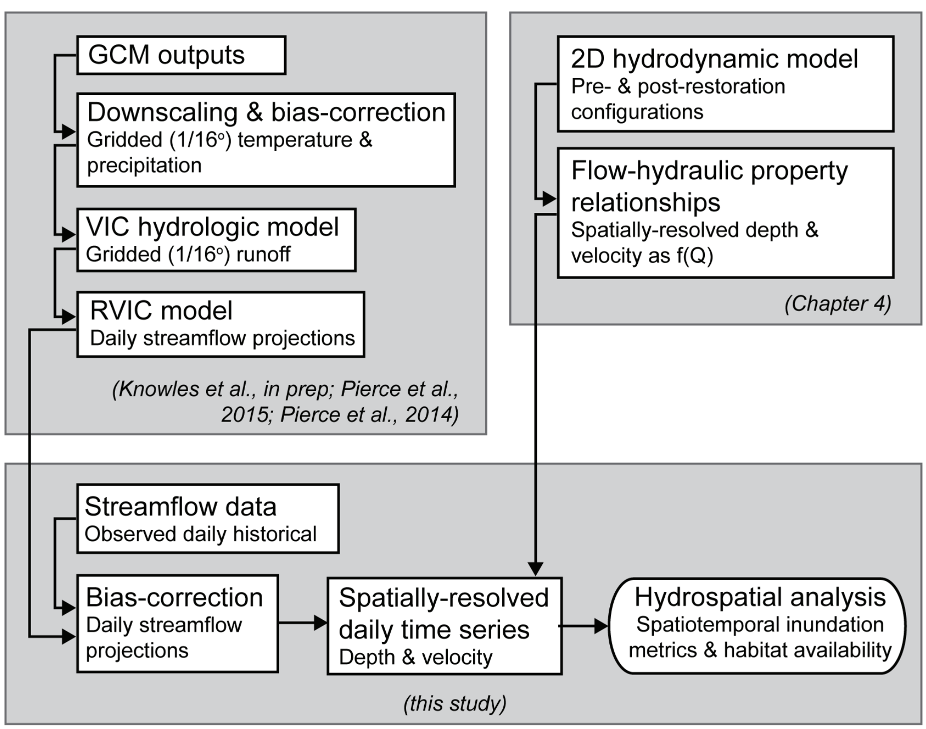

Figure 6-1. Conceptual diagram illustrating the process and components for evaluating spatiotemporal floodplain inundation patterns under climate change flow scenarios. Previously-completed elements are shown in the top two boxes and the bottom box contains steps completed for this study.

change scenarios and two topographic configurations), which were then analyzed and compared in space and time for the floodplain hydrospatial metrics of focus.

Study area

The Cosumnes River watershed drains 2,460 km^2 in the central Sierra Nevada of California (Figure 6-2). With a Mediterranean-montane climate, the Cosumnes watershed is characterized by cool, wet winters and hot, dry summers, generating strong seasonal runoff patterns. Precipitation averages 855 mm annually (1971–2000), most of which falls in higher elevations (PRISM Climate Group, 2006). Interannual variability is pronounced; just a few winter storms typically make the difference between a wet and dry year (Dettinger, 2011). Annual peak discharge for the 110-year historical streamflow record at Michigan Bar (MHB, #11335000; U.S. Geological Survey, 2017) ranges from just 6 m^3/s (1977) to 2,630 m^3/s (1997). The majority of runoff occurs between the months of December and May in response to rainfall events, though high flows in the late spring are boosted by snowmelt (Whipple et al., 2017).

The Cosumnes River remains largely unimpaired, unlike other rivers flowing into the Central Valley that are regulated by large dams serving California's vast water supply network. However, lower reaches of the Cosumnes River have followed a common history of channel incision and leveeing due to agriculture and development, aligned with the loss of most (~95%) of the nearly 4,000 km^2 of Central Valley floodplain wetlands (The Bay Institute, 1998; Whipple et al., 2012). Restoration efforts within the

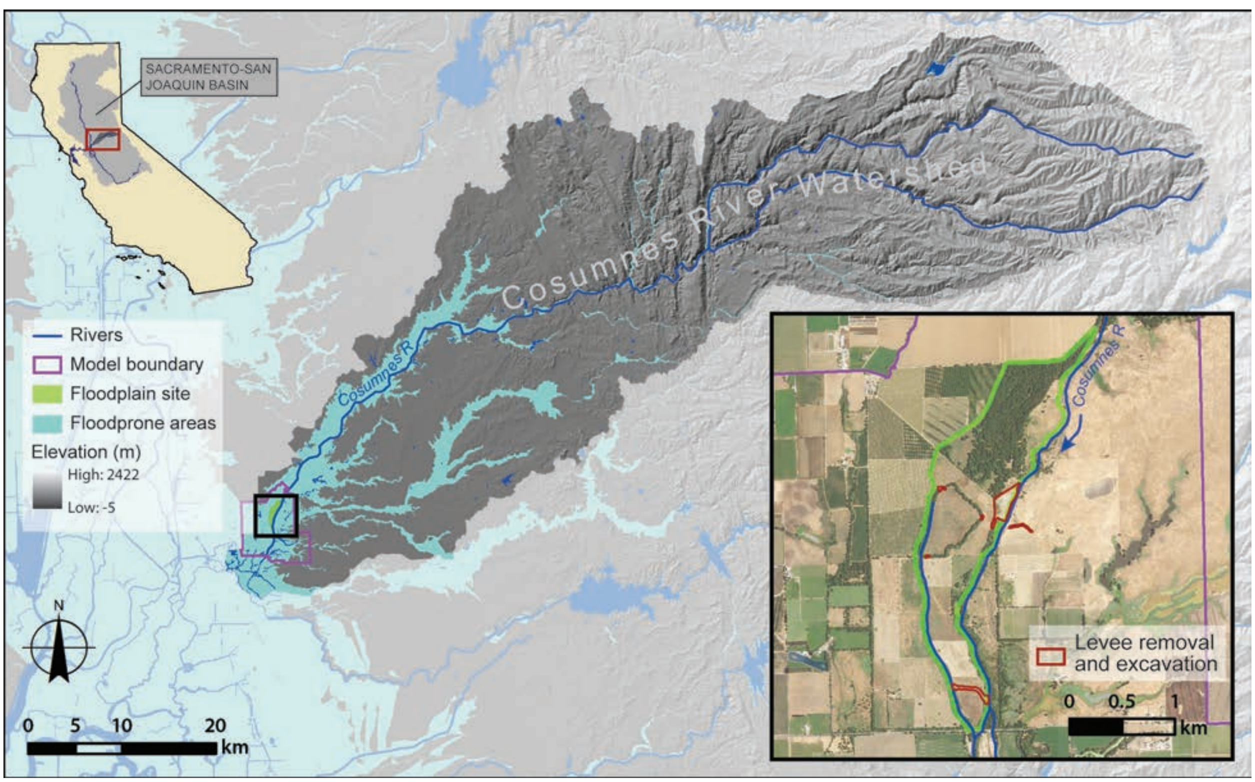

Figure 6-2. Study area within the Cosumnes River watershed. Floodplain restoration site position within the lower Cosumnes River watershed (2,450 km^2), one of the major river systems draining the west slope of the Sierra Nevada in the Sacramento-San Joaquin Basin, California. Inset map shows the floodplain site with the major levee removal and excavation elements (outlined in red) of the restoration project, implemented in the fall of 2014 by The Nature Conservancy. (Aerial imagery: USDA 2016)

Cosumnes River Preserve over the last several decades by The Nature Conservancy and a consortium of agencies has allowed the river to regularly access substantial portions of its former floodplain (Swenson et al., 2012). The most recent project – levee-removal on a 2.1 km^2 site implemented in the fall of 2014 – is the focus of the study here (see Figure 2).

The lower Cosumnes River is ecologically significant due to the high degree of connectivity between the river and floodplain (Opperman et al., 2010), which supports seasonal wetlands and riparian forest. This connectivity generates dynamic physical processes that rework the floodplain topography (Florsheim & Mount, 2002; Nichols & Viers, 2017) and promote ecological processes and functions, including hydrochorous dispersal and riparian vegetation succession (Trowbridge, 2007), primary and secondary productivity (Ahearn et al., 2006; Grosholz & Gallo, 2006), and spawning and rearing habitat for native floodplain fish, including Chinook salmon (*Oncorhynchus tshawytscha*) and Sacramento splittail (Jeffres et al., 2008; Ribeiro et al., 2004). The functional Cosumnes floodplain is particularly important within the larger context of dramatic declines in California's native fishes and severe ecological degradation and transformation of the Sacramento-San Joaquin Delta, into which the Cosumnes River flows (Moyle et al., 2011; Nichols et al., 1986).

Increasing temperatures and changing precipitation affect many aspects of the hydrologic cycle, including altered runoff patterns and groundwater recharge, and amplification of hydroclimatic cycling. Average air temperature in California has increased 1 °C over the last century and is expected to increase up to nearly 5 °C by the end of this century (Cayan et al., 2008; Dettinger et al., 2016). Overall precipitation changes are less clear, in part due to California's position at the transition between two large regions of the globe with opposing wet and dry trends in model scenarios (Berg & Hall, 2015). Climate change impacts are seen in California's changing hydrology in the form of declining snowmelt and earlier peak runoff (Barnett et al., 2008; Cayan et al., 2008; Mote et al., 2005; Mote et al., 2018). Hydrologic research specific to the Cosumnes River suggests shifts toward larger floods and reduced spring snowmelt floods over the last century (Booth et al., 2006; Whipple et al., 2017). Projections for California indicate increased winter streamflow and declining spring and summer flow as hydrologic regimes shift from snow- to rain-dominated (Dettinger et al., 2016; Georgakakos et al., 2012; Maurer, 2007). Years of low snowpack are more severe under anthropogenic warming (Berg & Hall, 2017), snowpack declines of up to 90% by the end of the century have been projected (Hayhoe et al., 2004; Knowles et al., 2018), and spring snowmelt runoff is projected to shift one to nearly three months earlier (Schwartz et al., 2017). Climate projections also indicate intensification of shifts between extreme wet and dry periods within an already variable hydroclimate (Berg & Hall, 2015; Swain et al., 2018; Yoon et al., 2015). More extreme floods are expected as precipitation extremes increase in number and magnitude (Cayan et al., 2008; Pierce et al., 2013). Using a 16-member ensemble, Das et al. (2013) found 50-year return-interval floods increases of 30-90% in the northern Sierra Nevada, with enhanced variability attributable to increased peak flows.

Altered timing and magnitude of runoff under climate change will impact ecosystems directly, as well as indirectly through human responses as balancing the demands for California's scarce water resources becomes increasingly challenging. Moyle et al. (2013) estimated that over 80% of California's native fishes are highly vulnerable to climate change. Understanding potential climate change impacts on runoff in the Sierra Nevada and its cascading effects is necessary for successful adaptation of California's water management for humans and the environment (Vicuña et al., 2007). The Cosumnes River provides a useful case study for exploring potential climate change impacts to a Sierra Nevada river flow regime and subsequent consequences for floodplain landscapes.

Climate change projections

This study uses climate change projections of daily streamflow for major rivers flowing into the Central Valley. These projections were derived from the Intergovernmental Panel on Climate Change (IPCC) Coupled Model Intercomparison Project Phase 5 (CMIP5; Taylor et al., 2011) in support of California's Fourth Climate Change Assessment (Pierce et al., 2016). To provide guidance for studies too computationally limited to use the full 32-model ensemble of the CMIP5 archive, two subsets of ten and four models were previously systematically selected as most appropriate for California water resource assessments (Cayan et al., 2018; Kravitz, 2017). In this study, the subset of four GCMs were used, which span the 1951-2099 water years (WY; defined as October 1 through September 30; Table 6-1). This subset includes the following GCMs: HadGEM2-ES ("warm/dry"), CNRM-CM5 ("cool/wet"), CanESM2 ("middle"), and MIROC5 ("diversity"; Cayan et al., 2018; Kravitz, 2017). Figure 6-3 shows Sacramento/Central Valley average precipitation and temperature changes associated with each of these scenarios in the end of century (WY2070-2099) period relative to modeled 1976-2005 conditions under representative concentration pathway (RCP) 8.5 (Cayan et al., 2018). The baseline historical period (using historical greenhouse gas emissions levels) spans WY1951-2005, after which future emissions scenarios forced the models. To reduce computation, runs from only the highest emissions scenario (RCP8.5) were analyzed. Current emissions align with the trajectory of this scenario (Peters et al., 2012). To further reduce computation and simplify comparisons, two 30-year time periods were used in this study, a historical (WY1951-1980) and future (WY2070-2099) period (resulting in eight 30-year model and time period combinations).

Development of daily flow projections, by Cayan et al. (2018) and Knowles et al. (2018) and supporting California's Fourth Climate Change Assessment, followed a typical progression of downscaling daily gridded temperature and precipitation from GCM output and subsequent hydrologic modeling using the downscaled climate projections as input (Xu et al., 2005). As biases are usually present in the GCM output at regional and local scales, bias-correction is standard practice. The GCM output was bias-corrected by Pierce et al. (2015) and downscaled following the Localized Constructed Analogs (LOCA) method (Pierce et al., 2014). Hydrologic modeling to generate daily unimpaired runoff projections was performed with the commonly-applied Variable Infiltration Capacity (VIC) land surface model (Liang

Table 6-1. The four global climate models, and the institutions that developed them, from the CMIP5 (Coupled Model Intercomparison Project Phase 5) archive used in this study and selected by California's Department of Water Resources Climate Change Technical Advisory Group for their ability to represent California's climate well (CDWR-CCTAG, 2015; Kravitz, 2017).

Global Climate Model	Institution
CNRM-CM5	CNRM (Centre National de Recherches Meteorologiques, Meteo-France, Toulouse, France) and CERFACS (Centre Europeen de Recherches et de Formation Avancee en Calcul Scientifique, Toulouse, France
CanESM2	CCCma (Canadian Centre for Climate Modelling and Analysis, Victoria, BC, Canada)
HadGEM2-ES	Met Office Hadley Centre, Fitzroy Road, Exeter, Devon, EX1 3PB, UK
MIROC5	JAMSTEC (Japan Agency for Marine-Earth Science and Technology, Kanagawa, Japan), AORI (Atmosphere and Ocean Research Institute, The University of Tokyo, Chiba, Japan), and NIES (National Institute for Environmental Studies, Ibaraki, Japan)

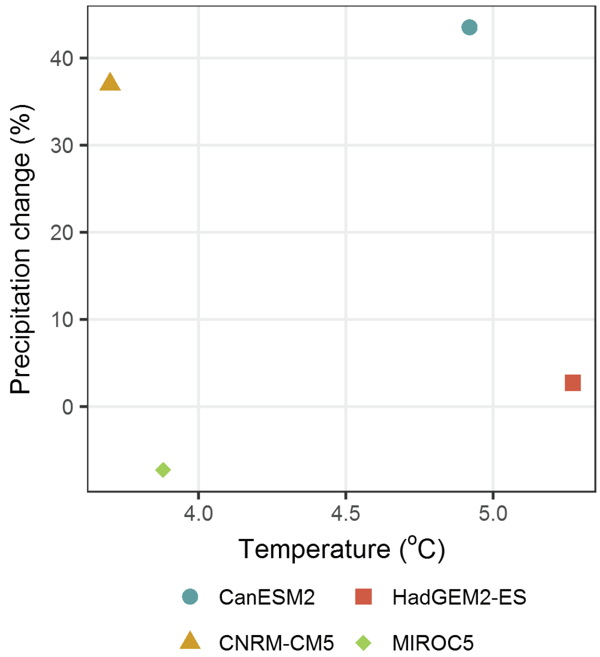

Figure 6-3. Temperature and precipitation changes for the four climate change scenarios associated with the late century period (2070-2099) over historical modeled 1976-2005 climatology (Cayan et al., 2018). Data are shown for the Sacramento/Central Valley region.

et al., 1994). Researchers then routed the 1/16°-gridded runoff output to points within the Central Valley watershed channel network, including the lower Cosumnes River, using the RVIC routing model (Cayan et al., 2018; Lohmann et al., 1996). These modeling products are currently used to study water management and ecological impacts of climate change in the Central Valley (Knowles et al., 2018).

In this study, the routed VIC daily streamflow projections for the Cosumnes River were bias corrected with a reference time series of observed historical flows (USGS MHB streamflow gage; WY1908-2017; Appendix C; Figure 6-4). This bias correction step was necessary because the VIC model is not well-calibrated for the west slope of the Sierra Nevada, resulting in a general overestimation of high flows and underestimation of low flows (Knowles & Lucas, 2015). This follows other climate change assessment studies that have applied bias correction of VIC hydrologic output (Miller et al., 2012; Snover et al., 2003). The common bias correction technique of quantile mapping was applied here (Maurer et al., 2013). This method adjusts the model distribution variance to align with observed variance of a reference data set by matching quantiles of non-exceedance probability distributions between the observed and modeled values and using this relationship to map the future period values. For daily correction, a 31-day moving window was used, following Thrasher et al. (2012). An important assumption is that the GCM biases are adequately characterized through comparisons between historical observed and modeled values. Bias correction was implemented in *R*, using the *hyfo* package (R Core Team, 2016; Xu, 2017). The bias-corrected products are not flow predictions, but rather represent a range of plausible future Cosumnes River streamflow regimes.

Across the GCM scenarios evaluated by Knowles et al. (2018), projected annual flow increases by <10%, following precipitation trends (10 GCMs, RCP4.5 and RCP8.5). These changes reflect substantial loss of Sierra Nevada snowpack as temperatures warm over the next century, including a projected increase of 33% on average in the fraction of total flow arriving prior to April 1. The four simulated Cosumnes River streamflow projections (referred to as climate change scenarios hereafter) evaluated for this study include both wetter and drier futures. Mean annual flow volume between the WY1951-1980 and WY2070-2099 periods increases 75% for CanESM2, increases 60% for CNRM-CM5, increases 11% for HadGEM2, and decreases 13% for MIROC5 (Figure 6-5). Compared to the basin-wide assessment of Knowles et al. (2018), the average increase in pre-April flow fraction is less pronounced for the Cosumnes River (12-28%). This is likely attributable to the lower fraction of runoff from snow in the Cosumnes watershed relative to the other higher-elevation watersheds of the Sierra Nevada. Most flow

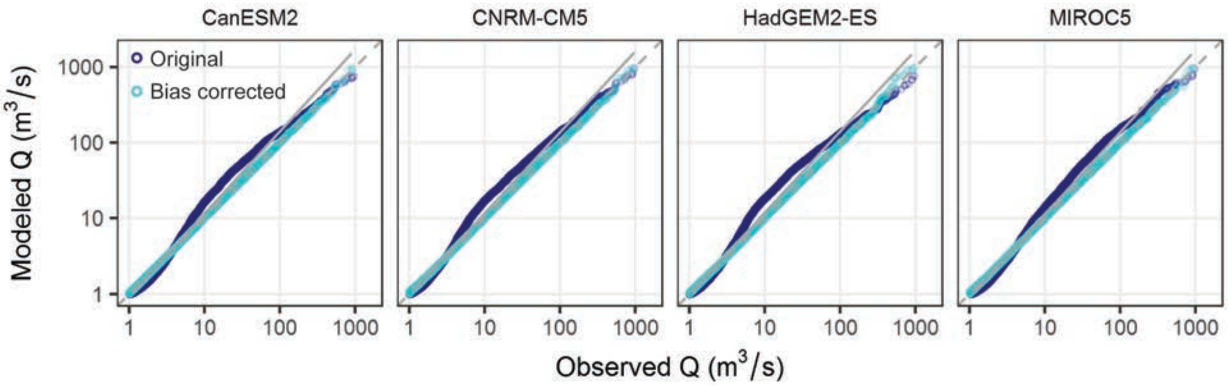

Figure 6-4. Bias corrected daily flow for the Cosumnes River. Observed data from the Michigan Bar streamflow gage (MHB, #11335000; U.S. Geological Survey, 2017) is plotted against modeled original (dark blue) and bias corrected (light blue) daily flow derived from the four climate change scenarios during the full historical period in common (1950-2005). All data are sorted prior to plotting. Only values >1 m^3/s are shown in the log-transformed plots. Plots include linear trends for original data (solid gray) and the 1:1 line (dashed gray).

increases projected for WY2070-2099 are due to increases in January and February (Figure 6-6). Monthly distributions for the WY1951-1980 and WY2070-2099 periods are significantly different for the four climate change scenarios (using the non-parametric two-sample Anderson-Darling test, p-value < 0.05). Pronounced variability increases occur in early winter, particularly in the CanESM2 and CNRM-CM5 scenarios (Figure 6-7, see Figure 6-6). From quantiles of total annual flow (water year types) established from both time periods, the number of water years associated with the wettest water year type (0.8-1.0 quantile) changes most substantially between the two periods, from a 50% decline (MIROC5) to a 400% increase (CanESM2 and CNRM-CM5). The projected number of days of flood flows inundating the floodplain – using the previously-determined threshold of 23 m^3/s (Florsheim et al., 2006; Whipple et al., 2017) – decline between the WY1951-1980 and WY2070-2099 periods (from 4-21% or 4-21 days annually). However, aligned with other research (Das et al., 2013; Knowles et al., 2018), projections for extreme daily flows also increase, as represented by increases in the number of days exceeding the 99th-percentile daily flow of the USGS MHB streamflow gage (136 m^3/s; 3.67 days/year; WY1951-1980), ranging from 0.3-18.1 days (Figure 6-8). Separating these flood days into flood events (defined by at least one day below the floodplain inundation threshold) and classifying them according to the flood types in Whipple et al. (2017) shows pronounced changes in flood type distribution. In particular, the frequency of the wettest flood type increases between 100-1400% and the later season flood types decline 12-76% between the two periods. Changes to frequencies of other flood types varied depending on the scenario.

Hydrodynamic modeling before and after restoration

Two-dimensional (2D) hydrodynamic modeling was performed for the Cosumnes floodplain restoration site in order to generate depth and velocity estimates for a range of flows, which were then used to establish gridded time series for the flow scenarios. The flow scenarios were therefore not used directly as modeling input. Two separate hydrodynamic models were developed for the pre- and post-restoration floodplain configurations using the U.S. Army Corps of Engineers' Hydrologic Engineering Center River Analysis System (HEC-RAS 5.0; Brunner, 2016). The basis of the model geometry was 2007 LiDAR, which was combined with in-channel cross-sectional and floodplain surface RTK (Real Time Kinematic GPS)

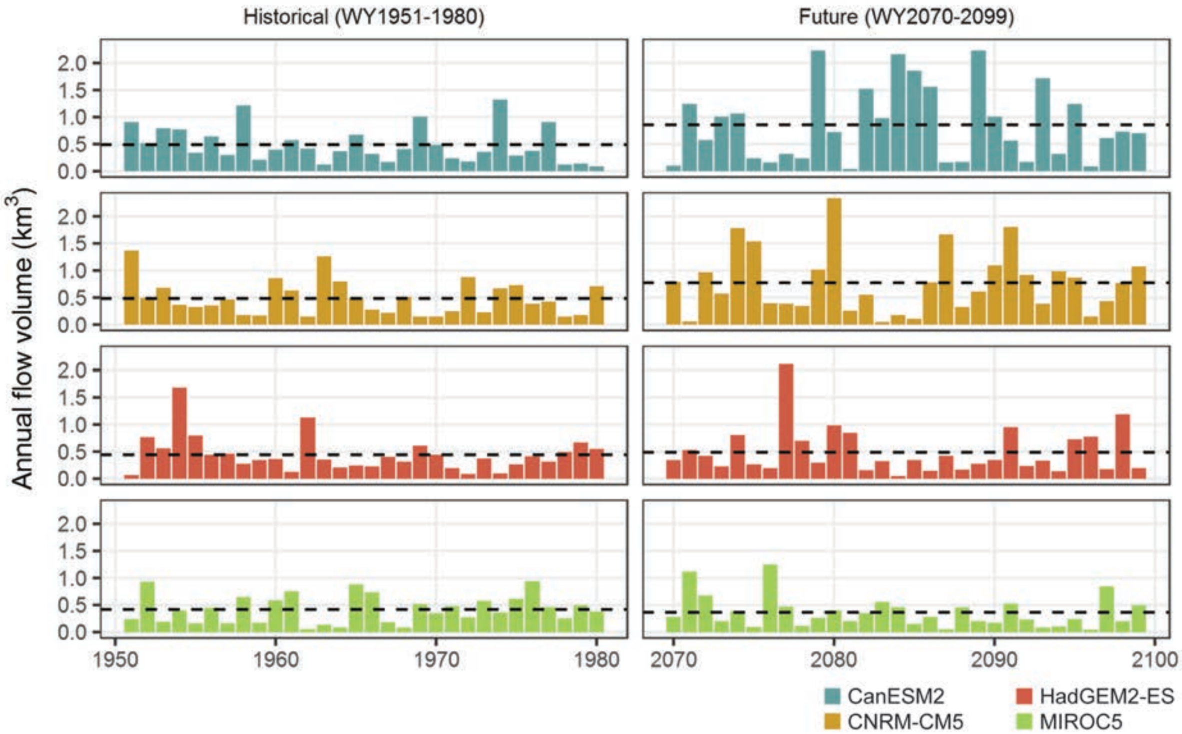

Figure 6-5. Comparison of Cosumnes River annual total flow volume between the WY1951-1980 and WY2070-2099 periods for the four climate change scenarios. Mean annual volume for each period and scenario is shown as dashed lines.

survey data (CDWR, 2010). The model structure consisted of one-dimensional (1D) channel flow coupled via lateral weir structures with the 2D modeled floodplain area. The computational grid for the floodplain site was unstructured, and the subgrid capacity of the HEC-RAS model allowed for relatively large computational cells (110-4,200 m^2) while maintaining model output at the topographic input resolution of 1 m^2. Input hydrology was derived from 15-minute USGS MHB streamflow gage data. Unsteady model simulations were run using a 10-second computational time step. Calibration for pre- and post-restoration models was performed by iteratively adjusting roughness values and weir coefficient parameters using observed water surface elevations at several in-channel and floodplain locations, which were monitored both before and after restoration.

The pre- and post-restoration hydrodynamic models were subsequently run for an artificial hydrograph that spanned a range of flows. At intervals ranging from 10-100 m^3/s, flow was held constant for several days (of model simulation time) to allow depth and velocity on the floodplain to become representative of the inflow. Intervals between selected flows were smaller where the areal extent of flooding changed rapidly with flow. Modeling was conducted this way (as opposed to steady state for a range of flows) in order to capture the substantial areas that become ponded on the receding limb of the hydrograph. Further description of the hydrodynamic modeling applied in this study can be found in Chapter 4 and Appendix A.

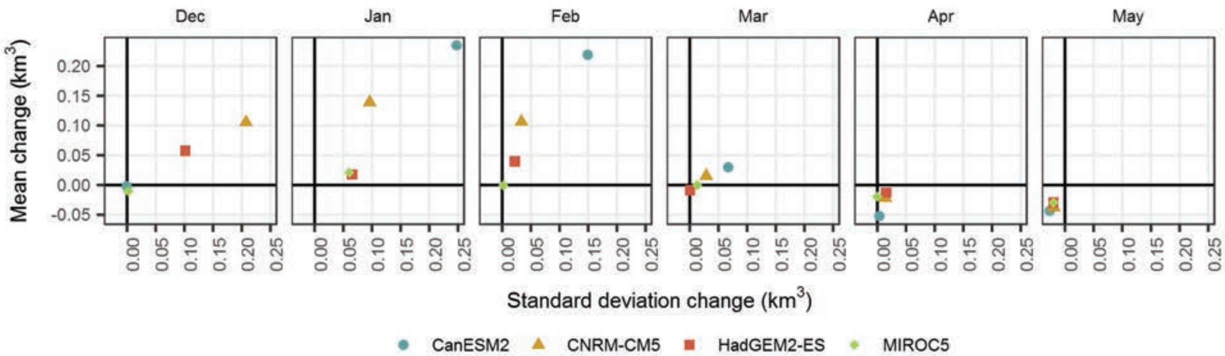

Figure 6-6. Monthly flow volume change. Differences between the WY2070-2099 and WY1951-1980 (future minus historical) period mean and standard deviation of monthly flow volume for each climate change scenario.

Hydrospatial analysis

The approach for spatiotemporal analysis of floodplain inundation patterns followed in this study applies the hydrospatial analysis framework established in Chapter 4. The 2D hydrodynamic modeling output was used to generate daily time series of spatially-resolved depth and velocity associated with both pre- and post-restoration topography for the WY1951-1980 and WY2070-2099 periods of the four climate change scenarios (for a total of 16 30-year periods; see Figure 6-1). Gridded depth and velocity estimates (9 m^2 resolution, resampled from 1 m^2) were made using cell-by-cell piece-wise linear interpolation based on the hydrodynamic modeling output at known flows on the rising and falling limb of a hydrograph. This interpolation routine was performed for each day of flow above the 23 m^3/s floodplain inundation threshold (Florsheim et al., 2006; Whipple et al., 2017), averaging 2,695 days per 30-year period and scenario (which is on average ~90 d/yr).

The multi-metric hydrospatial analysis approach here and introduced in Chapter 4 follows ecohydrological and environmental flows literature that evaluates flow regime characteristics and human-induced change using a range of variables (e.g., Dyer et al., 2013; Kennard et al., 2010; Olden & Poff, 2003; Poff, 1996; Richter et al., 1996). The application of spatially-dependent criteria aligns with recent research evaluating various physical characteristics of floodplain and wetland inundation

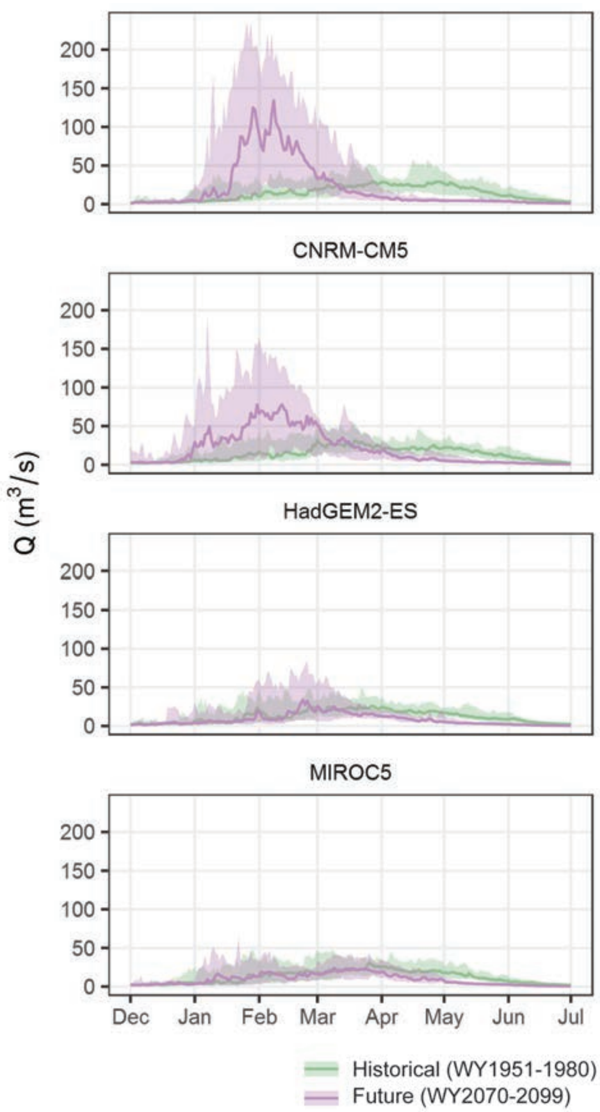

Figure 6-7. Median daily flow comparison for the WY1951-1980 and WY2070-2099 periods across high flow months for the four climate change scenarios. Shading shows the interquartile range.

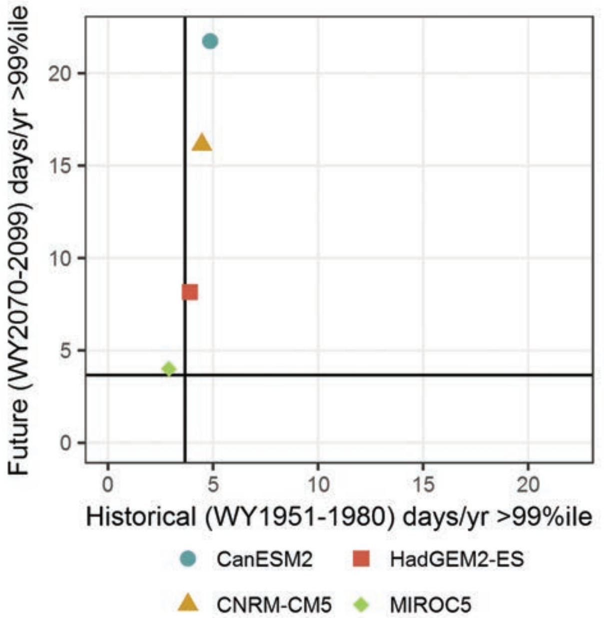

Figure 6-8. Change in extreme flows. Annual average number of days for each climate change scenario exceeding the historical observed 99th-percentile daily flows (black lines, MHB, #11335000; U.S. Geological Survey, 2017) for the WY2070-2099 versus WY1951-1980 periods.

(e.g., Cienciala & Pasternack, 2017; Coleman et al., 2015; Stone et al., 2017). The metrics for this analysis were selected to describe physical floodplain inundation dynamics valuable for ecological interpretation and management, including extent, depth, velocity, duration, timing, frequency, and connectivity (Table 6-2; see Chapter 4; Bouska et al., 2016). For extent, daily inundated area, maximum inundated area, and area-days (area summed over time) was calculated. Inundation depth and flow velocity were assessed in space and time. Spatially distributed inundation duration was calculated for each flood event and for each water year. The timing of flood inundation was assessed using the water year day of the centroid floodplain flood volume. Frequency of inundation was calculated at each grid cell for flood events and years. Connectivity was assessed from daily determinations of whether inundated cells maintained a surface water connection to the river.

The physical floodplain metrics were further evaluated for a specific floodplain ecological function: rearing habitat availability for the native floodplain fish species, Sacramento splittail. Splittail is the primary native fish species occupying the lower Cosumnes River floodplain, and has been identified as an important indicator species for floodplain functionality (Cloern et al., 2011; Matella & Merenlender, 2015; Moyle et al., 2004; Sommer et al., 2014). Habitat availability calculations involved the common technique of habitat suitability criteria applied to physical parameters (e.g., Guse et al., 2015; Stalnaker, 1979). Criteria for depth, velocity, duration, timing, and connectivity were used, which were established based on published literature and expert opinion (Figure 6-9; see Appendix B; Moyle, 2017; Sommer, 2017; Suddeth, 2014). Criteria were applied cell-by-cell on a daily basis and combined using the geometric mean, resulting in daily gridded cell suitability index (*CSI*) values that were then summed to estimate daily weighted usable area (*WUA*; see Chapter 5). Given the similarity between requirements for spawning and rearing, only habitat availability for the juvenile rearing life stage is reported here.

Metrics were summarized and compared across space and time, producing tabular, graphical, and geospatial output. Comparisons between time periods, climate change scenarios, and restoration configurations were supported by non-parametric measures of coefficient of dispersion (CD), deviation factor (DF), and the non-parametric two-sample Anderson-Darling test to detect differences in distributions of annual values (p-value < 0.05; Anderson & Darling, 1954; Stephens, 1986; The Nature Conservancy, 2009). All spatial and statistical analyses were performed in *R* (R Core Team, 2016), with use of the *raster* package for processing the large spatial datasets (Hijmans, 2015).

Table 6-2. Metrics used for physical description of spatiotemporal inundation patterns, including spatial and temporal characteristics and examples of ecological relevance.

Metric Group	Metric	Description	Spatially integrated?	Temporally integrated?	Examples of ecological relevance
Inundation extent	MaxA (km^2)	Maximum daily inundated area in a water year	X	X	Potentially available aquatic habitat; Primary, secondary, and fish productivity; Riparian vegetation community composition (Bayley, 1991; Sommer et al., 1997; Opperman et al., 2010; Ward and Stanford 1995)
	ADay (km^2-day)	Daily inundated area summed over a water year	X	X	
	A (km^3)	Daily inundated area	X		
Depth	MaxDm (m)	Spatial mean of maximum flood event depth, as mean for a water year	X	X	Fish habitat suitability; Riparian vegetation community composition (Opperman et al. 2010; Florsheim and Mount 2002; Maddock 1999; Guse et al. 2015)
	Dm (m)	Daily spatial mean depth	X		
	MaxD (m)	Maximum water year depth		X	
Velocity	MVm (m/s)	Spatial mean of mean flood event velocity, as mean for a water year	X	X	Fish habitat suitability, Riparian vegetation community composition (Opperman et al. 2010; Florsheim and Mount 2002; Maddock 1999; Guse et al. 2015)
	Vm (m/s)	Daily spatial mean velocity	X		
	MV (m/s)	Mean flood event velocity, as mean for a water year		X	
Duration	Durm (day)	Mean water year spatial mean flood event duration	X	X	Fish life history requirements; Primary and secondary productivity; Riparian vegetation community composition (Mahoney and Rood 1998; Junk et al. 1989; Gallardo et al. 2009)
	Dur (day)	Mean water year flood event duration		X	
Timing	WYDCnVol (WY day)	Water year day of centroid flood volume	X	X	Life history requiremnts (e.g., spawning, germination) (Moyle et al. 2004; Stella et al. 2006)
Frequency	IFN$_m$ (count)	Mean water year spatial mean of flood event number of times inundated	X	X	Primary and secondary productivity; Nutrient exchange; Riparian vegetation community composition (Poff et al. 1997; Robertson et al 2001; Mahoney and Rood 1998; Junk et al. 1989)
	IFN (count)	Water year sum of flood event number of times inundated		X	
Connectivity	CADay (km^2-day)	Daily connected inundated area summed over a water year	X	X	Habitat accessibility; Habitat diversity; Primary and secondary productivity, Nutrient exchange (Ahearn et al. 2006; Junk et al. 1989; Ward and Stanford 1995)
	DCADay (km^2-day)	Daily disconnected inundated area summed over a water year	X	X	
	CA (km^2)	Daily connected area	X		
	DCA (km^2)	Daily disconnected area	X		
	C (%)	Percent of water year connected		X	
	DC (%)	Percent of water year disconnected		X	
Habitat availability	WUADay (km^2-day)	Daily weighted usable area summed over a water year	X	X	Physical habitat suitability relating to depth, velocity, duration, timing, and connectivity (see Appendix B; Moyle et al. 2004; Sommer et al. 2014; Suddeth 2014)
	WUA (km^2)	Daily weighted usable area	X		
	CSI (-)	Mean water year cell suitability index		X	

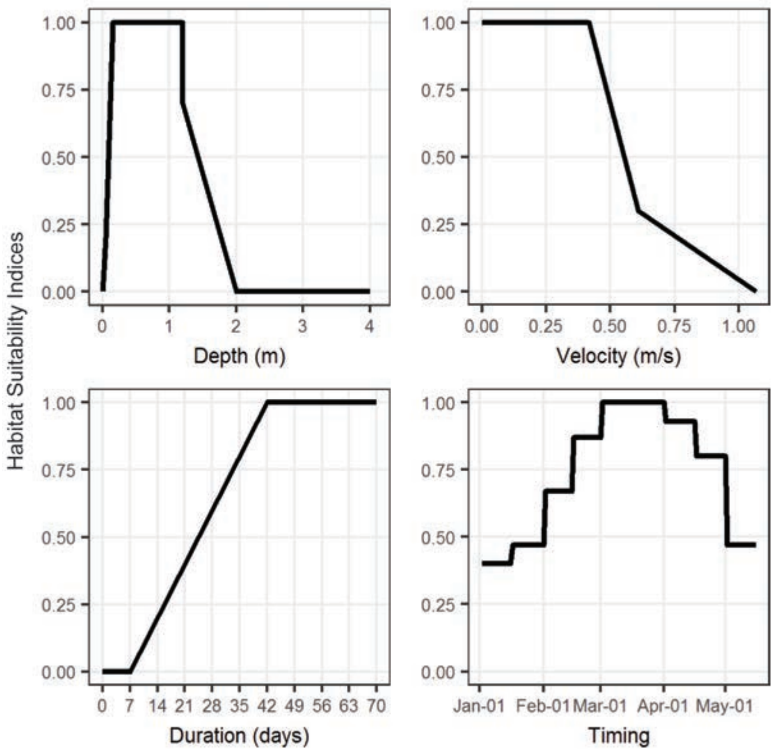

Figure 6-9. Habitat suitability curves for juvenile Sacramento splittail constructed from indices of suitability for depth, velocity, inundation duration, and seasonal timing based on literature synthesis and expert opinion (see Appendix B).

RESULTS

Inundation extent

The water year maximum extent of inundated area (*MaxA*; see Table 6-2) within the Cosumnes River floodplain site showed significant increases (as indicated by the two-sample Anderson-Darling test, p-value < 0.05) between the historical (WY1951-1980) and future (WY2070-2099) period for the three wetter climate change scenarios (CanESM2, CNRM-CM5, and HadGEM2-ES; Table 6-3; Figure 6-10). Annual accumulated inundated area (*ADay*) increased and annual distributions were significantly different between the two time periods for CanESM2 and CNRM-CM5 under the pre-restoration floodplain topographic configuration and only CNRM-CM5 under the post-restoration configuration. Deviation factors (DFs) for median conditions were greatest (an over two-fold increase) for CanESM2 and CNRM-CM5 under pre-restoration conditions. Lower interannual variability of *MaxA* and *ADay* was found for CanESM2 and CNRM-CM5, indicated by deviation factors (DFs) of coefficient of deviation (CD). For each climate change scenario and time period, post-restoration inundation extent was higher than pre-restoration. And, for MIROC5 (the drier climate change scenario) and HadGEM2-ES, post-restoration *ADay* in the WY1951-1980 period was greater than pre-restoration *ADay* in the WY2070-2099 period. These results are supported by exceedance probabilities generated from the water year summaries (Figure 6-11). For example, for CanESM2 and CNRM-CM5, exceedance probabilities of floodplain inundation reaching an accumulated area of 50 km^2-day ranged from 25-50% in the WY1951-1980 period, but 50-75% in the WY2070-2099 period. Exceedance probabilities for MIROC5 declined slightly between the periods, but the magnitudes of change were in line with differences between restoration configurations. Compared to differences in water year flow volume, DFs for median conditions were less in absolute terms for *MaxA* except for HadGEM2-ES, while *ADay* DFs were greater for CanESM2 and CNRM-CM5 (see Table 6-3).

Across the climate change scenarios, the spread in DFs for median conditions was greater than between restoration configurations, which is particularly pronounced for *ADay* (Figure 6-12). This difference is also evident in cumulative distributions for the two time periods (Figure 6-13). While in the WY1951-1980 period all post-restoration accumulated inundated area was higher than pre-restoration, CanESM2 and CNRM-CM5 accumulated inundated area over the WY2070-2099 period very rapidly exceeded that of HadGEM2-ES and MIROC5. However, the post-restoration configuration had less of

Table 6-3. Summary statistics for hydrospatial metrics. Bootstrapped 95% confidence intervals (CI) for the median values and coefficients of dispersion (CD; (75th%ile-25th%ile)/median). Deviation factors are computed for medians and CDs of the WY1951-1980 and WY2070-2099 periods. The two-sample Anderson-Darling test results are also shown for purposes of comparing time periods, with those associated with a *p*-value < 0.05 bolded. Water year projected Cosumnes River flow volume (top rows) is included for comparison against the floodplain metrics.

Metric group	Metric	Restoration configura-tion	Climate change scenario	Historical (WY1951-1980)				Future (WY2070-2099)				Deviation factors		Anderson-Darling p-value
				Mean	Median	CI	CD	Mean	Median	CI	CD	Median	CD	
Flow volume	Vol (m^3)	*NA*	CanESM2	4.89E+08	3.87E+08	(2.11e+08-4.66e+08)	1.07	8.58E+08	7.10E+08	(3.80e+08-1.10e+09)	1.41	0.83	0.33	**0.029**
			CNRM.CM5	4.83E+08	4.06E+08	(2.39e+08-5.49e+08)	1.12	7.73E+08	6.91E+08	(4.39e+08-9.91e+08)	0.94	0.70	-0.16	**0.034**
			HadGEM2.ES	4.42E+08	3.68E+08	(2.85e+08-4.45e+08)	0.76	4.88E+08	3.38E+08	(1.99e+08-4.25e+08)	1.52	-0.08	0.99	0.569
			MIROC5	4.19E+08	3.88E+08	(2.69e+08-5.24e+08)	1.02	3.65E+08	2.72E+08	(1.16e+08-3.42e+08)	1.07	-0.30	0.04	0.474
Extent	MaxA (km^2)	pre	CanESM2	1.39	1.78	(1.55-2.39)	0.80	1.79	2.05	(2.03-2.07)	0.06	0.15	-0.93	**0.002**
			CNRM.CM5	1.49	1.81	(1.58-2.26)	0.63	1.83	2.04	(2.01-2.12)	0.13	0.13	-0.80	**0.008**
			HadGEM2.ES	1.37	1.58	(1.24-1.99)	0.78	1.85	2.04	(2.02-2.28)	0.17	0.29	-0.78	**0.004**
			MIROC5	1.39	1.73	(1.52-2.27)	0.74	1.27	1.49	(1.08-2)	0.88	-0.14	0.19	0.739
		post	CanESM2	1.52	1.72	(1.44-2.04)	0.48	1.79	2.02	(1.99-2.04)	0.08	0.18	-0.84	**0.002**
			CNRM.CM5	1.62	1.72	(1.44-1.99)	0.40	1.86	2.01	(1.98-2.13)	0.17	0.17	-0.57	**0.009**
			HadGEM2.ES	1.49	1.52	(1.18-1.64)	0.45	1.84	2.01	(1.99-2.33)	0.21	0.32	-0.53	**0.004**
			MIROC5	1.48	1.63	(1.38-1.86)	0.46	1.38	1.48	(1.15-1.62)	0.50	-0.09	0.08	0.745
	ADay (km^2-day)	pre	CanESM2	35.52	21.04	(0.71-28.86)	2.14	64.89	57.27	(25.62-90.58)	1.44	1.72	-0.33	**0.040**
			CNRM.CM5	34.05	22.95	(5.61-35.63)	1.86	63.95	56.80	(29.89-78.24)	1.17	1.48	-0.37	**0.009**
			HadGEM2.ES	30.05	22.56	(10.74-27.72)	1.29	36.27	24.62	(9.16-33.23)	1.47	0.09	0.14	0.495
			MIROC5	29.62	23.61	(11.38-34.95)	1.56	26.08	15.97	(3.14-21.41)	1.46	-0.32	-0.06	0.458
		post	CanESM2	52.81	36.45	(10.54-52.45)	1.83	72.51	73.04	(44.62-115.36)	1.27	1.00	-0.31	0.203
			CNRM.CM5	51.09	40.08	(11.39-63.42)	1.65	78.82	75.69	(45.75-101.52)	0.97	0.89	-0.41	**0.043**
			HadGEM2.ES	44.26	36.88	(21.08-49.05)	1.18	45.29	32.84	(9.97-44.86)	1.56	-0.11	0.32	0.832
			MIROC5	45.56	39.62	(28.1-64.63)	1.45	37.47	25.76	(2.33-36.68)	1.51	-0.35	0.04	0.523
Depth	MaxDm (m)	pre	CanESM2	0.52	0.48	(0.35-0.53)	0.42	0.81	0.66	(0.44-0.7)	0.80	0.39	0.88	**<0.001**
			CNRM.CM5	0.50	0.45	(0.4-0.49)	0.38	0.70	0.65	(0.55-0.76)	0.52	0.44	0.37	**<0.001**
			HadGEM2.ES	0.47	0.44	(0.37-0.5)	0.43	0.61	0.56	(0.45-0.65)	0.47	0.27	0.09	**0.005**
			MIROC5	0.51	0.46	(0.35-0.51)	0.43	0.48	0.44	(0.37-0.47)	0.38	-0.04	-0.11	0.623

Metric group *(cont.)*	Metric	Restoration configura-tion	Climate change scenario	Historical (WY1951-1980)				Future (WY2070-2099)				Deviation factors		Anderson-Darling p-value
				Mean	Median	CI	CD	Mean	Median	CI	CD	Median	CD	
Depth (*cont.*)	MaxDm (m, *cont.*)	post	CanESM2	0.50	0.44	(0.3-0.48)	0.54	0.78	0.63	(0.42-0.65)	0.67	0.43	0.25	**0.002**
			CNRM.CM5	0.48	0.44	(0.41-0.47)	0.51	0.67	0.63	(0.56-0.71)	0.42	0.44	-0.17	**<0.001**
			HadGEM2.ES	0.45	0.45	(0.38-0.52)	0.55	0.59	0.53	(0.41-0.61)	0.43	0.20	-0.22	**0.004**
			MIROC5	0.50	0.48	(0.37-0.55)	0.51	0.46	0.43	(0.36-0.48)	0.41	-0.11	-0.18	0.495
Velocity	MVm (m/s)	pre	CanESM2	0.029	0.026	(0.02-0.04)	0.84	0.064	0.050	(0.03-0.06)	0.97	0.91	0.16	**<0.001**
			CNRM.CM5	0.028	0.023	(0.01-0.03)	0.91	0.050	0.044	(0.04-0.05)	0.60	0.96	-0.33	**<0.001**
			HadGEM2.ES	0.028	0.023	(0.01-0.03)	0.97	0.047	0.036	(0.02-0.04)	0.91	0.53	-0.06	**0.001**
			MIROC5	0.031	0.026	(0.02-0.04)	0.80	0.026	0.019	(0.01-0.02)	1.23	-0.28	0.54	0.485
		post	CanESM2	0.046	0.040	(0.02-0.05)	0.82	0.088	0.070	(0.04-0.08)	0.76	0.74	-0.07	**<0.001**
			CNRM.CM5	0.043	0.046	(0.04-0.06)	0.60	0.080	0.074	(0.06-0.08)	0.59	0.61	-0.02	**<0.001**
			HadGEM2.ES	0.045	0.043	(0.03-0.05)	0.73	0.071	0.066	(0.06-0.08)	0.51	0.52	-0.31	**<0.001**
			MIROC5	0.049	0.049	(0.04-0.06)	0.61	0.042	0.040	(0.03-0.04)	0.53	-0.20	-0.12	0.328
Duration	Durm (day)	pre	CanESM2	8.32	7.19	(6.43-8.26)	0.53	15.74	9.16	(1.63-11.68)	1.52	0.27	1.89	0.103
			CNRM.CM5	7.66	6.82	(6.09-7.43)	0.42	11.17	8.91	(6.69-10.52)	0.78	0.31	0.84	0.088
			HadGEM2.ES	6.96	6.60	(5.61-7.34)	0.43	7.44	5.84	(4.36-6.3)	0.40	-0.12	-0.05	0.266
			MIROC5	8.04	7.22	(6.43-7.54)	0.38	7.10	6.52	(5.91-6.88)	0.20	-0.10	-0.47	0.203
		post	CanESM2	9.23	7.08	(4.01-8.3)	0.67	16.50	10.56	(3.43-14.21)	1.48	0.49	1.21	0.134
			CNRM.CM5	8.37	7.04	(5.83-8.24)	0.67	12.40	10.53	(8.45-12.93)	0.56	0.50	-0.17	0.061
			HadGEM2.ES	7.17	6.56	(4.9-7.55)	0.52	7.53	5.40	(2.57-5.93)	0.67	-0.18	0.29	0.350
			MIROC5	9.05	7.53	(5.7-9.12)	0.53	7.34	6.77	(6.39-7.63)	0.30	-0.10	-0.44	0.343
Timing	WYDCnVol (WY day)	pre	CanESM2	145.73	148.50	(128-170)	0.39	132.77	130.00	(128-137.5)	0.12	-0.12	-0.70	0.135
			CNRM.CM5	133.37	133.00	(106.5-151)	0.50	116.33	120.00	(112-137.5)	0.31	-0.10	-0.39	0.107
			HadGEM2.ES	137.40	131.00	(97-144.5)	0.45	125.60	128.00	(111-146.5)	0.40	-0.02	-0.12	0.277
			MIROC5	145.47	143.50	(118-164)	0.43	133.83	141.50	(131-162)	0.32	-0.01	-0.27	0.455
		post	CanESM2	149.53	157.00	(145-182.5)	0.36	133.87	131.00	(129.5-139.5)	0.10	-0.17	-0.72	0.053
			CNRM.CM5	136.83	141.00	(122-166.5)	0.42	117.63	122.00	(113-140.5)	0.28	-0.13	-0.34	**0.048**
			HadGEM2.ES	139.43	132.50	(99-147.5)	0.41	131.50	133.50	(120.5-146)	0.29	0.01	-0.30	0.438
			MIROC5	147.03	147.00	(125-163.5)	0.41	136.23	145.00	(135-166.5)	0.29	-0.01	-0.29	0.460
Frequen-cy	IFNm (count)	pre	CanESM2	6.67	5.97	(3.81-7.08)	0.74	4.76	4.56	(3.83-5)	0.46	-0.24	-0.38	**0.015**
			CNRM.CM5	7.33	6.53	(4.34-8)	0.71	5.70	5.87	(5.08-7.24)	0.56	-0.10	-0.22	0.088
			HadGEM2.ES	6.98	7.60	(6.68-9.19)	0.44	6.24	6.38	(5.73-7.51)	0.45	-0.16	0.03	0.159
			MIROC5	5.94	6.19	(4.81-7.75)	0.58	5.74	5.92	(4.81-6.72)	0.56	-0.04	-0.04	0.889

Metric group *(cont.)*	Metric	Restoration configura-tion	Climate change scenario	Historical (WY1951-1980)				Future (WY2070-2099)				Deviation factors		Anderson-Darling p-value
				Mean	Median	CI	CD	Mean	Median	CI	CD	Median	CD	
Frequen-cy (*cont.*)	IFNm (count, *cont.*)	post	CanESM2	6.86	6.17	(3.94-7.12)	0.79	4.63	4.41	(3.63-4.91)	0.47	-0.29	-0.40	**0.003**
			CNRM.CM5	7.49	6.86	(4.34-8.67)	0.68	5.65	5.40	(3.98-6.35)	0.52	-0.21	-0.23	0.061
			HadGEM2.ES	7.27	7.89	(6.87-9.26)	0.40	6.28	6.36	(5.83-7.44)	0.46	-0.19	0.17	0.068
			MIROC5	6.23	6.59	(5.37-8.47)	0.57	5.90	6.12	(5-7.08)	0.53	-0.07	-0.07	0.644
Connec-tivity	DCADay (km^2-day)	pre	CanESM2	7.23	5.66	(2.3-7.1)	1.20	6.86	7.60	(6.94-9.67)	0.66	0.34	-0.45	0.901
			CNRM.CM5	7.02	6.63	(4.95-9.49)	0.94	9.10	9.20	(6.42-10.26)	0.71	0.39	-0.25	**0.033**
			HadGEM2.ES	6.80	6.54	(4.12-8.7)	1.05	7.50	7.18	(5.83-9.13)	0.65	0.10	-0.38	0.572
			MIROC5	6.73	6.21	(3.84-8.93)	1.09	5.54	5.20	(3.45-6.8)	0.78	-0.16	-0.29	0.449
		post	CanESM2	6.82	6.80	(5.8-8.08)	0.49	5.33	5.32	(4.6-5.78)	0.45	-0.22	-0.08	**0.046**
			CNRM.CM5	7.11	6.44	(3.73-8.42)	0.96	7.11	7.48	(6.13-8.66)	0.54	0.16	-0.44	0.809
			HadGEM2.ES	7.08	7.64	(6.95-9.26)	0.65	6.32	6.04	(4.81-6.77)	0.57	-0.21	-0.13	0.224
			MIROC5	6.43	6.78	(5.92-8.84)	0.85	5.55	5.68	(4.68-6.67)	0.73	-0.16	-0.14	0.358
Habitat	WUADay (km^2-day)	pre	CanESM2	12.15	5.64	(-1.75-8.52)	2.68	31.28	20.69	(-2.46-36.99)	2.30	2.67	-0.14	0.055
			CNRM.CM5	12.30	4.67	(-4.12-7.66)	4.00	23.58	18.27	(2.07-28.64)	1.70	2.91	-0.57	**0.041**
			HadGEM2.ES	9.72	6.60	(3.39-10.44)	1.80	10.59	3.41	(-3.75-5.5)	3.67	-0.48	1.04	0.965
			MIROC5	9.47	6.09	(1.4-10.73)	1.83	8.76	4.12	(0.13-6.93)	2.22	-0.32	0.21	0.542
		post	CanESM2	23.23	10.62	(-4.54-16.76)	2.94	40.70	32.55	(11.49-56.76)	2.17	2.07	-0.26	0.127
			CNRM.CM5	23.00	10.14	(-11.56-17.19)	3.32	36.59	35.21	(14.49-53.03)	1.37	2.47	-0.59	0.061
			HadGEM2.ES	17.38	11.83	(2.77-18.78)	2.02	17.08	7.12	(-8.01-12.1)	3.42	-0.40	0.69	0.848
			MIROC5	19.44	10.19	(-2.55-18.18)	2.44	15.79	7.45	(-2.56-12.67)	2.48	-0.27	0.02	0.504

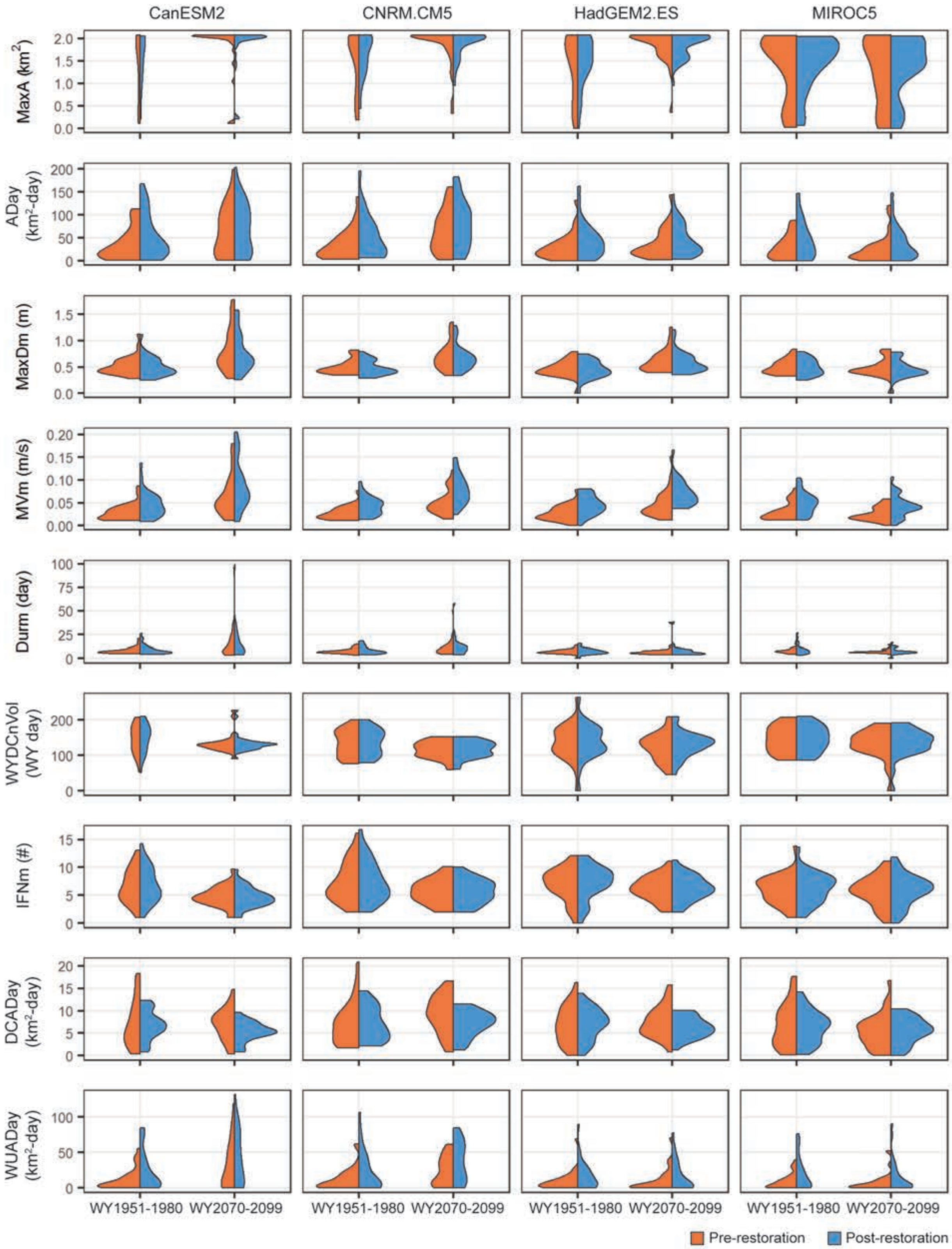

Figure 6-10. Violin plots showing the distribution of hydrospatial metrics summarized by water year under pre- (orange) and post-restoration (blue) configurations for the WY1951-1980 and WY2070-2099 periods of each climate change scenario. Note the different scales and units on the metrics.

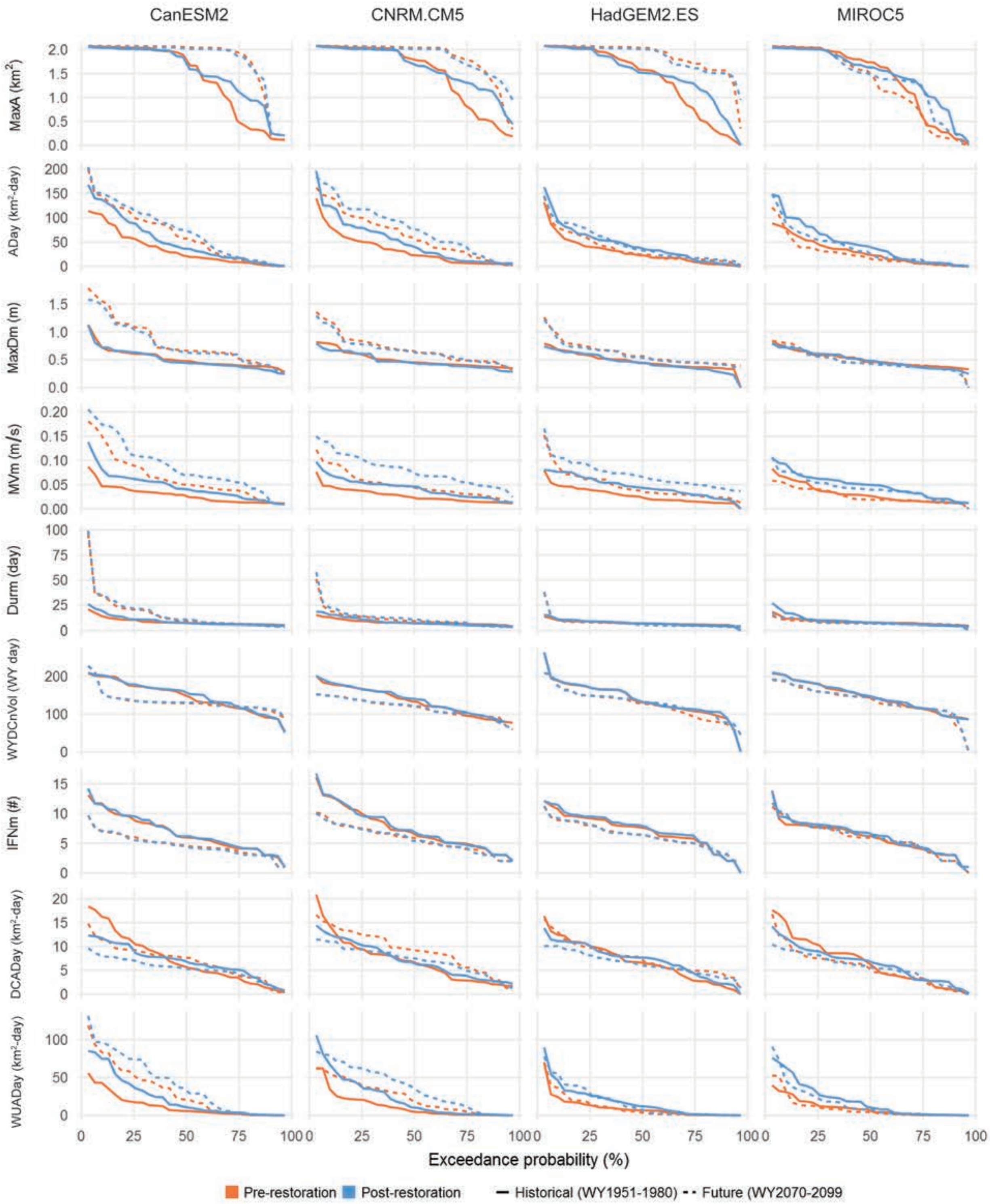

Figure 6-11. Empirical exceedance probability plots for each combination of hydrospatial metric, climate change scenario, time period, and restoration configuration.

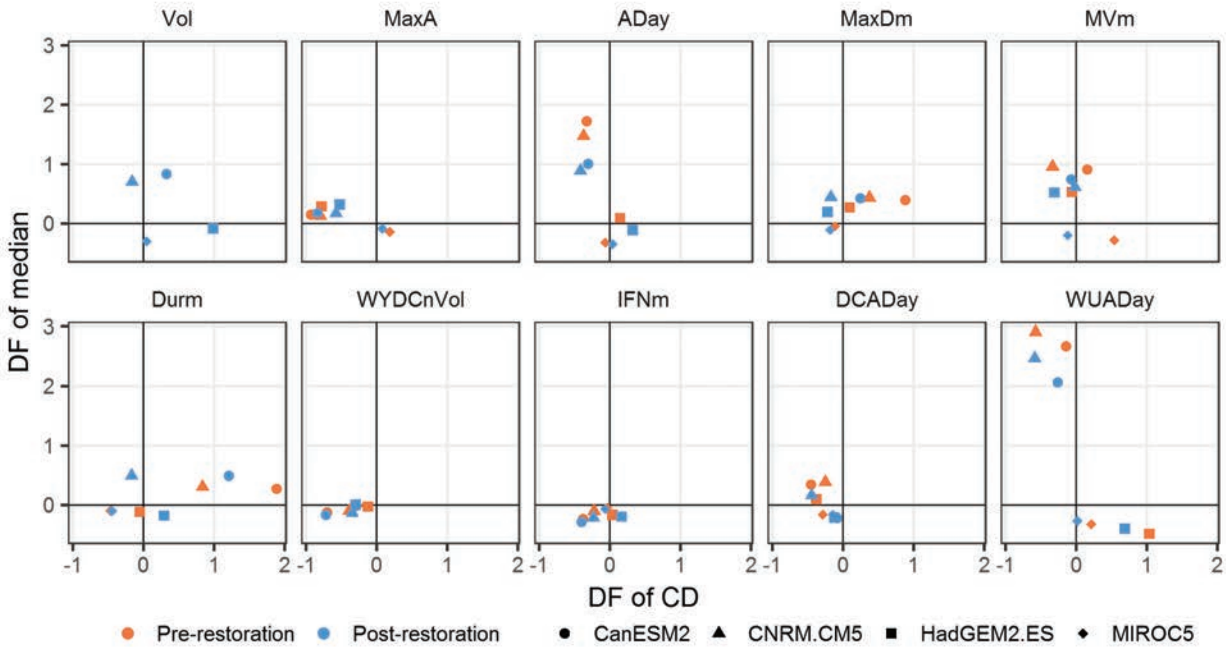

Figure 6-12. Change between WY1951-1980 and WY2070-2099. Deviation factors (DFs) of medians and coefficients of deviation (CD) for hydrospatial metrics comparing the WY2070-2099 and WY1951-1980 periods for each climate change scenario and restoration configuration. The upper left plot shows the DFs for the Cosumnes River water year flow volume projections for comparison purposes.

an overall impact with regard to increasing inundation extent in the WY2070-2099 over WY1951-1980 period.

Across all climate change scenarios, the median and interquartile range of daily inundated area was substantially reduced in the spring months between the WY1951-1980 and WY2070-2099 time periods, aligned with changes in the daily flow projections (Figure 6-14). Also, CanESM2 and CNRM-CM5 stand out with much greater median and extreme conditions in January and February, again largely aligned with the flow projections. Median inundated area was substantially higher in February for CNRM-CM5 post-restoration compared to pre-restoration for the WY1951-1980 period. Whereas median conditions were positive through early- to mid-May in the WY1951-1980 period, median conditions dropped to zero much earlier in the WY2070-2099 period across all climate change scenarios, coming to zero in mid-March (CanESM2) to mid-April (CNRM-CM5). The post-restoration configuration resulted in increased inundated area in the early spring period (March). Daily DFs of median and mean conditions present very different distributions (Figure 6-15). For median conditions, positive deviations were pronounced for CanESM2 and CNRM-CM5 in late January to February. DF of mean was much less in magnitude and higher in the earlier winter for HadGEM2-ES and MIROC5 (with large spikes in November), and somewhat higher in January to February for CanESM2 and CNRM-CM5.

Inundation depth and flow velocity

Maximum depth (*MaxDm*) and mean velocity (*MVm*) increased significantly in the WY2070-2099 period over the WY1951-1980 period for the three wetter climate change scenarios (CanESM2, CNRM-CM5 and HadGEM2-ES; see Table 6-3). These climate change scenarios are associated with pronounced increased metric values for water years with extremely high values (i.e., low exceedance probabilities; see Figure 6-10, Figure 6-11). *MaxDm* increased from approximately 0.5 m to 0.6-0.8 m and median *MVm* increased from 0.03 m/s to 0.05-0.06 m/s. Though median annual *MVm* for MIROC5 decreased between the two time periods, the annual distributions were not significantly different. Median conditions less than doubled, with the highest being *MVm* pre-restoration (see Figure 6-12). Results suggest that variability decreased fairly consistently for *MVm*, while *MaxDm* variability increased for CanESM2, CNRM-CM5 and HadGEM2-ES under pre-restoration conditions, but increased only for CanESM2 under post-restoration conditions. These trends show some similarities to DFs of annual flow volume, but are not consistent across climate change scenarios or restoration configurations. Daily distributions do show similar patterns when comparing daily flow (and inundated area) to depth and velocity (see Figure 6-14). However, median depth has a stepped pattern in time, with mean depths around 0.25 m for most flows except for extreme conditions in late-January through February for CanESM2 and CNRM-CM5. Distributions shifted to earlier times within the water year overall. Daily DFs were highly variable and positive in the winter months, with negative DFs in the later spring months (see Figure 6-15). Spatial distribution of *MaxD* (mean water year maximum depth) shows that patterns were more

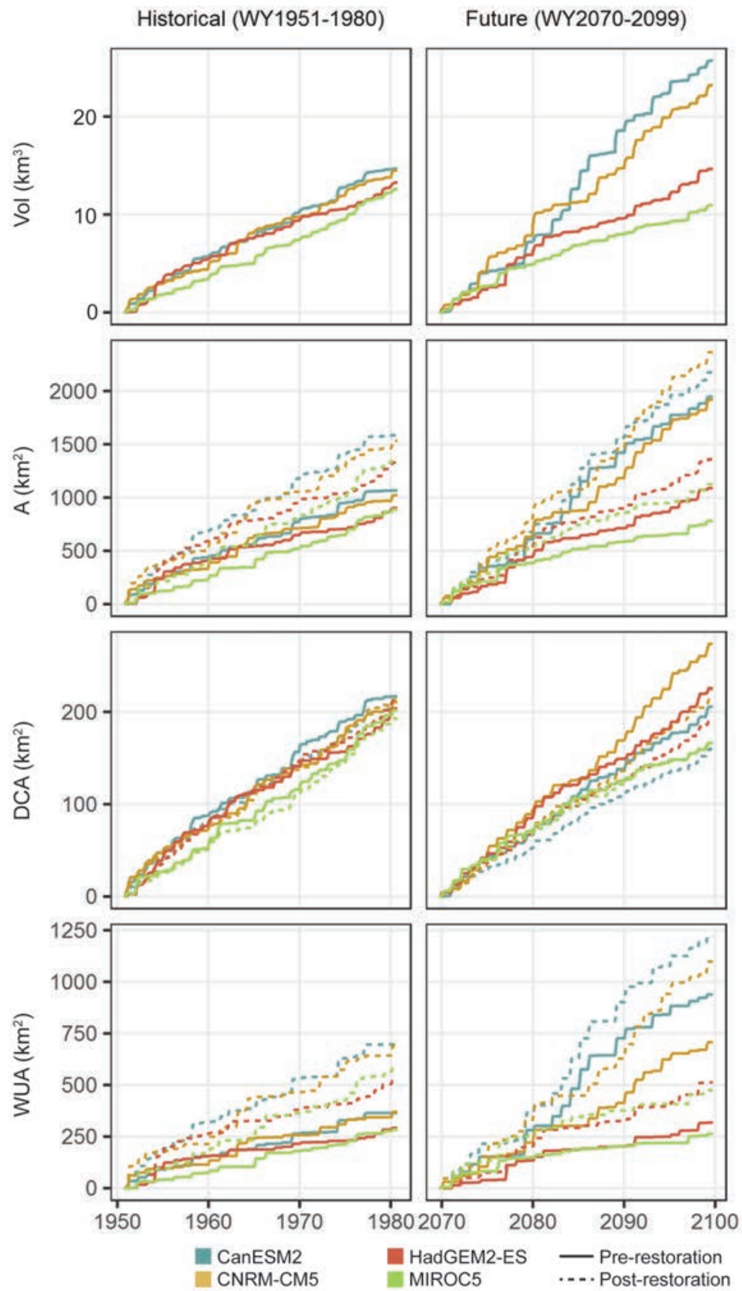

Figure 6-13. Cumulative distributions for temporally-resolved metrics for the WY2070-2099 and WY1951-1980 periods. Each row shows flow volume (*Vol*, for comparison purposes), inundated area (*A*), disconnected inundated area (*DCA*), and weighted usable area (*WUA*), respectively. Separate distributions are shown for each combination of climate change scenario and restoration configuration.

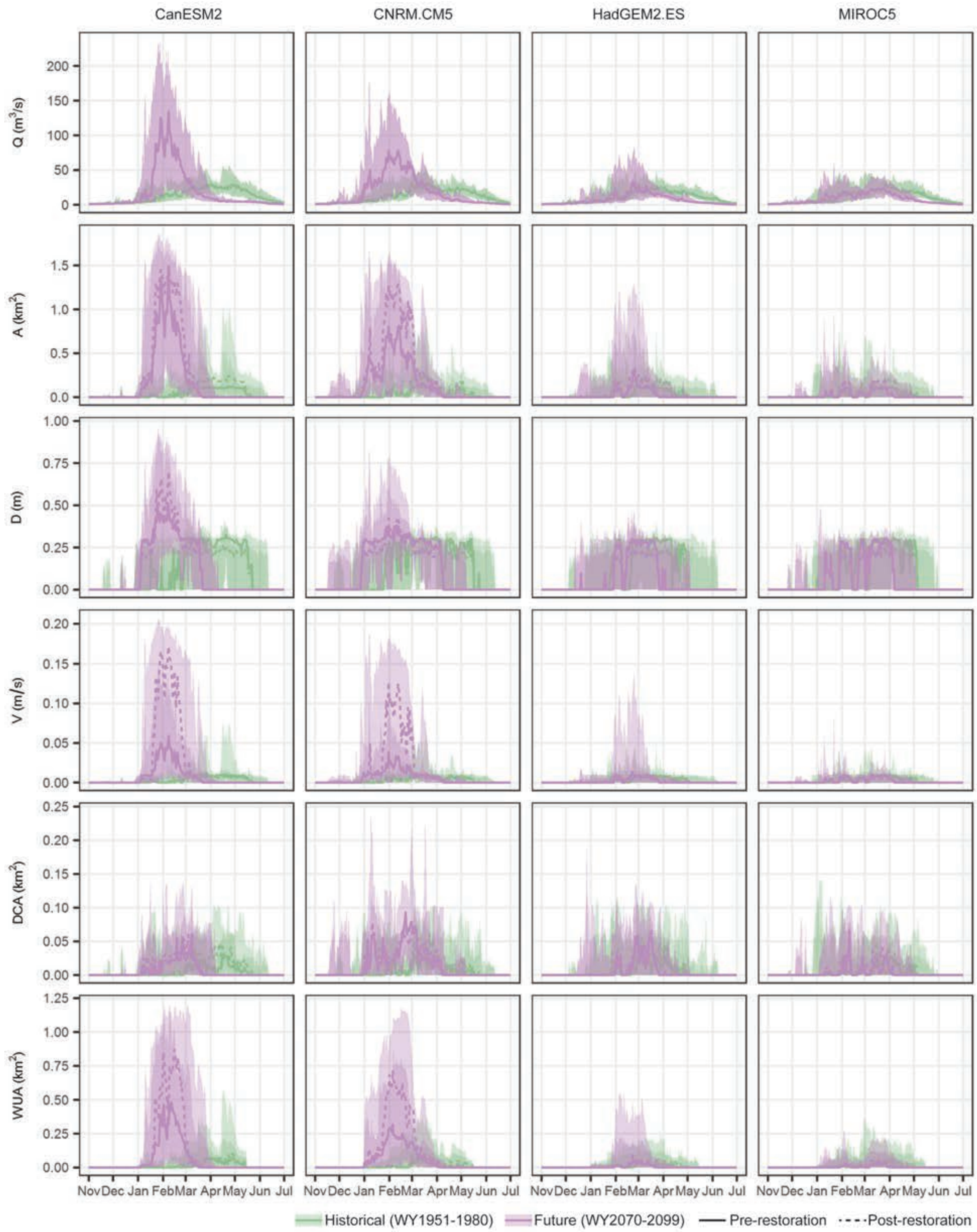

Figure 6-14. Daily median distribution comparison for selected hydrospatial metrics and flow volume projections comparing WY1951-1980 and WY2070-2099 period conditions for different climate change scenarios. Distributions are also distinguished by restoration configuration. Shading represents the 25th-75th percentile.

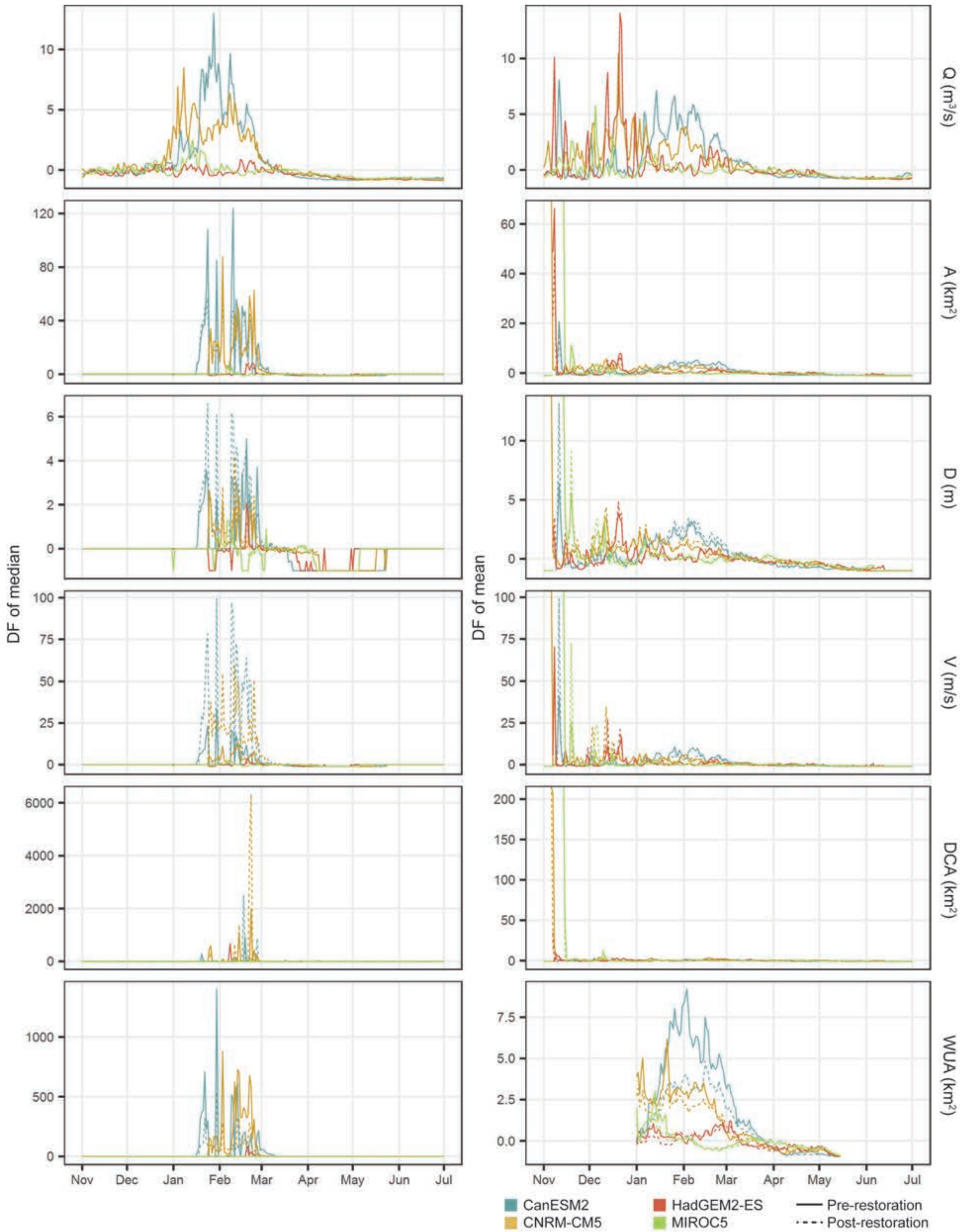

Figure 6-15. Daily deviation factors between the WY1951-1980 and WY2070-2099 periods for daily median and mean values for selected temporally-resolved hydrospatial metrics and flow volume projections. Distributions are also distinguished by restoration configuration.

similar between the two time periods than they were between pre- and post-restoration configurations (Figure 6-16). In general, deeper inundation was more evenly spread across the floodplain under the post-restoration configuration. However, deeper areas expanded overall for CanESM2, CNRM-CM5 and HadGEM2-ES. For *MV* (mean water year mean velocity), changes were similar to *MaxD*, where areas of higher velocity expand in the WY2070-2099 period and areas of high and low velocity are visually more patchy for the pre-restoration configuration (Figure 6-17).

Duration

For inundation duration (*Durm*), the WY2070-2099 period was associated with longer median flood event duration by 2-3 days for CanESM2 and CNRM-CM5 and shorter duration by about one day for HadGEM2-ES and MIROC5 (see Table 6-3). The direction of change is the same for duration as it is for water year flow volume, but the DFs are different in magnitude. Comparing pre- and post-restoration configurations, duration increased relatively more for CanESM2 and CNRM-CM5 post-restoration between the two time periods, but the post-restoration difference was significant only for CNRM-CM5. Water years associated with extremely long average event durations increased in both number and magnitude for CanESM2, CNRM-CM5 and HadGEM2-ES (see Figure 6-10 and Figure 6-11). Duration variability (as indicated by CD) increased for four of the eight flow scenarios and restoration configuration combinations, with variability increasing over two-fold for CanESM2 (see Figure 6-12). Spatial distribution of mean duration shows the substantially lengthened duration in terms of magnitude and spatial extent for CanESM2 and CNRM-CM5 (Figure 6-18). The maps also show that for HadGEM2-ES and MIROC5, post-restoration duration for the WY1951-1980 period was longer in most parts of the floodplain than pre-restoration duration for the WY2070-2099 period.

Timing

All combinations of climate change scenarios and restoration configurations produced an earlier timing of centroid volume (*WYDCnVol*) in the WY2070-2099 period, and significant differences in the two time periods were found for CanESM2 and CNRM-CM5 (see Table 6-3). In the most extreme cases, median *WYDCnVol* decreased by 26 days (March 6 to February 8; CanESM2, post-restoration) and mean *WYDCnVol* decreased by 19 days (February 14 to January 25; CNRM-CM5, post-restoration). While in the WY1951-1980 period, the post restoration scenario advanced median timing by 1.5-8.5 days, that advance shrunk in the WY2070-2099 period, to 1.0-5.5 days. Variability was also found to decrease across all scenarios, mostly attributable to the decline in water years with late centroid timing (see Figure 6-10, Figure 6-11, and Figure 6-12). This shift toward earlier timing is also exhibited in the daily distributions of other metrics shown in Figure 6-14.

Frequency

Inundation frequency (*IFNm*) declined between the two time periods across all flow scenarios and restoration configurations, from about 6-8 to 4.5-6.5 times a season (see Table 6-3). Significant differences in frequency between the two time periods were found for CanESM2 under both pre- and post-restoration configurations and for CNRM-CM5 post-restoration. Future conditions were less variable, except for HadGEM2-ES, primarily following declines at the upper extremes by >5 times (see Figure 6-10, Figure 6-11, and Figure 6-12). Spatially, frequency also appears to decline between the two time periods (Figure 6-19). Small areas of the floodplain with high frequency in the WY1951-1980 period show declined frequency in the WY2070-2099 period. However, the upstream floodplain area

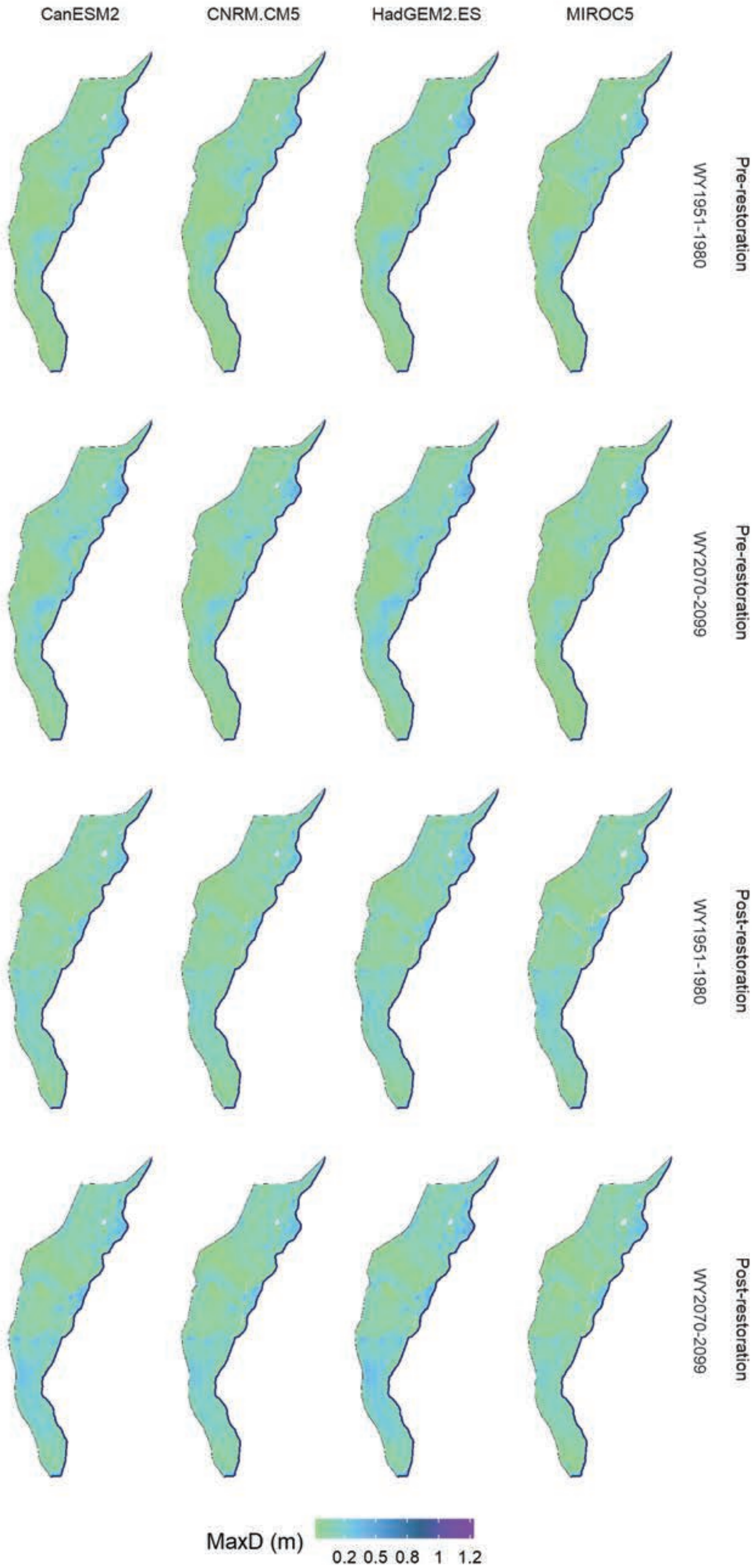

Figure 6-16. Spatial distribution of mean of maximum depth (*MaxD*) for water year maximum.

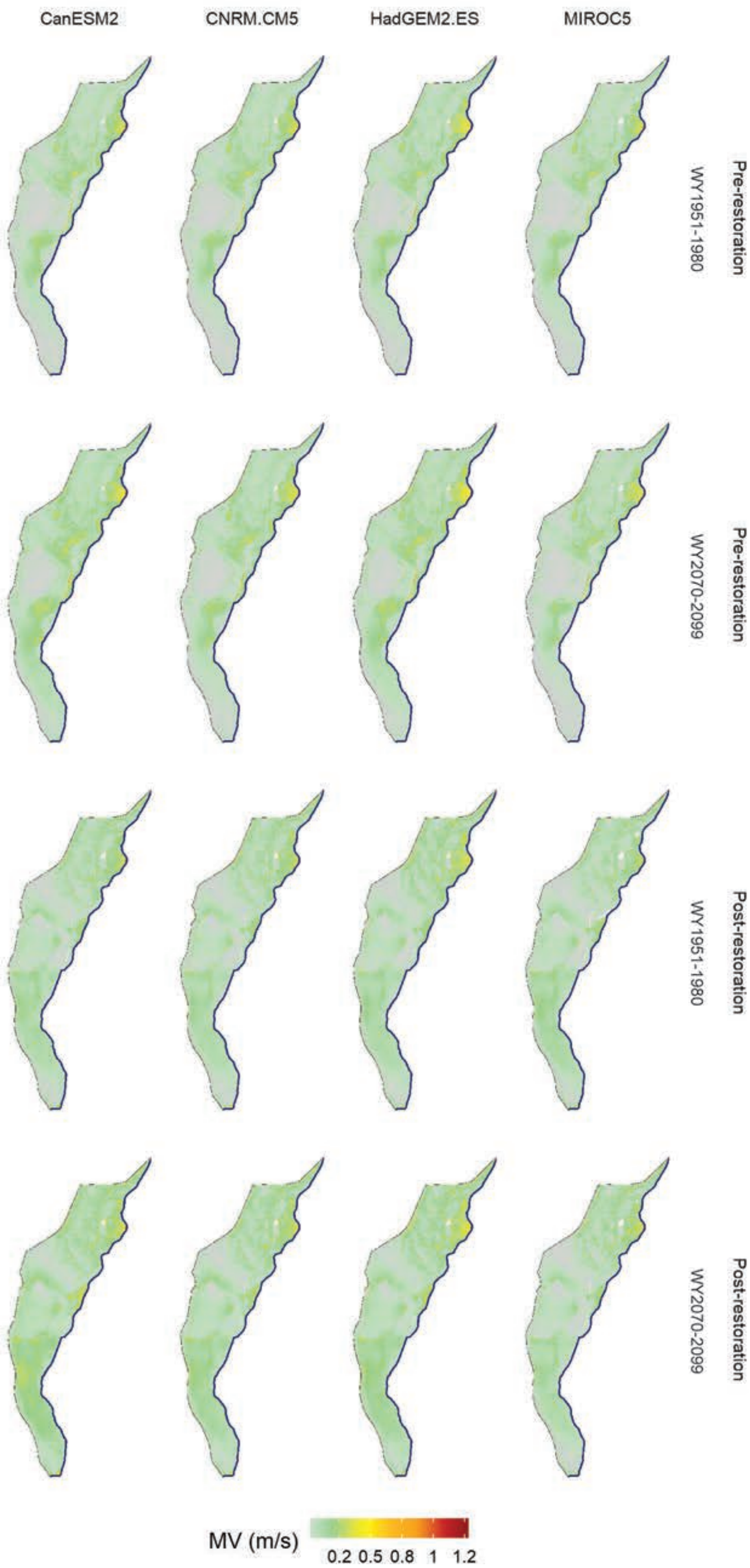

Figure 6-17. Spatial distribution of velocity mean of water year mean of flood event mean (*MV*).

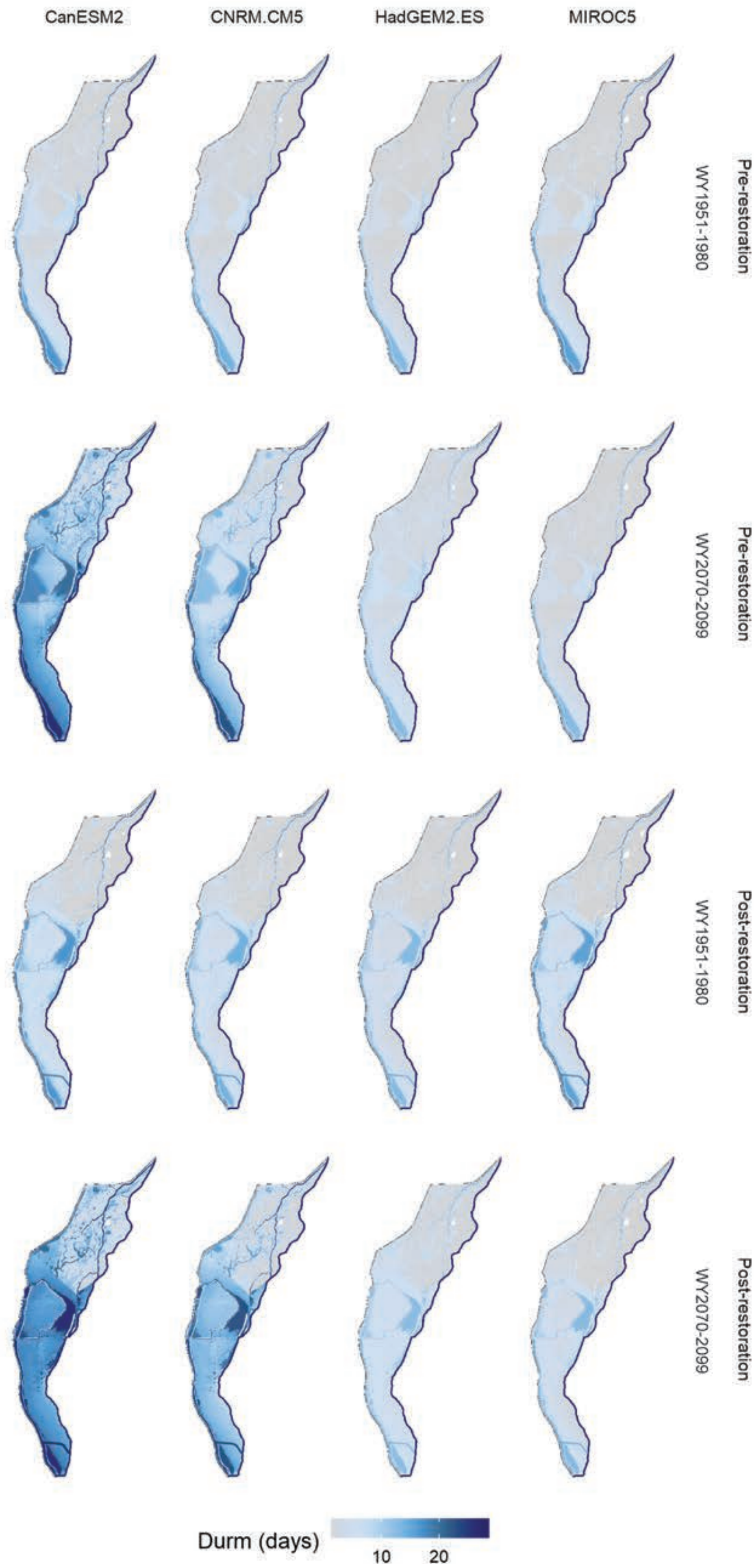

Figure 6-18. Spatial distribution of flood event duration mean of water year mean of (*Dur*).

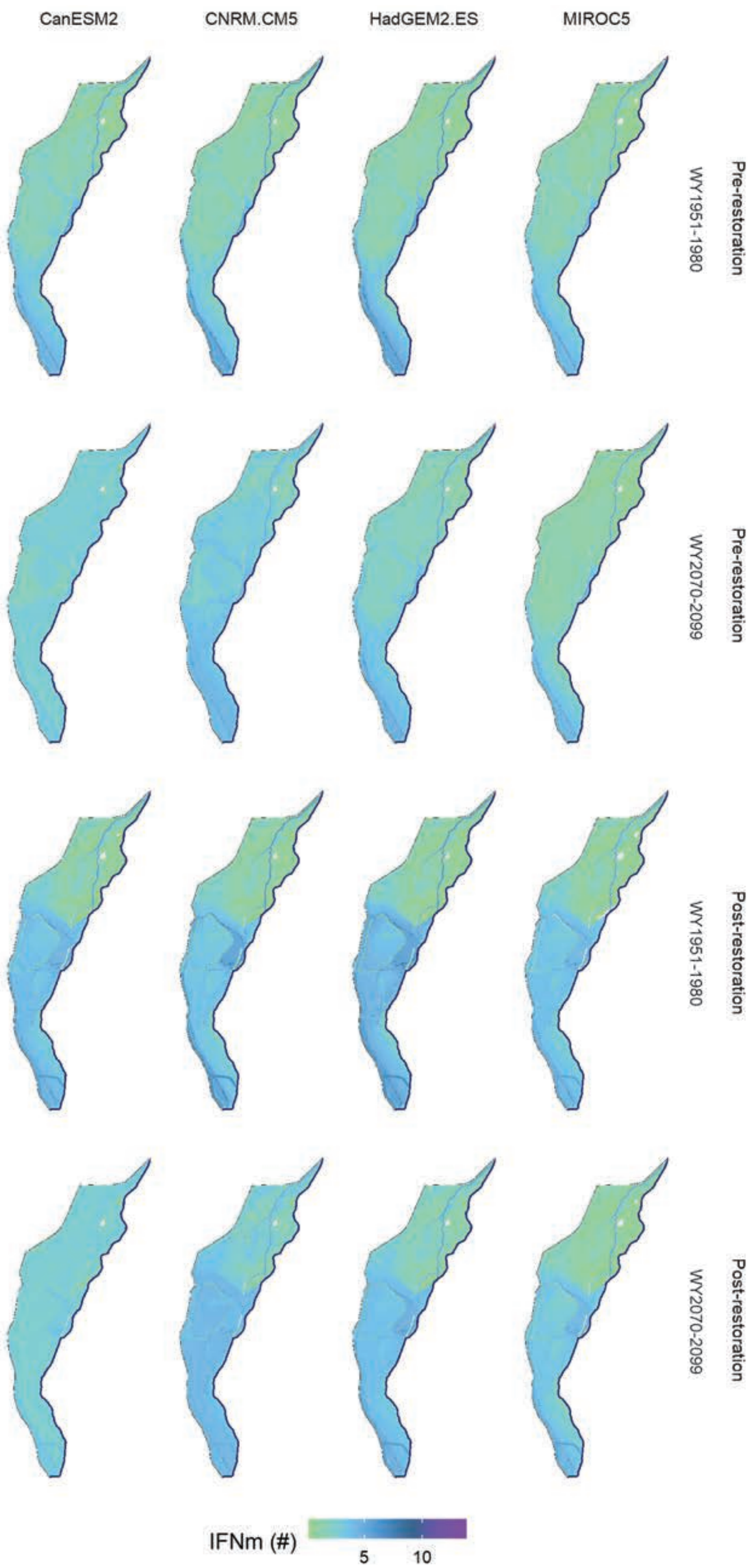

Figure 6-19. Spatial distribution of inundation frequency, mean of water year (*IFN*).

generally increased in frequency. Also, the difference between CanESM2 pre- and post-restoration was less substantial for the majority of the floodplain in the WY2070-2099 period than for other scenarios.

Connectivity

Hydrologic connectivity, as measured by area disconnected from the river channel (*DCADay*), did not show consistent trends between the two time periods (see Table 6-3). Significant differences between the two periods were found for increased connectivity with CNRM-CM5 pre-restoration and decreased connectivity for CanESM2 post-restoration. Connectivity variability across water years decreased consistently, however, which appears primarily a result of decreased upper extremes in the WY2070-2099 period (see Figure 6-10 and Figure 6-12). Exceedance probability plots in Figure 6-11 show that the relative influence of restoration configuration and time period differed depending on the climate change scenario and specific exceedance probabilities. For temporal distributions, pre-restoration CNRM-CM5 had distinctly higher cumulative disconnected area in the WY2070-2099 over the WY1951-1980 period in comparison to the other climate change scenarios, which were generally similar between the two periods (see Figure 6-13). CNRM-CM5 also had pronounced peaks in the 75th percentile of daily area disconnected (see Figure 6-14). Otherwise, daily distributions were fairly similar across climate change scenarios, with earlier timing in the WY2070-2099 compared to WY1951-1980 period. Median values were, however, higher for CanESM2 and CNRM-CM5. Spatially, the areas of higher average disconnectivity were less pronounced in the WY2070-2099 compared to the WY1951-1980 time period for most climate change scenarios (Figure 6-20).

Habitat availability

Habitat availability (*WUADay*) for juvenile rearing Sacramento splittail – determined using depth, velocity, duration, timing, and connectivity requirements – increased substantially under CanESM2 and CNRM-CM5 between the WY1951-1980 and WY2070-2099 periods (see Table 6-3 and Figure 6-12). Compared to these increases, reductions in *WUADay* for HadGEM2-ES and MIROC5 were less in magnitude. Significant differences between the two time periods were found only for increased habitat availability with CNRM-CM5 pre-restoration. Across all climate change scenarios, post-restoration conditions reduced the direction of change over pre-restoration conditions (positive DFs of median conditions were lower and negative DFs were higher). Trends in variability were opposite in direction to changes in median conditions (i.e., declines for CanESM2 and CNRM-CM5). With regard to extremes, water years in the CanESM2 WY2070-2099 period had substantially higher overall habitat availability relative to the WY1951-1980 period and other climate change scenarios (see Figure 6-10 and Figure 6-11). The exceedance probability plots also show that for probabilities below 75%, the WY2070-2099 period tended to be higher than the WY1951-1980 period, regardless of restoration configuration. However, for HadGEM2-ES and MIROC5, habitat associated with the post-restoration configuration was consistently higher than the pre-restoration configuration, regardless of the time period. Cumulative habitat over each of the 30-year periods shows this difference as well and demonstrates how habitat availability generally followed the spread and rate of change over time for flow volume (see Figure 6-13).

Habitat variability in space and time differed substantially between the wetter and drier climate change scenarios. The daily distribution of habitat availability (*WUA*) generally tracked that of inundated area, though the differences between pre- and post-restoration configurations were greater for *WUA* (see Figure 6-14). Under post-restoration, median daily *WUA* reached around 0.75 km^2 in late January and early February. Daily median conditions for HadGEM2-ES and MIROC5 were much less during this

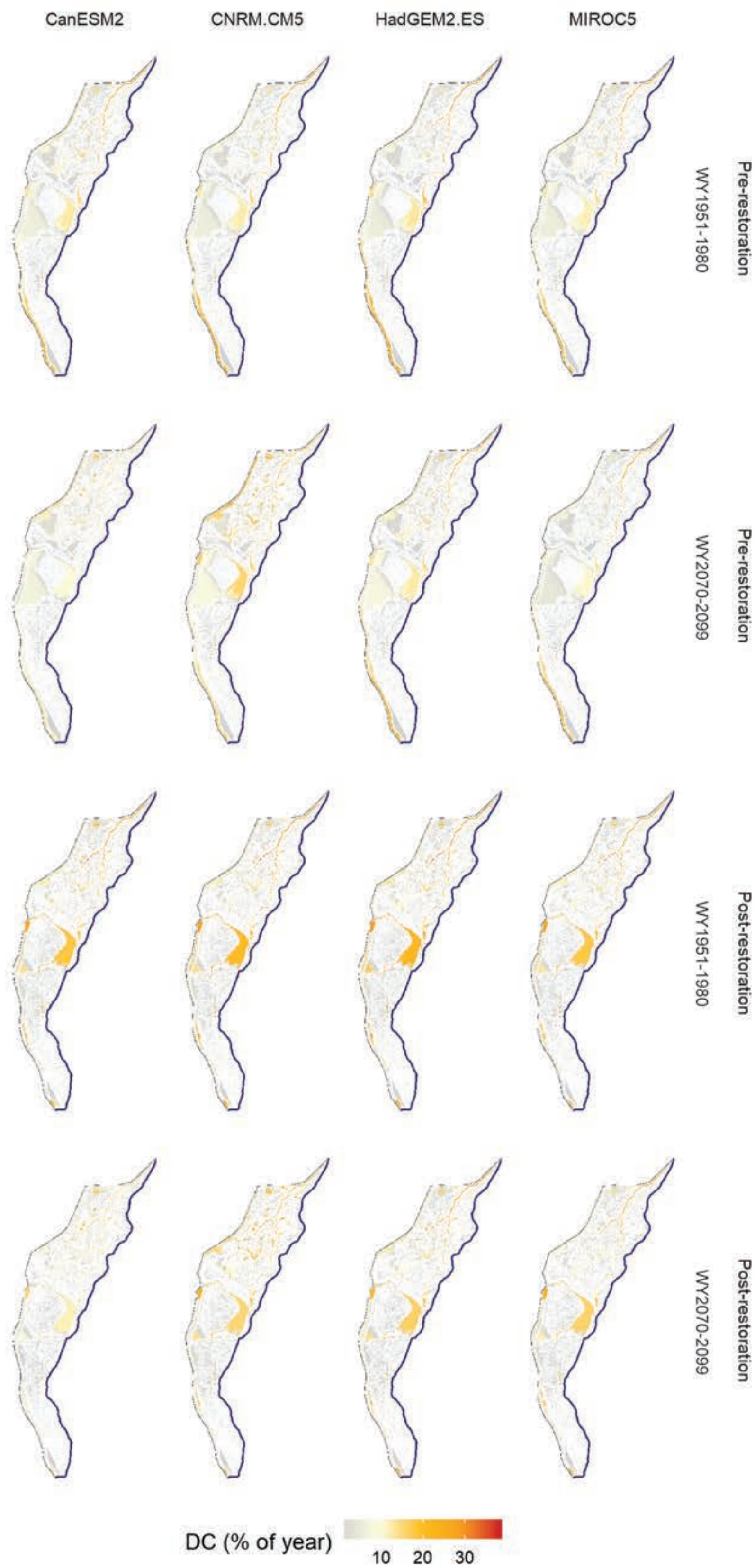

Figure 6-20. Spatial distribution of mean of percent of water year disconnected (*DC*) from river channel.

period, though the HadGEM2-ES 75th percentile in February to early March approached 0.5 km^2. All distributions shifted toward earlier timing. Daily DFs show high deviation in late January and February (mostly associated with CanESM2 and CNRM-CM5), which is similar to other metrics. Because inundated area prior to January 1 was not counted as habitat, DFs for mean began in January and were generally positive before becoming negative in late March (see Figure 6-15). Spatial distributions of annual habitat availability show the extensive spatial expansion of higher average *WUADay* for CanESM2 and CNRM-CM5 between the WY1951-1980 and WY2070-2099 periods (and expansion for the post- relative to the pre-restoration configuration; Figure 6-21). Reduced *WUADay* is shown for HadGEM2-ES and MIROC5, most noticeable in areas with high values.

DISCUSSION

Floodplain hydrospatial response to climate change

The multi-metric hydrospatial analysis presented here shows that responses of floodplain inundation patterns and habitat to climate change reflect projected losses of spring flooding and increased extreme winter floods across the four climate change scenarios. The magnitude and direction of responses differed, however, across floodplain metrics. For example, overall changes to mean velocity were larger than for maximum inundated area and depth. Maximum inundated area for HadGEM2-ES increased while duration and habitat availability for juvenile Sacramento splittail decreased. Results also show considerable variation across climate change scenarios, such that the magnitude of differences between climate change scenarios was greater for some metrics than the change between the WY1951-1980 and WY2070-2099 periods. The variation can be attributed to uncertainty in changes to magnitude and frequency of extreme flood events. Changes in metric values were generally similar – and most were strongly positive – for CanESM2 and CNRM-CM5 (the two wet flow scenarios), reflecting high winter extreme flood flows in the WY2070-2099 period relative to the WY1951-1980 period that overwhelmed springtime flow losses and nearly doubled median annual flow volume. Climate change responses under HadGEM2-ES were more mixed, likely because increased extreme winter floods did not consistently offset springtime losses. In the case of habitat availability, the lost habitat in the spring was not made up for by increased habitat in the winter. The dominating signature for MIROC5 was reduced springtime flows, such that water year median conditions declined across all metrics. Interannual variability declined across most metrics between the WY1951-1980 and WY2070-2099 periods, though this decline depended on the climate change scenario and restoration configuration. For some, like maximum inundated area, variability decline followed increasing frequency of extreme floods, which increased the number of water years where the entirety of the floodplain was flooded (2.1 km^2). Exceptions included accumulated inundated area and habitat availability, which accompanied increased interannual variability for HadGEM2-ES and MIROC5.

The examination of spatial and temporal variability in addition to average conditions informs understanding of factors contributing to overall change. The daily resolution illustrates the consistent loss of springtime flooding, causing the median and 75th percentile to reach zero a month or two earlier in the WY2070-2099 period. Further, only a short window in time in late January and February is primarily responsible for large increases in metrics including inundated area, depth, velocity, and habitat availability. Depth, in particular, appears to have a threshold effect, where median daily flow above 50 m^3/s relates to a sharp bump in depth above the otherwise fairly consistent 0.25 m. The interquartile range for daily metric distributions expanded substantially in the WY2070-2099 period in the early winter months for CanESM2 and CNRM-CM5. Regarding spatial distribution, some metric changes appeared to originate

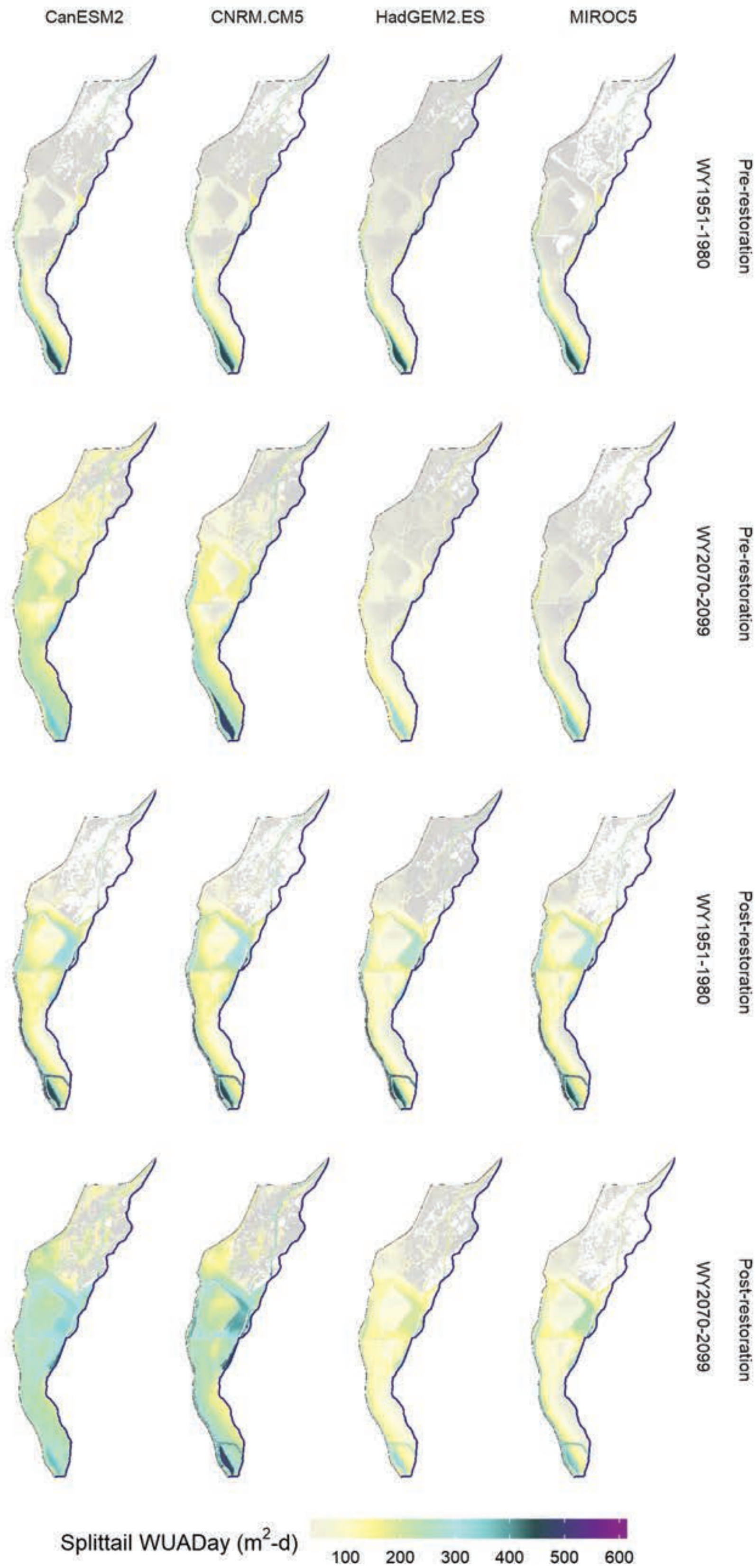

Figure 6-21. Spatial distribution of annual habitat availability for juvenile Sacramento splittail (*WUADay*, m^2-day).

from accentuated conditions in particular parts of the floodplain (e.g., disconnected area), while, for other metrics (e.g., velocity), the spatial expansion of higher (or lower) values can have the overall impact of driving up spatial mean.

Furthermore, this study shows that flow and floodplain inundation changes may differ in magnitude and direction under climate change, and how those flows change may affect the sensitivity of floodplain conditions. This concept is exemplified by changes in climate change flow scenarios that did not map well to all metrics. And, it seemed to matter what the flow changes were. For example, a less than doubling of water year flow volume produced a more than doubling of cumulative inundated area, but a slight decline in flow volume did not show similar sensitivity in a negative direction. A slight decline in median flow volume in the case of HadGEM2-ES was associated with a much greater decline in habitat availability. Essentially, understanding changes to the magnitude, timing, and duration of flows as well as how floodplains variably respond to changes in those different flow characteristics is necessary to assess potential floodplain change. Overall, results illustrate that the interaction of flows and the floodplain mediates the changes in spatiotemporal inundation resulting from hydroclimatic change. Therefore, inferring climate change impacts to floodplains from flow regime change alone may not provide an adequate understanding of potential change.

Variable influence of floodplain restoration

Between WY1951-1980 and WY2070-2099, the relative effects of levee-removal floodplain restoration to promote hydrologic connectivity were not consistent across metrics or climate change scenarios. However, shifts toward greater peak winter flood flows and loss of long-duration intermediate flood flows in the spring months reduced the impact of restoration overall. For example, the springtime habitat benefits under post-restoration (Chapter 5) are overwhelmed in the WY2070-2099 period by loss of spring flood flows. Primary causes for the reduced effects of restoration are that the post-restoration configuration is most influential for intermediate flood flows, which generally decline for WY2070-2099, and that post-restoration configuration affects conditions relatively little at high flood flows because most of the floodplain is inundated regardless of restoration configuration. That is, for the floodplain studied here, the levee-removal mostly expands inundation extent at lower flood flows, and such flows are projected to be less common in the late winter and early spring across flow scenarios, and are replaced by higher flood flows in the earlier winter of CanESM2 and CNRM-CM5. Consequently, CanESM2 and CNRM-CM5, with much higher peak winter flows, have greater relative impact of climate change effects over restoration. One exception is centroid timing of flow, which declined relatively more in the WY2070-2099 period with the post-restoration configuration. The HadGEM2-ES is a particularly interesting flow scenario, however, because peak winter flows increased while median flow volume declined between the WY1951-1980 and WY2070-2099 periods. This discrepancy is apparent in the results: exceedance probabilities for metrics such as maximum inundated area and depth increased in the WY2070-2099 period regardless of restoration configuration while exceedance probabilities for cumulative inundated area or habitat availability were higher for the post-restoration configuration for both time periods. In the case of MIROC5 (the drier flow scenario), the post-restoration configuration had an overall counteracting and overwhelming effect on climate change impacts. Taken together, results from this study indicate that floodplain levee removal generally dampened climate change impacts, whether the future was wetter or drier.

Assessing riverine response to climate change

The research presented here contributes to a growing literature addressing climate change impacts to riverine environments and their ecosystems. It involves a novel approach, introduced in Chapter 4, to analyze spatiotemporal inundation conditions, which allows floodplain characteristics to be explicitly evaluated and extreme conditions more comprehensively assessed. This study offers a flexible approach for evaluating the response of floodplain inundation patterns and to consider implications for habitat availability using multiple climate change scenarios.

Results align with other research that has assessed climate change impacts on hydrodynamic-dependent conditions. Spatially- and temporally-variable response and susceptibility to change are common threads. For example, Karim et al. (2016) concluded that changes to wetland connectivity under wet and dry climate change scenarios varied spatially across the floodplain, which was determined via 2D hydrodynamic modeling. Under projected future Pacific Northwest climates, Hatten et al. (2014) found that salmon habitat change varied spatially and temporally and also that floodplain habitat may be relatively more important with future as opposed to current hydrology. Similar to the conclusion here that hydrospatial response was not directly correlated with flow changes, Guse et al. (2015) found that, under climate change, hydraulic change was less sensitive and not clearly related to hydrologic change.

Other studies have taken multi-factor approaches similar to that taken here to evaluate the relative impact of climate change within the context of management alternatives. Matella and Merenlender (2015) found that additional environmental flows were needed in combination with floodplain restoration in order to meet salmon and Sacramento splittail habitat objectives under climate change. Boughton and Pike (2013) used flow and bioenergetics modeling in combination with climate change projections on the Pajaro River, California, to examine salmon migration impacts and found that floodplain rehabilitation buffered against the potential negative effects of reduced magnitude or frequency of floods. Another salmon habitat study combined restoration scenarios with climate change projections and determined that the restoration actions had the capacity to offset temperature increases due to climate change (Justice et al., 2017). Results from this research support the emphasis placed on restoring floodplains to build ecosystem resilience to change and offset negative impacts of climate change (e.g., Beechie et al., 2013; Wade et al., 2013; Williams et al., 2015).

Climate change impact assessment challenges

Top-down climate change impact assessments, which typically involve linked models similar to this study (e.g., from GCM to hydrologic model to hydrodynamic model), inherently include sources of uncertainty. Climate change studies now typically employ an ensemble or multi-model approach to partly address uncertainty associated with individual GCMs. Through an ensemble approach, the degree of consistency across models can provide more actionable information for management and planning (Brekke et al., 2008). Carrying each GCM scenario through a full chain of models can be computationally burdensome, however. Hence, the State of California supported a process to evaluate and select a limited set of scenarios for state-level planning purposes, which are the four GCMs used in this study (Kravitz, 2017). Uncertainty across these models taken to the point of floodplain hydrospatial metrics is reflected in differences in historical distributions across scenarios, but further analysis of this uncertainty would be informative (see Figure 6-10).

However, while multi-model approaches for GCM scenarios are becoming standard practice, potential limitations or uncertainty associated with the often single hydrologic and hydrodynamic models applied subsequently in the model chain are less scrutinized. Also, developing streamflow products at spatial and temporal scales necessary for hydrodynamic or reservoir operations modeling often requires bias correction to adjust for errors associated with hydrologic modeling. Recent research has cautioned that this process may distort the climate change signal in some cases (Nijssen & Chegwidden, 2017). More broadly, research has also begun evaluating effects of methodological choices in assessing hydrologic impacts of climate change, resulting in suggestions that multiple downscaling methods, ensembles of regional models as well as hydrological models, and multiple parameter estimation methods be used to assess uncertainty (Buytaert et al., 2010; Chegwidden et al., 2017; Chiew et al., 2010; Mendoza et al., 2016). Uncertainty related to climate change scenarios is compounded by uncertainty associated with hydrodynamic modeling and further evaluation of hydrodynamic modeling output, such as habitat quantification.

A notable limitation, which is involved in each modeling step, is the assumption that model parameters and the entire modeling domain do not change over time (Xu et al., 2005). For example, the quantile mapping procedure used to bias correct the daily flow projections presumes that the bias structure of the historical baseline period persists in the future period. Even more open-ended and potentially more impactful are the landscape changes (e.g., watershed-level land cover changes, geomorphic changes), shifts in antecedent conditions, and other ramifications of climate change (e.g., human responses) not accounted for in the hydrodynamic modeling and the chain of models overall. Thus, it is essential to recognize that this assessment does not provide predictions of future conditions, but rather evaluates relative impacts of plausible future flow regime change for particular floodplain topographic configurations using a set of metrics to represent the floodplain hydrospatial regime.

Water management and restoration planning implications

The predominate shifts in climate change affecting floodplain hydrospatial conditions are reduced magnitude, frequency, and duration of spring flood flows and increased extreme winter flood flows. In a drier future dominated by loss of springtime flooding, floodplain restoration could help counter this loss, though shifts in inundation timing would still occur. With a wetter future, extreme winter flooding increases relatively more than spring flooding declines, and floodplain restoration may counter more extremes. It may be appropriate to develop floodplain restoration strategies that better accommodate and take advantage of higher peak flood flows. Results also imply that flow management options on regulated rivers could be considered alongside restoration options to promote more intermediate flood flows with duration and timing to mitigate floodplain climate change impacts. Furthermore, results showing reduced floodplain restoration impact under future flow conditions, spatiotemporally variable floodplain responses, and non-linear relationships between flow and floodplain hydrospatial conditions raise the importance of coordinating mutually-beneficial physical restoration and environmental flow strategies (see Chapter 2).

The research here illustrates how climate change may counter some effects of prior human modifications of floodplain landscapes. For example, increased extreme flood flows enhanced overall floodplain inundation extent and frequency, typically in a diminished state from past levee-building and channelization. However, depending on the flow scenario, time of year, or position within the floodplain, this effect is not equivalent to the effect of levee-removal to promote floodplain connectivity with historical flows. The findings imply that rather than assuming future extreme flows lessen the need for restoration, working to understand potential trajectories of change under various flow and landscape management scenarios should be a part of planning.

Though floodplain inundation and related habitat is likely to be substantially affected by climate change, results also show that restoration can help mitigate those impacts. This observation is in line with other work that suggests restoration becomes more relevant within the context of climate change (Beechie et al., 2013; Seavy et al., 2009). For many systems, freshwater ecosystem degradation from past and current human actions has been more severe over the last century than what is expected under climate change alone (e.g., Balcombe et al., 2011; Dyer et al., 2013; Rheinheimer & Viers, 2014). Thus, there is great potential to reduce climate change impacts through addressing other human stressors. The cross-comparison between climate change and restoration scenarios presented here helps inform such discussions.

The approach taken here can be adapted as a water management and restoration planning decision-support framework for evaluating the relative impacts of management responses – including habitat restoration, environmental flows, or both – in space and time and for evaluating consistency across climate change scenarios. Findings support the concept that flow regime analysis alone is inadequate for assessing changes to floodplain inundation patterns and therefore for developing appropriate restoration strategies. This approach provides spatially and temporally explicit evaluation of hydraulic metrics (e.g., flooding extent, depth, duration, and timing), as opposed to hydrologic metrics, factors important to planners and managers (Bouska et al., 2016). It can also be used to assess which strategies perform best across a range of possible future scenarios (Boughton & Pike, 2013; Schindler & Hilborn, 2015). Overall, this research points to opportunities available to planners and managers to adjust flows and landscapes to lessen potential ecological impacts of climate change. It also suggests that considering physical restoration alternatives alongside flow management alternatives under climate change can broaden the capacity for enhancing resilience to riverine and floodplain impacts of climate change.

CONCLUSIONS

Hydroclimatic change expressed in shifting flow regimes translates to changing conditions within floodplain environments. Floodplain hydrospatial regime responses are mediated by spatial configuration and topography of the river-floodplain landscape. Therefore, understanding potential climate change impacts to floodplains and the ecosystems they support requires quantitative tools that assess spatial and temporal change and variability of hydraulic metrics (such as inundation extent, duration, depth, velocity, timing, and connectivity) as opposed to hydrologic metrics. Physical requirements for various species of interest can be quantified from these hydraulic metrics. Successful management of floodplains needs new methods and scenario-based tools to determine the relative influence and robustness of habitat restoration and environmental flow strategies against climate change impacts. This study addresses these needs by quantifying the change in floodplain hydrospatial conditions across climate change scenarios. Results from the analysis here show that floodplain responses to hydroclimatic change – both in magnitude and direction – depended on the metric examined and varied in time and space. The counteracting forces of reduced spring flooding and increased winter extreme flooding drove overall floodplain response to climate change projections. While the spread across climate change scenarios was large, results show that the centroid timing of flood volume consistently shifted to an earlier timing and variability was reduced for most metrics, regardless of restoration status of the floodplain. For some floodplain variables, effects of changes to spring flooding were counteracted and even overwhelmed by increased winter floods projected in some climate change scenarios. Also, results show that levee-setback restoration can dampen climate change impacts, under both wetter and drier futures. Overall, this research shows the utility of hydrospatial analysis to examine potential climate change implications for floodplain environments and to evaluate the relative influence of levee-removal restoration actions to mediate response. The research approach and findings here support more informed planning and management of flows and the landscapes with which they interact under a variable and changing hydroclimate.

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CHAPTER 7

CONCLUSIONS AND FUTURE RESEARCH

Conclusions and future research

MOTIVATION AND APPROACH

This dissertation was motivated by the grand challenge of reestablishing dynamic and complex riverine environments that sustain functional ecosystems within the constraints of competing demands for water, contemporary human-dominated landscapes, and a changing climate. Science and management traditionally approach water resources and land management separately, which has limited capacity for addressing the interdependent problems facing large river systems. Interdisciplinary thinking and new techniques are required to adequately develop and assess management alternatives under current and future conditions that best support ecosystems.

The body of work herein informs these needs by examining the interaction of flow regime and floodplains to better characterize spatiotemporal variability, determine response to change, and establish links to ecological functions. The research objectives were to establish a methodological framework to characterize the lateral dimension of a river's flood regime, and to demonstrate the approach through a multi-metric quantification of floodplain restoration response, restoration habitat benefits, and relative impacts of climate change scenarios. Established methods for hydrologic classification, hydrodynamic modeling, and spatial analysis are applied in new ways. Studies were conducted for a floodplain restoration site along the lower Cosumnes River, California, which is uniquely suited for investigating spatiotemporal floodplain inundation patterns as an unregulated river in an agricultural-dominated yet ecologically important landscape. Overall, this research enriches understanding of complex floodplain environments and provides insights and tools to enable more effective and efficient integrated land and water management.

RESEARCH SUMMARY

Conclusions that emerged from this dissertation support broad concepts as well as provide specific insights concerning the floodplain restoration site of focus. **Chapter 2**, a literature review, juxtaposed the river restoration sub-fields of habitat restoration and environmental flows with the purpose of synthesizing scientific consensus and encouraging integrated approaches for restoration of highly modified large rivers. The discussion followed the parallel transformation of these fields of literature from static to process-based goals and highlighted the recognition that both flows and the landscapes with which they interact must be managed together to achieve objectives in highly modified rivers (Beechie et al., 2010; Palmer et al., 2014; Wohl et al., 2015). This contribution coalesced current thinking, presented examples of integrated restoration approaches, and identified paths forward, including coupling watershed-scale modeling with reach-scale hydraulic modeling, advancing the use of metrics representing hydrogeomorphic processes, inclusion of food web dynamics in modeling approaches, and overcoming institutional barriers that limit coordination of water and land management.

The recently published flood typology presented in **Chapter 3** pursued an informative description of types of floods inundating floodplains, showing that a variable flood regime can be characterized using ecologically-relevant metrics (Whipple et al., 2017). While flow regime classification is common (Olden et al., 2012), few studies explicitly focus on describing the flood component of the flow regime, and thus, the primary methodological contribution was the application of hydrologic classification techniques to describe a river's flood regime. Flood events were systematically classified using unsupervised cluster analysis with metrics relating to flood magnitude, timing, duration, and rate of change. Six flood types were successfully identified as best describing the variability in the flood event metrics for the daily flow record of the Cosumnes River (1908-2014 water years). Although specific flood typologies are expected to be different for other systems, this flood regime characterization allowed examination of changing flood type occurrence over time, improved interpretation of driving physical processes, described prevailing flood conditions relevant to floodplain ecosystems, and can be used to develop more informed management strategies.

The hydrospatial analysis framework of **Chapter 4** described a process for evaluating spatiotemporal floodplain inundation patterns and is the core methodological contribution of this dissertation. The approach is distinguished from common floodplain inundation quantification techniques in that spatial and temporal resolution is retained in the analysis. The basic framework consisted of two-dimensional (2D) hydrodynamic modeling to establish spatially-resolved flow-depth and flow-velocity relationships combined with a daily flow time series to produce gridded estimates of depth and velocity, which were then summarized in space and time for a range of physical metrics. Inundation changes due to levee-removal restoration within the lower Cosumnes River floodplain were evaluated in the study. Reflecting variable hydrology interacting with floodplain topography, a primary outcome was that responses to the restoration actions were not consistent in space, in time, across metrics, or in relationship to flow. While levee removal generally increased inundation extent, other variables such as depth did not respond substantially. Changes were most substantial at intermediate flood flows. This work demonstrated the utility of the hydrospatial approach for effectively characterizing floodplain inundation at spatiotemporal scales relevant for ecological interpretation and management.

In **Chapter 5**, the hydrospatial analysis approach of **Chapter 4** was taken a step farther to quantify how the levee-removal restoration of the lower Cosumnes River altered floodplain habitat availability for Sacramento splittail (*Pogonichthys macrolepidotus*), a native fish species. Physical habitat modeling derived from 2D hydrodynamic modeling is often used to link changes in the environment or management alternatives to their ecological implications. Whereas most studies develop flow-habitat area relationships, spatial and temporal resolution was retained in the analysis, allowing for the fact that flow and habitat relationships are not necessarily one-to-one. Habitat suitability criteria were applied that related to depth and velocity, as well as duration, connectivity, and timing. Furthermore, this effort contributes to growing attention toward floodplain as opposed to in-channel habitat quantification (e.g., Erwin et al., 2017; Matella & Jagt, 2014; van de Wolfshaar et al., 2010). Research outcomes specific to the study site included the finding that overall floodplain habitat availability for the Sacramento splittail nearly doubled with restoration, though benefits varied substantially in space and time. For example, not all areas of the floodplain generated gains in habitat, revealing spatial tradeoffs. Results also corroborated that the relationship between available habitat area and flow could not be expressed as a simple function.

Chapter 6 explored potential floodplain responses to climate change, supporting that the floodplain landscape mediates the impact of flow regime change, with the Cosumnes River levee-removal restoration dampening responses overall. This application also demonstrated that the hydrospatial analysis approach could be used to evaluate effects of flow regime change in addition to or in combination with physical landscape alteration (e.g., habitat restoration). Specifically, this study compared metrics relating to inundation extent, depth, velocity, duration, timing, connectivity, and splittail habitat for four climate change scenarios (global climate model induced daily flow regimes), historical (water years 1951-1980) and future (water years 2070-2099) time periods, and pre- and post-restoration topographic configurations. Magnitude and direction of change were highly variable across metrics, though they were tied to opposing trends of declining spring flooding and increasing extreme winter flooding in future climate scenarios. Though the earlier flow timing and reduced spring flooding was consistent across future climate scenarios, the overall spread across scenarios was substantial. For the extreme wet scenarios, climate change impacts overwhelmed those of the restoration action. This research supports that floodplain restoration actions can be used to mitigate effects of flow regime change under climate change and points to the utility of multi-metric spatiotemporal analysis such as applied in this chapter to help evaluate alternatives.

MANAGEMENT IMPLICATIONS AND FURTHER RESEARCH DIRECTIONS

This dissertation informs several broad themes relating to the restoration and management of rivers. As a part of these conversations, this work inspires further research toward promoting land-water interactions that support ecological resilience within the complex realities of human-dominated landscapes.

Managing for variability and complexity instead of individual stressors or species

Riverine landscapes are defined by variability and complexity. By fundamentally altering these conditions, human activities have induced extensive ecological changes with far-reaching and interacting ramifications. Restoration activities have been largely unsuccessful in reversing freshwater ecosystem decline, attributed in part to piece-meal approaches that address only individual stressors or species. Especially within the context of climate change and novel ecosystems, there are great risks associated with increasingly precise management that attempts to target specific functions, species, or static images of what habitat should look like (Hiers et al., 2016). Scientists encourage restoration focused on ecological resilience, productivity, and biodiversity (Harris et al., 2006; Poff, 2017). Process-based restoration is seen as a means to attain such ends, addressing root causes of change and encouraging the re-establishment of natural ecosystem dynamics. Management for processes and functions at the landscape scale are now articulated goals in planning documents and substantial investments are being made toward enhancing hydrologic connectivity through the implementation of environmental flows and/or physical floodplain or in-channel modifications. However, research is needed to determine what natural variability and complexity means in any given system as well as how specific management actions may ultimately affect those conditions.

The hydrospatial analysis approach and applications in this dissertation help address these broad challenges through improved quantification of variable floodplain inundation patterns in space and time. Broadly, showing that different parts of the floodplain have different characteristics at different times supports the concept that by managing for variable conditions, appropriate floodplain processes and habitat can be found at least in some location most of the time. Specific to the hydrospatial framework, extending methods to allow for assessments at larger spatial scales (e.g., satellite imagery analysis and targeted monitoring rather than 2D hydrodynamic modeling) would benefit regional- and watershed-scale planning efforts. Additionally, further refinement of a more limited set of priority physical metrics and summaries would reduce redundancy and make it more practical for application in a management setting. Ecological inference was limited in **Chapter 5** due to evaluation of a single species, and further research could extend the study to address a suite of species and/or ecological functions. Such research could help optimize restoration actions for multiple ecological functions. As hydrodynamic modeling power increases and computation of large spatial datasets becomes more feasible, more involved analysis relating to spatial pattern and landscape ecology principles, evaluation of a suite of ecological functions, and larger spatial scope or finer resolution are more achievable.

Evaluation of flow-ecology relationships

Tracing the many layers of interacting changes that have produced contemporary landscapes and their ecosystems and predicting future change are fundamental interdisciplinary challenges (Harden et al., 2014). As simplified representations of these connections, modeling can at times be illuminating in its capacity to test hypotheses and at other times opaque due to numerous technical as well as ecological uncertainties. The hydrospatial analysis approach responds to recent appeals for the use of hydraulic metrics over hydrologic metrics for evaluating river management alternatives (Bond et al., 2014; Brewer et al., 2016; Turner & Stewardson, 2014). With habitat quantification still largely dependent on simple descriptions of area as a function of flow, research remains to develop and apply metrics based on multi-dimensional hydraulic conditions. This is especially critical for complex floodplain environments, where landscape patterns mediate hydraulic conditions and complicate flow-habitat relationships. Additionally, the spatiotemporal resolution of physical metrics must match the scales relevant to an individual organism or the ecological process to be modeled.

Beyond using more explicit ecologically-relevant metrics, substantial advances in connecting flow and landscape changes to ecological response can be made through new research and methods that include landscape evolution, food web dynamics and biotic interactions (Naiman et al., 2012), as well as human systems' response to change. Continued advances in modeling overall and the capacity to dynamically couple hydrodynamic models with hydrologic models as well as ecological or socio-economic response models expand opportunities for more spatially-resolved analyses and more explicit links to the ecological implications of anthropogenic stressors and restoration actions.

Decision-support tools for scenario analysis that address multiple factors together

Further research is needed to successfully adjust flow regimes and landscapes together to generate complex and heterogeneous conditions to drive diverse and productive ecosystems. This is especially important in highly modified rivers, which have been substantially homogenized by flow regulation as well as landscape change. Increasing demands for integrated water and land management that addresses multiple and often competing objectives must be met with more sophisticated decision-support tools that provide detailed quantification of target metrics. Scenario analysis is widely used to weigh management alternatives. The hydrospatial analysis framework presented in this dissertation can provide a flexible method for comparing multiple scenarios and metrics quantitatively as well as visually. By retaining spatial and temporal resolution, visualizations in different dimensions are possible. The four-dimensional nature of floodplain inundation patterns can make it difficult to discern overriding patterns of change, which the research here attempts to address through single-value summaries interpreted through time series and geographic representations of the data. These options are useful in communicating information to scientists across disciplines, resource managers, policy-makers, and the public. Technological advances, such as interactive web-based visualizations, could facilitate the use of hydrospatial analysis results in planning processes, where information could be quickly gathered and presented to explore different questions.

Potential climate change-hydrology-ecosystem impacts at scales relevant for planning and management

In the area of freshwater ecosystem impacts of climate change, considerable research and resources are devoted to improving modeling and downscaling techniques, with growing attention toward the hydrologic modeling component (Chegwidden et al., 2017). Considerable uncertainties associated with projecting climate change impacts at the level of detail required for understanding floodplain ecosystem response and identifying appropriate management strategies remain (Xu et al., 2005). Some uncertainties will persist, however, and planning for an uncertain future requires improved techniques to evaluate relative impacts of various possible courses of action across climate change and management scenarios.

Specifically stemming from the studies here, there are several promising avenues of further research. With regard to the flood typology presented in **Chapter 3**, floods under future climate scenarios could be classified and trends in frequency of occurrence, loss of types, or potential emergence of new types explored. Such analysis would facilitate examination of changing flood characteristics relevant for floodplain ecosystems beyond typically-applied flow frequency analysis. Considerable expansion in the scenario comparison of **Chapter 6** is possible. One useful addition would be the examination of a near-term (e.g., 2030-2059 water years) climate period in addition to the historical (1951-1980 water years) and future periods (2070-2099 water years) applied here. As increasing flood extremes is understood to be a primary hydrologic consequence of climate change, potential insights could be gained from more extensive examination of how these extremes might be reflected in changing floodplain inundation patterns. More sophisticated bias correction techniques to address hydrologic model bias could also be applied that would potentially improve the flow projections (Pierce et al., 2015). Also, as computational speeds and data storage improve, the number of models, number of years, and spatial resolution of the analysis can increase. Further, performing the analysis across multiple watersheds could help draw conclusions about prevailing changes. Daily projections of flow based on the global climate models (GCMs) have only recently become readily available and further research remains to understand what those projections mean, how they vary between GCMs, and what the primary uncertainties are. More broadly, adding representations of landscape evolution as hydrologic regimes change as well as human-responses to climate change (e.g., water management alternatives or shifts in land use) within the hydrospatial analysis framework would expand the set of understood plausible futures.

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APPENDIX A

MODEL DEVELOPMENT AND CALIBRATION

Model development and calibration

INTRODUCTION

For purposes of studying flow-dependent ecosystem processes and functions, hydrodynamic modeling provides an efficient and effective description and analysis tool for physical conditions. Hydrodynamic modeling is increasingly applied within river and floodplain restoration contexts to better understand the dynamics of hydrologic connectivity between rivers and their floodplains, to evaluate restoration options, and to establish expected project benefits. Two-dimensional (2D) hydrodynamic modeling allows for the evaluation of more complex, multi-directional flows across laterally heterogeneous topography, which is particularly important for floodplain environments. The objective of the work described here was to establish calibrated pre- and post-restoration 2D hydrodynamic models representing a restoration site within the lower Cosumnes River floodplain, located within the Cosumnes River Preserve (CRP) managed by The Nature Conservancy and a consortium of agencies. The development of these models was part of a larger study to evaluate effects of the restoration project, which was conducted by the UC Davis Center for Watershed Sciences in partnership with The Nature Conservancy and funded by the California Department of Fish and Wildlife (Grant No. E1120001). Specifically, this modeling informs research quantifying floodplain inundation patterns in space and time, or the hydrospatial regime, for evaluating restoration impacts including habitat availability.

Restoration applications of hydrodynamic models commonly involve habitat availability assessments (Petts, 2009; Tharme, 2003). Of such approaches, the instream flow incremental methodology (IFIM) and physical habitat simulation (PHABSIM) are the most well-known (Bovee, 1982; Leclerc et al., 1995). Recently, one-dimensional (1D) modeling of California's Central Valley floodplains has been used to evaluate restoration options through relationships between flow and inundated area that serve particular ecological functions (Matella & Jagt, 2014; Merenlender & Matella, 2013). As computing power and availability of high-resolution data improves, the range of applications for hydrodynamic modeling within river restoration science increases. This research takes advantage of the capacity of 2D hydrodynamic modeling with high spatial and temporal resolution to evaluate floodplain inundation patterns.

MODEL DESCRIPTION

Modeling was performed using the U.S. Army Corps of Engineers' Hydrologic Engineering Center River Analysis System (HEC-RAS 5.0) software with 2D modeling capability (Brunner, 2016). The 2D version of this model was relatively recently released, but has already been widely adopted by consultants and state agencies for applications in California. Developed for floodplain modeling applications, models can be run using the Diffusion Wave equations or the full St. Venant equations. It employs an implicit finite volume solution algorithm, allowing for longer computational time steps, enhancing stability, and allowing for wetting and drying (common within floodplains). The computational grid can be structured or unstructured, providing greater flexibility in cell size and shape. An important characteristic of this model is its subgrid capability, where detailed computational hydraulic tables for each cell face are established using the underlying topography of the cell and cell faces, allowing for larger cell sizes while model output can remain at the resolution of the underlying DEM. The 2D options are flexible, where the model can be run fully in 2D or with 1D channel flow represented by cross sections and attached to 2D floodplain areas. The latter method was used in this application as the area of focus was the floodplain.

This modeling builds on previous efforts in the region. Most recently and specific to this research, the restoration site was subject to a 2D hydrodynamic modeling effort for environmental impact assessments (Robertson-Bryan Inc., 2011). However, this model was established using proprietary software (FLO-2D) and does not provide the level of resolution desired for this research. Earlier modeling for the region includes a 1D hydraulic analysis of the lower Cosumnes River using the proprietary MIKE11 software (Blake, 2001; Moughamian, 2005). This was extended downstream by Hammersmark et al. (2005) to evaluate restoration scenarios at the McCormack-Williamson Tract along the lower Mokelumne River. Another restoration site downstream from the restoration project of focus was subject to 2D hydrodynamic modeling to examine floodplain residence times (Andrews, 2007). While these previous efforts informed the establishment of the model with regard to model domain, boundary conditions, and general flow patterns, no elements of these prior models were applied directly to the HEC-RAS modeling here.

STUDY AREA

The lower Cosumnes River is a low-gradient river intersecting Pleistocene alluvial deposits and flowing to meet the tides just upstream of its confluence with the Mokelumne River in the Sacramento-San Joaquin Delta, California (Figure A-1). Its watershed is approximately 2,460 km^2, with headwater elevations at approximately 2,300 m. Unlike other Sierra Nevada rivers, the Cosumnes River is largely unregulated and frequently accesses its floodplain. Historically complex and dynamic floodplain topography reflecting a tidal to fluvial gradient was occupied by extensive distributary channels and abandoned channels, perennial freshwater marsh, willow swamps, and riparian forest lining channel banks (Whipple et al., 2012). This landscape has been altered by over a century of farming, leveeing, and channel incision, resulting in drier and homogenized conditions. However, in addition to agricultural fields, large areas of the lower watershed comprising the CRP consist of perennial and seasonal marshes, successional riparian forest, and oak woodlands.

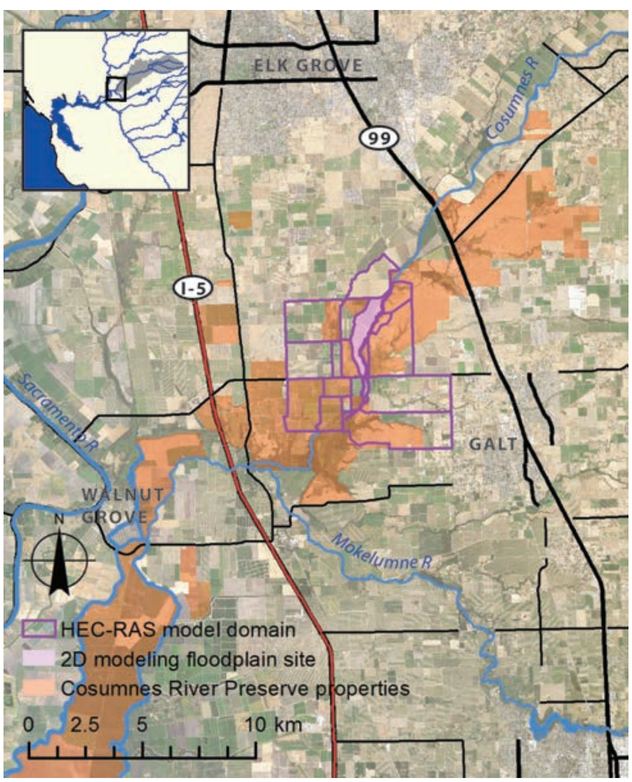

Figure A-1. The lower Cosumnes River, including the Cosumnes River Preserve and lower Cosumnes River floodplain restoration project area. (Aerial imagery: USDA 2016)

The study area for this modeling effort is the most recent restoration project within the CRP – one of a series of levee breach and levee setback projects conducted over the last several decades to restore floodplain wetlands and riparian forests through a more natural flood regime (Mount et al., 2002; Swenson et al., 2012). The restoration project was implemented by The Nature Conservancy and constructed in the fall of 2014 (Robertson-Bryan Inc., 2011). The main construction elements within the ~1.2 km^2 site consisted of the removal of 370 m of levee at the upstream end of the floodplain for increased flood frequency, the breaching of an interior "ring" levee at three locations to promote flooding, and the creation of a 330 m long swale and 90 m breach at the downstream end for drainage (Figure A-2). The downstream swale and levee breach were included due to concerns about fish stranding. Similar to previous projects, the goal was to increase hydrologic connectivity between the river and its floodplain, increasing the extent, frequency and duration of inundation within the restoration area through levee breaching to promote floodplain processes and functions. This includes enhancing sediment depositional and erosional processes, nutrient export downstream, generation of wetlands and successional riparian forests, and creation of habitat for fish spawning and rearing.

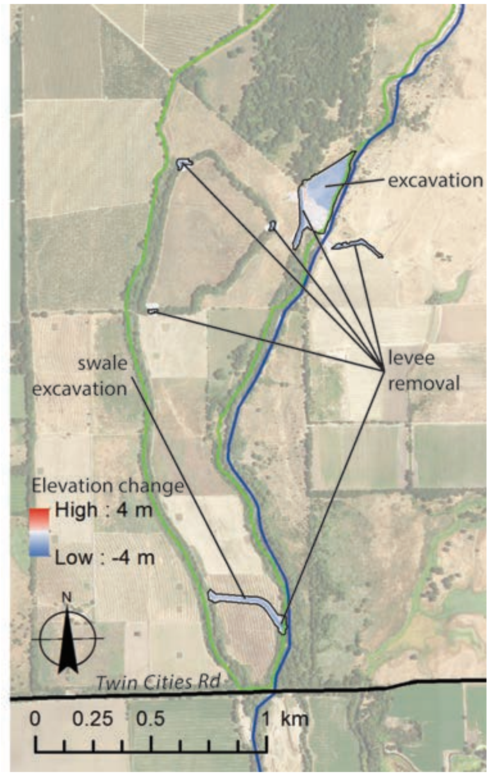

Figure A-2. Locations of the construction elements for the lower Cosumnes River floodplain restoration project. Elevation change resulting from the construction is shown where blue indicates reduction in elevation post-restoration and red indicates increased elevation post-restoration. The floodplain area modeled in 2D is outlined in green. (Aerial imagery: USDA 2016)

MODEL DEVELOPMENT

Model geometry

The model domain consists of >8 km of Cosumnes River channel extending from 3.4 km downstream of the USGS McConnell gage (MCC; USGS #11336000) to 2.8 km downstream of where the river flows under Twin Cities Road (Figure A-3). Elevation ranges between approximately 15 m at the upstream boundary and along high levees to approximately 0 m in the downstream river channel (NAVD88; Figure A-4). Model domain extent was determined through a balance of achieving a large enough area to allow flood flows to move appropriately within the model, while maintaining a limited extent to reduce model complexity and run times. The total model domain area is approximately 42.4 km^2.

The floodplain site of interest for 2D modeling is approximately 2.1 km^2. It consists of the restoration site as well as the Shaw Forest to the north. The Shaw Forest was included in the 2D area because of its hydrologic connection to the restoration site. No major levees separate the Shaw Forest from the Cosumnes River or from the restoration site. In addition, a large side channel at the upstream end of the Shaw Forest branches out and flows into the restoration site, and another side channel downstream also conveys water through the Shaw Forest and into the restoration site. All other floodplain areas were modeled as 1D storage areas. Storage area boundaries were set along defining levee or road features and modeled as weir structures. A total of 19 storage areas were included in the model. The 2D area was connected to the 1D channel using lateral weir structures along the full extent of the floodplain. Its other boundaries were connected to storage areas, also through weir connections.

Several significant in-channel features were also included in the model. In the vicinity of the upstream levee breach, the Low Water Crossing is a concrete structure crossing the channel that serves as a road in times of low water. It extends about a meter above the channel and contains a 1.8 m wide rectangular culvert. Also, two bridges span two branches of the Cosumnes River just downstream of the

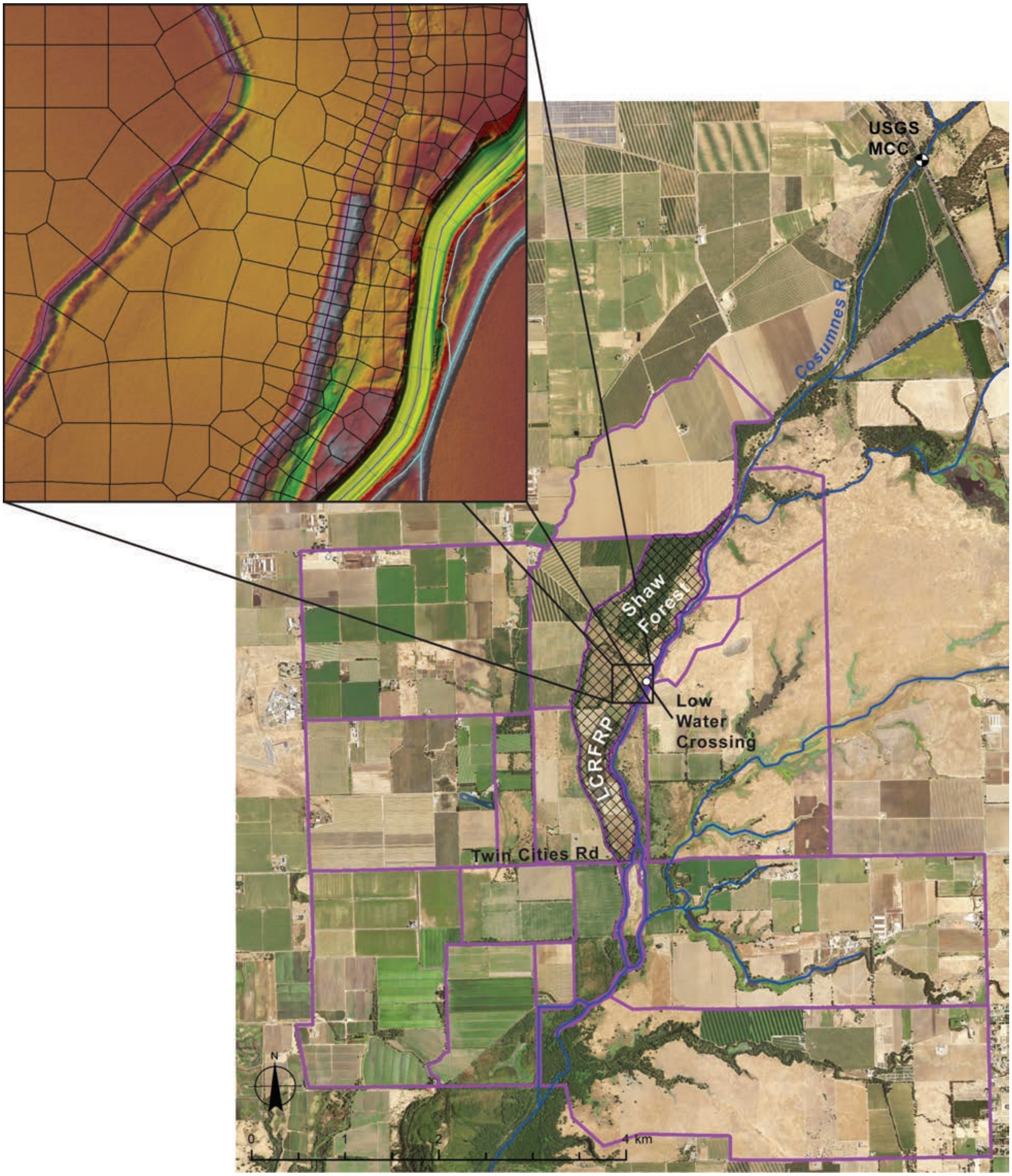

Figure A-3. HEC-RAS model geometry, with 1D storage areas (purple outline) and the Lower Cosumnes River Floodplain Restoration Project (LCRFRP) area and Shaw Forest modeled in 2D (hashed area). The inset depicts the mesh structure (black lines) over the 1 m resolution DEM for the 2D area with breaklines (pink lines) along levees to establish the mesh orientation. (Aerial imagery: USDA 2016)

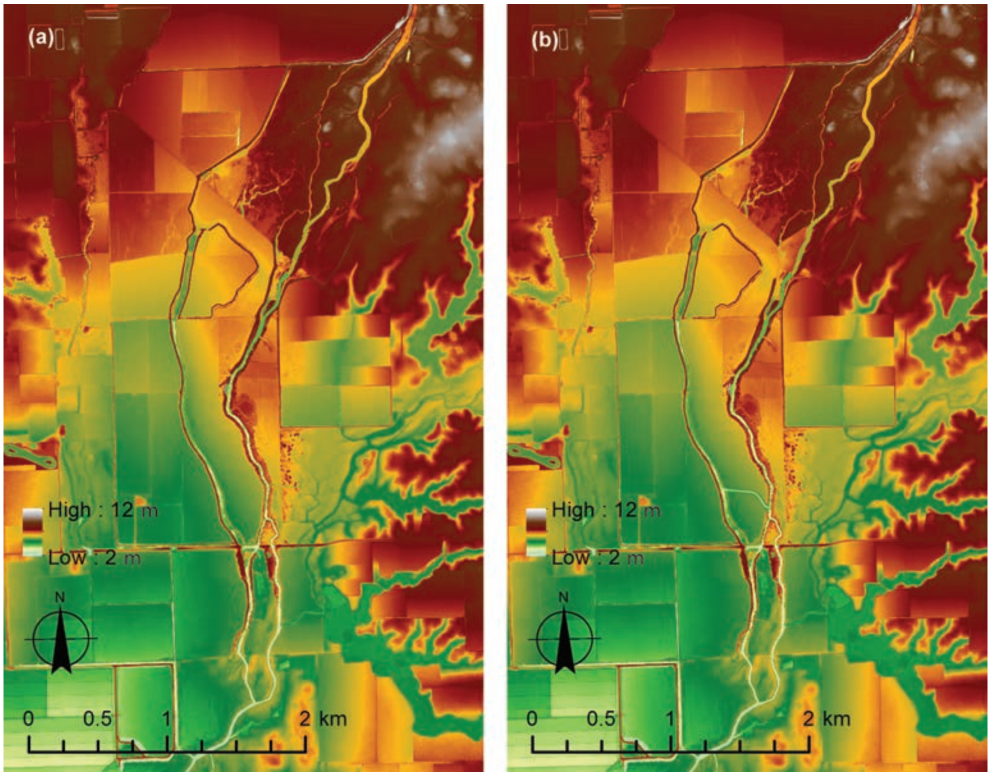

Figure A-4. Topography within the floodplain area of focus and the surrounding area for pre- (a) and post-restoration (b) conditions. (LiDAR: CDWR, 2010).

restoration area. These bridges were surveyed and represented in the model. Culverts along Twin Cities Rd were also included at connections between the model storage areas (e.g., for Lower Laguna Creek).

Development of the model geometry relied most heavily upon 2007 LiDAR available for the region from the California Department of Water Resources (DWR) Central Valley Floodplain Evaluation and Delineation (CVFED) project (CDWR, 2010; Hegedus & Simmons, 2011), processed at a 1 m resolution. Unfortunately, despite the channel bed being largely dry when the data were gathered, the dense riparian tree canopy distorted bare earth measurements in some locations, resulting in anomalous bumps and an error in bed elevation on the order of 1 m in some locations. To correct these issues, topographic surveying was performed using Real Time Kinematic (RTK) Global Positioning System (GPS) equipment (Topcon HiperLite+ and HiperV) for channel cross sections at an approximate spacing of 200 m within the model domain. The channel surface interpolation tool within HEC-RAS was applied to these cross-sections to establish a channel surface DEM, which replaced the LiDAR data in those locations. High resolution RTK surveys were also performed post-restoration, in the areas that were substantially altered topographically. These survey points were used in these areas in place of the 2007 LiDAR to establish a post-restoration DEM for use in the modeling.

The computational mesh within the 2D floodplain area was revised from an initial rectangular grid at a resolution of 50 m to conform to local topography using breaklines. Higher resolution cells were added in topographically complex areas. This resulted in cell sizes ranging from approximately 110-4,200 m^2 (~10-65 m resolution). The total number of cells was 1,277 for the pre-restoration geometry and 1,395 for the post-restoration geometry. An important aspect of developing the computational mesh was to make sure that cell boundaries aligned with higher topographic areas and that topographic ridges did not extend across cells. Given that the model uses computational hydraulic tables established from the cell edges, any obstructions to flow (such as ridges of high land) are not seen by the model within cells.

Boundary conditions

METHODS

To estimate inflow boundary conditions, discharge at the nearest streamgage in operation was adjusted to account for lag time and tributary inflows. The gage on the Cosumnes River at Michigan Bar (MHB, #11335000; U.S. Geological Survey, 2017) lies approximately 45 km upstream of the model boundary, with daily discharge available continuously from the 1908 water year and 15-min discharge available continuously from the 1984 water year. To determine lag times and scaling factors for tributary inflows based on MHB discharge, results from previous hydrologic modeling of the Cosumnes River watershed were used (David Ford Consulting Engineers, 2004). The purpose of the hydrologic modeling performed by David Ford Consulting Engineers (DFCE) was to determine flood frequency flows for the North Delta HEC-RAS model, developed by Sacramento County, California Department of Water Resources, and Sacramento Area Flood Control Agency. The tributary runoff hydrographs used to determine model inflow include upper and lower Deer Creek, local input near Highway 99, upper and lower Laguna Creek, and Badger Creek (Figure A-5).

Correlations with the 15-min discharge data at MHB were developed, including a lag to account for the travel time between the gage and model boundary and scaling and lagging for additions of tributary flow determined from relationships established using the hourly DFCE flood hydrology. Travel time lags were established by comparing the time between MHB peak flood flows and the associated peaks in stage at two sites (LWC and WB; see Figure A-8) in the upper reach of the model. The highest flows within 36 hours of each other were used, so as not to confuse downstream peaks. Lags for the discharge of each tributary were estimated by comparing the time of local peaks between MHB and each site in the DFCE hydrology. These were added to the travel time lags. To determine the scaling factor (percentage of MHB discharge) for each tributary, peak discharge from the DFCE hydrology for MHB and each tributary were compared using linear regression. Once the scaling factors and lags were determined, MHB hydrographs selected for modeling were adjusted to estimate each input. Deer Creek and local inflows near Highway 99 were added to the lagged MHB hydrographs and Laguna and Badger Creek hydrographs were kept separate for use as lateral inflows in the model. Since the DFCE

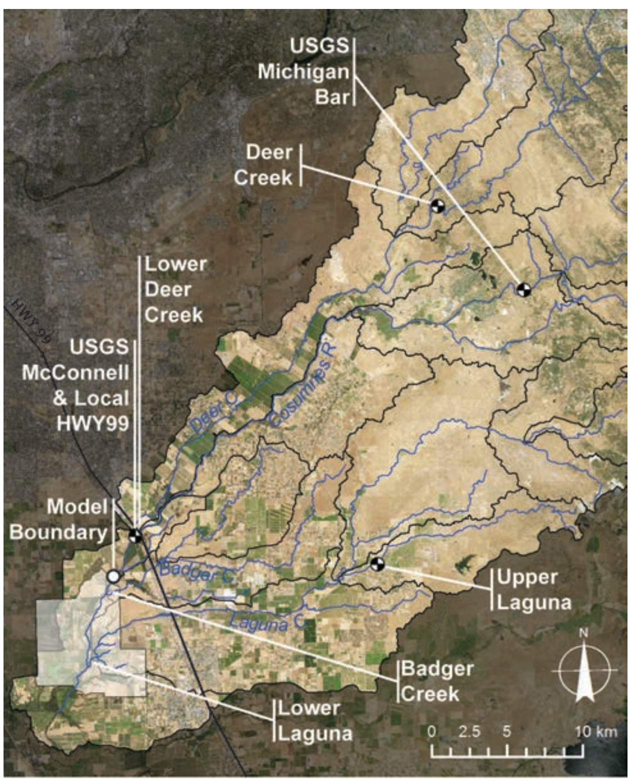

Figure A-5. Map of tributaries to the Cosumnes River, for which time lags and scaling were determined based on correlations to USGS MHB discharge to predict model inflow. (Aerial imagery: USDA 2016) hydrologic modeling considered lower Laguna Creek and Badger Creek together, these two contributions were separated using the percentage of watershed area. All processing was performed in *R* (R Core Team, 2016). This approach is similar to that applied in previous hydrodynamic modeling of the lower Cosumnes River floodplain (Robertson-Bryan Inc., 2011).

To evaluate the effectiveness of this transformation, predicted daily discharge at the upstream model boundary for 199 flood events (2,775 days) isolated from the historical record were compared against daily discharge recorded at the MCC gage, which was in operation between the 1943 and 1982 water years. This gage lies about 3.25 km upstream of the model boundary and no major tributaries enter the river along the reach, so the comparison was considered appropriate. Performance measures used to assess goodness of fit between the predicted inflow and the measured MCC discharge included the Nash-Sutcliffe Efficiency (NSE), percent bias (PBIAS), and the ratio of root mean square error (RMSE) to the standard deviation of observations (RSR; Moriasi et al., 2007).

For downstream boundary conditions, the normal depth assumption was applied, with the friction slope used as a calibration parameter. Importantly, the distance between the 2D floodplain site of focus and the downstream boundary (~3 km) is great enough such that the downstream boundary does not appear to affect conditions within the floodplain. Using the normal depth was necessary as a rating curve is unavailable for higher flood flows. Also, the streamflow gage downstream has a relatively short record period and is influenced by tides and the Mokelumne and Sacramento Rivers, preventing establishment of a satisfactory relationship between it and the downstream boundary.

INFLOW PREDICTION PARAMETERS

Estimated parameters for scaling and lagging MHB discharge to predict model inflow, established by correlating MHB discharge with the DFCE flood hydrology, are shown in Table A-1. They indicate that the most substantial tributary contributions come from lower Deer Creek (~15% of MHB discharge) and Upper Laguna (~11% of MHB discharge). Travel time between MHB and the model boundary is approximately 13.2 hours. Lag times for the tributary sites are relative to MHB discharge and include travel time to the model boundary.

Several uncertainties exist in determining model inflow using this approach. As no hydrodynamic modeling is involved, flood wave attenuation as the hydrograph moves downstream is not included, and lag times do not vary with flow. Another issue is that precipitation and runoff patterns for the tributary watersheds can be quite different from those of the mainstem Cosumnes River, so the simple scaling factors only approximate the central tendency. That is, the percentage contribution of each tributary to total discharge at the model boundary varies depending on the storm magnitude as well as individual storm characteristics. It is likely that tributary contributions are over-predicted at lower flows and under-predicted at higher flows. In general, the lower elevation tributary watersheds do

Table A-1. Time lag and scaling factors for MHB discharge to account for travel time and tributary inflow used to determine model inflow boundary conditions.

Site Name	DFCE Code	Lag (hours)	MHB scaling factor (%)
Michigan Bar	JCN29	13.23	100.00
Deer Creek	CN30	7.58	6.53
Lower Deer Creek	JCN35A	0.70	14.67
Local HWY99	CN36	-3.80	2.59
Upper Laguna	R380	8.17	11.06
Lower Laguna	LAGLOW	-3.87	4.46
Badger	LAGLOW	-3.87	3.56

not receive as much orographic precipitation, are less influenced by precipitation falling as snow, and are generally flashier in character. Further, travel times are also approximations, which will vary depending on the individual event, discharge, and channel slope. Additional uncertainty originates from the DFCE hydrologic modeling upon which these relationships are based.

INFLOW PREDICTION PERFORMANCE

The assessment of predicting model inflow based on lagging and scaling discharge at the MHB gage revealed a good fit overall. The predicted model inflow fit the MCC discharge with a NSE of 0.69, a PBIAS of 9.8%, and an RSR of 0.55, within the criteria set by Moriasi et al. (2007) for a satisfactory fit (see "Goodness-of-fit Measures" section below). A linear fit of observed and predicted discharge shows that discharge is, on balance, under-predicted, though over- versus under-prediction appears to be fairly evenly distributed (Figure A-6). Figure A-7 illustrates several of the larger-magnitude predicted flood hydrographs compared to MCC discharge, which shows peak flows both over- and under-predicted. In some cases, the flood peaks don't match on the same day, which can be attributed to the necessity of using mean daily discharge as opposed to 15-min data for this assessment (15-min data are unavailable for the MCC gage). Overall, in the absence of full hydrodynamic modeling for the lower watershed, this approach provides a reasonable approximation of inflow at the model boundary based on the upstream MHB gage.

CALIBRATION AND VALIDATION

Methods

Model calibration and validation relied primarily upon water surface elevation (WSE) monitoring at 15-minute intervals at several in-channel and floodplain locations over the three years leading up to the restoration (2012-2014 water years) and two years after restoration (2015-2016 water years). Of the monitoring performed, six stations were used for the primary calibration and validation process, which included one in-channel location along the Oneto-Denier floodplain reach (LWC), one downstream in-channel location (TC), and four sites across the upstream to downstream gradient of the floodplain surface (Site115/ODF1, NWSplay/ODF2, Site208/ODF3, and Site108/ODF4; Figure A-8). The number of monitoring locations was expanded over the course of the project. The locations of several floodplain monitoring sites changed before and after restoration.

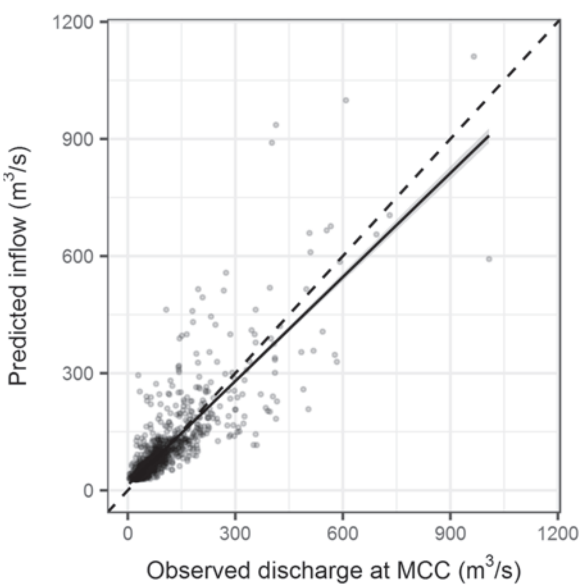

Figure A-6. Relationship between observed discharge at MCC gage versus predicted inflow for the upstream model boundary, which are geographically close and thus offer an assessment of the prediction effectiveness. The linear fit (including 95% confidence interval) is shown along with the 1:1 line (dashed line).

One flood event for each restoration scenario (pre- and post-restoration) was used for calibration and another from each for validation. Relatively few floods of substantial magnitude occurred, attributable to the severe 2012-2016 state-wide drought. In particular, this limited the floods available for pre-restoration model validation.

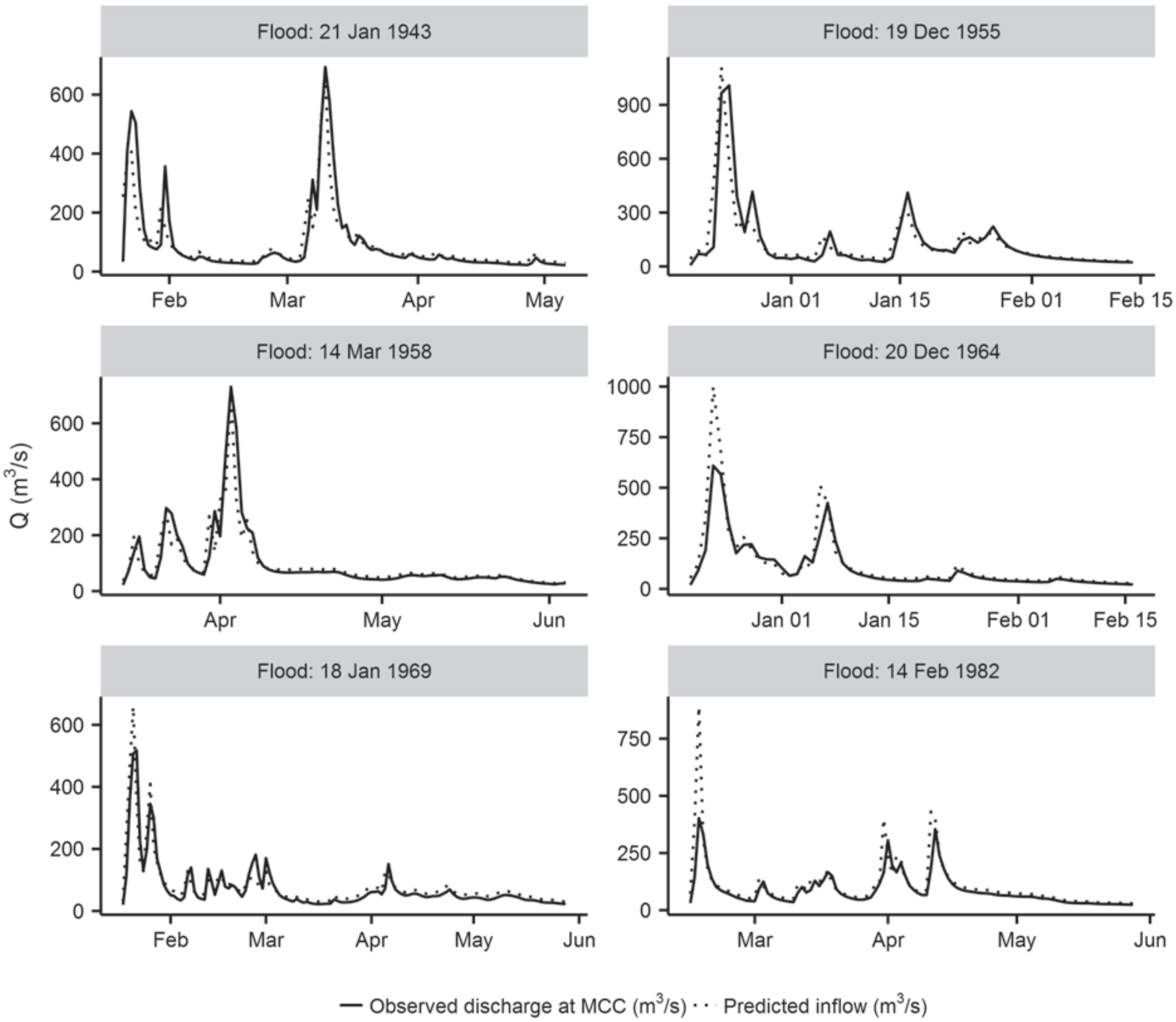

Figure A-7. Six selected flood hydrographs comparing observed (MCC discharge) and predicted inflow at the model boundary, based on daily discharge at MHB, approximately 45 km upstream.

For pre-restoration calibration, a single-peak three-day flood (9–12 February 2014) was selected, which reached a maximum mean daily flow of 71.4 m^3/s at MHB (~1.27-yr recurrence interval). This was the only flood with substantial floodplain inundation observed during the pre-restoration monitoring period. A larger magnitude flood event (15–19 March 2012), which reached a maximum mean daily flow of 139.9 m^3/s (~1.55-yr recurrence interval) at MHB, was used for validation. Floodplain WSE observations are unavailable for this period, however. For post-restoration calibration, a single peak three-day flood (21–24 December 2015) was selected, which reached a maximum mean daily flow of 82.7 m^3/s at MHB (~1.28-yr recurrence interval). A multi-peaked, larger-magnitude event, with a maximum mean daily flow of 202.7 m^3/s at MHB (~1.85-yr recurrence interval) served as a validation flood.

Other available information was also used in model calibration. This included aerial photographs taken during floods, which helped assess the spatial extent of inundation. Specifically, the pre-restoration model was evaluated with aerial photography, courtesy Roberson-Bryan, Inc., taken a day after peak flows of 362 m^3/s at MHB on 24 March 2005 (~300 m^3/s at upstream model boundary) and the post-restoration

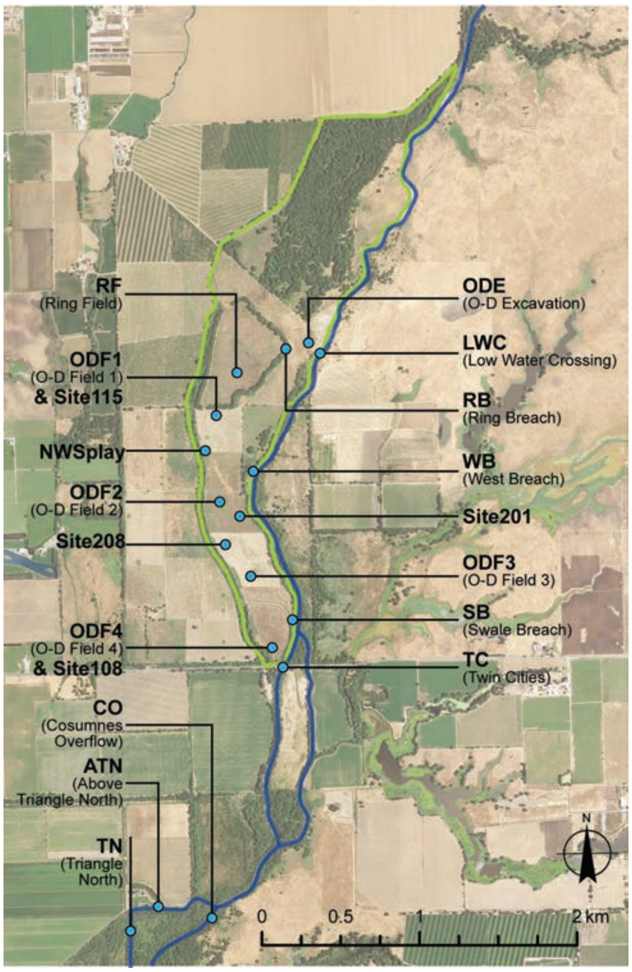

Figure A-8. Locations of WSE measurements within the floodplain restoration area (green outline) and downstream. Five stations are located in-channel: LWC, TC, CO, ATN and TN. See Table A-2 for information on when stations were in operation (pre- or post-restoration). (Aerial imagery: USDA 2016)

model was evaluated with UAV photography, courtesy UC Merced, from 15 March 2016 (~200 m^3/s at upstream model boundary). Each flood event was modeled, such that inundation depth at the approximate time of two sets of aerial photographs could be used. Also, model performance at low flows was evaluated based on flow observations taken at the in-channel LWC site adjacent to the floodplain of focus.

MODEL PARAMETERIZATION AND SENSITIVITY

A number of model parameters were used in the process of calibration. Surface roughness, typically represented by Manning's *n*, is the most common calibration parameter for hydrodynamic models. Adjustments were made to both surface roughness of the channel as well as the floodplain surface (spatially varying with land cover type; Figure A-9). With normal flow set as the downstream flow boundary condition, friction slope was another parameter adjusted during calibration. Several weir coefficients were also used as calibration parameters, including the LWC and the levees (modeled as lateral weir structures) along the river. Each parameter was adjusted independently to assess the relative sensitivity of WSE at the channel and floodplain sites under consideration. In some cases, changing a parameter improved model performance at one site, but decreased it in another. A generalized summary of the response to parameter adjustments is given in Table A-2.

As expected, increasing friction slope at the downstream boundary decreased WSE and disproportionately affected the WSE at downstream sites. For most sites, WSE increased in response to increasing roughness along channel reaches, though the relative response varied depending on the site and reach altered. Response to increasing floodplain surface roughness differed across vegetation type and site. Responses varied somewhat depending on the actual value of the parameter, the values of other parameters, and the flow magnitude. A formal sensitivity analysis was too resource intensive given the many parameters involved and not necessary for model calibration adequate for the purposes of the research.

GOODNESS-OF-FIT MEASURES

To evaluate model performance, several commonly-used measures of goodness-of-fit were applied to observed and modeled WSE. It is well understood that multiple measures should be used to assess multiple aspects of model performance (Jain & Sudheer, 2008). As recommended by Moriasi et al. (2007), the Nash-Sutcliffe efficiency (NSE), percent bias (PBIAS), and the ratio of root mean square error (RMSE) to the standard deviation of measured data (RSR) can together provide a reasonable evaluation of model fit. The NSE compares the variance in observed data with the residual variance, PBIAS quantifies the overall tendency to over- or under-predict the data, and RSR is a measure of the RMSE relative to standard deviation of observations. Moriasi et al. (2007) considers model fits satisfactory if NSE is greater than 0.50, PBIAS is within 25%, and RSR is less than or equal to 0.70. For simulation of single events, as is performed here, a measure of the percent error in peak flow rate is also useful (Moriasi et al., 2007). The use of these four measures fits the recommendation that both goodness-of-fit and absolute error measures are used to evaluate model performance (Legates & McCabe, 1999). Visual inspection also aided evaluation of where and under what conditions the models over- and under-predicted relative to observations. Evaluation of model performance was also considered within the context of the modeling purpose, which is most concerned with relative differences between the pre- and post-restoration scenarios. Thus, evaluation considered the representation of the interaction of in-channel flow and floodplain topography as represented in floodplain inundation patterns, depth, and velocity.

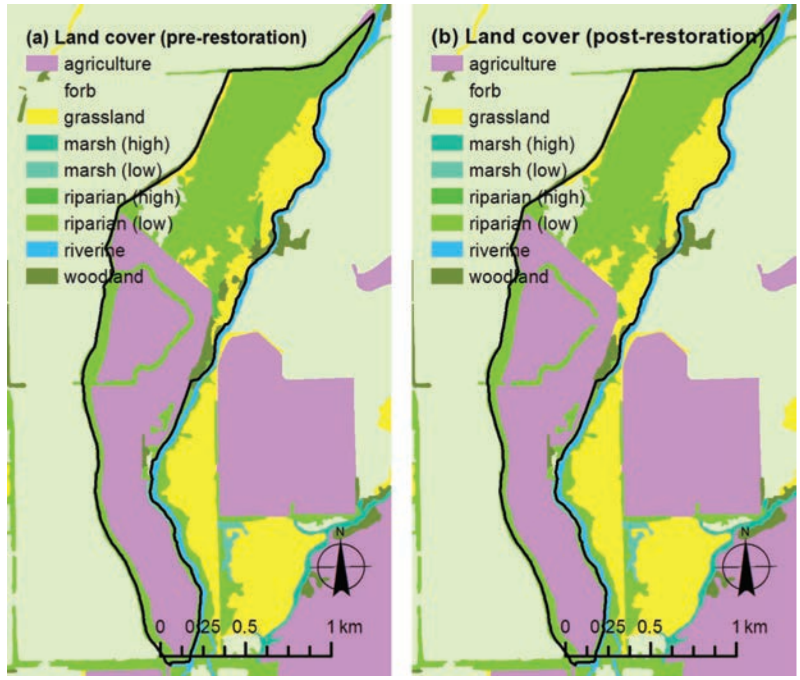

Figure A-9. Land cover used for spatially variable Manning's n surface roughness within the 2D floodplain area of the pre- (a) and post-restoration (b) model. (Land cover: GIC 2012)

Results

CALIBRATION AND VALIDATION FLOODS

Overall, successful model calibration for both the pre- and post-restoration conditions was achieved through iterative adjustments to model parameters, including downstream friction slope, Manning's *n*, and weir coefficients. This was an involved process given the many individual parameters that could be adjusted. Calibrated friction slopes ranged from 5×10^{-5} and 1×10^{-4}. In-channel Manning's *n* values of 0.03 and 0.038 were used and varied depending on the scenario and reach. Slightly higher Manning's *n* values were used for most land cover types of floodplain surface in the post-restoration model compared to the pre-restoration model (0.055, 0.06, and 0.15 compared to 0.035, 0.05, and 0.08 for agriculture, grassland, and woodland types, respectively). A Manning's *n* of 0.1-0.16 was used for riparian forest. The LWC weir coefficient was 1.7 and 1.3 for the pre- and post-restoration model, respectively. Weir coefficients of lateral structures along the river channels varied between 0.3 and 0.7 (all based on SI units). As noted in the HEC-RAS model documentation, appropriate weir coefficients for lateral structures

Table A-2. Effect of model parameters on WSE. The relative strength of direct (green) or inverse (red) relationships between model parameters (grouped by friction slope, channel Manning's *n*, floodplain Manning's *n*, and weir coefficient) and WSE is given for each restoration scenario and gaging site. The degree to which WSE at sites responded varied depending on the actual value of the parameter, the values of other parameters, and flow magnitude, necessitating the more general summary provided here. See Figure A-8 for the site locations.

	Friction slope		**Manning's *n* (channel)**			**Manning's *n* (flood-plain)**				**Weir coefficients**					
Site	**Main channel**	**Side channel**	**Upstream reach**	**West branch**	**East branch**	**Agriculture**	**Grassland**	**Riparian**	**Woodland**	**Low Water Crossing**	**2D area south border**	**2D area below Twin Cities Rd**	**Cosumnes branches**	**Upstream reach W levee**	**Downstream reaches**
Pre-restoration															
LWC															
NWSplay															
Site108															
Site201															
Site208															
TN															
Post restoration															
ATN															
CO															
LWC															
ODE															
ODF1															
ODF2															
ODF3															
ODF4															
RB															
RF															
SB															
TC															
TN															
WB															

are important for the 2D modeling. Namely, the weir coefficients should generally be lower than what is commonly used for inline weirs; otherwise, too much flow will move from the 1D channel into the 2D area (Brunner, 2016).

As summarized in Table A-3, both pre- and post-restoration models performed well for most sites across all four performance measures. The NSE for the LWC in-channel site was between 0.7 and 0.8 for both model calibration runs, and 0.93 for the model validation runs. The floodplain sites averaged 0.58 for pre-restoration model calibration, averaged 0.85 for post-restoration model calibration, and averaged 0.84 for post-restoration model validation. Overall, the post-restoration model appears to perform

Table A-3. Summary of goodness-of-fit measures to evaluate model performance for the pre- and post-restoration scenarios. Model performance was evaluated using the Nash-Sutcliffe Efficiency (NSE), Percent bias (PBIAS), RMSE relative to standard deviation of observations (RSR), and Percent error in peak flow (% Peak Error). Bolded values represent a satisfactory model fit following Moriasi et al. (2007).

Site	Calibration				Validation			
	NSE	PBIAS	RSR	% Peak Error	NSE	PBIAS	RSR	% Peak Error
Pre-restoration								
LWC	**0.8**	**-1.6**	**0.45**	**0.04**	**0.93**	**1.4**	**0.27**	**5.75**
NW Splay	**0.53**	**-1.2**	**0.68**	**-0.2**	*NA*	*NA*	*NA*	*NA*
Site 108	**0.86**	**-0.7**	**0.38**	**1.02**	*NA*	*NA*	*NA*	*NA*
Site 201	0.27	**0**	0.85	**0.29**	*NA*	*NA*	*NA*	*NA*
Site 208	**0.66**	**0.45**	**0.58**	**0.45**	*NA*	*NA*	*NA*	*NA*
TN	-0.49	**4.4**	1.22	9.16	-0.01	**4.5**	1.01	**9.25**
Post-restoration								
LWC	**0.73**	**1.9**	**0.52**	**-1.34**	**0.93**	**-0.3**	**0.27**	**-0.24**
ODF1	**0.76**	**0.5**	**0.49**	**-0.17**	**0.83**	**-1.1**	**0.41**	**-1.47**
ODF2	**0.8**	**0.3**	**0.45**	**1.04**	**0.84**	**-2.5**	**0.4**	**-0.99**
ODF3	**0.89**	**1.2**	**0.34**	**1.94**	**0.84**	**-2.9**	**0.4**	**-1.5**
ODF4	**0.96**	**1**	**0.21**	**1.06**	**0.86**	**-3**	**0.37**	**-1.95**
TN	-0.19	**3.9**	1.09	7.73	**0.89**	**0.9**	**0.34**	**1.53**

better than the pre-restoration model within the floodplain area, but observational data are limited for the pre-restoration model. The model performed less well at the downstream model boundary (TN), which is attributed to the normal flow assumption. This did not appear to substantially affect model performance within the upstream floodplain area of interest. Site 201 in the pre-restoration model was also unsatisfactory for NSE and RSR, which may be explained by the complexity of the area near this site due to a previous levee breach.

Visual assessment of model fit for the pre-restoration scenario shows good matches for the peak of the hydrograph (apart from the TN downstream boundary; Figure A-10a). The broader shape of the observed in comparison to the modeled hydrograph is likely attributable to the method of predicting model inflow based on upstream flows as opposed to particular issues within the model. Unfortunately, evaluation at the floodplain sites is limited given the absence of a validation flood during the period the floodplain was monitored. However, the calibration flood used shows an over-prediction of the model peak, again likely related to the method of predicting model inflow. The over-prediction of the lower flows may be because low flows are approximately the elevation of the road bed of the LWC. Visual inspection of the post-restoration calibration and validation flood hydrographs suggests the model performs quite well for flood flows at the upstream channel site and within the floodplain of focus (Figure A-10b). Floodplain peaks are slightly over-predicted for the calibration flood but somewhat under-predicted for the validation

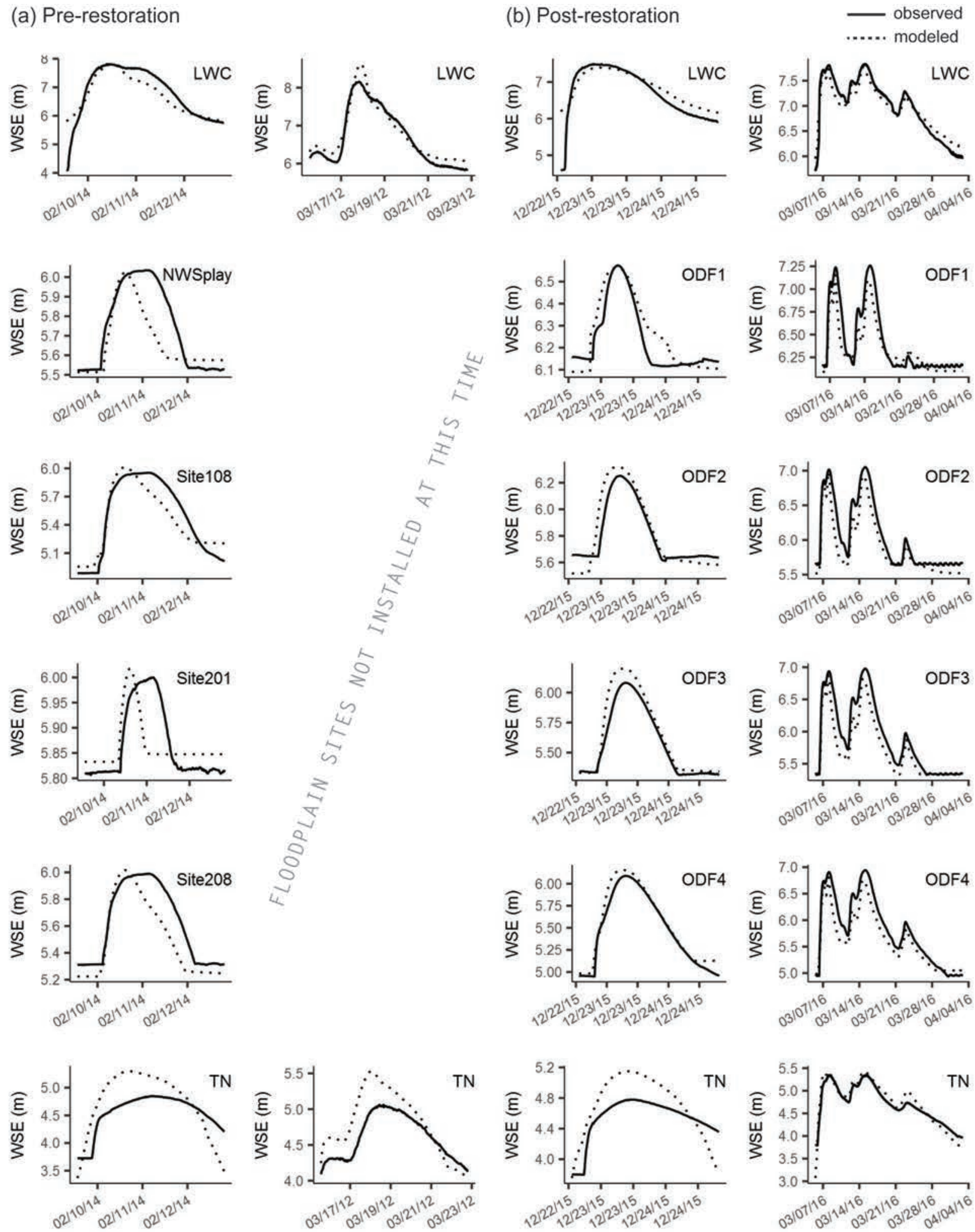

Figure A-10. Modeled and observed WSE for the pre- (a) and post-restoration (b) models at in-channel and floodplain sites for the calibration (left column) and validation (right column) floods.

flood. As with the pre-restoration model, the downstream boundary conditions (TN) do not fit the calibration flood well, though the higher flows of the validation flood do fit the observed data well. The differences in modeled and observed WSE at the floodplain sites before and after the flood hydrographs represent uncertainty in the bare surface elevation of the LiDAR dataset and the surveyed RTK elevation.

OTHER CALIBRATION

Modeled inundation extent for discharge was compared to aerial photography of two flood events, one of which occurred prior to restoration (photographs taken on 24 March 2005 by Robertson-Bryan, Inc), and the other of which occurred after restoration (photographs taken on 15 March 2016 by the University of California, Merced; Figure A-11). For the pre-restoration event, which peaked at 362 m^3/s at MHB a day before the photography was taken, the modeled inundation shows a reasonable match with the photographs. It appears as though the model may slightly over-predict depth and inundation extent based on some areas that do not appear inundated within the floodplain in the top photograph (though vegetation may be obscuring water). For the post-restoration event, the extent of flooding appears to match the available photography well. Modeling is performed in 1D outside the main floodplain area of interest, so some of the inconsistencies in those areas may be a result of this difference.

For evaluating low flows, seven discharge measurements (ranging from 1.31 to 6.51 m^3/s) and associated WSE at the LWC gaging location taken prior to restoration are available. To evaluate model performance, the observed WSE at these flows were compared with modeled WSE for those same flows (Figure A-12). The associated goodness-of-fit measures for these seven points include a NSE of 0.92, a PBIAS of 1.7%, and an RSR of 0.26. Modeled WSE tends to be over-predicted, which may be related to datum or flow issues. Also, challenges associated with accurately representing the weir structure and culvert at the LWC site may account for some of the uncertainty in the modeling at this location. Unfortunately, discharge measurements at higher flows are not available. Also, no discharge measurements were taken during the post-restoration period, so a similar comparison was not possible for the post-restoration model (though model geometry is nearly identical between the two models for low flows and should therefore be similar to these results).

ADDRESSING MODEL PERFORMANCE

Further improvement to the models would likely require improved model inflow prediction (see "Boundary Conditions" section) or simply extending the model to the upstream MHB gage. It is likely that the mismatches in the shape of hydrographs are primarily attributable to this issue. Useful model improvements could also include extension of the downstream boundary conditions to the downstream gage on the Mokelumne River or the development of a rating curve at the current downstream flow boundary. An overall larger domain may also allow for more realistic routing of flows because, in large flood events, floodwaters extend and communicate with areas outside the model domain. Furthermore, backwater from the Delta during these high flows complicates matters and also causes stage records to be inadequate representations of channel discharge in the downstream reaches. Surveyed cross sections were limited at the downstream end of the model and debris jams were not well represented, which would affect the conveyance capacity of the channel. Incorporating additional survey data for these reaches may therefore improve the model fit to observed data. Also, random or systematic errors are inherent in recorded field data, including the WSE data used for calibration. Further efforts in model calibration may also yield improvements to model performance. For example, varying hydraulic connectivity between floodplain storage areas modeled in 1D through weir coefficients was limited. These and other parameters

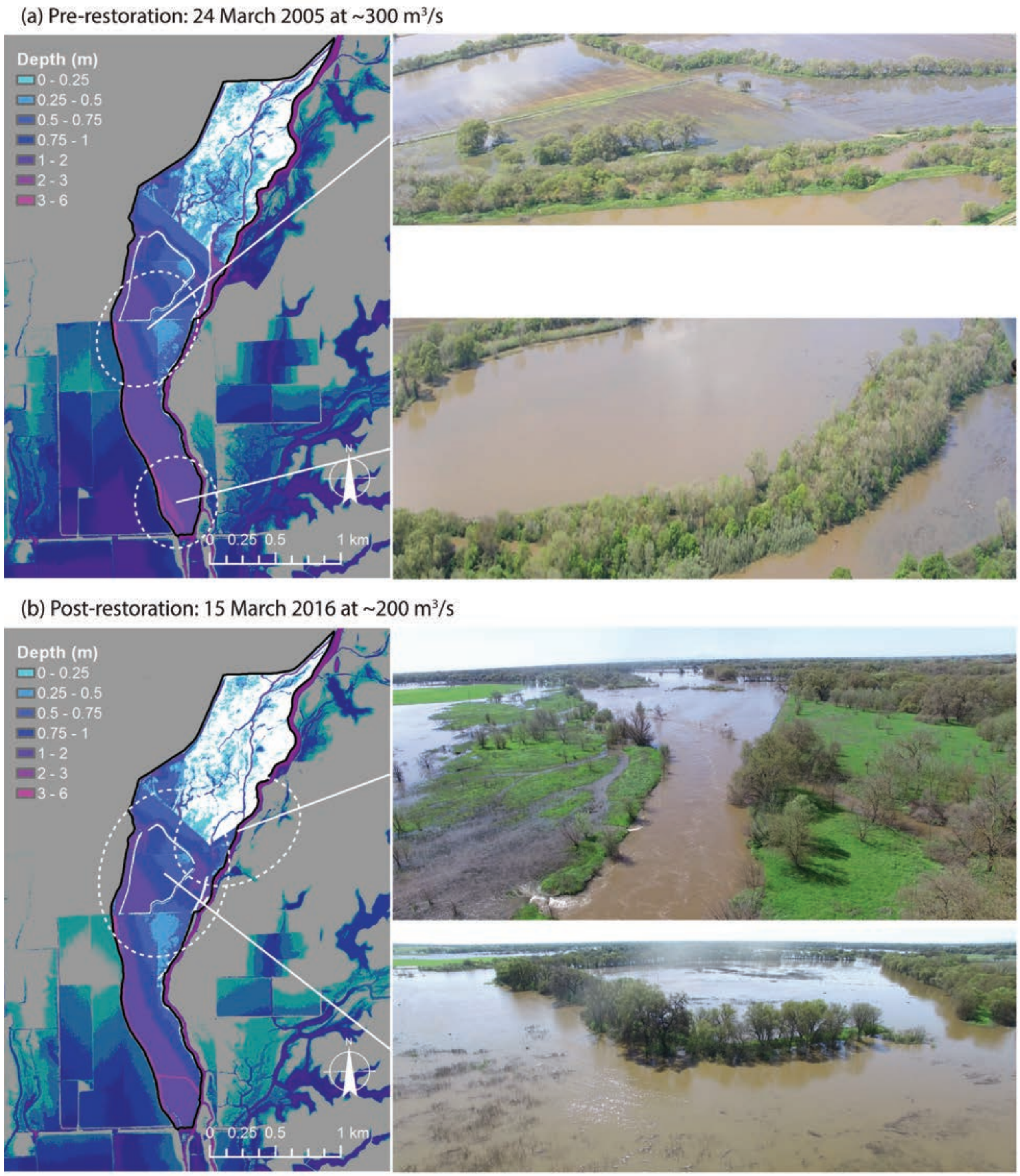

Figure A-11. Pre- (a) and post-restoration (b) model inundation depth maps compared against aerial photography of flood events before (24 March 2005) and after (15 March 2016) restoration. Areas modeled in 1D outside the 2D modeled floodplain of focus are shaded. Photos by Robertson-Bryan, Inc. (a) and University of California, Merced (b).

could be explored more thoroughly through a more formal sensitivity analysis.

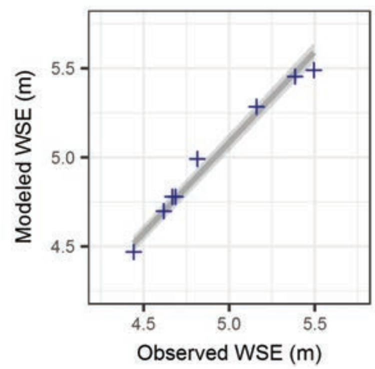

Figure A-12. Low flow observed versus modeled WSE at the LWC site based on seven discharge measurements made in the pre-restoration period. The model fit is associated with a NSE of 0.92, a PBIAS of 1.7%, and an RSR of 0.26.

CONCLUSIONS

The successful development of hydrodynamic models for the pre- and post-restoration conditions of the lower Cosumnes River restoration site, using the recently developed 2D capability of HEC-RAS 5.0, provides a valuable tool for research quantifying spatiotemporal floodplain inundation patterns to evaluate effects of restoration on physical processes and habitat availability. Establishing these models involved the creation of two model geometries, which relied primarily on a pre-restoration LiDAR DEM and post-restoration RTK surveys for the construction areas. With the model inflow boundary lying approximately 45 km downstream of the nearest active streamflow gage (MHB), correlations between discharge at this gage and the inflow boundary were established using existing streamflow data and previous hydrologic modeling for tributary inflows. With these relationships, a reasonable estimation of the downstream model boundary inflow hydrograph was determined from the upstream MHB hydrograph. For model calibration and validation, model parameters relating to the downstream friction slope, channel and floodplain Manning's *n* surface roughness, and weir coefficients were adjusted to improve the model fit for flood events occurring before and after restoration for which WSE data were available for two in-channel sites and four floodplain sites. Model performance was evaluated using goodness-of-fit measures commonly applied in hydrodynamic modeling, including the NSE, PBIAS, RSR, and percent error of peak flow. Model performance for both the pre- and post-restoration model was found to be satisfactory for the floodplain area of focus and the intended applications, which are primarily concerned with capturing relative differences in floodplain inundation extent, depth, and velocity pre- and post-restoration. Overall, these models provide the needed output for quantifying the hydrospatial regime of a floodplain and can be used in other research to better understand restoration impacts at this site.

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APPENDIX B

SACRAMENTO SPLITTAIL HABITAT SUMMARY

Sacramento splittail habitat summary

INTRODUCTION

The Sacramento splittail (*Pogonichthys macrolepidotus*) is a cyprinid fish native to California and listed as a state Species of Special Concern. The splittail is endemic to the Central Valley, occupying the San Francisco Bay-Delta estuary and lower tributaries. Though it still persists in much of its historical range, the splittail has been restricted due to dams and other land use changes (Moyle et al., 2010). It has been found as far upstream as Redding on the Sacramento River, Oroville on the Feather River, and Friant on the San Joaquin River. When not spawning or rearing, splittail primarily occupy the San Francisco Bay-Delta estuary, in the Delta and Suisun (Moyle, 2002). In addition to the populations existing within the Suisun Bay and Delta, a distinct population has been identified within the lower Napa and Petaluma Rivers. Some have suggested that the splittail has avoided extinction in part because of its still relatively broad distribution (Sommer et al., 2007). It is the only remaining species of its genus, since the Clear Lake splittail (*Pogonichthys ciscoides*) became extinct in the 1970s. Alongside Chinook salmon (*Oncorhynchus tshawytscha*), splittail are known for their use of floodplain habitats (Jeffres et al., 2008; Moyle et al., 2004). The splittail is understood to be the only California native fish species today that requires floodplains for its persistence (Moyle et al., 2007) and is therefore a particularly relevant species for assessing habitat benefits of floodplain restoration.

STATUS

Research showing population declines prompted the splittail being listed as threatened in 1999 by the US Fish and Wildlife service, but population increases in the 1990s – likely related to wet years – and an improved understanding of their population resilience caused their delisting in 2003 (Sommer et al., 2007). The splittail continues to be a Species of Special Concern, with long term declines related to extensive floodplain habitat loss, as well as passage impediments, food limitation, and food web changes (Moyle et al., 2007). The splittail is listed in a number of state habitat restoration plans, including the Delta Plan and EcoRestore, and various county-level habitat conservation plans. Historically, splittail were harvested by native peoples and later fished commercially (Sommer et al., 2007). Though the population is estimated to be only a fraction of historical numbers, splittail were the most abundant native large-bodied fish in a recent study of the Yolo Bypass in the Central Valley and there is still a recreational fishery on the Sacramento River (Moyle, 2017; Sommer et al., 2014).

HABITAT

Sacramento splittail are found in rivers, side channels, sloughs, and seasonal floodplains. Splittail migrate upstream in the winter and use seasonally flooded vegetation on floodplains for spawning and rearing. Though vegetation preference appears to vary depending on the age of the juvenile and the time of day, research suggests that submerged terrestrial vegetation is preferred over submerged aquatic vegetation and wetland vegetation such as tule (Sommer et al., 2008). Supporting the importance of floodplain habitats, a 2004 study on the Cosumnes River showed that juvenile splittail were in better condition in floodplain habitats than in riverine habitats (Ribeiro et al., 2004). Abundance appears to respond positively to the extent and duration of floodplain inundation, higher in wet years and lower in dry years (Sommer et al., 1997; Sommer et al., 2007). Consequently, it is thought that floodplain inundation (i.e., environmental factors) is the primary control on the population, as opposed to the number of adults (Sommer et al., 2007). Spawning begins in late January and continues into the spring (Moyle et al., 2004). Regular access to floodplain habitat is needed in the spring (February-May) for long durations of at least two weeks and ideally more (Matella & Jagt, 2014; Opperman, 2012; Sommer et al., 1997). Based on a survey of scientists, Suddeth (2014) proposed a minimum of two weeks of inundation, with added benefits up to eight weeks. Annual access to floodplain habitat is not required to maintain populations because splittail have been known to recover from several years of drought with limited floodplain access (Moyle et al., 2004; Sommer, 2017).

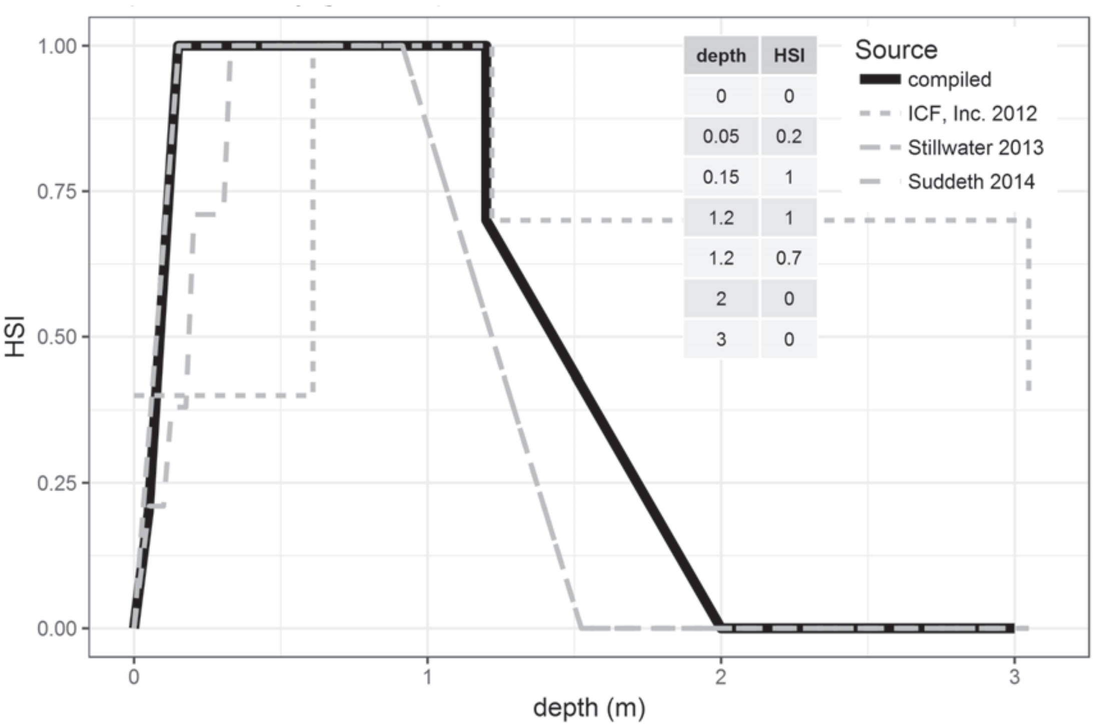
Different water depths are used depending factors such as life stage and time of day (Sommer et al., 2008). In general, juveniles are thought to prefer water <2 m deep and adults water less than four meters deep (Moyle et al., 2004). Splittail are thought to spawn in water <1.5 m deep (Opperman, 2012). They prefer relatively cool (<15 °C) and turbid water though they have been shown to tolerate temperatures ranging from 7-33 °C. Splittail also have relatively high salinity tolerances (18 ppt; Moyle et al., 2010). Given the variety of habitat preferences, managing for mosaics of interconnected habitat that provide a range of conditions in space and time is considered to be an important restoration strategy (Moyle et al., 2010; Sommer et al., 2008).

Valuable food sources for splittail are found in floodplain environments. As benthic foragers, splittail feed on zooplankton, opossum shrimp, benthic amphipods, copepods, terrestrial insects and detritus. Larval and juvenile splittail primarily consume cladocerans and chironomid larvae. The long duration of flooding not only allows for life history requirements of spawning and rearing, but also promotes the production of zooplankton. In particular, flood pulses spaced at two to three week intervals have been shown to maximize zooplankton production (Grosholz & Gallo, 2006).

Splittail habitat requirements and preferences based on existing literature and expert opinion have been compiled into depth, velocity, duration, and timing habitat suitability indices for juveniles and adults (Figures 1-5). In developing habitat suitability indices, restoration management planning typically focuses on depth either exclusively or with velocity, though duration and timing have also been used (Suddeth, 2014). Despite some existing research (Sommer et al., 2008), sufficient information to develop robust suitability indices for vegetation types is lacking. Temperature and salinity are not thought to be restrictive in general and less is known concerning turbidity preferences (ICF Inc., 2012). Also, interactive factors between physical variables, such as depth preferences changing with vegetation height or type (Moyle, 2017), are often not accounted for in typical habitat suitability criteria.

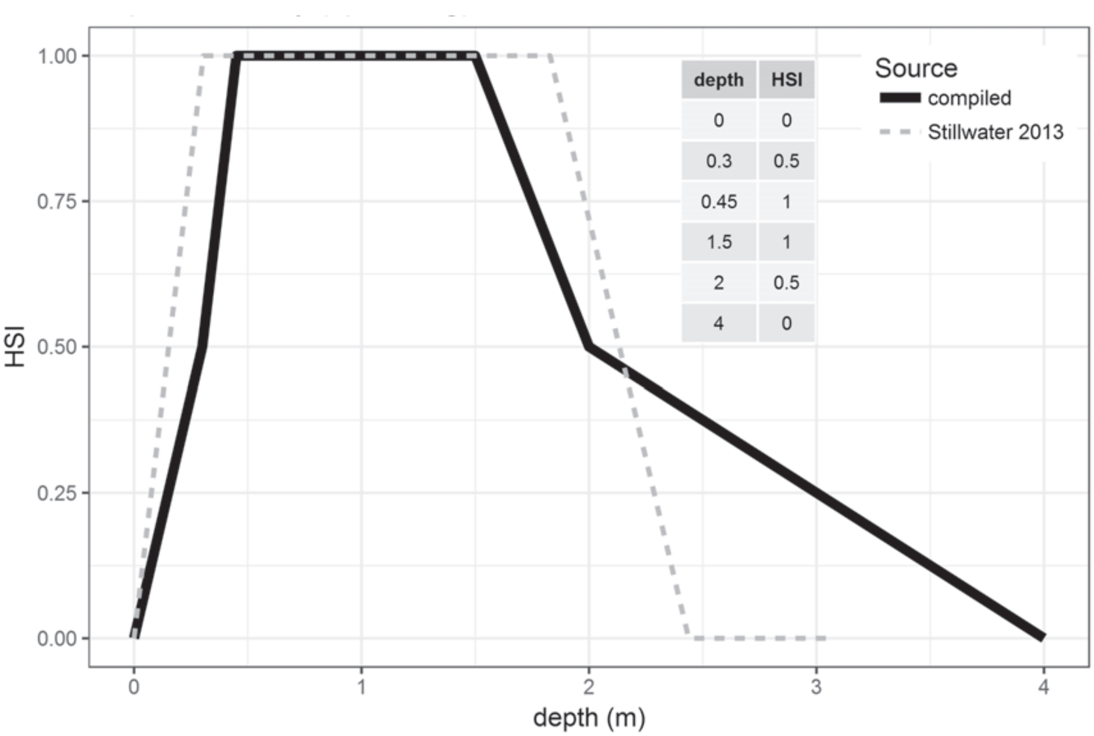
LIFE HISTORY

Within the inundated floodplains, spawning occurs on submerged vegetation and eggs stick to the vegetation while the eggs incubate for five to ten days. After hatching, the larvae remain in shallow inundated vegetation for another one to two weeks before moving into deeper water (Moyle et al., 2004; Sommer et al., 1997). Splittail are notable for their longevity (5-7 years) and fecundity (approximately 100,000 eggs per female). Adults can be up to 40 cm long and reach sexual maturity in their second year (Sommer et al., 1997). Because they are long lived, have high fecundity, and are opportunistic feeders, splittail are thought to be more resilient than other native fish such as longfin smelt or delta smelt (Sommer et al., 1997). That they have been found to spawn successfully in marginal habitat also suggests their ability to withstand dry years (Moyle et al., 2010). Overall, the life history of splittail is adapted to the high degree of variability and seasonally predictable conditions once found in the Delta (Sommer et al., 1997). Scientists have noted that native fishes, including splittail, generally rear earlier in the spring than non-native fishes, which is timed with when floodplain inundation typically occurs (Sommer et al., 2001).



depth	HSI
0	0
0.05	0.2
0.15	1
1.2	1
1.2	0.7
2	0
3	0

Figure B-1. Floodplain depth suitability for juvenile splittail compiled (bold black line) from existing literature, expert opinion, as well as several existing suitability curves (gray dashed lines; ICF Inc., 2012; Moyle et al., 2004; Sommer, 2017; Sommer et al., 2008; Stillwater Sciences, 2013; Suddeth, 2014).



depth	HSI
0	0
0.3	0.5
0.45	1
1.5	1
2	0.5
4	0

Figure B-2. Floodplain depth suitability for spawning splittail compiled (bold black line) from existing literature, expert opinion, as well as an existing suitability curve (gray dashed line) (ICF Inc., 2012; Moyle et al., 2004; Stillwater Sciences, 2013).

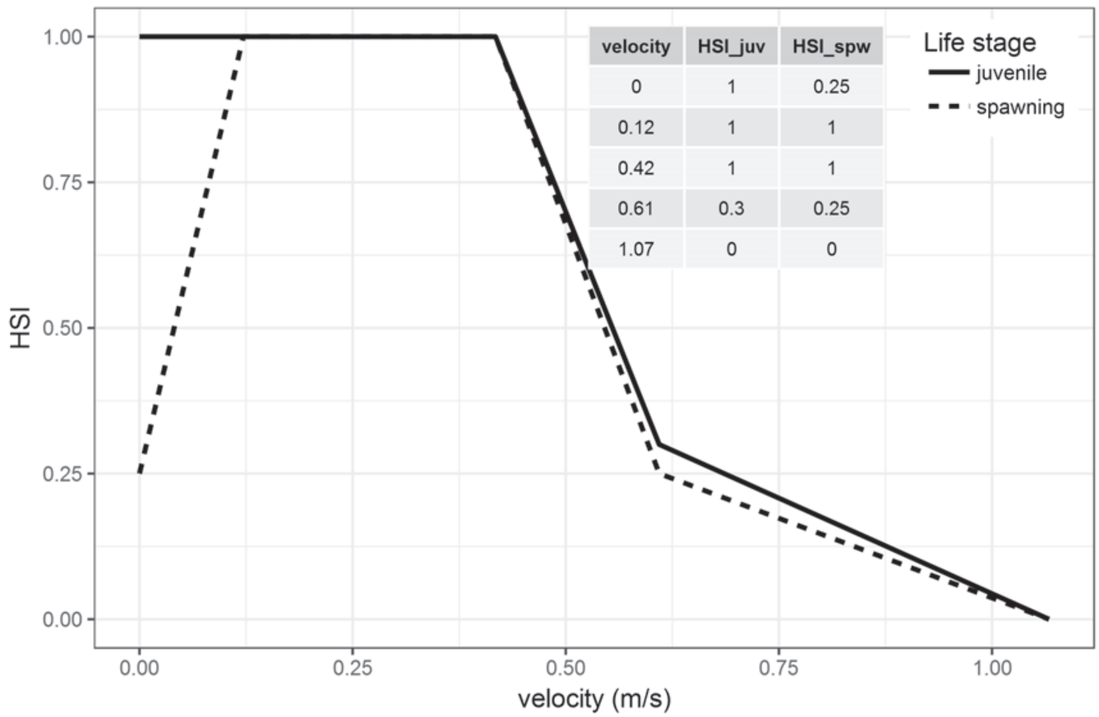


Figure B-3. Velocity suitability for juvenile and spawning splittail on floodplains adopted from Stillwater Sciences (2013).

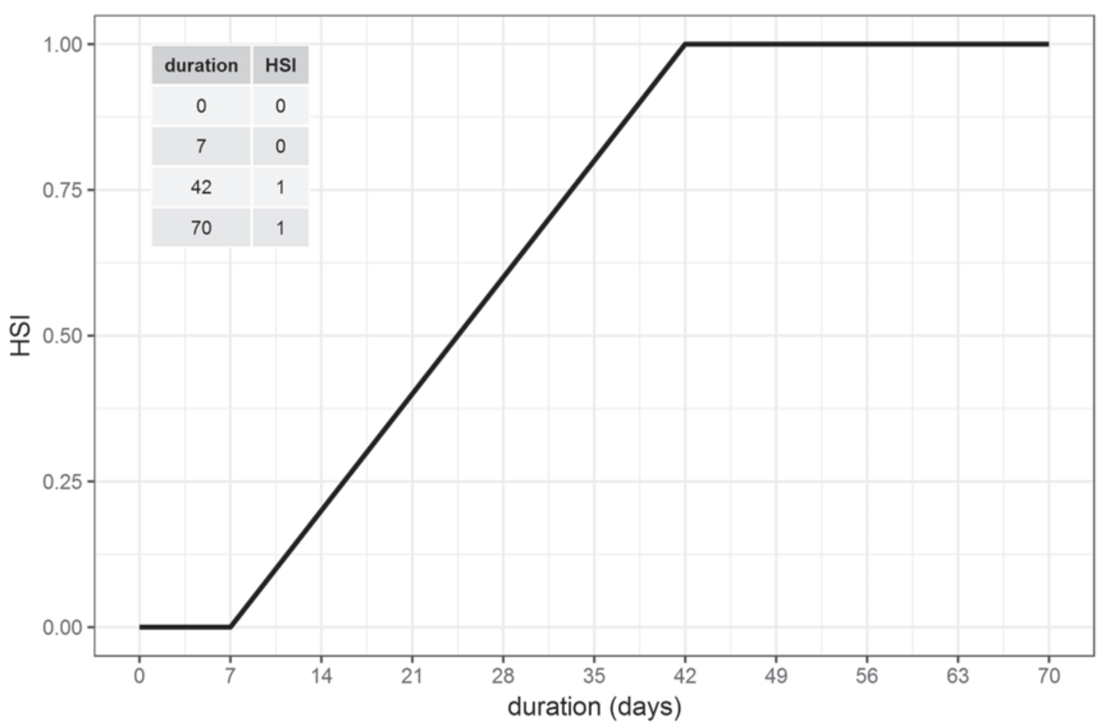


Figure B-4. Suitability of floodplain inundation duration for spawning and rearing splittail based on existing literature and expert opinion (Matella & Jagt, 2014; Sommer, 2017; Suddeth, 2014).

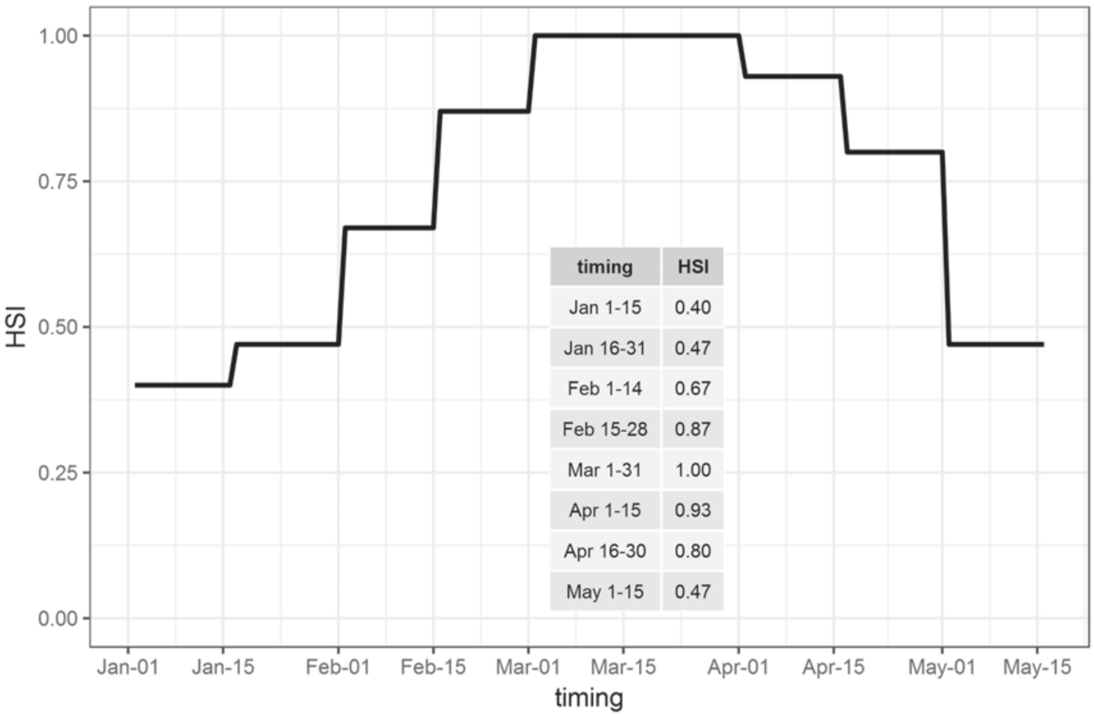


Figure B-5. Suitability of floodplain inundation timing for spawning and rearing splittail based on existing literature and adopted from Suddeth (2014).

CONCLUSIONS

The physical habitat requirements for splittail summarized here provide the basis for the habitat suitability indices used in hydrospatial analysis of Cosumnes River floodplain inundation patterns. For reasons outlined here, the splittail is a natural choice as a focal species to illustrate how hydrospatial metrics can be used to describe and quantify habitat availability. Splittail and salmon are the two species commonly targeted in floodplain restoration efforts. Though much attention is focused on salmon, which use floodplains as rearing habitat for juveniles on their migration to the ocean, splittail have been recognized as the most floodplain dependent species in the Central Valley (Sommer et al., 2001). Moyle (2004) summarized general restoration recommendations to better support splittail: 1) early season flooding, 2) complete drying of floodplain in the late spring, 3) reduction of permanent water bodies, 4) a mosaic of habitats, 5) high variability but annual flooding. Restoring the variability and complexity of floodplain inundation patterns beneficial for splittail is likely to also support processes that will benefit other native species and generally increase the resilience of native populations and ecosystems.

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APPENDIX C

BIAS CORRECTION AND MODEL COMPARISON FOR COSUMNES RIVER HYDROLOGY

Bias correction and model comparison for Cosumnes River hydrology

INTRODUCTION

Changing hydrology under climate change is challenging California's complex water management already inadequately balancing multiple, often competing human water demands and the need to sustain ecosystems. A region well-known for its variable climate, increasing volatility threatens the current system's capacity to adapt to change. More specifically, it is unclear whether actions taken today to restore freshwater ecosystems will have desired effects or whether other actions may be more beneficial. To better understand potential implications and improve strategies to manage and adapt, assessments of potential changes to the amount and timing of water availability and associated water management and ecosystem impacts are essential.

Rising temperatures as well as changing precipitation patterns have direct implications for irrigation, drinking water, flood protection, hydropower, recreation, and ecosystem support. By mid- to late-century, annual average temperatures are expected to increase several degrees C, with most of the warming occurring during the summer months (Cayan et al., 2008; Hayhoe et al., 2004; Pierce et al., 2013). Precipitation trends are less clear, though models indicate decreasing precipitation overall, particularly in southern California. In northern California, seasonal shifts are likely to be pronounced, with more precipitation occurring in the winter months and less occurring in the spring months. These seasonal shifts, coupled with increasing temperature, indicate more precipitation will be falling as rain and snowmelt will occur earlier in the season (Knowles & Cayan, 2002; Maurer, 2007; Miller et al., 2003). Increasing variability is also likely, with increases to floods and droughts, and rapid swings between wet and dry periods (Diffenbaugh et al., 2015; Swain et al., 2018). For example, Das et al. (2013) found a 16-model ensemble suggests larger floods, with 50-year return period floods increasing by 30-90%.

Regional and local climate change impact assessments rest on an array of products derived from multiple scenarios of future climate generated by the most recent global climate models (GCMs). They are best performed using multi-model ensembles to capture the range of model projections and reduce the effects of natural internal climate variability (Brekke et al., 2008; Pierce et al., 2009). Prior to regional- or local-scale applications, systematic errors in the data are removed in a process called bias correction, and the data are downscaled to a spatial resolution that accounts for topography and other effects at scales finer than the GCMs. The non-linear hydrologic responses to precipitation changes make hydrologic projections particularly sensitive to these biases. Further, many impact studies require hydrologic variables that are not part of the meteorological variables available from GCMs. These are obtained through the Variable Infiltration Capacity (VIC) land surface model (Liang et al., 1994). Studies examining hydrologic impacts of climate change, including the research supported by the work described here, often require daily flow time series for rivers, requiring additional effort to route the VIC surface runoff.

This document describes the process developed to bias correct routed VIC (RVIC) daily flow time series for each of 10 downscaled and bias-corrected GCMs, for application to hydrospatial analysis of lower Cosumnes River floodplain inundation patterns under climate change. The VIC model has not been well-calibrated for California's west-slope Sierra Nevada rivers, producing flow estimates that are systematically biased such that high flows are generally overestimated while low flows are underestimated (Knowles & Lucas, 2015). To address this issue, bias correction using quantile mapping is applied to the RVIC data using a reference time series based on observed flows at the Cosumnes River Michigan Bar streamgage (MHB, #11335000; U.S. Geological Survey, 2017).

HYDROLOGIC DATASETS

This work uses recently available daily flow projections for major rivers flowing into California's Central Valley, which were derived from the Intergovernmental Panel on Climate Change (IPCC) Coupled Model Intercomparison Project, Phase 5 (CMIP5) multi-model archive (Taylor et al., 2011) and were produced to support California's Fourth Climate Change Assessment (Pierce et al., 2016). These model outputs are currently being used to study water management and ecological impacts of climate change in the Central Valley and Sacramento-San Joaquin Delta.

GCM projections and VIC simulations

The climate change projection datasets used in this work are derived from GCM output of the CMIP5 archive. Of the larger CMIP5 ensemble of 32 models with daily data, 10 GCMs were selected by the California Department of Water Resources (DWR) Climate Change Technical Advisory Group (CCTAG) as most capable of capturing processes relevant to California water resources, and are to be used in regional or local California climate change impact studies (Table C-1; DWR CCTAG 2015). These 10 have been systematically subset further using a variety of metrics, for studies where a complete model ensemble is too resource intensive or otherwise impractical (Kravitz, 2017; Pierce et al., 2016). These selected four are: HadGEM2-ES ("warm/dry"), CNRM-CM5 ("cool/wet"), CanESM2 ("middle"), and MIROC5 ("diversity"; Cayan et al., 2018; Kravitz, 2017). For the 10 selected GCMs, researchers at USGS and Scripps Institution of Oceanography, UCSD produced bias corrected and downscaled output for the

Table C-1. The 10 global climate models and the institutions that developed them, that were selected by California's Department of Water Resources Climate Change Technical Advisory Group for their ability to represent California's climate well. Bolded text indicates the four models selected as a representative subset.

Global Climate Model	Institution
ACCESS1-0	CSIRO (Commonwealth Scientific and Industrial Research Organization), Australia, and Bureau of Meteorology, Australia
CCSM4	The National Science Foundation, The Department of Energy, and the National Center for Atmospheric Research, United States
CESM1-BGC	The National Science Foundation, The Department of Energy, and the National Center for Atmospheric Research, United States
CMCC-CMS	Centro Euro-Mediterraneo per I Cambiamenti, Italy
CNRM-CM5	CNRM (Centre National de Recherches Meteorologiques, Meteo-France, Toulouse,-France) and CERFACS (Centre Europeen de Recherches et de Formation Avancee en Calcul Scientifique, Toulouse, France
CanESM2	CCCma (Canadian Centre for Climate Modelling and Analysis, Victoria, BC, Canada)
GFDL-CM3	NOAA Geophysical Fluid Dynamics Laboratory, Princeton, N.J., USA
HadGEM2-CC	Met Office Hadley Centre, Fitzroy Road, Exeter, Devon, EX1 3PB, UK
HadGEM2-ES	Met Office Hadley Centre, Fitzroy Road, Exeter, Devon, EX1 3PB, UK
MIROC5	JAMSTEC (Japan Agency for Marine-Earth Science and Technology, Kanagawa, Japan), AORI (Atmosphere and Ocean Research Institute, The University of Tokyo, Chiba, Japan), and NIES (National Institute for Environmental Studies, Ibaraki, Japan)

Sacramento-San Joaquin Basin (SSJB) at a gridded 1/16° resolution (Cayan et al., 2018). This was done using newly developed methods using the localized constructed analogs method (LOCA), which together better preserve daily extremes and variability and seasonality compared to other methods (Pierce et al., 2015; Pierce et al., 2014). The bias correction and downscaling procedure relied on a gridded historical observed dataset of temperature and precipitation produced by Livneh et al. (2015). These observation-based daily data, including daily precipitation and maximum and minimum temperature, were developed specifically for the U.S. Bureau of Reclamation for purposes of downscaling GCM output. The data for each GCM spans 1950-2100, including the historical period (1950-2005), and two Representative Concentration Pathway (RCP) future projections ("medium" RCP 4.5 and "high" RCP 8.5; 2006-2100).

The bias corrected and downscaled GCM climatology, along with the historical Livneh dataset, was used by USGS researchers to force the VIC land surface model at a daily time step, which produces gridded variables including rain, snow, evaporation, and runoff (Cayan et al., 2018). The VIC model is a distributed, physically-based hydrologic model that balances energy and water budgets and allows for soil moisture capacity distribution at the subgrid scale, a factor important for areas with complex terrain (Das et al., 2011; Maurer, 2007). It simulates surface and shallow sub-surface hydrologic processes, including vegetation impacts, simplified baseflow (groundwater and groundwater-surface water interactions are not included), and snow processes. The VIC model is widely applied for climate studies in the western United States (e.g., Cayan et al., 2010; Dettinger, 2011; Hamlet & Lettenmaier, 2007; Maurer et al., 2002; Vano & Lettenmaier, 2013), but as with any model, the process of using the downscaled GCM variables to force the VIC model adds a layer of uncertainty to results. The gridded VIC hydrologic output was subsequently routed, via the RVIC routing model, to various points within the SSJB channel network, producing unimpaired daily streamflow estimates (Cayan et al., 2018; Knowles et al., 2018; Lohmann et al., 1996). One of these pour points is the Cosumnes River (38.28, -121.4). It is this RVIC daily time series output that has been made available for use here.

Historical observed hydrology

For the historical observed data set, which is required for bias correction, Cosumnes River daily flow observations at the USGS MHB gage were used for the corresponding GCM historical period 1950-2005. As the Cosumnes River is largely unregulated, these data represent "unimpaired" flows and are thus comparable to the RVIC flows. The importance of bias correcting output from the VIC model can be seen comparing the Livneh dataset against the MHB gage data. Based on this analysis, it is clear that the Livneh routed flows substantially overpredict the MHB flows on an annual basis, and in general overestimate higher flows and underestimate lower flows. On an annual basis, the flow volumes are associated with a bias of approximately 46%, with an R^2 of 0.97, and a Nash-Sutcliffe efficiency of 0.48 (Figure C-1). At a monthly scale, winter high flow months are overpredicted overall, while late spring and early summer lower flow months are under-predicted overall (Figure C-2; Table C-2). This high bias for higher flows and low bias for lower flows is also observed in comparing the daily flow duration curves for the two datasets (Figure C-3). This bias is consistent with the VIC model bias observed by others (Knowles & Lucas, 2015). Some of the difference may also be attributable to the fact the RVIC flows are routed to a point on the Cosumnes (38.28, -121.4) that is downstream (>45 km) of the MHB gage. Using the MHB daily flow record for bias correction of the RVIC hydrology is particularly relevant here considering that Cosumnes floodplain hydrodynamic modeling has been performed using this dataset.

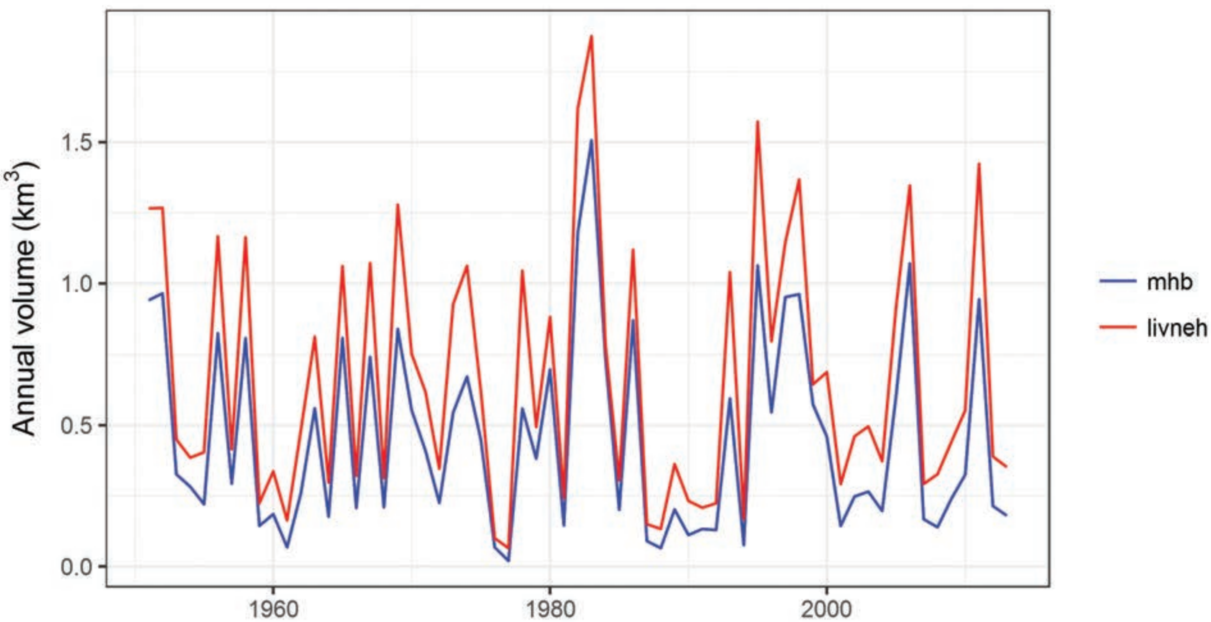

Figure C-1. Observed and modeled historical flows. Annual volume for observed historical flows at USGS Michigan Bar gage (mhb; #11335000) and for the routed VIC flows (livneh) based on the Livneh et al. (2015) dataset.

METHODS

Hydrologic modeling results often reveal systematic errors, or bias, in comparison to observations. Similar to bias correction of GCM output (temperature and precipitation), bias correction of hydrologic model output using these data has also been applied (Miller et al., 2012; Nijssen & Chegwidden, 2017; Snover et al., 2003). This is especially important for more localized studies where errors in aspects of the time series can be particularly apparent, especially when the results are compared to the historical observed time series. The bias correction process allows for more appropriate comparisons between historical conditions and future projections.

To perform the bias correction of the daily flow projections, quantile mapping was used, which is one of the most commonly used bias correction techniques (Maurer, 2007; Maurer et al., 2013; Snover et al., 2003; Thrasher et al., 2012; Wood et al., 2002). This method addresses variance, where the model distribution variance is brought into closer alignment with the observed variance. To transform the distributions, modeled values in the overlapping historical period are mapped to the corresponding observed values by matching their associated quantiles, and this relationship is then applied to the future flow time series. An important assumption in this approach is that the structure of the bias of the model in the historical period persists in the future time period. Previous research has demonstrated that quantile mapping for GCM precipitation can potentially alter some trends in the GCM output, which has the potential to be of issue for flows as well, though this has not yet been assessed (Maurer & Pierce, 2014).

The quantile mapping procedure was implemented within *R*, using the *biasCorrect* function of the *hyfo* package (R Core Team, 2013; Xu, 2017). The *biasCorrect* function takes input time series of observed (here, MHB observed daily flows dataset), hindcast (historical period of GCM RVIC datasets), and forecast data (full period of GCM RVIC datasets), where the time period of observation and hindcast

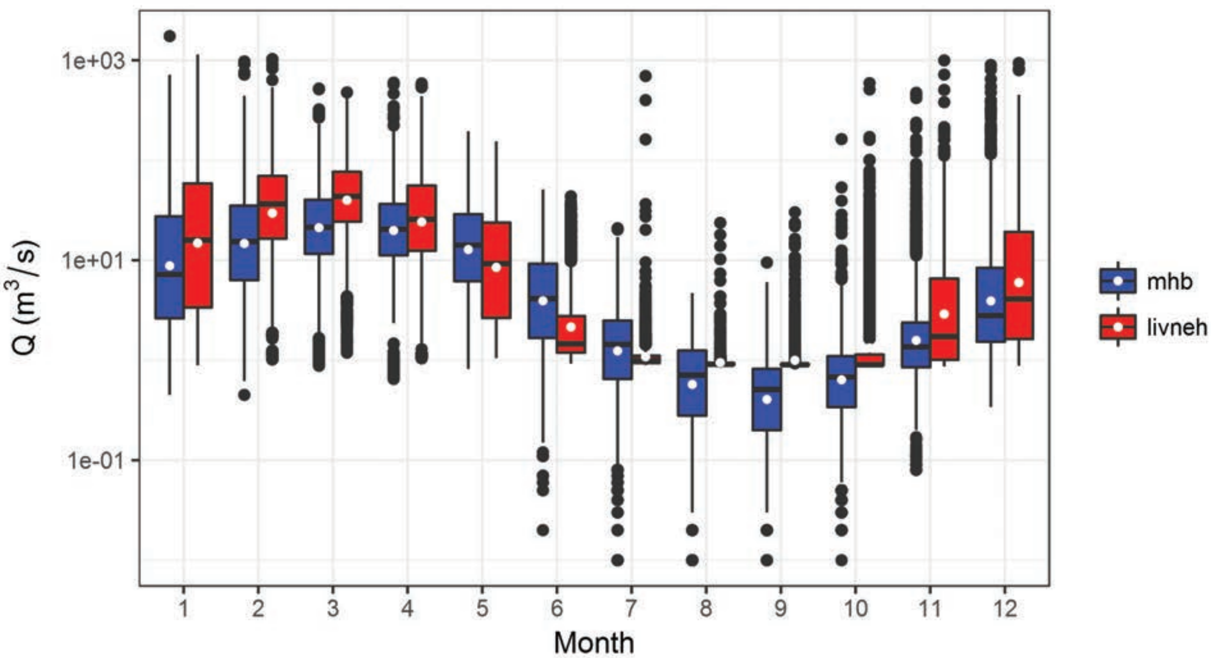

Figure C-2. Daily flow comparison by month between observed historical flows at USGS Michigan Bar gage (mhb; #11335000) and for the routed VIC flows (livneh) based on the Livneh et al. (2015) dataset. Boxes extend from the first to third quartiles and the whiskers extend to 1.5 times the inter-quartile range. The mean is shown in white dots.

data match. The function returns the transformed forecast data through a variety of methods, including quantile mapping. This function also provides the option to bound the modified forecast data by the range of observed data, which was applied here. When simulated values are outside the range of the historical period, they are either mapped to a value within the range of observed data or the quantile distributions can be extended and values mapped from this theoretical distribution. In some cases, unbounded mapping can lead to unrealistic values where the extreme values are in error. Quantile mapping was applied using a 31-day moving window, following Thrasher et al. (2012). Cumulative distribution functions representing each day of the year were constructed using the historical GCM simulated and historical observed values from 15 days prior to 15 days after the day in question. The bias correction transfer function for each day and model was derived for the historical (1950-2005) period of available data and then applied to the full 1950-2099 period.

Table C-2. Goodness-of-fit statistics comparing daily flows by month for observed historical flows at USGS Michigan Bar streamflow gage (#11335000) and the routed VIC flows based on the Livneh et al. (2015) dataset.

Month	PBIAS (%)	R^2	NSE
October	215.3	0.65	-14.49
November	112.1	0.74	0.06
December	43.0	0.84	0.79
January	54.1	0.79	0.71
February	72.9	0.85	0.66
March	69.2	0.81	0.38
April	40.7	0.84	0.52
May	-15.8	0.81	0.76
June	-47.3	0.73	0.55
July	-12.5	0.04	-54.40
August	22.9	0.01	-1.13
September	106.5	0.12	-6.82

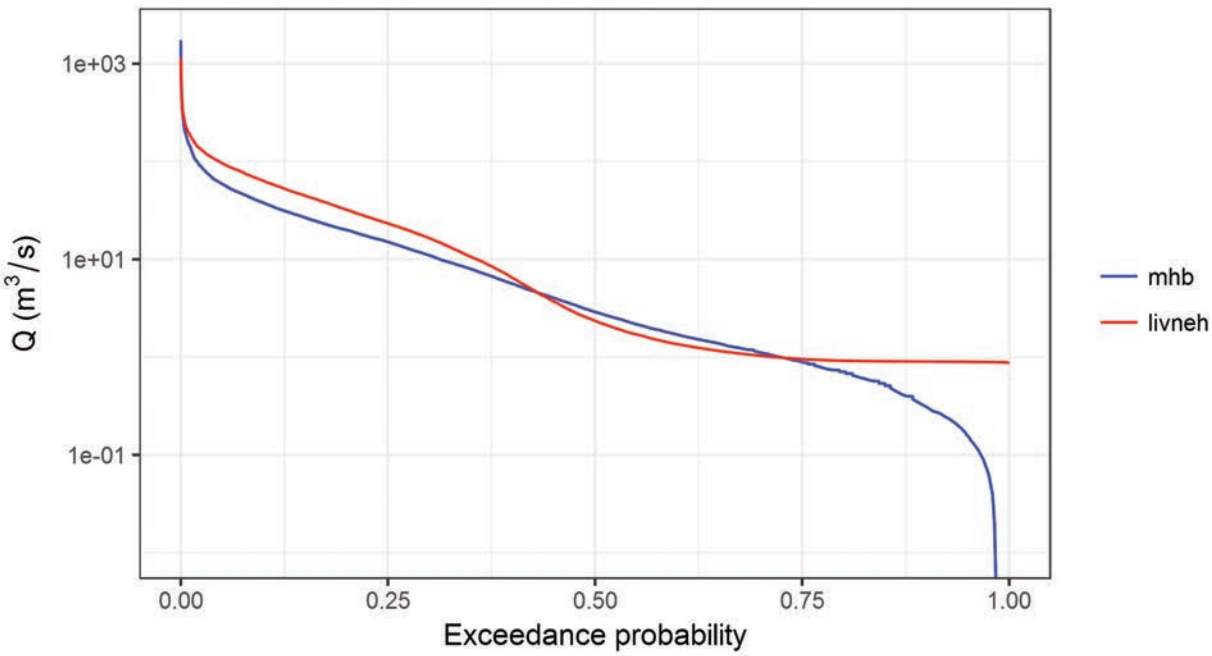

Figure C-3. Flow duration curves for observed and routed modeled historical flows. Data are from observed historical flows at USGS Michigan Bar gage (mhb; #11335000) and the routed VIC flows (livneh) based on the Livneh et al. (2015) dataset.

RESULTS AND DISCUSSION

Bias correction

The bias correction procedure results in substantial reduction in RVIC GCM model output biases during the historical period. Figure C-4 shows the matching observed USGS MHB gage data and bias corrected GCM RVIC data distributions by month relative to the original GCM RVIC data. All models in the wet season high flow months show a bias toward higher median flows in the original GCM data but lower median flows in the late spring and early summer when lower flows predominate. These biases are also apparent in flow duration curve comparisons (Figure C-5). Simple scatterplots of the sorted observed MHB flows against the sorted original and bias corrected GCM data illustrate the overall overestimation of flows, but also reveal that while overestimation occurs for high flows, exceptionally high flows (>500 m^3/s) are underestimated in many models (Figure C-6). Differences in the models in this respect lead to differences in how well linear trend lines align with the 1:1 line. Overall, these plots also illustrate how these high flow values are brought into closer alignment with the observed data through the bias correction process.

To assess statistical significance of the differences in these datasets, Wilcoxon signed rank tests were conducted for each model and month for the original and bias corrected GCM data in comparison to the observed USGS MHB gage data. At a significance level of 0.05, the null hypothesis that the two datasets have the same distributions was not accepted in any of the 120 model and month combinations with the original data, but was accepted for all 120 model and month combinations with the bias corrected data.

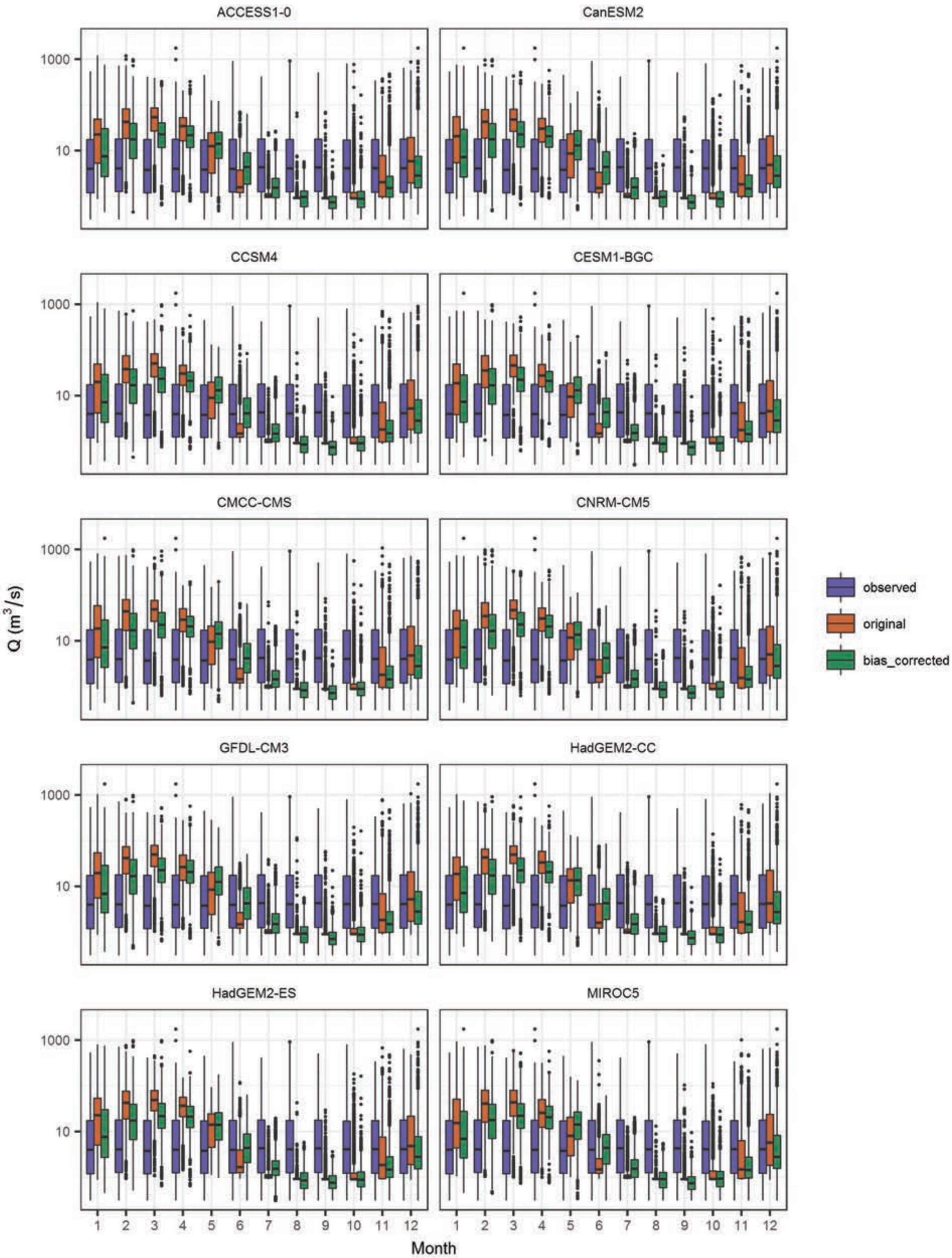

Figure C-4. Historical period observed and bias corrected comparison summarized by month. Daily data include the historical period (1950-2005) at USGS Michigan Bar gage (#11335000), the original GCM output, and the bias corrected GCM output.

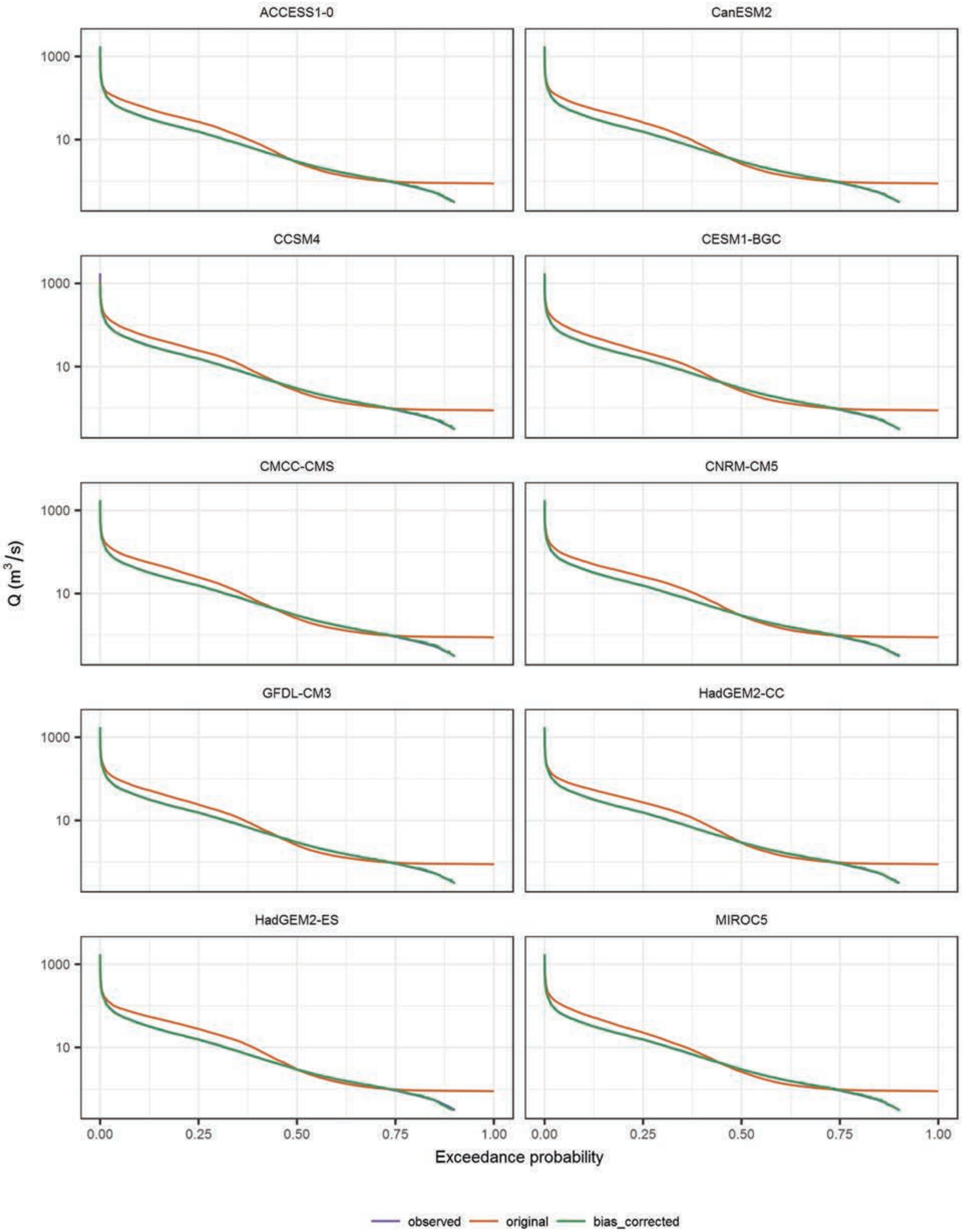

Figure C-5. Flow duration curves for the historical period observed and bias corrected daily flows. Daily data shown are observed daily historical flows at USGS Michigan Bar gage (#11335000) against the GCM original and bias corrected daily RVIC flows. High flows are consistently overestimated for higher flows and somewhat underestimated for lower flows.

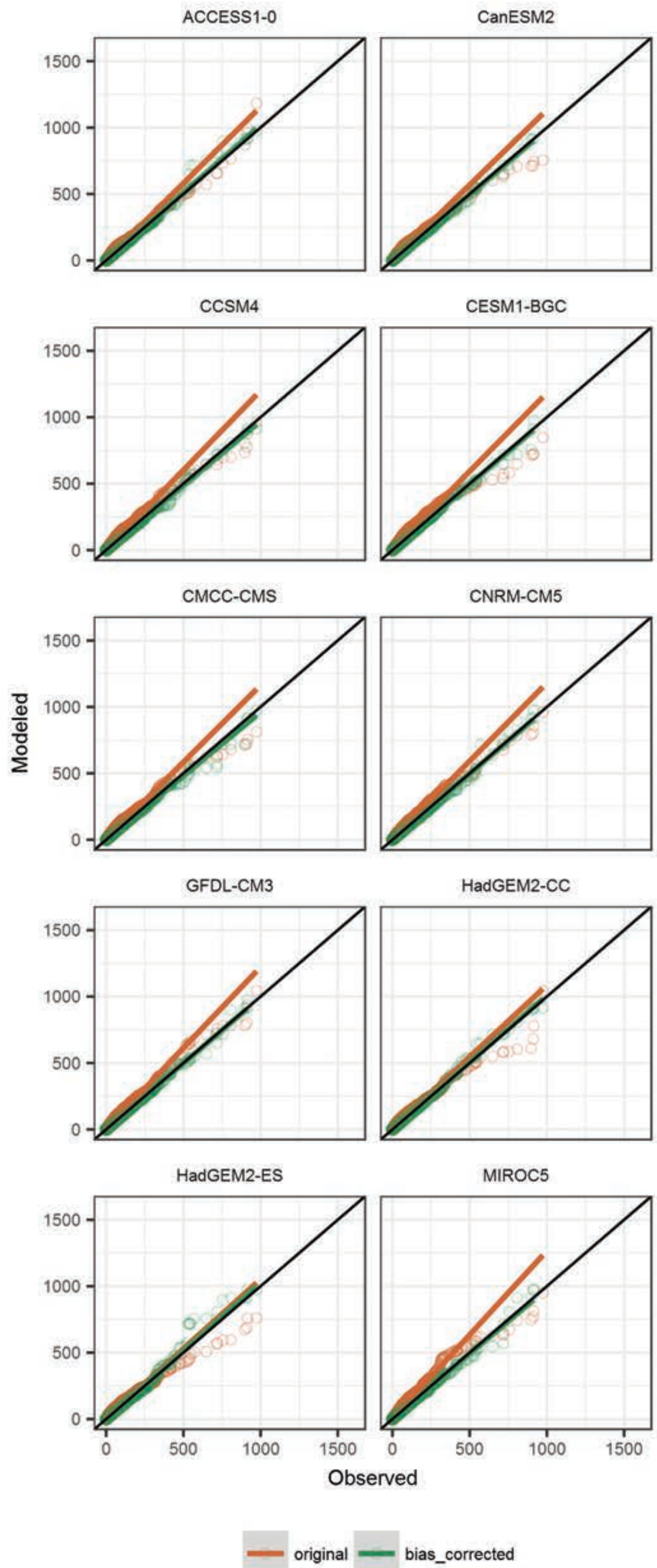

Figure C-6. Observed versus modeled daily flows. Scatterplots of observed Livneh daily flows against the GCM original and bias corrected daily flows. Linear trendlines are fit to the data and the 1:1 line is shown in black.

Model comparison

A limited exploration of the bias corrected GCM RVIC flows reveals declining median flows for most models in both the near (2010-2039) and far (2070-2099) future compared to the historical period (1950-1979; Figure C-7). Of the 10 models and two RCPs each, only CCSM4 RCP4.5 and HadGEM2-CC RCP4.5 show median flow increases in the far future over the near future median flows. Changes in seasonality and further differences between models and RCPs are apparent when comparison is performed by month (Figure C-8). For winter months, many models show relatively little change or some increases in both the near and far future periods. However, for later spring and early summer months, models consistently show declining median flows for both periods, some of which are quite pronounced. Comparing the flow duration curves for each of these periods shows a general pattern of declining flows for exceedance probabilities greater than 0.10, while some models suggest lower exceedance probability flows are associated with higher flows in the future periods (e.g., CanESM2, CESM1-BGC, CNRM-CM5, HadGEM2-ES; Figure C-9). As would be expected, many of these differences are more pronounced for the higher emissions scenario (RCP 8.5).

CONCLUSIONS

The application of this procedure produced adjusted Cosumnes River GCM RVIC daily flow projections for the period 1950-2099 that are statistically consistent with the observed dataset during the historical period of 1950-2005. The removal of the systematic bias in the daily flow projections allows for more appropriate evaluation of subsequent impact analysis, including the lower Cosumnes River floodplain hydrospatial analysis for which this bias correction was performed. Comparison of historical

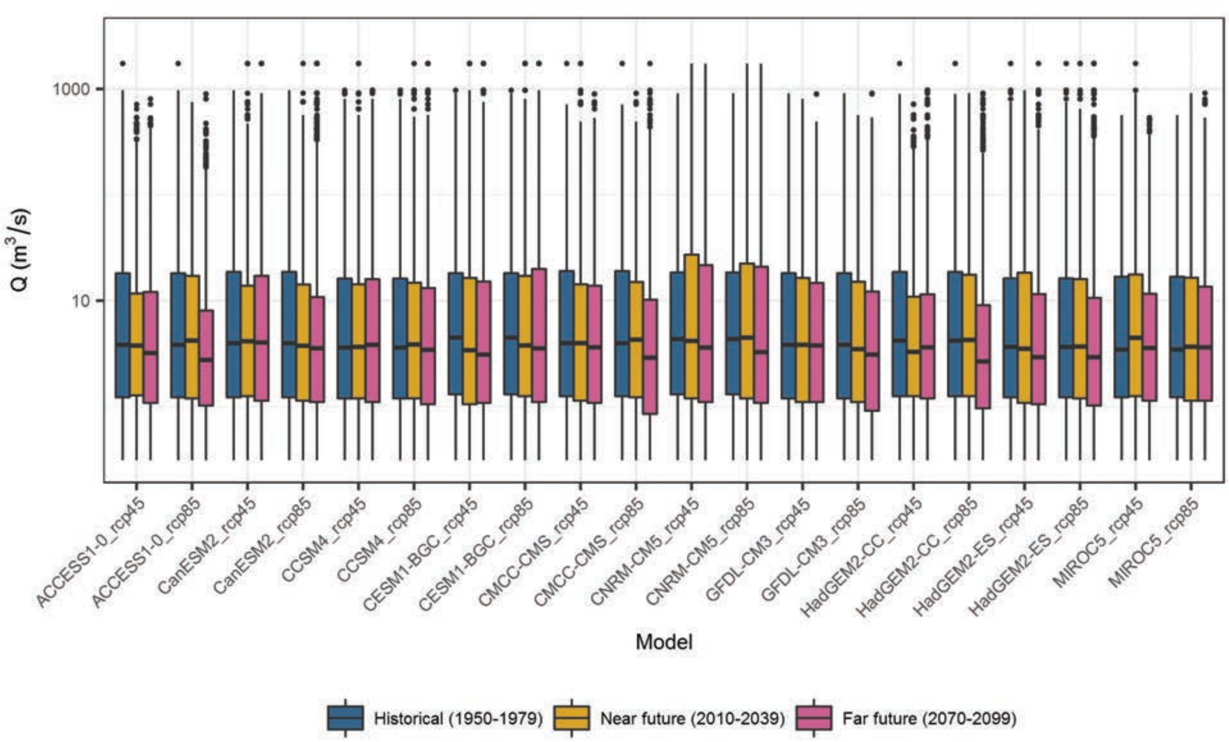

Figure C-7. Daily flow summaries of bias corrected flows for each GCM and RCP combination, for three time periods.

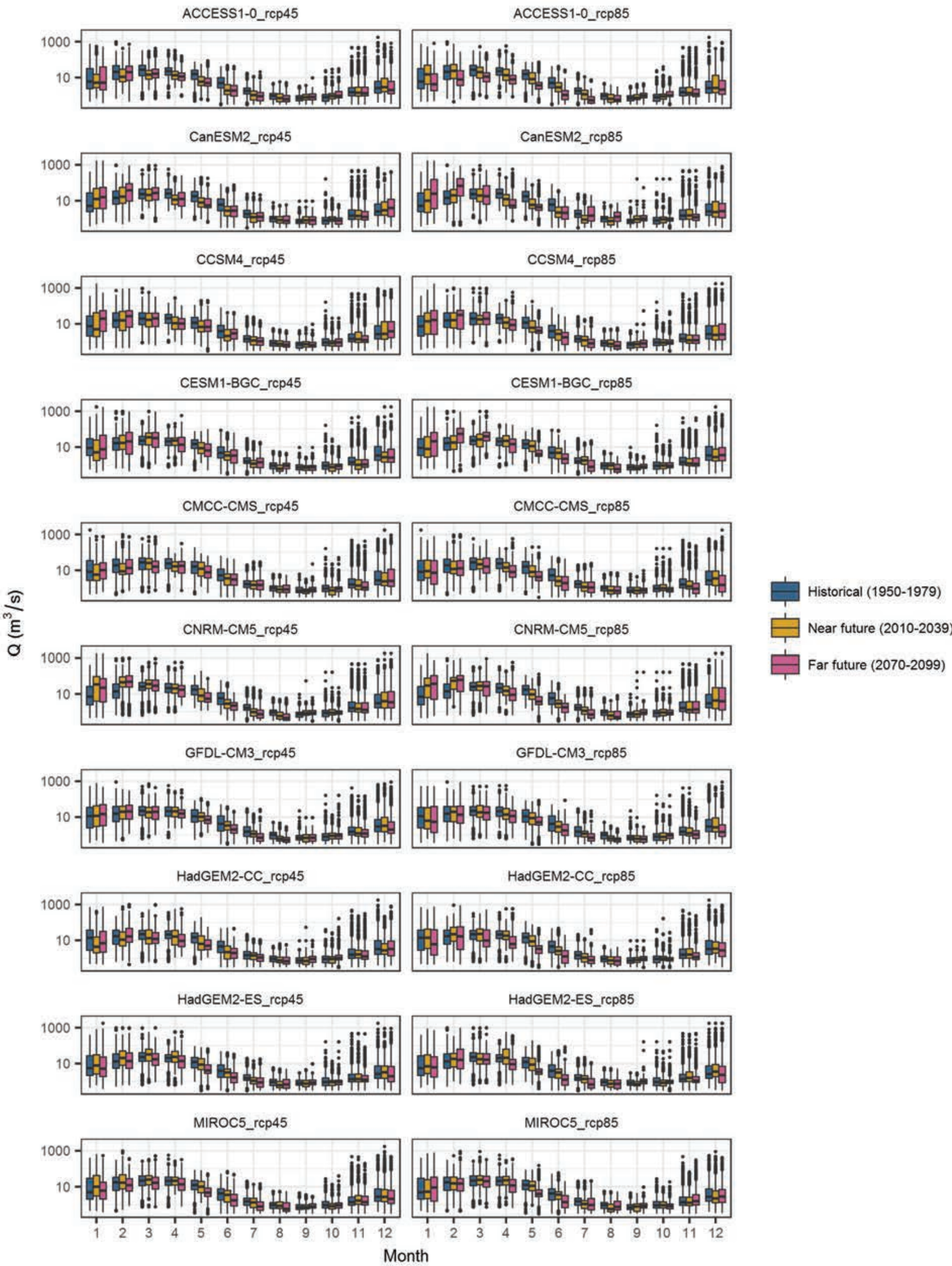

Figure C-8. Daily flow summaries by month of bias corrected flows for each GCM and RCP combination, for three time periods.

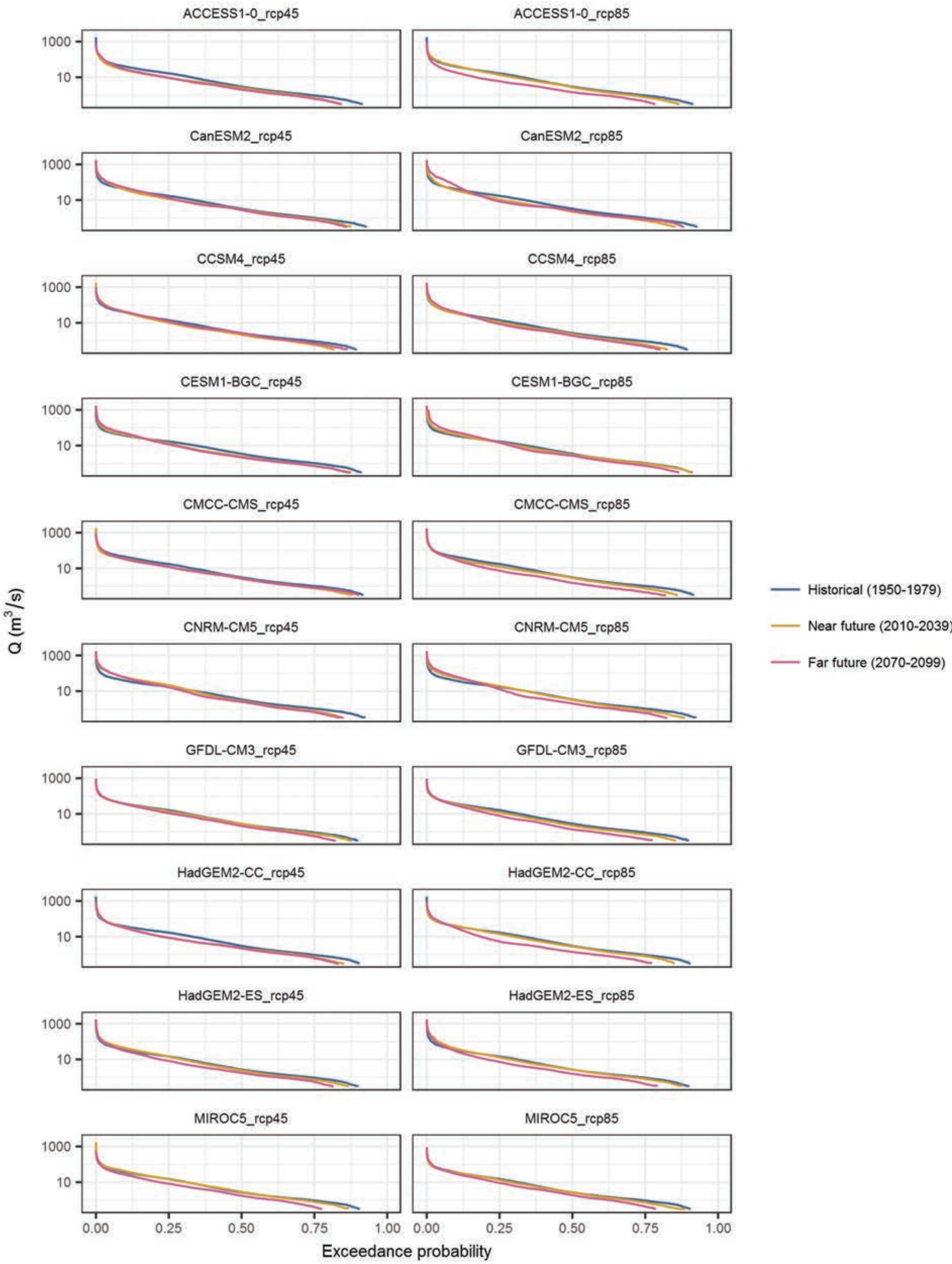

Figure C-9. Flow duration curves of bias corrected flows for each GCM and RCP combination, for three time periods.

and future flows from the 10-model ensemble are in general agreement with other studies, suggesting shifts in seasonality with model-dependent changes to winter flows and consistently declining flows in the later spring months. The application of this method to bias correct GCM RVIC daily flow projections and preliminary examination of the resulting datasets support the research goals to evaluate potential climate change impacts to lower Cosumnes River floodplain inundation patterns in space and time.

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