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Abstract

In the Preface to volume 1, we discuss the development of riparian ecology as one of the newest of ecological fields that gained significant momentum in the 1950s and 1960s as part of the general "riparian movement" in the United States. The field expanded rapidly throughout the latter half of the 1900s. Volume 2 involves more than two dozen authors—most with decades of experience—who expand upon riparian and other topics introduced in volume 1. Two important recent developments are global climate change and impacts of introduced tamarisk leaf beetles (*Diorhabda* spp.) in the American West. Other chapters in volume 2 that provide current information evaluate the losses of riparian habitat, including "extirpation" of a large number of mesquite bosques (woodlands) in the Southwest; the restoration of riparian ecosystems damaged by anthropogenic activities; the importance of a watershed; and the importance of riparian ecosystems to recreation. The combination of volumes 1 and 2 examines the evolving understanding of scientific implications and anthropogenic threats to those ecosystems from Euro-American settlement of the region to present.

Keywords: riparian, ecosystem, ecology, riparian processes, riparian losses, restoration, aquatic, arid, semiarid, upland, freshwater, groundwater, hydrology, watershed, tamarisk, tamarisk leaf beetles (*Diorhabda* spp.)

Cover: A comparison of the effects of tamarisk leaf beetle defoliation. Top photo 2005, before leaf beetle defoliation. Photo by Steven W. Carothers. Bottom 2017, after defoliation. Photo by Robb Irwin Eidemiller. Colorado River, mile 7.5; Glen Canyon National Recreation Area, Arizona.

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Foreword

I was honored when Roy Johnson and Steve Carothers asked me to write a Foreword for volume 2 of *Riparian Research and Management: Past, Present, Future,* but then realism hit me. When I read this book, I realized how much time, experience, and knowledge had passed me by. But, in my memories I could still envision the day in 1968 when Doug Morrison called me from the Flagstaff office of the Coconino National Forest to request funds for two “birders” to make a census of breeding birds along a section of the Verde River. The two “birders” were ornithologists Steve Carothers, Museum of Northern Arizona, and Roy Johnson, Prescott College.

Salt River Project, a Phoenix-based water management company, had been cutting down mature cottonwood trees along the Verde River in central Arizona. Salt River Project was determined to rid Arizona of the “phreatophytes that were robbing the people of Arizona of their water,” but when they hit the Coconino National Forest boundary, they found out there were other resources to be considered. Our search for money to do the survey came up with a measly $528. I called Doug back knowing no biologist worth his salt would agree to such a meager sum! I don’t know if it was Doug’s pleading, or Steve’s and Roy’s knowledge that this inventory was far more important than money, but they agreed to make this survey. Steve and Roy found a large number of species of birds nesting in the cottonwood forests of the Verde Valley. Even more exciting, they also found the highest population densities of breeding birds ever reported for North America! These results changed the minds of a great number of people about the vital importance of riparian habitat, including mine.

In 1977, I moved from my position as Wildlife, Fish, and Endangered Species Director for the Southwest Region of the United States Forest Service in Albuquerque, New Mexico, to the same position for the Chief of the Forest Service in Washington, DC Early that year, Roy Johnson, Steve Carothers, and I planned a riparian conference for the western United States that was held on July 9 in downtown Tucson, Arizona. We were unaware of concurrent planning for another riparian conference on “Riparian Forests in California,” held 2 months earlier, on May 14, 1977, at the University of California at Davis. The proceedings from those two conferences were the first riparian proceedings published for the United States. This current two volume publication is an example of the U.S. Forest Service’s continuing commitment to research in riparian ecosystems.

While I was reviewing volume 2, it was interesting to see that the 1970s were the years that riparian studies, conferences, and publications happened in many areas of the country. The symposia in California and Arizona may have influenced this, but a lot of the awareness just seemed to happen. The other shocking thing to me as I read volume 2 was that this manuscript was sometimes difficult to understand for biologists of my era. Such rapid advances have been made in riparian ecology since my retirement that I spent more time reading the dictionary than I did the manuscript of some chapters; but the great news is that today and tomorrow’s biologists will find volume 2 of *Riparian Research and Management: Past, Present, Future* extremely helpful.

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Preface for Volume 2

This is the second of two volumes of scientific papers on semi-arid West and wet Pacific Northwest riparian ecosystems. Volume 1 entailed a primarily historical approach to the evolution of the science of riparian ecology and concerns and conservation needs for and on riparian habitats. It also reviewed past effects of anthropomorphic impacts to these highly productive ecosystems (chapters 1, 2, and 3). Another chapter discussed the turn-of-the-century invasion of the nonnative tree/shrub *Tamarix* spp. (tamarisk or salt cedar) and the past several decades of mostly failed attempts at its suppression and habitat restoration with native woody species (chapter 4).

Following this was a detailed description of biocontrol efforts by the Department of Agriculture's Animal and Plant Health Inspection Services (APHIS) on tens of thousands of acres where this exotic plant has invaded throughout the western United States over the past 100-plus years (chapter 5). The success of biocontrol, beginning in earnest in 2001, has surprised everyone, including program proponents with how rapidly introduced Asian tamarisk leaf beetles (*Diorhabda* spp.) have spread and defoliated their target. Results of biocontrol reveal a surprising management conundrum in the plight of an endangered songbird, the southwestern willow flycatcher (*Empidonax traillii extimus*), and its apparent reliance on nonnative *Tamarix* spp. for nesting habitat. Without this "invasive" tree species, there is little suitable nesting habitat remaining throughout much of the flycatcher's range.

As *Tamarix* declines under the influence of biocontrol, it has become obvious that original assumptions that native plant species would eventually replace the invader have not come to pass. Most areas, but not all, where *Tamarix* has been defoliated revegetate with a mixture of native and nonnative forbs and grasses, poor wildlife habitat replacement for the declining invader (chapter 4). Chapters 6 and 7 in volume 1 focused on beavers, their near-extirpation by early Euro-Americans, and how reintroduction of beavers can have many positive benefits on riparian health.

Volume 1 also provided a review of the extensive watershed alterations by dams and diversions in Arizona (chapter 8), reminding us that one of the more common and insidious impacts associated with dams is the inundation of riparian habitat behind the dam and change in the natural hydrograph, or seasonal patterns of flow, temperature, and sediment accumulation and distribution downstream. Chapter 9, using spatially extensive matched photo sets, provided the details of changes in riparian vegetation in canyon bound reaches of the Colorado River below these dams. The penultimate chapter, (chapter 10) discussed the breeding water birds of the Mexican portion of the Colorado River delta and the insidious effects that the last 100 years of reduction in freshwater flows have had on delta habitat.

The final chapter to volume 1 (chapter 11) documented the species richness and available data on population densities of land vertebrates in mesquite (*Prosopis* spp.) bosque habitats compared to other riparian habitat types. The appendices to volume 1 also provided a chronology of southwestern and northwestern United States riparian symposia and the riparian ecologists and other "pioneers" who made those symposia possible.

In volume 2, our first chapter explores conclusions of a widely distributed book on southwestern riparian systems, the *Ribbon of Green* (2007) by Robert Webb, Stanley Leake, and Raymond Turner. By analyzing changes in streams and rivers through time by photographic comparisons, this book reached a conclusion that more riparian habitat exists today than was present in the past. This chapter re-assesses riparian status, considering the proliferation of dams that have inundated thousands of miles of riparian habitat, diversions of surface water for agriculture, falling water tables, wide-spread invasion of nonnative species, and ever-increasing demands on water resources. Although we recognize the value of the book's contributions on the status of southwestern streams and rivers in general, the results of our assessment reveal that the amount of riparian habitat present today has been appreciably reduced compared to earlier times, and we explain why our conclusion differs from Webb et al. (2007).

Throughout the remainder of volume 2, we apply a contemporary view of present-day riparian habitat by discussing the need to understand better the short- and long-term effects to wildlife habitat, riparian vegetation succession, and ecosystem recovery under the vagaries of climate change as well as remarkably successful biocontrol as was discussed in detail in chapter 5 of volume 1. The chapter discussing defoliation of Grand Canyon Colorado River corridor *Tamarix* and its potential influence on native vertebrates illuminates ongoing and ever-changing challenges to which native wildlife is exposed (chapter 2). Chapter 3 continues the discussion from volume 1 on the disappearance of mesquite bosques and suggests habitat restoration where possible with mesquite as a replacement for *Tamarix*. One of the more creative approaches to riparian habitat maintenance and restoration is discussed in the chapter on the recent development of experimental gardens in arid lands and attempts to identify and deploy plant genotypes that are likely to survive vagaries of climate change. The combined effects of climate change undoubtedly will have an impact on our ability to maintain and restore riparian ecosystems, even to the level of defining specific ecotypes of plants that can be used in each specific location (chapter 4).

Additional volume 2 chapters detail and discuss complexities of the watershed continuum model and how that model may provide a better understanding and approach to habitat restoration (chapter 5). Chapter 6 describes a wealth of opportunities in the Anthropocene (the period during which human activity has been a dominant influence on the environment) for riparian restoration using agricultural runoff, unlined irrigation canals, waste water outfalls and orchards, that are often a part of human communities. Threats to riparian communities are not just limited to arid land systems, as are detailed in the two chapters describing unique challenges in California streams and rivers (chapters 7 and 8).

The final two chapters in volume 2 discuss riparian ecosystem function where and how recreation and habitat restoration can be conflict-free. A compelling argument is made that as we engineer physical or biological habitat, we are also engineering recreation habitat, either by default or by design (chapters 9 and 10).

In both volumes of this publication, various threats to riparian ecosystems have been mentioned and discussed. From our viewpoints, threats are those processes, natural and anthropogenic, that disrupt or interfere with dynamic processes of native biotic and abiotic constituents functioning together in a healthy riparian ecosystem. Our concepts of threats to riparian habitats are value-based and biased by our professional and personal experiences as biologists and ecologists. With that viewpoint, it becomes obvious that there are many threats and most of these are interacting with each other simultaneously or sequentially. Our understanding of losses and gains in habitat through time are difficult to quantify because of different viewpoints and inadequate historical records. We know that things are not what they were in riparian areas before the coming of modern civilization, but accurate quantification of losses and threats is impossible, as is a complete return to those former conditions. Throughout both volumes, an emphasis on the natural hydrograph of healthy riparian systems is a common theme. The altered hydrograph usually results in a loss of native woody riparian vegetation that is often replaced by nonnative invasive species, e.g., *Tamarix* spp. or *Elaeagnus angustifolia* (Russian olive).

It has become clear that without a natural flow regime, most native riparian vegetation cannot become reestablished as a viable community and, in its absence, wildlife productivity along our rivers and streams is seriously compromised. Invasive species are often pointed to as threats, but as is evidenced by the volume 1 chapter on unintended consequences of biocontrol and volume 2 chapters on Grand Canyon habitats, sometimes invasive species, especially *Tamarix* spp., can have multiple effects— “good” and “bad” in the context of riparian habitat and wildlife it supports.

Collectively we are learning how to restore formerly disturbed riparian ecosystems. Many people, including most of those involved in the production of these two volumes, had various roles as “pioneers” in developing our current understanding of the functions and values of healthy native riparian areas. Our culture appears to have learned that riparian woodlands are not evil phreatophytes, needing control by removing all vegetation so that more water might be available for human usage. But these woodlands are valuable natural resources to be protected. Management orientation of riparian resources has shifted from control to protection and appreciation, throughout the Southwest, Pacific Northwest, and Mexico. Remnants of riparian areas are now being protected, and new riparian areas have been created, either intentionally or inadvertently, and then managed for conservation and aesthetic objectives and recreational uses. The collective consciousness of American culture appears to be moving toward a broader understanding of the role of riparian areas in the context of the watershed continuum and the world in which most of us want to live.

Clearly, one of the greatest challenges riparian ecologists will face for the next several decades will be in evaluating impacts of *Tamarix* biocontrol and developing prescriptions for habitat restoration and recovery in areas where existing native wildlife resources have been significantly compromised due to unintended and unanticipated results of biocontrol. As documented in this volume, with few exceptions, passive revegetation with wildlife-enhancing native woody species does not occur post-biocontrol. Depending upon the quality of soil and water resources available, most areas where *Tamarix* has been defoliated by the tamarisk leaf beetle will likely require active, and expensive, revegetation efforts to achieve wildlife productivity approaching the levels that invasive *Tamarix* has heretofore accommodated.

We, the authors and editors of these two volumes, hope that this work will serve as a foundation for continuing research, directed management, and societal appreciation of our western riparian resources.

Steven W. Carothers and R. Roy Johnson
Co-Directors
The Western Riparian Project

Acknowledgments

This is the second of a two-volume technical report on riparian habitats in the western United States. The idea for this series of technical reports was an outgrowth of The Western Riparian Project, a project initiated by a group of riparian ecologists and resources managers, all of whom have decades of experience in the ecological aspects of riparian habitat productivity, management, and conservation. Beginning in 2013, the Western Riparian Project is in its seventh year and has benefitted from participation by more than 50 scientists who have collectively spent thousands of uncompensated hours to ensure the success of the Project. Several of us have been involved with riparian ecology since the earliest days of its establishment as a science in the 1960s. These two volumes examine the progress made in western aridland riparian ecology, management, and conservation since those earliest studies. The idea for the beginnings of the project are outlined in the Acknowledgements of volume 1.

Many of the authors of the chapters in this General Technical Report (GTR) have spent decades documenting the importance of conserving and managing riparian habitats due to their disproportionate value to wildlife and people compared to upland habitats. We greatly appreciate the efforts of our reviewers. Professionals that reviewed the entire volume 2 include Dale A. Jones, Patricia M. Woodruff, and Harley G. Shaw. Others that reviewed one or more chapters include Bertin W. Anderson, Kai Caraher, Bruce K. Orr, and Richard Valdez. We thank David L. Hawksworth from the Rocky Mountain Research Station for help in formatting the chapters. Special thanks to Lane Eskew, contracted editor for the Forest Service, who did a brilliant job combing through every word, number, figure, caption, table, etc. and made many suggestions that improved the product.

Space prohibits us from thanking all of the authors, individually, of the various chapters but their affiliations can be found in this GTR's front matter. Their dedicated efforts have greatly contributed to the success of this project. Finally, we thank Dale Jones for writing the insightful Foreword to this volume. Dale is a real riparian pioneer, one of the earliest to recognize the importance of riparian ecosystems and undertake steps for their conservation and management.

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Chapter 1. Understanding Gains and Losses of Riparian Habitat: Interpreting Change, Its Causes and Consequences

Steven W. Carothers and Dorothy A. House

Introduction

One of the most wide-ranging scientific treatments of riparian vegetation of the American Southwest (exclusive of New Mexico and southwestern Colorado) can be found in the 2007 book, *The Ribbon of Green* by Robert Webb, Stanley Leake, and Raymond Turner (Webb et al. hereafter). In *The Ribbon of Green,* the authors use sets of matched photographs to evaluate and interpret historical changes in riparian vegetation along southwestern streams that have taken place since the late 19th century. As stated in the book's preface, the authors "concentrate on the question of long-term changes, addressing whether we have endured a net loss of woody riparian vegetation" over that time (Webb et al. 2007:x).

More narrowly, the authors challenge the accuracy of a widely-repeated claim that 90 percent (or a similarly large percentage) of riparian vegetation in the southwestern United States has been lost due to human activities. In *The Ribbon of Green,* Webb et al. either explicitly or implicitly conclude that: (1) the Southwest has not endured a net loss of woody riparian vegetation over time; (2) claims of such loss have been exaggerated and misguided; (3) the 90 percent loss figure is myth; (4) riparian vegetation has changed to some degree, but change is to be expected in a dynamic ecological system and is neither bad nor good; and (5) the role of human actions in bringing about change in riparian systems has been overblown relative to the influences of natural processes. In this chapter, we examine the validity of those conclusions and the methodology that led to them, and offer further information and interpretation.

For their study, Webb et al. compared 2,724 sets of repeat photography with native and nonnative woody riparian vegetation visible in at least one of the matched photos. Of the 2,724 sets examined, 215 sets totaling 480 individual photographs are published in the book. Lengths of time spanned by the published sets vary widely, from a few recent decades (e.g., 1988 compared to 2000) to more than a century (e.g., 1863 compared to 2000). The book is more than just a comparison of photographs, however. Webb et al. also summarized the biology and distribution of riparian ecosystems in the study area, drawing upon numerous publications and research projects completed prior to 2007. They look at human influences, climatic fluctuations, hydrologic changes, and the life history strategies of dominant woody vegetation species in the selected riparian ecosystems.

The Ribbon of Green is a remarkable reference book for many things riparian in the geographic areas they address. That said, we believe Webb et al. are doing a disservice to the very resource they celebrate by painting too sanguine a picture of the history and status of southwestern riparian ecosystems. The scientists who initially raised concerns about the historical trajectory, condition, and future of riparian habitats were leaders who drew attention to conservation needs for these ecosystems. In their preface, Webb et al. dispute the concerns of those scientists and generally understate the level of threat to riparian vegetation that existed in the mid-20th century. At its core, the book is revisionist, an attempt to correct a record that, in the authors' view, had been distorted. But in their enthusiasm to counter often-exaggerated estimates of historical riparian losses and their interest in promoting the effects of climate and other natural processes on riparian systems as opposed to human impacts, the authors risk tipping the scales too far in the opposite direction.

There is danger in understating human impacts on southwestern riparian habitats. The risk is a desensitization of policy makers, resource managers, and the public to existing and future threats. Another risk is a weakening of the political will to avoid, minimize, and mitigate such threats. Vigilance is particularly important now because southwestern riparian habitats are undergoing highly visible changes resulting from release of the tamarisk leaf beetle (*Diorhabda* spp.), a biocontrol effort that was in its infancy when Webb et al. wrote *The Ribbon of Green.* The long-term ecological consequences of the beetle release are unknown and much debated, but in the short term it is clear that the amount of woody riparian vegetation available to wildlife is shrinking as nonnative, invasive tamarisk (*Tamarix* spp.) defoliates and dies across the western United States (see Carothers et al., this volume; McLeod volume 1).

Have We Endured a Net Loss of Woody Riparian Vegetation?

Webb et al. pose the question of riparian losses in the book's preface, and then they answer it in the book's final two chapters with the following statements:

> "In general, riparian vegetation has had either an [a small] increase (49 percent [of the photographs]) or a large increase (24 percent [of the photographs]) in comparisons involving all years ...." (Webb et al. 2007:388)
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> "The time series of change ... suggests that, at least in the specific reaches for which we have documentation, riparian vegetation began to increase after about 1940. Increases in density of plants appear to have accelerated after the 1970s, followed by increases in the size of plants. Many deviations from this general pattern have been noted in the chapters on specific reaches." (Webb et al. 2007:388)
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> "Our interpretations of the available imagery ... suggest more gains than losses within our photographic views." (Webb et al. 2007:403)
>
> "Although we observed local decreases in riparian vegetation in many places, reach-scale decreases in the 20th century were present only in a few locations." (Webb et al. 2007:405)

In these statements, the authors couch their findings in terms of the specific locations photographed, or, at most, the river reach in which the photographs were taken. These geographic constraints are appropriate considering the spatial limitations of repeat photography, which we discuss at length below. However, a stated purpose of the study is to address allegations of riparian habitat loss across a very large area, the southwestern United States. Given this regional premise, readers can be forgiven if they understand the scope of authors' ultimate conclusions to be region-wide as well. The book's unmistakable message is we have not endured a net loss of woody riparian vegetation in the American Southwest despite more than a century of intensive agricultural, urban, and water resource development.

In the book's preface and accompanying notes (page 413), Webb et al. place the responsibility for originating the "perception of diminishing or degraded riparian areas" with academic and agency scientists who, for the most part, began publishing their concerns in the late 1970s. Specifically, Webb et al. report the claim of a 90 percent loss as having been traced back to a single paper (i.e., Ohmart et al. 1977). Webb et al. imply that the concerned scientists—many of whom pioneered the discipline of riparian ecology—exaggerated the extent of riparian losses right from the beginning. How does one account for the conflicting views of Webb et al. and those they disagree with?

Several explanations are possible. First, Webb et al. and the early concerned scientists came to the issue from different professional perspectives, and those perspectives carried with them different priorities and biases that likely colored their conclusions. Second, Webb et al. and the early concerned scientists defined the issue somewhat differently. The scientists who first raised the alarm were concerned about the loss of *native* riparian vegetation, not all riparian vegetation, and they were concerned about the reduction of wildlife habitat *quality*, particularly that posed by the proliferation of nonnative plant species. Webb et al., although they differentiate between native and nonnative species, concentrate on changes in the total amount of riparian vegetation. They pay little attention to habitat quality.

Third, Webb et al. and the concerned scientists arrived at their conclusions via different pathways, and their divergent approaches influenced their conclusions. Fourth, Webb et al. and the concerned scientists operated in different timeframes. The circumstances affecting riparian areas in the early 2000s, when *The Ribbon of Green* was written, differed in many ways from those in the 1970s, when the alarm was first raised. Moreover, Webb et al.'s temporal scope was necessarily limited to the periods when photographs were taken, while the concerned scientists were troubled by changes that have taken place since first European settlement. We discuss these points in the following sections and revisit the question of where that 90 percent loss figure originated.

Different Perspectives, Value Judgements, and the Issue of Habitat Quality

Robert Webb and Stanley Leake are hydrologists/geomorphologists, and their interest and expertise in physical processes is evident in the attention they pay to climatic variation, streamflow, flood records, and stream channel morphology. Raymond Turner, a plant biologist, has long favored climate shifts as the primary driver of vegetation changes in the Southwest (see Bahre 1991; Hastings and Turner 1965). In *The Ribbon of Green,* a focus on physical processes as opposed to human influences (as well as a conviction that relatively little riparian vegetation has been lost) is revealed in this statement: "That some patches of vegetation, growing as they do in a dynamic geomorphic milieu, are gone is to be expected" (Webb et al. 2007:ix). This is not to say that Webb et al. do not acknowledge human impacts—they do—but they place decidedly more emphasis on climatic fluctuations, hydrology, and geomorphological changes to explain the differences in riparian vegetation between compared photographs.

When describing the vegetation visible in their photographs, Webb et al. make no overt value judgments about what they see. From their perspective, differences observed between compared photographs reflect changes that are neither desirable nor undesirable, neither good nor bad. In their preface, Webb et al. decry the fact that some scientists *do* make judgments about the rightness or wrongness of changes they perceive. The authors criticize the "... value judgments and assumptions that creep into the issue of long-term change in the region's rivers. Some authors use pejorative words such as *destroyed* to describe what has happened to these watercourses historically and *lost* to describe changes in riparian vegetation.... These judgments derive from some scientists' perception that humans are the ultimate cause of regional change and therefore that any changes must be 'bad'" (Webb et al. 2007:ix)

Webb et al. are correct that value judgments have been made by others, and words such as "destroyed" and "lost" have been (and are) commonly used, but the authors' presumption that fellow scientists believe "humans are the ultimate cause of regional change and therefore that any changes must be 'bad'" is unsupportable and unmerited. One need not look far in the literature to find a different and more obvious motive among the scientists who launched the movement to conserve native riparian systems. Those scientists were generally ecologically-oriented ornithologists who saw riparian areas primarily as habitat for wildlife, particularly birds (e.g., Anderson and Ohmart 1976; Hehnke and Stone 1979; Hubbard 1971; Johnson and Simpson 1971; Knopf 1985; Phillips and Monson 1964; Rea 1983; Schmitt 1976; Stevens et al. 1977). Their orientation was (and is) to view riparian systems primarily through the lens of services provided, not to humans directly, but to something humans (biologists in particular) care about: the health and functioning of ecological communities.

From that perspective, a reduction in either the amount of riparian habitat or a diminishment in the quality of that habitat—as measured in species richness, diversity, and density—is undesirable. That is a commonly held view whether the undesirable condition is caused by human activity or not. Value judgments did not creep into the assessments of the concerned scientists; value judgements were an overt and legitimate aspect of their assessments.

Different Pathways—Repeat Photography

Webb et al. approach the issue of riparian change through a methodical study using compared photographs. An implicit premise of the study is that the collection of historical photographs available to the authors reliably documents conditions of riparian vegetation across the Southwest at the time of Euro-American settlement (any effects of earlier Native American, Spanish, and Mexican activities are necessarily moot because such activities predated photography). We believe this premise and other aspects of the study are open to question and explain why below.

First, while a comparative photographic record provides valuable visual documentation of two points in time for a specific location, such a record has limited application and utility for extrapolation over a large area. Constrained by nonrandom snapshots that are rigidly confined both spatially and temporally, repeat photography is not well suited to provide a systemwide assessment of riparian condition and change over time. Achieving adequate geographic coverage is difficult if not impossible for such an assessment. On page 404 of *The Ribbon of Green*, the authors state, "In this book, we used thousands of repeat photographs to address the claim that 90 percent of riparian vegetation has been lost in the southwestern United States" (Webb et al. 2007:404). On the same page, however, they themselves cast doubt about the appropriateness of their approach by acknowledging they could not fully meet this objective for reasons that include lack of "full regional coverage."

The acknowledgment also alludes to an important weakness of the study and the conclusions based on it. The regional coverage falls far short of "full." While 2,724 sets of photographs may seem to be a great many, the actual amount of riparian environment documented in those photographs barely signifies when compared to the tens of thousands of miles of rivers, streams, and washes in the American Southwest. In Arizona alone, there are approximately 14,605 miles of perennial and intermittent waterways and another 112,900 miles of ephemeral washes (Arizona Department of Environmental Quality 2011). Coverage of the rivers and streams included in Webb et al.'s study is sparse and spotty, and many waterways (most notably the Rio Grande and its tributaries) are not represented at all.

Moreover, the images used in the study suffer from the spatial limitation common to all photography. That is, the area depicted in each image is only as large as the camera's field of view. Each photograph in this study represents only one narrowly circumscribed point in a reach of stream that may extend scores or hundreds of miles. A photograph cannot speak to riparian conditions that exist a few yards, let alone miles, upstream or downstream of the camera station.

The study also suffers from what practitioners of repeat photography call "spatial bias" (Villarreal et al. 2013). The most obvious example of this is the fact that 46 percent (1,251 of 2,724) of the matched photographs used in the study were taken in a single river reach: the Colorado River through Grand Canyon. That overrepresentation biases the result toward circumstances that are not characteristic of southwestern rivers and streams in general. Grand Canyon is a severely canyon-confined reach, while most southwestern waterways are more alluvial and open in nature. Spatial bias is an inherent weakness of a repeat photography project that uses pre-existing baseline imagery (Bahre 1991; Pickard 2002; Pupo-Correia et al. 2014; Villarreal et al. 2013). The camera stations in such studies are not purposefully and randomly selected to achieve a spatially unbiased sample. Rather, they are accidents of history: functions of photographs that happen to have been taken years earlier, usually (as in this case) for reasons unrelated to the question at hand. In the Webb et al. study, just two river reaches (Grand Canyon and Cataract Canyon) of the over 37 reaches covered account for over half (55 percent) of all matched images. The historical photographs in those matches happen to exist because photographers accompanied two expeditions in the 19th century.

Other locales that appear to be overrepresented (based on the published images) include stream gaging stations and readily accessible sites near bridges and communities. Types of environments that appear to be underrepresented in those images include river reaches and associated riparian vegetation lost or altered by reservoirs, urban development, and bottomland agriculture. For example, four Colorado River reservoirs (Lakes Powell, Mead, Mohave, and Havasu) that collectively inundated approximately 400 mainstem river miles are represented in the book by only seven matched pairs of photographs.

In the book's preface, Webb et al. use a term implicative of a second common problem of repeat photography: limited temporal coverage. The authors write: "... [repeat photography] is the only method that provides a *glimpse* [our emphasis] into predevelopment conditions in our region, a glimpse that is required to address the question of gains or losses of riparian vegetation" (Webb et al. 2007:iv). "Glimpse" implies a fleeting moment, and a moment in time is all each photograph in the study captures. The validity of the study, however, rests on the premise that each image represents the usual conditions at that location in the “past” compared to the usual conditions in the “present.” This assumption is difficult to defend for environments as dynamic as southwestern riparian systems, systems that are subject to unpredictable scouring floods and characterized by fast-growing plant species. The vegetation incidentally captured in a photograph may be very different from the vegetation that would have been shown if the photograph had been taken at the same location a year, a month, or even a week earlier or later.

The problem of temporal coverage is aggravated by temporal inconsistency. Webb et al. maintain that their “repeat photography establishes the visual condition of woody riparian vegetation at the end of the 19th century and the beginning of the 20th century” (Webb et al. 2007:403). However, this narrowly defined baseline does not exist. There is no common historical baseline of any kind. Far from it. Webb et al. report that the earliest dates of the initial photographs used to assess change since “predevelopment” range from 1863 (for the lower Colorado and Mohave rivers) to 1940 (for the Agua Fria River). The oldest images for all other stream reaches fall somewhere between those 2 years (Webb et al. 2007:44).

Using dates ranging as far apart as 1863 and 1940 to represent riparian conditions “predevelopment” is questionable, but the temporal inconsistency is even greater than that. In individual matched sets reproduced in the book, dates of the older or oldest photographs range well into the 1980s. Overall, the “baseline” dates number in the dozens and are scattered over more than a century. Dates of the “new” photographs in matched sets, those documenting recent conditions, generally range from the late 1990s to the early 2000s. Thus, the time span of individual matched sets is highly variable, ranging from over 130 years (e.g., 1863 compared to 2000) to under 20 years (e.g., 1988 compared to 2000).

The relatively recent dates of many baseline photographs point to yet another problem with the study. Webb et al. refer to it when they admit that “for many watercourses ... the initial photography postdates a strong human influence on the watershed” (Webb et al. 2007:403). Certainly, assessing the role of human activity in what is essentially a “before and after” exercise is hindered when the baseline (i.e., “before”) photographs were taken after—sometimes long after—human activities (e.g., cattle grazing, beaver trapping, damming) commenced in the Southwest.

Different Pathways – Different Times

The initial concern over historical losses of riparian habitat in the Southwest and the realization that what was left was seriously threatened took place amidst a nationwide groundswell of environmental awareness beginning in the 1960s and 1970s. Ecology was a young, rapidly expanding science, and the role of humans as an ecological factor was gaining recognition. In the Southwest, ecologically oriented ornithologists were among the first to recognize a link between riparian habitat value, human activities, reduction in available riparian habitat, and species decline. They began to see a decrease in riparian avian species richness and abundance about the same time they were discovering the disproportionate importance of riparian habitat for birds (see, e.g., Johnson et al. 1977; Rea 1977).

Biologists assessed avian population status and trajectories in riparian areas by comparing the results of recent bird surveys with historical accounts and distribution lists (e.g., Stephens 1885; Swarth 1914). Following one of these assessments, Hunter et al. (1987) reported that, since the year 1900, 17 riparian bird species had either declined or been extirpated from one or more of 16 southwestern river reaches. The lower Colorado, lower Gila, lower Salt, and lower Santa Cruz Rivers were hardest hit, with 12 to 13 species lost or declining in each system. These were the reaches most altered by dams, diversions, channelization, bottomland agriculture, urban and industrial development, and/or lowered water tables due to heavy groundwater withdrawals. In contrast, the river reaches upstream of dams and in less developed areas (e.g., the Upper Gila, Upper Salt, Upper Santa Cruz, and San Pedro Rivers) experienced the least change in riparian avifauna, with 0–2 species lost or declining in each system Hunter et al. (1987).

The fact that riparian bird species declined concurrently with human modification of river systems was not lost on observers. A reduction in the availability of suitable riparian habitat due to human activities was the logical inference, an inference supported by observed deviations from historical descriptions of vegetation found in written accounts from the 18th through the early 20th centuries (see, for example, Bartlett 1854; Bell 1869; Bolton 1919; Emory 1857; Froebel 1859; Johnston 1848; Mearns 1907; Rusling 1875; Swarth 1905; Willard 1912).

By the middle of the 20th century, the stands of riparian trees—mostly cottonwood (*Populus* spp.), willow (*Salix* spp.), and mesquite (*Prosopis* spp.)—described in those accounts had often dwindled or disappeared (Bahre 1991; Dobyns 1981; Rea 1983; Webb et al. 2014). The reduction in the amount of streamside vegetation was strikingly apparent and troubling because of the rarity and relative importance of that vegetation in the Southwest compared to other regions. In the Southwest, riparian habitats make up a very small fraction of the generally arid and semiarid landscapes and are naturally more disjunct than those of wetter climes. At best, they are connected only along narrow ribbon-like bands and usually separated by vast expanses devoid of similar vegetation (Skagen et al. 1998).

The most easily attributable (although not the earliest) human-caused changes in southwestern riparian vegetation resulted from the dams and diversion structures that were built on nearly every river in the Southwest following passage of the Federal Reclamation Act of 1902. That Act created the Bureau of Reclamation and authorized construction of Roosevelt Dam, the first of seven dams built on Arizona's Salt River and its largest tributary, the Verde River (see table 1). Combined, the seven dams and associated diversions choked off all but flood flow in the once perennial lower Salt River. By the end of the 1920s, loss of discharge from the Salt River (the Gila River's largest tributary), combined with damming of the Gila mainstem, converted the lower Gila River into a mostly dry channel (Huckleberry 1996; Werner volume 1). The Agua Fria River, another Gila River tributary in Arizona, suffered a similar fate. Altogether, the dewatered portions of just these three rivers totaled more than 340 miles (see table 1).

The dewatering of the Salt, Agua Fria, and Gila River channels had a particularly egregious effect on downstream riparian systems, but only a few dams in the Southwest impeded flow to that degree. Every dam, however, altered downstream flow in some fashion, typically by disrupting the natural hydrograph, modulating and decreasing downstream flow, and reducing the magnitude of peak flows. Downstream sediment loads were diminished, stream channel and floodplain morphology altered, and water tables lowered. These modifications affected the persistence and recruitment of native riparian vegetation downstream of dams. Upstream of dams, impounded water drowned whatever riparian vegetation once grew along the flooded mainstem channel (over 250 miles on the

Table 1—Approximate linear miles of selected rivers impacted by dam construction in Arizona.

River	Total miles in Arizona	Inundated channel upstream	Regulated flow downstream[a]	Dewatered channel downstream[b]	Total miles (%) not free-flowing[c]
Colorado[d]	704	250+[e]	454	0	704 (100%)
Gila[f]	440	20	55	265	340 (77.3%)
Salt[g]	200	40	11	38	89 (44.5%)
Verde[h]	170	20	20	0	40 (23.5%)
Agua Fria	120	12	0	38	50 (41.7%)
Total	1,634	342+	540	341	1,223 (74.7%)

Source: R. Roy Johnson

[a] Reaches of rivers below dams into which controlled releases are made for irrigation, domestic and industrial water, and power production.

[b] Total miles of stream channel that flow only during flood stage and that have been dewatered and desertified, losing nearly all their riparian habitat.

[c] Total includes previous three categories in table.

[d] From Glen Canyon Dam to U.S.-Mexico boundary below Yuma.

[e] Includes only reservoirs behind Hoover, Davis, Parker, and Headgate Rock Dams (i.e., Lakes Mead, Mohave, Havasu, and Moovalya, respectively), four of the nine major storage and diversion dams on the Lower Colorado River.

[f] Includes Coolidge Dam, San Carlos Reservoir; Ashurst-Hayden Diversion Dam.

[g] Includes Roosevelt Dam, Roosevelt Lake; Horse Mesa Dam, Apache Lake; Mormon Flat Dam, Canyon Lake; Stewart Mountain Dam, Saguaro Lake; Granite Reef Diversion Dam.

[h] Includes Horseshoe Dam, Horseshoe Reservoir; Bartlett Dam, Bartlett Reservoir; Granite Reef Diversion Dam at confluence with the Salt River.

lower Colorado River alone) as well as along flooded tributary channels (e.g., the Virgin River arm of Lake Mead and the San Juan arm of Lake Powell) (see table 1).

The impacts of dams and diversions on riparian vegetation were perhaps the most obvious when concern about riparian habitat loss and impairment was first raised, but those impacts were not the only ones. Large swaths of bottomlands capable of supporting native vegetation had been converted to agriculture and other human uses. No region-wide estimate of the magnitude of this conversion is available, but local studies have been done. For example, Ohmart et al. (1988) quantified the shift to agriculture for a 21,504-ha, 71-mile-long stretch of the lower Colorado River. Between 1938 and 1986, the proportion of that area covered by native riparian vegetation decreased from 88.6 to 19.8 percent, while crops increased from 0.3 to 61.1 percent. Even greater losses of native riparian vegetation to agriculture (ca. 95 percent) were calculated along the lower Gila River (Lacey et al. 1975). Riparian displacements of similar magnitude were thought to be typical of fertile, easily irrigated bottomlands throughout the Southwest (Ohmart et al. 1988).

Additional reductions in native riparian vegetation were reported due to urbanization, groundwater pumping, bank stabilization, channelization, levee construction, and other activities (National Research Council 2002; Poff et al. 2011). Throughout the 19th and early 20th centuries, cattle grazing and agriculture contributed to erosion and loss of riparian integrity in many waterways of the Southwest, including small streams and arroyos (Dobyns 1981). Even woodcutting took a toll. As early as 1910, for example, it was noted that woodcutting for fuel had resulted in the “practical disappearance” of mesquite along much of the lower course of the Colorado River (Grinnell 1914), an area where 50 years earlier observers reported large, at times impenetrable, mesquite woodlands (Bartlett 1854; Bell 1869; Emory 1857).

A huge mesquite bosque on the Salt River estimated to cover 25 square miles had been replaced by metropolitan Phoenix and surrounding fields and orchards (Jacobs and Rice 2002). Additional large, well-documented bosques that had been decimated or obliterated by the mid-1900s included the San Xavier Thicket (or Great Mesquite Forest; Webb et al. 2014) on the middle Santa Cruz River; the Komatke Thicket (or New York Thicket; Rea 1983) at the confluence of the Santa Cruz and Gila Rivers; and the Casa Grande National Monument bosque just south of the Gila River (Johnson et al., this volume; Judd et al. 1971). These bosques succumbed to a combination of activities, including wood cutting, clearing for agriculture, upstream dam construction, and finally groundwater overdraft and cessation of flowing surface water (Phillips et al. 1964; Rea 1983; Webb et al. 2014).

In the mid- and late 1900s, scientists and others certainly lamented the riparian habitat that had been lost and degraded up to that time, but it was ongoing and accelerating threats that incited a call for action. Demands for water in the arid Southwest were growing along with the human population and economic development, and those demands were being met by ever more ambitious water development and storage projects. The projects authorized by the U.S. Bureau of Reclamation in the Colorado River basin during that period included eight[1] new large dams, numerous smaller dams, and a score of large water diversion projects, including the massive Central Arizona Project (Bureau of Reclamation 2018a,b).

Unprecedented threats to the integrity of riparian habitat included the rampant spread of nonnative tamarisk along southwestern drainages and an intentional assault on riparian vegetation called "phreatophyte control." The spread of tamarisk was troubling for biologists largely because, in the early days of riparian study, tamarisk was almost universally believed to out-compete native vegetation and replace it with inferior wildlife habitat (Bateman et al. 2013; Haase 1972; Harris 1966; Rosenberg et al. 1991; Sogge et al. 2013). Phreatophyte control was a coordinated strategy of land and water managing agencies, water suppliers, and industrial water users to increase water yield for beneficial (i.e., human) use by removing deep-rooted, high-water-consumptive plants (phreatophytes) from along streams and rivers. While tamarisk was the principal focus of eradication efforts, native species were targeted as well. Woody riparian plants would be replaced, if possible, with low-water-consumptive and/or economically useful vegetation (e.g., grass forage for livestock).

In Arizona, phreatophyte control was first field-tested in the 1940s as a means to provide water needed to expand wartime copper production at the Phelps Dodge Morenci Mine (Gatewood et al. 1950). By 1969, thousands of acres of riparian vegetation had been treated and many more projects were planned. In that year, the Pacific Southwest Inter-Agency Phreatophyte Subcommittee conducted a survey of phreatophyte control projects in Arizona. The results of the survey were reported by Affleck (1975) and are summarized here in table 2. A total of 11,909 acres of riparian vegetation had been treated as of the date of the survey; 20,415 acres of treatment were underway; and 80,191 acres of treatment were planned, for a total of 112,515 acres (table 2).

[1] Three of the big dams were never built. Marble Canyon Dam and Bridge Canyon Dam (aka Hualapai Dam), both planned for the Grand Canyon, were removed from consideration in 1968. Hooker Dam, slated for the Upper Gila River, remained a possibility into the 1980s. Serious interest in building an alternative to Hooker Dam for water storage on the Upper Gila was revived with passage of the Arizona Water Settlements Act of 2004 (P.L. 108–451), but nothing has materialized as of this writing (Pearson 1994; Utton Transboundary Resources Center 2014).

Table 2—Acres of phreatophyte treatments completed, underway, and planned in Arizona according to a survey conducted by the Pacific Southwest Inter-Agency Phreatophyte Subcommittee in 1969.

Phreatophyte type	Treatment completed	Treatment underway	Treatment planned	Total acres (%)
Native	4,657	7,710	0	12,367 (11)
Tamarisk	5,894	7,455	2,711	16,060 (14)
Mixed native and tamarisk	1,358	5,250	77,480	84,088 (75)
TOTAL	11,909	20,415	80,191	112,515 (100)

Source: Calculated from data provided by Affleck (1975) in table 4.

Of that total, 11 percent of the vegetation was native (cottonwood, willow, mesquite); 14 percent was tamarisk; and 75 percent was mixed tamarisk and native. At the time, estimates of the total coverage of riparian vegetation in Arizona outside of National Parks and other protected areas ranged from 279,600 acres (Babcock 1968) to 320,000 acres (Lower Colorado Region Comprehensive Framework Study, Appendix VI 1971). Based on these estimates, it would have been reasonable to conclude that between 35.2 and 40.2 percent of the remaining unprotected riparian habitat in the State was threatened with imminent destruction.

In short, in the mid-20th century, the perception of loss and decline of native southwestern riparian habitat was well justified, and the threat of substantial additional harm was acute. By the time Webb et al. launched their study in the early 21st century, the context was quite different. The ecological importance of riparian systems had become universally recognized, and some of the threats that loomed largest in the mid-20th century had diminished. Public and Federal enthusiasm for new dam construction had dwindled; intentional destruction of native riparian vegetation for water salvage had become socially and politically unacceptable; conservation of riparian habitats had become the norm in natural resource management; and several restoration projects were planned and underway. This is not to say old threats had not persisted (e.g., escalating water demand associated with human population growth), or that new threats had not surfaced (e.g., predicted increases in drought occurrence and severity), but by the turn of the 21st century, riparian habitats were widely valued and had powerful advocates.

So ... Have We Endured a 90 Percent Loss of Woody Riparian Vegetation in the Southwest or Not?

Answering this question has been hampered by the lack of a credible baseline from which to measure change. To our knowledge, no one has attempted to estimate the total amount of woody riparian vegetation that existed throughout the Southwest before European settlement, and no one has attempted to quantify the amount of change that has taken place since that time. What researchers have done is estimate the extent of pre-settlement riparian vegetation and historical change for a few individual river reaches.

One of these estimates, an effort by Ohmart et al. (1977) to quantify the loss of cottonwood-willow in the Lower Colorado River Valley, is of special interest. It is this publication that Webb et al. identifies as the ultimate "scientific justification" for the statement, "We've lost 90 percent of the riparian vegetation in the southwestern United

States" (Webb et al. 2007:ix, endnote 3). To be fair to Ohmart et al. (1977), they made no such claim, nor did they present information that justified such a claim.

To begin with, although Webb et al. placed quotation marks around the statement, "We've lost 90 percent of the riparian vegetation in the southwestern United States," they gave no attribution. Rather than an actual quotation, this appears to be a generic representation of a type of claim seen in the literature. The connection Webb et al. make between this generic "statement" and Ohmart et al. (1977) is through a chain of citations associated with an actual statement made in support of an executive order issued by Arizona Governor Rose Mofford for the protection of riparian areas (Governor's Riparian Habitat Task Force 1990). The actual statement reads as follows: "According to most estimates, over 90 percent of the native riparian areas along Arizona's major desert watercourses have been lost, altered or degraded as a result of man's activities (SCORP 1989; TANC 1987; Warner 1979)"[2]. The actual statement varies significantly from Webb et al.'s generic statement. Note that the actual statement refers: (1) only to "native riparian areas," not to all riparian areas; (2) only to "Arizona's major desert watercourses," not to the entire "southwestern United States"; and (3) to "lost, altered or degraded" riparian areas, not just to "lost" riparian areas.

The first two references cited by the Task Force (SCORP 1989 and TANC 1987) both lead to the third reference (Warner 1979). Warner (1979) discusses local losses (not Statewide losses) of riparian vegetation in Arizona on the order of 95 percent and cites Ohmart et al. (1977) and Lacey et al. (1975). The analysis conducted by Ohmart et al. (1977) was limited to 200 miles of the lower Colorado River and to only one component of riparian habitat (the cottonwood-willow community). Ohmart et al. (1977) hypothesized a reduction of that community over time but did not calculate a percentage loss. They stated that:

"Cottonwood communities have declined from high abundance (5,000 acres plus) along the lower Colorado River in the 1600s to scattered groves containing a few mature individuals today. Anderson and Ohmart (1976) have estimated that only 2,800 acres of cottonwood-willow community remain along the lower river. If one was to consider pure cottonwood communities, it would be less than 500 acres" (Ohmart et al. 1977:45)[3].

Lacey et al. (1975) performed a similar analysis for predominantly mesquite riparian habitat along a 58-mile reach of the lower Gila River and concluded that only about 5 percent of the theoretical 1860 riparian base remained.

In subsequent years, the findings of both Ohmart et al. (1977) and Lacey et al. (1975), as well as the understanding of those findings reported in Warner 1979, were misunderstood and distorted in numerous publications. For example, the "cottonwood-willow" and "pure cottonwood" communities of Ohmart et al. (1977) were misunderstood to represent all riparian vegetation in the lower Colorado River (Council on Environmental Quality 1978:317–318). The 5 percent of original riparian habitat

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[2] SCORP is the Arizona Statewide Comprehensive Outdoor Recreation Plan, and TANC is The Arizona Nature Conservancy.

[3] The amount of vegetation cited by Ohmart et al. (1977) as having been mapped by Anderson and Ohmart in 1976—i.e., 2,800 acres of cottonwood-willow community and 500 acres of pure cottonwood for a total of 3,300 acres—is suspect. The acreage numbers conflict with those in Ohmart et al. 1988 (table 22), which lists a total 3,354 ha (8,288 acres) of cottonwood-willow (apparently inclusive of "pure cottonwood") mapped in the same river reach (Lower Colorado River) by the same researchers (Anderson and Ohmart) in the same year (1976). The similarity of numbers (3,300 acres and 3,354 ha) suggests Ohmart et al. (1977) confused hectares and acres. If so, their entire analysis is thrown into question.

remaining on the lower Gila River (Lacey et al. 1975) became 5 percent of original riparian habitat remaining in Arizona (Arizona State Parks 1989:60).

Eventually, it became common in both scholarly and popular literature to refer to a 90 percent (or over 90 percent, or 90–95 percent) loss (or loss and degradation) of original (or native) riparian habitat in Arizona (or in Arizona and New Mexico or in the western United States) (e.g., Fleishner 1994; Johnson 1989; Mac et al. 1998; National Research Council 2002; Noss 1995; U.S. Fish and Wildlife Service 2000; Yong and Finch 1996; Zaimes 2007). Few authors cite Ohmart et al. (1977) and Lacey et al. (1975) directly, but citation trails commonly lead back to them or to a more comprehensive analysis of riparian change on the lower Colorado River completed by Ohmart et al. (1988).

Citing Mearns (1907), Ohmart et al. (1988) assumed 160,000 ha (400,000 acres) of indigenous riparian vegetation, not just cottonwood-willow, covered lower Colorado River bottomlands at the turn of the 19th century.[4] Based on vegetation mapping completed by Anderson and Ohmart (1984) and Younker and Andersen (1986), Ohmart et al. (1988) concluded that only 40,000 ha of the original 160,000 ha of riparian vegetation remained. Of that 40,000 ha, only about 17 percent (ca. 6,800 ha) consisted of native trees and shrubs. The balance of the remaining vegetation (ca. 33,200 acres) consisted of tamarisk and mixed native/tamarisk communities. These figures suggest that, in less than a century, the total volume of riparian vegetation along the lower Colorado River was reduced by 75 percent, and native riparian vegetation, unmixed with tamarisk, was reduced by more than 95 percent.

These few regional analyses are clearly inadequate to support the claim that 90 percent (or a similarly large percentage) of riparian vegetation has been lost, or lost and degraded, across the entire Southwest. In this regard, we agree with Webb et al.; the “90 percent loss” claim is unsupported by quantitative data or analysis. It simply became conventional wisdom legitimatized through repetition.

Postscript

The urgent concern that led to Governor Mofford’s 1990 executive order for the protection of riparian areas was about the loss of indigenous cottonwood, willow, and mesquite-dominated riparian woodland communities due to human activities. These woodlands are known to have increased in some drainages, notably along the Upper San Pedro River (Stromberg and Tellman 2009), but the preponderance of evidence suggests a net decrease across the Southwest. The impacts of dams, water diversions, channelization, bank stabilization, groundwater withdrawals, agriculture, urban development, grazing, invasive species, and a multitude of other factors have been extensive and well documented, if not well-quantified.

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[4] Ohmart et al. (1988:24) state that, “In 1894, Mearns (1907) estimated that there were about 160,000 to 180,000 ha (400,000 to 450,000 acres) of alluvial bottomland between Fort Mohave and Fort Yuma covered by riparian vegetation.” To set the record straight, the estimate was not made in 1894, nor was it made by Mearns, nor was the maximum 450,000 acres. What Mearns (1907:125) actually wrote was, “As a result of an investigation along the Colorado River, made in January, 1902, by the hydrographic branch of the U. S. Geological Survey, the extent of the alluvial bottom land ... was found to be from 400,000 to 500,000 acres.” Mearns, who did not venture north of Yuma during his visit to the Colorado River in 1894, likely obtained the cited information from the Reclamation Service’s First Annual Report, in which Lippincott and Davis (1903) refer to 400,000 to 500,000 acres of “irrigable lands” surveyed along the Lower Colorado River in the preceding year.

Webb et al. endeavored to address the issue of riparian change through time in a more systematic way than has been attempted in the past. In our judgement, however, repeat photography is not the appropriate tool. It is not sufficiently comprehensive, rigorous, or objective to measure change over such a large area and long period of time. Weaknesses of that approach for addressing this question include sparse and inadequate spatial and temporal coverage, spatial and temporal bias, temporal inconsistency, and a multitude of baselines from which to measure vegetation change in a highly changeable ecosystem.

What is called for is a comprehensive, stream-by-stream quantitative analysis that encompasses, at a minimum, the entire States of Arizona and New Mexico. While we currently lack a well-reasoned baseline from which to quantify historical change, the tools needed to accomplish that task are fast improving. Geospatial technology designed to reconstruct pre-settlement riparian conditions has become increasingly refined (e.g., LANDFIRE 2008) and shows promise for assessing historical change across a broad, multi-state scale. Macfarlane et al. (2016), for example, used such technology to determine the ratio of existing native riparian vegetation cover to pre-European settlement riparian vegetation cover for 25,685 km of perennial streams and rivers in Utah. They found that approximately 62 percent of riparian vegetation along those streams showed significant (> 33 percent) to large (> 66 percent) departures from historical conditions. The departures were predominantly attributed to human land use impacts and conversion of native riparian vegetation to nonnative or upland vegetation types. A comparable analysis for the Southwest would be welcome, not only as a window on historical riparian conditions and past changes, but as a tool to help assess the effects of tamarisk biocontrol as it plays out in the coming years.

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Chapter 2. A Naturalized Riparian Ecosystem: Consequences of Tamarisk Leaf Beetle (*Diorhabda* spp.) Biocontrol

Steven W. Carothers, R. Roy Johnson, and Kenneth J. Kingsley

Introduction

Presence of the introduced genus *Tamarix* has been a perplexing problem for decades along rivers of the southwestern States. It is clearly an invasive species occurring along most perennial, ephemeral, and intermittent drainages of the Southwest including rivers, small streams, and normally dry washes. It seems to reach highest densities and form monocultures along waterways with altered flow regimes, but it can also invade unaltered streams and small springs, where it may become the dominant or exclusive woody species. Approximately 150 years after *Tamarix* was first introduced into the Southwest, it was being reviled as a notorious phreatophyte that was thought to measurably deplete ground and surface water at the expense of native riparian habitat (Chew 2013; Robinson 1952).

Over the past 60-plus years, much research and effort have gone into understanding *Tamarix* biology and the largely failed struggles to eradicate this nonnative shrub/tree's species (Chew 2009, 2013; El Waer et al. volume 1; Horton 1964, 1977; Sher and Quigley 2013; Zavaleta 2000, 2013). On the issue of water consumption by *Tamarix*, recent findings discourage generalizations as to excessive water use. Findings indicate that the species' complex is a stress-adapted group with a low to moderate water consumption that primarily replaces native vegetation when conditions within watersheds become unsuitable for native species colonization (Nagler and Glenn 2013; Nagler et al. 2012). Nagler and Glenn (2013) also demonstrated that from 1967 to 1982, water salvage projects by *Tamarix* removal did not achieve a sustainable recovery of water. Analysis of more recent water salvage/*Tamarix* removal literature concluded that increases in water yield after removal are only likely to occur when a *Tamarix* stand with high leaf area is replaced with low leaf area recolonizers (Shafroth et al. 2005).

It is also well documented that its two dominant species, *T. ramosissima* and *T. chinensis* and their hybrids, can become a naturalized part of the landscape in many areas and provide a unique *Tamarix*-dominated habitat type (Brown et al. 1987; Chew 2013; Johnson and Carothers 1987; Scott et al. volume 1). Although not all riparian birds find *Tamarix* to be suitable nesting habitat (i.e., woodpeckers, most cavity nesters, large raptors), studies over the past 30 years show that remarkably productive wildlife habitat is provided where *Tamarix* dominates and mixes with native riparian woody species (Bateman et al. 2013a; Brown et al. 1987; Brown and Trosset 1989; Darrah and van Riper 2017; Hunter et al. 1988; Johnson et al. 2012; Sogge et al. 2005, 2008, 2013; van Riper et al. 2008). In this chapter, we call this a "naturalized community," which is a unique vegetation assemblage or group of woody riparian plant species, largely dominated by the nonnative *Tamarix*, but intermixed with native species like mesquite (*Prosopis* spp.), cottonwood (*Populus* spp.), willow (*Salix* spp.), arrowweed (*Pluchea sericea*), seepwillow (*Baccharis* spp.), and others that are repeatedly found together within a riparian corridor.

The relatively brief reign of *Tamarix* appears to be coming to an end. In 1999, the Department of Agriculture through its division of Animal and Plant Health Inspection Service (APHIS) issued research permits for the release of *Diorhabda elongata*[1] (Coleoptera: Chrysomelidae), a species of nonnative tamarisk leaf feeding beetle, in secure field cages at 10 sites in six States. Based on the success of the field trial, in 2001 permits for the release of the beetle from field cages were issued by APHIS to the Agricultural Research Service, Bureau of Land Management, and Bureau of Reclamation. By August of 2005, the biocontrol program was officially underway. This was the culmination of a *Tamarix* biocontrol effort that had been in development for at least two decades (Bean et al. 2013; DeLoach et al. 2003; McLeod volume 1). These beetles of Asian origin rapidly spread throughout the West, beginning in limited areas of Colorado, Nevada, and Utah.

By 2015, the expanding beetle invasion and their repeated episodic defoliation of *Tamarix* eventually led to varying rates of dieback. This included significant levels of plant mortality in some riparian ecosystems throughout the West (Bloodworth et al. 2016; Hultine et al. 2015). The leaf-beetles have not only become more widespread than once predicted, they are also rapidly adapting their life cycles and behavior to facilitate penetration into areas where they were never expected (Hultine et al. 2015; Meinhardt and Ghering 2012). Thus, it finally appears as if the *Tamarix* invasion of wet places in aridland regions of North America has met its first serious challenge as a result of successful biocontrol.

Many biocontrol efforts have had unforeseen consequences (Howarth 1983, 1991; Louda et al. 2003; Simberloff and Stilling 1996; many others). We believe that one unforeseen consequence of the *Tamarix* biocontrol triumph has been the rapid alteration of the functional benefit of thousands of acres of riparian habitat as breeding, stop-over, and wintering habitat for many species of wildlife, especially birds, in areas where the beetle impacts have led to repeated defoliation and plant mortality (McLeod volume 1). Moreover, we are not as optimistic as biocontrol proponents that the long-term benefit of the *Tamarix* destruction will be unassisted native species habitat restoration. With limited exceptions (see discussion on Grand Canyon below), most available evidence to date indicates that without massive efforts at active habitat rehabilitation, once *Tamarix* is removed, it is most often replaced by a mixture of native and nonnative grasses and herbaceous cover, not the woody vegetation necessary to support riparian wildlife (Gonzales et al. 2017).

In the arid Southwest, the beetle biocontrol has had a significant adverse impact on riparian habitat and wildlife conservation. Contrary to the assurances of the original proponents of the biocontrol efforts (DeLoach and Tracy 1997), there is little evidence that reestablishment of native woody species will naturally occur in most areas where

[1] *Diorhabda elongata* was later reclassified as *D. carinulata* and after that species release, three additional species (*D. carinata, D. elongate,* and *D. sublineata*) were tested in cages between 2002 and 2009. All four species were eventually released at about 70 sites in Texas (see McLeod volume 1). Today, *D. carinulata* and *D. sublineata* are known to be established and are responsible for most *Tamarix* defoliation with the former occupying the Colorado River Basin and expanding in range south and east while the latter is moving north and west from its original release sites in Texas.

biocontrol has resulted in significant defoliation and mortality of *Tamarix*. No serious efforts at habitat restoration are currently planned in any reasonable timeframe.

In this chapter, we have three objectives: first, we document the value of nonnative *Tamarix* as summer habitat for birds compared to native riparian habitats of mesquite bosques and cottonwood/willow, and mixed deciduous gallery woodlands; second, we specifically focus on the unintended consequences to native avifauna of dam construction, *Tamarix* invasion, native vertebrate colonization of the *Tamarix*-dominated riparian habitat, and subsequent biocontrol along approximately 300 miles of the Colorado River in Grand and Glen Canyons; and, third, we briefly review current allelopathic studies on the potential long-term fate of native woody riparian vegetation when growing alongside *Tamarix*.

Importance of *Tamarix* to Riparian Birds

Tamarix has not always been appreciated for its importance as wildlife habitat. Beginning in the late 1950s, investigations into the avian use of *Tamarix*-dominated riparian areas consistently demonstrated that the loss of native riparian vegetation and invasion of *Tamarix* had a negative effect on the population sizes of riparian birds (Anderson and Ohmart 1977). However, as verification of the fact that ecological generalizations concerning *Tamarix* are elusive, there were exceptions. Rosenberg et al. (1991) reported 14 pairs of nesting white-winged doves per acre in *Tamarix*-screwbean mesquite (*Prosopis pubescens*) communities in the Lower Colorado River Valley. In the late 1950s, Shaw (1961) and Shaw and Jett (1959) reported nesting white-winged dove (*Zenaida asiatica*) populations as high as 60 nests per acre in the "saltcedar" thickets adjacent to declining mesquite bosques near the mouth of the Salt and Gila Rivers near Gillespie Dam in central Arizona.

Hunter et al. (1988) compared avian use on three river systems, the Lower Colorado, Rio Grande, and Pecos Rivers. They found that on the Pecos River in New Mexico, where riparian trees were mostly rare before the invasion of *Tamarix*, several species of obligate riparian birds expanded their range into the Pecos River Valley coincident with the arrival of *Tamarix*. The Hunter et al. (1988) findings were surprising as avian use of *Tamarix* on the Lower Colorado and Rio Grande was comparatively low. Brown et al. (1987) and Carothers and Brown (1991) also documented avian range and population increases along the Colorado River in Grand Canyon and linked those increases to the arrival of *Tamarix*. For both the Pecos River and Colorado River in Grand Canyon, quantitative estimates of native vegetation cover and bird species density prior to the invasion of *Tamarix* were mostly non-existent. In the Pecos River Basin, *Tamarix* is reported to have invaded in 1912 (Hildebrant and Ohmart 1982). In Grand Canyon, *Tamarix* was first reported as relatively rare with only isolated occurrences along the river corridor in 1938 (Clover and Jotter 1944); however, as early as 1936 a pair of southwestern willow flycatchers was found nesting in *Tamarix* below Glen Canyon at Lees Ferry (Woodbury and Russell 1945).

Quantitative estimates of bird life along the Pecos and Colorado Rivers are only available long after the respective *Tamarix* invasions. However, prior to the invasion of *Tamarix* along the Pecos River, Hildebrandt and Ohmart (1982) report, without citing the source for their conclusion, that there were few existing tall and mature stands of vegetation that could be used by riparian birds. In the Grand Canyon, *Tamarix* only proliferated significantly in the riparian zone after 1963, coincident with the construction and operation of Glen Canyon Dam and the significant reduction of annual scouring floods (see more below on Glen Canyon Dam river corridor impacts).

It was not until the surprising rate of spread and rapid and repeated defoliation and tree mortality caused by the *Diorhabda* became obvious that the value of *Tamarix* as wildlife habitat was quantified. Sogge et al. (2008) reviewed some available literature (see Corman and Wise-Gervais 2005) for nesting birds in Arizona and elsewhere and determined that 49 species used *Tamarix* as nesting habitat. Over 75 percent of low- and mid-elevation riparian birds in Arizona are known to build their nests in *Tamarix* (Bateman et al. 2013a; Corman and Wise-Gervais 2005; Sogge et al. 2008).

In this chapter, we have expanded the *Tamarix*-bird-use literature search to include: (1) over 50 publications spanning several decades; (2) more southwestern States than just Arizona; and (3) birds recorded as using *Tamarix* for foraging and roosting as well as for nesting. For reference, we compare bird use of *Tamarix* to that of riparian vegetation associations dominated by native species, including mesquite, cottonwood/willow, and mixed deciduous habitats (table 3). Of a total of 143 species of lowland birds in the southwestern United States normally found during the spring and summer months associated with breeding activities, 105 (73 percent) have been recorded in *Tamarix*, 98 (69 percent) in mesquite, 81 (57 percent) in cottonwood-willow, and 67 (47 percent) in mixed deciduous habitat types. While not all the species listed in table 3 necessarily build their nests in the specific tree types or associations, the lists do serve to emphasize the relative importance of the *Tamarix* shrub/tree species as a wildlife resource.

The value of *Tamarix* to riparian wildlife is especially evident where native riparian species have significantly declined over the past decades. It is also apparent in disturbed drainages where native vegetation species can no longer recruit and survive due to land conversion by urbanization, agriculture, and/or alteration of the natural hydrograph (Sogge et al. 2013). Where native riparian species are specifically precluded, *Tamarix* is a far superior habitat to no woody habitat at all. It provides structural diversity for riparian wildlife species that does not exist when there are no woody tree species and/or low-growing native and nonnative grasses and forbs. We believe increased use by birds of *Tamarix*-dominated habitats versus exclusively native species-dominated habitats is the result of increased general habitat structure provided by the increased foliage volume and foliage height diversity. A *Tamarix*-dominated understory is normally unavailable in the exclusively native stands. The *Tamarix* growth form provides vegetative cover and foraging areas from the ground up, while riparian habitat consisting of mostly native species normally has low-growing herbaceous plants under the gallery forest or mesquite bosque canopies (see Bateman et al. 2013a). In addition, *Tamarix*'s ability to rapidly establish after disturbance and its high stress tolerance compared to native vegetation has resulted in the rapid proliferation of vegetative cover. Thus wildlife habitat is quickly available where previously either low-growing, non-woody cover predominated, or only bare ground was present (Hultine and Dudley 2013).

Moreover, it has been demonstrated that invertebrate density and diversity can increase within the riparian ecosystem in some areas where *Tamarix* has invaded (Stevens 1976, 1985; Strudly and Dalin 2013). Within a little over two decades following the closing of the floodgates of Glen Canyon Dam, *Tamarix* supported a relatively low species richness of invertebrates compared with native species, but it supported

Table 3—A comparison of breeding birds recorded in lowland riparian habitats) of the southwestern United States.

Table Key

* Birds recorded from the Colorado River and its tributaries in Grand Canyon (Brown et al. 1987; Corman and Wise-Gervais 2005).

Riparian dependency at lower elevations:

F = facultative, occurring approximately equally in or out of riparian habitats;

O = obligate, occurring in riparian or similar wetland habitats >90 percent of the time;

P = preferential, occurring in riparian habitats < 90 percent of the time but more often than in nonriparian or other wetland habitats;

W = may occur in riparian habitats but often occurring in other wetland types, e.g., marshes, open water, or openings near water.

Although most species nest in the habitat in which they have been listed they may forage in or over the vegetation types listed below.

			-------------- Vegetation type --------------			
Common name	**Scientific name**[b]	**Riparian dependency code**	***Tamarix***[c]	**Mesquite**[d]	**Cottonwood-Willow**[e]	**Mixed-Deciduous**[f]
Abert's towhee	*Melozone aberti*	O	2,11,15	X	X	
Acorn woodpecker	*Melanerpes formicivorus*					X
American coot	*Fulica americana*	* W	6,13,16	X		
American dipper	*Cinclus mexicanus*	* O	12,17,21			
American kestrel	*Falco sparverius*	* P	4,18	X	X	X
American robin	*Turdus migratorius*	P			X	X
Anna's hummingbird	*Calypte anna*	P		X	X	X
Arizona woodpecker	*Picoides stricklandi*					X
Ash-throated flycatcher	*Myiarchus cinerascens*	* F	2,3,15,16	X	X	X
Bald eagle	*Haliaeetus leucocephalus*	O	6,13	X	X	
Barn owl	*Tyto alba*		2	X	X	X
Bell's vireo[k]	*Vireo bellii*	* P	1,11,15	X	X	X
Belted kingfisher	*Megaceryle alcyon*	O	12,17			
Bendire's thrasher	*Toxostoma bendirei*	F		X		
Bewick's wren	*Thryomanes bewickii*	*	2,10,11	X	X	X
Black phoebe	*Sayornis nigricans*	* O	6,12,15	X	X	X
Black rail	*Laterallus jamaicensis*	O W	21	X		
Black-caped gnatcatcher	*Polioptila nicriceps*	P		X	X	X
Black-chinned hummingbird	*Archilochus alexandri*	* P	1,2,17,18	X	X	X
Black-crowned night-heron	*Nycticorax nycticorax*	* O	1,12,13	X	X	
Black-headed grosbeak	*Pheucticus melanocephalus*					X
Black-tailed gnatcatcher	*Polioptila melanura*	* P	4,12,22	X	X	
Black-throated sparrow	*Amphispiza bilineata*			X		
Blue grosbeak	*Passerina caerulea*	* O	1,2,3,15	X		X
Blue-gray gnatcatcher	*Polioptila caerulea*	*	1,5	X	X	
Botteri's Sparrow	*Peucaea botterii*	F		X	X	X

Table 3—*Continued.*

Common name	Scientific name[b]	Riparian dependency code	Vegetation type: *Tamarix*[c]	Mesquite[d]	Cottonwood-Willow[e]	Mixed-Deciduous[f]
Bridled titmouse	*Baeolophus wollweberi*	O			X	X
Broad-billed hummingbird	*Cynanthus latirostris*	P		X	X	X
Bronzed cowbird	*Molothrus aeneus*	O		X	X	
Brown thrasher	*Toxostoma rufum*		12,17,20		X	
Brown-crested flycatcher	*Myiarchus tyrannulus*	P	12,18	X	X	X
Brown-headed cowbird	*Molothrus ater*	* F	1,2,3,16	X	X	X
Bullock's oriole	*Icterus bullockii*	* O	2,3,15,18	X	X	X
Cactus wren	*Campylorhynchus brunneicapillus*	*	3,12,15	X		
Canyon towhee	*Melozone fusca*	*	18	X		
Canyon wren	*Catherpes mexicanus*	* F		X		
Cassin's kingbird	*Tyrannus vociferans*	F	18		X	X
Cattle egret	*Bubulcus ibis*	P	17,20,21	X		
Chihuahuan raven	*Corvus cryptoleucus*		12,17,18	X	X	X
Clapper rail	*Rallus crepitans*	O W	21	X		
Clark's grebe	*Aechmophorus clarkii*	* W	5			
Cliff swallow[l]	*Petrochelidon pyrrhonota*	* O	4,12,17	X	X	X
Common black hawk	*Buteogallus anthracinus*	O		X	X	X
Common ground-dove	*Columbina passerina*	P	18,21	X		
Common moorhen	*Gallinula chloropus*	W	6,12,13	X		
Common poorwill	*Phalaenoptilus nuttallii*			X		
Common raven	*Corvus corax*	*	4,10,18	X	X	X
Common yellowthroat	*Geothlypis trichas*	* O W	11,12,15	X	X	X
Cooper's hawk	*Accipiter cooperii*	* O	18	X	X	X
Costa's hummingbird	*Calypte costae*	*	14	X		
Couch's kingbird	*Tyrannus couchii*				X	
Crissal thrasher	*Toxostoma crissale*	* P	3,9,12,15	X		
Curve-billed thrasher	*Toxostoma curvirostre*	F	7	X		
Double-crested Cormorant	*Phalacrocorax auritus*	* O	21	X	X	
Dusky-capped flycatcher	*Myiarchus tuberculifer*					X
Eastern bluebird	*Sialia sialis*				X	X
Elf owl	*Micrathene whitneyi*	P	12,20,17	X	X	X
European starling	*Sturnus vulgaris*	* P	4	X	X	X
Ferruginous pygmy-owl	*Glaucidium brasilianum*	P		X	X	
Gambel's quail[g]	*Callipepla gambelii*	* P	6,11,13	X	X	X
Gila woodpecker	*Melanerpes uropygialis*	P	4,15,18	X	X	X

Table 3—*Continued.*

Common name	Scientific name[b]	Riparian dependency code	-------------- Vegetation type -------------- *Tamarix*[c]	Mesquite[d]	Cottonwood-Willow[e]	Mixed-Deciduous[f]
Gilded flicker	*Colaptes chrysoides*	F	4	X	X	
Golden-fronted woodpecker	*Melanerpes aurifrons*	P				
Gray hawk	*Buteo plagiatus*	O	18	X	X	X
Great blue heron	*Ardea herodias*	O	4,6	X	X	
Great egret	*Ardea alba*	O	21			
Great horned owl	*Bubo virginianus*	*	4	X	X	X
Great-tailed grackle	*Quiscalus mexicanus*	* O	11,15			
Greater roadrunner	*Geococcyx californianus*	* F	3,12,22	X	X	X
Green heron	*Butorides virescens*	O	6,13		X	
Harris hawk	*Parabuteo unicinctus*	P		X	X	
Hooded oriole	*Icterus cucullatus*	* P		X	X	X
House finch F	*Haemorhous mexicanus*	* F	11,14,15	X	X	X
House sparrow	*Passer domesticus*	*		X	X	
Hutton's vireo	*Vireo huttoni*	F				X
Indigo bunting	*Passerina cyanea*	* O	12,17	X	X	X
Killdeer	*Charadrius vociferus*	W P	6,14,16	X		
Ladder-backed woodpecker	*Picoides scalaris*	* P	2,3,15,16	X	X	X
Lazuli bunting	*Passerina amoena*	* O	12,17		X	X
Least bittern	*Ixobrychus exilis*	W	5,10	X	X	
Lesser goldfinch	*Spinus psaltria*	* P	11,15,18	X	X	X
Lesser nighthawk	*Chordeiles acutipennis*	*	4,6,13	X	X	X
Loggerhead shrike	*Lanius ludovicianus*		3,21	X		
Long-billed thrasher	*Toxostoma longirostre*			X		
Lucy's warbler	*Oreothlypis luciae*	* O	1,2,15,18	X	X	X
Marsh wren	*Cistothorus palustris*		4,5			
Meadowlark	*Sturnella* sp.		7			
Mexican jay	*Aphelocoma wollweberi*					X
Mississippi kite	*Ictinia mississippiensis*	O	19,21	X	X	
Montezuma quail	*Cyrtonyx montezumae*					X
Mourning dove	*Zenaida macroura*	* P	2, 6,11,13	X	X	X
Northern beardless-tyrannulet	*Camptostoma imberbe*	O	18	X	X	X
Northern cardinal	*Cardinalis cardinalis*	P	9,18	X	X	
Northern flicker	*Colaptes auratus*			X	X	X
Northern mockingbird	*Mimus polyglottos*	P	3,11,7,18	X	X	X

Table 3—*Continued.*

Common name	Scientific name[b]	Riparian dependency code	---- Vegetation type ---- *Tamarix*[c]	Mesquite[d]	Cottonwood-Willow[e]	Mixed-Deciduous[f]
Northern rough-winged swallow	*Stelgidopteryx serripennis*	P	6,12,13	X	X	X
Orchard oriole	*Icterus spurius*	P		X	X	
Painted bunting	*Passerina ciris*		22			
Peregrine falcon[j]	*Falco peregrinus*	* P	5			
Phainopepla	*Phainopepla nitens*	* P	4,14,18	X	X	X
Pied-billed grebe	*Podilymbus podiceps*	W	5,6,10			
Prairie falcon	*Falco mexicanus*	*				
Purple martin	*Progne subis*			X		
Pyrrhuloxia	*Cardinalis sinuatus*	P	3,12,22	X		
Red-tailed hawk	*Buteo jamaicensis*	*	14,21	X	X	X
Red-winged blackbird	*Agelaius phoeniceus*	* W P	2,8,14,16	X		
Ring-necked pheasant	*Phasianus colchicus*		7			
Rose-throated becard	*Pachyramphus aglaiae*	O			X	X
Ruddy duck	*Oxyura jamaicensis*	W	5,13			
Rufous-winged sparrow	*Peucaea carpalis*	P		X		
Say's phoebe	*Sayornis saya*		14,15,16			
Snowy egret	*Egretta thula*	O	21			
Song sparrow	*Melospiza melodia*	* O W	2,6,10,15	X	X	X
Sora	*Porzana carolina*	O W	5	X		
Spotted sandpiper	*Actitis macularius*	* W	1,21		X	
Spotted towhee	*Pipilo maculatus*		7			
Sulphur-bellied flycatcher	*Myiodynastes luteiventris*	O				X
Summer tanager	*Piranga rubra*	* O	2,9,12,17	X	X	X
Swainson's hawk	*Buteo swainsoni*		21	X		
Thick-billed kingbird	*Tyrannus crassirostris*	P	21			
Tricolored blackbird	*Agelaius tricolor*	W P	12,17,20			
Tropical kingbird	*Tyrannus melancholicus*	P	9	X	X	
Turkey vulture	*Cathartes aura*	*	4	X	X	X
Varied bunting	*Passerina versicolor*	F	18	X		
Verdin	*Auriparus flaviceps*	P	2,14,15	X		
Vermilion flycatcher	*Pyrocephalus rubinus*		2,18	X	X	X
Violet-green swallow	*Tachycineta thalassina*	*				X
Virginia rail	*Rallus limicola*	O W	5	X		
Western grebe[h]	*Aechmophorus occidentalis*	* W	5			

Table 3—*Continued.*

Common name	Scientific name[b]	Riparian dependency code	-------------- Vegetation type --------------			
			Tamarix[c]	Mesquite[d]	Cottonwood-Willow[e]	Mixed-Deciduous[f]
Western kingbird	*Tyrannus verticalis*	* F	3,15,18	X	X	X
Western meadowlark	*Sturnella neglecta*		4			
Western screech-owl	*Megascops kennicottii*	* P	2	X	X	X
Western wood-pewee	*Contopus sordidulus*	O		X	X	X
White-breasted nuthatch	*Sitta carolinensis*					X
White-faced ibis	*Plegadis chihi*	W	12,20,17			
White-throated swift	*Aeronautes saxatalis*	*	5			
White-winged dove	*Zenaida asiatica*	P	2,6,8,13	X	X	X
Willow flycatcher	*Empidonax traillii*	* O	9,17		X	
Yellow warbler	*Setophaga petechia*	* O	1,2,15,18	X	X	X
Yellow-billed cuckoo[i]	*Coccyzus americanus*	* O	2,3,7,17	X	X	X
Yellow-breasted chat	*Icteria virens*	* O	1,13,15	X	X	
Yellow-headed blackbird	*Xanthocephalus xanthocephalus*	O W	4,5,12,17			
Zone-tailed hawk	*Buteo albonotatus*	P		X	X	X
*58 Grand Canyon	Total species: 143		105	98	81	67
			Total	Total	Total	Total

[a] After Johnson et al. (1977, 1987) with modifications from information gathered since those publications. Several species of cavity nesters are obligate or preferential riparian nesters except for using saguaros (*Carnegiea gigantea*), e.g., western screech-owl, elf owl, Gila woodpecker, gilded flicker, and American kestrel.

[b] Common and scientific names after the 2017 checklist of North and Middle American birds by the North American Classification Committee (NACC) of the American Ornithologists' Union.

[c] Tamarisk or saltcedar; after 3 or 4 references are listed for a given species additional references are not cited.

[d] Velvet (*Prosopis velutina*), honey (*P. glandulosa*), or screwbean mesquites (*P. pubscens*).

[e] Fremont (*Populus fremontii*) and plains cottonwoods (*P. deltoides*), and Goodding willow (*Salix gooddingii*).

[f] Largely Arizona (*Platanus wrightii*) and California sycamores (*P. racemosa*); ash (*Fraxinus* spp.), and walnut (*Juglans* spp.).

[g] Although numerous historic records of Gambel's quail the species was extirpated by the mid-1900s.

[h] Western and Clark's grebes have been recently separated so there is confusion about which occurs (one or both?) on upper Lake Mead at the lower end of Grand Canyon.

[i] Formerly the Yellow-billed cuckoo was a rare summer resident in the Canyon but last recorded in 1971 (Brown et al. 1987).

[j] Peregrines commonly nest on cliffs close to water from where they hunt for birds on and over water.

[k] Bell's vireo has progressed steadily upstream in *Tamarix* thickets since the construction of upstream Glen Canyon Dam; one of the most noticeable species because of its persistent song (Brown et al. 1983).

[l] The cliff swallow was formerly a common breeding species along the Colorado River in Grand Canyon but by the mid-1970s it was extirpated, apparently due to sediment being entrapped by upstream Glen Canyon Dam, thus a lack of mud for nest-building (Brown et al. 1987).

[m] The overall period covered was from 1978 to 2012 with the states of CA, AZ, NM, TX, and NV all represented and although the same locations were sometimes sampled more than once each sampling listed different species.

[n] van Riper et al. (2008) does not list how species use riparian habitat or differentiate between use of *Tamarix* and other riparian vegetation for each species.

[o] https://birdsna.org/Species-Account/bna/species/

Table 3—*Continued.*

I. Tamarix citations[m]	**Rivers and states**
1. Brown et al. (1987)	Colorado R. in Grand Canyon, AZ
2. Rosenberg et al. (1991)	Lower Colorado R. Valley, AZ & CA
3. Hunter et al. (1988)	Pecos, Rio Grande, lower Colorado rivers, TX, AZ & CA
4. Anderson (2017)	Lower Colorado R. Valley, AZ & CA
5. R. Roy Johnson (Pers. observ.),	unpublished Various, SW U.S.
6. Johnson and Simpson (MS b)	Salt R., AZ
7. Livingston and Schemnitz (1996).	Pecos R., NM
8. Rea (2007).	Gila R., AZ
9. Hunter et al. (1987)	Colorado, Gila, Pecos, Rio Grande, Salt, San Pedro, Santa Cruz, Verde rivers, TX, AZ & CA
10. Johnson et al. (2000)	Lower Salt River, AZ
11. Engel-Wilson and Ohmart (1979)	Rio Grande, TX
12. Sogge et al. (2008)	Various, SW U.S.
13. Johnson and Simpson (MS a)	Gila, Salt, Verde rivers, AZ
14. Cardiff et al. (1978a)	Mojave R., CA
15. van Riper et al. (2008)[n]	Lower Colorado R., AZ & CA
16. Cardiff et al. (1978b)	Mojave R., CA
17. Paxton et al. (2011)	Southwestern U.S.
18. Brand et al. (2008)	San Pedro R., AZ
19. Glinski and Ohmart (1983)	San Pedro R., AZ
20. Birds of North America accounts[o]	Various, SW U.S.
21. Corman and Wise-Gevais (2005)	Arizona rivers, AZ
22. Hunter et al. (1985)	Colorado, Pecos, Rio Grande rivers, SW U.S.

II. Mesquite citations: (Arnold 1940; Bendire 1872, 1892; Brandt 1951; Brown 1987; Brown et al. 1984; Carothers and Brown 1991; Carothers and Sharber 1976; Carothers et al. 1976; Corman and Wise-Gervais 2005; Dawson 1921; Gavin and Sowles 1975; Glinski and Ohmart 1983; Huels et al. 2013; Hunter et al. 1987; Johnson and Simpson Ms A, Ms B; Johnson et al. 2000; Monson and Phillips 1981; Ohmart et al. 1988; Phillips et al. 1964; Rea 1983, 2007; Rosenberg et al. 1991; Stamp 1978; Webb et al. 2014; Willson and Carothers 1979).

III. Cottonwood-willow citations: (Brandt 1951; Carothers and Johnson 1976; Carothers et al. 1974; Corman and Wise-Gervais 2005; Engel-Wilson and Ohmart 1979; Glinski and Ohmart 1983; Hunter et al. 1987; Johnson and Simpson Ms A, Ms B; Johnson et al. 2000; Monson and Phillips 1981; Ohmart et al. 1988; Phillips et al. 1964; Rea 1983, 2007; Rosenberg et al. 1991; Stamp 1978; Webb et al. 2014).

IV. Mixed deciduous citations: (Bock and Bock 1984; Carothers et al. 1974; Monson and Phillips 1981; Phillips et al. 1964).

a 10-fold higher biomass of invertebrates compared with the co-occurring native willow (*Salix exigua*) (Stevens 1985). Most of the invertebrate biomass increase found in *Tamarix* reported by Stevens (1985) was due to high density of the *Tamarix* host-specific, nonnative leaf hopper (*Opsius stactogalus*)—a species that Grand Canyon birds commonly feed upon during the breeding season (Yard et al. 2004). In addition, *Tamarix* is a rapid invader and fast to grow in conditions otherwise marginal for most native riparian species. These conditions include inflows and outflows to reservoirs like Lakes Mead and Powell, where annual tailwater flows are severely reduced from pre-dam flows, and where receding reservoirs expose bare ground.

The unintended and unexpected consequence of a major reduction in wildlife habitat caused by the rapid and widespread loss of *Tamarix*-dominated habitat in many areas has now been well documented (Bateman et al. 2015, 2013b; Darrah and van Riper 2017; McLeod volume 1; Paxton et al. 2011; Sogge et al. 2008, 2013; van Riper et al. 2008; Yard et al. 2004). Sogge et al. (2008) also determined that there were no negative effects from breeding in *Tamarix* habitats expressed in several studies on the southwestern willow flycatcher (*Empidonax traillii extimis*). This finding contradicted the conventional wisdom that birds that nested in *Tamarix* had lower reproductive fitness as measured in nest and fledgling survival (see Brand et al. 2008; DeLoach et al. 2000). In Grand Canyon, the earliest southwestern willow flycatcher breeding record (1936) reported by Woodbury and Russell (1945) in the Lees Ferry area was in *Tamarix*, and every nest found along the Colorado River in Grand Canyon by Brown (1988) was also in *Tamarix* (table 4)[2]. Sogge et al. (2008) warned that the overall ecological costs and benefits of *Tamarix* control were difficult to predict and that restoration projects that resulted in removal of *Tamarix* without replacement with high quality habitat had the potential to reduce net riparian habitat value for local and/or regional bird populations. Their predictions were prophetic.

Darrah and van Riper (2017) proposed that the impacts to riparian bird species resulting from *Diorhabda* biocontrol will continue until regrowth of native vegetation is established. Whereas they recognized that active restoration may be necessary in some areas, they did not discuss the challenges that will be faced by private, State, and Federal organizations in providing human and financial resources necessary for active habitat restoration[3] on thousands of acres of *Tamarix* habitat destroyed by the beetle. One estimate for comprehensive eradication and restoration of *Tamarix*-dominated habitat prior to biocontrol concluded that, although costs per acre are difficult to average depending on local site conditions and other factors, they normally reach thousands of dollars per acre (Zavaleta 2000).

An early review of the potential for habitat restoration following anticipated successful biocontrol questioned whether the habitat that occurs following *Tamarix* control and revegetation was any better for wildlife than the original habitat (Shafroth et al. 2005). Indeed, in their review, Shafroth et al. (2005) observed that following *Tamarix* control efforts, failure to plan and implement restoration efforts could result in recolonization of a site by other exotic species. This prediction has been largely verified by Gonzales et al. (2017) in their study of 244 sites where *Tamarix* was removed or disadvantaged by one means or another. In another study of a restoration project on Las Vegas Wash in Nevada where *Tamarix* was removed from several sites, replaced by native trees and shrubs, and then monitored for avian use, benefits to birds were not as evident as predicted (Shanahan et al. 2011). However, the apparent lack of wildlife benefits in *Tamarix* removal projects reviewed to date may be partially a consequence of the assessments being too early after control impacts. There may have been insufficient time for revegetation following the defoliation and mortality of the exotic species.

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[2] Nesting records for the endangered southwestern willow flycatcher are important not only because of the rarity of the species but because of its nesting in *Tamarix* in Grand Canyon rather than in willows or other native species. The flycatcher had formerly occurred throughout much of the state (Phillips et al. 1964) but had largely disappeared by the 1970s. For example, 34 nests were collected along the Colorado River in the Yuma area in 1902 by Herbert Brown (Unitt 1987) but the species was later extirpated as a breeding bird for that region (Rosenberg et al. 1991). Monson and Phillips (1981) wrote, "No nests found [in Arizona] since 1970," obviously unaware of our nesting records (table 2) and those published by Carothers and Johnson (1975). Thus, the southwestern willow flycatchers nesting in Grand Canyon during the 1970s and into the 1980s represented the only known breeding population for the state at that time.

[3] We are distinguishing between *passive* restoration and *active* restoration as defined by Shafroth et al. 2013; i.e.: "Passive restoration refers to facilitating the return of desirable system dynamics and species composition by removing one or more underlying stressor(s). Active restoration approaches include manipulating a site to prepare it for restoration; revegetating the site by introducing seeds, transplant stock, or cuttings; or irrigating or otherwise manipulating the site to enhance recovery" (Shafroth et al. 2013, p. 411).

Table 4—Summer records of the endangered southwestern willow flycatcher (*Empidonax traillii extimus*) from the Colorado River in Grand Canyon prior to 1989; see especially Brown (1988).

Date	Location	Observation	Observer and reference
June 7, 1933	RM 2[1]	Female collected	Woodbury and Russell (1945)
August 18, 1936	Near Lees Ferry	Nest collected from *Tamarix*	Woodbury and Russell (1945)
June 16, 1953	Lees Ferry	Specimen collected	R.W. Dickerman; Monson (1953)
June 17, 1953	Mouth of Little Colorado River	Specimen collected	R.W. Dickerman; Monson (1953)
July 12, 1971	Cardenas Marsh	Nest with eggs in Goodding willow	Johnson; Carothers and Johnson (1975)
July 27, 1971	Cardenas Marsh	1 pair	Carothers; Carothers and Johnson (1975)
May 20, 1974	Cardenas Marsh	1 pair	Johnson; Carothers and Johnson (1975)
1974-1976	225 mi of Colorado R.	1 known pair	Carothers et al. 1976; Carothers and Sharber (1976)
1974-1976	225 mi of Colorado R.	1 known pair	Carothers et al. 1976; Carothers and Sharber (1976)
1982	Saddle Can. to Nankoweap Crk.	1 singing male	Brown (1988)
1982	Cardenas Marsh	1 singing male	Brown (1988)
1983	Saddle Can. to Nankoweap Crk.	4 singing males	Brown (1988)
1984	Saddle Can. to Nankoweap Crk.	3 singing males, 2 nests* In *Tamarix*	Brown (1988)
1984	Cardenas Marsh	1 singing male	Brown (1988)
1985	Saddle Can. to Nankoweap Crk.	7 singing males, 4 nests* In *Tamarix*	Brown (1988)
1985	Cardenas Marsh	1 singing male	Brown (1988)
1986	Saddle Can. to Nankoweap Crk.	8 singing males 2 nests* In *Tamarix*	Brown (1988)
1986	Nankoweap Crk. to Kwagunt Crk.	1 singing male	Brown (1988)
1986	Cardenas Marsh	2 singing males	Brown (1988)
1987	Saddle Can. to Nankoweap Crk.	4 singing males 2 nests* In *Tamarix*	Brown (1988)
1987	Cardenas Marsh	3 singing males	Brown (1988)

[1] River Miles downstream from Lees Ferry.

*Exact location of nests not given by Brown (1988).

Whereas biocontrol appears to be satisfying the long-desired eradication of this invasive species much more quickly than expected, there is little evidence at this time that native riparian vegetation will, without active restoration, eventually colonize most areas as originally assumed (DeLoach and Tracy 1997). Beetle-release proponents admitted early in the development of the biocontrol that evidence for native vegetation species to replace *Tamarix* was circumstantial and not well supported (see McLeod volume 1) and clearly underestimated the value of the *Tamarix* to native riparian wildlife species. Recent revegetation efforts attempted along the Colorado River in Glen Canyon indicate that once *Tamarix* is gone, reestablishment of native woody vegetation can be rapid and effective, but only coincident with labor intensive planting, fencing, and watering (Stevens et al. 2015).

However, in other areas where active restoration has not been implemented after *Tamarix* removal, passive revegetation is almost exclusively limited to recolonization by a mixture of native and nonnative grasses and herbaceous cover with the conspicuous absence of woody species (González et al. 2017; Sher et al. 2018). In a study in Grand Canyon tributaries, Belote et al. (2010) found that 1 to 3 years after mechanical removal of *Tamarix*, there was no recruitment or increase in the number of native species; however, there was approximately a 50 percent decline in precipitation between pre- and post-restoration periods that could have prevented a greater success in restoration of native species.

Tamarix-Dominated Riparian Habitat In Grand Canyon: A New Wildlife Resource

Prior to the closing of the flood gates of Glen Canyon Dam in 1963 and creation of Lake Powell, annual flood scour in the river channel precluded riparian growth in all but a narrow margin at the edge of the high water line. Composed of honey mesquite (*Prosopis glandulosa*), netleaf hackberry (*Celtis reticulata*), four species of Baccharis, and western redbud (*Cercis occidentalis*), this narrow band of vegetation constituted the extent of riparian habitat in the river corridor. Below this narrow band of high water line vegetation, the annual scour zone was largely devoid of woody vegetation and composed of annual forbs and grasses that could establish between flood flows. After the dam began to hold back annual highwater and annual scouring floods were largely controlled, a *Tamarix*-dominated vegetation zone became established in the river corridor. This included *Tamarix* (*T. chinensis/ramosissima*) intermixed with native species, arrowweed (*Pluchea sericea*), *Baccharis* spp., coyote willow (*Salix exigua*), and Goodding's willow (*Salix gooddingii*) where previously only forbs and grasses occurred. Figures 1 and 2 provide a graphic representation of the riparian vegetation condition of the river corridor in Grand Canyon before and after the Dam. Scott et al. (volume 1) demonstrate these vegetation changes with photographic comparisons.

By the time Glen Canyon Dam had been in place for 20 years (1963-1983), natural resources studies below the Dam were focused on attempting to quantify changes that dam-controlled flow had on aquatic and terrestrial ecosystems within Grand Canyon National Park. At that time, approximately 300 miles of river corridor from the dam to Lake Mead were under control and management of the National Park Service, but dam releases were under control of the Bureau of Reclamation and fluctuated according to power needs by cities. Also, during this time, researchers first recognized that the terrestrial *Tamarix*-dominated habitat was becoming increasingly more utilized by river corridor wildlife in Grand Canyon. The New High Water Zone (NHWZ; see fig. 2) habitat was recognized as a boon to native wildlife species.

Within only two decades of the closing of the flood gates at Glen Canyon Dam in 1963, *Tamarix*-dominated habitat below the dam became remarkably productive for native vertebrates and invertebrates and some nonnative invertebrates as well (Brown et al. 1987; Stevens 1985; Yard et al. 2004). The NHWZ vegetation introduced an extensive "new" river margin zone of vegetation along the river corridor from Glen Canyon Dam to the inflow of Lake Mead. Moreover, studies soon demonstrated that the NHWZ not only increased the overall density of canyon birds, but also contributed to the range expansions of several avian species that invaded from the south where native habitats and some avian species were in rapid decline (Brown et al. 1987; Rosenberg et al. 1991). Table 5 presents a list of the birds and mammals that have colonized or experienced range extensions into *Tamarix*-dominated habitat along the Colorado River in Grand Canyon since the closing of Glen Canyon Dam floodgates.

Remarkably, it soon became obvious that NHWZ was expanding rapidly and that it was significantly more productive and robust in vegetative growth and in carrying capacity for riparian birds and other vertebrates and invertebrates than the native mesquite-dominated Old High Water Zone (OHWZ; fig. 1) (Brown and Trosset 1989; Carothers and Brown 1991; Holmes et al. 2005; Ruffner et al. 1978; Stevens 1995; Yard et al. 2004). Studies targeted specifically on OHWZ tree/shrub species indicated that plants in this zone were slowly senescing, presumably due to the absence of flows no longer reaching the OHWZ habitats \(Anderson and Ruffner 1987, 1988).

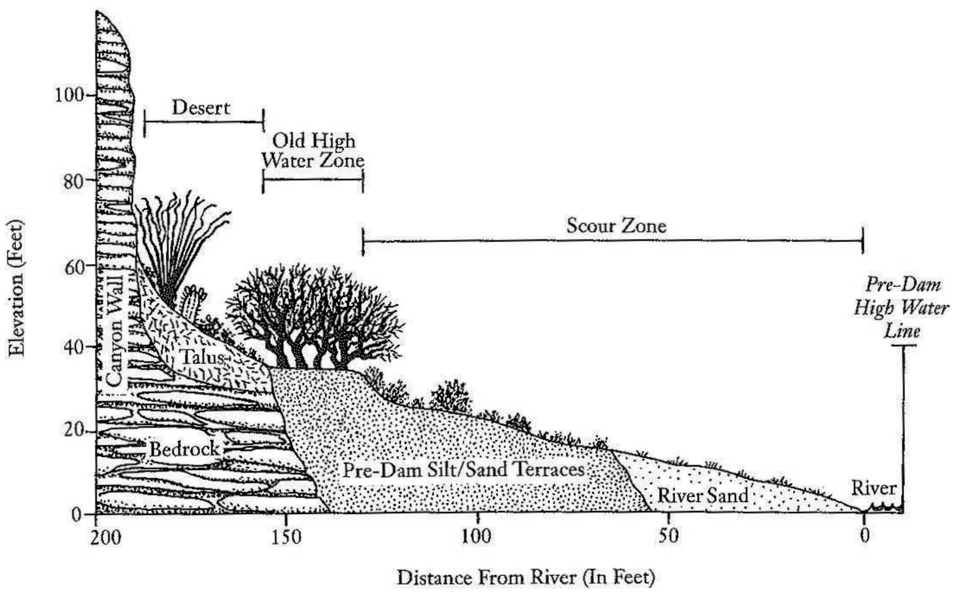

Figure 1—Pre-dam floods normally scoured the lower-lying river terraces of all woody vegetation, allowing only a few annual grasses and herbs to develop in the pre-dam flood zone. From *Grand Canyon Birds* by Bryan T. Brown, Steven W. Carothers, and R. Roy Johnson, © 1987 The Arizona Board of Regents. Reprinted by permission of the University of Arizona Press.

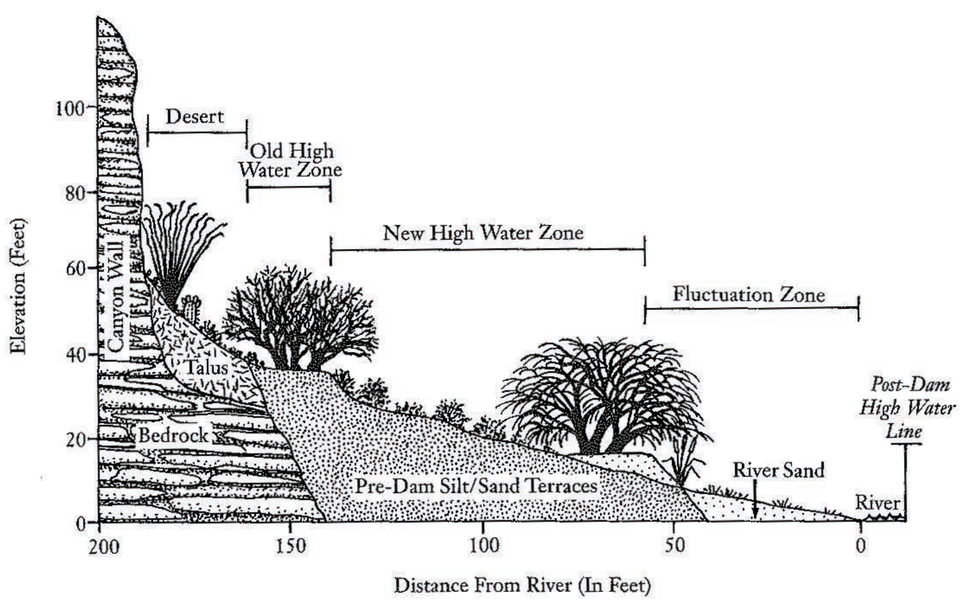

Figure 2—Beginning in 1963, Glen Canyon Dam prevented the annual scouring floods. A new zone of riparian vegetation, dominated by the exotic *Tamarix*, developed in the pre-dam flood zone and was quickly colonized by riparian breeding birds and other native vertebrates. From *Grand Canyon Birds* by Bryan T. Brown, Steven W. Carothers, and R. Roy Johnson, © 1987 The Arizona Board of Regents. Reprinted by permission of the University of Arizona Press.

Table 5—Birds and mammals that have colonized (newly arrived species) or experienced range extensions (moved upriver from downstream breeding locations) into *Tamarix*-dominated habitat along the Colorado River in Grand Canyon since the closing of Glen Canyon Dam floodgates in 1963 (modified from Johnson and Carothers 1987).

Common name	Scientific name	Status and notes
	Birds	
Western grebe[a]	*Aechmophorus occidentalis*	Colonized, nesting in dead *Tamarix* at upper Lake Mead
Clark's grebe[a]	*A. clarkii*	Colonized, nesting in dead *Tamarix* at upper Lake Mead
Black-chinned hummingbird	*Archilochus alexandri*	Colonized
Green heron	*Butorides virescens*	Nonbreeding
Black-crowned night-heron	*Nycticorax nycticorax*	Colonized, nesting in dead *Tamarix* at upper Lake Mead
Willow flycatcher	*Empidonax traillii*	Colonized, endangered species
Vermilion flycatcher	*Pyrocephalus rubinus*	Range expansion
Bell's vireo	*Vireo bellii*	Range expansion
Lesser goldfinch	*Spinus psaltria*	Colonized
Yellow-breasted chat	*Icteria virens*	Colonized
Hooded oriole	*Icterus cucullatus*	Range expansion
Bullock's oriole	*I. bullockii*	Colonized
Red-winged blackbird	*Agelaius phoeniceus*	Colonized
Brown-headed cowbird	*Molothrus ater*	Colonized
Great-tailed grackle	*Quiscalus mexicanus*	Range expansion
Lucy's warbler	*Oreothlypis luciae*	Colonized
Common yellowthroat	*Geothlypis trichas*	Colonized
Yellow warbler	*Setophaga petchia*	Colonized
Summer tanager	*Piranga rubra*	Range expansion
Blue grosbeak	*Passerina caerulea*	Colonized
Lazuli bunting	*P. amoena*	Colonized
Indigo bunting	*P. cyanea*	Range expansion
	Mammals	
Beaver	*Castor canadensis*	Colonized
Brush mouse	*Peromyscus boylei*	Colonized
Deer mouse	*P. maniculatus*	Colonized

[a]The western grebe was recently split into two species and it is not known if only one or both are nesting here.

Table 6—Comparison of species richness and population densities for spring and summer breeding birds showing preferences for *Tamarix*-dominated habitat relative to mesquite-dominated habitat along the Colorado River in Grand Canyon; after Brown (1987) and Brown and Trosett (1989).

Type of study plot and year	Total species	Species/site	Average species/site	Population (densities/site pairs/40 ha)	Average (density/site pairs/40 ha)
Tamarix 1984	21	1-15	7.9	200-1,200	611
Tamarix 1985	23	1-16	8.0	100-1,200	565
Prosopis 1984	20	2-14	8.6	182-986	449
Prosopis 1985	20	3-17	8.0	73-943	379
Prosopis 1985	20	3-17	8.0	73-943	379

[1] River Miles downstream from Lees Ferry.

*Exact location of nests not given by Brown (1988).

Table 6 provides a summary of the number of species and relative density of birds found in both the *Tamarix*- and mesquite-dominated habitats (NHWZ vs. OHWZ, respectively). While species richness was similar in both habitats with 20-23 nesting species, surprisingly, and against all conventional wisdom at the time, avian density in the *Tamarix*-dominated habitat was significantly greater than in the mesquite-dominated habitats with similar cover and height attributes (Brown 1987; Willson and Carothers 1979). Additionally, most species demonstrated a preference for *Tamarix* as nesting habitat. For example, black-chinned hummingbird (*Archilochus alexandri*), blue-gray gnatcatcher (*Polioptila caerulea*), Lucy's warbler (*Oreothlypis luciae*), yellow warbler (*Setophaga petechia*), and common yellowthroat (*Geothlypis trichas*) occurred most consistently in the *Tamarix*-dominated NHWZ, with these five species comprising 51.5 percent of the total density and 31 percent of the breeding species in 1985 at the inner canyon study sites.

Lucy's warbler and black-chinned hummingbird were the most abundant and widespread species in *Tamarix*-dominated habitats during 1984 and 1985 and remarkably, of 24 Lucy's warbler nests found during that time, 15 (62.5 percent) were in *Tamarix*. What is most remarkable about the Lucy's warbler nests in *Tamarix* is the fact that the warbler is primarily a cavity-nesting species and *Tamarix* is not known for an abundance of cavities. The warblers were all found forming a cavity nest (pseudo-cavity) within the clumps of leaf litter debris caught in the forks of upper canopy branches (Johnson et al. 1997). In addition, all 12 nests of the southwestern willow flycatcher in the Grand Canyon were in *Tamarix* (Brown 1988).

The overriding preference for birds nesting in *Tamarix* in the Grand Canyon led Brown and Trosset (1989) to conclude that *Tamarix*-dominated habitats can be the ecological equivalent of native communities that are required habitat of some obligate riparian nesting birds. This is especially true in a situation like Grand Canyon where the NHWZ vegetation community did not displace a community composed of native plants but became established where previously there was no nesting habitat.

While the impact of Glen Canyon Dam on terrestrial habitats of the river corridor clearly had created habitat for native species where previously habitat did not exist, dam-related changes to the aquatic ecosystem were not as favorable to the native aquatic species, especially the native fish. The river had been dramatically changed from an aquatic ecosystem driven by an annual hydrograph with seasonal flow periodicity, sediment laden flows, and dramatically fluctuating water temperatures to a relatively steady flow, sediment starved, perpetually cold system that was largely incapable of nurturing the native aquatic species (Carothers and Brown 1991; Gloss et al. 2005; Johnson and Carothers 1987). Within the National Park, the post-dam environment bore no semblance to the pre-dam river. One of the major findings from research on changes to the aquatic and terrestrial ecosystems of the riparian corridor below Glen Canyon Dam concluded that the exotic *Tamarix* species had become a wildlife-valuable "naturalized" element of the land-based ecosystem. This *Tamarix*-based ecosystem has been termed a naturalized ecosystem, supporting high densities of native vertebrates, while the aquatic ecosystem was termed an exotic ecosystem due to the loss of indigenous species and addition of numerous exotics (Johnson and Carothers 1987).

Management of *Tamarix* in Grand Canyon: A Dilemma

Describing the dilemma of resource management in a National Park, where intense efforts are normally expended to eliminate all nonnative plants and animals, Johnson and Carothers (1987) opined on the futility of attempting to remove *Tamarix* from Grand Canyon. They argued for accepting and managing the relatively new and wildlife-supporting *Tamarix*-dominated riparian habitat as a "naturalized" community. Johnson and Carothers (1987) described the dam-influenced habitat as follows:

"… a naturalized ecosystem contains biotic communities with both indigenous and exotic plants and/or animals. In these communities, dominance or predominance is not a function of species origin (that is, native or nonnative), and the indigenous biota is not threatened either in species richness or population sizes by exotic species. In naturalized ecosystems biotic and abiotic processes have either reached or are evolving toward an equilibrium in which exotics do not restrict or interfere with native organisms or ecological processes, rather than evolving toward the destruction of components and processes of the original, natural ecosystem. If native species are extirpated or their populations are greatly reduced, the ecosystem cannot be considered naturalized. In Grand Canyon new post-dam riparian vegetation has led to larger populations of native species and generally has been beneficial to wildlife as well as recreationists."

In Grand Canyon, the "new" habitat became so productive for wildlife, especially birds, small mammals, reptiles, and amphibians that the National Park Service (NPS) eventually recognized the new vegetation zone as a naturalized ecosystem that would never be returned to its natural state as long as Glen Canyon Dam controlled the river (Sharrow 1990). As such, the NPS recognized the new riparian habitat as a wildlife resource and made no efforts to remove or attempt to control *Tamarix* in the dam-influenced river corridor.

The *Tamarix*-dominated riparian zone in the Grand Canyon is an unusual situation compared to most areas that are invaded by the exotic plant. In general, conditions allowing colonization by and proliferation of *Tamarix* are varied. Contrary to popular opinion, *Tamarix* is a weak competitor compared with native riparian species in systems where the stream hydrograph is largely unaltered (Sher et al. 2000; Stromberg et al. 2007). Rarely does *Tamarix* have an opportunity to colonize when perennial stream systems are in their native state (hydroriparian ecosystems), except where portions of a stream are naturally intermittent (Johnson et al. 1984).

Sometimes even minor changes in groundwater levels, stream flow and extended periods of stream intermittency can tip the scales away from native species dominance to conditions that favor the nonnative plant (Stromberg et al. 2007). However, in highly altered rivers—like the Lower Colorado River, where dams, channelization, groundwater pumping, and other factors leave no semblance of a natural hydrograph—*Tamarix* reaches its highest density and is often found in monotypic stands (Anderson 2017; Ohmart et al. 1988). Normally, in moderately altered systems that still maintain natural seasonal periodicity—but where flood control structures preclude large floods—*Tamarix* is typically intermixed with native woody vegetation (Nagler et al. 2011).

The Invasion of Tamarisk Leaf Beetles in Grand Canyon

The first arrival of *Tamarix* beetles in Grand Canyon after their release in 2001 is unknown, but by 2009 they were commonly seen, and the effects of year-after-year defoliation, although unquantified until 2015, were becoming more evident. By 2015, the

Tamarix along the river corridor was dying in many areas, with the mortality increasing with each passing year (L.E. Stevens, personal communication to S.W. Carothers). To quantify the rate and levels of defoliation of *Tamarix*, Flesh and Stevens (unpublished data, see fig. 3) visually estimated the percent defoliation by randomly selecting 266 points in the 240 miles between Glen Canyon Dam and Diamond Creek. Where it occurred, defoliation on individual trees was conspicuously evident as the once luxuriant foliage was reduced to dead branches and twigs showing no living leaves over most of the plant. The results of the 2015 defoliation estimate showed varying rates of plant mortality depending on location within the 240-mile reach (fig. 3). Mortality reached very high levels in the Glen Canyon reach down to about river-mile 50 where 70 to 80 percent of *Tamarix* was dead or dying. Below river-mile 50 to river-mile 225 defoliation was less, but mostly ranging from 10 to 50 percent.

Below river-mile 225 to river-mile 280 high levels of defoliation were like those above Lees Ferry in Glen Canyon (S.W. Carothers 2005-2018, personal observations, and fig. 4). The relatively low levels of defoliation found sporadically on some beaches in the canyon represent areas where *Tamarix* density is either relatively low, or limited areas where the leaf beetle has not yet invaded. It is expected that within the next few years, all the *Tamarix* in Glen and Grand Canyon will be defoliated and either dead or dying and no longer a significant wildlife habitat resource. It is important to clarify that some once dense and widespread stands of *Tamarix* on upper terraces of Lake Mead sediments have been completely dead for the last decade or more as the lake has reached all-time low levels. The water table has dropped so rapidly that even fast-growing and drought-hardy *Tamarix* has not been able to survive.

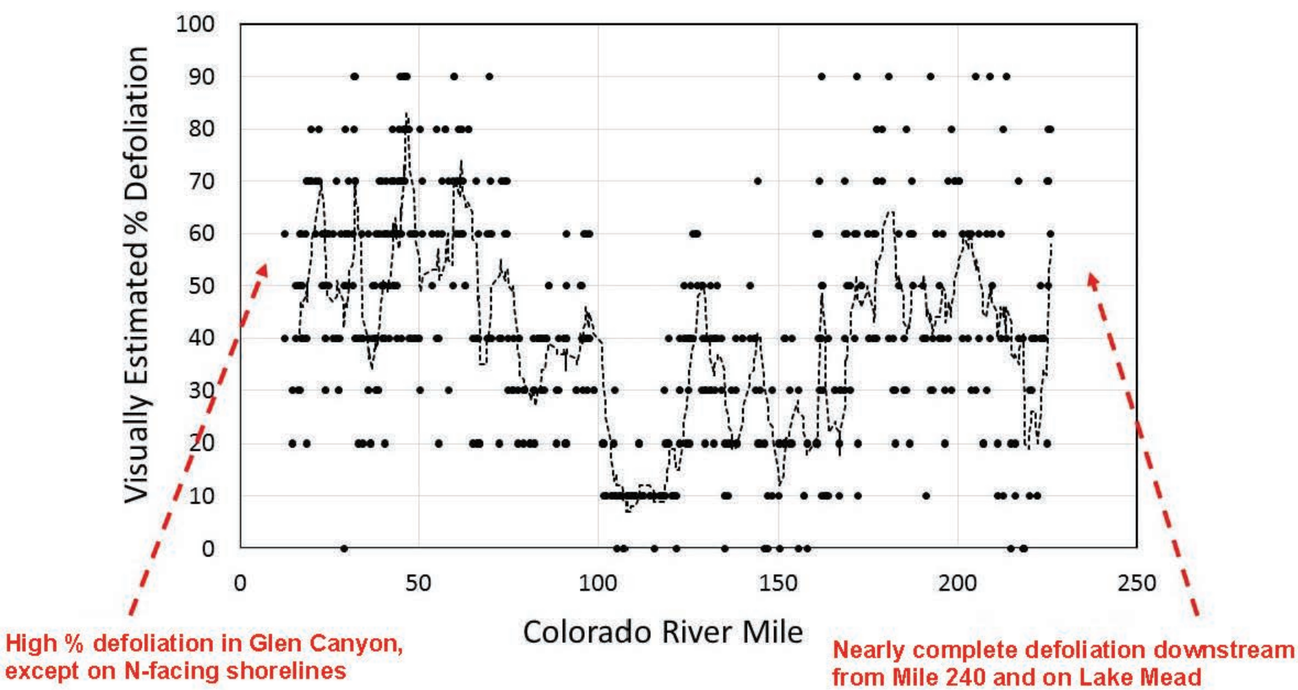

Figure 3—Percent defoliation of *Tamarix* by *Diorhabda* beetles along the Colorado River from below Glen Canyon Dam to the upper reaches of Lake Mead as determined by visual estimates in 2015. Defoliation was most extreme in the first 50-70 miles below the dam on south facing beaches in the upper river areas and high in the upper reaches of Lake Mead. Defoliation was less in the interior reaches of Grand Canyon. Data after Flesh and Stevens (unpublished manuscript 2015). Figure by Lawrence E. Stevens.

Figure 4—A comparison of the effects of tamarisk leaf beetle defoliation. Top photo 2005, before leaf beetle defoliation, photo by Steven W. Carothers. Bottom 2017, after defoliation, photo by Robb Irwin Eidemiller. Colorado River mile 7.5; Glen Canyon National Recreation Area, Arizona.

While it is too early to estimate the ground cover of the future in these canyon areas, OHWZ species are slowly moving downslope into the NHWZ and these species are expected to continue to increase in both size and area occupied (Holmes et al. 2005). In addition, the area occupied by and the structural robustness of the native shrub arrowweed, typically a NHWZ species, is steadily increasing in the shadow of declining *Tamarix*. Arrowweed is not generally recognized as nesting habitat for birds, but that may change as more of the moisture and soil nutrients previously consumed by *Tamarix* become available and arrowweed takes on a more robust growth form.

Tamarix Disruption of Mycorrhizal Fungal Communities of Cottonwood Trees

The iconic riparian gallery forest species of southwestern riparian habitats is the cottonwood tree. Though riparian ecologists often refer to the cottonwood-willow gallery forest community, the overriding foliage of the community is almost always provided by the cottonwood tree. It is the cottonwood tree, rather than the willow, that provides most canopy cover and root systems; and it is this species that accounts for unusually high wildlife productivity of the gallery forest (Johnson and Jones 1977). Willow trees require more water than cottonwoods and are usually found in a narrow band, one or two trees deep adjacent to the sides of a river or stream. Cottonwood trees can often be found throughout the entire floodplain where the water table is sufficient. In a recent estimate of the relative frequency of cottonwood versus willow trees within the gallery forest of the Upper San Pedro River, Carothers (2016) estimated that cottonwood trees contributed up to 90 percent of the riparian woodland, with willow only accounting for 5 to 10 percent. Thus, if cottonwood trees are disadvantaged or cannot recruit, the entire forest ecosystem is compromised. Cottonwood recruitment largely depends on natural river flow regimes (Stromberg 1993).

Recent studies have firmly established that healthy plant communities include a wide variety of interacting and interdependent species at many taxonomic levels, some of them cryptic and poorly understood (Corenblit et al. 2014; Franklin et al. 2016). Mycorrhizae and fungal endophytes, as well as a host of microbes, appear to be important participants in healthy plant communities. The contribution of mycorrhizal and other soil fungi to establishment, growth, and survival of cottonwood and other trees is becoming somewhat better understood as a result of recent and current research.

Conventional mycorrhizal fungi include the arbuscular endomycorrhizal (AM) fungi, which have been found in upward of 80 percent of terrestrial plant species, and ectomycorrhizal fungi (EM), which are found in about 2 percent of species (Smith and Read 2010). Cottonwood trees apparently have both, with probably many other associated microbes and fungi. The relationship is mutualistic. In this symbiotic relationship, fungi gain energy and carbon from tree photosynthesis while providing the tree with enhanced nutrient and water gathering capabilities from soil (Buckling et al. 2012). The trees depend on fungi for growth (Ghering et al. 2006; Meinhardt and Ghering 2012, 2013). Other biotic components of the rhizosphere and rhizoplane are only beginning to be understood, and knowledge of their roles in establishment and growth of native riparian plants is very limited. A few endophytic components are beginning to be understood (Lau et al. 2013), as are the complex interactions between species and individuals in establishment and maintenance of riparian communities (Corenblit et al. 2014).

Studies on the influence of *Tamarix* on beneficial mycorrhizal fungal communities of native cottonwood trees reveal one of the more insidious capabilities that this invasive tree/shrub employs to gain a competitive advantage over native riparian woody species. For example, when cottonwood trees are found in close association with *Tamarix*, the mycorrhizal relationship is disrupted by the invasive species' ability to alter soil chemistry by concentrating chemical compounds like salts, nitrogen, and phosphorus in leaf litter (Hultine et al. 2015; Meinhardt and Ghering 2012, 2013). Researchers have demonstrated that cottonwood trees growing in association with even low-density of *Tamarix* have two-fold reduced mycorrhizal colonization compared to native trees that do not have the invasive tree/shrub as a neighbor (Meinhardt and Ghering 2012). The impact of *Tamarix* on the other gallery forest species is not as well studied, but both willow (Beauchamp et al. 2006) and mesquite (Titus et al. 2003) are known to benefit from mycorrhizae and both are likely negatively impacted when in association with *Tamarix*.

Whereas studies on complex mycorrhizal interactions within the soil and roots of cottonwood trees in association with *Tamarix* are still in their infancy, it appears that the long-term survival of the mixed native/nonnative *Tamarix*-dominated riparian habitat was uncertain even before the *Tamarix* leaf-eating beetle was released. As Hultine et al. (2015) emphasized, the combination of climate change and the ability of *Tamarix* to disrupt the integrity of mycorrhizal relationships in the native gallery forest foundation species is a harbinger of long-term deterioration of native riparian habitats when growing in the presence of *Tamarix*. Thus, as the *Tamarix* declines because of the biocontrol efforts, the possibility of long-term native species recolonization of suitable habitat may eventually be enhanced.

However, the length of time and suite of conditions needed to remove or sufficiently attenuate the disruptive effects of *Tamarix* on mutualistic rhizosphere and rhizoplane species are not well studied. Therefore, the potential for reasonably complete restoration of native riparian communities is still largely unknown. Wildlife ecologists in the past several decades have promoted the benefits of the *Tamarix*-dominated riparian habitats, measured as increases in range and density of many vertebrate and invertebrate species. While we promoted the wildlife benefits, the *Tamarix* has apparently been altering the soil around and below the remaining native plant species. Hultine et al. (2015) suggest that the combination of mycorrhizal disruption by *Tamarix* and the vagaries of climate change constitute threats to riparian habitats that we had not previously anticipated.

These recent findings have clear implications on the future restoration and survival of the cottonwood forest type that is already considered one of the most threatened vegetation communities in the United States (Stromberg 1993; Webb et al. 2007). Dixon et al. (2009) reviewed the most up-to-date model predictions on climate change and found that the drier, hotter southwestern climate of the future would result in decreases in cottonwood-willow forests within the Upper San Pedro River by two-thirds. The additional mycorrhizal impacts to the gallery forest ecosystem when *Tamarix* is present are yet another threat to the declining riparian habitat throughout the southwestern United States.

Conclusions

The human life span is short, and the attention span of researchers is even shorter compared to the time needed for ecological change. The reign of *Tamarix* spp. as a dominant or co-dominating invasive species in western riparian areas has also been short. Longer-term consequences of this invasion are unknown. Along some rivers, especially those with drastically altered flow regimes, *Tamarix* has created a unique ecosystem that has been exploited by many wildlife species, including some that have lost habitat in the period since human activities started to have a significant impact on the earth's geology and ecosystems (the Anthropocene, see Kingsley, this volume). Currently (2019) we are witnessing major changes in the *Tamarix*-dominated ecosystem because of the very successful biocontrol effort. We have documented potential riparian wildlife impacts associated with the removal of *Tamarix* and cited studies documenting that the *Tamarix*-dominated native/nonnative riparian community can support more avian species than native species-dominated habitats of cottonwood-willow, mesquite, and mixed deciduous woodlands. It is now apparent that the biocontrol program was pursued with the removal or disadvantaging of *Tamarix* as a major goal without a clear understanding of revegetation dynamics post-biocontrol.

It appears so far that in the absence of active restoration efforts, recolonization of the post-biocontrol *Tamarix*-dominated habitat is almost exclusively limited to a mixture of native and nonnative grasses and often weedy, herbaceous cover with the conspicuous absence of woody species. Thus, at least in the short term, the biocontrol effort has resulted in the loss of important wildlife-producing habitat without replacement. In most areas where *Tamarix* has proliferated, both human-caused alteration of natural flow regimes and *Tamarix*-caused alteration of soils are complicating or prohibiting the establishment of native riparian woody plants.

In the specific case of the Grand Canyon, an unprecedented naturalized riparian community became established after construction of upstream Glen Canyon Dam. The ultimate disposition of that community is largely unknown at present; however, short- and long-term changes in riparian vegetation along the river corridor are the subject of regular inventory and monitoring (Palmquist et al. 2018) and we are assured that the answer to what comes after *Tamarix* will eventually be known.

What we do know now however, is that the *Tamarix*-dominated riparian community developed since Glen Canyon Dam became operational is now dramatically changing. It is possible, in time, that the OHWZ species (principally the mesquite and acacia) will continue to move downslope, as they have done for decades, and eventually replace *Tamarix* as it is reduced or eliminated. With the native OHWZ mesquites and acacias available to continue moving into the NHWZ, the Colorado River corridor in Glen and Grand Canyons may eventually be composed of mostly native species. They may provide an example, albeit rare so far, of the biocontrol effort realizing the potential that was originally anticipated by APHIS. At present, however, the decline of the *Tamarix* in Glen and Grand Canyons represents a loss of this unique naturalized habitat. Hence, we create a requiem to that declining community and the suite of wildlife species that were a part of it. Future research will document the development and replacement of an evolving community of vegetation and wildlife and hopefully the riparian habitat of the river corridor of the future is likely to be more dominated by native riparian vegetation species than it has been since Glen Canyon Dam altered the hydrograph.

Thus, as we contemplate an answer to the question, "What have we lost and what have we gained as a result of the biocontrol?" – or, simply put – "Is *Diorhabda*-induced biocontrol an ecological disaster, or will it eventually lead to a more robust native riparian community?" – at this time we do not find a clear answer.

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Chapter 3. Vanishing Riparian Mesquite Bosques: Their Uniqueness and Recovery Potential

R. Roy Johnson, Tanner S. Carothers, and Steven W. Carothers

Introduction

The “mesquite bosque” (Spanish for “forest” or “woodland”), one of the most unique and productive southwestern riparian habitat types, was once far more abundant than it is today. Twenty-five years ago, Stromberg (1993), with a focus on Arizona, provided an excellent review on the ecology, decline, existing threats, and potential for recovery of these mesquite forests. By 1993 the iconic mesquite bosque riparian habitat was in serious decline, due primarily to anthropogenic activities. Stromberg (1993) observed that previous attempts at habitat restoration were of limited success and indicated that much of the significant bosque habitat loss was largely the result of human-induced changes in the biotic and abiotic conditions and processes in river floodplains specifically required by species of mesquite (*Prosopis* spp.).

In this chapter, we update elements of Stromberg’s 1993 review and provide a classification between two types of bosques based on distinct vegetation associations along a relatively dry to wet riparian continuum. We also discuss the uniqueness of mesquite bosques as wildlife habitat and chronicle the loss of some of the more distinctive of these forests in Arizona as well as conditions that led to their disappearance. Lastly, we suggest opportunities for a timely approach to mesquite habitat restoration that will likely arise as a result of recent litigation resolution between the Department of Agriculture and the Center for Biological Diversity and the Maricopa Audubon Society.

The Mesquite Bosque

Mesquite (*Prosopis* spp.) forests, or bosques, historically represented one of the most widespread of riparian communities in the Southwest. At one time these bosques occurred on floodplains, often elevated 2 to 15 m above a relatively stable water table (Stromberg 1993) within riparian ecosystems. These ecosystems historically often spanned widths of several kilometers and stretched for tens of kilometers along the lower reaches of many southwestern rivers (Anderson and Ohmart 1977; Brown 1982; Rea 1983; Webb et al. 2007, 2014). Today, surviving mesquite bosques are reduced to remnants of their former grandeur.

The three native mesquite species found in association with bosques in the American southwest are bipinnately compound “legumes” that can occur as shrubs to very large trees. These species represent some of the more common facultative phreatophytes in the southwestern United States and northwestern Mexico. Honey mesquite (*Prosopis glandulosa*, formerly *P. juliflora*) and velvet mesquite (*P. velutina*, also formerly classified as a variety of *P. juliflora*) occur as components of both riparian and upland ecosystems. A third species, screwbean mesquite or tornillo (*P. pubescens*), is more closely associated with riparian ecosystems and largely restricted to floodplains (Benson and Darrow 1981; Correll and Johnston 1970; Foldi 2014; Kearney and Peebles 1960; Stromberg 1993).

Historically, mesquite bosques provided a unique and valuable ecosystem for a variety of animal and plant species; however, in recent times many of these woodlands have been either eliminated or heavily reduced and fragmented by anthropogenic influences. Associated with loss and reduction of these bosques is a decrease in populations of many plant and animal species dependent upon this riparian type (Stromberg 1993; Johnson et al. volume 1).

The importance of mesquites has been demonstrated by the fact that, although there are only three native species in the United States, they are prevalent and important enough that several plants and animals have "mesquite" in their names. Two such eponymous grasses include curly mesquite grass (*Hilaria belangeri*) and vine mesquite grass (*Hopia obtusa*). Insects include the giant mesquite bug *(Thasus neocalifornicus; Coreidae*) and the mesquite borer, larva of the longhorned beetle (*Placosternus difficilis; Cerambycidae*). Vertebrates include the mesquite lizard (*Sceloporus grammicus*), mesquite warbler, also known as Lucy's warbler (*Oreothlypis luciae*), and mesquite mouse (*Peromyscus merriamii*).

A Classification of Mesquite Bosques

Riparian mesquites may occur in hydroriparian (perennial streams), mesoriparian (intermittent streams), or xeroriparian (ephemeral streams) situations (Johnson et al. 1984a). Stromberg (1993) recognized that along typically ephemeral or intermittent streams, mesquite bosques are normally in association with Sonoran riparian scrub plant species (see Brown 1982). However, bosques found in association with perennial streams and rivers typically have more riparian vegetation assemblages, often including cottonwood and willow (*Populus-Salix*) trees. We suggest herein that bosques may be divided into two basic categories, Type A and Type B mesquite bosques. We base the classification on two factors: first, we readily agree with Stromberg's (1993) initial observations on the different vegetative associations found in bosques within ephemeral/intermittent versus perennial steamflow segments. We recognize the substantial differences in wildlife productivity between the bosque types. Clearly, the two types of bosques range along a riparian continuum from relatively dry to relatively wet. Type A bosques are normally found where non-perennial flow conditions exist. At the other end of the continuum, with perennial flow and generally wetter conditions, Type B bosques typically have an increased upper canopy cover of hydroriparian species such as cottonwood and willow.

Type A bosques consist almost entirely of mesquites as the tallest woody plants, while Type B bosques have a relatively large number of other, often taller tree species. These categories are important because higher groundwater conditions, or more permanent flow in the stream in all or portions of Type B bosques, facilitate the addition of taller trees. This results in an increase in foliage volume and habitat structure. The increased foliage volume usually results in an increase in biodiversity, especially bird species (Anderson 2017; Carothers et al. 1974; MacArthur and MacArthur 1961; MacArthur et al. 1962; Willson and Comet 1996).

The greater ecological value of Type B bosques to breeding birds can be illustrated by the number of species recorded in each of the two classes of bosques. The only Type A bosque in its near-historical condition for which we find breeding bird records is along the San Pedro River near Mammoth, Arizona (Gavin and Sowls 1975). We also have breeding bird records for three Type B bosques—San Xavier, Ft. Lowell, and Blue Point Cottonwoods. In the three Type B bosques, the number of breeding species are: San Xavier 73, Ft. Lowell 70, and Blue Point Cottonwoods 65, respectively, compared to only 45 breeding species for the San Pedro Type A bosque (Johnson et al. volume 1).

Type A bosques: In addition to mesquite, there may be a small number of scattered trees, e.g., cottonwoods (*Populus* spp.), willows (*Salix* spp.), Arizona walnut (*Juglans major*), or other species, but not in sufficient density to break the homogony of a mesquite-dominated upper canopy. Other woody species, most often shrubs or smaller trees, may include acacias (*Senegalia* spp., *Vachellia* spp.), wolfberry (*Lycium* spp.), desert hackberry (*Celtis ehrenbergiana*), saltbush (*Atriplex* spp.), and numerous other species (Stromberg 1993). Examples of Type A bosques include the surviving large mesquite woodland along the San Pedro River near Mammoth, Arizona, and the "extirpated" Santa Cruz River arm of Komatke Thicket, also known as the New York Thicket (Rea 1983; Wigal 1973).

Type B bosques: In addition to mesquite trees and the same basic shrub understory species also found in Type A bosques, several broad-leaf trees often occur that are commonly taller than the mesquites. These include cottonwoods, willows, net-leafed hackberry (*Celtis reticulata*), blue palo verde (*Parkinsonia florida*), velvet ash (*Fraxinus velutina*), Arizona walnut, desert willow (*Chilopsis linearis*), and other species (Stromberg 1993). These trees may be scattered throughout the bosque, such as at Blue Point Cottonwoods on the Salt River. Or they may occur largely along the streamside as was the situation with: the now "extirpated" San Xavier Bosque, also known as the Great Mesquite Forest near Tucson; the Phoenix Bosque (Jacobs and Rice 2002); and the Yuma Bosque (Mearns 1907).

Unfortunately, few studies were conducted in most of the larger bosques prior to their loss, and detailed accounts of their biodiversity focused primarily on studies of the avifauna. We know that species rare in the United States often occurred in these bosques. For example, three primarily Mexican (or neotropical riparian breeding bird species that may nest in Type B bosques) include the ferruginous pygmy-owl (*Glaucidium brasilianum*), tropical kingbird (*Tyrannus melancholicus*), and rose-throated becard (*Pachyramphus aglaiae*). Coincident with the loss of the Phoenix and Tucson bosques, the owl no longer occurs in those areas, but it formerly occurred north to the Blue Point bosque (Johnson and Simpson 1971; Johnson et al. 2000; Johnson et al. 2003), and in both the Phoenix (Breninger 1898) and Ft. Lowell bosques (Bendire 1892). The northernmost breeding record for the becard was in the San Xavier bosque, which is now completely gone (Webb et al. 2014). The kingbird is the only remaining of these three species that still nests along the San Pedro River among other spots throughout Arizona (Corman and Wise-Gervais 2005).

Mesquite Resilience

Despite the widespread decline of mesquite bosques throughout their former range in the Southwest and Arizona specifically, mesquite species are well adapted to arid environments and are excellent candidates for habitat restoration where conditions allow. In riparian areas, mesquite seems to thrive when alluvial groundwater is between about

2.5 and 12.2 meters. The higher the groundwater, the more robust is their growth form (Stromberg et al. 1992; Stromberg et al. 1993). Mesquite trees have a dimorphic root system, including a deep tap root that can reach alluvial aquifers as well as a system of lateral roots that can take advantage of precipitation and flood waters (Leenhouts et al. 2006). Thus, mesquite species are capable of extracting water from both surface soil and deeper strata long after obligate phreatophytes like cottonwood and willow can no longer be sustained during severe droughts and/or falling water tables. These factors are especially important since a great deal of uncertainty currently exists on the future health of riparian habitats in the arid Southwest with climate change and tamarisk or saltcedar (*Tamarix* spp.) biocontrol (Carothers et al. this volume; Dixon et al. 2009).

The relative resilience and deeper root system of mesquite may allow the species to replace declining cottonwood/willow gallery forests if climatic conditions are such that available groundwater can no longer sustain gallery forests of obligate hydroriparian and mesoriparian phreatophytes (Stromberg et al. 2012). We also suggest that one or more of the three mesquite species discussed above will be excellent candidates for riparian restoration in those suitable areas where *Tamarix* has been or will be eventually decimated by biocontrol (see McLeod volume 1).

Mesquites also have a shoot system adapted for high temperatures and low moisture conditions. It is widely accepted that microphylls (small leaves), such as those of mesquite species, with a reduced surface area, result in reduced evapotranspiration rates (Thoday 1931). This allows plants to survive in more arid environments than related species with larger leaves (megaphylls). In addition to microphylls, mesquites also regulate their stomata, pores used for gas exchange, by opening them wider when conditions are more mesic and closing or partially closing them to reduce water loss when more arid conditions prevail (Leenhouts et al. 2006).

A less known adaptive strategy for surviving in a variety of hydrologic conditions occurs in the root system of mesquites. Research along the San Pedro River in southeastern Arizona has documented that mesquites have the ability to redistribute soil moisture (hydraulic redistribution) throughout the year to achieve maximum growth, with the direction of sap flow differing depending on environmental conditions (Hultine et al. 2004). For example, before the monsoon season, sap flow in the deeper reaching tap root of a mesquite is upward (hydraulic lift) during both day and night, and from there outward toward the periphery of the plant through lateral roots nearer the surface of the soil. After the start of the rainy season, when surface soil is moist, the plant's water distribution pattern is reversed, and water moves toward the stem through lateral roots and then downward in the taproot toward the water table (Hultine et al. 2004; Leenhouts et al. 2006). Thus, mesquites are well adapted to grow in both moist riparian zones where they can reach very large sizes, as well as relatively dry upland conditions where robust growth and tree size is relatively limited.

Riparian Mesquites

Riparian mesquites commonly occur on floodplains and terraces, often separated from the stream course by cottonwood-willow riparian gallery forests or mixed deciduous woodlands, such as sycamore-ash (*Platanus-Fraxinus*). On their more arid side, away from the stream course, bosques commonly grade into upland desertscrub or semidesert grassland (Brown 1982). Velvet mesquite trees once rivaled mature cottonwoods in size, reaching more than 4 feet in diameter and over 70 feet in height (Brandt 1951; Stromberg 1993; Swarth 1905; see fig. 5). Such large size is at least partially because they are among the least disturbed woody species by floods of the major southwestern riparian trees, owing largely to their extensive root systems, thus attaining ages exceeding that of many other riparian trees (see Smith and Finch volume 1). At the time of early settlement by Euro-Americans, bosques with such large trees were historically widespread throughout the Southwest, especially along larger perennial rivers (Dobyns 1981); however, we know of no remaining trees approaching such large sizes.

Although mesquites growing in riparian situations may have the same genotype as those occurring in adjacent upland situations, with more access to groundwater and nutrients, riparian mesquites attain much greater size than upland plants. Willard (1912) wrote of trees in the San Xavier bosque that “the mesquite trees are wonders of their kind. There were some whose trunks at the base scaled over four feet in diameter.” Later, Brandt (1951) wrote that “here there are, indeed, trees of heroic dimensions; the bole of one stately specimen that we measured reached a girth of 13 feet, 6 inches, and a diameter of more than 4 feet, 3 inches; while the height of another capitol-domed giant was calculated to be 72 feet.”

In Arizona, the larger iconic bosques, especially along the perennial sections of the Salt, Gila, Santa Cruz, and Lower Colorado Rivers, are either gone or greatly reduced in size as second/third-growth trees. The best remaining examples of mature forests of velvet mesquite known to us are still found along several reaches of the San Pedro River where drought and declining water tables may eventually pose a threat to their long-term survival along usually dry desert watercourses. Mesquites often occur along

Figure 5—Photograph taken in the mid-1970s of Dr. Charles H. Lowe, one of the earliest riparian ecologists, standing next to a 72-foot tall, 2.5-foot diameter at breast height velvet mesquite tree (*Prosopis velutina*) at the confluence of Sycamore Creek and the Verde River on the Fort McDowell Indian Reservation, Maricopa County, Arizona. (Photo by David E. Brown.)

margins adjacent to normally dry arroyos as a single line of trees along with species such as acacias, paloverdes, and ironwood (*Olneya tesota*) as well as a variety of other trees and shrubs (Simpson 1977; Stromberg 1993). Even mesquites of these xeroriparian ecosystems have been impacted by urban and rural development, especially by larger cities such as Phoenix and Tucson. Some of the larger remaining xeroriparian bosques are on protected lands such as along Growler Wash in Organ Pipe Cactus National Monument (Johnson et al. 1984b).

Compared to xeroriparian situations where one or two trees line a generally dry desert wash, more species of herbaceous riparian plants occur in Type A and B mesoriparian bosques that do not generally occur with xeroriparian mesquites. These include some relatively uncommon plants like hoary bowlesia (*Bowlesia incana*), Watson's dutchman's pipe (*Aristolochia watsonii*), and several grasses, e.g., Arizona brome (*Bromus arizonicus*) and vine mesquite grass.

Upland Mesquites

Within the past several decades, mesquite species have generally migrated out of the mesoriparian and xeroriparian areas and successfully invaded many upland habitats. The increase in mesquites in upland areas has been largely due to the natural adaptability of the species for surviving in a wide range of hydrologic conditions discussed above. Clearly, upland mesquites are the same species as riparian mesquites but have been able to invade the upland areas as grasslands have declined under grazing pressures or drought. Their spread into uplands has been aided by the foraging of cattle on mesquite seed pods in riparian areas. The seeds can then be carried by these cattle into adjacent grassland (Cable and Martin 1973). Examination of cattle droppings showed as many as 1,671 mesquite seeds in a single cow-chip (Glendening and Paulsen 1955).

Loss of a Mesquite Bosque: Loss of an Ecosystem

The middle Santa Cruz River near Tucson, the middle Gila, and the Lower Colorado River from Davis Dam to the Mexican border once supported large mesquite bosques. The widespread loss of many of these once common bosques with trees of a far larger size and distribution than can be found today is well documented, especially for central and southern Arizona. Additionally, the extirpation of vertebrate species from portions of these river systems has been particularly well documented (Anderson 2017; Carothers et al. 2013; Ohmart et al. 1988; Rea 1983; Rosenberg et al. 1991; Stromberg et al. 2004; Webb et al. 2014).

Many, perhaps most, of the larger bosques once common along stream courses in Arizona have been "extirpated" through Euro-American settlement patterns (Dobyns 1981; Webb et al. 2014). Stromberg (1993) lists numerous threats to mesquite bosques. The value of mesquite firewood is exceeded in the Southwest deserts only by desert ironwood and both of these species have been significantly reduced by wood-gathering for domestic as well as commercial use. Thousands of acres of mesquites were cut to provide fuel for steamships that navigated the Lower Colorado River during the last half of the 1800s and early 1900s (Lingenfelter 1978). Much of those mesquite woodlands in the Lower Colorado River Valley and along other major southwestern rivers were replaced by invasive *Tamarix* (Poff et al. 2011; Rosenberg et al. 1991; Stromberg et al. 2007), which is now in decline as a result of biocontrol (Bloodworth 2016). Some bosque remnants are still extant along waterways for larger ephemeral and intermittent streams, though the woodlands are usually smaller and not as widespread as those that historically occurred along larger perennial and intermittent streams. Some of the best developed bosques remaining in Arizona occur along perennial streams with neither upstream dams nor urban developments, such as along the Gila River upstream from Coolidge Dam (Minckley and Clark 1984) and along the San Pedro River (Gavin and Sowls 1975; Stromberg and Tellman 2009).

Perhaps the most severe losses of riparian mesquite ecosystems have been to screwbean mesquite. A recent study has shown that tornillo, or screwbean mesquite, has disappeared from 53 percent of the localities in which it was common more than a century ago (Foldi 2014). The decline of screwbean mesquite along portions of the Rio Grande in New Mexico and Lower Colorado River in Arizona has been as recent as the past few decades and this once dominant species can now only be found in isolated patches (Anderson 2017). In a recent study attempting to determine the cause of the decline of screwbean mesquite in the Lower Colorado River watershed, Madera (2016) concluded that some unknown pathogen is likely the cause of the species decline in that area.

Vanishing Bosques

San Xavier Bosque: The Great Mesquite Forest, Santa Cruz River

Located on the Santa Cruz River, approximately 10 miles south of Tucson and adjacent to San Xavier del Bac Mission, this Type B bosque was called the Great Mesquite Forest because of its outstanding avian biodiversity and sheer abundance of birds (Brandt 1951; Webb et al. 2014). This once vast forest represents one of the best documented losses of an entire riparian/aquatic ecosystem in the Southwest (Webb et al. 2014). Between the earliest journal records in the mid-1800s and its complete demise by the 1960s, it was the best-known mesquite woodland in the Southwest. Although numerous accounts of ornithological work in the bosque were published, little is known of its size at the time of Euro-American settlement. Willard (1912) spent “the whole day… skirting along the edge of the mesquite forest a distance of some six miles.” Brandt (1951) reported that it “bordered both banks of the Santa Cruz for a number of miles and appeared to be four or five miles in width at its broadest part, tapering back to the river on either side.” An aerial photograph taken in 1936 showed the bosque to be approximately 7 square miles, but agricultural fields were already encroaching on it at that time (Webb et al. 2014). Biological reports by Euro-Americans that stopped at San Xavier del Bac Mission to camp along the Santa Cruz River became available by the mid-1800s. In 1850, Judge Benjamin Hayes of California wrote in his journal of this bosque that wolves “… were howling all around us, and one of very large size, was seen” (Davis 1982). In 1872, Major Bendire wrote of camping in this bosque on a trip from his post at Ft. Lowell in Tucson to the army outpost at Tubac (Bendire 1892).

By the beginning of the 1900s, the Great Mesquite Forest became one of the best-known sites for ornithological work in the United States. In addition to a few birds mentioned by Bendire (1892), the first attempt to record all the breeding birds of the bosque was in 1902 and 1903 (Swarth 1905; Webb et al. 2014). From Swarth’s first visit in 1902 into the early 1960s, several of the best-known ornithologists in the United States; including W.L. Dawson (1921) and A.C. Bent (1919-1958), visited and published lists of birds breeding in this bosque. However, by 1917, 2,500 cords of mesquite were being cut and removed annually from the San Xavier bosque by woodcutters (Dawson 1921), and by 1960, the bosque consisted mostly of second growth mesquites, approximately 15 to 20 feet in height with none of the previously described giants remaining (R. Johnson, personal observation). The final death knell of the forest occurred during the 1970s, when construction of Interstate 19 bisected the remains of this once great forest (Webb et al. 2014). It was also during the 1970s when decades of groundwater pumping for domestic water supplies by the city of Tucson finally drained the alluvial aquifer below the levels that once supported these grand mesquite trees.

One of the unusual animals of the Great Mesquite Forest was the mesquite mouse, which occurs only in the States of Arizona and Sonora, Mexico (Hoffmeister 1986; Kingsley 2006) and is one of the rarest of native mice in the United States. The largest known population of this mouse lived in the Great Mesquite Forest before its destruction. In addition, by the start of the 20th century, several species of birds that had been recorded earlier along the Rillito River (Bendire 1872a, 1872b, 1892), a major tributary of the Santa Cruz River, had apparently been lost from the Great Mesquite Forest. These were largely riparian or wetland species, including killdeer (*Charadrius vociferus*), black phoebe (*Sayornis nigricans*), common yellowthroat (*Geothlypis trichas*), song sparrow (*Melospiza melodia*), gray hawk (*Buteo plagiatus*), and willow flycatcher (*Empidonax trallii*). Although the now endangered southwestern willow flycatcher (*E. t. extimus*) was known to breed along the Lower Colorado River during the late 1800s and early 1900s (Huels et al. 2013; Unitt 1987), it is not clear if the species was breeding or a migrant along the Santa Cruz River (Dawson 1921; Swarth 1905; Webb et al. 2014).

Komatke Thicket: New York Thicket, Confluence of Santa Cruz with Gila River

Located at the confluence of the Santa Cruz with the Gila River on the Gila River Indian Reservation, Komatke, or New York Thicket, was so named by local Pima Indians because it was so crowded with trees and wildlife (Rea 1983; Webb et al. 2014). It was said to be approximately 8 miles long from east to west and 3 miles wide from north to south (Webb 1959; Webb et al. 2014). Neff (1940) wrote that "in some places it is said to be fully six miles in diameter." Records are inadequate to determine if this was a Type A or Type B bosque. Unfortunately, by the late 1970s, apparently about 90 percent of the mesquites were dead, killed by the lack of both surface water and groundwater (Rea 1983). In a 1970s photograph showing much of the bosque (Wigal 1973), there were no live mesquites and little more than dead stumps since most trees had been cut for firewood. Wigal (1973) called the once grand woodland "… 6,800 acres of a dying bosque." At that time the Gila River had ceased to be free-flowing after the completion of upstream Coolidge Dam in 1928 and water flowed in the Santa Cruz only on rare intervals during upstream flooding (Webb et al. 2014). In addition, starting in 1902, deep wells were drilled on the Reservation and nearby, rapidly lowering the water table (Rea 1983). By the late 1970s, live mesquite trees were only found at the edge of the bosque, where water drained from the bajada of the Sierra Estrellas, and in an ancient Hohokam canal over 2 meters wide (Rea 1983). Although there were studies of a large population of nesting white-winged doves (*Zenaida asiatica*) in the bosque (Neff 1940), in contrast to the Great Mesquite Forest, we find no general ornithological studies or other biological studies of any type from this large bosque before its demise.

Ft. Lowell Bosque, Rillito-Tanque Verde Creek

Ft. Lowell, a Type B bosque, was a commonly visited site by ornithologists looking for rare birds during the late 1800s and early 1900s. Although not as well studied ornithologically as the Great Mesquite Forest, we find 70 species of nesting birds reported for the Ft. Lowell area, only three species fewer than 73 species for the Great Mesquite Forest (Webb et al. 2014). Published ornithological studies at Ft. Lowell started earlier than those along the Santa Cruz River; we have records of birds from this area for three decades before records from the Great Mesquite Forest. The area was perhaps best known for the discovery of the first ferruginous pygmy-owls for the United States in “the heavy mesquite thickets bordering Rillito Creek, near the present site of Camp Lowell” (Bendire 1872b, 1892). In addition to the owl, Bendire added another breeding bird of mesquite bosques (Bibles et al. 2002) to the regional avifauna, the gray hawk, and he also recorded the painted redstart (*Myioborus pictus*), a montane breeding species that occurs in mesquite bosques during migration. Bendire also added two species new to science from this area that occur in mesquite bosques: Bendire’s thrasher (*Toxostoma bendirei*) and rufous-winged sparrow (*Peucaea carpalis*) (Fischer 2001).

After Bendire’s work in 1872-1873, the next report of birds for the Ft. Lowell area included several species collected by Frank Stephens in 1881 (Brewster 1882, 1883; Webb et al. 2014). Stephens was the second person to collect a ferruginous pygmy-owl in the area and he also collected the type specimen of a new subspecies of brown-crested flycatcher (*Myiarchus tyrannulus magister*) near Ft. Lowell (American Ornithologists’ Union 1957). Including the flycatcher, there were seven new subspecies of birds named from the Ft. Lowell bosque (Brandt 1951). By the start of the 1900s, groundwater withdrawal for the city of Tucson and surrounding Tucson Basin had resulted in a lowering groundwater table throughout the Basin (Webb et al. 2014). This resulted in an ever-increasing loss of mesquites from both Rillito/Tanque Verde Creek and the Great Mesquite Forest. Along with the loss of mesquites, several nesting riparian birds such as the gray hawk and willow flycatcher also disappeared. By the 1930s, Willis (1939) wrote that “the tree stand here [on the Rillito floodplain] is nearly pure mesquite and is probably a relic of the old forest.” Between the mid-1950s and 1980s, there was a continuing decline in mesquites along the Rillito floodplain, apparently due to urban development and decreases in both groundwater and surface flow. In 1988, Tucson increased its rate of withdrawal of groundwater, further stressing the remaining mesquites (Stromberg et al. 1992). Although the Ft. Lowell Bosque has survived longer than the Great Mesquite Forest on the San Xavier Indian Reservation, it is a shadow of its once great size. Urban encroachment and increasing groundwater withdrawal is a continuing threat to the remaining woodland.

Casa Grande Bosque, Gila River

Formerly located on the south bank of the Gila River, on an old floodplain (Judd et al. 1971) and adjacent to Coolidge, Arizona, the Casa Grande Type A bosque has been “extirpated” by groundwater overdraft and rural and urban development. Tree-ring dating aged three trees in this bosque between 110 and 137 years old (Judd et al. 1971). As with bosques at San Xavier and Komatke, a declining water table also had much to do with the loss of this habitat. The water table declined from a depth of 44 feet in 1923 to approximately 100 feet in 1952 and 150 feet by 1960 (Judd et al. 1971). Mistletoe (*Phoradendron* spp.) infestation may have further hastened the death of mesquite trees (Judd et al. 1971; Stromberg 1993). With the completion of upstream Coolidge Dam in 1928, flow in the Gila River ceased, thus cutting off surface as well as groundwater to this bosque. Although there were later records of the ferruginous pygmy-owl from various localities along the Gila River, one of the earliest specimens of this rare tropical owl for the United States was taken here by Mearns on 10 May 1885 (Fisher 1893; Johnson et al. 2000).

Phoenix Bosque, Salt River Valley

The Salt River Valley of central Arizona is an example of a region that retains almost none of the area's former wet riparian habitat or the mesquite bosques that were once widespread. Three of the State's major rivers—Gila, Salt, and Verde—have all been dammed upstream from Phoenix to withdraw water for irrigation, municipal, and industrial usage in Phoenix and the surrounding area. This has resulted in desertification and dewatering of much of the stretches below the dams and inundation of potential riparian areas above the dams. In addition, several thousand square miles of xeroriparian ecosystems have been replaced in the Salt River Valley by urban and rural development. Phoenix Bosque is the name we have chosen for this Type B bosque that formerly lined much of the Salt River during the mid-1800s before it was entirely displaced by Phoenix. This large bosque covered approximately 25 square miles, extending northward for approximately 3 miles at its widest point, and from current Central Avenue westward for approximately 13 miles. A narrow band of cottonwood-willow grew immediately adjacent to the river with the mesquite on the large floodplain and river terraces to the north (Jacobs and Rice 2002). Phoenix was established amid this large bosque in 1868, and by 1871, "98 blocks, each 300 feet long" were for sale (Barnes 1935, 1988). These lots each supported an average of six giant mesquite trees and early maps showed dense mesquite stands extending from the Salt River northward beyond Five Points (junction of Van Buren Street with Grand Avenue; Douglas 1938).

Before construction of dams on the Salt and Verde rivers, but long after the settlement of Phoenix began compromising the bosque, early irrigation canals and earthen ditches were commonly lined with cottonwoods and other species of trees, thus resembling natural watercourses (R. Johnson, personal observation). These artificial watercourses mimicked natural stream courses. Breninger (1898) wrote of the Phoenix area that "since trees planted by man have become large enough to afford nesting sites for woodpeckers, this Owl [cactus ferruginou*s* pygmy-owl] has gradually worked its way from the natural growth of timber bordering the rivers to that bordering the banks of irrigating canals, until now it can be found in places ten miles from the river."

Yuma Bosque, Lower Colorado River Valley

At the turn of the 20th century, Edgar A. Mearns, a member of the U.S./Mexican Boundary Survey party, published his findings for this region and reported that the extent of the alluvial bottom land between Camp Mohave and Yuma was approximately 400,000 to 500,000 acres in size. This was a Type B bosque with the river channel marked by a line of tall cottonwoods and a lesser fringe of willow. He also observed that the adjacent bottom lands upslope of the river were covered more or less with mesquite including screwbean mesquite (Mearns 1907). Mearns marveled that the size of some of the larger tree canopies covered an area of 50 meters or more in diameter.

Since Mearns' work, hundreds of thousands of acres in the Lower Colorado River Valley have been invaded by nonnative *Tamarix*. By 1986, a group of biologists working in the Lower Colorado River Valley estimated that the total riparian vegetation comprised only about 40,000 ha (88,000 acres). Thus, at that time approximately 22.5 percent remained of Mearns' 1907 estimate. Of that 88,000 acres, 40 percent was covered by pure *Tamarix* with only 43 percent consisting of native plants mixed with *Tamarix*, and 16.3 percent was covered by honey mesquite and/or native shrubs. Only 0.7 percent (307 ha) could be considered mature cottonwood or willow habitat. Thus, less than 15 percent of the original estimates of native riparian vegetation at the time of Mearns remained by the late 1900s, and much of that was mixed with nonnative *Tamarix* (Rosenberg et al. 1991).

Unnamed Bosques

In addition to the above discussed bosques, others are occasionally mentioned in the literature without references to size or other factors. For example, Stromberg (1993) mentions several and Minckley and Clark (1984) mentioned a large mesquite bosque at Texas Hill on the lower Gila River, in the Wellton-Mohawk area. Minckley and Brown (1982) wrote of losses of mesquite and cottonwood forests along the lower Gila and Colorado Rivers without naming them or giving better locations and sizes. Historical documents are woefully incomplete in detailing the extent of the change affected on southwestern mesquite bosques. It is clear that the retreat of bosques is inextricably tied to the westward expansion of American settlers and subsequent ventures requiring large amounts of both water and land for agriculture and urbanization. The future of southwestern mesquite bosques is uncertain and full of challenges, but with effective conservation and restoration practices, opportunity exists to restore some areas with one or more of the three mesquite species. As stated above, mesquite species make promising candidates in the restoration of riparian corridors and springs, able to send roots deeper than cottonwoods and willows.

Perhaps the two best known surviving large bosques are a Type A bosque on the San Pedro River near Mammoth (Gavin and Sowls 1975) and a Type B bosque at Blue Point Cottonwoods, on the Salt River (Johnson and Simpson 1971; Johnson et al. 2000). Whereas mesquites found in these existing bosques are often smaller than those from the historical accounts, they still support a high degree of biodiversity. By examining extant bosques, one may extrapolate conditions that existed before the pumping of groundwater, the introduction of exotic competitors, and the large-scale clearing of mesquite for firewood and building material. From these case studies of historical and extant bosques, their distribution, extent, and size (per tree) has changed dramatically through the Anthropocene. Whereas any single threat has the potential to dramatically change the structure of bosques, these ecosystems are beset by polymorphic threats.

The Future of Mesquite Bosques in the Shadow of Tamarisk Biocontrol

The remarkably successful biocontrol efforts to slow down invasion and/or eliminate the nonnative tamarisk (*Tamarix* spp.) shrub/tree throughout the range of the mesquite species (see McCloud volume 1; Carothers et al. this volume) has provided heretofore unavailable opportunities for habitat restoration of native riparian species, especially cottonwood and willow, but may eventually include mesquite. Potentially suitable conditions for cottonwood-willow and mesquite restoration are now available as the *Tamarix* declines in tens of thousands of acres. (However, see Carothers et al. this volume, on the potentially lasting inhibitory effects of *Tamarix* on reestablishment of native species.) The specific adaptations exhibited by mesquite (described above) for thriving in a variety of habitat conditions on a wet to dry riparian continuum allow for mesquite to be an obvious native riparian species for habitat restoration.

Significantly, Federal funding for riparian habitat restoration has come available in 2018 at unprecedented levels. As of June 2018, the Department of Agriculture is under a court order to fund the rehabilitation of riparian habitat to benefit the Federally threatened southwestern willow flycatcher in Arizona and New Mexico. The recently released United States District Court Remedial Order (Case No. 2:13-cv-1785-RFB-GWF), settling a lawsuit brought by the Center for Biological Diversity and Maricopa Audubon Society against the Department of Agriculture, is required to result in progress toward specifically restoring/creating cottonwood-willow habitat used for nesting by the flycatcher. Restoration of mesquite species as a buffer to the preferred habitat of cottonwood and willow could also benefit. The Court has ordered the Department of Agriculture to fund “intensive third-party riparian restoration efforts” in most of the living rivers and streams in Arizona, California, and New Mexico where the flycatcher was known to utilize or nest in the now-disadvantaged and dwindling *Tamarix* “bosques.” While the focus of the court-ordered flycatcher habitat restoration is directed toward cottonwood-willow trees, buffers of mesquite trees around the former, as is found in Type B bosques, will be an obvious solution to long-term protection of plantings of the hydroriparian species. Whereas in many of the sites required by the court order for planting of cottonwood-willow habitat will not have the seasonal flooding necessary for recruitment of these hydroriparian species, the mesquite buffer restoration will persist and recruit once they are established.

Stromberg (1993) earlier recommended prevention of further degradation of mesquite bosques by removing the ongoing threats of recruitment inhibiting groundwater overdraft. She also recommended eliminating woodcutting, livestock grazing, and impacts originating from recreation. As far as restoration of mesquite habitat, Stromberg (1993) was far less sanguine; for example, she correctly cited one of our early papers (Carothers et al. 1990) as providing examples of high post-restoration mortality in largely failed efforts to reestablish mesquite. In the late 1980s and throughout the 1990s, habitat restoration science in riparian habitat was in its infancy. Those engaged in early attempts at restoration were often unaware of the dynamics between restoration success and a myriad of project-defeating biotic and abiotic factors. Not all potential restoration sites are equal and few are harbingers of rapid success. The previous occupation of a site by a particular plant species does not in and of itself offer predictions of restoration success. Even after evaluation of hydrologic and edaphic factors indicate restoration potential, beaver, elk, rabbits and pocket gophers can quickly decimate the efforts of an expensive, well-planned revegation project. It may take years for a restored site to become established and self-recruiting.

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Chapter 4. Using the Southwest Experimental Garden Array to Enhance Riparian Restoration in Response to Global Environmental Change: Identifying and Deploying Genotypes and Populations for Current and Future Environments

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Introduction

The role of genetics in ecosystem restoration has largely revolved around the policy of using local genotypes (i.e., those sourced from areas near the restoration site), based on the logic that local plants are best adapted to local conditions (Johnson et al. 2004; Meffe and Carroll 1977). In a relatively stable environment, this is a sound practice; however, global environmental change impacts on the landscape are increasingly rendering this policy as inadequate at best and damaging at worst (Whitham et al. 2010). Here we define “global change” as ongoing changes in temperature, moisture, interactions with invasive plants and pathogens, increases in the frequency and severity of environmental extremes (e.g., fires, droughts), and other challenges (Cayan et al. 2010; Gutschick and BassiriRad 2003; Jones and Monaco 2009). Because of rapid environmental change, plants that are locally adapted to current environmental conditions are likely to become increasingly maladapted to their changing native environments. For example, in Arizona, Grady et al. (2011, 2015) found that Fremont cottonwoods (*Populus fremontii*) were currently locally adapted along an elevational gradient, and that tree genotypes from lower elevations transplanted to higher elevations were predicted to outperform the local genotypes at higher elevations under climate change conditions. In regions of especially rapid change such as the American Southwest (Garfin et al. 2013; Seager et al. 2007), local populations are likely to lack sufficient genetic variation to adapt to these new environments (Aitken et al. 2008; O’Neill et al. 2008a). Similarly, with the rapid velocity of climate change coupled with fragmented landscapes (Loarie et al. 2009), many species cannot migrate fast enough to reach favorable environments (Aitken et al. 2008; Davis and Shaw 2001).

Ignoring the reality of global change sets up restoration efforts for failure. Given that most restoration projects focus on foundation plant species that, by definition (Dayton 1972), shape the communities and ecosystems around them, discounting global change impacts to these foundation species will also lead to high biodiversity loss and altered ecosystem function. Failed restoration projects could also result in the loss of future restoration funding. Riparian and aquatic ecosystems support some of the highest biodiversity worldwide, and incorporating future climate change impacts in the restoration planning process is essential for successful preservation of these valuable habitats and the wealth of species they support. These problems are recognized as a global issue that affects riparian and aquatic organisms worldwide (Davies 2010).

Robust experiments are a key component of identifying which species, populations, and genotypes are most likely to survive both current and future environmental conditions. Just as agricultural field trials have been essential for crop improvement, the same basic field trial approach is being used for forest trees and other species in wildlands (e.g., Marris 2009; O'Neill et al. 2008a). In the American Southwest, we have established field trials of riparian cottonwoods and willows that are embedded within larger landscapes to be restored. We recommend such field trials as a first step in restoration to minimize future risks and enhance project success on adjacent lands.

The Southwest Experimental Garden Array (http://www.sega.nau.edu) was specifically funded by the National Science Foundation to develop this approach. Established along a 1,400-meter elevational gradient, the SEGA gardens allow researchers and managers to understand predicted climate change impacts across five major vegetation zones in Arizona. Based on findings quantified in gardens (Evans et al. 2016; Grady et al. 2011, 2015) and in the wild (Ikeda et al. 2017), restoration biologists can deploy genotypes from populations that show adaptive potential and are most likely to survive in a changing environment. In so doing, both the foundation species and the diverse communities they support can be preserved for future generations (Ikeda et al. 2014).

The following sections develop the logic of this approach. We: give evidence that global change results in maladaptation; emphasize the importance of focusing first on foundation plant species that are the community and ecosystem drivers; describe experimental approaches that can be used to identify adapted genotypes for use in restoration; and discuss the importance of maintaining high genetic diversity for the evolutionary potential and highest biodiversity of foundation species' communities.

Local Adaptation in a Stable Environment and Maladaptation with Global Change

Many studies show that plants are generally locally adapted to their environment (e.g., Clausen et al. 1940; Hereford 2009; Joshi et al. 2001; Savolainen et al. 2007). Thus, the demonstration of local adaptation is a primary justification for using only local stock in restoration, which is the current practice for restoration within most national parks. However, climate change and other global change stressors such as invasive species, drought, and other extremes are causing dramatic shifts, such that locally adapted plants are becoming increasingly maladapted (Jones and Monaco 2009). This decoupling of plants from the environments they were adapted to is resulting in landscape-level die-offs (Allan et al. 2010; Gitlin et al. 2006; van Mantgem et al. 2009) and a shift in the elevational and/or latitudinal distributions of species to environments that match the site conditions of their former home environments (e.g., Allen and Breshears 1998; Brusca et al. 2013; Chen et al. 2011; Walther et al. 2002).

A mismatch between the environments that species are currently adapted to and what the environment has or will become in the future is illustrated in figure 6, showing a photograph of the Bill Williams River that is suffering from upwards of 85 percent mortality of Fremont cottonwood. This mortality is associated with an ongoing drought that began about 5 years prior to a high mortality event that first became evident in the summer of 2015. Concurrent with this mortality event was a lack of surface stream

Figure 6—Photo of cottonwood riparian forest taken March 28, 2017, along the lower reaches of the Bill Williams National Wildlife Refuge in Arizona. Photo by co-author Hillary Cooper.

flows, a water table that had receded below 5 m, and an increase in extreme summer temperatures events ≥ 50 °C. These conditions have continued through the present (September 2017) and are associated with continuing mortality. High mortality of these foundation trees is of great concern since this forest is home to many sensitive and listed species (e.g., Lower Colorado River Multi-Species Conservation Program: LCR-MSCP 2004).

As such drought conditions persist or worsen into the future as species distribution models predict for the Sonoran Desert ecotype of *P. fremontii* (Ikeda et al. 2017), restoring with local stock would likely result in restoration failure for both the cottonwood and its many dependent species. Similar forecasts are predicted for many forest trees throughout western North America (Rehfeldt et al. 2006). Thus, experimentally identifying source populations and genotypes from non-local stocks that can survive both current and future conditions is crucial for restoration projects to be successful. Furthermore, when local plants are no longer locally adapted as figure 6 emphasizes, concerns about genetic pollution are largely negated. The focus should shift toward finding stocks that can survive in altered environments and simultaneously maintain the listed and sensitive species that depend upon this habitat for their survival (e.g., Bangert et al. 2013).

The Importance of Focusing on Foundation Species

Riparian foundation species such as cottonwoods and willows support hundreds of other organisms including sensitive and listed species (e.g., Bangert et al. 2013; Patla 2014; Skagen et al. 2005). Therefore, a first priority should be to identify the foundation species that play a central role in defining their respective communities and ecosystems and focus on their preservation as a first priority (Dayton 1972; Ellison et al. 2005; Whitham et al. 2003). Furthermore, many studies have shown that different plant genotypes support different communities and ecosystem processes (Crutsinger 2016; Lamit et al. 2015; Whitham et al. 2012). The communities that these different genotypes support represent a heritable plant trait (Gehring et al. 2017; Keith et al. 2010; Shuster et al. 2006). Thus, it is essential that we consider preserving not only species, but also the intraspecific genetic variation within these species. High genetic diversity in foundation species drives higher biodiversity in their associated communities and enhanced ecosystem processes (e.g., Ferrier et al. 2012; Fischer et al. 2014; Wimp et al. 2004). Thus, the problem of conserving riparian communities (and other vegetation types) is three-fold:

- Preserving foundation species.
- Preserving the genetic diversity of foundation species essential for maintaining their adaptive potential and role as a foundation species supporting diverse associated communities and ecosystem processes.
- Understanding the relative importance of genetics versus the environment and how they interact (G x E interactions) to affect foundation species, associated communities, and ecosystem processes.

To achieve these three goals in a changing environment, we must identify the individual plant genotypes and populations that can survive both current and future environmental conditions for a given restoration site (Grady et al. 2011; Ikeda et al. 2017; O'Neill et al. 2008a) and identify the corridors that maintain gene flow among populations as landscapes become more fragmented (Bothwell et al. 2017) and may suffer from the loss of genetic diversity (Vranckx et al. 2012). By focusing on these foundation species that are community and ecosystem drivers, we can also save many other species that are dependent upon the critical habitat they provide. In the next section, we describe how we can rigorously and experimentally identify the populations and genotypes of foundation species most useful in restoration.

Quantifying Which Stocks Will Survive Future Environments

Experiments are essential for identifying which species, populations, and genotypes are most likely to survive both current and future environmental conditions. The basic experimental approach pioneered by Clausen et al. (1940) is to establish field trials on portions of lands that are slated for restoration. Because the findings from field trials are most valid for the biotic and abiotic environment in which they are measured, having these sites embedded in larger restoration sites ensures that the findings from the field trials have the same soils, communities, and other variables that are most relevant to the adjacent sites to be restored. Based on quantification of plant performance (e.g., survival, growth, biodiversity, community stability, ecosystem functions) from such field trials, managers can identify stocks that exhibit adaptive potential to changing environments so they can be used to restore adjacent lands. This approach maximizes both short- and long-term plant performance goals and reduces risk of failure. Field trials have long been a standard practice in agriculture for maximizing yield, but they are much less common in wildlands restoration that necessitate more complex metrics of success (e.g., biodiversity, ecosystem services). Long-term monitoring of these field trials is especially important with long-lived species, as genotypes that do best in the short term may exhibit higher relative mortality in the long term or vice-versa (e.g., Sthultz et al. 2009).

Just as agricultural field trials have been essential to identify superior genetic lines to maximize crop yield under current and future conditions, the same basic field trial approach is being used with commercially important forest trees (e.g., Marris 2009; O'Neill et al. 2008a,b). As an example of the success of this approach, O'Neill et al. (2008a) developed a model using a network of provenance trials to match seed source populations of lodgepole pine with planting sites where they will perform best under future climatic conditions. Based on their findings, it is now reforestation policy in British Columbia to plant with trees adapted to a 2 °C hotter environment than the local stock (Marris 2009; O'Neill et al. 2008b; Pedlar et al. 2011), and these recommendations are being further refined to obtain even better matches with projected future environments (O'Neill et al. 2017).

Research in the United States on foundation tree species has greatly benefited from this field trial/common garden approach, exemplified in a new research facility, the Southwest Experimental Garden Array (SEGA). The provenance trial approach specifically involves collecting genotypes from each of several different populations throughout a plant's range and reciprocally transplanting them in common garden test sites to quantify their performance in different environments. Northern Arizona University (NAU) has undertaken a long-term program to develop SEGA as a research platform to study and provide solutions for utilizing genetics-based approaches to mitigate climate change, invasive species, and other global challenges. This research platform (fig. 7) allows collaborating scientists and institutions (e.g., USGS, USFS, NAU) to quantify the ecological and evolutionary responses of species to changing climatic conditions using emerging technologies.

SEGA was initiated in 2012 with $4.5 million in seed funding from the National Science Foundation (NSF) and NAU to create a network of 10 gardens along an elevation gradient in northern Arizona. The SEGA model is flexible, and a growing network of satellite sites are being added by users that expand upon and increase the resolution of the core sites. Because data are archived to NSF standards and are available online, this array is ideally suited to the experimental study of gene by environment interactions that can best be addressed using a common garden network. Study organisms span from soil microbes, grasses, forbs, shrubs, and trees.

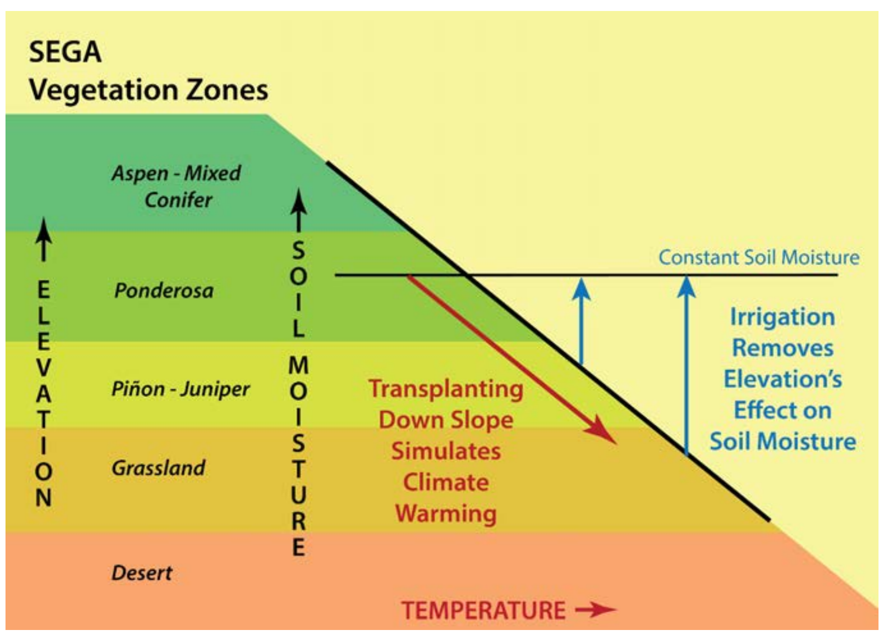

Figure 7—The Southwest Experimental Garden Array (SEGA) spans a steep elevational gradient of 1,400 m in northern Arizona that encompasses desert to aspen-mixed conifer vegetation types. Each site has irrigation to help establish plants and experimentally eliminate the effect of soil moisture with increasing elevation if desired. A weather station at each site, sensor arrays, live video, and other instrumentation allow researchers to monitor their plots remotely from their home institutions. Illustration by Paul Heinrich from the SEGA website http://www.sega.nau.edu.

Because temperature and moisture predictably change with elevation, these common gardens arrayed along an elevation gradient reflect climatic differences from desert to aspen-mixed conifer forest biomes that mimic the effects of climate change. Thus, this elevation gradient allows researchers to simulate the effects of climate change. For example, by planting the same genotypes of plants in multiple gardens, especially at lower elevations where it is hotter and drier, plants can be challenged with expected future climatic conditions. Researchers can then identify genotypes that show adaptive potential consistent with survival in a changing environment for use in restoration. Similarly, by adding supplemental water at a site or removing water with the use of rainout shelters, researchers can quantify the combined effects of temperature and moisture gradients with even greater control, or quantify just the effects of temperature alone. Experimental separation of temperature and moisture effects is especially important since increasing temperature has been demonstrated in virtually all climate models, while precipitation predictions are more variable as some areas might get wetter or have a change in the distribution of precipitation.

In combination, this experimental design allows researchers to identify optimal populations and genotypes that will perform best on adjacent restoration sites. Experimental designs also allow more complicated studies to include the examination of nurse plant effects and mycorrhizal mutualists (e.g., Gehring et al. 2017), the planting of co-adapted communities (i.e., multiple species co-evolved at the same site) planted and tested together versus randomized plantings of species from different sites (Grady et al. 2017), and planting of genotypes to promote biodiversity, community stability, and network structure (Keith et al. 2017).

SEGA enables a new generation of genetics-based climate change research that allows researchers from diverse disciplines to quantify the ecological and evolutionary impacts of climate change on (1) foundation plant species, (2) their associated communities, (3) native-exotic species interactions, and (4) the ecosystem processes that emerge from these interactions. Additionally, the Northern Arizona Research Greenhouses support plant propagation for use in SEGA, and the Environmental Genetics and Genomics center (EnGGen) allows researchers to genetically characterize their target species and identify candidate genes for functional traits of special interest (e.g., drought and temperature tolerance, productivity). These integrated facilities greatly enhance the research potential of SEGA.

In short, SEGA and related facilities allow unprecedented opportunities to address issues important in riparian systems, and other vegetation types, that can enhance restoration success for future climates and other global challenges. SEGA emphasizes interdisciplinary-interagency collaborations, which can help shorten the turn-around time between discovery and implementation by land managers.

SEGA sites have provided important findings that can be used by restoration biologists and managers to select high performing tree genotypes for a given restoration site under future conditions. For example, using trees collected from 14 source populations of Fremont cottonwood and grown at a Bureau of Reclamation riparian restoration site on the Lower Colorado River (fig. 8), Grady et al. (2011) showed that different genotypes and populations of Fremont cottonwood predictably varied in their performance when grown at the lower edge of their distribution where record temperatures can exceed 50 °C. From these studies, five major findings emerged:

(1) Transfer distance has an important impact on annual above-ground net primary

productivity (ANPP). A transfer distance of 6.5 °C means that a high-elevation, cool-site population was transferred to the warmer, low-elevation field trial, and the greater the transfer distance, the greater the decline in ANPP (fig. 8). ANPP is important because it reflects the growth rate of individual tree genotypes, but it is also positively correlated with survivorship (Grady et al. 2015) and biodiversity (Ikeda et al. 2014; Stone et al. 2010; Swaty et al. 2004). Comparing the same genotypes from different transfer distances further revealed that local populations had genetics-based leaf economic traits that made them adapted to the local site (Grady et al. 2013).

(2) Lower elevation source populations are currently better adapted to temperatures that can exceed 50 °C than higher elevation populations where temperatures are cooler. This finding of local adaptation is supported by the fact that populations like the low-elevation field site on the Lower Colorado River (i.e., sites that have a long, hot growing season) achieve higher ANPP than populations from higher mountain populations (i.e., sites that are genetically adapted to frost and a shorter growing season; fig. 8). However, with ongoing and projected climate change, local stock is expected to become maladapted.

(3) The slope of the regression line shows the sensitivity of cottonwoods from the source populations used at the planting site to temperature (mean annual maximum temperature, MAMT) (figs. 8, 9). Foundation species characterized by high sensitivity (i.e., with a steep regression slope) are key candidates for conservation/restoration strategies because of their sensitivity to climate change. Less sensitive species with a shallow regression slope might receive lower priority (Grady et al. 2011). In other words, populations of species with a more shallow regression slope can be transferred greater distances without suffering a loss in productivity compared with populations of species with a steep slope.

(4) The trait-based variance of growth is illustrated by standard error vertical bars in figure 8, which represents a population's potential to adapt to climate change. Thus, for any population shown in this figure, if the predicted change in climate is less than the variance, then the population has the potential to adapt (evolve) to predicted changes. Such rapid evolution has been demonstrated in pinyon pine (Sthultz et al. 2009). After the record drought in 2002 that resulted in landscape-level mortality, the genetic composition of the survivors was significantly different than the populations prior to the drought; similar findings have been found with other plants in response to rapid climate shifts (e.g., Franks et al. 2007) and introduced mammalian herbivores (Smith et al. 2015).

Populations with high genetic variation in climate change related functional traits allow rapid evolution to new conditions. However, if predicted climate changes exceed the genetic variation in a population's climate change related functional traits, then that population is likely to die out once conditions exceed the ability of all members to cope with the hotter environment. In other words, the pace of climate change leaves little room for adaptation beyond existing genetic variation. Thus, in regions of the world where global change is highest, it would be a grave mistake to depend upon rapid evolution to solve this problem; once the trait variation in a population is exceeded (e.g., temperature or drought tolerance), the local population will likely be extirpated. Genotypes already adapted to a hotter environment represent the logical source of stock for long-term restoration in an altered environment. Thus, if the temperature of the planting site is projected to increase by 2 °C, source populations already adapted to sites that are already 2 °C warmer than the planting site would be good sources of propagation stock.

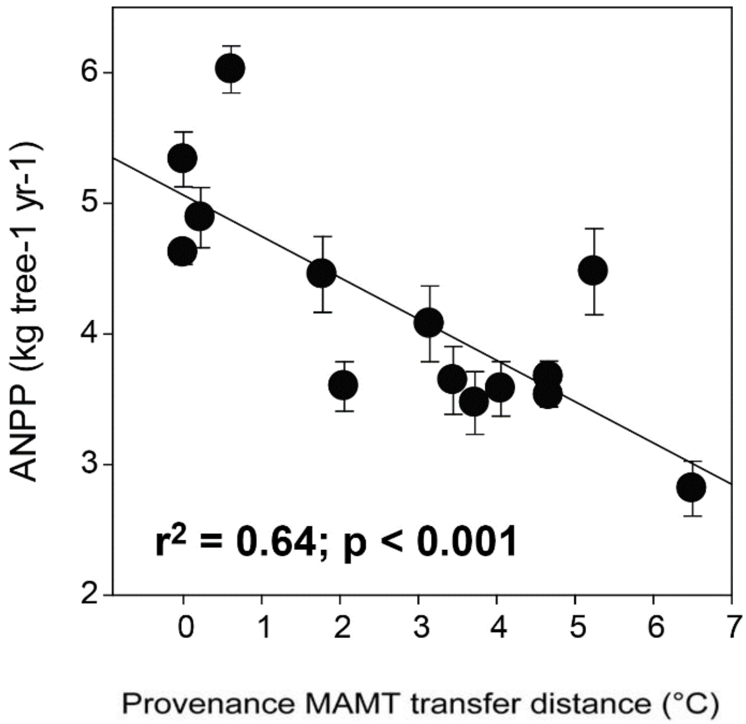

Figure 8—At a low elevation site at the lower edge of the distribution of *P. fremontii,* mean Aboveground Net Primary Productivity (ANPP) per population is plotted as a function of the temperature transfer distance of the source population to the restoration site. A transfer distance of 0 °C indicates a source population from a hot site similar to the hot planting site at low elevation. A transfer distance of 6.5 °C indicates a cooler, high elevation mountain population transferred to the hotter, lower desert planting site and a mismatch in environmental conditions that each population is locally adapted to (adapted from Grady et al. 2011).

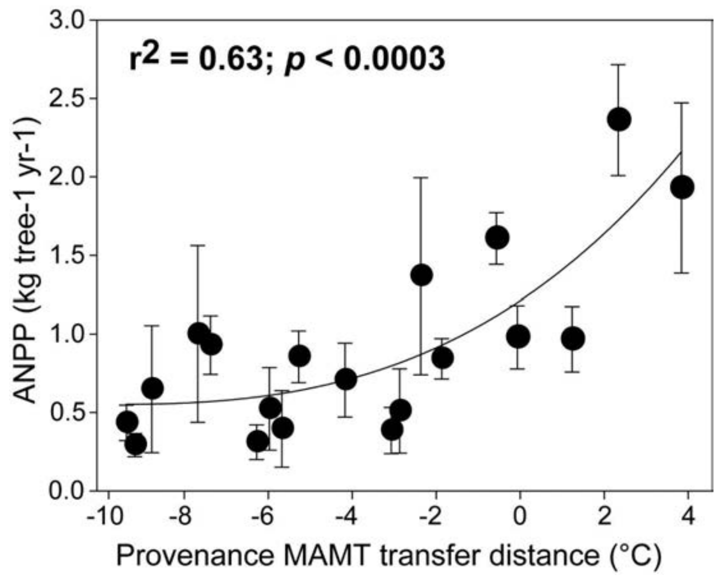

Figure 9—At a high elevation site nearer the upper, cold edge of the distribution of *P. fremontii,* mean Aboveground Net Primary Productivity (ANPP) per population is plotted as a function of the temperature transfer distance of the source population to the restoration site. A transfer distance of 0 °C indicates a source population from a cooler site similar to the cooler planting site at high elevation. A transfer distance of -10 °C indicates a hotter, low elevation population transferred to the cooler, higher elevation planting site and a mismatch in environmental conditions that each population is locally adapted to (adapted from Grady et al. 2015).

The problem of identifying which populations to use in restoration is especially grave at the lower edge of the species' distribution, because there are no other populations that are currently known to exist in even hotter environments. In other words, managers may have few options for finding individual genotypes or populations that are already adapted to what the environment will become in the future. This suggests that efforts to restore cottonwoods at the lower edge of their distribution are likely to fail with further climate change that exceeds the physiological tolerances of the species that would be reflected in widespread mortality and recruitment failure. In such cases, researchers and managers should use field trials to identify and experimentally deploy the most functionally equivalent species that is adapted to an even hotter environment to potentially fill the functional role of cottonwood as a foundation species (e.g., perhaps mesquite). Although not a preferred solution, this may be the best option available to achieve some restoration goals of supporting community biodiversity and providing ecosystem services.

(5) While the options for lower elevation edge populations facing climate change may be limited due to a lack of genetically appropriate stock, higher elevation populations have more promising options, because there are numerous populations at lower elevations that are already adapted to a hotter environment. For example, in another field trial study conducted near the upper end of the elevational distribution of Fremont cottonwood, Grady et al. (2015) demonstrated local adaptation in which high-elevation populations outperformed low-elevation populations in the high-elevation garden (fig. 9). With projected climate changes, these high-elevation populations would also be expected to become maladapted to the new environment. Because numerous populations live at lower elevations with a hotter environment, managers have the opportunity to select among these populations to find the one(s) that are the best candidates to plant at a higher-elevation site undergoing climate change, by considering both transfer distance that equates with projected warming as well as similarities in site characteristics (e.g., soils).

Using the information in this graph, we can select source populations for a predicted level of climate change. For example, if temperature is predicted to increase by 3 °C at the high-elevation planting site, using the empirical data from this graph, populations from lower elevation sites that have evolved for a 3 °C hotter environment should do best in the hotter future environment of the high-elevation planting site (see -3 °C populations indicating a hotter to cooler temperature transfer distance of 3 °C). Given our findings, we recommend a maximum 3 °C transfer distance to balance success between current and future climates. While trees from greater transfer distance would be better adapted to even hotter environments, trees would suffer from productivity losses in the current cooler environment. However, transfer distances of ≤ 3 °C would suffer minimal productivity losses in the current environment and would perform best in a 3 °C warmer future environment. If climate change is projected to increase by 6 °C, a phased or stepwise approach over time is recommended, in which increasingly more temperature tolerant genotypes and populations are planted as the climate continues to change.

In combination, these findings provide powerful tools to assist managers in the selection of genotypes and source populations that can survive a given level of climate change. Based upon the above and other studies, we recommend planting a combination of local stock for the current environment and stock from sites that are 1-3 °C hotter than the planting site, which would perform best in predicted future climates. The combination of these genotypes and populations provides important genetic variation that selection can act upon as climate change and other stressors determine who wins and who dies. Such strategies are especially important in regions where global changes are great such as the arid Southwest.

Insistence on using only local stock—where the mismatch of the local populations and genotypes with the new environment are great as in figure 6—greatly diminishes the probability of successful long-term restoration. We emphasize that field trials should be embedded in the sites where extensive restoration is proposed because they standardize soil conditions, community species pools, and other factors that could confound planting strategies based upon findings from sites farther removed from the actual restoration site.

The Importance of Maintaining Genetic Diversity for Associated Communities

We emphasize that restoration plantings must maintain high genetic diversity both for the foundation species as well as their associated communities. Genetic variation provides the essential building blocks for natural selection to act upon (Fisher 1930), thereby allowing species to evolve in response to unforeseen challenges including future climate change, new invasive species, and currently unknown biotic and abiotic effects.

For example, Sthultz et al. (2009) documented rapid evolution in pinyon pine, *Pinus edulis,* in response to record droughts, in which drought-tolerant genotypes suffered only 21 percent mortality, whereas drought intolerant genotypes from the same population suffered 68 percent mortality resulting in a rapid shift in the genetic structure of the population. Without such genetic variation naturally occurring in the population, the impacts of drought could have been much more severe, resulting in a major bottleneck event and potential local extinction had all genotypes been drought intolerant. These and other studies clearly demonstrate the importance of maintaining high genetic variation in foundation species populations, even when the roles of genetics in community diversity and the environmental conditions likely to favor certain foundation species traits are poorly understood.

For example, Orians and Fritz (1996) showed that under one set of environmental conditions, one group of willows was found to be resistant to insect herbivory, but under another set of environmental conditions, another group of willows was most resistant to insect herbivores. Similarly, antagonistic pleiotropy refers to a single gene controlling many traits, with at least one trait conferring positive fitness effects and at least one resulting in negative impacts to the organism (Yanchuk et al. 2011). For example, a plant may exhibit suboptimal performance under normal environmental conditions, but outperform the "optimal" genotypes under conditions of extreme stress. Planting only the most productive genotypes tested under "normal" conditions could result in large losses under drought conditions. In other words, high genetic variation is an insurance policy against unknown future biotic and abiotic conditions.

In addition to high genetic diversity being crucial for the long-term survival of foundation and other species, many studies have also demonstrated that high genetic variation in foundation species promotes greater biodiversity in their associated communities (e.g., Ferrier et al. 2012; Crutsinger et al. 2006; Wimp et al. 2004). A primary reason for this relationship between genetic diversity in the foundation species and the diversity of the associated community is that different genotypes support different communities. Because many plant traits such as phenology, phytochemistry, growth, and morphology have a strong genetic component and can covary and interact with each other (Endler 1995), the combination of all these traits is referred to as the multivariate phenotype (Holeski et al. 2012). As this multivariate phenotype differs greatly among individual genotypes, it is not surprising that different organisms and communities are found on different plant genotypes and that interspecific indirect genetic effects (IIGEs) among species in a community further lead to differences in community assembly on different plant genotypes and evolution (Allan et al. 2012).

Because these multivariate plant traits and interactions are genetically based and passed to their offspring, the communities they support represent heritable traits that can be quantified (Keith et al. 2010; Shuster et al. 2006). Numerous studies of plants around the world from the tropics to the tundra and desert to alpine environments exhibit community heritability in which related individuals tend to support the same communities (review by Whitham et al. 2012). Numerous studies with cottonwoods have demonstrated the importance of individual genotypes supporting different communities of diverse organisms including mycorrhizal mutualists, decomposing soil fungi and bacteria, lichens, twig endophytes, and insects that are closely tied to the multivariate phenotypes of individual plant genotypes (Lamit et al. 2015).

Studies of other systems such as conifers and eucalypts have shown similar patterns. Gehring et al. (2014) showed that different genotypes of pinyon pine (*Pinus edulis*) supported different mycorrhizal communities, and Barbour et al. (2009) showed that different genotypes of blue gum (*Eucalyptus globulus*) supported different communities of arthropods and fungi. These genetic effects even extend to beavers (Bailey et al. 2004), elk (Bailey et al. 2007), nesting birds (Martinsen and Whitham 1994) and seed-eating birds and mammals (Christensen and Whitham 1993). Thus, if maintaining high biodiversity is desirable, managers can select for a combination of different genotypes and populations that support the greatest biodiversity.

Another important principle is the concept of managing foundation species for both genetic and community connectivity. Genetic connectivity considers the degree to which populations of foundation species share genes and associated alleles across their distribution. Similarly, community connectivity considers the degree to which populations of foundation species share communities across their distribution, which is a concept supported by the genotypic-community associations described above. Thus, it is critical to understand the biotic and abiotic factors that either facilitate or inhibit genetic connectivity in foundation species and how it may change over time in conjunction with environmental change.

Cushman et al. (2014) and Bothwell et al. (2017) recently demonstrated the utility of this approach via the identification of genetic corridors in two widely distributed foundation species, *P. fremontii* and *P. angustifolia.* Like many cottonwood species, both *P. fremontii* and *P. angustifolia* support large and diverse communities of organisms whose connectivity may depend, in part, on the maintenance of genetic connectivity in their respective foundation species. Although such studies are in their infancy, “managing for connectivity” in both foundation species and their associated communities constitutes a “best approach” for preserving biodiversity into the future.

In combination, maintaining high genetic diversity in restoration plantings is crucial to both the survival of the foundation species and the diverse communities they support. These findings illustrate why it is so important for restoration biologists to maintain high genetic diversity in plantings, not only to support evolutionary potential in the trees, but also to support a diverse community of organisms. Experimental field trials (common gardens) represent a robust tool that allows researchers to quantify these relationships and develop a truly “adaptive management” strategy that is based on both ecological and evolutionary principles.

Summary

Despite their intuitive appeal, ecological restoration policies encouraging the use of locally derived stock may be misguided in the face of climate change, invasive species, altered flood regimes, fire, and other biotic and abiotic challenges. Rapidly changing environmental conditions are likely to favor plant traits that are distinct from the historical conditions in which local species have evolved. Failure to recognize these crucial environmental differences can lead to long-term failure of restoration projects. While the focus of this chapter has been on riparian restoration, global change is a universal challenge and the same principles presented here apply to all ecosystems facing similar challenges worldwide.

We propose the following stepwise restoration approach to mitigate the impacts of global change:

(1) Focus first on regions that are recognized as undergoing rapid environmental changes such as riparian habitats in the American Southwest.

(2) Identify the foundation plant species that are recognized as the drivers of their respective ecosystems (e.g., cottonwoods, willows, and other dominant plants on the landscape). This is relatively easy to do as there is a wealth of published data and knowledge about the species that have the greatest functional roles in defining their respective ecosystems. As we continue to become more knowledgeable about these ecosystems in identifying hidden players such as endophytes, fungal mutualists, and pathogens (e.g., Bailey et al. 2005) that also play foundational roles, they can be added into more sophisticated studies.

(3) On Federal, State, and private lands where change is imminent or has already occurred, such as the Little Colorado River that is inundated by invasive tamarisk (*Tamarix* spp.) and camelthorn (*Alhagi maurorum*), field trials should be established to determine which populations and genotypes can survive in these altered environments. These plantings will be most effective if they comprise the range of genotypes present throughout the species' range, especially from hotter, drier climates that have similar hydrology, soils, and invasive species.

(4) Based on these field trials, a diverse set of adapted genotypes and source populations can be identified and propagated for use on adjacent lands that can best survive both current and future environmental conditions.

(5) Because different genotypes support different communities, individual genotypes and source populations can also be selected that support high biodiversity, community stability, and connectivity. Thus, as supporting high biodiversity (especially of sensitive and listed species) becomes increasingly important, managers can move from restoring at the level of species to restoring at the genotype level. They do this by using specific plant genotypes that are preferred by community members of special interest such as the southwestern willow flycatcher (Bangert et al. 2013). However, managers must be careful to avoid selecting only optimal genotypes for a specific purpose. As noted above, sometimes the suboptimal genotypes under one set of conditions outperform the optimal genotypes under another set of conditions.

The Southwest Experimental Garden Array funded by the National Science Foundation was specifically established to achieve these goals of incorporating genetics approaches in land management with the major vegetation types on the Federal, State, and private lands that the array encompasses. We emphasize that such genetic approaches have long been used in agricultural crops in which yield is the major desired outcome. The goals of wildland restoration generally emphasize creating native habitat to support high biodiversity are different than agricultural fields. But the genetic principles are the same, and land management can be improved using such methods. In the face of great environmental challenges, genetics-based approaches are cost effective and represent a key step forward to help mitigate the impacts of global environmental change.

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Chapter 5. The Watershed Continuum: A Conceptual Model of Fluvial-Riparian Ecosystems

Lawrence E. Stevens, R. Roy Johnson, and Christopher Estes

Mountains of music swell in the rivers, hills of music billow in the creeks, and meadows of music murmur in the rills that ripple over the rocks. Altogether, it is a symphony of multitudinous melodies. All this is the music of waters ... sounds that span the diapason from tempest to tinkling raindrop, from cataract to bubbling fountain.

John Wesley Powell 1895: 394, 397

Introduction

Terrestrial fluvial-riparian ecosystems (FREs) are riverine landscapes that integrate aquatic, riparian, and upland domains within watersheds, linking physical, biological, and cultural-economic processes (Tockner et al. 2002). FRE characteristics and processes intergrade through the entire watershed, from the river's headwaters to its terminus into an endorheic basin or the sea, and can extend far out into the submarine environment (e.g., Canals et al. 2009; Vannote et al. 1980; Ward et al. 2018; fig. 10). FREs include all sources of water that contribute to the basin's riverine ecosystem, including springs, surface runoff, lakes, and atmospheric sources, such as humidity and fog. Only an average of 2,120 km^3 (0.0002 percent) of the world's water exists in river systems at any given time (Shiklomanov 1993).

While rivers process only a tiny fraction of the Earth's fresh water and occupy only a minute proportion of the Earth's terrestrial surface, FREs are highly productive and ecologically interactive, often supporting diverse, densely packed biotic assemblages across short to vast time scales (Behrensmeyer et al. 1992; Sabo and Hagen 2012). Elevated biodiversity in FREs is linked to, and influenced by, many factors, including hydrogeomorphological, ecotonal, and shifting habitat mosaic effects (Gregory et al. 1991; Naiman and Décamps 1997; Naiman et al. 1987, 2005). Humans rely on FREs, and our species' evolutionary history and modern demography clearly demonstrate that reliance. As human domination of the Earth has progressed, rivers have been subjected to a host of anthropogenic alterations, including resource extraction, groundwater withdrawal, flow diversion and regulation, water quality degradation, and introduction of nonnative species. The need to integrate understanding of groundwater, fluvial, and lacustrine interactions within basins, and to sustainably manage rivers, has never been more urgent (Famiglietti 2014).

The natural dimensions and human impacts on FREs have stimulated a rich history of basic research, generating a vast literature, and prompting initiation of many local to international riparian research and stewardship organizations. Diverse ecohydrogeological models have been synthesized, tested, and predictively applied to FREs stewardship. Although sometimes biased by the expertise of the authors, these syntheses have advanced the collective understanding of FRE ecology. However, much remains to be learned. With the growing anthropogenic pressures on rivers and associated riparian ecosystems, we require an expansive, hierarchically organized conceptual framework to generate advanced understanding and management of FREs.

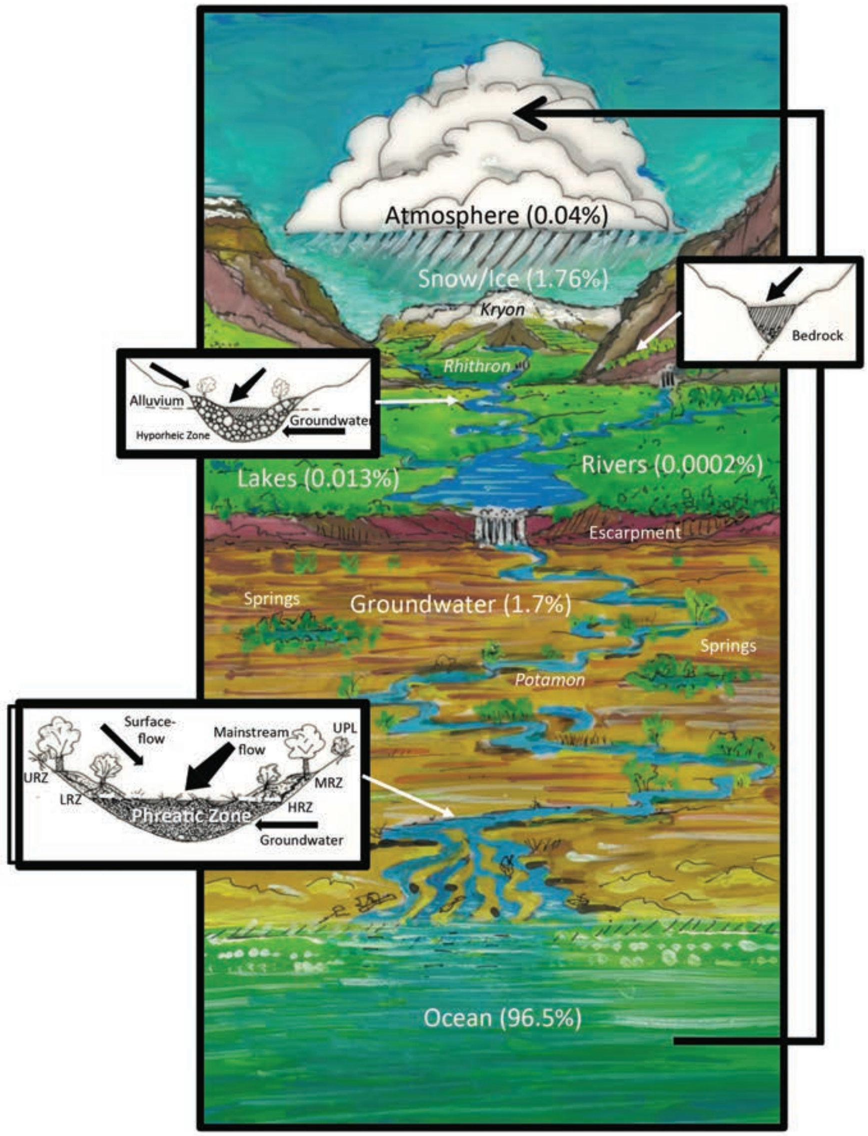

Figure 10—The hydrological cycle, emphasizing stream-order transition and lateral processes within watersheds between constrained, alluvial, and lentic reaches. Numbers are percent of Earth's water in different settings (Shiklomanov 1993). Illustration by Lawrence E. Stevens.

Here, we introduce the watershed continuum model (WCM) to expand interdisciplinary collaboration in ecological resource watershed system science and to improve science-based natural resource stewardship. The WCM describes matter, energy, and socio-values subsidy exchange within an entire FRE basin through the physical, ecological, and cultural processes that influence ecosystem geomorphology, biota, and society, across four-dimensional temporal and spatial scales. FRE connectivity extends from groundwater emergence and surface-derived flow, across the watershed, to the mouth, into the receiving basin or sea, and into the atmosphere. This integrative approach has been emphasized by Annear et al. (2004, 2009), Hynes (1975), Stanford (1998), Ward (1989), and others to interrelate physical hydrology, geochemistry, geomorphology, sedimentology, and aquatic and riparian domain linkage. We further advance this framework by linking aquatic and biological domains across stream order within watersheds (*sensu* Horton 1945; Strahler 1957) and over time.

We restrict our discussion in this chapter to natural, unmanipulated FRE processes and characteristics, independent of the many anthropogenic impacts affecting FRE ecology. Anthropogenic impacts on rivers are widespread and merit much more scientific and societal attention, but we focus on natural processes and components to clarify and describe the interdisciplinary integration of hydrogeology, geomorphology, ecosystem ecology, and evolution. These components collectively interact within the watershed to generate FRE processes, assemblage structure, and stewardship decision-making. It is impossible here to do full justice to the broad array of FRE characteristics, themes, and models that contribute to this complex view; rather, we seek to provide a synthesis of patterns and insights, and to identify information gaps that affect FRE ecology and evolution. Collectively, the WCM couples multidirectional, trans-temporal material, and energy interchange from uplands to fluvial habitats throughout the basin, ultimately influencing the evolution of both aquatic and riparian FRE biota, as well as human cultures.

Although our focus is global, we draw on examples from western North America, especially the Colorado River basin in the American Southwest where we have done much of our work. We begin this chapter by summarizing basic fluvial-riparian concepts, and then we describe the physical, biological, and ecological elements that provide the basis for the WCM. Next, we review and discuss biologically focused FRE models using illustrated schema. Finally, we discuss issues that remain understudied or unresolved. Given the focus of this publication, and because integration of aquatic and terrestrial domains across scales within FRE ecology is complex and incomplete, we add additional emphasis on the riparian domain. We recommend further considerations and actions and discuss conceptual model dimensions and limitations.

We present the WCM to enhance ecological resource watershed system science and improve natural resources management. Full predictive capacity of the WCM is not likely to be soon forthcoming, because some components have not been measured, numerically modeled, or assessed in relation to process interactions. Nonetheless, we welcome insight and collaboration from all who are interested in expanding and refining this integration, and we hope to stimulate further research needed to advance FRE ecology and stewardship.

Physical FRE Elements and Processes

Overview

Many supporting models have been proposed to describe the array of physical processes associated with FREs. FREs are terrestrial dendritic surface-water flow paths transporting matter and energy downslope through their channels within watersheds, with flow contributed by groundwater and multiple surface water sources. Riverine processes and geomorphology vary in relation to the geologic setting, gradient, flow, channel bed material, and sediment loads (Hey et al. 1982; Leopold and Mattuck 1953; Li et al. 1976; Schumm 1985; Tinkler and Wohl 1998; Wohl 2010).

We initiate this discussion with a brief overview of physical flow and hydrogeomorphological models (HGMs). Early work in fluvial hydrogeomorphology focused on classification and process-based modeling at microsite to basin-wide spatial scales, and on understanding and describing the interrelationships between basin characteristics and climate-based flow on channel form, function, and development. These

HGM topics remain under intensive study and debate, particularly in relation to physical FRE management of rivers (e.g., Leopold et al. 1964; Leopold and Maddock 1953; Lewin 1978; Morisawa 1968; Schumm 1985). More recent emphases have shifted to tributary- and reach-based HGMs, with growing recognition that channel geomorphology and function depend on the extent of geologic constraint (Bellmore and Baxter 2014; Montgomery 1999; Sabo and Hagen 2012). In some situations, bio-ecological processes may reciprocally influence FRE geomorphology (e.g., Beschta and Ripple 2006). We follow this literature review with a description of how these elements relate in an integrated watershed approach.

Fluvial Hydrogeomorphology

While trophic cascades can influence fluvial geomorphology, study of "bottom-up" physical processes has stimulated a long history of HGMs and much ongoing effort to improve understanding and description of FREs. The initial foci of HGMs was on understanding channel landform organization and distribution and the processes that generate them (e.g., Buffington and Montgomery 2013; Knighton 1998; Rosgen 1994, 2008). HGMs were primarily considered in unconstrained alluvial rivers, but Rosgen (1996), Tinkler and Wohl (1998), Wohl (2010), and others extended channel formation description to constrained and bedrock-dominated settings. Additional focus has been placed on the extent of groundwater-dependence (Eamus and Froend 2006). The Committee on Riparian Zone Functioning and Strategies for Management (2002), Giller and Malmquist (1998), Malanson (1993), and Naiman and Décamps (1997) also emphasized the importance of lentic habitat (river-related lakes and wetlands) relationships to rivers. That discussion was expanded by the Instream Flow Council (Annear et al. 2004, 2009) to include variability in lateral and vertical flow in relation to instream flow assessment and management strategies, particularly for salmonid fish.

Discipline-based or regional specificity has limited the applicability of some HGMs. For example, Bennett and Simon (2004) focused on engineering-based analyses of riparian ecology, without considering the implications of evolutionary ecology, as discussed by Steiger et al. (2005) and Stromberg et al. (2004). Harvey and Gooseff (2015) recognized this difficulty and emphasized integration of physical and ecological models. However, increased interdisciplinary dialogue is needed to further integrate hydrogeomorphic models with fluvial engineering, aquatic and riparian ecosystem ecology, as well as socio-cultural, economic, and evolutionary models (e.g., Lubinski 1993).

The process domain concept (Bellmore and Baxter 2014; Montgomery 1999) shifted attention to FRE structure and process from the spatial scale of the basin to that of reaches. Alluvial reaches often support broader riparian zones and filter, store, and process organic matter from upstream (Nilsson and Svedmark 2002). In contrast, constrained reaches function more as conduits for material transport. At a coarse scale, Bellmore and Baxter (2014) compared aquatic-riparian organic matter dynamics in five confined versus five alluvial reaches, reporting that the former had twice the allochthonous (extrinsic) organic input, but had a reduced capacity to retain and process that material. As an across-scale approach, the process domain concept posits that spatial variation among geophysical processes shapes the disturbance regime, ecosystem structure, and ecosystem dynamics. This approach resolves some of the limitations of the river continuum concept (Vannote et al. 1980; below) that were attributed by Minshall et al. (1983) to the influences of location-specific lithology and geomorphology.

FRE structure is strongly dominated by physical processes operating in a bottom-up fashion to shape riverine landforms and the habitats on which aquatic and riparian assemblages develop (figs. 10, 11). The geologic context and climate of the watershed control basin position, geomorphology, topography (elevation), configuration, and the supporting aquifers are physical state variables. Collectively, those physical state variables regulate flow, hydrography, water quality, and sediment transport. They also generate the FRE templates that are organized through hierarchically flow-linked reaches (Fisher et al. 1998; Montgomery 1999; Vannote et al. 1980). Ecologically, nearly all terrestrial FREs exhibit four-dimensional subsidy exchange, including: (1) upslope eolian and zoochorous transport processes; (2) capillary rise of groundwater through fine sediments; (3) lateral and vertical transport through the watershed, often with subsurface or tidal influences at the confluence of tributaries, in parallel flow with head from adjacent rivers, in lowermost reaches, and into the atmosphere; and (4) artesian groundwater or, in very low-gradient reaches, downstream tributary flooding that initiates upstream-directed flow (Ward 1989; figs. 10, 11).

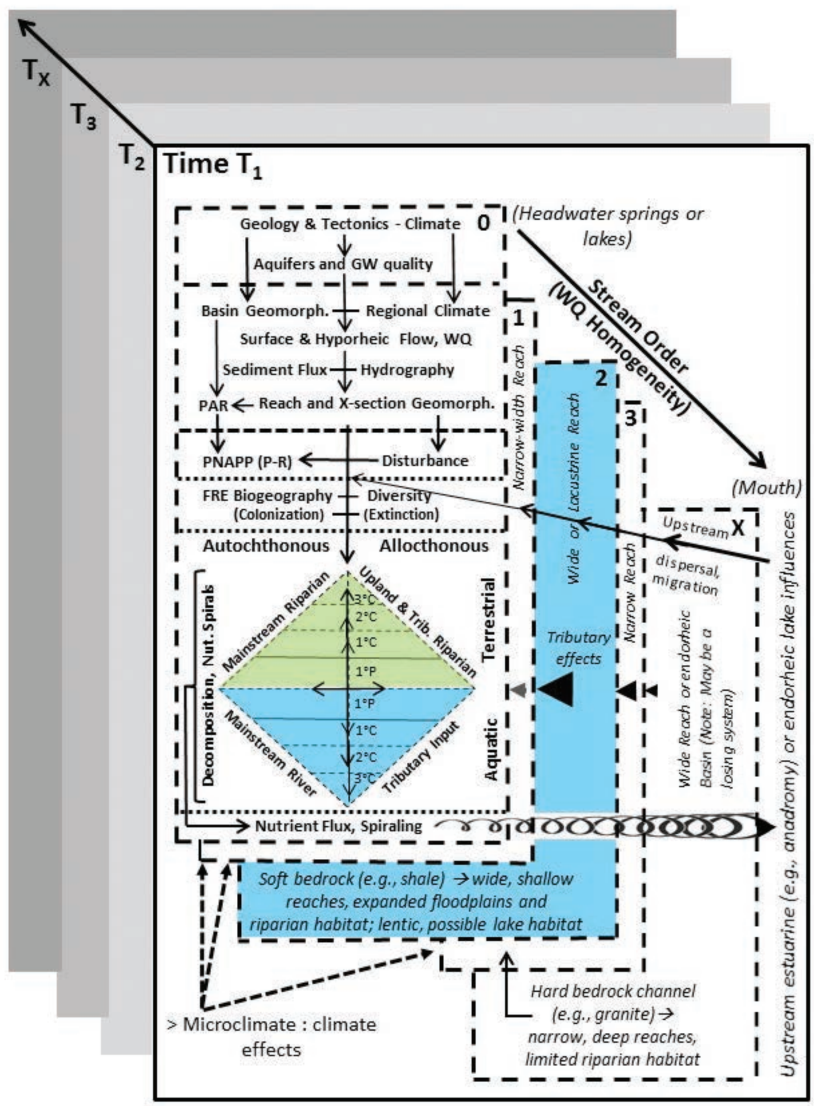

Figure 11—Conceptual fluvial-riparian ecosystems model depicting interactions among independent and dependent physical and ecological variables and processes, across stream order (0 headwaters to X- mouth) and time (T1 to TX). Black arrow points indicate relative impacts of tributaries of different sizes.

A comprehensive FRE conceptual model must integrate these dimensions over time among aquatic, riparian, and upland domains within the watershed. It also must link independent and dependent physical, biological, and interacting ecosystem processes. All of these processes respond dynamically toward equilibration of fluvial boundary conditions, matter transport, and energy dynamics. Consequently, we depict FREs as primarily driven by physical factors. This emphasizes the foundational role of physical processes with dependent biotic composition, structure, function, and trophic interactions (figs. 10, 11). Although little-discussed in the RCC, lower stream order FRE changes often occur in a punctuated, stepped, or reach-boundary fashion, as flowing water emerges from groundwater and FREs receive tributary contributions. In contrast, higher order streams terminating at the confluence with the sea (but not those terminating in endorheic basins) generally change more gradually, both spatially and over time.

Fluvial Hydrogeology Model Elements

Overview

Hydrogeological influences on FREs vary in relation to latitude, elevation, season, stream order, and the derivation of reach source waters (e.g., precipitation and runoff, groundwater, springs, or lakes). This variation exists on scales ranging from microsite to local stream cross sections, among reaches, across entire basins, and through different climate/humidity regimes (figs. 10, 11). Impacts also vary as a result of differing regional and global tectonic histories. Hydrogeological FRE models have developed incrementally over the past century, building on each other without any major concept-changing epiphanies. Careful experimentation and analyses have gradually expanded the logic, empirical background, and predictive capabilities of groundwater and surface water models. However, additional research is needed to incorporate ecological and evolutionary concepts into fluvial ecohydrology.

Basin Geography

The geologic setting in which groundwater and surface water basins exist control FRE form and function. Tectonics, geography, sources, elevation range, basin structure, climate, parent rock stratigraphy, glacial rebound status, structural geology, and aspect all influence rivers to varying degrees (figs. 10, 11). Tectonic controls on river development were reviewed by Holbrook and Schumm (1999), but coarse-scale configuration of groundwater and surface water basins under different tectonic regimes needs further elucidation. Large, high order rivers typically interconnect intracontinental basins through periods of tectonic quiescence. Such periods of tectonic inactivity can persist across evolutionary and geologic time scales. Integration of large river systems across complex landscapes has involved 10+ million years in the Nile, Colorado, Rio Grande, and Mississippi River basins (Mack et al. 2006; Timmons and Karlstrom 2012; Wickert et al. 2013; Woodward et al. 2007).

Orogenies that dominate continental margins often produce abundant low to moderate stream order basins. In contrast, extensional terrains in East Africa and western North America generate streams of moderate order from high elevations that discharge into isolated, endorheic basins. While aquifers that develop in these landscapes often are assumed to be constrained by surface catchment boundaries (e.g., Springer et al. 2016), such is not always the case. For example, groundwater passes more than 200 km beneath several endorheic basins in Nevada before emerging in the lower White River (Winograd and Thordarson 1975).

Tributary Effects

Tributary influences on mainstream rivers vary in relation to stream order and the extent of differences in hydrography, water quality, sediment loading, and biota (Benda et al. 2004b; Dye 2010; Rice et al. 2001; Thorpe and DeLong 1994; Ward and Stanford 1983). Habitat complexity at tributary confluences increases ecological productivity and biodiversity, and it sustains biogeographic habitat connectivity in four dimensions (Naiman et al. 1993; Rice et al. 2008; Sabo and Hagen 2012; Stevens and Ayers 2002; Thorpe and DeLong 1994). Tributary impacts on mainstream water quality are likely greatest when flows of the two are similar; however, differences in biota may follow the opposite pattern: aquatic macroinvertebrates in a small, springfed tributary may substantially differ from that in the large, adjacent mainstream. In figure 11, we depict the magnitude of tributary influences as dark triangles of varying size, but we also note that the impacts of large or influential tributaries occur across mainstream order.

Source Waters

Along with precipitation, snowmelt, and lake source contributions, springs play a far more important role in FRE ecology than has previously been recognized. Many of the world's major rivers arise from discrete springs, springfed marshes, or groundwater-fed lakes (Junghans 2016). Rivers with spring-sourced baseflow include the Amazon, Colorado, Mississipi, Rhine, Volga, Murray, and many others. For example, the Mississippi River heads in springfed Nicollet Creek and two tributaries of Elk Lake in Minnesota, as well as springfed Lake Itasca. The Missouri River heads at Browers Spring on the Montana-Idaho border. One of its major tributaries, the Yellowstone River, heads in the Absaroka Mountains in northwestern Wyoming, passes through Yellowstone Lake (which is partially springfed by biota-rich thermal vents; Morgan et al. 2007), and eventually joins the Missouri River. Both the Nile and Colorado Rivers head in springfed helocrene marshes. Lake Tahoe in the Sierra Nevada Mountains of California and Nevada receives about 0.05 km^3/year from groundwater inflow (0.033 percent of its volume; Thedol 1997). That lake is the source of the Truckee River, which terminates in endorheic (and also partially springfed) Pyramid Lake in Nevada's Great Basin Desert. Both subaerial and subaqueous springs contribute to river flows, although subaqueous springs are difficult to quantify and study (Springer and Stevens 2009). Headwater springs often are distinctive habitats with unique water quality that may influence mainstream processes, such as imprinting among larval fish (Teears 2016; but see Muller-Schwarze 2006), and they often provide headwater baseflow for rivers in non-ice-dominated regions.

FRE hydrologic models often assume that river baseflows arise from diffuse emergence of groundwater. Such a generalized assumption is questionable, because geologic structure usually brings groundwater to the surface at discrete points (springs), generating focused point-sources of groundwater. In 2016, the senior author found that the headwaters sources of the Fiume Tagliamento in northeastern Italy, the last undammed river in the Alps, arose from a suite of high-elevation rheocrene and hillslope springs, with some summertime flow augmented by snowfield runoff. This was despite reports of its source flow arising from diffuse groundwater (Tockner et al. 2003). In addition, rivers often support infiltration, adding to groundwater supplies. However, the assumption that diffuse groundwater largely provides the baseflow sources of rivers has hampered integration of knowledge of groundwater-surface water interactions in FRE stewardship, obscured understanding of the contributions of springs to surface water, and biased river science and policy (e.g., U.S. Army Corps of Engineers and Environmental Protection Agency 2015).

Reaches and Stream Order

River drainage networks are often subdivided into segments and reaches, which are important organizational units of FREs. We define a river segment (*sensu* Stevens et al. 1997b) as one or a group of river reaches that are collectively subject to an abrupt change (usually) in one or more ecosystem characteristics (e.g., water temperature, geochemistry, suspended sediment load, or gamma diversity species assemblage). Such changes often are introduced by a tributary, thereby affecting downstream ecology (Bruns et al. 1984; Rice et al. 2008). Reaches lie within segments and are distinguished geomorphologically on the basis of differences in parent rock geology, shoreline erodibility, slope (gradient), and thalweg position. Schmidt and Graf (1990) analyzed several hundred cross sections to define the geomorphic reaches of the Colorado River in Grand Canyon in relation to bedrock geology. Thus, both segment and reach boundaries commonly occur at tributary confluences with the mainstream. Division of watersheds into sub-basins at various spatial scales (e.g., the hydrologic unit classification system used in the United States) may not well reflect reach geomorphology or segment configuration. Agreement on definition of these terms remains an important issue for further progress in FRE ecology.

FRE complexity is defined, in part, by stream order, which increases when two stream channels of the same magnitude meet (Horton 1945; Strahler 1957) (fig. 11). We regard springs as "zero order" streams, and springbrooks are typically first order channels. Stream order increases erratically downstream: the largest rivers can exceed a stream order of 10. For example, with more than 1,000 tributaries more than 1,000 km long, the Amazon River is considered to be a 12th order stream. FREs lacustrine reaches can occur at any order within a basin, through which limnological processes like thermal or chemical stratification can influence downstream flow and geochemistry.

Higher stream order FRE characteristics and processes vary primarily in relation to the extent of geologic constraint, stream order, and reach geometry (figs. 10, 11). In alluvial rivers, reach characteristics are generally influenced by stream order, exhibiting distinctive, sinuous meanders, or braiding. Shales and other soft bedrock strata allow reach width to expand in unconstrained river channels, resulting in less variation in flow dynamics, reduced responses to aspect and microhabitat effects, and reduced species extinction probability. Such streams usually have increased solar energy, productivity, and colonization potential. In contrast, in geologically constrained reaches, the bedrock geology and geologic structure exert stronger influences over reach geometry. Narrow reaches often form in harder bedrock, where their channels sustain greater variation in flow, stronger responses to aspect, reduced solar energy budgets and productivity, and decreased colonization potential coupled with increased species extinction probability. Riparian zones in narrow, constrained reaches typically exist in a state of perpetual succession, a process suspended by high disturbance intensity (Campbell and Green 1968).

Large rivers are typically high order streams, and the large quantity of water they transport exert unique ecological influences on their ecology, buffering changes in water temperature, geochemistry, and equilibration timing. Johnson et al. (1995) questioned whether the RCC that Vannote et al. (1980) applied to large (high order) rivers, due to the temporally and spatially nested hierarchical organization, the potentially increasing influences of physical bottom-up ecological controls, increased equilibration time, and the inadequacy of understanding and modeling physical process interactions.

Water Quality

River water quality varies across lithology, latitude, elevation, humidity province, season, and stream order within basins, and among reaches, and springs and lakes can influence river waters. Water quality characteristics transition over stream order and are important determinants of macrophyte composition and life history cues for aquatic macroinvertebrates, fish, and amphibians,. In turn, they influence food web linkage (but see Heino et al. 2015) and riparian groundwater quality. Surface flow geochemistry generally dominates higher stream orders, with river water quality change occurring at tributary confluences (Dye 2010), and to a lesser extent in side channels and shallow, low-velocity shoreline habitats. Limnologically influenced water quality dominates lake-sourced rivers, but we know of little research on natural downriver responses to such alteration. River water trends toward a universal quality across stream order, creating relative similar geochemistry among the world's major rivers at their mouths. However, the contributions and evolution of FRE water quality depends in large measure on subbasin geology and the relative contributions of tributaries (e.g., Giller and Malmqvist 1998; Kabede et al. 2005).

As zero order streams, springs often exhibit strikingly different temperature and geochemical characteristics than those in the adjacent higher order streams with which they are confluent (e.g., Lowe and Likens 2005). The ecological transition from headwater springs into first order streams is highly individualistic, often occurring at a chemically and thermally discrete distance from the source (Morrison et al. 2013). The quantity and quality of riverside or in-stream springs, as well as seasonal flow changes driven by precipitation, also can affect stream channel geomorphology. For example, limestone (travertine)-depositing springs shape stream geomorphology by precipitation of calcium carbonate ($CaCO_3$), forming dams and creating pools similar to those behind beaver dams (Cantonati et al. 2016). Such springs have received global attention as important aquatic ecosystems, and often add much dissolved load to rivers. They also affect downstream channel landforms, morphology, and interstitial pore space in mainstream bed sediments.

Geothermal springs, such as those at Yellowstone National Park and Mammoth Hot Springs, often deposit travertine (Sorey 1991). $CaCO_3$ precipitation rates in Fossil Creek in central Arizona exceeded 0.11 g/L of discharging flow, actively depositing travertine that shapes channel geomorphology (Malusa et al. 2003). Travertine deposition appears to be facilitated by algal growth, a deposition process that quickly oxidizes organic carbon, either through microbial activity or through as-yet-unrecognized chemical processes, enhancing the role of travertine deposition in CO2 release. Larger springs can dominate or strongly influence or dominate riverine water quality. For example, Silver Springs provides nearly 13 m^3/sec of baseflow to the Silver River in Florida (Odum 1957), and the newly discovered Shanay-Timpishka River is a large geothermal river arising on the floor of the Amazon basin (Ruzo 2016).

Hydrography

Flow, stage, and flow variability are the primary physical factors affecting FRE geomorphology and ecological processes and components (e.g., Junk et al. 1989; Tockner et al. 2000; Topping et al. 2003). Flows of most temperate and many tropical rivers vary seasonally, and factors that regulate flow include drainage area, geographic position, humidity province, underlying lithology and groundwater hydrology, vegetation cover, seasonality, and regional climate. Two useful analyses for understanding individual FREs are annual hygrographs (plots of the average daily flow and variation across days of the year) and flow duration curves (plots of the annual frequency distribution of flows). These plots vary in relation to climate, basin structure, and location within the basin. Such analyses are most robust when data represent long time series, clearly displaying interannual variability among wet and dry years. Failure to incorporate a sufficient duration of flow data and the range of variation can lead to serious mis-management of water. For example, the Colorado River was famously over-allocated among basin States in 1922, based on an insufficiently long flow time series that overemphasized wet-year data.

Physical disturbance strongly influences FRE geomorphology and biotic assemblages, but how does disturbance intensity vary across the channel and across stream order? Magilligan (1992) used HEC-2 flood modeling to describe variation in channel boundary shear stress and unit stream power on an array of stream channels across 2- to 500-year floods. She included analyses of channel cross section and distance downstream though basins with varying lithology, noting at least three-fold variation in flood power through the basin due to valley width, which is largely controlled by lithology. Wide valleys with broad, alluvial channels were subject to lower flood power, because of reduced percent of channel conveyance during large magnitude floods and decreased rates of depth change.

In contrast, such events generated increased flood power in narrow valleys with constrained channels, a pattern influenced both by basin size and by local controls, such as dams. Magilligan estimated minimum shear stress and unit stream power for "catastrophic" floods in humid, alluvial channels of 300 W/m2. Interestingly, she suggested that maximum flood impacts on channel geomorphology occur at discrete points in the reach or drainage and that such points are likely to shift over time (headward movement of such points seems most likely). Her insights suggest that geomorphic evolution of a drainage network occurs most intensively at a highly localized scale, affecting reach-based disturbance dynamics and FRE biotic assemblages.

Antecedent high flows exert lasting impacts on FRE structure and ecology. Foster et al. (1998) highlighted the importance of antecedent imprints from large, infrequent disturbances on ecosystem structure and function. Parsons et al. (2005) reported that large, infrequent floods following volcanic eruptions, wildfires, or hurricanes increased heterogeneity in fluvial riparian habitat, creating multiple trajectories for vegetation renewal along the Sabie River in Kruger National Park in South Africa and Mozambique. Thus, river geomorphology may continue to respond to flood impacts from the distant past—impacts that often are difficult to discern.

The frequency, duration, magnitude, and timing of low flows and dewatering events also are critical determinants of FRE ecology. If flows are sufficiently low, the river may become a series of pools, with limited connectivity, or may entirely desiccate. A full dewatering event can eliminate FRE macroinvertebrate and fish species, and prolonged dewatering alters a host of geomorphic and ecological processes and characteristics. However, long-term flow data on droughts and their ecological effects are limited and difficult to correlate. These data challenges sometimes can be addressed through dendrochronology. Long-term flow modeling of dendrochronology within basins among a wide array of arid to humid environments around the world have provided insight into drought frequency and duration (e.g., Akkemik et al. 2004; Case and MacDonald 2003; D'Arrigo et al. 2009; Melo et al. 2012; Therrell and Bialecki 2015). Such studies have the advantage not only of evaluating wet years and extreme precipitation events, but also of drought characteristics that strongly affect water resource supplies management. The studies highlight the need for adaptive strategies to cope with climate change (e.g., U.S. Bureau of Reclamation 2012).

The impacts of natural, regular, short-term stage fluctuations in rivers are generally poorly known, but they are of great consequence in rivers impounded for hydroelectric power production (e.g., Kennedy et al. 2016). Natural semi-daily tidal bores are a common phenomenon in the lowermost reaches of low-gradient rivers that reach the sea, and less regular stage fluctuations may extend upstream into low-gradient streams that open into wind-influenced lakes. Daily variation in flow stage in such settings may desiccate or freeze macrophytes, or macroinvertebrate habitats, and eggs. It may also interrupt aquatic and riparian faunal feeding and other behaviors, leading to reduced or fluctuating primary and secondary consumer production. Given the frequency of hydropower-driven flow fluctuation impacts on regulated rivers, understanding the effects of natural fluctuating flows is an important topic for future FRE and WCM research.

Sedimentology

The erosion, and deposition, of bed, suspended, dissolved loads, and flotsam are related to watershed geology, aquifer properties, flow dynamics, and channel configuration. In addition, ice processes, upland wildfire, forest pest insect outbreaks, and overgrazing affect fluvial sedimentation and nutrient transport (e.g., Bormann and Likens 1979). Cumulatively, including anthropogenic materials, the world's rivers deposit about 20 billion metric tonnes of solid material into the sea each year (Gray and Simões 2008). However, rather than being solely a function of basin area, sediment deposition is disproportionately the result of discharge from thousands of small, relatively high-gradient rivers (i.e., drainage areas < 10,000 km^2) that open directly into the ocean (Milliman and Syvinski 1992). Large rivers deposit proportionally less sediment due to subaqueous storage in deltas. Alluvial reaches often have relatively uniform bed materials and channel landform configuration, and often are closer to equilibrium than are constrained channels. Parent rock lithology, geologic structure, and less predictable gradients and bed loads control reach characteristics and channel geometry in constrained channels (e.g., Hey 1982; Schmidt and Graf 1990; Tinkler and Wohl 1998; Vogel 1981).

Models of sediment deposition and erosion are diverse (reviewed by Merritt et al. 2003, among others), and can provide adequate two-dimensional prediction of suspended sediment transport through channels with varying bed roughness, channel steepness, and sediment transport. Some two-dimensional models accurately predict flow and sediment transport for some rivers (e.g., the Glen and Grand Canyon reaches of the Colorado River) (Rueda 2015). However, most rivers have insufficient historical flow and hydrographic resolution to permit high-precision modeling (Alley et al. 2013). Variation in turbulence, sheer stress, transport capacity, and bed and suspended loads results in more highly variable channel landform and FRE habitat variation among bedrock-defined reaches. In addition, three-dimensional flow modeling is needed to relate discharge and sediment transport to sedimentation and channel landform development.

FRE suspended sediment transport is a power function of flow, increasing and varying dramatically during floods (e.g., Leopold et al. 1964; Topping et al. 2003, 2013). Suspended and bed load grain sizes vary from clay, silt, sand, and pea gravel to boulders and bedrock, typically with grain size negatively related to elevation and stream order. Fluvial sediment grain sizes are non-randomly erodible, with clay and cobble-to-larger sediments far less erodible than are silt or sand (Hjulstrøm 1939). Suspended sediments are actively redistributed by high flows and strongly influence FRE channel landforms and the availability and quality of germination and maturation microhabitats needed for riparian plant and faunal assemblages. Deposition of fine-grained sediments tends to occur on the falling limb of flood hydrographs and, in constrained reaches, at discrete microsites, commonly in relation to release of channel constrictions (e.g., Schmidt and Graf 1990; Topping et al. 2013).

FREs also transport extensive allochthonous (upland- and upstream-derived) and autochthonous dissolved and fine-to-coarse organic matter, depositing large quantities that may affect flow, channel geomorphology, and riparian habitat development (Stevens 1997; Tockner et al. 2003). A vast literature exists on the quantity, distribution, and ecological roles of woody debris (reviewed by Stevens 1997). For example, coarse woody debris provides refugia for aquatic macroinvertebrate larvae during upper Colorado River basin floods.

Channel Geometry

Geomorphologists have identified an array of river channel types based on the extent of geologic constraint: geologically unconfined (alluvial) and highly erodible channels and mobile beds differ geomorphically and ecologically from constrained (fixed shoreline to canyon-bound) channels with relatively immobile (low erodibility) margins and beds (e.g., Rosgen 1996; Tinkler and Wohl 1999) (figs. 10, 11). Alluvial streams often have relatively broad, meandering, lower gradient, shallow channels with highly mobile beds, and sometimes with anastomose channels. In contrast, geologically constrained rivers often have narrow, low-sinuosity channels with higher slopes, deep channels, and largely immobile beds. The primary variables of interest in understanding development of fluvial channels include: channel slope, the stage-to-discharge relationship, the hydrograph, bed materials, channel roughness, and the history and frequency of catastrophic disturbances (e.g., ice damming, earthquakes, volcanism and lava flows, and mass wasting slope failures) (Leopold et al. 1964).

Many channel classification systems have been proposed (reviewed by Buffington and Montgomery 2013), providing a robust description of channel types and ranging from non-symmetrical spring channels (e.g. Griffiths 2008) and bedrock channels (Tinkler and Wohl 1998), to a wide array of alluvial, highly sinuous, braided, or anastomosing channels (e.g., Rosgen 1996). From a stream ecology perspective, rivers can be divided into rhithron (headwaters) and potomon (lowland) zones.

Ward (1994) identified three low-order stream segment types in European headwater alpine habitats. Kryal (icemelt) reaches were characterized by glacially sourced flows with low temperatures, variable discharge, and turbidity; limited and largely allochthonous food sources; and limited benthic assemblages. Rhithral (channel stream) reaches were dominated by snowmelt hydrograph, varying temperature, and more complex and variable stream invertebrate assemblages. Krenal (springbrook) segments were steadier-flow, groundwater-dependent reaches, with high water clarity and water quality, dominated in his study by calcium carbonate, and supporting a richer assemblage of benthic macroinvertebrate species derived from multiple sources. Rhithron and potomon segments can be either constrained or alluvial. However, few studies have quantified or modeled channel geometry across stream order within basins in different tectonic settings or across latitude and additional comparative studies are needed to reveal broader scales of channel geomorphological organization and the implications on FRE ecology.

Springs that emerge along river channels can increase the frequency of bank failure and redirect mainstream current direction (L.E. Stevens, unpublished observations). Hillslope springs along the edges of smaller perennial or many ephemeral FREs may exert feedback on channel geomorphology by increasing dense wetland plant cover or woody phreatophytic shrubs and trees (plants rooted in the water table or on the phreatic surface) along adjacent low-order mainstream channels. Such vegetation may be sufficiently extensive to reduce stream power during floods, causing local deposition and geomorphic alteration of channel geometry.

Fluvial Landforms

Channel landforms include pools, eddy deposits, debris and cobble islands, side channels, paleo- and contemporary terraces, and other features, whose development can be mapped but not yet reliably modeled. Predicting fluvial landform development and responses to flow events remains limited, due in part to divergence among turbulence physics equations. Post-flood flow subsidence sequentially exposes an array of draining shoreline habitats and generates terrace development along alluvial, sand-dominated channels. As flow stage declines, lateral channels change from being through-flow features to being open at the downstream end, and then becoming isolated pools. This annual flood cycle allows a progression of vegetation, invertebrates, and fish assemblages changes (e.g., Stella et al. 2011). In constrained, sand-dominated segments, return current channels are distinctive side channel features that form in recirculating zones and mouth upstream (Schmidt and Graf 1990). Under low flows, these features can undergo fluvial hydrarch succession into marshes and, with prolonged dewatering, into terrestrial habitats (Nilsson 1984; Stevens et al. 1995).

Riparian vegetation is often strongly zoned in response to stage (fluvial groundwater availability) and the return frequency and magnitude of flooding or ice scour disturbances. Although definition of riparian terrace structure and habitat has been attempted several times in various hydrological and ecological contexts over the past century (Carothers et al. 1979; Hupp and Osterkamp 1988, 1996; Johnson 1991; Nilsson 1983; U.S. Natural Resources Conservation Service 2013), there appears to be little consensus in the river science community about terrace lexicon and classification. For example, the phrases "first overbank terrace," "geolittoral zone," and "hydrologically active zone" all apply to what we refer to (below) as the "lower riparian zone." Such lack of agreement on basic terminology impedes understanding of FRE landform and vegetation interactions. It also hinders interdisciplinary collaboration. Here and in the figures, we describe and define physical and biological terminology and concepts to help clarify, assess, and descriptively model the WCM as the basis for proposing this FRE model. In general, we prefer the simplest, most intuitive landform descriptions, and therefore we propose the following stage-terrace definitions:

The aquatic zone (AQZ) is the wetted channel, downslope form the lowest stage elevation.

The hydrologically active zone (HAZ) is inundated and scoured one to many times/year, and it extends upward from the baseflow water's edge up to the bankfull stage. This zone, when vegetated, usually is occupied by wetland and flood-tolerant plant species.

The lower riparian zone (LRZ) is the first overbank terrace. It is usually flooded or scoured every 1-2 years and in the arid regions often is occupied by a combination of drought-tolerant deciduous species (e.g., clonal *Phragmites australis*) and shallow-rooted phreatophytes (e.g., *Acer negundo, Alnus, Platanus,* some *Fraxinus* and *Quercus* species, and various *Salicaceae* in the American Southwest) as well as deeply rooted, nonnative *Tamarix* and *Eleagnus*. In mesic regions of eastern North America, the LRZ often is occupied by riparian understory herbs and graminoids, as well as *Acer, Platanus, Quercus, Populus, Salix,* and other bottomland hardwoods, *Larix*, and other species. Patagonian LRZs often support *Salix* (Stevens, unpublished data). Australian LRZs also are occupied by a wide array of herb and woody species, depending on the region (Lukacs et al. 2008).

The middle riparian zone (MRZ), sometimes referred to as the second overbank terrace, is usually flooded or scoured every 2-10 years and in arid regions often is occupied both by moderately deeply rooted phreatophytes, and opportunistic and weedy upland species. In mesic regions, it often is dominated by upland species.

The upper riparian zone (URZ) is only flooded or scoured by large, rare (> 10 years) events, and in arid regions it is generally occupied by deeply rooted phreatophytes and both facultative and non-facultative upland-riparian plant species. In mesic regions, it usually is dominated by upland species.

The upland zone (UPL) occupies non-flooded/non-scoured stages above the river channel, and it is dominated by non-phreatophytic plant species.

FRE Soils

Organic riparian soils develop in relation to flood and saturation frequency, as well as organic litter deposition and decomposition. Those processes vary by humidity province, stage elevation (scour disturbance intensity and frequency), and microhabitat type. Because pedogenesis requires considerable time, riparian soil development is negatively related to flood and ice scour return frequency and may increase non-linearly with distance and elevation above the active floodplain. Soils differentially influence plant functional group distribution (e.g., Andrew et al. 2014). With sufficient time, geologic and climate conditions, geomorphology, and vegetation, riparian soils can develop into deep, fertile loams and serve as a sink for carbon fixation.

Riparian substrata, soils, and microhabitats affect germination niche availability and quality (Grubb 1977). Day et al. (1988) reported that soil fertility (as grain size and nutrient status) and flood scour decreased germination and growth of fluvial wetland plant species. Bagstad et al. (2006) examined soil-vegetation relationships along the San Pedro River in southeastern Arizona, reporting that electrical conductivity, silt content, organic matter content, and biologically available phosphate all increased with patch age, as determined by the age of associated plants, although silt content more likely was a function of the geomorphic setting. Stevens (1989) experimentally demonstrated that germination and ecesis probabilities of common southwestern United States riparian plant species increased in relation to the concentration of silt in sandy shoreline sediments, due to reduced desiccation, susceptibility to flood scour, and increased nutrient availability. Jacobs et al. (2007) and Stella et al. (2011) reported that complex interactions among upland and river environmental variables, soils, microsite geomorphology, and large animal behavior all affected N and P processing in mesic and semiarid African riparian zones.

Soil salinity can be a strong organizing factor in riparian and lacustrine plant assemblages, influencing plant assemblage composition and structure. Brotherson (1987) identified five zones along the shore of saline Utah Lake in central Utah. The lowest elevation zone was occupied by halophilic saltgrass (*Distichlis spicata*) and dysclimax weeds, followed upslope (in order of increasing elevation) by saltgrass-alkaligrass, saltgrass-forb, saltgrass, and spikerush assemblages. Soluble salt concentration and soil pH decreased downslope, while moisture and organic matter increased. Forb distribution was correlated with micronutrient concentration.

Fluvial Microclimate

Fluvial climate is influenced by global- to local-scale phenomena, the latter including nocturnal cool air subsidence and upriver mountain valley wind patterns (e.g., Draught and Rubin 2006; Whiteman 1990), as well as local microclimate interactions. However, we know of few meso-scale studies of river influences on basin, reach, or local microclimates. Stevens (2012) and Yard et al. (2005), respectively, reported that riparian and in-stream interception of photosynthetically active radiation (PAR) varied temporally and by reach in the deep, narrow Colorado River in Grand Canyon, influencing in-canyon air temperature, relative humidity, aquatic and riparian production. Although not yet studied, variation in PAR flux also may influence slope failure frequency, cliff retreat, and canyon landform evolution.

At the local microclimate scale, Stanitski-Martin (1999) demonstrated that flow and channel aspect influenced cross-sectional fluvial solar energy flux, creating seasonally discrete patterns of diel temperature and relative humidity on the Colorado River in Arizona. In particular, she noted a discrete belt of humid air that developed over the river surface. This belt of humid air is evident because it conveys sound more effectively than less humid air farther above the surface. The humid air belt increased in thickness and extended into the riparian zone, particularly during nocturnal hours, and it often shrank during daytime. Occasionally visible as fog, it may positively influence riparian biotic productivity, particularly in arid regions. Such river-based microclimate patterns also affect riparian trophic dynamics. For example, increased riparian humidity was positively associated with avian species diversity in the Murray-Darling River basin in Australia (Selwood et al. 2016). Microclimate gradients and soil moisture availability contribute to the “riparian effect” in which the lowest elevation occurrences of aridlands upland shrub and tree species occur in ephemeral riparian channels.

Microsite variation in PAR flux and microclimate also may be a common phenomenon at tributary confluences, particularly in temperate constrained or canyon-bound rivers, affecting FRE aquatic and terrestrial productivity. The confluences of east (E) or west (W)-flowing tributaries in temperate rivers that flow north (N) or south (S) may not receive any direct solar radiation during the winter months, particularly in deep canyons at latitudes greater than 23° (Stevens 2012). In contrast, in the northern hemisphere, the confluences of N-flowing tributaries on mainstream rivers that flow E or W are likely to receive a shaft of light throughout the year, potentially greatly increasing PAR, increasing ambient temperature, and reducing the duration of winter freezing, all of which add to productivity and habitat diversity at ecologically interactive tributary mouths. The opposite pattern (a south-flowing tributary confluent with an east- or west-flowing canyon-bound mainstream) should obtain in the southern hemisphere. Such solar radiation patterns clarify why tributary confluences can serve as biological hotspots (Thorpe and DeLong 1994).

Dynamic Geomorphological Equilibrium

Rare coarse-scale events, such as lava flows, peak flows, tectonic events, ice dam failures, fire, and other types of physical catastrophic events exert formative and long-lasting influences on channel establishment and development. Antecedent events, including the failure of natural dams or large paleofloods, are long-term drivers of fluvial geomorphology (Foster et al. 1998; Parsons et al. 2005). FRE channels undergo geomorphic transitions after destabilizing flows, moving toward spatially uniform energy dissipation, as described in Morisawa's (1968) dynamic equilibration model (DEM). The DEM is a cornerstone of our WCM, in which stream channel geomorphology and ecosystem development trend toward equilibrium but never reach that condition because destabilizing flooding and other disturbances are frequent, vary in intensity, and are erratic.

Variation in high flow return frequency (FRF) in rivers with erodible banks affects FRE channel geometry and geomorphology by creating or maintaining shoreline habitats, point bars, gravel and cobble islands, and terraces, all of which can support riparian assemblages. River landforms are maintained or readjusted through hydrographic time and change at rates ranging from instantaneous to the geological lifespan of the watershed (107 years or more), depending on FRE and the stage elevation of the landform. However, and in part due to the power function of suspended sediment load to flow rate (e.g., Gray and Simões 2008), discrete FRF intervals (particularly the bankfull 1.5-2.5 year interval) generate new channel bedforms and scour or reset riparian vegetation on hydrologically active and lower riparian terraces (Campbell and Green 1968). Channel geometry and landforms respond over long periods, with rare, large, antecedent floods exerting long-lasting impacts on channel landform distribution.

Physical Elements and Processes: Conclusions

Much widely divergent, interdisciplinary expertise is required to understand rivers as ecosystems, and the incomplete integration of physical science elements with ecological disciplines remains an impediment to FRE ecology and stewardship. The next steps in refining and testing HGM concepts are likely to involve intensified study. This includes improved modeling of groundwater-surface water interactions and interactions among reaches (including lentic and frozen landscape transitional reaches) among watersheds in relation to tectonic history and under changing climates. In addition, more refined communications among FRE physical and ecological researchers may help clarify future science directions. This dialogue requires improved long-term monitoring data on hydrographic, hydrologic, and other physical and biological variables. Such emphases will enhance understanding of channel landform development, as well as the interactive spatial and temporal scales of hydrogeomorphological and ecological influences on FRE development.

Bio-Ecological FRE Concept Modeling

Historical Synthesis

Building on the abiotic processes discussion above, we review and discuss biological elements and processes of FRE ecology and evolution. In addition, we use a description of the details of figures 11 and 12. We examine continuity and other models of FRE processes and characteristics within the watershed across stream order and hierarchical frameworks of stream habitat classification (e.g., Frissell et al. 1986). Many river ecosystem models have been developed over the past several decades, including recent, highly integrative approaches. While many of these models have substantially advanced understanding of FRE structure and function, none are mutually exclusive, and many broadly but incompletely overlap.

Hutchinson (1967) and Hynes (1970)

While G.E. Hutchinson (e.g., 1967) and his students and colleagues set the stage for much of modern ecology and limnology, the first comprehensive synthesis of flowing water ecology was presented by Hynes (1970). Hynes' *Ecology of Running Waters* was a pioneering compendium focused largely on biological elements and ecological processes associated with streams and smaller rivers. Hynes (1970) described and related many discrete biological studies of flowing water, and his subsequent work included consideration of the watershed spatial scale and groundwater influences, advancing both basic and applied research. Hynes' research included the role of the watershed, lakes, and ecological risk assessment in FRE ecology and management (e.g., Hynes 1974, 1975; Stanford 1998; Ward 1989). However, Hynes focused less attention on non-flowing watershed components, like the importance of the riparian zone or a unified FRE model. Wurtsbaugh et al. (2015) reported that focus on Hynsian (lotic) versus Hutchinsonian (lentic) freshwater limnology created differences in interpretation about the roles of habitat and biotic factors in fish population biology and freshwater aquatic research. Nonetheless, both Hutchinson's and Hynes' contributions initiated a rush of integrative thinking about FRE ecology that followed their 1970s work and continues today.

The River Continuum Concept (RCC)

Beginning in the 1970s, the RCC was developed in the 1970s by Cummins (1976) and Meehan et al. (1977), and was formalized by Vannote et al. (1980), integrating aquatic and riparian domains of FREs. The RCC "... described the entire fluvial system as a continuously integrated series of physical gradients and associated biotic adjustments as the river flows from headwater to mouth" (Annear et al. 2004: 9-10). The RCC described "... a series of responses within the constituent populations resulting in a continuum of biotic adjustments and consistent pattern of loading, transport, utilization and storage of

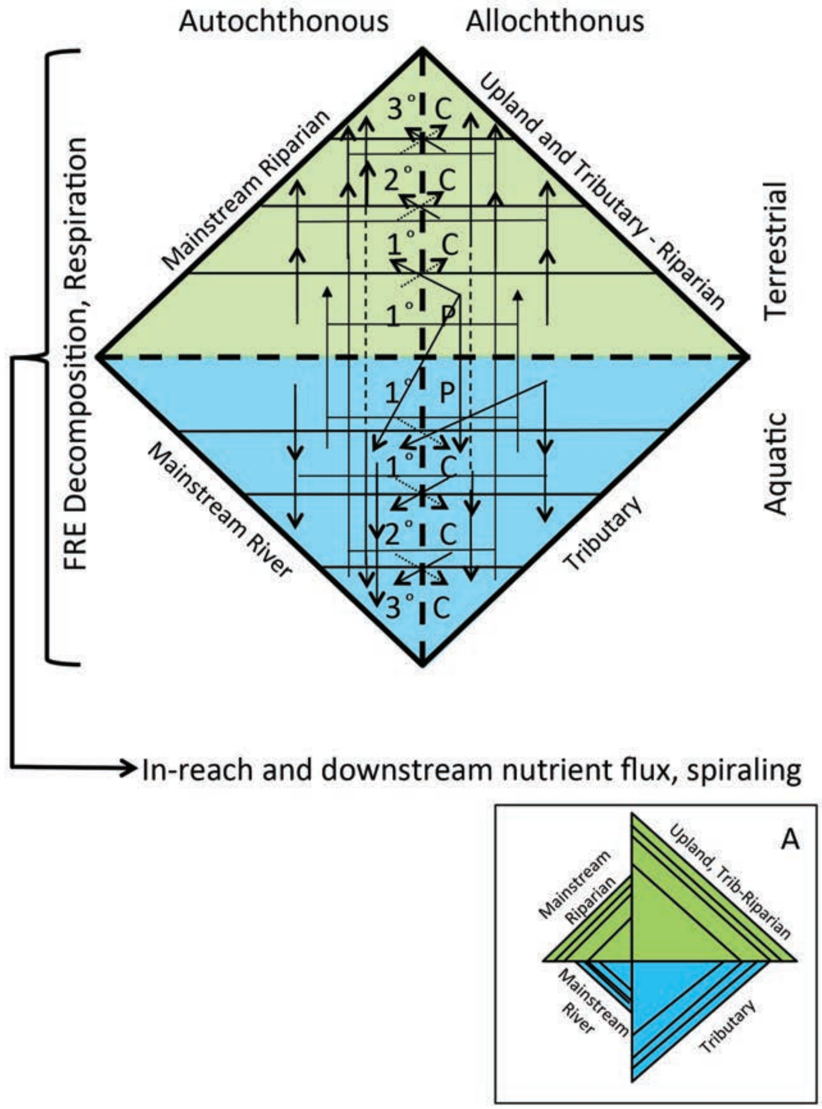

Figure 12—Expanded detail of linkage of the FRE, contrasting allochthonous (uplands and tributary) vs. autochthonous (mainstream) ecosystem energy inputs with aquatic vs. riparian domain interactions. Arrows indicate common energy pathways among trophic levels in the four FRE arenas. Not all interactions occur in every FRE, and other trophic interactions that are not depicted here may exist in some FREs. Inset A depicts spatial change in reach-based FRE structure and function in response to watershed changes. For example, upland fire can result in sediment, ash, and nutrient loading through tributaries, processes that may diminish FRE productivity and ecological role in the watershed. Similarly, reduction in precipitation or groundwater contribution through climate change or aquifer depletion may reduce mainstream and riparian function.

organic matter along the length of a river" (Vannote et al. 1980:130). The focus of the RCC was on lotic (riverine) processes supporting the aquatic ecology of invertebrates and fish and did not relate streams to groundwater and lentic sources or to potential hyporheic refugia (Palmer et al. 1992). Quickly recognized as a useful overall FRE model (Merriam 1984), the RCC was broadly supported by studies of low-medium order streams, primarily in mesic regions (e.g., Minshall 1988, Minshall et al. 1983, 1985). Applicability of the RCC to higher stream orders has been questioned (e.g., Sedell et al. 1989).

The RCC described perennial streams (AQZ) bordered by hydroriparian (HRZ) to URZ ecosystems from headwaters to mouth (Johnson et al. 1984) and generally supporting large, important fish populations, particularly salmonids. Although concentrating on aquatic ecosystems, it emphasized interactions between aquatic and riparian domains. The RCC considers the riparian zone as a dependent contributor of organic matter, shade, and fish food, sourced from upstream reaches and adjacent uplands. Riparian zones provide important ecotonal interfaces between aquatic and terrestrial domains (reviewed in Committee on Riparian Zone Functioning and Strategies for Management 2002). In addition to autochthonous primary productivity, FRE aquatic domains receive essential contributions of nutrients and bio-available leaf litter, wood, detritus, and dissolved organic carbon, as well as recalcitrant carbon from the adjacent riparian zone. The bio-available portions of those contributions are processed by stream microbes and macroinvertebrates, providing food resources for fish. In turn, the adjacent riparian ecosystem depends on subsurface water from the hyporheic zone, other groundwater, and surface flow events including overbank flooding. As originally described, the RCC did not explicitly extend into the hyporheic zone, but a plethora of FRE macroinvertebrates, such as Plecoptera (stoneflies) and amphipods, can occur there (Palmer et al. 1992), some being found in 10 m-deep wells as far as 2 km from the channel on the floodplain of the Flathead River, Montana (Stanford 1998; Stanford and Ward 1988; Annear et al. 2004).

The RCC quickly became the leading paradigm explaining FRE ecology, and it stimulated a vast wave of research, assessment, metrics development. It also encouraged policy relating water quality, aquatic macroinvertebrate feeding guilds, and fish populations as management indicators of stream health (Annear et al. 2004, 2009). This emphasis has included Karr's (1991, 1999) development of the index of biotic integrity (IBI), which assembles multiple layers of information to provide a quantifiable metric of stream ecosystem health to guide management and stream rehabilitation. Reviews of the success of IBI and other metrics by Merritt et al. (2008), and of the success in understanding gradient responses of aquatic macroinvertebrates by Heino et al. (2015), are somewhat reserved because of the growing recognition that these metrics and relationships often only weakly describe patterns of macroinvertebrate biodiversity and distribution. Interactions occur among physical and biotic variables, not to mention anthropogenic factors, operating at microsite to watershed scales. These interactions influence the ecological integrity of FRE aquatic habitats in sometimes incomprehensible ways.

The RCC champions the concept of ecological connectivity, which was recognized in the 1980s as "… maintenance of longitudinal, lateral, and vertical pathways for biological, hydrological, and physical processes" (Annear at al. 2004:215). However, spatial focus on watershed and river flow and geomorphology in the RCC has somewhat neglected the temporal dimension, which is integral to FRE development, function, and dynamic seasonal and interannual geomorphic adjustment (Morisawa 1968). Although connectivity is recognized as important to perennial streams and adjacent riparian zones, it also is applicable to ephemeral and intermittent streams and their adjacent riparian zones, as well as river source lakes, lentic zones, and groundwater sources, including springs (National Research Council 2002).

Subsequent to Hynes' (1970) work and formulation of the RCC, many FRE syntheses have been undertaken, including comprehensive edited volumes and reviews by: Annear et al. (2004); Bouwman et al. (2013); Fisher et al. (1998); Humphries et al. (2014); Johnson and Jones (1977); Johnson et al. (1985); Karr (1991, 1999); Karr and Chu (1999); Malanson (1993); Minshall et al. (1983, 1985, 1988); Naiman et al. (1998, 2005); National Research Council (2002); Sedell et al. 1989; Stanford (1998); and Thorp et al. (2006, 2008). While a full review of these syntheses is beyond the scope of this document, we note that each review has emphasized particular aspects of FRE ecology, somewhat to largely overlapping previous reviews, but sometimes shifting focus away from conceptual integration.

Contributing Ecological Models and Syntheses

During and since the initial formulation of the RCC, a number of important additional models have been proposed. These contributions appear to us and others (e.g., Sedell et al.1989; Ward 1989) to refine FRE ecology as critically definitive axioms, corollary processes, and addenda to the RCC, although several offer alternative perspectives on FRE ecology.

The nutrient spiraling concept was formulated by Newbold et al. (1981; reviewed in Ensign and Doyle 2006) to describe the helical ecological pathways of material and nutrient processing through FREs. Autochthonous (endogenic) and allochthonous (exogenic) production is processed, released, and taken back up as water flows through uplands and tributaries into riparian zones and mainstream rivers. Nutrient spirals lengthen with stream order and with higher flows, thus, spiraling is a process that directly links upstream to downstream reaches, in addition to lateral shoreline habitats.

Although developed to describe the impacts of impoundment on regulated rivers, Ward and Stanford's (1983, 1995) serial discontinuity concept (SDC) illuminated the issue in many natural rivers of the roles and impacts of natural dams that form lacustrine reaches and affect FRE channel geomorphology, flow, and population dynamics, both upstream and downstream from the dam. Lacustrine reaches can occur anywhere in a basin as a result of tectonism, lava dams, slope failure, or glacier development, and the natural dams may persist for short to long durations. Lake Victoria in Uganda formed as a result of tectonic rifting, interrupting the flow of the Kagera and other Nile River headwater streams. Lago de Nicaragua (L. Cocibolca), the largest lake in Central America, formed as a result of tectonic uplift in the lower Tipitapa and San Juan River basins. Prominent examples of North American lava flow dams include the formation and collapse of basalt dams; large to enormous paleo-impoundments in lower Grand Canyon (Crow et al. 2008; Dalyrmple and Hamblin 1998; Fenton et al. 2004); and in the Verde River basin in central Arizona (Elston et al. 1974). Costa and Schuster (1987) claimed that natural impoundments created by landslides, glacial ice, and glacial moraines constitute substantial threats to human life and property. They identified six types of slope failure dams around the world, ranging from relatively common single events that half-impounded a valley, to rare, simultaneous impoundment of multiple valleys, creating several to many natural lakes. The largest slope failure dams in the Colorado River basin in the American Southwest occurred in the Surprise Valley region in the middle of Grand Canyon (Rogers and Pyles 1980). Due to several triggering mechanisms among several events, nearly 15 km of the north rim failed (at least once catastrophically), slid into the canyon, and repeatedly dammed the river, causing it to shift its deeply incised course. Other large dam-forming landslides are known throughout Grand Canyon and elsewhere in the region, including a 0.3 km^3 dam-forming landslide in the Virgin River Canyon downstream from Zion National Park (Castelton et al. 2016; Hereford et al. 1995). Enormous Pleistocene glacial dam outburst floods from Lake Missoula swept through the lower Columbia River drainage, creating scablands geomorphology (Benito and O'Connor 2003; Bretz 1923).Ice damming floods are well known on fjord rivers as well (e.g., Reeburgh and Neburt 1987).

Like anthropogenic dams, natural impoundments change river seasonal water quality and flow, hydrography, stage relations, velocity, habitat quality and distribution, and FRE biogeography. The SDC posits that the location and size of a dam resets and influences downstream recovery of the FRE "normal" conditions through tributary contributions of flow, water quality, and biota (Dye 2010; Rice et al. 2008). Furthermore, the extent of river recovery from natural impoundment varies in relation to the number, size, and flow characteristics of downstream tributaries. Like large, influential tributaries, natural dams are not controlled by stream order (they can occur anywhere within a basin), and a large and persistent natural dam may exert long-term direct and antecedent impacts on FRE geomorphology and ecology. Like anthropogenic impoundments, natural dams also seasonally influence upstream FRE habitat quality and connectivity.

Also related to Ward and Stanford's (1983) SDC, the discontinuity impacts of larger tributaries often are abrupt and generate multi-reach alterations of mainstream FRE ecology, as incorporated in the link discontinuity concept (LDC; Rice et al. 2001). The LDC posits that rivers are networks of tributary confluence nodes linked by the mainstream. This perspective was advanced in the network dynamics hypothesis (Benda et al. 2004b), which integrates drainage networks of channel and confluences, and proposes that the complexity of the overall basin shapes tributary contributions to the mainstream, resulting in minor to overwhelming influences on mainstream characteristics (Benda et al. 2004a,b; Clay et al. 2015).

The flood pulse concept (FPC; Junk et al. 1989; Tockner et al. 2000) added the importance of high flow pulses to the FRE function. Floods regularly restructure river landscapes, remove encroaching channel margin vegetation, transport and deposit large woody debris, trigger river biota life history events, and open new habitat for germination and establishment of fluvial and riparian biota. Floods vary in frequency, duration, timing, and magnitude, sometimes on catastrophic spatial scales (e.g., regular seasonal flooding), accounting for the state of suspended succession recognized as a common characteristic of natural riparian vegetation by Campbell and Green (1968).

Ward (1989) clarified four dimensions of spatial and temporal scale operating in most lotic ecosystems, including: (1) the "longitudinal" dimension up- and downstream through rivers; (2) across-channel, riparian-aquatic domain interactions; (3) vertical interactions with hyporheic habitats and groundwater; and (4) a broad temporal dimension (fig. 11). Dynamic interactions among all of these dimensions contribute to the individuality of character of FREs.

Ecosystem research and stewardship require detailed and long-term understanding of geologic, hydrographic, biota, and land use history within a basin. White (1979) emphasized this in relation to forest management, and Décamps et al. (1988) and Petts et al. (1989) described the relevance of such understanding in western European river basins.

Following Stanford (1998) and Ward et al. (1989, 1998, 2002), Malard et al. (2002) expanded discussion of the biodiversity and roles of FRE hyporheic exchange through microbial and macroinvertebrate distribution and activity (figs. 10, 11).

Focusing specifically on river riparian zones but related to Ward's (1989) considerations, Nilsson and Svedmark (2002) recognized that four major processes or characteristics interactively function in FREs: (1) The flow regime (hydrograph) regulates FRE ecological and geomorphological processes, including riparian succession. (2) The channel provides a corridor for inorganic and organic transport, primarily downstream but also upstream, and including dispersal of propagules. (3) The riparian zone functions as a filter and boundary between upland and riverine processes. Naiman et al. (1993) and

Nilsson and Svedmark (2002) also recognized (4) that riparian zones are particularly rich in biodiversity and species interactions, and play important roles in watershed biodiversity.

From advances in nearshore marine ecology patch dynamics concept (e.g., Pringle et al. 1988; Townsend 1989), Thorp and DeLong (1994) proposed and refined the river productivity model (RPM), which posits that production, as well as decomposition, recruitment, and other important river processes are distributed nonrandomly within the channel. These processes occur at specific points or in specific zones in the channel, such as at tributary confluences, along shorelines, or in certain depositional settings. Thus, their RPM proposes that river ecosystems are mosaics of microhabitats with differing ecological functions. They and their colleagues subsequently expanded this description to more fully describe the complexity of river food webs (Thorp et al. 2006, 2008), and much subsequent research has proceeded from this area of inquiry.

Fisher et al. (1998) proposed the telescoping ecosystem model, in which the FRE includes nested, concentrically positioned subsystems of the stream, hyporheic zone, channel margins, and riparian zone. Collectively, these function like a telescope, extending or retracting in relation to flood disturbance. Their model implies stepped transitions at tributary confluences and reach boundaries across stream order.

Following the lead of Jacobs et al. (2013) and Meybeck (1982), Bouwman et al. (2013) emphasized the importance of integrated biogeochemistry in FRE conceptualization. Rather than regarding a river simply as a single thread of flow or as a coupled mainstream and floodplain landscape, they considered all parts of the flow system (river mainstreams, floodplains, lakes, wetlands, etc.) as a biogeochemical retention and processing network.

The river wave concept (Humphries et al. 2014) posited that most phenomena within at least the FRE aquatic domain can be viewed in a wave context. For example, river flow can be characterized as a wave that varies in frequency, length, and shape, and travels lengthwise and laterally through the channel. Wave position determines or regulates production and transport of organic material, with allochthonous input occurring primarily in troughs and ascending limb of the wave, and with autochthonous production occurring on the crests and declining limbs of the wave.

Muehlbauer et al. (2014) defined the biological stream width as the distance of travel of resource subsidies from the FRE aquatic domain into the surrounding upland terrain. They used meta-analysis to model the spatial extent of this "stream signature," reporting that the 50 percent stream signature (the point at which subsidy resources are half the aquatic domain maximum) lay only 1.5 m from the stream edge, but that 10 percent of the signature can extend more than 0.5 km into the adjacent uplands. Thus, the biological stream width often is much larger than that defined hydrogeomorphologically.

While expansive in scope, the RCC is based on orderly, gradual downstream connectivity within a perennial alluvial channel network, and it does not well describe ephemeral and intermittent streams-riparian ecosystems or groundwater-surface water interactions (Stanford 1998). Ephemeral FREs are colloquially known as dry washes, arroyos, wadis, and other names throughout the world, and comprise more than half of the global stream channel network (Datry et al. 2014). Such stream ecosystems are becoming increasingly abundant as rivers are dewatered by human activities and subjected to a drying climate. A well-documented example was the conversion of the Santa Cruz River in Tucson, Arizona from a perennial to an ephemeral stream through groundwater pumping (Webb et al. 2014).

Flooding releases CO_2 sequestered by burial of organic matter and invertebrates, such as clams (Butman et al. in review; Smith et al. 2016), and that process is interrupted in ephemeral streams. Benthic invertebrates that shred, graze, or collect organic debris often are absent or rare in ephemeral streams, reducing decomposition rates, and their roles are often replaced by microbes and physical molar action when the streams flood.

Terrestrially, ephemeral versus intermittent riparian zones are bordered by distinctive suites of xeroriparian (dry riparian) to mesoriparian perennial plant species that provide cover and food resources (Johnson et al. 1984). Analysis of an ephemeral stream near Peshawar University in Pakistan revealed that deeply rooted woody perennial shrubs occurred in the wash, and weeds dominated the bed following winter rains, with drought resistant species occurring on terraces (Chaghtai and Khattak 1983). Aquatic productivity and trophic energetics of aridland ephemeral streams are reduced and interrupted during dry seasons (e.g., Jenkins and Boulton 2003), interrupting spatial and temporal “flow-through” RCC processes, but ephemeral channels commonly provide essential wildlife habitat connectivity. They perform as punctuated, rapidly functioning biogeochemical reactors (Larned et al. 2010). More study of ephemeral stream ecosystems continues to be warranted.

Biological Processes and Characteristics

Overview

Climatological, hydrogeological, and geomorphic physical processes generate the microscale-to-watershed template on which the microclimate and biological functions of FREs generate or develop nutrient dynamics, assemblage composition and structure, and trophic interactions (Merritt et al. 2010; Naiman et al. 1993, 1998, 2005). Here we describe and illustrate these processes and interrelationships through the figures presented below, with figure 11 as the template, figure 12 adding detail to figure 11, and subsequent figures to illuminate key ecological processes and interactions between linked aquatic and terrestrial domains. These biological processes and elements sometimes reciprocally influence physical factors, such as sedimentology, channel geometry, and microclimate (Merritt 2013; Montgomery et al. 2003; Pollen et al. 2004), as well as FRE characteristics, such as ecological resiliency. We discuss these biological processes and elements, placing additional emphasis on riparian vegetation formation and dynamics.

Nutrient and Organic Matter Production and Release

FREs export allochthonous (upland and upstream) and autochthonous fine to coarse woody debris, other organic matter, and nutrients downstream through ecological spirals (Newbold et al. 1981) (figs. 11, 12). Although the dominant direction of FRE matter transport is downslope and downstream, important returns of nutrients also can occur through upstream aerial or zoochorous transport of sediments, nutrients, and propagules (e.g., spawning salmon, wind-blown seeds, or adult forms of aquatic insects; review by Cederholm et al. 1999). N, P, and C are differentially stored in alluvial reaches but are generally exported from constrained reaches. Fluvial nutrient dynamics models are needed to relate nutrient fixation, storage, retention, transport, and recycling, but such models remain a largely overlooked dimension in FRE ecology (Bouwman et al. 2013).

Biodiversity

Elevated biodiversity in FREs is universally recognized. Nearly 7.5 percent of the nearly 1.5 million described species on Earth are aquatic and occur in freshwater habitats, and many more aquatic and riparian invertebrate species remain undescribed (Collon et al. 2014). In addition, many other species occur in adjacent wetland and riparian habitats. The high biodiversity and proportion of gamma diversity in FREs can be attributed to a suite of factors (Naiman et al. 1993). Gravity transports propagules downslope from the uplands and also downstream into different reaches and segments (Johansson et al. 1996; Stevens 2012). The dendritic network of a channel system provides a corridor for movement or range shifts for many species. As ecotones, FREs bring different suites of species together (Décamps and Tabacchi 1994; Gregory et al. 1991; Naiman and Décamps 1997). Also, rivers provide shifting mosaics of highly productive habitat, allowing different colonization opportunities over space and time (Naiman et al. 2005). Stevens and Ayers (2002) reported that nearly half of 1,400 vascular plant species in Grand Canyon occurred within 200 m of the Colorado River elevation, of which one-third were riparian and 10 percent were springs-dependent species. In addition, more than two-thirds of the regional fauna obligatorily or facultatively used riparian habitat. Thus, although the area of FRE habitat is trivial, it supports a vast proportion of regional biodiversity, particularly among aridland river basins (Jansson et al. 2007; Stevens 2012).

Nilsson et al. (1989) studied riparian plant species richness (*S*) along rivers in northwestern European rivers, reporting peak *S* at middle stream order. Substrate heterogeneity and fineness were the primary factors influencing total richness, with a unimodal peak in *S* at intermediate levels of substrate fineness. Renöfält et al. (2005) compared riparian-to-upland plant *S* in the Vindel River basin in northern Sweden, reporting that S was related to local, river-related processes and corridor-based dispersal. However, unlike Nilsson et al. (1989), they reported a monotonic decrease in *S* from the headwaters to the coast, and high floristic similarity between the uplands and the riparian zone. Differences between these studies highlight the roles of stream order and regional variation in upland plant diversity.

In an aridland comparison of riparian and upland vegetation, Bloss and Brotherson (1979) described plant assemblage composition in a Sonoran Desert valley near New River, Arizona. They reported that vegetation either intergraded or abruptly shifted from upland slopes into the river channel, based on the erosional structure of terraces. Leguminous shrubs and trees occurred differentially on middle portions of the moisture gradient. Although plant diversity was highest on upland slopes, diversity was positively related to soil moisture, and floodplain species exhibited the broadest niche widths. They concluded that disturbance as well as moisture availability influenced the transition from desert to riparian plant diversity.

Differences among the above and other studies highlight the need for comparative analyses of FRE plant diversity, composition, and structure across latitude and among humidity provinces.

Trophic Energetics and Structure

Sunlight is the primary source of terrestrial FRE energy, not only generating the hydrologic cycle, but also powering photosynthesis (e.g., Stevens 2012; Yard et al. 2005).

The proportional contribution of autochthonous versus allochthonous production and nutrients varies between aquatic and riparian domains in relation to the physical setting of the watershed, stream order, season, and reach-based channel geometry and connectivity (Fisher and Likens 1973), changing over time in a successional fashion if not disturbed by flooding (Fisher 1983; Fisher et al. 1982) (figs. 11, 12). High levels of intrinsic production may occur in clear headwater springs and low- to mid-order streams. Lower elevations in the basin may be warmer (or sometimes cooler if subsidence occurs) and often have longer growing seasons, but aquatic productivity may be reduced in higher order streams where organic and inorganic particles reduce PAR (e.g., Yard et al. 2005). Stevens (2012) also emphasized the PAR-limiting influences of cliff shading in large temperate river canyons.

FRE trophic structure varies among aquatic, riparian, and upland domains, and across the chemical and ecohydrological gradients occurring within them, as described above and illustrated in figures 11 and 12. Aquatic food-web structure is regulated by temperature (Glazier 2012), flow, sediment load, seasonality, shading, and many other factors. For example, Mustonen et al. (2016) used 12 experimental channels to examine the effects of flow and suspended sediment on primary producer, macroinvertebrate, and/or fungal production and decomposition. Flow and sediment impacts on the response variables were largely independent, but interaction effects were antagonistic (e.g., flow stimulated algal production, while sediment loading reduced production). While their results are largely intuitive, such studies help clarify and quantify the fundamental driving features of FRE ecology.

Gawne et al. (2007) tested predictions of the RCC, RPM, and FPC models on river metabolism through analysis of the ecological roles of microbial and macrophyte assemblages in three lowland tributaries of the Murray River in Australia. They concluded that all three models were supported to some extent, but the extent, causes, and consequences of aquatic primary production were varied, and no individual model fully explained the patterns observed. In contrast to the aquatic domain, riparian food-web dynamics are more diverse due to the more open nature of riparian-to-upland interactions and feedbacks. In a seminal paper, Carothers et al. (1974) reported that the highest diversity of breeding (primarily neotropical) birds in central Arizona occurred in cottonwood-willow riparian habitats bordered by agricultural field in central Arizona, with riparian vegetation structure affecting not only bird assemblage composition but also social organization. Although not yet studied to our knowledge, the noise levels generated by rivers in steep canyons also is likely to influence FRE avifaunal assemblages (e.g., McClure et al. 2013).

Trophic cascades are regularly observed in fish-dominated ecosystems and in some low-order fishless systems (e.g., Blinn 2008) but are limited in FREs by physical processes (e.g., hydrology, sediment transport, ice impacts), where average sheer stress/unit area appears to be negatively related to stream order (Magilligan, 1992). However, turbidity generally increases with stream order, reducing downstream PAR availability and primary through tertiary aquatic production (e.g., Yard et al. 2005). Complex trophic relationships also develop in riparian zones, directly and indirectly influencing primary producer structure and composition. For example, leaf beetles, grasshoppers, beaver, and ungulates all can strongly influence riparian vegetation composition, structure, and decomposition/soil formation (e.g., Bailey and Whitham 2006; Sacchi and Price 1988), and such impacts can be modified by both secondary consumers (e.g., ants; Schweitzer et al. 2005) or top predators (e.g., wolves; Beschta and Ripple 2006, but see Marshall et al. 2012). In one example of complex riparian interactions, manna (honeydew) produced by the host-specific cicadellid tamarisk leafhoppers (*Opsius stactogalus*) stimulated soil fungal growth, which in turn killed germinating seedlings of both the host plant and other riparian plant species beneath *Tamarix* canopies (Simieon and Stevens 2015). Whether such multi-trophic-level interactions constitute an evolved process remains to be determined, but proximally it ensures *Tamarix* stand persistence through inhibitory (rather than facilitation or tolerance) successional mechanisms (*sensu* Connell and Slayter 1977). Just one of many complex riparian trophic interactions, this interaction is favored in relatively constant environmental conditions, and is most influential on the broad, low-gradient floodplains of higher order streams.

Trophic cascades also arise from interactions among other aquatic and terrestrial FRE taxa. For example, predatory aquatic invertebrates, such as hellgrammites (Megaloptera: Corydalidae), can function as top predators in clearwater streams, foraging actively at night when fish may be less able to detect prey. Amphibians can alter algal composition, algal production, and organic matter dynamics in small streams, and they can function as important aquatic and riparian predators (e.g., the giant aquatic salamander *Cryptobranchus alleganiensis*), thereby influencing aquatic-to-terrestrial ecosystem energy transfer (Whiles et al. 2006). The dual nature of amphibian life cycles may mean that the loss of an FRE amphibian species equates to the loss of two functional species, effects that may be greatest in neotropical stream ecosystems.

Finer-scale illustration of FRE trophic structure (fig. 12) depicts the complex food-web interactions among aquatic and terrestrial domains within reaches, which produce and receive ecosystem energy from autochthonous as well as from upslope, upstream (including tributary), and groundwater-derived allochthonous sources (e.g., Townsend et al. 2000). Four relatively discrete component triangles are illustrated in figure 12: autochthonous mainstream and riparian zone, and allochthonous tributary and upland plus tributary riparian triangles. These four triangles are depicted as being ecologically open and potentially interacting with each other (dashed lines). The most common inter-trophic interactions among the four components occur vertically (primary producers through tertiary consumers), but many other complex interactions exist among trophic levels among these four triangles. For example, across-triangle interactions include such food-web interactions as crocodilians feeding on riparian or upland ungulates and predators in tropical rivers.

Environmental variation within reaches and across spatial and temporal scales influences the relative contributions of the four FRE component triangles (fig. 12, inset A). For example, under a drying climate, decreased runoff may reduce or eliminate mainstream and riparian nutrient contributions. Conversely, increased wildfire frequency and severity may at least temporarily mask or inflate the influence of upland and tributary nutrient contributions to the watershed. Thus, relationships among component triangles are expected to vary over time and therefore are not expected to reach equilibrium.

FRE Biogeography

FRE biogeography involves colonization, recruitment, and population establishment overland by volant and other highly vagile species, as well as passive dispersal through gravity, aerial drift, or zoochorous transport of propagules through both overland and dendritic stream corridors (fig. 11). Regardless of the pathway, FRE population persistence and assemblage resilience is predicated on the ability of a species to remain in or disperse-recover their position in the watershed. Therefore, persistence of all FRE species requires some form of upstream dispersal, with eviction or extirpation the inevitable consequence of failed *in situ* or headwater recruitment strategies.

FRE dispersal by riparian plants is achieved through hydrochory, anemochory, or zoochory. FRE propagules of seeds, rootstocks, eggs, or larvae drift downstream through hydrochory (Merritt and Wohl 2002). But propagules must be blown upstream or be transported by animals if that species is to persist in the FRE. Larval aquatic macroinvertebrates may drift downstream, while adult aquatic insects often fly or are blown upstream as aerial drift. Dragonflies, salmonids, and many other fish taxa migrate upstream to spawn, against the dominant flow direction, and some fish transport larval unionid mussel larvae upstream. Migratory western North American warblers and other passerine birds intensively use aridland riparian habitat as stop-over habitat during migration (Carlisle et al. 2009; Skagen et al. 2005; Stevens et al. 1977), a pattern not strongly evident in mesic eastern North America (Kelly and Hutto 2005).

However, western North American songbirds generally migrate northward along broad fronts, rather than using FREs as navigation corridors, although "specific populations are likely restricted to narrower migration routes," such as riparian nesting species along FRE corridors (Carlisle et al. 200; RRJ, unpublished research). Front-based bird migrations also occur among some western North American shorebirds, but many waterbird species appear to follow FRE corridors, particularly through complex landscapes (e.g., Stevens et al. 1997a). In addition, many non-volant vertebrate species follow river corridors as dendritic pathways, although terrestrial faunal movements can be thwarted by steep cliffs, perilous crossings, and anthropogenic landscape interruptions (Stevens 2012).

Colonization and extirpation frequencies vary by taxon and life history strategy and on the basis of physiological, life history, mobility, and reproductive strategies, which collectively influence species dominance in a given FRE reach. In relation to insular biogeographic theory (MacArthur and Wilson 1967), colonization probability in FREs is enhanced by productivity through habitat "hospitability" to colonizing taxa, while local extinction (extirpation) probability is more related to disturbance, particularly from scour (flooding, ice scour), low flows, and random or erratic events.

Disturbance

Ecological disturbance is defined as events that kill mature individuals within habitats. It exerts direct and indirect controls over sessile species richness, composition, and structure, as described in the dynamic equilibrium concept (DEC) (Huston 1979, 1994; Laliberté et al. 2013; Lehman and Tilman 2000) (figs. 11 and 13). Floods, channel inundation or desiccation, glacial movements, ice scour, slope failure, and other disturbances affect sessile riparian species richness by removal or killing of adult organisms and resetting of environmental conditions (Connell 1978; Sousa 1984).

Scour impacts arise from unit stream power, applied as sheer across the channel surface (Bendix 1992). Scour impacts vary in relation to latitude, lithology, and stream order in alluvial versus constrained reaches, and across stage, differentially affecting

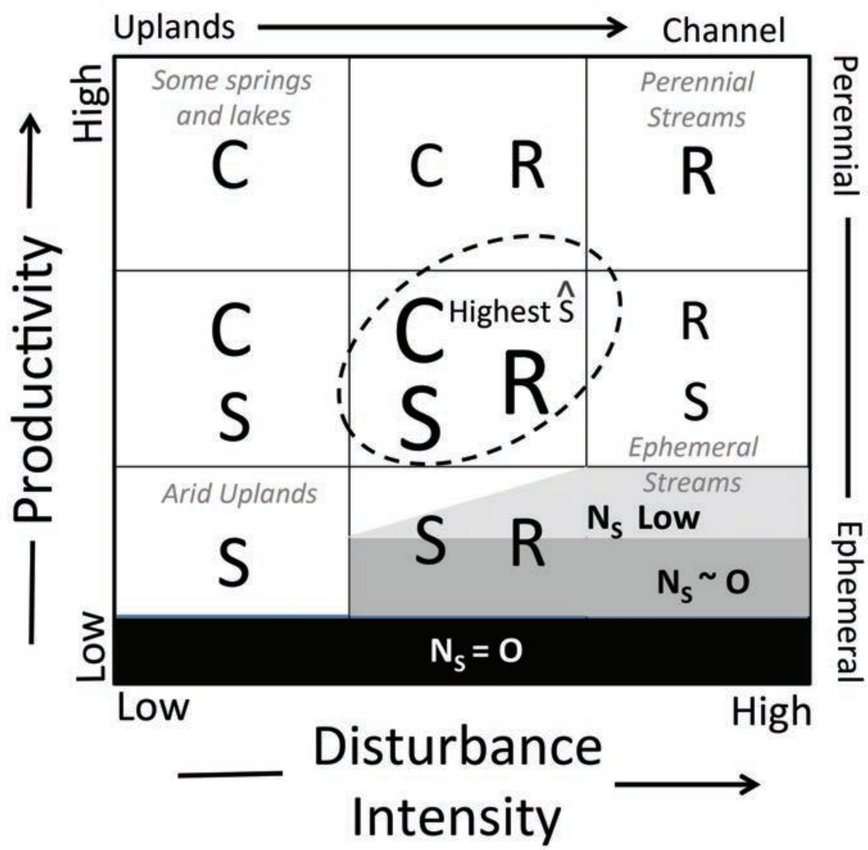

Figure 13—Plant life history groups (Grime 1977) in relation to disturbance intensity and productivity gradients (Huston 1979, 1994). C—competitive species, R—ruderal species, S—stress tolerators. Black areas support no plant species (NS = 0), gray areas have very low NS, and white cells have some to high NS. Highest "hat above S" [middle square]=maximum plant species richness.

lower terraces (Magilligan 1992). Due to shear stress and channel margin erosion, scour mortality of vegetation may differentially predominate lower terraces, while drowning mortality may be more likely in the latter. Stevens and Waring (1985) reported great variation in flood-related mortality among guilds of riparian plant species and across stage elevation in the Colorado River in Grand Canyon, Arizona. However, vegetation assemblage responses to flooding differ among terraces, reaches, and rivers. Pettit et al. (2001) reported reduced germination density, plant species richness, and riparian tree size in the intensively flood-scoured Blackwood River in southwestern Australia; however, they reported neutral germination responses and little reduction in species richness in response to flooding along the Ord River in northwestern Australia.

The impacts of ice formation and breakup on FREs are pronounced and are gaining attention in temperate and boreal rivers (Prowse and Culp 2003). At higher latitudes and elevations, ice formation and "shoving" routinely scour shorelines and bed surfaces and may dam channels, uplifting and redepositing fine to coarse substrata, including boulders and coarse woody debris. They may also alter channel geometry. In addition, processes such as melting black (benthic river) ice or surface ice breakup and scour can alter benthic and riparian composition complexity and structure (Scrimgeour et al. 1994).

Productivity

Riparian productivity is positively associated with biodiversity through intrinsic mechanisms, such as organism size distribution, niche specialization, assemblage history, as well as interaction with disturbance gradients (Fukami and Morin 2003; Hooper et al.

2005; Huston 1979, 1994; Marquard et al. 2009; Tilman et al. 2001) (figs. 11 and 13). Reach-based FRE productivity varies by stage (moisture and soil), aspect, and microclimate, as well as macro-scale climate, latitude, and elevation gradients. In lieu of a way to describe it, we refer to the colonization potential of a microhabitat as "ecological hospitability," referring to the productivity and receptivity of a site to colonization. Low levels of productivity and high levels of disturbance reduce the richness of sessile species through resource limitation and reduced survival. Competition limits species richness at low levels of disturbance and high levels of productivity (Connell 1978; Huston 1979) (fig. 13).

Disturbance-Productivity Interactions

FRE riparian zones exhibit steep ecological gradients in disturbance and productivity, particularly in arid regions. However, riparian habitats have both the highest potential disturbance and productivity nearest the water's edge. Gradients decrease with distance from, and elevation above, the shoreline (figs. 11 and 13). Stevens (1989) and Pollock et al. (1998) both reported support for the DEC (Huston 1979) for riparian plant species richness along western North American streams. However, Reice (1985) experimentally tested disturbance intensity on stream invertebrates, reporting no support for Connell's (1978) intermediate disturbance hypothesis. Although high levels of disturbance limited diversity in his study, competitive exclusion did not appear to reduce species richness at low levels of disturbance.

The intermediate disturbance hypothesis (reviewed by Wilkinson 1999), as well as the insular biogeography model of MacArthur and Wilson (1967), were developed for sessile taxa (such as plants and corals) but not for vagile species, some individuals of which can actively avoid disturbance events (e.g., stream abandonment behavior by the giant water bug, *Abedus herberti*) (Lytle 1999). However, Townsend (2003) found support for the intermediate disturbance hypothesis for benthic macroinvertebrates among 54 streams with varying histories of flood disturbance, reporting that bed disturbance accounted for the most variation in both sessile and vagile taxon richness. Nonetheless, the role of disturbance and the explanatory power of these FRE biodiversity models vary among aquatic and riparian taxa.

Thus, while efforts have been made to distinguish the impacts of disturbance and productivity gradients in FREs, spatial autocorrelation prevents clear separation of the individual impacts of these two gradients on the structure of riparian vegetation or other sessile taxa (e.g., ant hives). Steady flow systems, such as zero order headwater hillslope springs or seeps that flow into highly disturbed channels, provide a more refined study context than do FREs in which to distinguish disturbance from productivity impacts. The impacts of other gradients (e.g., nutrient availability) on FRE biodiversity and ecological functions can also be more accurately assessed.

Life History Strategies

Life history strategies among aquatic and terrestrial plant and animal species display complex responses to the environmental gradients and gradient interactions in FREs. A central focus of FRE ecology has been classification of guilds—groups of species with similar life history traits, particularly recruitment habitats and interactions among riparian plants (e.g., Johnson et al. 1984)—and feeding or habitat niche use among animals. For example, Hough-Snee et al. (2015) conducted an analysis of woody riparian plant life history traits in relation to environmental gradients and assemblage distributions in the Columbia and Missouri River basins in western North America. They identified five guilds of riparian plants based on rooting depth, canopy height, and resilience to flood disturbance, traits that may sort assemblages under a changing climate.

Grime (1977) identified three main plant life history strategies: short-lived, r-selected ruderal species; long-lived, K-selected competitors; and usually long-lived stress tolerators. Plant species characterized by these life history strategies vary spatially in relation to disturbance and productivity gradients, but disturbance-productivity interactions and competition for nutrients create conflicts within the Grime (1977) model (Craine 2005; Grace 1990; Huston 1979, 1994; Tilman 1988; Walker and Peet 1985). Plant species employ different strategies at different life history stages. For example, Salicaceae and *Tamarix* along southwestern United States streams have an exploitative, ruderal seedling establishment phase, but they also have more competitive and stress-tolerant mature phases (Stevens 1989). Nonetheless, Grime's (1977) three life history strategies may help explain some of Huston's (1979, 1994) dynamic equilibrium predictions: competitors are likely to dominate low disturbance environments, while ruderals are likely to dominate more highly disturbed habitats, and stress tolerators are likely to dominate in low-productivity settings (fig. 13). As a consequence, at least part of the reason that Huston's DEM predicts higher species richness at intermediate levels of both disturbance and productivity is that those gradient positions support all three of Grime's life history strategies.

Feeding guild and habitat use also have dominated classification and applied ecology of aquatic FRE macroinvertebrates. The RCC focused on the downstream transition of secondary production in relation to stream order (although with less emphasis on predators and drifting terrestrial invertebrates), stimulating a rigorous national effort to classify macroinvertebrates as bio-indicators of stream ecosystem health (e.g., Karr 1991; U.S. Environmental Protection Agency 2016; but see Heino et al. 2015 and Merritt et al. 2008). Water acceptable for human needs may support readily identifiable aquatic macroinvertebrate assemblages.However, U.S. water quality and quantity regulations fail to acknowledge that natural non-potable waters (like those commonly occurring in arid regions) support many common, important endemic and some endangered aquatic and wetland taxa and assemblages (e.g., Blinn 2008; Norment 2014).

While predatory invertebrates, amphibians, reptiles, birds, and mammals occasionally exert top-down trophic cascade influences on aquatic ecosystems, fish often are the most influential aquatic FRE species. Fish commonly affect lower trophic levels through both herbivory and trophic cascades. However, most conceptual modeling studies of river fish ecology involve applied species- and river-specific studies, usually directed toward flow regulation, pollution, harvest potential, and other anthropogenic impacts on socio-economically important fisheries. Continental-scale fluvial fish ecology studies (e.g., Dudgeon 2000) and global modeling across continents, river basins, and stream orders remain relatively rare. Ibañez et al. (2009) conducted a comparative study of river fish feeding guild structure in Africa, Europe, and North and South America, reporting general support for RCC-related hypotheses that overall species richness and the proportion of omnivorous species increased over stream order, while the proportion of invertebrate feeders declined. They also reported that piscivore and herbivore/detritivore taxa were relatively depauperate in smaller North American and European streams.

In an effort similar to that of Grime (1977), Bennett (2015) and Mims et al. (2010) used multivariate statistics to reveal three fish life history strategies among North American freshwater fish species: (1) equilibrium fish species with low fecundity and high juvenile survivorship (corresponding to Grime's competitive species), (2) opportunistic species with early maturation and low juvenile survivorship (corresponding to Grime's ruderal species), and (3) periodic species with late maturation, high fecundity, and low juvenile survivorship (somewhat corresponding to Grime's stress-tolerant species). Such convergence of life history strategies among terrestrial primary producers and aquatic consumers suggests that similar biotic and abiotic constraints analogously shape FRE guilds within trophic levels in both aquatic and terrestrial domains.

An ecologically intriguing group of species occupy FRE torrent habitats. Torrent species include organisms as diverse as: aquatic mosses; various riparian plants (e.g., Asteraceae: *Brickellia longifolia,* Poaceae: *Phragmites australis,* and Tamaricaceae: *Tamarix pentandra*); various insect taxa (some plecopteran stone flies; hemipteran belostomatids (Lytle 1999); gerrid *Merobates* and *Trepobates* small water striders; some dipteran tipulid crane flies and simuliid buffalo gnats; some trichopteran web-spinning caddisflies); immature anguillid eels and many salmonid, cyprinid, and other fish taxa; and some bird species (e.g., South American torrent duck [*Merganetta armata*], New Zealand blue duck [*Hymenolaimus malacorhynchos*], and American Dipper [*Cinclus mexicanus*]). Such taxa are specifically adapted for life in high velocity aquatic settings, some even occupying madicolous habitats (shallow cascading flows of white water).

Riparian Vegetation

Overview

Plant species vary enormously in flood and drought tolerance physiology, as well as architecture, reproductive strategies, recruitment strategies, and distribution (Malanson 1993; Reichenbacher 1984). As with benthic invertebrates, the wide array of plant adaptive traits has stimulated many attempts to classify and subdivide upland and riparian assemblages into functional guilds (e.g., Cody 1991; Grime 1977; Hook 1984; Johnson et al. 1984) or mapping units. Such efforts may aid in modeling or management, but life-stage differences and the substantial residual noise in such models reveal the diversity of life history strategies required to cope with FRE spatio-temporal non-equilibrium environmental variability.

Global FRE Riparian Vegetation Distribution

The Holdridge (1947) diagram of global vegetation is based on upland gradients of potential evapotranspiration and annual precipitation across humidity provinces, and it is scaled to latitude and elevation (e.g., Lugo et al. 1999) (fig. 14). Riparian and wetland habitats are not considered in that model. They pose an ecologically and evolutionarily important contrast to upland vegetation: abundant moisture availability in riparian habitats greatly reduces the constraints imposed by evapotranspiration and precipitation, collapsing the two primary axes of the Holdridge diagram to differences among humidity provinces.

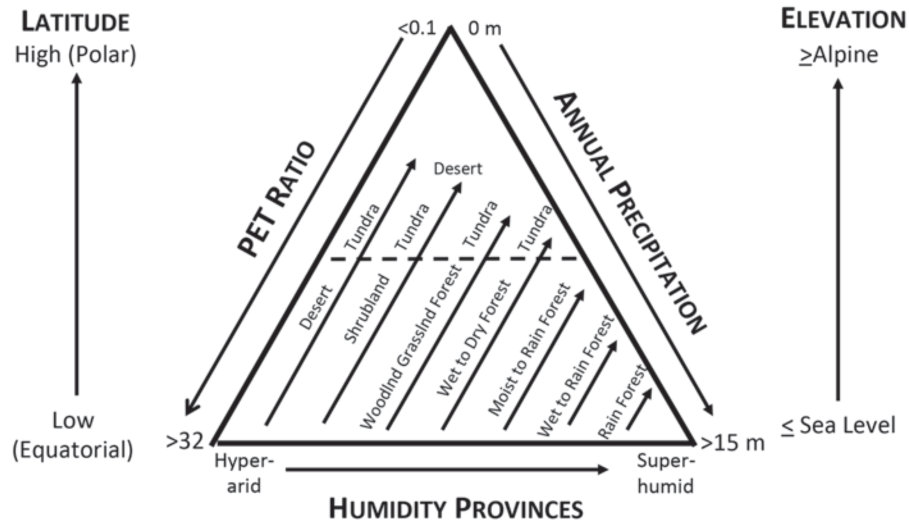

Figure 14—Simplified Holdridge (1947) global upland vegetation structure in relation to gradients of aridity, precipitation, evapotranspiration, latitude, and elevation.

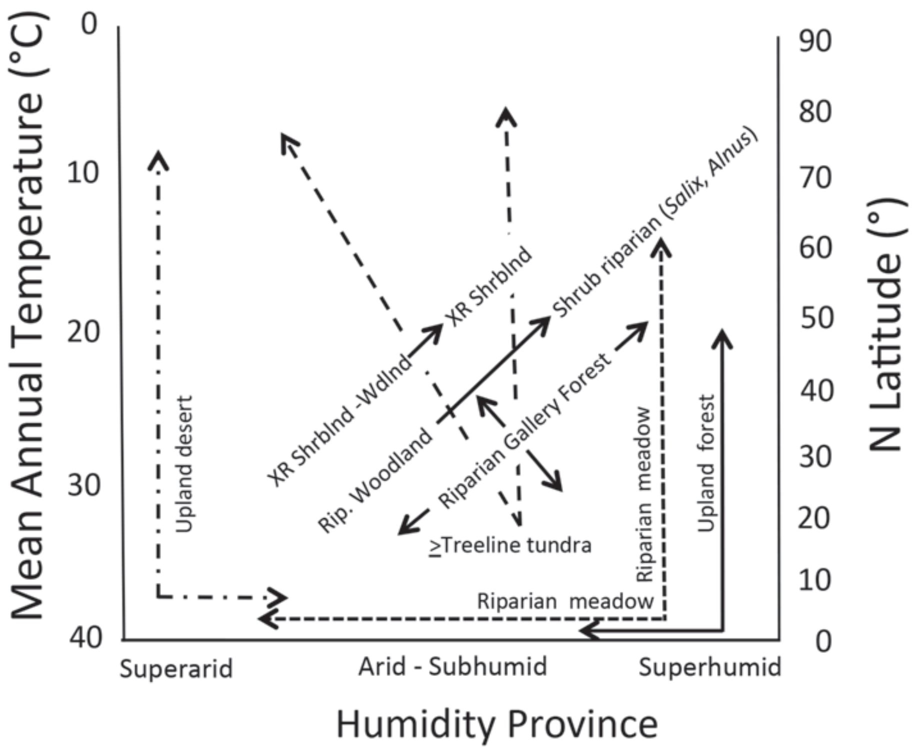

Figure 15—Modified Holdridge (1947) diagram describing the distribution and structure of riparian vegetation. Moisture availability limitations are reduced or eliminated by proximity to groundwater or surface water. Potential riparian vegetation structure is depicted in relation to humidity province, temperature regime, and latitude. Interactions among elevation, latitude, and stage are not depicted here.

Based on our observations across elevation throughout the New World, riparian vegetation appears to be predominately influenced by latitude, elevation, geomorphology, disturbance, groundwater and surface water sources, soil moisture, Holdridge humidity provinces. Riparian vegetation is also influenced to some extent by water quality (e.g., elevated aridity and soil-water salinity impose limits to vegetation at lowest elevations). Temperate regions, especially those in arid and semiarid regions in the New World Northern Hemisphere, often are dominated by galleries of usually deciduous forests (fig. 15). The spatial range of gallery riparian forests is remarkably broad, extending from 20-50° and across elevation at lower latitudes from 0.02-2.5 km. Riparian shrub and woodland vegetation extend even more broadly across latitude and elevation. In humid to superhumid regions, adjacent upland forest vegetation often dominates riparian and ephemeral tributary habitats.

In arid to superarid regions, ephemeral channel xeroriparian habitats sustain temporally varying fluvial groundwater moisture availability, generating reduced but often compositionally diverse channel vegetation. Subsurface fluvial soil moisture availability also often extends across the floodplain, producing aridland riparian zonation and a transition of vegetation—potentially from gallery riparian forest through xeroriparian woodlands and shrublands, to facultatively riparian upland vegetation, to true upland vegetation. Riparian zonation is less apparent but nonetheless evident in mesic habitats (e.g., Hook 1984), with riparian dominance by bottomland tree species, such as *Taxodium distichum* (bald cypress), *Nyssa aquatica* (water tupelo), *Larix decidua* (larch), *Platanus* spp. (sycamores), and several Salicaceae species.

Water Uptake and Flood Tolerance Physiology

Osmotic control on water uptake capacity differs markedly between upland versus phreatophytic or wetland plant species, affecting stand development, composition, habitat structure, and zonation, particularly in arid regions (Carothers et al. 1979; Woodbury 1959). Upland species in arid lands often lack the ability to restrict water uptake, and consequently, they quickly become waterlogged and drown when inundated for even short periods. However, arid land phreatophyte species have the ability to regulate water uptake and many taxa can persist for extended periods of time when inundated (Kozlowski 1984; Stevens and Waring 1985; Warren and Turner 1975). While the evolutionary directionality of this physiological divergence apparently has yet to be explored, we hypothesize that selection has repeatedly and independently favored loss of osmotic control among xerophytic taxa as a derived trait. At any rate, the substantial physiological differences between these two groups of species strongly sort contemporary aridlands FRE plant assemblage composition across riparian terraces in relation to inundation and the depth to groundwater.

Concomitant with upland-riparian variation in water uptake strategies, the diel range of variation in xylem water potential (Ψ) from midnight to midday varies widely among riparian species, and it is based on soil moisture and riparian groundwater depth. Fluvial wetland species exhibit a high, narrow range of Ψ, while middle and upper riparian zone species have progressively lower and broader ranges of Ψ. These trends were abundantly evident among 18 common riparian zone species along the Colorado River in Grand Canyon, sampled at 13 sites with known stage-discharge relationships (fig. 16). This variation in species Ψ ranges is largely responsible for the stage-elevation zonation commonly observed in southwestern aridland riparian plant assemblages.

Zonation

In relation to the above physiological issues, aridland riparian vegetation, as well as lacustrine and lentic aquatic macrophytic vegetation, are characterized by zonation: bands of discrete plant assemblages situated co-parallel to the mainstream channel existing in well-defined stage zones (e.g., Bayley 1995; Brotherson 1987; Carothers et al. 1979; Friedman et al. 2006; Johnson 1991; Spence 1982; Stevens 1989; Woodbury et al. 1959) (fig. 17). For example, Friedman et al. (2006) examined the relationship between flow and riparian vegetation along 58 km of the San Miguel River in southwestern Colorado. They found that riparian plant assemblages were arrayed along the hydrologic gradient, with species like coyote willow (*Salix exigua*) often occupying terraces with flood return frequencies of less than 2.2 years, and river birch (*Betula occidentalis*) occupying terraces just upslope. They also reported that proportional cover decreased upstream, where upland processes such as landslides dominated the channel.

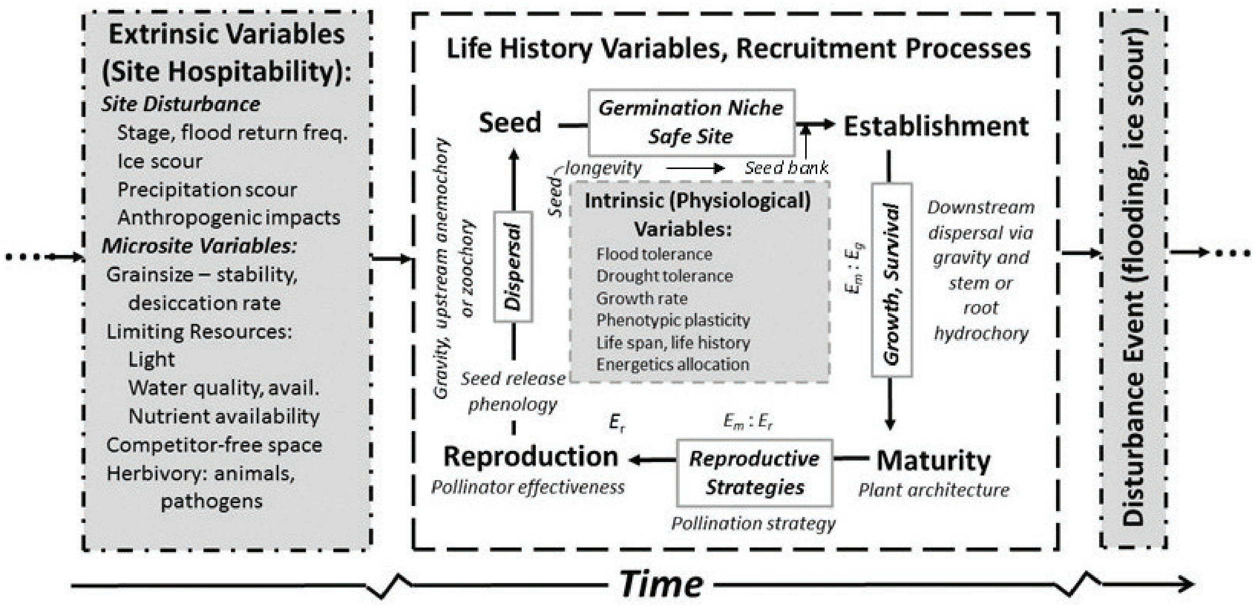

Figure 16—General conceptual model of individual riparian plant species colonization and stand replacement dynamics, with germination, establishment, growth, reproduction, and propagule dispersal occurring in relation to FRE hydrography. E_m, E_g, E_r, and ratios thereof are life history energetics (E) of plant growth (g), maintenance (m), and reproduction (r), respectively.

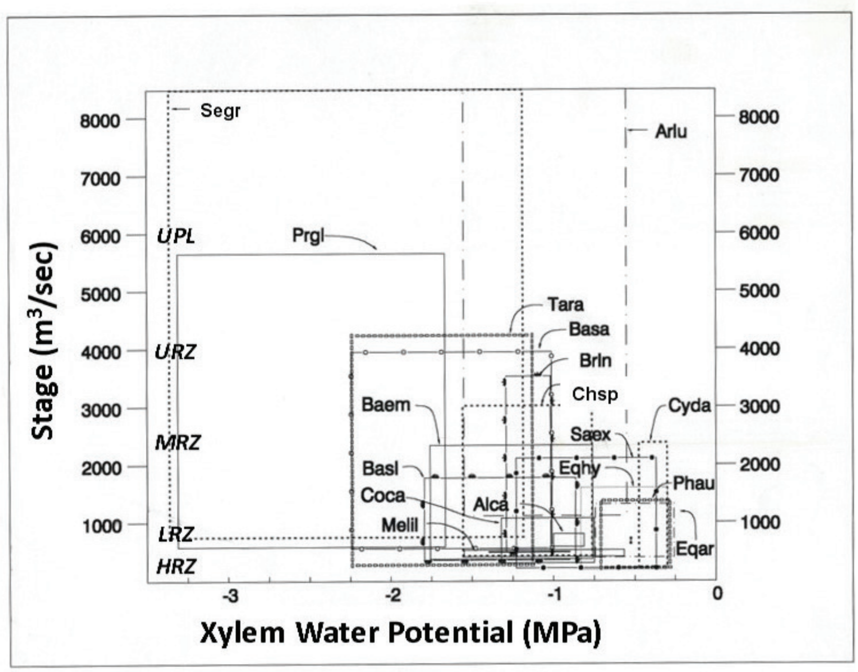

Figure 17—Range of stage elevations of 17 common obligate and facultative riparian plant species at 13 sites along the Colorado River in northern Arizona in relation to the full operating range of xylem water potential (MPa) for each species from midday (low) to night (high). HRZ—hydropriparian zone, LRZ—lower riparian zone, MRZ—middle riparian zone, UPL—upland zone, URZ—upper riparian zone. Plant species abbreviations: Alca—*Alhagi maurorum,* Arlu—*Artemesia ludoviciana,* Baem—*Baccharis emoryi,* Basa—*Baccharis sarothroides,* Basl—*Baccharis salicifolia,* Brln—*Brickellia longifolia,* Chsp—*Chlorocantha spinosa,* Coca—*Conyza canadensis,* Cyda—*Cynodon dactylon,* Eqar—*Equisetum arvense,* Eqhy—*Equisetum ferrissii,* Melil—*Melilotus* spp., Phau—*Phragmites australis*, Prgl—*Prosopis glandulosa,* Saex—*Salix exigua*, Segr—*Senegalia greggii,* Tara—*Tamarix* spp.

In addition to xylem water potential, riparian zonation also arises from variation in phenotypic plasticity of plant species in response to geomorphology and the disturbance regime, soil texture, and soil water and nutrient availability, as well as root architecture and rooting depth. Stevens (1989) grew 12-28 seedlings of 18 common southwestern riparian plant species for 1 month and measured the dry biomass of above-ground and root growth in fine, silty, nutrient-rich pre-dam Colorado River soil (the best naturally-available substratum) versus coarse, sandy, nutrient-poor post-dam soils (fig. 17). We used those data to calculate the phenotypic plasticity index (PPI_i) for each species I as the ratio of average dry post-dam belowground growth to average dry above-ground dry growth, as:

$$PPI_i = \frac{\sum_{i_{post}=1}^{i_{post}=N_{i\_post}} \left(\left(m_{ipost\_dbg}/m_{ipost\_dag}\right)\ldots\left(m_{Ni\text{-}post\_dbg}/m_{Ni\text{-}post\_dag}\right)\right)/N_{i\text{-}post}}{\sum_{i_{pre}=1}^{i_{pre}=N_{i\text{-}pre}} \left(\left(m_{i\_pre\_dbg}/m_{i\_pre\_dag}\right)\ldots\left(m_{Ni\_pre\_dbg}/m_{Ni\_pre\_dag}\right)\right)/N_{i\text{-}pre}}$$

where $m_{i\_post\_dbg}$ and $m_{i\_post\_dbg}$ are the average dry biomass of below- or above-ground growth of N_{i_post} seedlings of species *i* grown in (suboptimal) post-dam fine-medium sand, compared to that average for seedlings of species *i* grown in optimal pre-dam fine silty sand.

For example, seedlings of *Baccharis salicifolia* seepwillow, a weedy LRZ shrub with high phenotypic plasticity, increased its relative allocation from aboveground leaf and stem growth by 5.5-fold to roots when grown in nutrient-poor post-dam sand.

We also report species-specific PPI responses to reproductive strategy, seed size, and seed longevity measured under field conditions. Ruderal *Baccharis salicifolia* and some nonnative species (e.g., *Tamarix* spp.) generally had higher PPI than did long-lived K-selected tree species, such as *Fraxinus pennsylvanica* (velvet ash), *Populus fremontii* (Fremont cottonwood), or *Salix gooddingii* (Goodding's willow). K-selected *Prosopis glandulosa* (honey mesquite), which can live more than 800 years in the URZ in Grand Canyon (R. Hereford, U.S. Geological Survey, personal communication), has a large, long-lived seed but also a moderately high phenotypic plasticity, conferring upon it an adaptive advantage in unpredictable habitats. Collectively, this analysis indicates that no single plant life history trait explains the success of any plant species in the riparian environment. It suggests that compensatory options exist within the suite of a plant species life history traits to permit survival in non-equilibrium riparian ecosystems.

Riparian vegetation zonation also is common in mesic and humid-superhumid regions, but it is often less distinguishable than in arid regions. In humid environments, many upland species are tolerant of waterlogging and may have similar among-species variation in root architecture, plasticity, and Ψ range. For example, Dowe (2008) identified 263 plant species in three superhumid northern Australian watersheds, of which only 23 species (8.7 percent) were obligate riparian species. Oreliana et al. (2012) reviewed and synthesized water use models for groundwater-dependent plant species, suggesting that more research was needed to clarify riparian water use, and to differentiate water use from saturated versus unsaturated zones.

Riparian Plant Recruitment and Stand Replacement

The niche-box model (NBM) (Merritt et al. 2010) classified guilds of riparian plants in relation to similarities among life history traits. The NBM incorporates and compares many autecological elements for each plant species to improve prediction of vegetation assemblage development in relation to hydrography and riparian conditions. While successfully grouping some species, the large amount of variation in the niche-box model multivariate plots reminds us about the tremendous variation in life history strategies among riparian plants, variance that is highly adaptive but which does not readily lend itself to simple classification.

We describe and illustrate the life cycle of an individual riparian plant species to clarify autoecological life history, energetic constraints, and stand replacement potential (fig. 18). HAZ-MRZ plant recruitment often takes place on the descending arm of the hydrograph after the most recent flood or ice scour (e.g., Fenner et al. 1984; Rood et al. 2007). Freshly scoured, moist, fine-grained substrata are exposed and serve as potential regeneration niche sites for germination and establishment (Grubb 1977; Harper 1977). The characteristics of such "safe germination site" vary by species and season and are the most critical phase of successful recruitment. Safe sites for seed-reproducing phreatophytes with small or short-lived seeds generally require open, moist, silt-rich sediment deposits, and those species typically have high seedling mortality and extensive self-thinning (Type III survivorship curve; Deevey 1947). Common phreatophytes (e.g., wetland herbs, such as *Carex*, *Juncus, Phragmites, Schoenoplectus, Typha,* and some clonal *Salix*) often disperse through hydrochory as rhizomes or rootable stems, and sometimes through zoochory (e.g., beaver dispersal of rootable stems). Plant species with larger seeds (greater maternal investment; e.g., *Prosopis*) may germinate in shady habitats and display either Type I or II survivorship, vigorous growth, and less ecotypically plastic architectural responses (Stevens 1989).

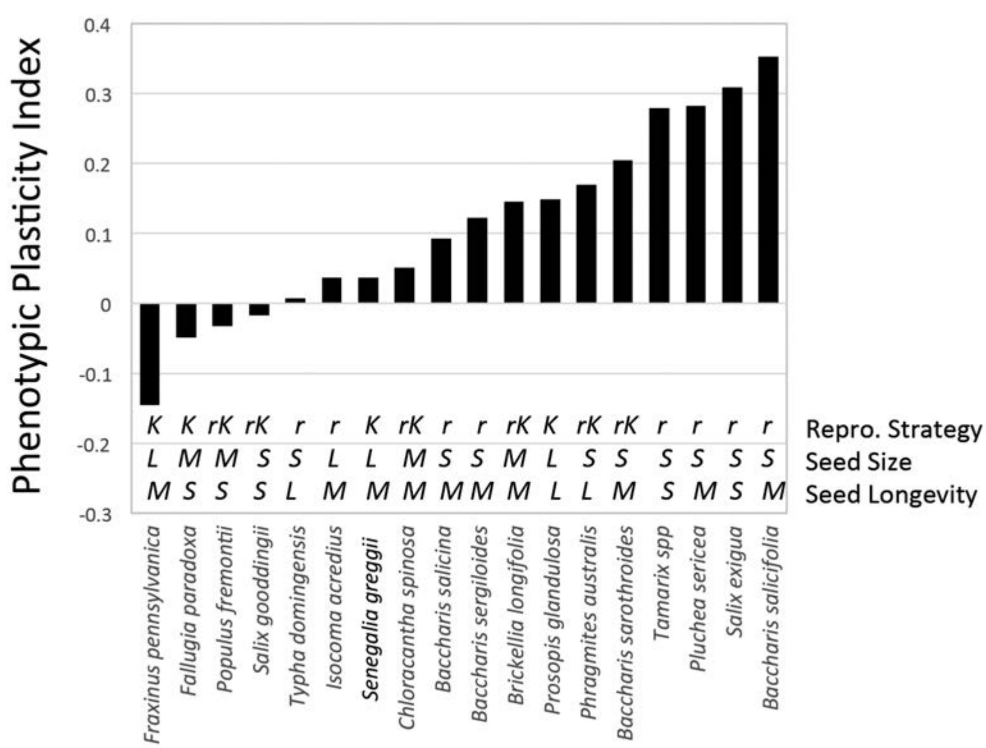

Figure 18—Phenotypic plasticity index of 18 common riparian plant species in the American Southwest (see text for calculation). Reproductive strategy: K selected, r selected, or rK intermediate. Seed size (relative): S small, L large, M medium. Seed longevity: L long-lived (> 2 years), M medium-lived (0.2-2 years), S short-lived (< 1 month).

Following germination, establishment takes place as stems and roots grow at species- and microsite-specific rates (Stevens 1989) (fig. 18). Energy for growth (E_G) is that remaining after the individual expends energy on maintenance and respiration (E_M and E_R, respectively), which are relatively greater in more stressful habitats. Subsequent survival to recruitment age varies by species, location (i.e., stage elevation), and environmental conditions, including disturbance, nutrient availability, competition, herbivory, and other factors. Annual riparian species, living between flood spates, face a boom-or-bust fate. They maximize E_G, E_R, and the energy devoted to propagule production (E_S), perhaps by reducing emphasis on E_M or maximizing ecotypic plasticity. The persistence of annual riparian species within the reach requires relatively high levels of maternal investment in seed production, perhaps coupled with low seed longevity (i.e., < 1 year) and either continuous recolonization from upstream sources or *in situ* propagule retention, recruitment dynamics that have been little studied. In contrast, biennial and perennial plant species can adaptively balance E_G, E_M and E_R in relation to E_S. Individuals may defer E_S costs during periods of unusual stress (e.g., exceptional flow years), allowing those individuals to reallocate E_S energy to E_G and E_M and even defer reproduction if necessary.

Riparian Succession

Vegetation succession (predictable change over time) occurs through three modes: facilitation, inhibition, and tolerance (Connell and Slayter 1977). Riparian successional trajectories may differ between Holdridge humidity provinces and in relation to stream order, fluvial hydrodynamics (disturbance frequency), geomorphic setting, grainsize distribution, depth to water table, and biological effects—conditions that collectively create the mosaic template on which vegetation develops (Stanford et al. 2005). However, variation in successional trajectories has been little studied across stream order. At one extreme, zero and first order streams have insufficient stream power to prevent wetland (if open) or woody (if forested) vegetation from colonizing stream terraces. In such low-disturbance settings, riparian vegetation may entirely overwhelm the FRE, eliminating surface water (e.g., Kodrick-Brown and Brown 2007). However, at higher stream orders (larger, more highly disturbed riparian settings), the trunks of trees and driftwood piles may resist scour and affect channel geometry by stalling flow and depositing sediment, even to the extent of creating mid-channel islands (e.g., Tockner et al. 2003).

Geomorphic setting exerts dominant influences over riparian plant successional processes and modes within reaches, but its role often has been obscured in floodplain studies. Succession in middle- to higher-order unaltered rivers may be suspended in hydrologically active to middle riparian zones because annual-biennial flooding or ice scour resets the riparian zone to an unvegetated state (Campbell and Green 1968; Prowse and Culp 2003) (fig. 19). Lower riparian terraces are highly productive, but the frequency of scour favors species capable of sweepstakes colonization through germination, rapid exploitation of available space and resources, and intense competition before the next scouring event, and therefore favor r- (ruderal) over K- (competitive) selected species. Rapid growth of or colonization by woody phreatophytes may reciprocally force channel narrowing or alter meandering in some alluvial reaches (e.g. Johnson 1994; Hupp and Osterkamp 1996). However, flow events shape channel geometry in more constrained reaches, forcing riparian vegetation there to respond to, rather than reshaping, bar configuration (e.g., Birkeland 1996).

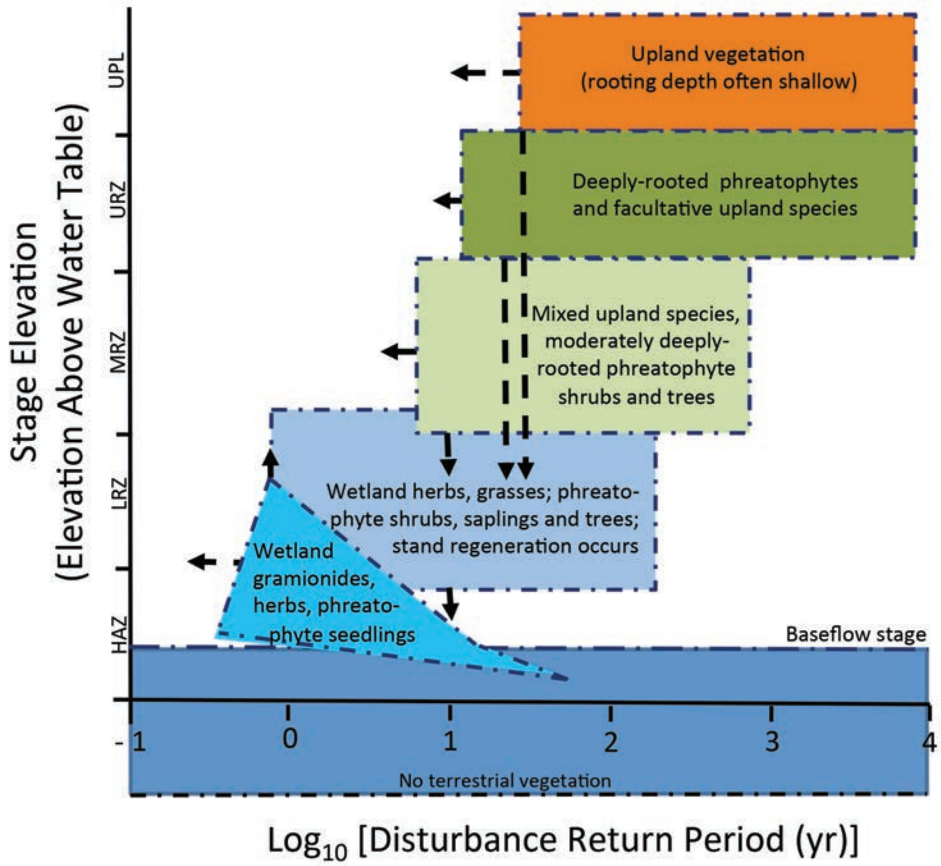

Figure 19—General model of temporal development or succession of aridland riparian plant zonation in relation to stage elevation and log10 (disturbance return period). Black lines show tendency of vegetation spatial and temporal distribution in mesic (humid) environments. HAZ—hydrologically active zone, LRZ—lower riparian zone, MRZ—middle riparian zone, UPL—upland zone, URZ—upper riparian zone.

Clonality in lower riparian zone species adds a further level of complexity to succession: Clonal *Phragmites* and *Arundo* grasses strongly resist scour and can deter sediment deposition during high flows, effectively armoring channels (e.g., cane breaks along the Lower Colorado, Rio Grande, and Mississippi Rivers). The root stocks of these and some woody phreatophyic clonal or rhizomatous species are highly flood tolerant and can persist belowground during scouring events, vigorously resprouting and growing out following disturbance, exploiting space and nutrients and quickly resuming dominance. Clonality also means that the genet (genetic individual) can move over time, as one portion of the root mass may be scoured away while another portion survives and regrows. Similar rapid colonization can occur on higher terraces after larger scouring events, but more time between disturbance events can allow other successional modes to occur.

Several recent studies have identified analogous successional patterns in riparian habitats. Naiman et al. (2005) identified four general stages of riparian succession: establishment, competitive stem exclusion, understory initiation, and maturity. Whether and how this progression occurs on all terraces remains unclear, as do the extent to which such patterns occur across latitude and humidity provinces. Egger et al. (2015) modeled terrace-based riparian succession in the Kootenai and Flathead Rivers in northwestern United States and southwestern Canada. They reported that individual species occupied similar hydraulic environments in different reaches, and that following impoundment (cessation of flood and ice scour), initial colonization and dominance by cottonwood (*Populus*) gave way over 60-150 year time scales to dominance by spruce (*Abies*). Although the mode was not identified, the latter transition likely occurred through facilitation as the conifers establishing under deciduous canopy shade, coupled with tolerance as the conifers out-lived the co-occurring cottonwoods. In those temperate latitudes, the two taxa apparently share similar flood tolerance and root depth-to-groundwater relationships.

Geerling et al. (2006) and Metz et al. (2016) reported three alternative successional trajectories on floodplains of the lower Allier River in France over 60 years of analysis. These alternative trajectories included: (1) progression (successional transformation from water to exposed sediment to colonization, first by pioneer species, to transitional grasslands and shrublands to riparian forests); (2) retrogression (a reversed trajectory toward bare soils or open water); or (3) stable (unchanging and stand-replacing patch conditions). They did not relate those three Markovian alternative states to the species-based successional modes of Connell and Slayter (1977), but such an analysis would likely be integrative.

Surrounding upland assemblages strongly interact with upper riparian terrace vegetation. Hence, the mature (equilibrium or climax) vegetation stage in the above studies becomes a mixture of downslope-colonizing upland species on upper riparian terraces and, in arid regions, dominance by deeply rooted, long-lived phreatophytes that maintain root connections to the river water table (e.g., Woodbury 1959, Carothers et al. 1979).

Riparian plant succession also can be directed by biological processes. Mycorrhizal succession has been identified along Montana rivers, with arbuscular mycorrhizae succeeding to ectomycorrhizae at decadal or longer time scales (Piotrowsi et al. 2008). The trajectory of riparian plant succession also can be altered by selective herbivory, particularly by ecosystem engineering rodents like beaver (Bailey and Whitham 2006), herbivorous insects (Simieon and Stevens 2015), plant diseases, and even some bird species (Stevens 1989). Overall, these studies indicate that reduction or elimination of flooding disturbance promotes as-yet-poorly-described stage-specific FRE succession.

FRE Evolutionary Ecology

FREs have been persistent habitats over evolutionary time, as demonstrated by the broad array of paleo-landforms and fossil deposits throughout the world (Behrensmeyer et al. 1992) and the enormous biodiversity and influence of rivers on upland biota. Evolutionary isolation and gene flow restriction can occur at microsite, reach- and among-basin scales, as well as between aquatic and riparian domains. Headwater elevation may influence along- and across-channel colonization (e.g., Vences et al. 2009). Stevens (2012) reported that the 32 km long, steep, canyon-bound Muav Gorge reach separated the Colorado River ecosystem in Grand Canyon into two basins: an isolated eastern basin that contains most of the river corrido's endemic plant and animal species, and the more open western basin dominated by Mohave Desert species and containing fewer endemic species.

Temporally, vicariance or large antecedent events, such as long-term drainage basin integration or natural or anthropogenic impoundment events, may contribute to isolation, restricting gene flow sufficiently to allow gene fixation. In addition, vegetation change in response to a drying climate across elevation in deep canyons may restrict across-basin colonization. An example is the well-known case of upland population isolation between

Kaibab and tassel-eared squirrels (*Sciurus aberti kaibabensis* versus *S. a. aberti*) on the North and South Rims of Grand Canyon, respectively, following repeated Pleistocene inter-glacial habitat separation events (Jones and Wettstein 1997).

At a finer scale, microhabitat isolation is particularly pronounced in large, deep canyons, resulting in isolation and formation of endemic species (Stevens 2012). Environmentally constant and harsh, in-canyon springs, caves, north- and south-facing slopes, and rim edges are settings in Grand Canyon that foster development of endemism. Aridland springs in particular are renowned as isolated hotspots of endemism due to their unique water quality and environmental constancy (Kreamer et al. 2015; Stevens and Meretsky 2008). FRE endemism in deep canyons may develop sequentially as microhabitats and associated species become increasingly isolated and rare during drying climate phases. Climate recovery can subsequently allow re-expansion of habitats and associated species—and in speciation events through adaptive radiation that increases biodiversity, as has occurred with hydrobiid springsnails (Hershler and Liu 2008) and cyprinodontid pupfish (Martin and Wainwright 2013).

Local genetic adaptation is likely an important characteristic of *r*- and intermediate *rK*-selected riparian species. For example, following flooding, LES monitored a 10 m^2 patch of Lower Colorado River riparian zone habitat that, after 1983 flood subsidence, supported germination of > 10^4 seedlings of woody riparian species, including *Tamarix*, *Salix*, and *Populus*, but on which only a single individual *Tamarix* seedling survived to reproductive age. Such intensive selection results in highly individualistic, site-specific local adaptation. Subsequent reproductive mixing of those traits is likely to enhance species-level fitness by generating elevated genetic heterozygosity. Consequently, local-scale endemism tends to be relatively rare among North American riparian plant species, although this may not be the case in tropical rivers due to higher productivity (e.g., Dowe 2008; Harrison and Grace 2007).

FRE Ecology Research Recommendations and Conclusions

Fluvial-riparian ecosystems are hierarchically and dynamically influenced by physical and biotic processes that vary spatially over stream order and time within the watershed, approaching but rarely achieving equilibrium conditions in channel geometry, fluid and matter transport, ecosystem energy dynamics and structure, and ecological developmental state. A wide array of conceptual models has been used to describe aspects of FRE ecology and responses to natural and anthropogenic perturbations. Most models have focused on single or a reduced suite of variables at site-specific, within-reach, or other incomplete watershed scales, and most often of anthropogenically altered streams. In the WCM, we emphasize the importance of understanding temporal and spatial scaling across the entire basin.

Despite much progress, a wide array of important ecohydrological processes, questions, and issues remain to be addressed or more fully investigated and integrated into the WCM. Not presented in prioritized order, this list of additional research topics includes but is not limited to: (1) the roles of self-similarity across reach and stream order spatial and temporal scales; (2) groundwater-surface water interactions and connectivity; (3) the significance, extent, and roles of groundwater and headwater springs as zero order streams in FRE ecology (*sensu* Gomi et al. 2002; Lowe and Likens 2005); (4) interrelationships among lentic and lotic habitats; (5) the distribution, importance and ecology of ephemeral streams and xeroriparian ecosystem ecology; (6) the interrelationships between interconnected ephemeral and xeroriparian riparian ecosystems; (7) the roles and importance of aspect, gradient, and photosynthetically active radiation in canyon-bound stream segments (e.g., Stevens 2012; Yard et al. 2005); (8) the ecological importance and roles of river microclimate; (9) the multi-dimensional roles of flooding, ice, and glacial effects, especially in boreal and high-elevation rivers; (10) multidirectional material and gene flow in dendritic pathways, which are dominated by downslope gravity and hydrochorous material, nutrient, and propagule transport, but that also have ecologically important upriver eolian and zoochorous transport mechanisms; (11) the role of plant physiology in riparian vegetation distribution; (12) the biogeographic significance of rivers as corridors, barriers/filters, and refugial systems (Stevens 2012); (13) stream order-driven and across-channel spatial impacts on biodiversity; (14) population and successional models among FRE biota and trophic levels; (15) FRE ecosystem genetics and evolutionary processes, including the development of endemism across latitude, longitude, and among tectonic landscapes; (16) climate change influences on FRE form and function; and (17) the role of noise on riparian songbird assemblage composition and structure.

Adequately incorporating the above research topics, and more fully constructing and testing the WCM, will require another generation of research, including collaborative discussion among hydrogeological, ecological, and socio-cultural disciplines (e.g., Fisher 1997). Such data and integration efforts are needed to improve understanding, modeling, and stewardship of FREs at local, regional, and global spatial and temporal scales.

River ecosystems are extraordinarily complex and vital to life on Earth. Informative and elegant as they are, the FRE models proposed to date remain incomplete. There also are many challenges associated with inadequacy of the physical and biological data required to calibrate and refine existing models and to develop new models. Here we attempted to summarize and illustrate the state of knowledge for FREs, and we point out additional elements that need further investigation and better integration. However, FREs cannot be readily, adequately, or usefully reduced to a suite of equations or simple illustrations. For example, some cultures commonly view rivers as living beings, supporting divine spirits. Integrating indigenous traditional ecological concepts and knowledge into improved stewardship has rarely been attempted. We suggest that improved comprehension of FREs may require consideration of other socio-cultural dimensions.

John Wesley Powell, whose quote opens this chapter, suggested that rivers can be understood as music. Rather than a Hutchinsonian stage on which the ecological play is enacted in the theater of evolution, a river ecosystem might better be considered as a time-transitive orchestral composition. Such a symphony might be composed of themes brought in by each major tributary and integrated in its watershed and geographic setting. In form, the symphony might be a fugue- like integration of variably self-similar subthemes amplified across stream order, with rhythms reflecting reach geomorphology, ecological character, and tempo. Each movement in the symphony might be a time-step in watershed development: the opening sonata describing the basin's geologic origination, the second movement portraying the FRE in its natural state, the third depicting the FRE under the terms of contemporary anthropogenic influences, and the final movement recapitulating the river's ultimate stratigraphic and evolutionary contributions. Given the precepts of Morisawa's (1968) dynamic equilibrium model, we do not expect this fluvial symphony to be either particularly orderly or melodic.

Many rivers, large and small, have come and gone on Earth. Seen in cross section, 400-million-year-old Devonian river channels in upper Grand Canyon that intersect the modern Colorado River still serve as conduits for groundwater flow and perhaps direct the course of contemporary tributary incision (Stevens 2013). Thus, the spatial and temporal impacts of large paleo-FREs may last for hundreds of millions of years.

Would such a symphonic model help move river and watershed science forward? Each river's symphony can help remind us of our species' evolution and history, the limits of our capacity to understand multi-dimensional reality. Rivers inspire our core sense of wonder, our fear of, and our deep need for flowing and lentic freshwater, elements that have stimulated abundant fundamental, integrative, and applied FRE science. Composition of symphonies for major rivers would be a worthwhile artistic endeavor that might more broadly engage the public and the artistic community in environmental stewardship. But like all models, even the most elaborate river symphonies would fall short of fully representing these remarkable, important, and dynamic ecosystems.

A rich array of research topics and endeavors awaits future students of FRE ecology, and scientific research, artistry, and public involvement all continue to be urgently needed. Improved stewardship of the world's rivers to sustain vital ecological functions is essential to sustaining life and socio-economic well-being. This effort remains a critical challenge and responsibility for all of humanity. We welcome comments and suggestions on this synthesis, and we hope this chapter helps to stimulate the research, synthesis, and communications needed to reach this goal.

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Chapter 6. It's Not All Bad News—Riparian Areas in the Anthropocene

Kenneth J. Kingsley

Introduction

An abundance of information, some of which is included in other chapters in these volumes, documents and laments the historic loss of or changes to riparian ecosystems. Historic impacts to most riparian organisms, ranging in size from cottonwood trees to microbes, are fairly well documented. Causes of these changes are also well documented or speculated upon based on the best available information. It is clear that the world, and specifically the riparian world, has changed because of human activities. It appears likely that changes will continue to occur as human activities continue, and that our impacts are sufficient to cause the planet to enter a new era of geology, atmospheric chemistry, and biology termed the Anthropocene (Crutzen and Stoermer 2008). Most of the changes that have occurred historically are generally considered "bad" because they involve a loss of biological communities that we usually consider "good." A small industry has developed involved in the protection of remaining riparian areas and wetlands and restoration or creation of new ones to compensate for historic losses and solve other problems.

This industry has met with mixed results, but complete restoration or creation of fully functional self-maintaining wetlands has probably not been achieved and may not be entirely desirable (Karpiscak and others 2004; Willott 2004). This industry is largely supported by funding by the U.S. Army Corps of Engineers and Environmental Protection Agency for restoration or wastewater treatment projects, although some may be supported by private, local, or regional sources. Much effort is currently being directed toward intentionally "restoring" riparian communities, or at least the most conspicuous desirable indicators of those communities, and protecting those remaining riparian communities that appear to be somewhat intact. These efforts continue to lead to a greater understanding of the nature of riparian communities. Many components from microbial to landscape and planetary are necessary to make healthy ecosystems.

Historically throughout the southwestern United States, rainwater sank into permeable ground or ran off impermeable or saturated surfaces into natural channels. These confluenced with increasingly larger streams until eventually the water sank into the streambed or ran to a sea or natural lake. Riparian communities developed along these natural drainage ways. The natural drainage scheme has been diverted, obstructed, reoriented, eliminated, or otherwise altered by human activities for thousands of years. In the most recent period of the Anthropocene, the permeability of the watershed substrate has been changed by overgrazing of watersheds, fires, invasion of alien plant species, trampling by livestock and people, and paving. Drainage ways have been channeled, blocked, driven underground, piped, dammed, and diverted. Especially in urban areas, what comes down from the sky no longer follows natural paths but travels through a maze of structures designed (or not) to put it somewhere else.

One could lament the loss of the natural riparian communities that were once part of the natural hydrologic pathways. One could also learn to appreciate some of the ways people and nature have combined to create new and different (and, frankly, often inferior) riparian communities. Here, my purpose is to look at some examples of riparian communities that have been developed, intentionally or not, in association with human attempts to deal with the hydrologic cycle. Most of what follows is based on personal observations over several decades, and may not be documented or supported by scientific or other literature. When appropriate references are available, they are given.

Wastewater Treatment Plants—A Source of Water

Example 1: Santa Cruz River Downstream From Nogales and Tucson

The Santa Cruz River was historically an interrupted perennial stream (Stromberg and others 2012; Webb et. al. 2014) that began in the Canelo Hills of southern Arizona, passed briefly into Mexico, and then made a northward turn to run some 200 miles to join the Gila River. Today, the same generalization can be made, but historically wet reaches have become dry, except during brief spates, and historically dry reaches are now perennially wet. Detailed descriptions of the historic river and anthropogenic changes to it are given in many sources, most notably the recent volume by Webb and others (2014) that includes a compilation of historic records, including photographs and descriptions by early settlers and naturalists. More recent, frequently updated descriptions of conditions can be found on the website of the Sonoran Institute (https://sonoraninstitute.org). Groundwater pumping and watershed changes are primary reasons for the drying of the historically wet reaches and treated urban sewage effluent is the sole reason for some reaches to be perennially wet today.

Over the past two decades, a growing body of literature examining the currently wet reaches has developed and is expected to expand as additional studies are conducted. Much of this literature is in the category of "gray literature," consisting of agency and consulting reports, synopses, or abstracts of meeting presentations, theses, dissertations, and web-based information. A review of this literature and description of conditions in 2013 was done by Pima County (2013). Updates of many current research projects and conditions are available from the Sonoran Institute website (https://sonoraninstitute.org) or by following leads on that website. It is clear that a new and dynamic river has emerged, differing greatly from the former in flow regime, water quality, substrate qualities, bank morphology, wildlife and plant uses and distribution, and human uses. Any description of "current" conditions will be of transitory value, and the situation will change, sometimes quite rapidly. This brief section describes conditions as they were primarily during the first decade of the 21st century. Several restoration projects of varying scales and approaches are in progress at this time (2018) and more are likely to be developed.

In 2018, there were two perennially wet reaches, one carrying effluent from the Nogales International Treatment Plant in Nogales, Arizona (Upper Santa Cruz), the other carrying effluent from treatment plants serving the Tucson area (Lower Santa Cruz).

Effluent input began on the Upper Santa Cruz in 1951 and on the Lower Santa Cruz during the 1970s, when the urban populations were less than half of present, and when treated effluent discharge into rivers was considered an expedient way to get rid of a problem. There were few regulations and little attention directed toward the discharge. Water flow in these reaches is subject to daily and seasonal fluctuations, depending on water use in the areas served by the treatment plants. Longer-term changes occur as the urban populations grow, demand for effluent for other uses varies, and sewage treatment processes develop. The amount of effluent in the Upper Santa Cruz was greatly reduced in 2012, changed in quality in 2013, and is expected to be practically eliminated as new technology is applied in Mexico. Added to input are occasional floods following precipitation events in which urban stormwater runoff and drainage from undeveloped portions of the watershed are added to the effluent flow.

Water is removed from these reaches by penetration into the substrate, diversion for irrigation, and evapotranspiration by vegetation. The depth to groundwater has been lowered so much by groundwater withdrawal that there is little or no immediate connection between the deep aquifer and surface water, and the zone of water availability to plants is limited to the immediate vicinity of flow channels. Water availability to plants and subsurface recharge may be reduced by buildup of a clogging coating of relatively impermeable material (schmutzdecke) caused by physical, chemical, and biological processes. This material may be removed at unpredictable intervals by flooding caused by summer monsoon storms. If there is no flooding, then riparian vegetation may die. This is an "inherently unstable" situation, which may result in rapid loss of riparian communities if no human intervention occurs to break up the clogging layer (Treese and others 2009). Innovations at the Nogales treatment plant in 2009, and subsequently, have led to an improvement in water quality, reduction of schmutzdecke, and return of native Gila topminnows (*Poeciliopsis occidentalis*) to a portion of the currently wet Upper Santa Cruz (U.S. National Park Service 2015).

Water quality is different from that of natural water sources and varies considerably with distance from the inlets, natural input events, changes in human population and water use, and sewage processing technology. A change in water quality for the Lower Santa Cruz followed the 2014-15 activation of new technology that is part of the Regional Optimization Master Plan. This resulted in a decrease of schmutzdecke and an increase of diversity of aquatic macroinvertebrates and fish including Gila topminnows (Sonoran Institute 2017). Prior to this change, water quality was such that many species of aquatic animals and plants that formerly lived in wet reaches of the river could not survive, whereas others, including nonnative invasives, thrived.

Propagules of native species that formerly inhabited the area may not be available, either because the currently wet areas never had them, or because populations have been extirpated from the watershed. Temperature is higher at effluent inlets than downstream. Nitrogen compounds are high near effluent inlets and attenuate downstream. Ammonia levels may be toxic to fish, amphibians, and invertebrates for a portion of the stream. Benthic macroinvertebrate communities vary with distance from inlets and concomitant changes in several parameters of water quality (Boyle and Fraleigh 2003), with taxa that are tolerant of pollutants present near the inlets and diminishing downstream, and intolerant taxa absent near inlets and becoming established downstream. Benthic macroinvertebrate communities are greatly limited by substrate condition (mostly sand and schmutzdecke, little gravel, few cobbles, very few boulders). They are also limited by flow amounts and distributions, which vary through a wide range hourly, daily, and seasonally.

Microbial communities, including potentially pathogenic microbes, may be greatly influenced by introduction of propagules and living organisms in effluent and by variation in conditions downstream from inlets (Duran and Spencer 2004). Unique mixtures of chemicals, including biologically active compounds such as pharmaceuticals and endocrine disruptors, occur and may impact the plants and animals in and along the stream.

Riparian vegetation, including cottonwood-willow forests, have developed in patches that formerly had only xeroriparian scrub, and a few areas (near Tubac and Tumacácori National Historical Park) of previously developed riparian forest and woodland have been nourished and enhanced by the effluent. Through much of the currently wet reach of the Lower Santa Cruz, the banks have been deeply incised and are now lined with soil cement to prevent bank erosion and flooding. Although the newly developed riparian areas have many characteristics of historic pristine conditions, they also differ in that they have a mix of native and nonnative plants and animals and lack many species that were historically found in the wet reaches of the river.

For the most part, riparian trees are smaller in stature under the new conditions than those found in more natural riparian areas, but that may be because they are younger. Native trees include Fremont's cottonwood (*Populus fremontii*), Goodding's willow (*Salix gooddingii*), velvet mesquite (*Prosopis velutina*), and blue palo verde (*Parkinsonia florida*). Nonnative trees include "Chilean" mesquite (*Prosopis* sp.) and possible hybrids with natives, Jerusalem thorn (*Parkinsonia aculeata,* apparently nonnative in this situation), athel tamarisk (*Tamarix aphylla*), and saltcedar (*Tamarix ramosissima*). The mix of species in any given reach may be unique and may change over time.

Because the depth to a zone of permanent available water is greater than the depth to which roots can reach, if the flow is reduced or changes channels slightly, riparian vegetation may be cut off from water and die. Changes in effluent amounts released by treatment plants may have profound impacts on vegetation. Minor channel changes caused by floodwaters rearranging the substrate can result in loss of blocks of riparian vegetation. Floodwaters cannot spread across a broad floodplain but are confined within soil cement walls through much of the urban area. This confinement increases the velocity of floods and damage to river bottomland vegetation when floods occur.

Many species of native birds have discovered and adapted to the newly established riparian areas. Some of the areas have become quite popular with local birders and appear on bird guidebooks and websites. A list of species found has been growing over many years, and migrating species that are outside their normal range are frequently spotted. In a yearlong study of birds using five different sites along the effluent-dominated Lower Santa Cruz, 133 species were recorded (SWCA 2000). There were differences between sites, between survey dates on each site, and between seasons. Most of the differences between sites can be attributed to the relative composition of vegetation associations at each site. Where trees, especially Fremont's cottonwood and Goodding's willow, were dominant, native bird species typically found in native riparian habitats were present. However, because blocks of trees were narrow and most of the trees young, many species of birds typically found in large blocks of native riparian habitat were missing (e.g. yellow-billed cuckoo *Coccyzus americanus*) or rare (e.g. summer tanager *Piranga rubra*). Twenty-seven species of wading birds, shorebirds, and waterfowl were documented. Many migrating birds were observed, typically only once or twice during the year.

Other wildlife is known to use the area, including nonnative American bullfrog (*Lithobates catesbianus*), but no native frogs. Native Sonoran mud turtle (*Kinosternon sonoriense*) and nonnative spiny softshell turtle (*Apalone spinifera*) and most native reptiles appropriate for riparian habitats are present. Some mammals, both native (coyote, woodrat, cotton rat, bats) and nonnative (domestic and feral dogs and cats and cattle) were seen. Native fish (Gila topminnow) were found for the first time in decades in 2017 (Sonoran Institute 2017), but several nonnatives have been long been present, including mosquitofish (*Gambusia affinis*). Presence of bullfrogs and mosquitofish probably precludes or at least complicates potential restoration of populations of natives.

Figure 20—Riparian habitat: effluent-dominated reach of Santa Cruz River in Tucson, historically dry, currently wet. Note narrow width of riparian vegetation and soil cement bank. Photo by author.

Figure 21—Riparian habitat and multi-use trail, along the historically wet, currently dry reach of the Lower Santa Cruz River in Tucson, which is now part of a linear park with multi-use trail. Photo by Amy Gaiennie.

The existing conditions of the Lower Santa Cruz River provide a valuable resource for the human community. A paved multi-use trail runs through developed parkland along both sides of the river for much of its course through urban Tucson, and it is very heavily used by pedestrians and bicyclists (figs. 20 and 21). The larger trees, especially athel tamarisk and Fremont's cottonwood, are frequently used as campsites by homeless people. As mentioned, bird watchers find the area of interest. Long-term residents understand and appreciate the changes that have occurred, but many young people and visitors may think the current conditions are normal and natural and appreciate their protection as parkland.

Example 2: An Artificial, Intentional Riparian Site—Sweetwater Wetlands

An active domain of wetland creation is constructed wetlands for processing of sewage wastewater (Kadlac and Wallace 2009). An example is Sweetwater Wetlands, Tucson, Arizona, which was created to process water that had been used to backwash filters at the Roger Road Wastewater Treatment Plant and put that water into the reclaimed water system. Considerable planning went into the effort to create an aesthetically pleasing wetland that would allow plants to process the water through a process of filtration, settling, and biological activity. Early on in the planning process, an expert on mosquito control was hired to help design a wetland that would not be plagued by mosquitoes. Unfortunately, mosquito control recommendations were scrapped in the later stages of planning because effective mosquito control was not consistent with effective water processing, the hoped-for wildlife and aesthetic values, and the project budget.

Sweetwater Wetlands was created, planted with native wetland plants including Fremont cottonwood and cattails (*Typha latifolia*) and put into operation. It included ponds with islands for nesting birds (fig. 22), dense cattail stands for various stages of water

Figure 22—One of the ponds at Sweetwater Wetlands, February 2014. Originally used to process filter backflush water as part of an older sewage treatment system, this and adjacent ponds are currently filled with reclaimed water to provide wildlife habitat and a recreational amenity for the community. Photo by author.

processing, a complex of several ponds surrounded by a walkway and piers for wildlife viewing, interpretive signs, restrooms, a public parking lot, and a system of aquifer recharge ponds. Educational programs were developed and thousands of schoolchildren have learned about wastewater processing, wetlands, and ecology on field trips. Informative brochures and an "Activity Book and Field Guide" have been produced. Processed water from the Treatment Plant flowed out into the riverbed close to this location, and the perennial lower Santa Cruz River began here. With some adjustments, the desired water processing, wildlife, and aesthetic values were achieved, and the area has become an outstanding popular recreation area for birders and a valuable educational resource.

Over 300 species of birds have been documented at the site (City of Tucson 2016). However, a mosquito problem also developed and has required continuous expensive effort (Karpiscak and others 2004; Willott 2004). Until recently, water quality could not support fish and may have been a population sink for migrating aquatic insects. Water quality improvements since 2014 have led to establishment of a dense population of mosquitofish and many species of aquatic insects that have probably greatly reduced the mosquito population. The sewage processing method changed in 2014, and Sweetwater Wetlands was no longer needed for its original function. Today, if the wetlands were allowed to dry and be removed, there would probably be a significant public outcry over the loss of recreational and educational benefits. Maintaining the wetlands as a park has become separate from the reason for their original creation because water treatment technology has changed. Continuous active maintenance and management are necessary to prune trees, keep pathways and structures in good condition, thin cattails, and prevent mosquitoes.

Urban Runoff—A Source of Water

In the urban environment, stormwater is often directed through a series of drains, sewers, canals, and culverts to run off eventually to a designated depository where it can enter its natural pathway or be used for aquifer recharge or other uses. Some of the structures created to handle urban stormwater can become important wildlife habitat if they are allowed or created to become so. Retention basins are designed to manage stormwater runoff to prevent flooding and downstream erosion and to improve water quality downstream. Retention basins are, at best, artificial lakes with vegetation around the perimeter, designed to include a permanent pool of water. Detention basins are designed to store water temporarily after a storm, but eventually to empty at a controlled rate to a downstream water body. Infiltration or recharge basins are designed to direct stormwater (or other water) to groundwater through permeable soils.

Vegetation may be intentionally planted or volunteer around the margins of the ponds or appear as emergent or aquatic plant communities. These utilitarian stormwater drainage features can be managed to improve the neighborhood and may be seen as valuable amenities, or they may be mismanaged to become major nuisances as mosquito breeding sites or other pest habitats.

The U.S. Fish and Wildlife Service, in cooperation with the Arizona Game and Fish Department and local government agencies, has developed a Safe Harbor Agreement for two species of endangered fish, Gila topminnow and desert pupfish (*Cyprinodon macularius*) that permits placing these fish in retention basins and other artificial bodies of water (Duncan and Voeltz 2005). As of early 2016, this agreement has not been used for placing fish in urban retention and detention basins because of the concern with fish moving downstream to another property (D. Duncan, U.S. Fish and Wildlife Service, personal communication). Fish have been utilized to control mosquitoes and other nuisance insects in artificial ponds and have been successfully established under this agreement in a few other artificial habitats. Also, their use in compliance with the terms of the Safe Harbor Agreement has expanded the range of remaining populations of these endangered fish and greatly increased the number of individuals of each species, which benefits the species especially if anything impacts the few remaining natural habitats.

Example: Kino Environmental Restoration Project (KERP)

An example of a detention basin developed for maximizing habitat values and minimizing nuisances is the Kino Environmental Restoration Project in Tucson, Arizona (Pima County 2015). Urban stormwater runoff from approximately 18 square miles around the Davis Monthan Air Force Base and the City of Tucson runs through a series of drainages, mostly concrete-lined box culverts or ditches, to collect in the Tucson (Ajo Road) Detention Basin. Prior to the inception of this project, the basin of approximately 50 acres was surrounded by concrete with a mostly mud substrate. Often the basin was dry, or filled with stagnant water and debris, and provided excellent mosquito larval habitat and dense weeds. In the mid-1990s, the U.S. Army Corps of Engineers, Pima County, and the Pima County Flood Control District collaborated in a cooperative agreement to create the Ed Pastor Kino Environmental Restoration Project (KERP). The goal was to create a facility that would meet three primary purposes: create native ecosystems, harvest urban stormwater, and control flooding. Elimination of mosquitoes was also an important consideration, as was elimination of trash dumping.

The project today is managed by the Kino Sports Complex/Stadium District in cooperation with Pima County departments, such as the Health Department, Regional Flood Control District, Department of Transportation, and the Regional Wastewater Reclamation Department, in addition to a number of State agencies to ensure KERP meets the State and Federal guidelines set forth for an environmental habitat. The final footprint of the KERP covers 141 acres and contains 28 acres of riparian and open water including: a 5.6-acre, 50-foot deep pond; 21 acres of grassland, mesquite bosque, marsh and upland vegetation; three short stream courses; and 92 acres that include flood control structures, a basin earthen berm, and a 2.2-mile recreational path that surrounds the basin (figs. 23 and 24).

Native plants associated with wetlands and riparian areas, as well as upland habitats, have been planted and are irrigated by a combination of floodwaters gathered by the basins and reclaimed sewage effluent. Some invasive plant species have come into the area but are managed. A pumping and valve system circulates and mixes reclaimed and stormwater within the basin. Stormwater coming into the project first passes a debris basin equipped with a trash rack to retain sediment and trap floating objects. It then passes into lined storage ponds that retain the runoff. Two small weirs and a weir gate regulate the flow of water into the deep pond. Water is circulated along the three stream courses that drain into the ponds, and cooler water is pumped from the bottom of the 50-foot pond to the top of the stream courses to improve water quality and aid with vector control. Water levels fluctuate depending on the availability of stormwater, and much of the area is designed to hold large volumes of rapidly arriving water during wet periods and survive as upland in dry periods. During periods of extended dryness, treated effluent (aka

Figure 23—Riparian habitat: Kino Environmental Restoration Project, deep pond and vegetation, February 2014. The pond was created primarily a receptacle for stormwater runoff from the upstream urban area, and is managed, in part, to provide wildlife habitat and as a recreational amenity. Its design and management allow for rapid filling following storm events and minimization of mosquito habitat. Photo by author.

Figure 24—Riparian habitat: Kino Environmental Restoration Area, shallow pond and vegetation, February 2014. This pond is off the stormwater channel, so is not subject to rapid inundation, but is filled by water intentionally diverted from the large pond for the purposes of aquifer recharge, water storage, and wildlife habitat. Photo by Amy Gaiennie.

reclaimed water) is purchased from Tucson Water to maintain pond levels and protect habitats. A 16-inch reclaimed water pipeline feeds water into the pond. In addition, the open water areas and marsh can be fed by three additional pipelines at KERP.

Harvested stormwater, when available, can be a low-cost alternative to purchasing and using groundwater and reclaimed water. This project has yielded approximately 19 million gallons per year of usable irrigation water (10-year average, range 0 to 39 million gallons). The water has been used to irrigate the basin's vegetation, the adjacent Kino Sports Complex ballpark and athletic fields, road median landscaping and easements, and landscaping for nearby public buildings. The project has become a popular recreational site and birding area, although access to the most interesting habitats is limited, and most viewing must be done from a distance. A casual visitor may see an assortment of birds representative of the variety of ecological communities present in the project, including migratory waterfowl, shorebirds, and wading birds. Some informative signs have been posted, and most have survived vandalism. Discussions are ongoing between agencies regarding placing native fish under the Safe Harbor Agreement.

Areas With Water Supplied by Agricultural Irrigation

Agriculture has historically been the largest destroyer of riparian habitat, directly by conversion of riparian forests and woodlands to croplands, and indirectly by damming streams and rivers to create reservoirs, reduce flooding, and manage water availability. In arid lands, riparian areas are clearly the most suitable for agriculture because the land is relatively level, close to water, and has suitable soils. In Arizona and much of the arid Southwest, the primary crops have been cotton, alfalfa, vegetables, or grass. These have replaced native woodlands with monoculture shrublands, fields of forbs, or grassland. Tree crops, primarily citrus and pecans, have created nonnative woodlands. Whereas a great deal of research has gone into studying how to grow these crops, very little has looked at the ecological effects of the conversion of native riparian areas to croplands and the novel communities of wildlife and native plants that develop.

Ohmart and others (1985) found that agriculture created new habitats for wading, water, and shorebirds, and that the type of agricultural development and crops grown were important in determining species composition and population sizes. Conine and others (1978) found that some riparian bird species used agricultural land as foraging habitat, whereas others did not. Wells and others (1979) examined avian use of citrus orchards and found that citrus was an important habitat for doves but generally lacked flycatchers, small insectivores, and woodpeckers, which were abundant in native riparian habitat.

Obviously, birds that require trees do not do well in cotton fields and many other animals once common in native riparian communities cannot survive the conversion to agriculture. Other creatures, however, do manage to adapt in various ways, perhaps losing nest or den sites but gaining feeding areas. If nearby sites can provide the missing components (e.g., nest sites near grain fields, ditch banks, and culverts near alfalfa fields), some animals may survive and even thrive. It appears unlikely, if not impossible, that intact communities with all previously present species in healthy numbers can survive, but some agricultural crops can provide suitable habitats for many of the species native to our riparian communities.

Example: Pecan Orchards as Riparian Habitats

Kingsley (1989, 1985a,b) studied wildlife uses of a large irrigated pecan orchard along the Santa Cruz River south of Tucson (fig. 25). At the time, this was said to be the world's largest irrigated pecan orchard and consisted of approximately 2,200 hectares with tree densities varying from 59 to 119 trees per hectare. Fields were flood irrigated approximately every 2 weeks during the growing season (April-October typically). Trees were even-aged, tall, with the understory cleared of most vegetation. Stumps of removed trees sometimes sprouted and created a shrub story and a mixture of forbs and grasses were the understory. Dense tall weeds, including several species of seed-providing forbs and grasses, occurred in places along irrigation ditches and tailwaters. Slash piles consisting of pruned branches surrounded many fields and provided a structural equivalent of a dense shrub story that was utilized by some wildlife species.

Insects were abundant, most of them considered beneficial or neutral, although aphids that feed on pecans and mosquitoes that bite people were present, and outbreak populations had caused problems (Kingsley 1985a). Many aquatic insect species migrated into the flooded fields and irrigation ditches but probably could not survive and reproduce successfully since these were temporary waters. Amphibians were present and abundant, but the only species documented as successfully reproducing was Couch's spadefoot (*Scaphiopus couchi*), which can develop in transitory water lasting less than 2 weeks. Reptiles were relatively rare, probably because they cannot survive and reproduce under conditions of frequent inundation, absence of topographic diversity, or lack of some microhabitat components (Jones and Glinski 1985).

Kingsley (1989) listed 83 species of birds found in the orchard with populations ranging from rare and transitory to greater than those found in native riparian woodlands. Flycatchers and small insectivores were present, but woodpeckers were absent as nesting birds although frequently seen foraging in the orchard. Birds characteristically found in dense riparian thickets were not found. No dense midstory was present. Hole-nesting birds were not found breeding, but several were observed foraging. Tree holes were not allowed to develop.

Predatory mammals such as coyote (*Canis latrans*) and raccoon (*Procyon lotor*) and bats of several species were abundant as foragers, but burrowing mammals were generally absent, probably because they could not survive frequent inundation. No survey of mammals was conducted, so the number of small mammals is unknown. Bats foraging in pecan orchards in Texas were studied by Braun de Torrez and others (2011) and Braun de Torrez and Kunz (2009), who found bats present in pecan orchards, especially organic orchards, at higher numbers and diversity than in native juniper-mesquite woodlands and the adjacent semiarid landscape.

The orchard has been a popular place for local birders, especially those on their way to Madera Canyon, an especially popular area that is reached by a road passing through the orchard. The orchard and processing plant supported approximately 250 workers and was one of the largest industries in the area at the time. Much of the historic orchard has been converted to residential development as the communities of Sahuarita and Green Valley have grown, and the long-term plan is to phase out the orchard except in floodplain-zoned sites where agriculture may continue indefinitely.

Figure 25—Flooded pecan orchard, Green Valley Pecans, Arizona. Photo by author.

Figure 26—Century-old irrigation canal, still in use, with cottonwoods. Originally put in to irrigate an agricultural field, the canal is now part of Zion National Park. Cottonwoods apparently invaded after the canal was created and are now well established. Photo by author.

Example: Unlined Irrigation Canals as Riparian Habitat

That irrigation canals support a new riparian resource has long been considered (fig. 26). Breninger (1898 p. 128), referring to the cactus ferruginous pygmy-owl, stated: "And since trees planted by man have become large enough to afford nesting sites for woodpeckers, this Owl has gradually worked its way from the natural growth of timber bordering the rivers to that bordering the banks of irrigating canals until now it can be found in places ten miles from the rivers."

Recently, Sueltenfuss and others (2013) examined such wetlands in northern Colorado using quantitative and qualitative measures. The wetlands they examined were marsh, wet meadows, and salt flats. Eighty-nine percent of the canals examined were unlined earth. They found wetlands associated with irrigation canal leakage, pond and reservoir dam seepage, wetlands located on the fringes of reservoirs, tailwaters, ephemeral channels adjacent to canals, and a few intentionally created wetlands. They found large fluctuations in water level in wetlands to be associated with the amount of water flowing in canals, sometimes with delays of up to 50 days, and not associated with precipitation events. They found one instance in which irrigation of an adjacent field was associated with water table levels in a wetland.

The same types of wetlands have been documented in southern Arizona (K.J. Kingsley, personal observations, and Kingsley 1985a), and some have cottonwood trees or other trees and shrubs growing along and watered by them.

Johnson (1972) presented a review of changes occurring along irrigation ditches in the Phoenix area, where ditches were being converted to concrete-lined conduits or underground pipes that would not support vegetation. Water conservation by lining or burying conduits has resulted in loss of these riparian resources and is likely to continue and expand as demand for water increases and availability decreases. Many of the irrigated farmlands subsequently are converted to urban and suburban communities that may be very different ecosystems.

Example: Tailwaters

Tailwaters are wetlands created by excess irrigation and are wet areas at the low points of irrigated fields or immediately downstream from the fields. Along the Gila River in southern Arizona, some tailwater wetlands have dense thickets of tamarisk (*Tamarix* spp.) that are known to be used by nesting southwestern willow flycatchers (*Empidonax trailii extimus*) (K.J. Kingsley, personal observations). Tailwaters appear to be better habitat for this species than areas along the nearby river. Water moves slowly in these areas and provides abundant habitat for mosquitoes, which are fed upon by this endangered species, whereas the river moves too quickly to provide good mosquito habitat. In addition, tailwaters are fairly stable environments, compared to the unstable river flows, and are not as susceptible to flooding with concomitant loss of vegetation structure. Water is provided even during the worst droughts when water might not be available in the river. Several other species of riparian and wetland birds may also be observed in this environment, but the typical tailwater is an unstable habitat with a dense thicket of weeds or weed trees that is kept in an early stage of succession and affords little habitat for species that prefer large blocks of habitat or more open spacing.

Example: Abandoned Fields Persisting as Riparian Habitats

Agriculture is not always successful or permanent, and when it fails, it leaves behind land that will transform from the regular rows of monoculture crops to a hodge-podge of early succession if left to itself. More than 96,356,000 ha of farmland were abandoned in the United States in the 20th century (U.S. Department of Agriculture 1998 cited in Richter and Stutz 2002). Especially in riparian areas, however, abandoned agricultural fields can be returned to some semblance of the former natural community. Sometimes, if the area is subject to a more-or-less natural flow regime and flood timing is right, and especially if the water table has not dropped to a depth inaccessible to roots, it can grow back to cottonwood-willow forest or woodland (e.g., several historic fields along the Verde River in the Cottonwood-Camp Verde area, now included in the Verde River Greenway State Natural Area, K.J. Kingsley, personal observations). Some efforts have been made to encourage or restore riparian communities to abandoned farmland.

Of some concern in abandoned agricultural lands is contamination by chemicals, including undocumented historic chemical dumps. Other concerns are reduced water input because of increased efficiency by active agriculture, unnatural flow regime, long-term unpredictability, and possible attraction or harboring of agricultural pests. To my knowledge, none of these has been documented.

Conclusion

Little scientific attention has been directed toward the study of anthropogenic riparian areas, especially those that have been inadvertently created. Many more examples could have been selected from a wide range of sites. The number, diversity, and quality of anthropogenic riparian areas are likely to change over time as human demand for water resources increases with increasing population and aspirations for amenities. These environments warrant further study and some measures of protective management to enhance their suitability as wildlife habitat and recreational resources.

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Chapter 7. The Development of Riparian Ecosystem Restoration in California

John Stanley, F. Thomas Griggs, and John Rieger

Introduction

The evolution of our modern-day understanding of riparian ecology and the development of the field of riparian ecosystem/habitat restoration underwent significant advances from 1970 to 2000. The four parts of this chapter include: (1) a chronological overview of the major activities, events and publications that documented, and punctuated, the progress of our understanding of riparian ecology and riparian ecosystem restoration in California; (2) a discussion of the various types of riparian restoration projects with the projects categorized by the primary reason they were undertaken; (3) examples of the various types of research, experimentation, field investigations, and monitoring programs associated with the early riparian habitat restoration projects; and (4) concluding remarks.

The Society for Ecological Restoration (SER), incorporated in 1988, defines "ecological restoration" as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER 2004). We use the term "restoration" in this chapter in a broad sense. We use the same terminology as used by the early "restorationists" to describe their projects even though many of the early projects do not meet the exact intent of SER's definition. For example, many of the early projects were described as rehabilitation, enhancement, creation/fabrication, reclamation, ecological engineering, etc., rather than "ecological restoration" and/or "habitat restoration."

In the broadest sense, restoration activities have been occurring within California's rivers, creeks, and riparian ecosystems for centuries. The cultural stewardship practices of California Indians directed at promoting ecological services also promoted continuance of the ecosystems (rivers, creeks, riparian areas) upon which they relied. Anderson (2005) documents a number of traditional cultural practices and horticultural techniques (traditional resource management) used by California Indians within the riparian corridor. For example, Stevens (2003) studied the relationship between California Indian groups that tended white root—an herbaceous perennial understory plant in valley oak riparian woodlands with long roots used for basket weaving—and how they affected the distribution and ecology of the plant. Unfortunately, the traditional ecological knowledge of California's Indian Tribes was seldom, if ever, incorporated into riparian area restoration projects implemented in the 20th century[1].

We recognize that prior generations of landowners performed remedial work on their properties (e.g. stabilization of eroding streambanks using woody live and dead plant material) within the riparian corridor. In addition, workers employed under the Works Progress Administration between 1935 and 1942 and Civilian Conservation Corps between 1933 and 1940 performed various tasks along rivers and streams in California (Riley 1998).

[1] An example of how this might occur in the 21st century is the partnership developed in 1997 between the U.S. Forest Service Lake Tahoe Basin Management Unit (LTBMU) and the Washoe Tribe of Nevada and California to conduct the Meeks Meadow Washoe Restoration Project to restore the ecological and cultural function of the Meeks Meadow.

The field of riparian ecosystem/habitat restoration did not evolve solely on its own, but rather, in combination with the development of the fields of: (1) stream restoration; (2) erosion control; 3) (water quality/nonpoint source pollution control; (4) biotechnical/ soil bioengineering; (5) watershed management; (6) landscape ecology; and (7) greenway design and management. Practitioners attempting to restore riparian areas drew from the experience of professionals involved in these related activities and interacted with many of these professionals during the course of restoration project planning and design.

There is an extensive body of gray literature associated with the field of riparian restoration: project proposals, conceptual plans and detailed planning documents, project plans and specifications, as-built plans, monitoring programs, and monitoring reports. We avoided referencing these documents since they are difficult to locate. In almost all instances the literature cited herein is from books, conference proceedings, and other available documents.

Over the years, a number of conferences were convened in California to address issues related to the conservation and restoration of riparian ecosystems. Conferences and symposia pertaining to riparian ecosystems, riparian ecology, riparian habitat restoration, and riparian area conservation throughout the western United States including California are presented in Volume 1 Appendix B.

Part 1—Publications, Activities, and Events: 1960s to 2000s

The 1960s

Smith (1977) stated: "Prior to 1960, few people showed any concern for the demise of California's Riparian Forest communities." Warner and Hendrix (1984) asserted that "descriptions of historical extent and character of the Sacramento Valley's riparian systems by Kenneth Thompson (1961) were among the first writings to demonstrate their importance."

The California Department of Water Resources (CWDR) recounted, "During 1961, a flood of protests against the denuding of levees in the Sacramento-San Joaquin Delta was launched by sportsman's organizations, wildlife conservation groups, and the public in general."(At that time, "Levee maintenance regulations dictated that virtually all shrubs and trees be cleared from levees to insure (sic) that the flood control protection provided by the levees would not be impaired" (CDWR 1967).

The CDWR (1966) reported: "In 1961, the California State Legislature authorized the Sacramento River and Delta Recreation Study. One of the recommendations was that 'a program of pilot studies on selected reaches of levee be initiated to test various types of vegetation, determine control measures necessary, study methods for these controls, and determine the costs of this type of maintenance'. In response to that legislative recommendation, the Department of Water Resources initiated the Pilot Levee Maintenance Study in 1962."

Thus, "The Pilot Levee Maintenance Study was begun to conceive and test alternative methods of levee maintenance that would provide for multiple use of levees. A number of tests[2] were conducted to determine if vegetative growth could be allowed on levees and

[2] Refer to part 2 below (Riparian Restoration Projects–Bank Stabilization/Erosion Control Projects) for a brief discussion of the types of tests conducted and the types of plant materials used in these tests.

berms to preserve and enhance esthetic, recreational and wildlife habitat values without impairing the flood control capacity of the levees" (CDWR 1967).

Two annual progress reports were prepared in 1963 and 1964 and a Preview to Bulletin 167 was published in 1966 (CDWR 1966). Findings during the last year of study and the conclusions and final recommendations of the Pilot Levee Maintenance Study were presented in CDWR Bulletin 167 (CDWR 1967). The study "concluded that with proper vegetative management programs, certain delta levees can be adapted and maintained to serve the needs of esthetics, recreation and wildlife as well as the primary purpose of flood control."

One of the earliest acknowledgments of the ecological values and severely limited extent of riparian systems appeared in Volume III of the 1965 California Fish and Wildlife Plan (CDFG 1965 per Warner and Hendrix 1984. Smith (1977) cited this document when he stated that "of the 29 habitat types listed in the 'Inventory of Wildlife Resources, California Fish and Wildlife Plan' (Vol. III), riparian habitat provides living conditions for a greater variety of wildlife than any other habitat type found in California."

The 1970s

The 1970s were the period of heightened awareness of riparian habitat loss and the importance of riparian habitat for many of the wildlife species in the State of California.

In *An Island Called California,* Elna Bakker pointed out that "no natural landscape of California has been so altered by man as its bottomlands. The grass-rich stretches of the great Central Valley are, for the most part, lost to orchards and vineyards, cotton and alfalfa fields. Many miles of curving green ribbon along its watercourses have been eradicated, replaced by the sterile concrete of flood control and navigation channels. Most of the tule marshes of the Delta country are now neatly diked rice paddies. On the freeway between San Francisco Bay and Sacramento one forgets that this was once wild land with golden beavers going about their industrious ways and great blue herons on guard with that watchful immobility so peculiarly their own. To recreate this world of slough, bank, and riverway takes more than the simple listing of what can be recalled, or guessed, was there. It needs imagination coupled with a persistent searching for the last few remnants of the original river country" (Bakker 1971).

Concerns arose over the loss of riparian habitat and the degradation of remaining riparian areas due to land clearance for orchards and field crops, logging for wood products, grazing by livestock, streambank stabilization, channelization and other flood control activities, and altered water flow due to dams and irrigation throughout California (especially California's Central Valley). Consequently, scientists, conservationists, and agency personnel came together to share their observations, express their concerns, and seek solutions in the early and mid-1970s.

Warner and Hendrix (1984) stated: "In the early 1970s, studies by the Nongame Wildlife Investigations Section of the Department of Fish and Game (e.g., Gaines 1974) began to bring into focus the impact of riparian system loss to the State's wildlife populations. At about the same time, disturbing figures of riparian vegetation loss along the upper Sacramento River were reported (e.g., McGill 1975)." Burns (1979) reported, "Based on historical accounts, there were nearly 775,000 acres of riparian forests along the Sacramento River and its tributary streams in 1850." Others (Katibah 1984; Roberts et al. 1977) cite a figure of 800,000 acres of riparian forest remaining after 1848. By 1952, only about 20,000 acres of riparian forest remained (Smith 1978). During the 20-year period between 1952 and 1972, 53 percent of the mature riparian forests that existed in 1952 had been removed and the land converted to agricultural uses (Burns 1979). By 1972, only about 12,000 acres of riparian forest remained in the Sacramento Valley (Gaines 1976; Roberts et al. 1977). Those interested in greater detail on the decline of California's Central Valley riparian forests should refer to Katibah (1984).

In the fall of 1975, the California Secretary of Resources established the Upper Sacramento River Task Force "to solve the acute resource problems centered primarily along the 170-mile section of the river below Shasta Dam" (Burns 1979). The initial task force was made up of State agencies[3] within the Resources Agency. Soon after its formation, the task force was enlarged to include Federal and local agencies and later enlarged again to include special interest groups. The objectives of the task force were "to coordinate intergovernmental activities and to take actions to ensure the protection of the fish, wildlife, recreation, and aesthetic values of the river while considering the other beneficial used of the river and adjacent lands, for such uses as water conveyance and agriculture," (Burns 1979). Encouraged by Napa County's 1974 adoption of an ordinance allowing for the protection of riparian vegetation along water courses (Burns 1977; Dunlap 1977; Gaines 1976), the task force "drafted a model, county general plan element, and ordinance that would bring the removal of riparian vegetation under a permit process," (Burns 1979). The task force gave these models to the boards of supervisors of five counties along the Sacramento River and asked them to adopt similar regulations. Only the northernmost county adopted a riparian protection ordinance (Burns 1979).

The first riparian forest conference in California titled "Conference on the Riparian Forests of the Sacramento Valley" was held in Chico in May of 1976. Cosponsored by the Davis and Altacal Audubon Societies, the conference was organized by David Gaines. There were about 70 participants (Abell 1989). There were no published conference proceedings; however, David Gaines prepared abstracts of the presentations (Gaines 1976). During this one-day conference, speakers gave a historical perspective on Sacramento Valley's riparian forests, described riparian vegetation, and discussed the animals that rely on riparian habitat. Staff from the CDWR described land use changes in the Sacramento River riparian zone and discussed the activities of the Sacramento River Task Force. A representative from the Army Corps of Engineers described Corps project work for flood control and bank protection on the Sacramento River. The President of the Sacramento Valley Landowner's Association presented a landowner's perspective of the bank erosion and loss of agricultural land due to the management of the river for water transport. CDWR staff described the recreational values of the Sacramento River and State Department of Parks and Recreation staff addressed boating on the river. A land agent with the California Wildlife Conservation Board discussed riparian forest acquisition along the Sacramento River, a supervisor from Tehama County discussed the county's approach to riparian forest habitat, and a U.S. Fish and Wildlife Service field supervisor addressed the need to mitigate habitat loss due to bank protection. Summing up, an attorney encouraged everyone to fight for their vision of the future of the Sacramento River and its riparian forests.

[3] Department of Fish and Game, Department of Water Resources, Department of Parks and Recreation, Department of Navigation and Ocean Development, Wildlife Conservation Board, Reclamation Board, Water Resources Control Board, and State Lands Division.

The second riparian forest conference in California titled "Riparian Forests in California: Their Ecology and Conservation" (Sands 1977) was held at the Davis Campus of the University of California in May 1977. Cosponsored by the Institute of Ecology, University of California, Davis, and the Davis Audubon Society the "symposium" was coordinated by Anne Sands. This was the first riparian conference for the United States with published proceedings. There were approximately 128 people in attendance (Abell 1989). Symposium topics were divided into two sessions; the morning session dealt with historical and ecological subjects followed by an afternoon session dealing with management and preservation. Presentations in the first part of the morning addressed the historical extent of riparian habitat loss (Roberts et al. 1977; Smith 1977), the diverse flora and fauna dependent on riparian areas (Roberts et al. 1977), accounts of the historical condition of Sacramento Valley riparian lands and remnants (Thompson 1977), and the concepts necessary for understanding the fluvial system (Keller 1977).

Following the morning break, experts went into greater detail describing the vegetation/flora of the Sacramento Valley (Conard et al. 1977), the importance of valley riparian forests to bird populations (Gaines 1977), and the habitats of native fishes in the Sacramento River Basin (Alley et al. 1977). Afternoon speakers addressed the need for legislation to study and protect California's riparian forests (Dunlap 1977), the activities of the U.S. Army Corps of Engineers (CE) protecting the Sacramento Valley levee system with rock bank protection (Kindel 1977), and the planting of trees and shrubs at selected sites along the Sacramento River (Kindel 1977). Speakers also described the progress made by the recently established Upper Sacramento River Task Force in coordinating the activities of the many governmental agencies that have jurisdiction over developments along the Sacramento River so as to ensure the protection of the fish and wildlife and recreational aspects of the river (Burns 1977). The symposium closed with a brief presentation on the value of riparian forests in today's society (Frost 1977) followed by a panel discussion.

It is noteworthy that no riparian restoration projects were mentioned, nor were the words "restoration" or "revegetation" used in any of the papers presented at the 1977 Symposium. The only mention of planting is by the Corps of Engineers—planting of trees and shrubs in 1967 along 3 miles of riverbank where the levee had been set back and a new berm had been created and protected by rock.

Anne Sands and Greg Howe presented a paper titled "An Overview of Riparian Forests in California: Their Ecology and Conservation" (Sands and Howe 1977) at the Symposium on the Importance, Preservation and Management of Riparian Habitat (Johnson and Jones 1977) held in Tucson, Arizona, in July 1977. This paper was comprised of abstracts from Sands (1977).

In 1978, Anne Sands (1979) presented a paper titled "Public Involvement in Riparian Habitat Protection: A California Case History" at a floodplain wetlands and riparian ecosystem symposium in Georgia. Anne Sands addressed the restoration[4] of damaged riparian areas as "probably the most difficult protection strategy, but certainly it is one of the most rewarding" (Sands 1979). In this paper, Anne Sands (1979) provided an example of how public concern over riparian habitat loss due to flood control project works had resulted in the State Reclamation Board commissioning a civil engineering study (Murray

[4] This is possibly the first mention of "riparian restoration" in the literature.

et al. 1978) on the retention of riparian vegetation as a means for controlling bank erosion along the upper Sacramento River.

Warner and Hendrix (1984) reported: "In 1978, a coalition of conservation organizations, led by the Riverlands Council chaired by Anne Sands, sponsored State legislation to protect riparian resources. The resulting legislation, AB 3147 (Fazio), appropriated $150,000 to the Department of Fish and Game (CDFG) for a study of riparian resources of California's Central Valley and desert." The stated goal of the CDFG in mounting its California Riparian Study Program (CRSP) was to "protect, improve, and restore the riparian resources of the State" (Warner 1984). There were multiple elements to the CRSP including: development of background information on California's riparian resources; mapping of riparian vegetation in the Sacramento and San Joaquin valleys; determination of the areal and linear extent of Central Valley riparian vegetation; conducting a remote-sensing survey and a ground inventory of riparian vegetation in the Central Valley; field surveys of California's desert riparian systems; reporting on State, Federal, and local programs affecting riparian systems; analysis of riparian conservation needs; and the development of a riparian conservation program.

The 1980s

In 1981, the California Reclamation Board, a governor-appointed body with statutory responsibility for maintaining Central Valley floodways, adopted a Riparian Vegetation Management Policy (King 1985). This policy recognized the benefit of riparian vegetation in maintaining the integrity of floodways and established a permit system for its removal. The Reclamation Board initiated a program, managed by the CDWR to ensure the retention of riparian vegetation at selected sites along the Sacramento River. Beginning in 1981, CDWR began installing native trees and shrubs in rock reinforced levees downstream of Sacramento (King 1985).

The third riparian forest conference in California titled "California Riparian Systems[12]: Ecology, Conservation and Productive Management" (Warner and Hendrix 1984) was held at U.C. Davis in September 1981. This conference drew 711 participants (Abell 1989). The fact that approximately 150 technical papers were presented at the conference of which 128 were included in the 1,035 pages of conference proceedings is evidence of the exponential growth in the fields of riparian ecology, conservation, management, and restoration. Although only seven papers were presented under the category of riparian restoration, many of the other papers reported on critical background research pertaining to the structure and function of riparian systems, especially hydrologic and hydraulic considerations. Also at this conference, Richard Warner (1981) reported on the structural, floristic, and condition inventory of Central Valley riparian systems conducted as part of the CRSP mentioned above (Warner 1984).

In the early 1980s, Randal L. Gray and Ron Schultze of the USDA Soil Conservation Service (now the Natural Resources Conservation Service) in Davis, California, organized the Riparian Revegetation Study Group (RRSG) to bring together individuals working on, or interested in, the reestablishment of riparian vegetation along degraded streams in California. Beginning as an interagency group, RRSG quickly expanded to

[5] This was the first conference convened at the University of California, Davis, under the "California Riparian Systems" title (Abell 1989).

include Federal, State, and local agency personnel, staff of non-profit organizations, and consultants. Semiannual meetings were hosted by members in San Jose, Hayward, Marin County, and Sacramento. The agenda for most meetings included short presentations of restoration projects and discussions of difficult issues associated with the restoration of riparian areas (for example, vandalism). Some meetings included field tours of recently completed revegetation projects. A subgroup of RRSG was organized in southern California because travel time and distance to meetings held in the northern part of the State precluded the involvement of most southern Californians. RRSG became the Riparian Guild of the California Society for Ecological Restoration (SERCAL) after SERCAL's formation in 1991.

Several of the papers presented at the Native Plant Revegetation Symposium (Rieger and Steele 1985) held in San Diego in November 1984 dealt with the restoration of riparian areas (Barry 1985; Capelli 1985; King 1985; Riley and Sands 1985). Two hundred and two people were in attendance. This meeting was the first that addressed restoration as the theme of the symposium. It was an effort to survey and promote communication between restoration practitioners in the State.

In the late 1970s and early 1980s, it became apparent that the protection and restoration of riparian areas was in direct conflict with traditional floodplain management (Riley and Sands 1985). Conservationists encouraged Federal and State flood management agencies and local flood control districts to find ways in which the retention and/or planting of riparian vegetation within channels and on floodplains could be compatible with flood management. At the 1984 "Native Plant Revegetation Symposium," Ann Riley and Anne Sands (1985) suggested that flood control planners and engineers should adopt the concept that "restoring streams is environmentally less damaging and less expensive than traditional channelization," an approach that was being promoted by Nelson Nunnally and Ed Keller (1979). Nunnally and Keller's central theme was "to preserve natural stream morphology wherever possible, and when change is essential, to design channels with morphological characteristics similar to those of natural channels." Riley and Sands (1985) argued for restoring the "natural balance between the river, its floodplain, and the riparian forest" and that to restore and maintain the riparian forest/fluvial system balance, we must recognize that floodplains must be allowed to function as they were intended. Riley and Sands (1985) also touted the combined use of vegetation and structural means for bank stabilization as presented in Gray and Leiser (1982) and Scheichtl (1980). Williams and Swanson (1989) proposed that flood management agencies should take a new approach to planning channel modifications for flood damage reduction. They pointed out that there were significant problems with conventional flood control project design including: (1) underestimation of the roughness of lined channels; (2) failure to account for channel bed erosion and deposition; (3) failure to account for debris; and (4) underestimation of maintenance requirements. Williams and Swanson argued that flood channel design should be multi-objective and should incorporate proper consideration of hydrologic, geomorphic, and biological factors that influence stream hydraulics. They pointed to the Wildcat Creek Flood Control Project as an opportunity to use the integrated design approach they discussed in their paper.

During the early- to mid-80s, some flood control agencies sponsored the preparation of design-it-yourself manuals so that flood control planners, engineers, and landscape architects could make decisions on the species to be planted along reconfigured channels and their appropriate planting locations without the input of biologists on each project site. In 1983, Harvey and Stanley Associates, Inc., prepared a “Revegetation Manual for the Alameda County Flood Control and Water Conservation District Revegetation Program” (Stanley and Stiles 1983). The manual contained detailed information on the characteristics of 97 species of riparian plants including their appearance, ecological relationships, wildlife habitat value, planting location, planting options, plant requirements, and maintenance requirements. A map and table allowed project planners to select plant species suitable to each of the planting zones within Alameda County. This was to be used in conjunction with a representative cross section of a typical channel and an accompanying table indicating the appropriate planting zone (streambed, toe of channel, lower, middle, and upper slope, top of bank, outside levee slope) for each species in order to select the appropriate native plants for each planting location. It was also possible to sort for native plant species to be used, or avoided, in special situations (for example, plants that are invasive, fire resistant, less than a certain height, colorful, a barrier to access, evergreen, good for erosion control, high in wildlife food and shelter value, and providing screening).

In 1984, the Urban Creeks Council sponsored the Urban Creek Restoration and Flood Control Act which was signed by California’s Governor in September 1984 (Riley and Sands 1985). The Urban Streams Restoration Program began in 1985 and was administered by the CDWR. “The purpose of the program is to provide grants and technical assistance to those local governments and community groups that want to implement less costly and more environmentally sensitive responses to erosion and flooding problems,” said Riley (1998). Over the next 10-year period, “the program funded 160 alternative restoration projects, including: innovative bank stabilization projects using live and dead plant materials; innovative channel design to increase flood capacities; culvert removal and stream daylighting (sic) to correct storm-water management problems; and land acquisition solutions to reduce flood damages.”

In May 1985, the CDFG released a Final Draft of “Riparian Resources of the Central Valley and California Desert” (Warner and Hendrix 1984). This report was the final product of Phase II of the CRSP that began in 1978 (see above). The report: (1) examined the structure and dynamics of riparian systems; (2) summarized the attributes, values, and vulnerabilities of riparian systems; (3) quantified their historical and present distribution in the Central Valley; (4) presented the major findings of field studies on riparian system distribution, structure, and condition in the Central Valley; (5) examined the nature and problems of desert riparian systems; (6) reviewed riparian resource conservation mechanisms available through Federal, State, and local laws, regulations, and programs; and (7) proposed a series of actions to reverse the chronic, long-term trends of riparian resource decimation and to restore some of these systems to their former status as productive major ecological elements in the California landscape.

In 1986, Aqua Resources Incorporated and Holton Associates (1986) prepared a “Riparian Planting Design Manual for the Sacramento River: Chico Landing to Collinsville” for the U.S. Army Corps of Engineers, Sacramento District. The purpose of the manual was to guide Corps personnel in designing plantings as mitigation for bank protection projects, mainly riprap, along the Sacramento River. The manual provided a list of recommended plant species for each of four plant communities (willow scrub, cottonwood riparian forest, mixed riparian forest, and valley oak riparian forest), which naturally occur in each of four hydrologic zones (river channel, low terrace, high terrace, upper high terrace) along the Sacramento River. The manual defined the planting zone for each plant community based on the average annual flood duration for each hydrologic zone. The manual also provided information on recommended planting densities and spacing patterns and guidelines for the preparation of revegetation plans and specifications. An appendix contained plant data sheets for 26 woody species recommended for riparian plantings (Granholm et al. 1988)

The "Second Native Plant Revegetation Symposium" was held in San Diego in April 1987 (Rieger and Williams 1988). In attendance were 257 people. Six papers dealing with riparian habitat restoration in California were presented at the conference. Results of several riparian revegetation projects implemented by the California Department of Transportation were reported as well as several private development mitigations.

The fourth riparian forest conference in California titled "California Riparian Systems Conference[6]: Protection, Management, and Restoration for the 1990s" (Abell 1989) was held at the University of California, Davis, in September 1988. This conference drew nearly 900 participants. Thirteen papers were presented in the session titled "Implementing Revegetation Projects" and six papers were presented in the session titled "Urban Streams." Other sessions dealing with channel dynamics, rangeland and desert riparian systems, and coastal streams also addressed the restoration of riparian areas.

In 1988, Faber and Holland (1988) published *Common Riparian Plants of California: A Field Guide for the Layman.* This book of photocopies of riparian plants and information on their characteristics and where they typically occur (distribution and elevation) was used by conservationists and community-based organizations seeking to restore riparian areas.

In January 1989, the Society for Ecological Restoration (SER) held its first annual meeting in Oakland, California. The conference proceedings titled "Restoration '89: The New Management Challenge" (Hughes and Bonnicksen 1989) contained a section on the restoration of riparian areas and a section on stream restoration. Other professional societies and conservation organizations also convened conferences in the latter part of the 1980s that included papers pertaining to riparian corridor and riparian ecosystem restoration. Some of these groups that recognized the relationship between restoring riparian areas and achieving the mission of their organization included: Association of State Wetland Managers, Inc.; California-Nevada Chapter of the American Fisheries Society; Salmonid Restoration Federation; Society for Range Management; and the Urban Creeks Council.

In 1989, the U.S. Fish and Wildlife Service published *The Ecology of Riparian Habitats of the Southern California Coastal Region: A Community Profile* (Faber et al. 1989). The purpose of this publication was to describe the structure and function of riparian habitat in southern California. This biological report: (1) described the physical setting and geofluvial processes in riverine systems; (2) outlined the effect of water regime on the establishment and succession of riparian plant communities; (3) described the most common species of riparian plants; (4) detailed the fauna dependent upon and that uses the riparian habitat; (5) summarized riparian ecosystem processes and values; and (6) spelled out the myriad of governmental jurisdictions and relationships that

[6] This was the second "California Riparian Systems Conference" convened by U.C. Davis Extension (Abell 1989).

affect the use of, and the ability to conserve, riparian habitat. The publication concluded with a section on riparian ecosystem restoration including a number of case studies. Unfortunately, no similar publication was ever produced for the riparian ecosystems of northern California.

Also in 1989, the United States Environmental Protection Agency published *Wetland Creation and Restoration: The Status of the Science.* Volume I (Regional Reviews) contained a paper on riparian wetland creation and restoration in California (Stanley 1989). An appendix to this paper contained profiles for 37 riparian restoration projects in California.

The 1990s

By the 1990s, some biologists, ecologists, and hydrologists had gained enough experience conducting riparian restoration projects that they began offering training in riparian habitat restoration. The Wetland Training Institute, Inc., offered workshops on Riparian Habitat Restoration at various locations in California throughout the 1990s. SERCAL and SER offered Riparian Habitat Restoration workshops in connection with conferences held in California during the latter part of the 1990s and 2000.

In 1991, State legislation created the California Riparian Habitat Conservation Program (CRHCP) within the Wildlife Conservation Board (WCB). The mission of the program is to develop coordinated conservation efforts aimed at protecting and restoring the State's riparian ecosystems. The goals of the CRHCP, as noted in its enabling legislation, are to protect, preserve, restore, and enhance riparian habitat throughout California. To accomplish the program's objectives, the WCB was authorized to award grants for riparian conservation purposes (acquisition and restoration) to non-profit organizations, local government agencies, State departments, and Federal agencies.

Also in 1991, the California State Lands Commission published *Delta-Estuary—California's Inland Coast: A Public Trust Report* (Argent 1991). This report described the Delta's geologic, hydrologic, biologic, and cultural history and the public trust uses that are dependent on these resources. Citing Madrone Associates (1980), the report stated that "riparian habitat is used by more vertebrate wildlife, 107 species, than any other Delta habitat type." The report indicated that the clearing of levees for maintenance or placement of rock revetment has resulted in a severe loss of riparian habitat and shaded riverine aquatic habitat. The report concluded with a survey of the institutions and entities that manage the delta's resources.

In 1992, the National Research Council's Committee on Restoration of Aquatic Ecosystems concluded: "Given that healthy, vegetated riparian habitat and bottomlands are essential to the natural ecological functioning of associated streams and rivers—and are among the nation's rarest habitats due to prior devastation—riparian habitat and bottomland restoration should be made a high national priority along with the restoration of the stream and river channel itself" (NRC 1992). The committee recommended that a national aquatic ecosystem restoration strategy be developed for the United States. By this time, California's State agencies were already actively moving to preserve, conserve, and restore California's rivers, creeks, and riparian ecosystems in cooperation with local agencies, academic institutions, non-governmental organizations, and professional consultants.

In 1992, representatives of 28 agencies gathered at the request of California Resources Secretary Douglas Wheeler. These agencies recognized that diverse programs, goals, missions, regulations, and geographic regions required diverse information to support decisions regarding the management and conservation of California's rivers. They recommended a process that began with a survey of professional judgment of California's river conditions. They continued with the accumulation, organization, and internet publication of a large and diverse body of facts and tools dedicated to the analysis and management of California's rivers. The Information Center for the Environment (ICE) the University of California, Davis, conducted the California Rivers Assessment[7] (CARA) Project under the oversight of the WCB. ICE submitted the Final Report for the CARA Project titled *California Riparian Habitat Inventory and Assessment* (ICE 2009) to the WCB in April 2009.

The first phase of the CARA Project, a Professional Judgment Assessment (PJA), drew upon the knowledge, expertise, and opinions of resource managers, scientists, and other river experts to assemble a database of information about the condition of riparian and aquatic resources for California's 196 largest rivers. Riparian criteria included the presence (or absence) of a natural flow regime, vegetation size and land coverage, the trend in riparian habitat distribution over the past 25 years, and the impact of human activities on the areas. The PJA succeeded in collecting information for 616 segments on 145 rivers. Numerical scores for each river segment were distributed into quartiles labeled "Outstanding," "Substantial," "Moderate," or "Limited." A pilot comparative evaluation of the PJA responses was performed to illustrate how the PJA survey information could provide a Statewide perspective on the relative condition of riparian and aquatic river resources. In the final report, ICE (2009) indicated that out of California's 172,000 miles of rivers, 13,631 miles were rated by the CARA PJA. Of the 13,631 miles rated by PJA, 1,379 miles were rated "Outstanding" for riparian and 1,828 miles were rated "Limited" for riparian[8].

The fifth riparian forest conference in California titled "California's River Heritage[9]: A Conference on Conservation Issues, Policy and Implementation Strategies" (McCoy 1992) was held in Sacramento in May 1992. Concurrent sessions focused on: (1) Wild, Scenic and Recreation Rivers; (2) River Corridors and Parkways; and (3) Urban Riverfronts, Creeks, and Streams. Several of the speakers described river corridor restoration programs wherein riparian habitat restoration was, or would become, a major component including: Putah Creek (Sanders 1992); San Lorenzo River (Hall 1992); San Luis Creek (Jones 1992); and San Joaquin River (Dangermond 1992). Several stream restoration projects that received funding from the Department of Water Resources Urban Stream Restoration Program were described by Earle Cummings (1992).

In 1993, the California State Lands Commission published *California's Rivers: A Public Trust Report* (Argent 1993). The subject of this report was the condition of the

[7] CARA is a computer-based data management system designed to give resource managers, policy-makers, landowners, scientists, and interested citizens rapid access to essential information and tools with which to make sound decisions about the conservation and use of California's rivers. Cara contains 39 sets of mapped geographical information system (GIS) layers, 60 sets of tabular (database) and textual (text) data, as well as links to 510 additional maps, tables, and texts. All these data are organized by watershed and theme.

[8] These small numbers for the ranking of riparian and aquatic resources as "Outstanding" and "Limited" were the result of the limited number of miles of river segments evaluated. Funding was never obtained to complete the evaluation for the remainder of California river segments.

[9] This was the third "California Riparian Systems Conference" convened by U.C. Davis Extension.

rivers of California and their watersheds. The report documented the causes of their alteration and the nature and extent of their degradation; identified means by which their degradation could be avoided or reduced; and suggested measures to be taken for their restoration. The report addressed the need for riparian habitat restoration and explained the role of numerous Federal, State, and local agencies and organizations in the protection, management, and restoration of riparian areas. The report indicated that "in California, many riparian restoration projects have been implemented, but most are on a relatively small scale, rather than for whole systems" (Argent 1993).

Recognizing the importance of riparian habitat for landbirds in California, California Partners in Flight (CalPIF) initiated the Riparian Habitat Joint Venture (RHJV) Project in 1994. Eighteen Federal, State, and non-profit organizations signed a Cooperative Agreement to protect and enhance habitats for native landbirds throughout California. Modeled after the successful Joint Venture projects of the North American Waterfowl Management Plan, the RHJV reinforces other collaborative efforts that protect biodiversity and enhance natural resources as well as the human element they support. "The mission of the RHJV is to promote the protection, restoration and enhancement of riparian habitat sufficient to support the long-term viability and recovery of California's native landbirds and other associated species" (Chrisney 2003).

The sixth riparian forest conference in California titled "California's Riparian-River Ecosystems Conference IV:[10] Addressing Current Land Use and Resource Conflicts" (Laird et al. 1995) was hosted by U.C. Davis in Sacramento in November 1995. Speakers provided a historical perspective of the physical and fluvial processes and riparian and aquatic resources of California's rivers. There were updates on recent flooding and discussions of floodplain management. River management case studies addressed restoration of the Klamath, Trinity, Los Angeles, Russian, San Joaquin, and Sacramento rivers.

In October 1998, The Federal Interagency Stream Restoration Working Group, comprised of 17 Federal agencies, published *Stream Corridor Restoration: Principles, Processes, and Practices* (FISRWG 1998). Part II (Developing a Restoration Plan) provided suggested approaches for identifying problems and opportunities; developing goals, objectives, and alternatives; and planning the implementation, monitoring, and evaluation of restoration projects. Part III (Applying Restoration Principles) contained guidance on analysis of corridor conditions, restoration project design, and restoration project implementation, monitoring, and management. Although only one agency representative (USFWS) from California was on the Production Team for this document, much of its contents was in line with the approaches and practices already in use for stream and riparian restoration in California. This document was useful for those entering the field of stream and riparian restoration and managers and administrators of restoration projects.

The 2000s

The following noteworthy documents were published after 1999; however, they report on, or synthesize, knowledge that was generated in the latter part of the 20th century.

In August of 2000, CalPIF and RHJV released the first version of *The Riparian Bird Conservation Plan: A Strategy for Reversing the Decline of Riparian Associated Birds*

[10] This was the fourth "California Riparian Systems Conference" convened by U.C. Davis (McCoy 1995).

in California. "The Riparian Bird Conservation Plan seeks to synthesize and summarize the current state of scientific knowledge concerning the requirements of birds in riparian habitats. It provides recommendations for habitat protection, restoration, management, research, monitoring, and policy to ensure the long-term persistence of birds dependent on riparian ecosystems," (RHJV 2004). "The RHJV chose to emphasize the ecological associations of individual species as well as those of conservation concern. In doing so, the RHJV included a suite of focal species whose requirements define different spatial attributes, habitat characteristics, and management regimes representative of a 'healthy' system". (RHJV 2004).

Version 2.0 of the Conservation Plan (RHJV 2004) expanded the original list of 14 "focal" riparian bird species to 17 "focal species." Criteria for the selection of focal species included: (1) use of riparian vegetation as their primary breeding habitat in most bioregions of California; (2) warrant special management status—endangered, threatened, or species of special concern on either the Federal or State level; (3) have experienced a reduction from their historical breeding range; (4) commonly breed throughout California's riparian areas; and (5) have breeding requirements that represent the full range of successional stages of riparian ecosystems.

The seventh riparian forest conference in California titled "California Riparian Systems[11]: Processes and Floodplains Management, Ecology, and Restoration" (Faber 2003) was convened by the Riparian Habitat Joint Venture in Sacramento in March 2001. The conference focused on riparian and floodplain processes, habitat management and restoration, monitoring, and partnerships in riparian area activities. Seventy papers were presented in this volume. Most of the 18 papers in Section III (Restoration) report on work performed in the 1980s and 1990s.

In October of 2003, the CDFG published additional elements to the third edition of the *California Salmonid Stream Habitat Restoration Manual* (Flosi et al. 1998). The new Part XI (Riparian Habitat Restoration) addresses measures for the conservation and management of riparian habitats and measures for the the restoration of native riparian habitats. It provides guidance on riparian revegetation project planning, provides information on the sources of native plant material, and discusses revegetation techniques. Appendix XI-A contains fact sheets for numerous central and north coast native riparian plants while Appendix XI-B contains fact sheets for a number of invasive nonnative plant species common to riparian areas.

The eighth riparian conference in California was held by the American Water Resources Association (AWRA) in Olympic Valley, California, in June of 2004. Titled "Riparian Ecosystems and Buffers: Multi-scale Structure, Function and Management" (Dwire and Lowrance 2006), the conference included papers on the role of riparian ecosystems and riparian restoration projects in protecting the water quality of the Lake Tahoe Basin. Key papers were published in a special issue of the Journal of the American Water Resources Association (JAWRA 2006).

The ninth riparian conference in California was convened in Sacramento by the Riparian Habitat Joint Venture in December of 2007. Titled "Riparian Habitat Conservation and Flood Management in California[12]" (RHJV 2007), the conference proceedings

[11] This was the fifth "California Riparian Systems Conference."

[12] This was the sixth "California Riparian Systems Conference." It was mistakenly referred to as the "Fourth Conference on California Riparian Systems" in the introduction to the proceedings.

contained nearly 80 papers and discussions focused on integrating levee management, flood protection, riparian conservation, and wildlife protection so as to sustain a safe, vibrant, and healthy environment along the streams and rivers of California. The conference was specifically designed to address the critical need for restoring the essential biodiversity of riparian areas while also ensuring good water quality and flood safety.

In 2007, RHJV partners identified a need for guidelines for planning and implementing riparian restoration projects on the ground. RHJV convened a group of restoration experts for a workshop to produce a handbook of restoration strategies, standards, and guidelines. The goal was to provide practitioners, regulators, land managers, planners, and funders with basic strategies and criteria to consider when planning and implementing riparian conservation projects. River Partners, a RHJV partner, took the lead in developing this handbook. In July 2009, River Partners published the second edition of the *Riparian Habitat Restoration Handbook* (Griggs 2009). The goal of the handbook is to explain the proposal/planning process for a site-specific riparian restoration project for wildlife habitat to the first-time, as well as the experienced, restoration project manager. The handbook can be used for planning projects, creating budgets, and assessing restoration success. Ecological, biological, and regulatory components of a riparian restoration project are described. The handbook emphasizes the ecological river processes operating on floodplains and in river channels that create characteristic vegetation structure that forms wildlife habitat—as the foundation for planning a riparian restoration project. Case studies of Statewide riparian restoration projects that faced site-specific conditions illustrate implementation of the principles presented in this handbook.

Part 2—Riparian Restoration Projects: 1960 to 2000

The California Wildlife Habitat Relationships system of classification identifies seven major riparian habitats in California: montane riparian, valley foothill riparian, desert riparian, palm oasis, freshwater emergent wetland, wet meadow, and aspen (RHJV 2004). The vast majority of riparian restoration projects in the last half of the 20th century were conducted in the valley foothill riparian habitat type. This is because human impacts were the greatest in the Central Valley, in the lower foothills of the Cascades and Sierra Nevada, and in the Coast Ranges. Therefore, most of the riparian habitat restoration projects discussed below pertain to the valley foothill riparian habitat type.

In 1990, the Riparian Revegetation Study Group conducted a survey[13] of riparian restoration projects in California. Information was collected for more than 276 riparian restoration projects in California, of which 226 projects had been implemented or were in the process of being implemented. Fifty-nine percent of the projects were less than 5 acres in size, 72 percent of the projects were less than 10 acres, and 83 percent of the projects were less than 20 acres. The primary purposes for which these riparian restoration projects were conducted were: streambank stabilization (40 percent); mitigation for project impacts (34 percent); fish/wildlife conservation (13 percent); urban stream restoration (8 percent); and restoration of degraded public lands (5 percent).

[13] Survey conducted by John Stanley on behalf of the RRSG. Survey records and data were not published.

We have divided the projects discussed in this section into the following categories, which differ only slightly from the “purposes” mentioned above:

- Bank stabilization/erosion control
- River/stream/watershed restoration
- Urban creeks restoration
- Compensatory mitigation
- Habitat conservation/restoration
- Rangeland restoration
- Invasive plant removal/control

We recognize that many of these early projects were multi-purpose. Assignment of restoration projects to these categories is based on the primary reason for the restoration project. Refer to table 7 for additional information on the restoration projects mentioned in the text, including their general location (city/county), project sponsor(s), and the various actions involved in project implementation. Within each category in table 7, the projects are organized in order of the year(s) when they were implemented. Refer to table 8 for the scientific names of plant species mentioned in the text and in table 7.

We have placed our emphasis on “early” restoration projects—projects that were implemented in the 1970s and 1980s. However, in doing so we have not done justice to many of the projects implemented in the 1990s—many of which are larger projects that provided greater benefits to wildlife and the environment. We set an arbitrary cut-off date of the end of Year 1999 by which a project had to have begun construction/installation to be included in the text and table 7. This means that many significant projects that were planned in the 1980s and 1990s but which were not constructed until Year 2000 or thereafter are not mentioned in this chapter. We recognize that many of these early projects did not achieve all, or even most, of the ecological attributes of restored ecosystems as outlined in the SER Primer (SER 2004) and expanded upon in Clewell and Aronson (2013).

Additionally, we have focused on “Horticultural Restoration” (formerly referred to as “revegetation,” “active revegetation,” or “active restoration”) and have given much less attention to “Process Restoration” (formerly referred to as “passive restoration”). Griggs (2009) stated: “‘Horticultural Restoration’ refers to a high level of site management and external human inputs that include site preparation (land-leveling, disking), planting of nursery-grown trees and shrubs in predesigned patterns, irrigation, and chemical [and/or mechanical] weed-control for three or more years. Horticultural restoration is typically employed along rivers where the river’s physical processes have been severely modified by humans with dams, levees, bank stabilization, and water diversions.”

Griggs (2009) also specified: “‘Process Restoration’ strives to reestablish river processes onto the site. Process Restoration is appropriate on riparian sites along a river that retains functioning river processes (e.g., no dams, and few levees or water diversions). Process Restoration attempts to restore a site by working with existing river processes.” “Process Restoration” has also been referred to as “Process-based Restoration.” A Process-based Restoration approach might simply be the removal of a perturbation (for example, livestock) from a riparian corridor or a change in management (for example, grazing regime) followed by allowing native plants to reestablish on the site through natural processes. Process-based restoration projects often involve a single action (for example, removing livestock, breaching of a levee to reconnect the river to its floodplain, conducting a controlled burn, removal of invasive plants) followed by sitting back and letting natural regeneration occur.

Table 7—Examples of California riparian ecosystem restoration projects: 1967-2000.

Project name	Date(s)	Location	Project sponsor	Actions taken to implement project
			Bank stabilization/erosion control projects	
Monument Bend Demonstration Project	1967	Sacramento County	U.S. Army Corps of Engineers	Planted a variety of trees and shrubs along 3 miles of riverbank where the levee had been set back and the new berm protected by rock.
Bull Creek Bank Stabilization Project	1971-974	Humboldt County	CA Dept. Parks and Recreation	Planted willow cuttings and alders in riprapped banks along Bull Creek in Humboldt Redwoods State Park.
Intertidal Zone Levee Experimental Planting (UC Davis Environmental Horticulture Department)	1978-1980	San Joaquin County	CA Dept. Water Resources	In 1978, UC Davis researchers planted intertidal zone sites at the Webb Tract and Mandeville Island with buttonbush, three species of spikerush, Goodding's willow, and common tule. In 1980, planted intertidal zone on Terminous Tract with tules and willow cuttings and spikerush along with erosion control fabric. Seeded with alkalai bulrush and water grass.
Lost Canyon Rehabilitation	1983-1985	Near Wishon Reservoir, Fresno County	Pacific Gas & Electric Company	Restored eroded watercourse downstream from PG&E Helms Pumped Storage Project. Planted montane riparian vegetation along 3 miles of eroded streamside (90 acres) at elevations between 6,300 and 7,700 ft. Planted over 50,000 seedlings over a 2-year period. Riparian species planted included blue elderberry, mountain alder, mountain ash, black cottonwood, quaking aspen, and willow. Compared various planting treatments.
McDonald Creek Restoration Project	1983 & 1986	Humboldt County	Redwood Community Action Agency	Planted approx. 4,000 trees (alder, willow, spruce, redwood) along 1 mile of McDonald Creek and ½ mile along the north fork (total of 9 acres).
Tryon Creek Restoration Project	1984-1987	Del Norte County	Redwood Community Action Agency	Planted riparian trees and upland conifers on private property along one-mile reach of Tryon Creek, tributary to the Smith River. Fenced streambanks to exclude livestock.
Prairie Creek Restoration Project	1986	Humboldt County	Redwood Community Action Agency	Planted 8 acres of streamside deciduous forest (primarily willows and red alders) and adjacent conifer forest (Sitka spruce and coast redwood) along approximately 2 miles of eroding streambank on Prairie Creek (tributary of Redwood Creek). Installed fencing to exclude cattle. Installed log and hog wire bank stabilization structure on some of laid-back banks.
Red Clover Creek Erosion Control Demonstration Project	1986-1987	Plumas County	Plumas Corporation	Revegetation of streambanks along 1 mile of Red Clover Creek (elevation 5,497 ft.), tributary to East Branch of the North Fork of the Feather River, with montane riparian vegetation. Installed check dams to reduce channel gradient and downcutting of streambed. Studied survival rates of four hardwood species (coyote willow, mountain alder, black cottonwood, quaking aspen) native to Red Clover Valley. Compared survival of unrooted stakes vs. rooted liners planted in fall vs. spring. Installed 3 miles of exclosure fencing to control cattle access.
Georgiana Slough Pilot Bank Protection Project	1999	Near Walnut Grove, Sacramento County	California Bay-Delta Authority	Installed biotechnical bank protection measures (brush boxes, brush bundles, coir biologs) along 7,000 feet of Georgiana Slough. Established 8,000 feet of tules along Georgiana Slough and North Fork Mokelumne River.

Table 7—*Continued.*

Project name	Date(s)	Location	Project sponsor	Actions taken to implement project
			River/stream/watershed restoration projects	
Lagunitas Creek Watershed Restoration Program	1982-1988	Marin County	Marin County Resource Conservation District	There were 70 individual work sites in this project. Activities included gully repair, installation of check dams, post and wire fencing, gabion and riprap placement, and willow sprig planting. Treated 21 miles of unpaved roads to control erosion (regrading and installation of culverts, fords, and waterbars).
San Simeon State Beach Riparian Restoration Program	1985-1986	San Luis Obispo County	California Dept. of Parks and Recreation and California Coastal Commission	Removal of nonnative trees and shrubs along San Simeon Creek followed by installation of riparian trees and shrubs along both the north and south banks of the creek. Plant species installed included California sycamores, Fremont cottonwoods, California bay, coyote bush (sic), coast live oaks, willows, breadless (sic) wild-rye, and purple needle grass.
Walker Creek Watershed Restoration Program	1986-1990	Tomales, Marin County	Marin County Resource Conservation District	Rangeland erosion control and gully repair. Streambank repair including riprapping toes of banks, constructing of log crib wall interplanted with willows, and erecting gradient control structures. Rehabilitation of dirt road system.
Uvas Creek Park Preserve Restoration Project	1995	City of Gilroy, Santa Clara County	City of Gilroy Dept. of Parks and Recreation	Reconstructed sinuous, meandering channel on 0.6-mile reach of Uvas Creek within Uvas Creek Park Preserve in November 1995. In-stream channel improvements included rock vortex weirs (placement not as designed) and log and rootwad bank stabilization. Most improvements washed out in February 1996 resulting in an irregular, braided sand and gravel channel and eroding streambanks. Riparian plantings on channel banks did not have time to become established, or were not installed, prior to bank erosion.
Cold Creek/Pioneer Trail Stream Habitat Restoration	circa 1998	City of South Lake Tahoe, El Dorado County	California Tahoe Conservancy	Stabilization of eroding streambanks using boulders, logs and rootwads. Woody material and boulders were placed in the stream so as to improve fishery habitat. (Lake Tahoe Basin - Trout Creek Watershed)
Trout Creek Wildlife Enhancement and Stream Restoration Project	1999-2001	City of South Lake Tahoe, El Dorado County	City of South Lake Tahoe and California Tahoe Conservancy	Relocated Trout Creek to its historic position in middle of Trout Creek Meadow. Constructed approx. 3.5 miles of new sinuous stream channel and restored 107 acres of meadow. Creek was reengineered to reestablish hydrologic connectivity between the stream and its former floodplain thereby increasing flood frequency and duration, raising groundwater levels, and improving health of wetland and riparian vegetation. Used stacked sod revetments to stabilize streambanks along with planting of willow sprigs.
Bear Creek Meadow Restoration Project	1999-2000	Shasta County	California Dept. of Fish and Game	Restored historic floodplain and channel connection in Bear Creek Meadow. Constructed and revegetated 2.2 miles of channel to mimic pre-disturbance conditions. Filled incised gully to floodplain elevations. Propagated and planted 4,500 shrubs from 20 native species. Used sod mat transplants to stabilize new exposed banks. Transplanted large willows, rose, hawthorne, chokecherry, and spirea. Planted over 109,000 plugs of native sedge and rush species.

Table 7—*Continued.*

Project name	Date(s)	Location	Project sponsor	Actions taken to implement project
			Urban creeks restoration projects	
Strawberry Creek Park	1983	City of Berkeley, Alameda County	City of Berkeley Public Works Department	Daylighted 200-foot long section of Strawberry Creek by removing concrete culvert (20 feet below grade) in place since 1904. Revegetated with coastal riparian plant species along 1,200-foot long reach of reconstructed streambanks stabilized with concrete recycled from removal of cross-street.
Miller Creek Restoration Project	1985-1986	Marin County	Deerfield Park/ Lucas Valley Site Developer	Regraded deeply incised stream channel to create a compound channel with a low flow channel and an overflow terrace. Installed vegetated riprap and vegetated spur dikes to protect meander bends. Stabilized toes of eroding banks with rocks and willow cuttings. Planted willows and a mixture of native trees and shrubs and hydroseeded with native perennial grasses.
Wildcat-San Pablo Creeks Flood Control Project	1986-1988	City of Richmond, Contra Costa County	USACE and Contra Costa County Flood Control District	Restoration of a channelized stream. Constructed "natural-like," "two stage", flood control channel with a meandering bankfull channel and floodplain. Installed riparian plantings along both sides of low-flow channel.
Carmel River Biotechnical River Restoration	1986-1988	Monterey County	Monterey Peninsula Water Management District	Regraded pilot channel flanked by second-stage floodplain at level inundated by approx. 2-year flood. Planted 25-acre Schulte Road Restoration Project site with a series of willow rows at toe of eroding streambanks and willow groins in herringbone pattern on floodplain to narrow excessively wide sections of channel and stabilize river meanders. Also, installed biotechnical revegetated riprap and post and wire revetment with revegetation on outside bends. Installed some Fremont cottonwoods translocated from Central Valley whereas Carmel Valley had only a native black cottonwood forest.
Strawberry Creek Restoration Project	1987-1988	City of Berkeley, Alameda County	University of California, Berkeley	Repaired old check-dams and installed new ones on section of Strawberry Creek on UC Berkeley Campus. Stabilized eroding streambank using redwood log crib wall. Crib wall was backfilled and planted with about 25 species of native plants.
Strawberry Creek Management Plan/ Program	1987+	City of Berkeley, Alameda County	University of California, Berkeley	UC Berkeley Campus watershed management strategies consisting of erosion control, non-point source pollution mitigation, and stormwater management. Gully control and repair through the use of biotechnical and soil bioengineering methods utilizing native vegetation and indigenous materials.
First San Diego River Improvement Project	1987-1989	City of San Diego, San Diego County	City of San Diego	Revegetation of 26.8 acres of riparian woodland along a 7,000-foot section of the San Diego River in Mission Valley. Planted willows, Fremont cottonwoods, sycamores, live oaks and other appropriate riparian corridor species.
Buena Vista Creek Restoration Project	1988	City of Vista, San Diego County	City of Vista and California State Coastal Conservancy	Installation of series of drop structures to slow water flow. Riparian habitat restoration along Buena Vista Creek to reduce sedimentation in Buena Vista Lagoon.

Table 7—*Continued.*

Project name	Date(s)	Location	Project sponsor	Actions taken to implement project
			Compensatory mitigation projects	
Sacramento River Maintenance Area 9	1981	Sacramento County	CA Dept. of Water Resources	Installed 1,142 native trees and shrubs on the upper banks of a 9.5-mile section of rock-reinforced levee located just downstream of Sacramento. Planted white alder, coyote brush, Oregon ash, western sycamore, live oak and valley oak. Plants were planted in clusters of 3-5 individuals, 100 feet apart in accordance with Reclamation Board standards. Monitored plantings through 1983.
Guadalupe River Revegetation Project	1981	Santa Clara County	Santa Clara Valley Water District	Planted California native plant species on bank slope along channelized section of Guadalupe River.
I-8/I-15 Mitigation	1982	San Diego County	Caltrans	Lowered upland adjacent to San Diego River in Mission Valley 10-14 feet in elevation to create floodplain planting bench. Planted 6 acres of willow/cottonwood riparian woodland with sycamores.
Caldecott Park Creek Revegetation	1983-1985	Alameda County	Alameda County Flood Control District	Installed riparian plant species along Caldecott Creek within Caldecott Park. Planted alder, toyon, big-leaf maple, coast live oak, and bay laurel.
Alamitos Creek Revegetation Project	1984	Santa Clara County	Santa Clara Valley Water District	Planted riparian trees, shrubs and groundcover along three miles of Alamitos Creek totaling approx. 20 acres.
Llagas Creek Watershed Mitigation	1984-1987	Santa Clara County	USDA SCS and Santa Clara Valley Water District	Revegetation with riparian trees, shrubs and herbaceous plants along 10-mile creek corridor.
Morena Street Site	1985 & 1988	San Diego County	Caltrans	Planted willow, sycamore, and cottonwood groves (3.5 acres) and Coastal Sage Scrub and Mixed Chaparral communities on 9-acre site along the San Diego River.
Crescent Bypass Riparian Revegetation Project	1985-1988	King County	Kings River Conservation District	Planted riparian vegetation along 6 miles of the Crescent Bypass between the south and north forks of the Kings River.
Saratoga Creek Flood Control Project Revegetation	1986 & 1988	Santa Clara County	Santa Clara Valley Water District	Installed 3,000 native riparian plant species in planting pockets and open bottom concrete planters within 7 acres of gabion-lined (stacked and mattress) flood control channel and in bare earth at top of bank.
Sweetwater Bridge Mitigation	1986-1987	San Diego County	Caltrans	Lowered 2-acre site alongside the Sweetwater River down 5-8 feet. Planted willow scrub woodland to replicate the habitat of least Bell's vireo.
Lower Coyote Creek Pilot Revegetation Project	1986-1987	City of San Jose, Santa Clara County	Santa Clara Valley Water District	Planted total of 3,640 plants comprised of 15 native riparian corridor plant species on 4.4-acre site on floodplain adjacent to lower Coyote Creek. Woody plant species included California box elder, white alder, Oregon ash, California black walnut, western sycamore, Fremont cottonwood, coast live oak, valley oak, red willow, yellow willow, blue elderberry, and California bay. Multiple types of plant materials (i.e., propagule types) were tested for each plant species. Comparison of approx. equal areas irrigated by overhead irrigation versus flood irrigation. Tested a variety of weed management strategies and techniques.

Table 7—*Continued.*

Project name	Date(s)	Location	Project sponsor	Actions taken to implement project
M&T Ranch Elderberry Mitigation	1987	Butte County	CA Dept. of Water Resources	Scattered plantings of blue elderberry on 167-acre parcel as mitigation for endangered Valley Elderberry Longhorn Beetle (VELB) habitat loss associated with Sacramento River Bank Protection Project.
Novato Creek Flood Control Project	1987-1988	Marin County	Marin County Flood Control District	Planted riparian trees and shrubs on approx. 7 acres of riparian corridor along Novato Creek. Plants included California buckeye coyote bush (sic), black walnut, Oregon ash, red alder, coast live oak, valley oak, bay laurel, and elderberry.
Spring Creek Flood Control Project	1987-1988	Sonoma County	Sonoma County Water Agency	Off-site revegetation of approx. 12 acres of riparian habitat along the Laguna de Santa Rosa. Primary plantings were oaks in the upper regions and Oregon ash in wetter areas.
Sacramento River Mile 154.6 Right	1987-1988	Colusa County	U.S. Army Corps of Engineers	Planting of blue elderberry and other high terrace riparian species at two sites totaling 2 acres as mitigation for loss of VELB habitat due to installation of rock slope protection.
San Joaquin Marsh Mitigation Bank	1987-1988	Irvine, Orange County	The Irvine Company	Planted 8.5 acres of willows, cottonwood and sycamore adjacent to San Diego Creek in Irvine out of ultimate total of approximately 30 acres.
SR-52 Mitigation/Mission Trail Park	1989-1990	San Diego County	Caltrans	Lowered 33 acres on 48-acre parcel adjacent to the San Diego River 10-17 feet. Planting of riparian species followed prescription for endangered least Bell's vireo habitat generated by Baird and Rieger (1989). Upland slopes created by the grading were seeded with coastal sage scrub which provided habitat for the threatened California gnatcatcher.
Hwy 85 Mitigation	1993	San Jose, Santa Clara County	Caltrans	Lowered 24-acre off-site mitigation area adjacent to middle Coyote Creek 10-15 feet in order to bring the final grade closer to the groundwater table. Constructed side channel to convey stream flows through project site. Revegetation included streamside, floodplain and valley oak riparian forest associations. A total of 10,484 container plants were installed.
		Habitat conservation/restoration projects		
Colorado River Dredge Spoil Revegetation	1979-1980	Near Palo Verde, Imperial County	USDI Bureau of Reclamation	Planted cottonwoods, willows, and quail bush on three sites located along the lower Colorado River including one 49-acre site on the Cibola National Wildlife Refuge. Studied effects of deep tillage and irrigation on plant growth and survival of rooted cuttings of Fremont cottonwoods on total of 79 acres of dredge-spoil.
Sepulveda Wildlife Reserve Revegetation Project	1984 & 1986	City of Van Nuys, Los Angeles County	US Army Corps of Engineers	Planted 17 acres of riparian woodland within the Sepulveda Flood Control Basin adjacent to the Los Angeles River. Installed 1,544 plants in 1984. Installed plants and cuttings in three 1-acre test plots in 1986 to compare irrigation methods (overhead vs. hand watering) with no irrigation. Plantings included box-elder, white alder, velvet ash, western sycamore, Fremont cottonwood, black cottonwood, coast live oak, Engelmann oak, valley oak, arroyo willow, and California bay.
Kern River Preserve Yellow-billed Cuckoo Habitat Enhancement	1986-1989	Near Weldon, Kern County	The Nature Conservancy	In 1986, Bertin Anderson planted 25-acre pilot project at TNC's Kern River Preserve situated along the south fork of the Kern River. Between 1986 and 1989, Bertin Anderson supervised the planting of 142 acres of cottonwoods and willows (four species) at the Kern River Preserve.

Table 7—*Continued.*

Project name	Date(s)	Location	Project sponsor	Actions taken to implement project
Cosumnes River Preserve Riparian Restoration	1988-2000s	Sacramento County	The Nature Conservancy and Partners	Planted 10 acres of valley oak riparian forest on fallow agricultural land adjacent to the Cosumnes River. New technology—collar and screen developed by Frank Chan of PG&E—was used to protect acorns and seedlings from predation by rodents and grasshoppers. First use of drip-irrigation by TNC. Documented growth of seedlings with irrigation.
Kopta Slough Preserve Riparian Restoration	1989-1995	Tehama County	The Nature Conservancy	Planted approx. 300 acres. Planted Fremont cottonwood, four willows (red, sandbar, arroyo, black), box-elder, Oregon ash, California wild rose, blackberry, coyote brush. First large-scale native grass plantings – blue rye, creeping rye, meadow barley, purple needlegrass. Soil moisture was studied as to depth to water-table and root growth rates into its surface, as revealed by backhoe pits. Irrigation movement through the soil was monitored by electrical moisture probes.
Stony Creek Preserve Riparian Restoration	1991	Glenn County	The Nature Conservancy and CA Dept. of Water Resources	Planted approx. 500 acres. Conducted more experiments with the timing of weed management and the timing of irrigation on plant growth. Refinement of implementation monitoring.
Sacramento River National Wildlife Refuge (Llano Seco Unit) Riparian Restoration	1991	Butte County	U.S. Fish and Wildlife Service and partners	Species planted included all of those mentioned for Kopta (above) plus additional understory species, such as mugwort, gumplant and evening primrose.
Rangeland restoration projects				
Willow Creek Restoration	1980	Near Adin, Modoc County	USDA Soil Conservation Service - (Now NRCS)	Dumped large rocks into slots cut across the eroded stream channel and keyed into the channel banks and bottom to create a series of rock sills across the channel. Disturbed channel banks were planted with willows and other woody shrubs. Area was fenced to prevent indiscriminate use by livestock.
Clark Canyon Riparian Demonstration Area	1984-1987	Mono County	U.S. Bureau of Land Management	Constructed multiple check dams along one mile of Clark Canyon Creek (tributary to Aurora Creek), East Walker River sub-basin, to control gully head-cutting, trap sediment, raise water table, and restore meadow riparian areas. Elevation 7,000 to 7,300 ft.
Invasive plant removal/control				
Thousand Palms Canyon Tamarisk Control Project	1986-1992	Riverside County	The Nature Conservancy and Partners	Volunteers and California Conservation Corps crews removed tamarisk in Thousand Palms Canyon—a 25-acre, 1-mile long wetland in the center of Coachella Valley Preserve. Infestation of tamarisk threatened native riparian community of desert fan palms, coyote willow, Fremont cottonwoods, common reed, honey mesquite, and screwbean mesquite. Outplanted mesquite grown in on-site nursery. Spread seeds collected on-site from natives including palm, cottonwood, mesquite, saltbush, quailbush, and alkali goldenbush.

Table 7—*Continued.*

Project name	Date(s)	Location	Project sponsor	Actions taken to implement project
Afton Canyon Riparian Restoration Project	1992-1996	San Bernardino County	Barstow Resource Area Office, Bureau of Land Management	300-acre pilot project within the Afton Canyon Area of Critical Environmental Concern (ACEC) on the Mojave River. Actions included construction of barriers to OHV travel; installation of cattle allotment exclusion fencing; use of prescribed fire in dense saltcedar stands; herbicide application to saltcedar resprouts; manual saltcedar stem cutting and herbicide application; revegetation of saltcedar removal areas using natural revegetation; pole planting of cottonwood and willow trees (7,000+); and seeding of native shrubs and grasses.
Russian River Watershed Giant Reed Eradication Program	1992+	Mendocino and Sonoma Counties	Circuit Rider Productions and Partners	Conducted research on control methods, including non-toxic approaches to giant reed removal. Conducted digital mapping of giant reed locations in riparian zones within the watershed. Prioritized sites for giant reed removal and follow-up habitat restoration. Educated landowners and community about values of riparian zones and problems associated with giant reed. Coordinated volunteer and community involvement in giant reed removal and habitat restoration. Conducted long-term monitoring.
Santa Ana River Watershed Arundo Habitat Management Program	1997+	Orange County and Riverside County	Santa Ana Watershed Project Authority (SAWPA) and Partners	Since 1997, SAWPA and its Partners (primarily the Santa Ana Watershed Association and the Riverside County Regional Park and Open-Space District) have removed over 3,000 acres (out of approximately 10,000 acres) of Arundo from the Santa Ana River Watershed. Cleared areas have been replaced with native riparian or wetland vegetation.

Table 8—Common and scientific names for plant species.

Common name	Scientific name
Alkalai bulrush	*Scirpus robustus*
Alkali goldenbush	*Haplopappus acradenius*
Arroyo willow	*Salix lasiolepis*
Arundo (see Giant reed)	*Arundo donax*
Athel	*Tamarix aphylla*
Beardless wildrye	*Leymus triticoides*
Big-leaf maple	*Acer macrophyllum*
Blackberry	*Rubus ursinus*
Black cottonwood	*Populus trichocarpa*
Black walnut	*Juglans hindsii*
Black willow	*Salix nigra* (aka *S. gooddingii*)
Blue elderberry	*Sambucus caerulea*
Blue elderberry	*Sambucus mexicana*
Blue rye	*Elymus glaucus*
Box elder	*Acer negundo*
Buttonbush	*Cephalanthus occidentalis*
California bay or bay laurel	*Umbellularia californica*
California blackberry	*Rubus ursinus*
California buckeye	*Aesculus californica*
California wildrose	*Rosa californica*
Chokecherry	*Amelanchier pumila*
Coast live oak	*Quercus agrifolia*

Table 8—*Continued.*

Common name	Scientific name
Coast redwood	*Sequoia sempervirens*
Common reed	*Phragmites australis*
Common tule	*Scirpus acutus*
Coyote brush	*Baccharis pilularis*
Coyote willow	*Salix exigua*
Creeping rye	*Leymus triticoides*
Engelmann oak	*Quercus engelmannii*
Evening primrose	*Oenothera elata* ssp. *Hirsutissima*
False bamboo (Arundo)	*Arundo donax*
Fan palm	*Washingtonia filifera*
Fremont cottonwood	*Populus fremontii*
Giant reed	*Arundo donax*
Goodding's willow	*Salix gooddingii*
Gumplant	*Grindelia camporum*
Honey mesquite	*Prosopis glandulosa*
Hawthorne (hawthorn)	*Crataegus douglassi*
Meadow barley	*Hordeum brachyantherum*
Mountain alder	*Alnus tenuifolia*
Mountain ash	*Sorbus scopulina*
Mugwort	*Artemesia douglasiana*
Oregon ash	*Fraxinus latifolia*
Palo verde (paloverde)	*Parkinsonia L.*
Purple needlegrass	*Stipa pulchra*
Quailbush	*Atriplex lentiformis*
Quaking aspen	*Populus tremuloides*
Red alder	*Alnus rubra*
Red willow	*Salix laevigata*
Saltbush	*Atriplex polycarpa*
Saltcedar (tamarisk)	*Tamarix ramosissima (*aka *T. pentandra)*
Sandbar willow	*Salix interior;* aka *S. exigua*
Screwbean mesquite	*Proposis pubescens*
Sitka spruce	*Picea sitchensis*
Spikerush	*Eleocharis* spp.
Spirea	*Spiraea* sp.
Tamarisk	*Tamarix ramosissima (*aka *T. pentandra)*
Toyon	*Heteromeles arbutifolia*
Tule	*Scirpus* spp.
Valley oak	*Quercus lobata*
Velvet ash	*Fraxinus velutina*
Water grass	*Echinochloa crusgalli*
Western sycamore	*Platanus racemosa*
White alder	*Alnus rhombifolia*
White root	*Carex barbarae*
Willow	*Salix* spp.
Yellow willow	*Salix lasiandra*

Rieger et al. (2014) refer to these two different approaches to restoration as “Construction and Installation Strategies” (horticultural/active) versus “Management Strategies” (process-based/passive). Actually, many riparian restoration projects employ both strategies on the same site at the same or different times.

Most of the early restoration projects employed horticultural restoration because the project planners were dealing with: (1) land surfaces that were well above active flooding; (2) lowered groundwater tables; (3) altered stream hydrology due to construction and operation of dams in the watershed and an increase in impervious cover in urban watersheds; (4) the presence of certain invasive plants that colonize a site so quickly so as to preclude the natural reestablishment of native plant species; and (5) pressure from clients and/or regulators to get riparian vegetation established quickly.

Notwithstanding our emphasis on horticultural restoration, it needs to be recognized that some restoration projects have operated under the principle that “revegetation projects can sometimes be best accomplished by planting nothing,” (Riley 1998). Riley (1998) went on to state, “The best revegetation project from a standpoint of ecological diversity, and the most economical, may be the project that simply creates the conditions needed for native vegetation to ‘reinvade’ a site.” Unfortunately, the presence of invasive nonnative plants on, or near, most riparian restoration project sites makes this a risky proposition for most project funders and sponsors.

Many of the early riparian restoration projects were not given formal names; in other instances, the authors of papers did not provide the project name but rather only the project location. We have assigned names to these projects in order to facilitate their discussion. Also, we have used the same terms (for example, revegetation) used by the authors in the literature describing the project.

Bank Stabilization/Erosion Control Projects

In 1960, at the request of the State of California, Congress authorized the Sacramento River Bank Protection Project to protect the Sacramento Valley levee system (Kindel 1977). Initially, bank protection was provided at the most critical areas (in other words, areas where erosion had extended well into the levees). Jannssen (1976) recalled: “In order to rebuild the levee and construct the rock protection (riprap), trees, and vegetation growing on the levee were removed. Public concern over this practice led to attempts by the U.S. Army Corps of Engineers (CE) and the state to establish experimental test planting areas … to determine if selected vegetative species could be found that would not constitute a threat to the structural integrity of the levees.” These test plantings were considered unsuccessful (Jannssen 1976).

As mentioned in part 1 above, DWR conducted a Pilot Levee Maintenance Study (CDWR 1966; CDWR 1967) between 1962 and 1967. Field testing for the Pilot Levee Maintenance Study was conducted within the Sacramento-San Joaquin Delta primarily on levees bordering the Sacramento River. The five test sites selected were at Garcia Bend, Ryde, Steamboat Slough, Hood, and Isleton. Much of the work done at the test sites was conducted by other governmental agencies under contract to CDWR. Between 1963 and 1965, tests were conducted under three categories: plant performance and maintenance; levee protection and repair; and revetment with vegetation. Specific types of experiments were conducted under each category. Most of the plant species used in these tests were nonnative groundcovers and trees, with the exception of tests involving the management of existing vegetation. The experiments conducted in this study led to the general conclusion that "alternative levee maintenance practices can be used to allow vegetation on levees" and that "this vegetation can be maintained for the multiple use of levees without jeopardizing the primary function of flood control" (CDWR 1967).

In the late 1960s, the Corps planted trees and shrubs at several selected sites along the Sacramento River to "demonstrate that such vegetation can be successfully grown, can be compatible with flood control requirements, and can offer a significant improvement to aesthetics and other environmental aspects of the river" (Kindel 1977). The most noteworthy project was the CE Monument Bend Demonstration Project in 1967 (Kindel 1977).

Most early streambank stabilization projects relied heavily on structural measures for controlling bank erosion. Riparian vegetation was planted within these structures to soften the visibility of concrete or to take over the function of slowing water adjacent to the banks initially provided by the revetment. The California Department of Parks and Recreation conducted some of the earliest projects; for example, Bull Creek Bank Stabilization in 1971-1974 (Barry 1984; Barry 1985; Barry 1988).

Between 1978 and 1980, researchers from the Environmental Horticulture Department at the University of California, Davis (Whitlow et al. 1984) investigated the potential for using vegetation as an agent for erosion control in the tidal zone on levees in the Sacramento/San Joaquin Delta. They conducted experimental plantings for the CDWR to identify species and planting techniques suitable for application in the intertidal zone[14]. The ultimate goal was a vegetative replacement for riprap. Unfortunately, both research sites were riprapped before the investigations could be completed.

The USDA Soil Conservation Service (SCS, now NRCS) commonly worked with landowners in rural areas and sometimes in urban areas to control streambank erosion. Patterson et al. (1984) described some of the more common streambank stabilization techniques used by the SCS during the 1970s and 1980s. Many of these measures provided "physical protection" (for example, rock riprap, post and wire revetment, gabion baskets), while others employed "vegetative protection" (for example, woody cuttings, rooted woody plants, herbaceous plants) but often in combination with some form physical protection at the toe and/or on the lower bank slope. This woody vegetation grew to cover the physical measures and provide valuable fish and wildlife habitat.

Severe storms in 1978, 1979, 1980, and 1982 caused considerable damage to streams in California. Drawing on Emergency Watershed Protection (EWP) funds, the SCS used structural and vegetative measures to stabilize severely eroding streambanks and reestablish riparian vegetation. From 1978 through 1982, a total of 371 EWP projects was completed. Some of the river systems on which these EWP streambank stabilization/riparian revegetation projects were constructed were the Cuyama River in Santa Barbara and San Luis Obispo counties, Carmel and Salinas rivers in Monterey County, Aptos Creek in Santa Cruz County, and the Santa Clara River in Ventura County (Gray et al. 1984). Although most of the streambank stabilization projects were small, SCS completed work on over 100 miles of streambanks. In 1983, Schultze and Wilcox (1985) evaluated the results of the revegetation work for 29 projects in California's central coast area. "Early SCS revegetation efforts used nonnative species of plants or species not considered riparian," (Gray et al. 1984). Although native species (primarily willows) were most

[14] Although research sites were within the intertidal zone, they were presumably influenced by fresh and/or brackish water based on the plant species used in the experiments. (See table 7.)

commonly used in SCS bank stabilization projects by the late 1970s, unfortunately, during this period, SCS sometimes incorporated invasive nonnative plant species in bank stabilization measures. For example, woody cuttings of saltcedar and athel[15] were planted behind pipe and wire revetment and at the toe of reshaped banks on over 2 miles of the Cuyama River as part of an EWP project constructed in 1979 (Gray et al. 1984). Also, *Arundo donax* (referred to as false bamboo) was planted along with willows in EWP projects constructed on the Carmel and Salinas rivers in Monterey County in 1978 and 1979 (Gray et al. 1984).

The Pacific Gas and Electric Company (PG&E) developed early revegetation techniques for riparian areas affected by its projects. The Lost Canyon Rehabilitation Project in 1983-1985 (Chan and Wong 1989), implemented as mitigation for erosion and habitat loss along a Sierra stream caused by a pipeline rupture, required the development of techniques for revegetation with Sierran montane riparian species in Fresno County. Frank Chan devised and tested some of the earliest innovative measures for native plant revegetation within the riparian corridor.

During the 1980s, citizens voiced opposition to the use of totally engineered structures for streambank erosion control. Revegetation of streambanks with native vegetation became an integral part of streambank stabilization projects. These types of projects often occurred in the northern part of the State and were driven by a concern to restore salmon and steelhead habitat, especially for listed fish species. Salmonid streams had been severely impacted by timber harvesting, road construction, and livestock grazing. The goal was to reduce nonpoint source sediment that was impacting salmonid spawning and rearing habitat. In some cases, the primary action was the construction of fencing to exclude livestock from the streambanks. Some of these projects employed the use of vegetative and quasi-vegetative bank/slope protection techniques such as presented in Schiechtl (1980) and Gray and Leiser (1982).

Several natural resources employment training programs were active in stream and riparian restoration in the 1980s. The Redwood Community Action Agency conducted a number of streambank stabilization and riparian revegetation projects along coastal streams in northern California in the early and mid-1980s. Reichard (1989) reported on restoration projects performed along McDonald Creek, Tryon Creek, and Prairie Creek. The Plumas Corporation performed similar streambank restoration projects in the northern Sierra. The Red Clover Creek Erosion Control Demonstration Project (Lindquist and Bowie 1989; Lindquist and Filmer 1988) involved the cooperation of multiple State and Federal agencies, landowners, and organizations through the use of the Coordinated Resource Management and Planning (CRMP) process[16].

More recent publications on the use of biotechnical/soil bioengineering techniques involving the use of riparian vegetation include Gray and Sotir (1996), Hoag and Fripp (2002), Schiechtl and Stern (1992), and Schiechtl and Stern (1997). An example of a biotechnical bank stabilization approach developed as an alternative to rock revetment (riprap) in the Sacramento-San Joaquin Delta is the Georgiana Slough Pilot Bank Protection Project conducted in 1999 (Hart and Hunter 2004).

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[15] According to Bossard et al. 2000, athel (*T. aphylla*) is not considered an invasive pest under most circumstances whereas saltcedar (*T. ramosissima*) is invasive.

[16] The CRMP process is a collaborative public-private project planning and implementation process that seeks to involve all interested parties in management and restoration decisions and in project implementation.

River, Stream, and Watershed Restoration Projects

A wide variety of river, stream, and watershed restoration projects were implemented in the 30-year period between 1970 and 2000. Many of these projects were components of a larger vision expressed in long-range river and watershed management plans prepared by Federal, State, and local agencies. The USDA Forest Service and the USDI Bureau of Land Management prepared management plans for Federally-designated Wild and Scenic Rivers while The California Resources Agency prepared management plans for State-designated Wild and Scenic Rivers; however, these plans dealt mostly with the preservation and management of the designated river section and not with the restoration of riparian resources. CDFG (now DFW) coordinated with local agencies to prepare State Protected Waterway Management Plans for the San Lorenzo River (Ricker 1979), Big Sur River (County of Monterey 1983), and Little Sur River (Harvey and Stanley Associates, Inc. 1983). Riparian vegetation management and riparian revegetation were components of these plans. Riparian habitat protection and management within the coastal zone was addressed in Local Coastal Plans prepared by counties. There were also locally-driven enhancement plans for watersheds such as the Garcia River (Mendocino County RCD 1992) and Huichica Creek (Napa County RCD 1993). Sometimes, it worked the other way around in which the success of the early restoration projects prompted the preparation of a management plan to address issues in the entire watershed.

In-stream restoration projects focused primarily on restoring fish spawning and rearing habitat; however, planting of riparian vegetation on streambanks was generally included in these projects so as to provide Shaded Riverine Aquatic (SRA) Cover for fish. Large woody debris (LWD) was often reintroduced into the stream system in the form of logs and rootwads to improve fish habitat and cover. The LWD was typically secured to the streambanks and woody cuttings were inserted into eroding bank slopes. Boulders were used for grade control and channel stabilization. Rock groins constructed to slow and redirect streamflow away from the banks were planted with woody cuttings. Streambank stabilization typically employed biotechnical measures incorporating live vegetation as well as dead woody material resulting in riparian habitat benefiting fish, aquatic life, and riparian dependent wildlife species. Many projects also involved the removal or modification of migration barriers such as log jams and culverts. The provision of adequate filter strips of riparian vegetation was a concern for lands managed for timber production, especially within USFS Streamside Management Areas.

Watershed restoration projects/programs generally focused on the installation and use of best management practices for the control of erosion and the prevention of sedimentation of streambeds as well as proper stormwater management for the control of pollutants (nonpoint source pollution prevention/control). Landslides, unstable slopes, and eroding streambanks were often stabilized with biotechnical slope stabilization measures incorporating the use of live and dead riparian vegetation. Local Resource Conservation Districts often served as project coordinators—working with citizen's advisory committees and governmental agencies, preparing and submitting grant applications, keeping track of expenditures, administering work contracts, and coordinating volunteer activities. The Lagunitas Creek (tributary to Tomales Bay) Watershed Restoration Program (Berger 1990; Marcus et al. 1987; Marcus 1989; Witkin 1990) is an example of a program using multiple approaches to control erosion coming from numerous sources throughout a watershed to improve coho salmon and steelhead habitat and reduce sedimentation in Tomales Bay. This program included preparation of an erosion control handbook (Prunuske 1987) containing gully control and streambank stabilization measures utilizing combined vegetative and structural solutions. Copies of the handbook were given to participating landowners and agencies. The Walker Creek Watershed Restoration Project (Berger 1990; Marcus 1989; Marcus et al. 1987) also involved the repair of many erosion sites through the use of riparian vegetation often combined with structural stabilization measures in order to reduce sedimentation in Tomales Bay.

During the 1970s, 1980s, and 1990s, there was a growing movement toward the development of Cooperative Watershed Associations often called watershed councils or alliances. These organizations generally sought to involve all stakeholders in a watershed or stream reach including private landowners, local, State, and Federal agencies, resource users, and citizen's groups. These associations were actively involved in the planning and implementation of river, stream, and watershed restoration projects.

James Barry (1985) described ecosystem restoration underway at multiple sites with the California State Park System for the purposes of erosion control, alien species eradication, and natural ecosystem enhancement. Revegetation within the riparian corridor on State Park lands was being performed as far back as the mid-1970s, mostly to compensate for former logging practices and overzealous stream clearance programs. One such project was the San Simeon State Beach Riparian Restoration Program (Capelli 1985).

Through the years, hydrologists and fluvial geomorphologists played an increasingly important role in riparian corridor restoration. Their expertise was essential in a number of ways, for example: restoring incised and leveed stream systems (Haltiner and Beeman 2003), which allowed for the creation of floodplain terraces that could be planted with riparian vegetation; restoring flooding to floodplain riparian systems including revegetated sites (Swenson et al. 2003); and reconnecting stream channels with their historic floodplains—aka re-hanging streams in meadows (Poore 2003). They recognized the importance of restoring the "physical integrity" (environmental health specific to a particular catchment river system) of rivers and their floodplains created by a "process of dynamic equilibrium punctuated by natural disturbances" (Haltiner et al. 1996), the "natural dynamic character" (management toward a more natural flow regime) of river systems (Poff et al. 1997), and the "natural stability" of stream channels (Rosgen 1996). Moreover, their emphasis on fluvial geomorphological principles caused many restoration practitioners to focus on a process-based approach to riparian habitat and stream restoration (Tomkins and Kondolf 2003). The preferred approach to restoration was to remove or ameliorate the effects of human interventions in the river system and "allow the natural processes to recreate desirable habitat" (Haltiner et al. 1996). Using this approach, success was redefined as the "restoration of key ecologic processes (physical and biological conditions) that are both resilient and evolving" (Haltiner et al. 1996). However, not all stream/riparian restoration projects withstood the test of time. Sometimes, in-stream channel improvements and bank stabilization measures did not withstand high flows long enough for riparian vegetation to become established as was the case with the Uvas Creek Park Preserve Restoration Project (Kondolf et al. 2001; Rosgen 2006; Rosgen 2008).

Stream and meadow restoration projects in the Tahoe Basin were conducted to restore habitat, but primarily to reduce the amount of sediment reaching Lake Tahoe. Projects such as the Cold Creek—Pioneer Trail Stream Habitat Restoration Project (Tahoe RCD 2015), designed with the assistance of Dave Rosgen, stabilized eroding streambanks with logs and rootwads, not only reducing sediment loads, but also creating valuable fisheries habitat. Projects such as the Trout Creek Wildlife Enhancement and Stream Restoration Project (CTC 2015) reconnected channelized streams with their floodplains, resulting in periodic overbank flows depositing sediment on the meadows rather than impacting the clarity of Lake Tahoe.

Similarly, the Bear Creek Meadow Restoration Project (Poore 2003) in eastern Shasta County restored the historic floodplain and channel connection, preventing channel erosion and buffering peak flow events, thereby reducing sedimentation downstream in Fall River. Both the Trout Creek and Bear Creek projects utilized stacked sod mat transplants cut from the meadows to stabilize the banks of the realigned stream channels in conjunction with riparian plantings.

Urban Creeks Restoration Projects

In rapidly growing California cities and counties, creek corridors, however severely impacted, were generally the last remaining undeveloped natural environments within urban and suburban development. Multiple demands were placed on these areas for flood control, active and passive recreation, and trail systems. Urban creeks were used as dumping grounds for trash and discharge of polluted waters. Additionally, urban creeks were subjected to the impacts of significant amounts of impervious cover within their watersheds, which often resulted in channel incision.

In the 1970s and 1980s, there were nationwide and regional movements for the improvement of urban creeks and the remnants of riparian habitat on their banks. Urban planners, citizen groups, etc., recognized the value of these areas as refuge from urban life and as buffers between conflicting land uses. At the same time, scientists, conservationists, environmentalists, and others recognized the importance of these streams and riparian areas as habitat for fish and wildlife. Nationally, these “natural” corridors were referred to as “greenways,” basically meaning linear open space. In California, use of the term greenways was not common; rather, these linear corridors were generally referred to as “urban creeks” or “urban creek corridors” or sometimes “parkways.”

Due to the nature of the problems impacting urban creeks, many urban creek restoration projects required extensive planning and multiple funding requests for various stages of the restoration including trash removal, sanitary engineering to resolve water quality issues, removal of invasive plants, erosion control, bank stabilization, in-stream aquatic habitat enhancement, and the installation of native plant species. Funding for some urban creek restoration projects became available in conjunction with other planning efforts: for example, flood control planning and park planning. Much of the work was also accomplished through the use of volunteers.

In 1984, the Urban Creeks Council was formed to present an alternative flood control option to the Corps of Engineers, which was planning to place a major section of Wildcat Creek (Alameda County) into a conventional dirt/riprap and concrete trapezoidal flood-control channel. The Council advocated for a multi-stage channel design with a bankfull channel, riparian corridor, floodplain, and berms (levees). This “natural-like” channel design for the Wildcat-San Pablo Creeks Flood Control Project (Fishbain and Williams 1988; Haltiner et al. 1996; Meyer 1989; Riley 1989a; Riley 1989b; Riley 1998; Riley 2003) was approved and constructed by the Corps.

The Urban Creeks Council was instrumental in getting numerous urban stream restoration projects implemented in the San Francisco Bay Area, the first project being Strawberry Creek Park (Berger 1990; Wolfe 1988), which involved the daylighting of a section of Strawberry Creek that had been underground in a culvert since 1904.

Many urban creek restoration projects were implemented in conjunction with the development of new residential subdivisions adjacent to deeply incised stream reaches. One of the earliest examples of the compound (multi-stage) channel approach was the Miller Creek Restoration Project (Haltiner et al. 1996; Yin and Pope-Daum 2004), wherein a low flow channel-floodplain system allowing some of the dynamics of a natural channel was constructed and then revegetated.

The CDWR Urban Stream Restoration Program began in 1985. Counties, cities, and non-profit organizations interested in improving the conditions of their watercourses were encouraged to submit grant proposals for restoration projects. Many of these projects involved the planting of riparian vegetation and the enhancement of existing riparian habitat in urban areas. An example of a CDWR-assisted (in other words, partial funding) stream restoration project with a significant riparian revegetation component is the Carmel River Biotechnical River Restoration (Cummings 1992; Haltiner et al. 1996; Matthews 1990).

The California State Coastal Conservancy was also actively involved in funding riparian habitat restoration in coastal areas especially when stream degradation was contributing to sedimentation of coastal lagoons. One such example is the Buena Vista Creek Restoration Project (Marcus 1987; Marcus 1988; Marcus 1989) to reduce sedimentation in Buena Vista Lagoon.

Many urban creek restoration projects involved work on only a limited portion or segment (sometimes referred to as a reach) of a stream. The Strawberry Creek Restoration Project (Berger 1990; Charbonneau and Resh 1992; Pollak 1990) and Strawberry Creek Management Plan/Program (Charbonneau and Rice 1989) conducted on the UC Berkeley campus and in its watershed are examples of urban creek restoration projects that included restoration work in the mid and upper watershed thereby reducing impacts on downstream reaches.

Locally-sponsored flood management projects evolved as alternatives to traditional flood control projects. These projects involved coming up with a greenbelt floodway design that provided flood protection but also provided for protection and restoration of riparian areas and public access to these "parkways." The First San Diego River Improvement Project (Burkhart 1989; City of San Diego 2001; Faber et al. 1989), a combination of flood control, natural area, and parkway, was initiated by private developers to allow commercial and residential developments in the Mission Valley to be approved by the City of San Diego.

The U.S. National Park Service River and Trail Conservation Assistance (RTCA) Program assisted local sponsors with the planning of urban creek restoration projects by bringing together diverse stakeholders and helping them find consensus solutions to restore degraded urban creeks. The RTCA published a book titled *How Greenways Work: A Handbook on Ecology* (Labaree 1992). The RTCA provided the impetus for the writing of *Ecology of Greenways: Design and Function of Linear Conservation Areas* (Smith and

Hellmund 1993), a book that bridges the gap between design and ecology. Publications like these were important since it was often landscape architects who were responsible for leading the design team for urban stream restoration projects.

Compensatory Mitigation Projects

Restoration of riparian habitat was undertaken as compensatory mitigation for riparian habitat loss due to unavoidable impacts resulting from infrastructure construction projects (for example, highway and bridge construction, flood control channel modifications, and utility corridors) and land development projects. These projects tended to be within, or adjacent to, urban areas where the stream corridors were often the only open space remaining. The major drivers behind these mitigation projects were NEPA and CEQA (both passed in 1970), the California Porter-Cologne Water Quality Control Act (1970), CCWA (1972), ESA (1973), CE Section 404 permitting, the California Coastal Act (1976), and CDFG stream alteration agreements. Often there was also a strong desire on the part of the community to develop public access to these "restored" stream corridors, sometimes leading to conflicts between regulators wanting the restored lands to be set aside for wildlife and park planners wanting to develop infrastructure (for example, pathways, lighting) for visitor use.

Some of the earliest projects involving native plant landscaping and native plant revegetation were installed by the Santa Clara Valley Water District (SCVWD) in the mid- and late-1970s. The District adopted a resolution in 1974 that set policy for the landscaping of District projects. In 1975, SCVWD published *A Landscaping Guide to Native and Naturalized Plants for Santa Clara County* (Stiles 1975). This guide contained pertinent information on characteristics and suitability of numerous plants native to the region, including species common to the stream systems in Santa Clara County. Dr. Bernie Goldner (1984) referred to projects installed in 1976 on Randol Creek, San Tomas Aquino Creek, Los Gatos Creek, and the Guadalupe River as "landscape projects" even though mostly native trees and shrubs were installed along these structurally modified channels. Various measures of structural bank protection were associated with these projects. Subsequent plantings of native vegetation installed in 1979 along Calabazas and Berryessa creeks were referred to as "revegetation projects" (Goldner 1984).

The fact that flood control managers were aware of the public's concern regarding the impact of flood control projects on riparian habitat at a local level is evidenced by the publication of *Valley Riparian Forests of California: An Overview of their Biological Significance and Physical/Chemical Processes* (Stiles 1978) by the SCVWD. In 1979, SCVWD adopted a new policy emphasizing mitigation of substantial adverse impacts, in conformance with CEQA (Goldner 1984).

There was a tendency for the early "revegetation" projects to end up appearing more like landscaping of structurally modified flood control channels than the creation of valuable fish or wildlife habitat. This was in part because there was an overreliance on engineered structures (for example, riprap, wire basket gabions) to prevent bank erosion and a resistance on the part of flood control maintenance personnel to allow the planting of riparian vegetation at the toe of the bank slope adjacent to the channel bottom. This prevented the establishment of shaded riverine aquatic (SRA) Cover.

Another reason why these "revegetation" projects often resembled landscaping is because many of these early projects were designed by landscape architects. Plantings were arranged for various visual purposes (for example, screening and aesthetics) without regard to the habitat requirements (for example, plant associations and vegetation structure) of riparian wildlife. Sometimes cultivars with showy appearance were installed instead of the native species or because of a failure to plan ahead for contract growing of native plant materials. Some common riparian plant species (for example, poison oak and stinging nettle) were almost always omitted because of undesirable characteristics (for example, poisonous and thorniness). Additionally, plantings were laid out in a linear fashion so as to be watered using drip irrigation.

Examples of "revegetation" projects installed along structurally protected flood control channels in the early and mid-1980s by the SCVWD include: Guadalupe River Revegetation Project (Goldner 1984); Alamitos Creek Revegetation Project (Berger 1990; Goldner 1988); Llagas Creek Watershed Mitigation (Berger 1990); and the Saratoga Creek Flood Control Project Revegetation (Berger 1990; Gray et al. 1984).

Up until the mid-1980s, almost all the SCVWD riparian revegetation projects were on the banks or immediate top-of-bank of flood control channels. The Lower Coyote Creek Pilot Revegetation Project (Berger 1990; Stanley et al. 1989), installed by the SCVWD in 1986-1987, was undertaken to determine the best means of revegetating historic floodplains that had been used for agriculture. Findings from revegetation on the 4-acre pilot project site were used for the selection of riparian species, propagule types, and planting, irrigation, and maintenance techniques on the remaining 28.5 acres of mitigation along lower Coyote Creek.

During the 1980s, other flood management agencies in the San Francisco Bay Region (for example, Alameda County Flood Control District, Marin County Flood Control District, and Sonoma County Water Agency) were also restoring riparian habitat as mitigation for flood control project impacts. Examples of these early revegetation projects include: Caldecott Park Creek Revegetation (Berger 1990); Novato Creek Flood Control Project (Berger 1990); and the Spring Creek Flood Control Project (Berger 1990).

Examples of early revegetation projects conducted as mitigations in the Central Valley in the 1980s include: California Department of Water Resources Sacramento River Maintenance Area 9 installed in 1981 (King 1985); and the Crescent Bypass Riparian Revegetation Project (Oldham and Valentine 1989; Oldham and Valentine 1990) installed between 1985 and 1988.

In southern California, Caltrans implemented a number of riparian revegetation projects as mitigation for impacts associated with highway construction. One of the earliest projects was the I-8 / I-15 Mitigation (Rieger 1988) constructed in 1982. Other Caltrans riparian revegetation projects in San Diego County included: Morena Street Site (Rieger 1988) installed in 1985; Sweetwater Bridge Mitigation site (Rieger 1988) installed in 1986-1987; and SR-52 Mitigation/Mission Trails Park (Rieger 1992) installed in 1989 and 1990.

Some mitigation projects involved significant alteration of the project site topography in order to create a planting bench or artificial floodplain with suitable flooding frequency and/or depth to groundwater to support riparian vegetation. Project designers were sometimes forced to take this option of converting upland areas to floodplain because of the no-net-loss of wetlands policy of the regulatory agencies. Caltrans projects involving significant lowering of the surface elevation include: I-8 / I-15 Mitigation in San Diego County (Rieger 1988); Sweetwater Bridge Mitigation (Rieger 1988) in San Diego

County; SR-52 Mitigation/Mission Trails Park (Rieger 1992) in San Diego County; and Hwy 85 Mitigation in Santa Clara County (National Research Council 2001). Frequently, the cost of the excavation and earth removal was applied to the highway budget and not the restoration project since the excavated material was need for fill for nearby highway construction.

Some mitigation projects created significant habitat for wildlife, especially migrant passerine bird species. Construction projects that created gaps in the riparian corridor (fragmentation) were often required to agree to mitigation acreage ratios of 2:1, 3:1, or greater, resulting in the planting of floodplain riparian habitat much wider than the remnant streamside vegetation that was impacted. For example, construction of a high flow bypass channel for the Lower Coyote Creek Flood Control Project necessitated the removal of approximately 15 percent of the existing riparian trees in the project area creating breaks in the riparian corridor. SCVWD was required to create 32.5 acres of new riparian habitat on the floodplain within the project levees.

Some revegetation projects focused mostly, or solely, on the mitigation of riparian habitat loss for special status species, especially Federally-listed endangered species. Caltrans Sweetwater Bridge Mitigation (Rieger 1988) and SR-52 Mitigation/Mission Trails Park sites were constructed primarily to provide habitat for the endangered least Bell's vireo. The planting regime at the Mission Trails Park site followed the prescription for least Bell's vireo habitat generated from extensive research conducted by Baird and Rieger (1989). Three pairs of least Bell's vireos nested at the Mission Trails Park site within a year of planting (Rieger 1992). The M&T Ranch Elderberry Mitigation Project (Stanley 1989) in Butte County and the Sacramento River-Mile 154.6 Right Project (Chainey et al. 1989) in Colusa County were installed to provide habitat for the endangered valley elderberry longhorn beetle.

In the latter part of the 1980s, we saw the creation of mitigation banks used by development projects that were unable to achieve on-site mitigation; for example, the San Joaquin Marsh Mitigation Bank (Stanley 1989) developed by The Irvine Company in Orange County in 1987-1988.

Habitat Conservation/Restoration Projects

A number of governmental agencies and conservation organizations saw an opportunity to restore large swaths (both in terms of length and width) of riparian habitat by purchasing available low-lying agricultural land within floodplains and creating wetland and riparian conservation areas. Much of this land was prone to periodic flooding and no longer profitable for farming. Typically, the objective was to create a mix of habitat types for a variety of bird species (waterfowl, waterbirds, and riparian dependent bird species).

Beginning in 1977, Bertin Anderson and Robert Ohmart undertook experimental revegetation projects for the reestablishment of cottonwood/willow forest on a number of sites along the Lower Colorado River, for the USDI Bureau of Reclamation (Ohmart et al. 1977). Their 1979-1980 Colorado River Dredge Spoil Revegetation (Anderson and Ohmart 1985a; Anderson and Ohmart 1985b) project sites totaled 74 acres, the largest being a 49-acre site on the Cibola National Wildlife Refuge. Interestingly, none of these plantings prospered—as of 2008—due to the modified soil and hydrology everywhere along this reach of the Colorado River. However, Anderson demonstrated hydrological needs and soil alkalinity levels for establishment of cottonwood and red willow on restoration sites.

The Army Corps of Engineers (CE) began revegetation at the Sepulveda Wildlife Reserve in the Los Angeles Basin in 1981. The CE planted 17 acres of riparian woodland in 1984 as part of the Sepulveda Wildlife Reserve Revegetation Project (Parra-Szijj 1990). Due to low survival rates of the riparian plantings, CE installed test plots in 1986 to demonstrate the need for irrigation and determine the best means of watering pole cuttings and seedlings, especially in light of the heavy growth of weeds.

In the early 1980s, The California Nature Conservancy (TNC) saw habitat restoration as a new tool for the conservation of natural areas. Successful habitat restoration would add acres of habitat for target wildlife species on nature preserves. Habitat restoration would change how preserve design would configure nature preserves (based on restoration potential) and allow more opportunity for process-based conservation of wildlife. This logic carried weight with private financial donors that supported TNC.

TNC's Kern River Preserve, situated along the south fork of the Kern River, was one of the earliest locations for experimentation for restoration technology in California. Cottonwood cuttings were planted as early as 1982; however, after initial poor results, TNC quickly realized the need for organized scientific testing of restoration methods—irrigation needs, weed control, and soil factors (texture, alkalinity, and water table depth). TNC hired Dr. Bertin Anderson of the Revegetation and Wildlife Management Center to develop the restoration technology.

TNC's Kern River Preserve Yellow-billed Cuckoo Habitat Enhancement Project (Anderson and Layman 1989; Reiner and Griggs 1989; Tollefson 2003) began in 1986 when Bertin Anderson implemented a 25-acre pilot project testing planting and irrigation methods for the establishment of cottonwoods and willows at the site. Anderson's quantitative approach allowed for the rapid development of methods that proved to be effective at establishing Fremont cottonwood and red willow woodlands required by the targeted species, yellow-billed cuckoo (YBC). From 1986-1989, 142 acres of cottonwoods and willows (four species) were planted at TNC's Kern River Preserve. YBC began using these stands within 2-3 years of growth of the trees. As of 2001, a total of over 330 acres of native riparian trees and shrubs had been planted on the higher floodplain surfaces (Tollefson 2003). In addition, "over 500 acres of native riparian forest have recovered at the Kern River Preserve through 'passive restoration,' by limiting or excluding livestock grazing in low-lying areas that had been converted to pasture through clearing and intensive grazing" (Tollefson 2003).

TNC became involved in the restoration of riparian habitat in the Sacramento Valley beginning in the late 1980s at TNC's Cosumnes River Preserve. TNC contracted with Harvey and Stanley Associates, Inc. (John Stanley and Harold Appleton) to assist TNC (Dr. Thomas Griggs) and Ducks Unlimited with the design and layout of the initial riparian revegetation at Cosumnes. The first phase of the Cosumnes River Preserve Riparian Restoration Project (Griggs et al. 1993) involved the planting of 10 acres of valley oak forest on fallow agricultural land in 1988. Tom Griggs supervised the initial restoration work at the Cosumnes River Preserve. TNC's Habitat Restoration Team directed volunteers in conducting plantings in each successive year. As of 2001, a total of 500 acres of oaks, willows, and other trees had been planted at the preserve by volunteers and school children (Swenson et al. 2003).

In early 1985, a levee protecting the farm field adjacent to the Cosumnes River Preserve failed. Cottonwoods and willows rapidly colonized about 15 acres of sediment deposited by the river on the farmland. Although the levee was repaired, the "accidental forest" was well established and through time provided habitat for a variety of wildlife species. This farm property was acquired by TNC in 1987. The rapidly growing "accidental forest" inspired TNC to explore how natural flooding processes could be enlisted to expand the riparian corridor (Swenson et al. 2003). In the mid-1990s, TNC reoriented its forest restoration program at the Cosumnes River Preserve to focus on areas where natural regeneration could be encouraged by reestablishing natural flooding. The Cosumnes River has close to a natural hydrograph since there are no major dams in the watershed—the entire Preserve area historically flooded in El Nino years, even with levees. In fall 1995, TNC intentionally breached the levee (created a 50-foot gap) and cut a shallow channel through the floodplain thereby reopening about 200 acres of bottomland to natural flooding. Natural "cuttings" of willow and cottonwood became established on the site (Mount et al. 2003; Swenson et al. 2003). The floods of 1997 caused many levee breaks along the Cosumnes River. The preserve and local farmers reached an agreement on an "unleveeing" project and convinced the CE to fund a nonstructural flood management project instead of traditional levee repairs. The project involved breaching and abandoning 5.5 miles of levees. Construction started in the fall of 1997 with the levee breaches and construction of a setback levee. This added about 100 acres to the floodway (Swenson et al. 2003).

After initiating the riparian habitat restoration at the Cosumnes River Preserve, Tom Griggs then moved on to plan the riparian plantings at Kopta Slough, Stony Creek, and the Sacramento River National Wildlife Refuge. "Cultivated restoration became necessary on the Sacramento River because Shasta Dam has altered natural hydrology, changing the patterns and extent of natural vegetation succession. Furthermore, mid to high floodplain soils are prone to support weedy herbaceous vegetation …which competes with native woody vegetation in natural and cultivated succession. TNC developed agricultural-style techniques to restore relatively large acreages of riparian vegetation in a logistically and financially efficient manner" (Silveira et al. 2003).

In 1989, TNC began planting riparian vegetation at the Kopta Slough Preserve (Griggs 1993; Griggs 1994). Kopta Slough Preserve was the R&D center for TNC for riparian restoration along the Sacramento River. The goal was to use all riparian plant species that are characteristic of riparian habitat and to study their survival and growth relative to a variety of factors: soil moisture, depth to water table, and irrigation water movement through the soil. Root growth rates and root architecture were studied by digging up selected plants. The first large-scale native grass plantings in riparian areas were undertaken at Kopta Slough. A total of about 300 acres were planted.

In 1991, TNC began riparian revegetation at the Stony Creek Preserve (Alpert et al. 1999; Griggs and Petersen 1997; Reiner and Griggs 1989). Stony Creek Preserve was the first site purchased by TNC with the intention of transferring property ownership to the USFWS after it was restored, which occurred after 1995. Restoration technology was further refined at Stony Creek as the soils were more variable and irrigation timing and amounts were refined for most species that were planted. Implementation staff was of 6-month interns (mostly recent college graduates). Time and costs of inexperienced implementation became obvious. Hiring of a professional field manager with experienced laborers was decided for future projects. Approximately 500 acres were planted.

TNC began work at the Llano Seco Unit of the Sacramento River National Wildlife Refuge in 1991 (Griggs and Golet 2002; Silveira et al. 2003). Established by the U.S. Congress in 1987, the Sacramento River National Wildlife Refuge consists of 28 Units in Tehama, Glenn, Butte, and Colusa Counties comprising 10,353 acres within the 100-year floodplain (recent mixed alluvium and gravel bars/sandbars) along 81-river miles of the middle Sacramento River from Red Bluff to below Princeton (J. Silveira 2016, personal communication). The USFWS obtained these fee-title properties from landowners who were willing to sell existing riparian forest and flood-prone agricultural fields adjacent to the forests. There are roughly 4,581 acres of remnant riparian habitats on the refuge. TNC established a cooperative management agreement with the USFWS that allowed TNC to restore former farm fields. "Propagules from indigenous plants and local ecotypes are being used in large-scale restoration layouts of various designs associated with site-specific hydrologic and edaphic conditions" (Silveira et al. 2003).

As of summer 2015, 5,033 acres of former flood-prone agricultural lands (primarily walnut, almond, and prune orchards, but in some cases row crops) have been restored to various riparian and floodplain vegetation types (J. Silveira 2016, personal communication).

Rangeland Restoration Projects

The planting of riparian vegetation for gully control in mountain meadows has a long history. Kraebel and Pillsbury (1934) published a handbook for erosion control in mountain meadows in the Sierra Nevada for the USDA Forest Service in which they included specifications for the selection and planting of willow cuttings and the construction of willow wattles.

There were a number of projects in the 1980s to restore wet meadows and their associated streamside riparian buffer strips in eastern California. Project work typically involved the installation of erosion control devices, typically rock, gabion, or fabric grade-control structures (in other words, check dams), across eroded stream channels in high mountain meadows to trap sediment and raise the water table in the meadow. These projects were generally accompanied by the installation of temporary or permanent fencing to control livestock use, a reduced stocking level, or a revised grazing regime (for example, season of use). Many of these projects were planned using the Coordinated Resource Management and Planning (CRMP) Process because they involved work on both public and private land, including public land grazing allotments.

Sample rangeland riparian restoration projects include: Willow Creek Restoration (Clay 1984) in Modoc County and the Clark Canyon Riparian Demonstration Area (Key 1987; Key and Gish 1989) in Mono County.

This work was similar in approach to the demonstration projects on Bear Creek (Elmore and Beschta 1987) and Camp Creek (Elmore and Beschta 1987; Winegar 1977), conducted by Wayne Elmore in the Prineville area of southeastern Oregon (Crook County) during the 1960s and 1970s. However, Wayne Elmore's emphasis was less on the installation of structures and more on the improved management of rangeland riparian areas (Elmore and Beschta 1989).

Invasive Plant Removal/Control

The removal or control of invasive nonnative plants was often an initial action at riparian restoration sites. Thus, most of the projects mentioned above and described in table 7 had some element of invasive plant eradication or control. Some riparian restoration projects only involved the removal of invasive plant species. In these cases, it was assumed that native riparian plants would reestablish on the site(s) after competition for light, nutrients, and water was eliminated. Some of these projects were followed by plantings of riparian plants after the invasive species had been removed entirely or were under control.

Many modern-day strategies for invasive plant control were developed during the latter part of the 20th century. The Nature Conservancy played a significant role in sharing information on control techniques with the preparation of Element Stewardship Abstracts initiated in the late 1980s and early 1990s. Formed in 1992, the California Invasive Plant Council (Cal-IPC) contributed to the development and exchange of information on the eradication of invasive plants in riparian corridors through its symposia, publications, and sponsored research and trainings. Bossard et al. (2000) provides technical information on physical, biological, and chemical control measures for most of the invasive plants that occur in California's riparian corridors.

There are many nonnative invasive plant species known to occur in California's riparian corridors and these were generally dealt with on a project by project basis. Of these, two species in particular had virtually overtaken many stream corridors: giant reed and tamarisk. Below, we mention only four projects of the literally hundreds, if not thousands, of invasive plant removal projects that occurred between 1970 and 2000.

Giant Reed

Giant Reed (also referred to as *Arundo* and false bamboo) was recognized early on as a threat to the success of riparian restoration projects, not only because it often occurred on restoration project sites, which could be dealt with during site preparation and site maintenance, but also because flooding is the primary mechanism of dispersal of stems and rhizome fragments (Rieger and Kreager 1989). In the 1980s and 1990s, giant reed control projects were undertaken in river drainages in many parts of the State including the southern California coast, the central coast, the San Joaquin and Sacramento valleys, and even the north coast. Most restoration projects dealt with a specific reach of a stream and in many cases funding and/or authorization was not available to address upstream infestations of *Arundo* in the watershed. However, there have been some efforts at watershed-wide invasive plant removal. In northern California, in the Russian River Watershed (Gaffney and Gledhill 2003), community-based organizations have worked in collaboration with agencies, landowners, and community members since 1992 to identify and map invaded sites, conduct experimental and demonstration projects, remove giant reed, restore native habitat, and conduct education and outreach programs. In southern California, a large-scale effort called the Santa Ana River Watershed *Arundo* Habitat Management Program has been underway since 1997. Its purpose has been to to rid an entire watershed (largest drainage in coastal southern California) of *Arundo* and restore riparian areas (SAWPA 2016). Native riparian habitat has expanded into at least 60 percent of the reclaimed floodplain, providing valuable habitat for birds such as the endangered least Bell's vireo (Zembal and Hoffman 2007).

Tamarisk

Removal or control of tamarisk or saltcedar and related species was one of the biggest challenges in the southeastern part of California, although it also had to be dealt with elsewhere in southern and central California. Saltcedar control in the Southwest had been attempted by various agencies and organizations since the 1950s (Rodman 1990), although not necessarily for the restoration of native ecosystems. TNC began removing tamarisk on the Coachella Valley Preserve in 1986 to restore native desert fan palm oases and associated riparian species through the Thousand Palms Canyon Tamarisk Control Project (Barrows 1993).

Numerous saltcedar removal projects were undertaken along rivers in southern California in the 1980s and 1990s. Some of these projects involved seeding or planting with native riparian species while others relied on "natural" recovery from nearby seed sources or the seed bank. Bay and Sher (2008) evaluated the success of "active revegetation" (with no irrigation) after Tamarix removal in riparian systems in the Southwest including sites along the Lower Colorado River in California. An example of a large-scale effort to remove saltcedar and restore riparian vegetation is the Afton Canyon Riparian Restoration Project (BLM 2015) on the Mojave River begun by the Bureau of Land Management in 1992.

Part 3—Research Associated With Restoration Projects

Various types of research, experimentation, field investigations, and monitoring programs were associated with early riparian habitat restoration projects. These programs addressed the following types of issues:

- Habitat requirements of target wildlife species
- Planting and irrigation techniques
- Plant survival and growth
- Plant, soil, and water relationships
- Competition from weeds and weed management techniques
- Wildlife usage of revegetated areas
- Proper functioning condition

Information on most of the projects used as examples below is presented in table 7.

Habitat Requirements of Target Wildlife Species

Significant amounts of data were often collected prior to the design of riparian revegetation projects, especially those projects intended to create habitat for special status species.

Lower Colorado River

In 1973, Bertin Anderson, Robert Ohmart, and John Discano began conducting studies of riparian vegetation-wildlife interactions on about 198,000 acres of riparian vegetation along the lower reaches of the Colorado River in an attempt to discover the vegetative characteristics to which birds were responding (Anderson and Ohmart 1977; Anderson and Ohmart 1979; Anderson et al. 1979). They developed a model for revegetating riparian areas from their 5-year database on vegetative-wildlife interactions (Anderson et al. 1979). In 1977, after clearing saltcedar from an area along the Lower

Colorado River, a revegetation design was developed for the site based on this model. While some palo verde volunteered on the site, willows, cottonwoods, and honey mesquite were planted in June-July of 1977. They monitored growth rates, root growth, plant mortality, and other factors for the first year and presumably longer. Anderson and Ohmart (1985a) later refined their modeling based on 7 years of data (1972-1979).

Yellow-billed Cuckoo Habitat at the Kern River Preserve

Anderson and Layman (1989) based the design of yellow-billed cuckoo habitat at the Kern River Preserve on data of cuckoo habitat requirements along the Colorado River from 1976-1983 (Anderson and Ohmart 1984) and on data collected along the South Fork Kern River from 1985-1988.

Least Bell's Vireo Habitat in San Diego County

In 1986, Hendricks and Rieger (1989) analyzed data of least Bell's vireo nesting sites on the Sweetwater, San Diego, and San Luis Rey rivers in San Diego County representing approximately 10 percent of the known species population. A variety of parameters at each nest site were measured to characterize the nesting habitat of the least Bell's vireo for the design of future restoration projects. Baird and Rieger (1989) used this baseline vegetation and habitat data for 30 nesting sites on the three rivers to develop a habitat restoration model for the creation of least Bell's vireo nesting habitat at Caltrans mitigation sites in San Diego County (Baird 1989).

Note: Chapter 8 (Conservation Recommendations) of the *Riparian Bird Conservation Plan* (RHJV 2004) provides recommendations pertaining to the design of riparian restoration projects for riparian dependent bird species.

Planting and Irrigation Techniques

Lower Colorado River

In 1979 and 1980, Bertin Anderson studied the effects of deep tillage (augered holes to various depths) combined with irrigation (daily irrigation discontinued after variable lengths of time) on the growth and survival of rooted cuttings of Fremont cottonwoods on 74 acres of dredge-spoil sites along the Lower Colorado River (Anderson 1989; Anderson et al. 1984). In 1981, Disano, Anderson, and Ohmart described the types of irrigation systems they used for riparian zone revegetation along the Lower Colorado River (Disano et al. 1984).

Kern River Preserve

Beginning in 1986-1987 at TNC's Kern River Preserve, Dr. Bertin Anderson conducted research on ways in which the physical characteristics of the site (soil salinity, soil texture, depth to groundwater) and the type and height of saplings affected plant survival and growth. He evaluated various methods for the propagation of cottonwoods and willows. He also evaluated plant survival and growth with various irrigation regimes at various depths of tillage (augered and then backfilled holes) with respect to the water table. In addition, he studied the effects of competition from weeds and the effects of browsing by wildlife and livestock (Anderson 1989; Anderson and Layman 1989).

Lower Coyote Creek

At the 4-acre Coyote Creek Pilot Revegetation Project site installed in 1986-1987, John Stanley, Larry Silva, Harold Appleton, William Lapaz, and others conducted a

3-year study on the effects of two different types of irrigation (overhead and flood) on the survival and growth of 3,640 plants comprised of 15 native plant species and multiple types of plant materials (propagule types) for each species. (Stanley et al. 1989)

Plant Protection

Frank Chan of PG&E developed the collar and screen plant shelter in the early 1980s. This device was used at the Lost Canyon Rehabilitation Project (Chan and Wong 1989). The collar (initially a cottage cheese container with the bottom cut out inserted into the ground) collected and concentrated precipitation into the root zone of the seedling and helped to deter gophers while the wire screen (tied to the collar) prevented damage from deer, rabbits, and insects. This device was used for direct seeding (for example, acorns) on numerous restoration project sites including at Coyote Creek and Cosumnes River.

Plant Installation

In the mid-1980s, Jonathan Oldham and Bradley Valentine of the Kings River Conservation District developed the "hydrodriller" for the planting of woody cuttings on streambanks at the Crescent Bypass Riparian Revegetation Project. This device was similar to the "waterjet stinger" described by the USDA Natural Resource Conservation Service (Hoag et al. 2001).

Translocation of Live Vegetation

Sutter et al. (1989) studied the survival of transplanted mature elderberry shrubs. The elderberries were transplanted as mitigation for the loss of habitat for the valley elderberry longhorn beetle along the Sacramento and American rivers.

John Rieger used a tree spade to translocate more than 2,000 cottonwood and willow trees that were being removed a mile upstream from the Mission Trails site for a previously approved development. The trees were installed at the Caltrans Mission Trails site and monitored for 5 years. Less than one-half of 1 percent mortality was observed (Rieger et al. 2014).

Application of Agricultural Methods

In 1989, the California Nature Conservancy gained management of the 700-acre Kopta Slough Preserve, which is owned by the Controller's Trust of the State of California. The Kopta property had been a productive almond orchard until the flooding of 1986 that killed over 60 percent of the orchard trees by drowning. The Nature Conservancy's goal was to demonstrate the feasibility of implementing large-scale riparian restoration by planting a minimum of 100 acres in one year. The farmer leasing the agricultural land at Kopta was hired to provide advice on farming technology. Only large acreage restoration would make a difference to target wildlife populations along the Sacramento River (Griggs 1993). Achieving large acreage restoration goals would require the use of modern agricultural technology and equipment. This resulted in the "farming" of native trees and shrubs for the first 3 years of growth to ensure their establishment, once irrigation and weed control were halted.

Plant Survival and Growth

Lower Colorado River

Bertin Anderson (Anderson et al. 1979) determined growth and mortality rates for palo verde, Goodding willow, cottonwood, and honey mesquite planted along the Lower Colorado River in 1977. In 1979 and 1980, Anderson (Anderson and Ohmart 1979; Anderson et al. 1984) measured the growth of rooted cuttings of Fremont cottonwoods planted in holes augered to various depths. Trees were measured (height and crown) three times during the growing season. Trees were irrigated daily with irrigation discontinued after varying lengths of time. Evaluations were made of the effects of deep tillage on tree growth and survival. The effects of differing periods of irrigation on growth and survival were also evaluated.

Lower Coyote Creek Pilot Revegetation Project

Each of the 3,640 plants installed at the 4-acre pilot revegetation site was monitored annually for the first three years (1987-1989). Assessments included survival, growth, vigor, and damage. A vegetation sampling program was employed to document semiannual changes in canopy height, canopy cover, foliage density, and diversity at different heights within the canopy and herbaceous cover. (Stanley et al. 1989)

First San Diego River Improvement Project (FISDRIP)

Monitoring at the 26.8-acre riparian revegetation component of FISDRIP involved the collection of data for comparison with the milestone performance standards that were established for the project (Burkhart 1989). Both quadrats and transects were used to evaluate the development of the riparian woodland and groundcover. Photo-documentation was an important component of the monitoring program.

TNC Sacramento Valley Project

Implementation monitoring involved daily evaluations of irrigation and weed control needs by the field manager. At the end of the growing season either a complete census, or a very focused sampling at preselected locations, was carried out at each project site. Survival and height growth were measured for each species. These data informed subsequent designs as to soil-plant placements.

Plant, Soil, and Water Relationships

Sacramento River National Wildlife Refuge

Lands purchased for the Sacramento River NWR were adjacent to the river channel and still flooded periodically. Channel deposits of sand and gravel were common on these lands, interspersed with the more productive floodplain deposits of fine sand and silt. In the first years, which species could grow on which soil texture was unknown. Likewise, the depth to water table affected the rooting behavior of each species differently. As irrigation was applied, careful monitoring was carried out of soil moisture in the soil profile by electrical moisture probes that had been placed at known depths previously. After 2 years of growth, backhoe pits were excavated to expose the root systems of selected saplings. Root system architecture and depth of development were mapped for each species, thereby informing future planning of design on sites with variable soils. For example, cottonwood “aggressively” grows roots deep to find the water table. Backhoe pits revealed abundant cottonwood roots down to 15 feet below the surface in less than 2 years; arroyo willow growing nearby had rooted only to 5 feet, given identical irrigation. Drought adaptations of native trees and shrubs was virtually unknown. Monitoring of growth on different soil textures and depths to water table informed future design based upon soil characteristics.

Competition From Weeds and Weed Management Techniques

Kopta Slough Preserve

Early on at Kopta Slough Preserve, TNC discovered what every farmer knows: weeds are not compatible with target plant growth. Even a "few weeds" are not conducive to optimum growth of the target species. Thus, aggressive weed control—herbicides, mowing, disking—throughout the season was required to achieve restoration and horticultural success. Lessons learned from several years of implementation were that herbicides are necessary only during the first year, with frequent mowing being sufficient for optimum plant growth during years 2 and 3. If herbaceous understory is to be planted, then herbicides or disking of the aisle between woody plants must be carried out each year to inhibit the production of weed seeds.

Wildlife Use of Revegetated Areas

A number of investigators implemented monitoring programs to document wildlife (especially avian) use of revegetation sites. Many of these programs also monitored avian use of mature riparian forest adjacent to, or nearby, revegetation sites.

Lower Colorado River Riparian Revegetation

Bertin Anderson, William Hunter and Robert Ohmart monitored avian use at three revegetation sites ranging in size from 25-74 acres along the Lower Colorado River from 1977-1984 (Anderson and Ohmart 1984; Anderson et al. 1979; Anderson et al. 1989)

Kern River Preserve Yellow-Billed Cuckoo Habitat Enhancement

William Humber, Bertin Anderson and Reed Tollefson censused birds utilizing naturally occurring cottonwood-willow habitats on TNC's preserve and on one 25-acre revegetation site implemented in 1986 and on two 25-acre revegetation sites implemented in 1987 (Hunter et al. 1989). Note: Appendix B of the *Riparian Bird Conservation Plan* (RHJV 2004) provides documentation on how birds responded to riparian restoration at the Kern River Preserve.

Lower Coyote Creek Pilot Revegetation Project

Based on a Fish and Wildlife Coordination Act Report issued by the U.S. Fish and Wildlife Service, the CE 404 permit for the Lower Coyote Creek Flood Control Project required the Santa Clara Valley Water District to monitor wildlife use of at least one riparian revegetation site for a minimum of 10 consecutive years after initial planting and then at year 15 and every 10 years thereafter for the life of the project[17]. In 1986, the Coyote Creek Riparian Station expanded a pre-existing bird banding program at lower Coyote Creek to include the 4.4 acre Lower Coyote Creek Pilot Revegetation Project site, which was installed in December 1986. In addition to the bird banding program, variable-radius circular plots were established in each of the three habitats (existing riparian corridor, newly installed pilot revegetation site, and the ruderal overflow channel landward of the revegetation site). A breeding bird census was also conducted in each of

[17] This permit requirement was later modified.

the three study habitats from March through July of each year. Other monitoring included mammal, reptile, and amphibian sampling and vegetation sampling within each of the 13 variable-radius circular plots. (Rigney et al. 1989)

Mission Trails Mitigation Project

Caltrans monitored the SR-52 (Mission Trails) Mitigation Site in San Diego for compliance with Section 404 Permit requirements. Vegetation composition and structure was monitored on the mitigation site to determine conformity with the Least Bell's Vireo Habitat Restoration Model (Baird and Rieger 1989). Bird populations were monitored to determine presence or absence of least Bell's vireos and nesting success. Success was defined as either a vireo pair nesting on site or no statistically significant differences between parameters on the mitigation site and those in functioning the vireo's habitat (Hendricks and Rieger 1989). Least Bell's vireos successfully nested on the site within 1 year with three territories and in subsequent years several other pairs nested in the remaining areas of the site. In addition, the vegetative parameters established in the habitat model were met for most of the monitored "cells" within the 5-year monitoring period.

Sacramento River National Wildlife Refuge

The Nature Conservancy hired Point Blue Conservation Science (PBCS, formerly PRBO) to monitor bird use of restoration plantings. Over 11 years of data were collected to show species use of restoration plantings at different times after planting (Golet et al. 2008). PBCS data also showed vegetation structural trends (tree-shrub ratios and densities) that affected bird diversity. These results informed future planting designs. Golet et al. (2003) reported on songbird use within the Sacramento River Project Area (over 100 river miles from Red Bluff to Colusa) including lands within the Sacramento River National Wildlife Refuge. At horticultural restoration sites, riparian bird diversity increased significantly over time as the revegetation sites matured.

Proper Functioning Condition

Afton Canyon Riparian Restoration Project

BLM conducted project monitoring using photoplot ground/canopy cover analysis and cross-sectional riparian plant frequency/cover trend analysis. BLM relied heavily on the use of the qualitative evaluation process referred to as Proper Functioning Condition Assessment conducted by an interdisciplinary team of specialists (BLM 2015).

Part 4 – Concluding Remarks

Progress in our understanding of riparian ecology and how to go about restoring riparian ecosystems proceeded at a rapid pace between 1970 and 2000. Many scientists, conservationists, land and resource managers, and volunteers contributed to the development of this field. Unfortunately, it was not possible to recognize the contributions made by many of the individuals involved. Between the 1970s and 2000s, some of this knowledge was recorded in the literature. However, since many restoration practitioners had little time and funding to publish their findings, many of the details of how restoration projects were conducted, their successes and failures, and lessons learned remain buried in project plans, specifications, monitoring reports, and unpublished final project reports. We offer our apology to anyone, and for any projects, we have overlooked in this chapter.

The authors hope that this overview provides those involved in promoting riparian conservation in the 21st century with an informative historical perspective regarding the evolution of the field of riparian ecosystem restoration in California. For those of our peers who played a role in the protection and restoration of riparian ecosystems in the 20th century, we hope that this read provided a pleasant trip down memory lane and reminded you that your contribution was worth the effort.

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Chapter 8. Sacramento-San Joaquin System

F. Thomas Griggs and Stefan Lorenzato

Introduction

The Great Central Valley of California occupies 22,500 square miles (58,000 square kilometers) in the interior of northern and central California. At the time of the Gold Rush in 1849, nearly 1 million acres (1,600 square miles, 4,000 square kilometers) of riparian vegetation covered the Central Valley floor along with approximately an equal area of wetlands. The riparian area flourished in the large river basins and along river channels (Katibah 1984; Thompson 1961). The Central Valley is partly defined by the Sacramento River in the north, the San Joaquin River in the south, and the Delta where the two rivers meet and turn westward toward San Francisco Bay (figs. 27a and 27b). The valley is made up of a series of basins connected by the rivers, which form a distributary floodway. Before development, heavy winter and spring runoff would flow out of the river channels and drain to the basins until waters were deep enough to continue their flow to the Delta. As flows subsided, water would sit in the basins until evaporated or it seeped into the ground. The wetland and riparian lands were nourished by these flows and extended across the low-lying areas in the valley trough and basin sinks. The land surface consisted of shallow undulating ridges and swales, creating complex soil-water-plant interactions that provided a great diversity of hydrology, vegetation, water depth and velocities, and timing. The result was a rich and dynamic system that dependably provided a mix of microhabitats and physical features (Kelley 1989; Thompson 1961).

These features that provided such a diversified ecology hindered agriculture. As cropping systems expanded to take advantage of the rich soils, riparian forests were cut down, land leveled for ease of production, and waterways rerouted. Today, the Great Central Valley of California grows over 200 crops and generates over 32 billion dollars annually in agricultural revenue (CDFA 2015), primarily due to its unique, and highly developed, hydrology and the complex patterns of alluvial soils that support diverse agricultural practices. But only about 2 percent to 4 percent of the riparian habitats remain (California Department of Fish and Wildlife's VegCamp Program 2011).

Central Valley River Basins and Forces of Change

The Great Central Valley is comprised of a set of linked basins (table 9). The basins' edges are marked by low rises in the land surface that control drainage patterns of overland flow. The rivers of the Central Valley flow through the basins in relatively small channels from the foothill watersheds to the Delta where north and south flows combine and head west into San Pablo and San Francisco Bays (figs. 27a and 27b).

Historically, flood flows in the valley were often flashy, flood peaks emerging from the upper watersheds in high volumes over short periods. The broad, flat basins of the valley floor readily absorbed the flows that escaped the channels. With these flashy flows came sediments that similarly were carried away from the channels and deposited in the basins. The resulting mosaic of soils and residual moisture provide large expanses of land

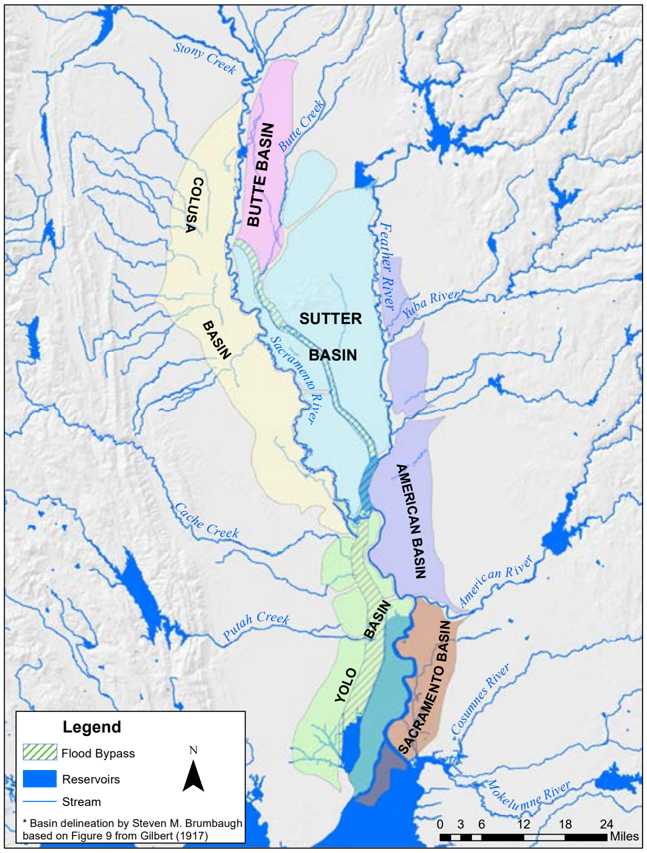

Figure 27a—Sacramento Historic Basins. All figures in this chapter are by Kevin Coulton, P.E. and CFM , Seungjin Baek, Ph.D., P.E., cbec eco engineering, as part of a USEPA Wetlands Development Grant to the California Department of Water Resources (grant CD-00T83701).

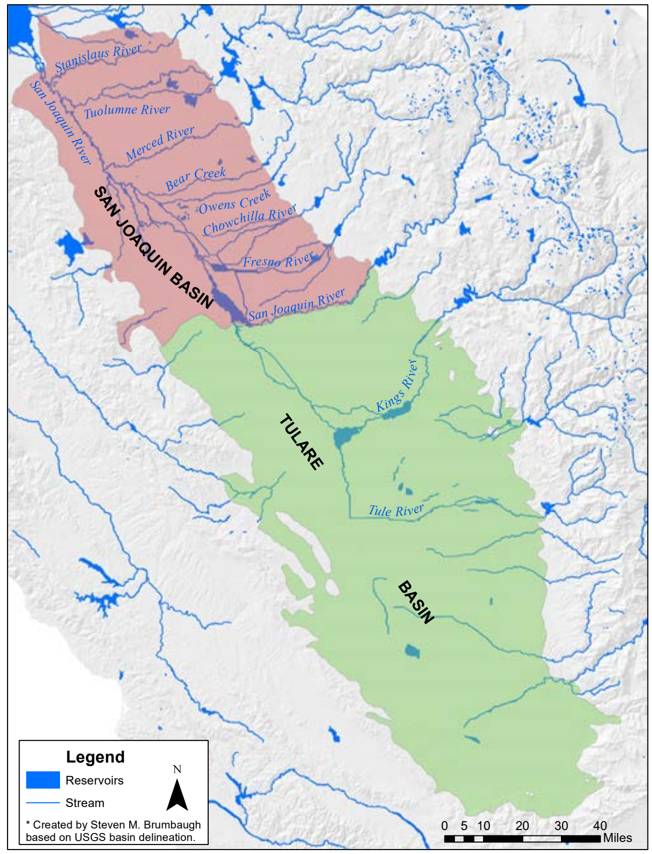

Figure 27b—San Joaquin and Tulare Basins.

Table 9—Hydrologic distributary basins of the Central Valley of California and their areas.

Basin	Approximate acres
Butte Basin	99,877
Colusa Basin	318,225
Yolo Basin	178,268
Sutter Basin	368,697
Sacramento Basin	95,545
American Basin	246,402
Lower San Joaquin	104,500
Middle San Joaquin	370,000
Upper San Joaquin	484,000
Tulare	540,000

where flood-adapted plants flourished and riparian forests established. The interaction of vegetation with the flood flows further influenced flow patterns and an even finer scale of microhabitats developed, supporting a great diversity of plants and animals (Scott and Marquiss 1984). Until the Gold Rush, people generally adapted to the patterns of flood and seasonal growth, exploiting the diversity and relying on natural cycles to produce the materials and food needed for their communities (Kelley 1989).

With the Gold Rush (1848-1855) came disturbance of land in the middle and upper watersheds and a tremendous amount—1.5 billion cubic feet (Mount 1995)—of soil, sediments, and rocks found their way into the streams and rivers from the mining operations. At the same time, agriculture was being established in the valley and farmers sought out the richest soils, which tended to be where the riparian forests stood. Farmers rapidly deforested the valley floors and carved canal systems into the land to move irrigation water diverted from the rivers, during summer low flows, into their fields.

A conflict rapidly emerged between valley farmers and miners over the sediments in the rivers and streams. Because the mining accelerated the erosion and transport of coarse sediment in the form of sand, gravel, and cobbles, when floods occurred, these sediments were deposited on fields that previously were only comprised of premier farm soils. The coarse sediments devalued the land and made farming difficult or, in some cases, not possible (James and Singer 2008).

The controversy was taken into court and in 1884 Judge Lorenzo Sawyer put a stop to the discharge of mine tailings into streams (Federal Reporter Vol 18, 1884; Mount 1995). While this stopped the input of new coarse sediment, by this time millions of tons of debris were already loaded into streams and on their way down to the valley. Farmers seeking to stop flooding of their land in order to reap more crop production also wanted to stop the deposition of sediments on their fields. Initially, the solution was for individuals to build berms and levees, but the haphazard product did not resolve the issue (Kelley 1989). Ultimately, in the early 1900s, a unified system of levees and floodways was designed to provide farmers with more predictable and manageable flood flows and to control and move sediment through the system (fig. 28).

While the farm interests were well served by the resulting flood protection systems, the riparian habitats of the Great Central Valley were rapidly succumbing to agricultural demand. The diversity and abundance of wildlife and plants that once graced the valley began to shrink and the dynamics of the ecosystem began to change (Grinnell and Miller 1944).

Another change force that emerged was the need for additional summer water for agriculture. By the mid-1900s, dams had become a common element of the landscape. With the large dam building era in full swing, major dams were placed on all the major rivers that drain into the Central Valley. The large dams created three very distinct alterations to the ecology of the Central Valley: (1) they restructured the pattern of flow, greatly limiting the medium sized peak winter and spring flows while increasing late spring and summer flows, but without significant peaks (Kondolf 1997); (2) they dramatically increased the acreage of land under irrigation; and (3) they cut off access by anadromous fish to their natal streams and the floodplains they used for rearing grounds (Sommer et al. 2003).

The plan for fish was to replace the loss of fish from lack of spawning habitat with hatchery bred fish. The result was a steady decline in populations of migrating fish. With this decline came a disruption of the nutrient cycles associated with fish returning from

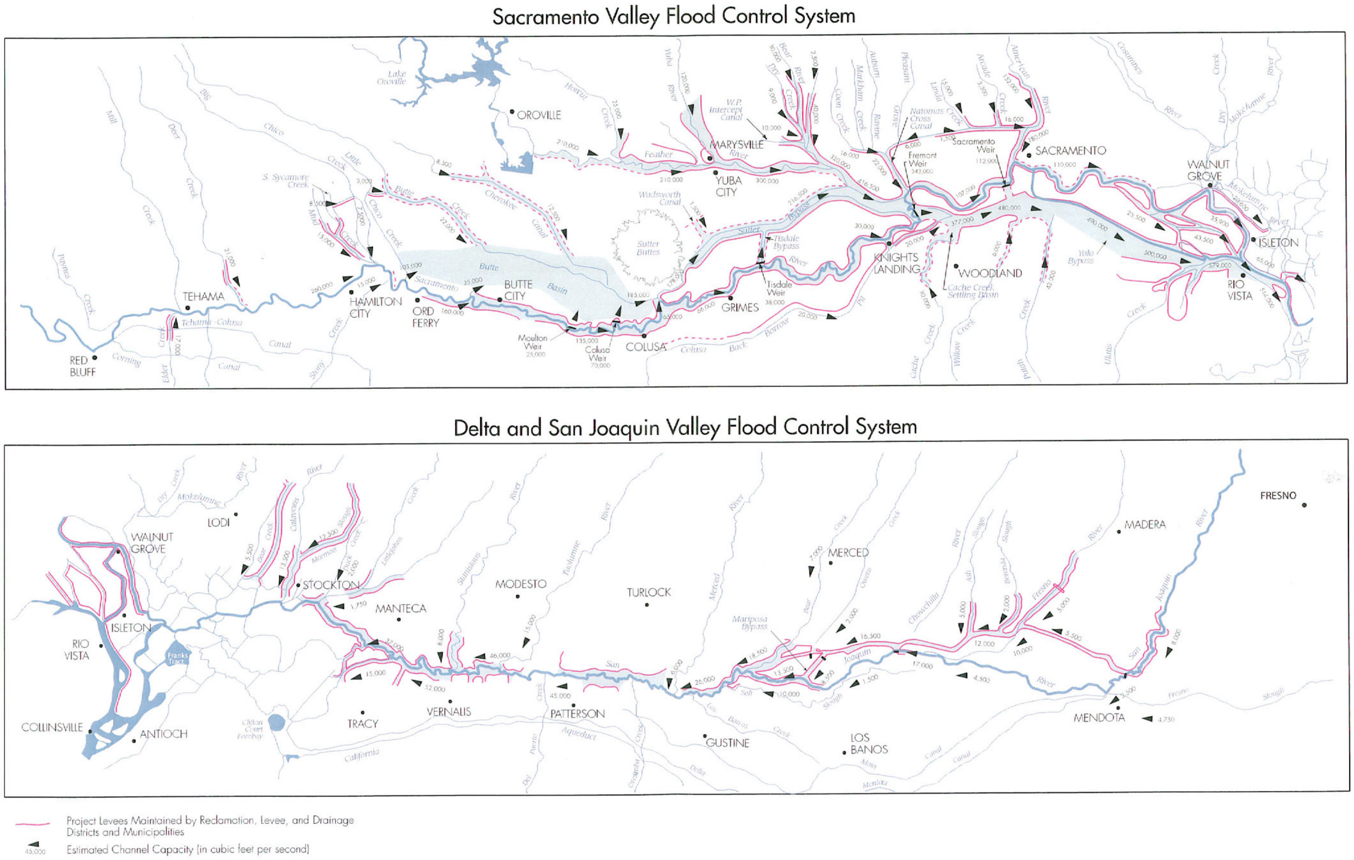

Figure 28—Sacramento and San Joaquin Valley Flood Control Systems.

the ocean and resulted in further reduction in riparian productivity (Merz and Moyle 2006). With the flow patterns disrupted, plants and animals that had evolved to be in synchronization with the natural cycles were left to find limited room to persist at the edges of the remaining floodplain resource. Of the nearly 1 million acres of riparian lands estimated to exist in the Sacramento Valley before the Gold Rush (Katibah 1984), by 2011 only 94,000 acres remained (VegCamp 2011).

What We Currently Face

The Central Valley is over 450 miles long, from its northern end at Redding to near its southern end at Fort Tejon (south of Bakersfield). The Sacramento River drains the northern portion of the Central Valley (27,500 square miles watershed), including the southern Cascade Mountains and northern Sierra Nevada (fig. 27a). Its primary tributary is the Feather River (6,000 square mile of watershed). The Sacramento empties into the Delta estuary between the City of Sacramento and San Francisco Bay. The southern portion of the Central Valley is drained by the San Joaquin River (31,800 square miles of watershed) and its larger tributaries—the Mokelumne, Cosumnes, Stanislaus, Tuolumne, and Merced rivers (fig. 27b). South of Fresno is the Tulare basin, home of historical Tulare Lake, once the largest freshwater lake west of the Mississippi River. The Tulare basin receives flows from the Kings, Tule, and Kern Rivers from the southern Sierra Nevada and overflows into the San Joaquin Basin southwest of Fresno. The San Joaquin River flows northward and empties into the Delta estuary from the south. The Delta is the single point of outflow for the entire valley.

Flow Patterns

Dams now control river flows on all these rivers. These dams serve both water supply and flood control purposes. The dams were conceived with the goal of providing water for agriculture and for urban uses. Two large water projects dominate the water conveyance systems: the Federal Central Valley Project (which has storage capacity of 13 million acre feet [maf] that delivers about 7.4 maf annually); and the California Water Project (30 dams, 20 reservoirs, 700 miles of canals, and storage capacity of 5.4 maf) (California DWR Bulletin 132-14, 2015). Together these projects serve over 3 million acres of irrigated land and supply over 25 million people with at least a portion of their water.

In an average year, the unimpaired runoff for the Sacramento Valley is about 18.2 maf and the San Joaquin Valley about 6 maf (based on SRR and SJR unimpaired flow estimates—DWR Bulletin 132-14). For the Tulare basin, which only flows into the San Joaquin in the wettest years, the average unimpaired runoff is 3 maf. Diversions dramatically change the timing of the water flowing through the river system. And much of the water release from the dams does not reach the Delta to become outflow. The 8 River index, a measure of water released from the Sierra Nevada watersheds compared with Delta Outflow (as estimated by the Day Flow model) (see CDWR web site https://water.ca.gov/Programs/Environmental-Services/Water-Quality-Monitoring-And-Assessment/RTDF-Summary) shows substantial losses as water moves through the system. For example, in 1993, the 8 River index showed about 19 maf of water entering the Central Valley but only about 5 maf leaving the Valley through the Delta).

Note that these models produce coarse estimates; actual amounts are likely higher. These model comparisons also indicate how context-specific the flows are. In drier years, the gap between inflow and outflow tends to be larger than in in wetter years. In consecutive wet years, the difference is even smaller (presumably due to lack of demand), and in consecutive dry years the differences again tend to be small (mainly due to lack of water supply). For riparian areas this means that in all but the wettest years there exists fierce competition for water needed to support riparian functions.

Dam building started in the 1800s and accelerated through the 1950s. By the early 1960s, dams had been built on the mainstems of the Sacramento and San Joaquin rivers and most tributaries before they entered the Central Valley from the mountains. Table 10 shows the amount of storage in several of the larger reservoirs created by these dams (Kondolf and Batalla 2005). Of particular interest, note the amount of storage on each river as a percent of average annual runoff for that stream. For example, reservoirs on the Stanislaus River are able to trap 2.94 times the average annual runoff; on the Tuolumne and Calaveras Rivers nearly twice the average annual runoff is contained in reservoirs. In contrast, note that Shasta reservoir can only capture 74 percent of the Sacramento's average annual runoff. Despite the flashiness of runoff in California, the large storage capacity in the San Joaquin reservoirs greatly diminishes the frequency of large floods and nearly eliminates the occurrence of modest floods on these rivers. Flooding is a more common occurrence on the Sacramento River.

Table 10—Central Valley rivers with impoundment ratios of average annual flow for each. (Excerpted from Kondolf and Batalla 2005.) Each watershed contains more than two major reservoirs.

River	Average annual runoff $M^3 \times 10^9$	Total reservoir storage capacity $m^3 \times 10^9$	Impoundment ratio (As percent of annual runoff)
		Sacramento Valley	
Sacramento at Keswick	7.278	5.384	0.74
Feather at Oroville	5.215	6.714	1.29
		San Joaquin Valley	
Mokelumne at Camanche	0.744	1.032	1.39
Calaveras at New Hogan	0.205	0.396	1.93
Stanislaus at Knights Ferry	0.202	3.518	2.93
Tuolumne at La Grange	1.772	3.444	1.94
Merced at Snelling	1.290	1.305	1.01
San Joaquin at Friant	2.095	1.140	0.54

In both the Sacramento and San Joaquin Valleys on the valley floor, a system of floodway bypasses are maintained to convey flows greater than what can be carried within the river channels. In the Sacramento Valley, the mainstem rivers are used to convey water supplies for agriculture and urban use while also functioning as a conduit for flood waters. The combined purpose has resulted in floodplains isolated from their river channels, with the timing of flows shifting away from winter and spring flood pulses to late spring and summer steady medium flows. This has simplified the hydrology of the rivers. Coupled with the transformation of riparian lands to farms, the riparian habitats have been left with truncated processes and limited area to support the historic mix of plants and animals (Hunter et al. 1999; Mount 1995). As a result, much of the riparian habitat is now also simplified and shrunken to a point that species that were once common are now absent or rare. Vaghti and Greco (2007) describe seven contemporary plant communities of the Central Valley and trends in species declines.

For example, the least Bell's vireo was the most common species in the willow thickets along the Sacramento River in the 1930s (Grinnell and Miller 1944; Howell 2010). It disappeared during the 1950s, retreating to coastal rivers in southern California. Yellow-billed cuckoo formerly ranged as far north as the Columbia River, but today the species persists in very low numbers only along the Sacramento River. Likewise, sawgrass (*Cladium maricus* ssp. *californicum*) was an important basket-making plant for the Yokuts of the San Joaquin Valley (Latta 1938). Today, sawgrass is apparently extinct from the Valley (F.T. Griggs, personal observation). Slough thistle (*Cirsium crassicaule*) formerly grew in riparian areas in the San Joaquin Valley from Kern County northward into the Delta with collections as recently as the late 1930s (CalFlora). Today one population persists on the Kern NWR. Numerous species of native fishes have disappeared in the Delta due to introduced aggressive competitors (Sommer et al. 2003).

River Processes

Riparian ecology is driven by physical river processes of flooding (magnitude and timing) and sediment transport (geology of watershed, bank, and surface erosion;

floodplain deposition; and channel meander) (Stillwater Sciences 2007). Riparian plants and wildlife are adapted to these river processes. River processes also determine plant community succession and the evolution of its physical structure. Flooding eliminates species that are not adapted to flooding, such as many invasive plants and animals, especially rodents. Native floodplain-adapted plants can tolerate flooding for weeks or months. Flooding recharges the local groundwater table that provides moisture to all plants through the growing season. The season after a flood typically sees season-long growth by trees and shrubs that tap into the abundance of soil moisture.

Flood flows disperse seeds of many plants and deposit them in appropriate locations for germination and establishment (Stella et al. 2011). The timing of flood flows, especially the recession limb of the hydrograph, is critical for seedbed preparation, seed germination, and seedling development (Mahoney and Rood 1998). Seeds generally need the moisture available in the soil during the recession of flood waters, but seeds also require the oxygen available in those drying soils. If the seeds instead fall on dry ground, they cannot sustain their growth, desiccate, and die. Similarly, if they are subjected to repeated flooding, as often occurs in rivers managed for irrigation water, the seeds are starved for oxygen and die.

Many plants have co-evolved with the flood timing of the Central Valley to release seed when flood waters recede, such as Fremont cottonwood and several willow species. With the dam-imposed flow patterns, much of the seed bed preparation and some of the timing queues have been removed, greatly reducing the success of floodplain-adapted plants establishing in recently flooded lands. This disrupts the cycles of disturbance and succession common to floodplains and limits the quality and extent of riparian vegetation communities. Birds, fish, and other animals similarly have developed life cycles that synchronize with the flood cycles and, with the lack of cycles imposed by dam and floodway operations, are showing similar declines in their populations (Strahan 1984). Plotting these natural cycles against river hydrographs of current and former flow regimes (figs. 29a and 29b) can give a good indication of the potential of disruption to riparian systems.

Riparian vegetation on the valley floor arranges itself in relation to the magnitude and duration of flow. As riparian plant communities age and flow changes, a predictable sequence of events unfolds. This sequence involves a shift in community structure from plants adapted to high flow unstable channels to plants suited to slow water and less frequent inundation (Griggs 2009). High flow plants tend to be highly flexible and of relatively short stature that can easily lay down in the face of fast flood waters. As the plant community ages, the flexible stemmed plants shift the hydraulics in very localized spaces, which then allow other plants to take advantage of the relative calmer flows. This leads to even more stable plants, eventually leading to a mixed elevation canopy that includes ground level plants, mid-height shrubs, and various trees.

The early stage of the full mixed canopy riparian forest is marked by a high density of trees. As the tree-canopy develops, light and shade become dominant characteristics of the forest and many of the trees succumb to the dense shade, leading to fewer and fewer stems per acre (Swiecki and Bernhardt 2003). The riparian gallery forest of 80-100 years old has relatively few large trees with various shade tolerant shrubs, vines, and ground covers underneath the canopy. It is one that grades for relatively short-lived flexible plants

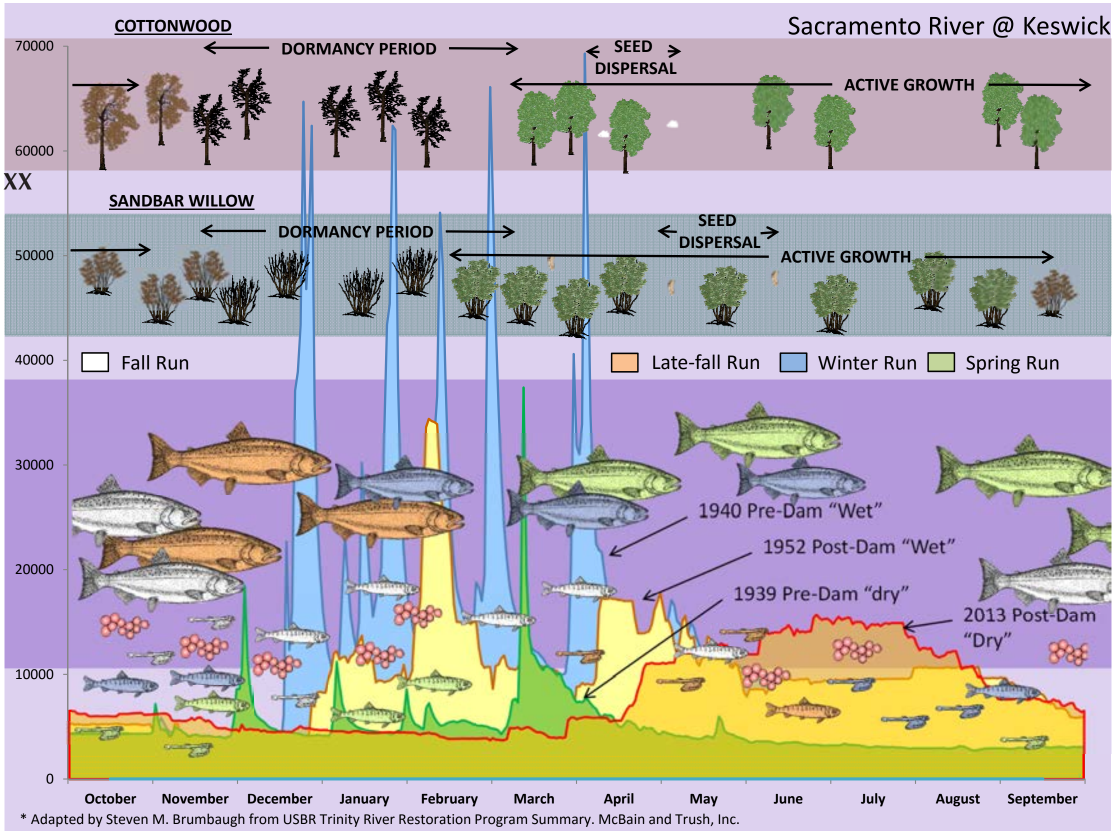

Figure 29a—Sacramento River Hydrograph.

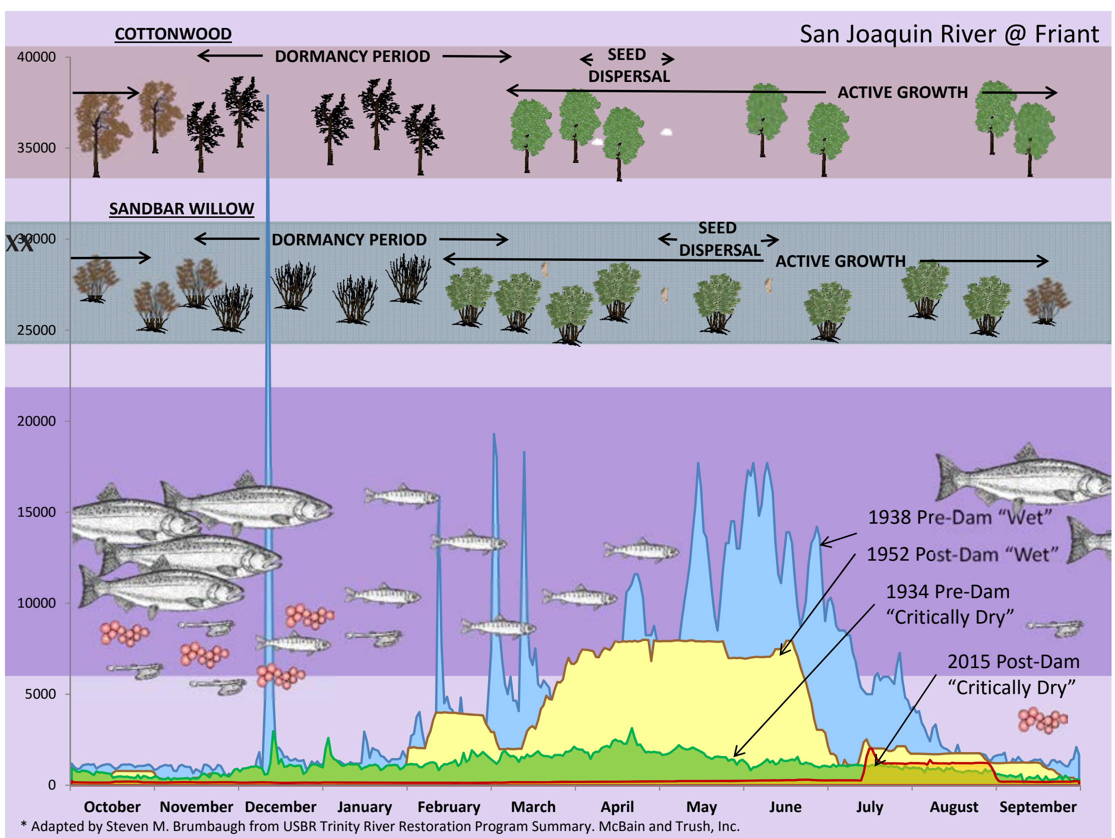

Figure 29b—San Joaquin River Hydrograph.

at the water’s edge to groves of oak and walnut in areas set away from the channels. Managing for flood suppression and water supply often disrupts this plant community evolution, diminishing the occurrence of gallery forests. (Griggs 2009). One of the important research questions is: What is the successional trajectory of the riparian plant communities under the modern hydrology?

Managing for irrigation supply will tend to produce an ecologically dysfunctional flow pattern that increases flows in spring and summer when the natural pattern would be declining flows, as on the Sacramento River (fig. 30). Many dammed rivers—Stanislaus, Tuolumne—have hydrographs that are essentially flat lines. We do not have indications that native species can adapt to such extreme changes. For example, because of the management of Shasta dam, today it is difficult to find stands of seedlings or saplings anywhere on the point bars or floodplain.

Sediment Transport

Dams, levees, bypasses, farming, road building, and urban encroachment into floodplains have all contributed to changing the patterns of sediment scour, transport, and deposition. Riparian assemblages take advantage of each of the variations in scour, transport, and deposition of sediment. The shifts in sediment dynamics have led to different niches and microclimates that in turn have shifted the character and quality of riparian areas (Rood et al. 2003; Stella et al. 2011) Most dams effectively capture sediment moving in the channels above the dam and leave the water released from the dam stripped of its sediment. The suspension and transport of sediment requires energy.

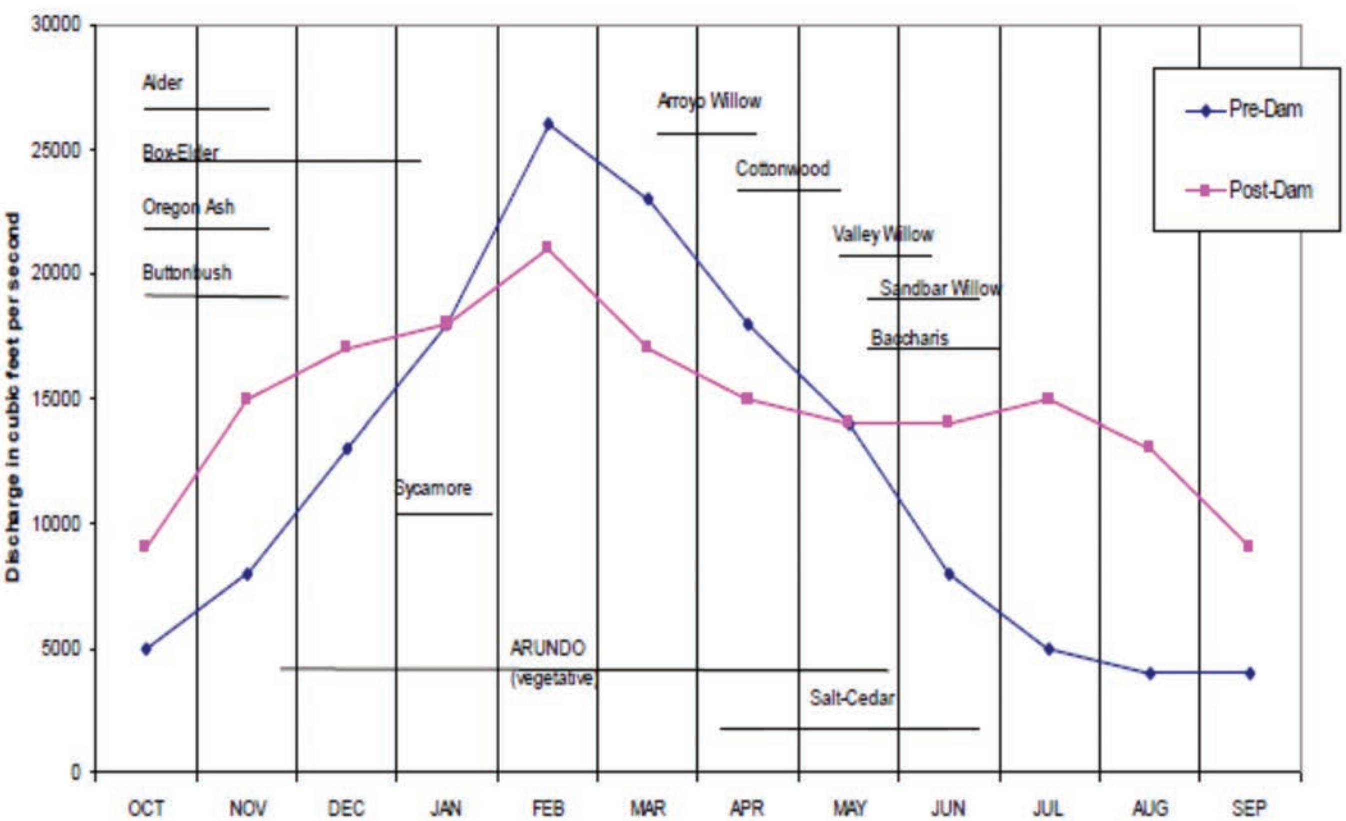

Figure 30—Dates for seed-release by water-dispersed riparian tree species and the mean monthly discharge for the Sacramento River pre-Shasta dam and post-Shasta dam.

The energy of flowing water provides the forces needed to keep sediments suspended or moving downstream. When sediment is deposited in reservoirs above dams, the water released has no sediment load to occupy its energy.

The water is then capable of working on the bed and banks and tends to erode these features until equilibrium in energy is reached where the energy of the water is balanced by the energy needed to suspend and transport sediment. The water released from the dam that contains little or no sediments has been termed “hungry water” (Kondolf 1997) because it is much more erosive than water that is full of sediments. Bank erosion and channel scour are often seen below dams. The sediment scoured away often exhibits particle size distributions different from what was being carried in the channel above the dam. Where these sediments fall out of suspension and are deposited, they can harm seedbeds for plants and nursery grounds for fish.

A feature of the shift in flow produced by dams is that the larger, more energetic flows of moderate floods are eliminated. Despite the impacts of hungry water, eliminating flood peaks stifles the process of channel meander, bank erosion, and bar formation that is a normal part of riparian areas. Rivers are dynamic systems; they continually adjust their bed and banks to the forces of flowing water (Knighton 1998). The result is a regular process of building bars and eroding them, moving banks, and rebuilding them. These processes depend on the more energetic, higher intensity flows to provide the energy needed to mobilize the bed and banks.

Deposition occurs as the storm events pass and the flows subside (Knighton 1998). Riparian vegetation has evolved to take advantage of these cycles. Species are adapted to colonizing newly formed bars, but they give way either to even greater flood flows or to encroachment from other plants better suited to the stability created by the colonizers. The result is a succession of plants with distinctly different characteristics. This plant sequence is similarly colonized by animals that align with the plant communities present. By making flows very constant, the dynamics of the system are eliminated and the ability for the plant succession to occur is greatly restricted. One example of this is when mid channel bars are covered with unnatural vegetation that would have been removed if more natural flow patterns were experienced. The loss of open bars has consequences for fish as well, limiting their spawning habitat, and often introducing algae or other organisms that pose problems for fish.

Levees and Flood Bypasses

Most of the river banks in the Central Valley mainstem rivers and tributaries are lined with levees. Together with the bypass areas, also bounded by levees, these features further restrict riparian vegetation. Many of the levees of the Central Valley have supported riparian trees in the past, but recent policy initiatives (Central Valley Flood Protection Plan 2016) have set in motion a process to remove trees from levees. Removing levee vegetation threatens to further fragment functional riparian areas and disaggregate riparian corridors. Levee construction along the river channel also separates, or disconnects, the floodplain from the river, eliminating flooding of the floodplain, sediment deposition on floodplains, groundwater recharge, and access to shallow water habitat that many organisms need for key life stages.

It has been argued that the flood bypasses provide floodplain functions (Sommer et al. 2003). But in many respects, the bypasses do not mimic floodplains. The bypasses are used for crop production. Farmers have rearranged the land surface to improve their farm operations. By grading and orienting flow to accommodate water inlets or drainage outlets, farmers have recast the surface that floods flow across, leaving uniform surfaces, sometimes tilted in directions inconsistent with natural flow, and without topographic variation. This simplified form removes potential for microhabitats and specialized niches. With this large-scale regrading of the land surface comes a dramatic decline of the area supporting riparian vegetation. The sparseness of native plant structure further fragments and isolates floodplain features and diminishes the processes that once supported riparian ecology.

Levees impose another significant ecological limitation. They isolate the floodplain from the river channel and thereby limit access of fish, plants, and other organisms to the highly productive shallow water habitat once found in vast areas of the Central Valley. For example, salmon have recently been shown to grow much faster on inundated floodplains than in the corresponding river channels (Jeffres et al. 2008). The floodplains (and inundated agricultural fields) produce vast quantities of food for fish that cannot be produced in the deeper, colder, and swifter river waters. Levees limit the ability for fish to gain access to this critical food supply.

Riparian Floodplain Vegetation

Simplified Native Structure

Due to these modifications in flow patterns and sediment transport, today's floodplain is variously covered by stands of native woody species within a matrix of both woody and herbaceous invasive species. The remnant stands of riparian vegetation are dominated by trees and shrubs of later successional development, such as valley oaks, elderberry, and box elder. Early-successional species—Fremont cottonwood and several willows—occupy a relatively small proportion of the vegetation today (Strahan 1984). Many of the terrestrial wildlife species that are rare or eliminated from the Central Valley riparian are dependent upon the early-successional vegetation (yellow-billed cuckoo, least Bell's vireo, southwestern willow flycatcher, yellow-breasted chat).

Invasive Exotics

Invasive, nonnative plants compete with native species for space and resources. Woody invasive species include giant reed (*Arundo donax*), Saltcedar (*Tamarix* spp.), Himalaya blackberry (*Rubus procerus*), hybrid black walnut *(Juglans* X *hindsii*), water primrose (*Ludwigia peploides*), and perennial pepperweed (*Lepidium latifolium*), with new invaders, as *Sesbania punicea* and Chinese pistachio (*Pistacia chinensis*), arriving annually. Numerous herbaceous agricultural weeds cover the floodplain to the exclusion of any native herbs. These species are well adapted to the current farm and floodway management practices, while native plants often struggle under these management paradigms.

Agricultural and Urbanized Floodplains

The cities of Sacramento, Stockton, and Marysville-Yuba City have covered thousands of acres of floodplain with homes and businesses, with more recent construction still occurring on high quality agricultural land west of Stockton (into the Delta) and north of Sacramento (Natomas area). These portions of the floodplain are cut off from the river by tall levees. The rich, alluvial soil of the floodplains that are protected by levees and not under residential and business development produces very large economic returns for agriculture and the surrounding community.

Summary

Severely broken river processes that change the ecology for plants and animals include the following:

(1) Water availability: flooding—reduced magnitudes, and shifted timing, diversions, and groundwater pumping.

(2) Sediment: trapped in reservoirs above dams, scour and erosion below dams (hungry water), particle size change and different deposition patterns, lack of floodplain deposits, channel and bank erosion.

(3) Hydrograph: inverted or flat, constant high spring summer flows, loss of flood events/peaks.

(4) Floodplain: land leveling and homogenization of land surface, loss of niches, loss of hydrologic connections (primarily due to levees).

(5) Native vegetation: loss of vegetation community connectivity, simplification of species complex, significant loss of extent (95 percent gone) of flood-adapted vegetation.

(6) Exotic/invasive species: competition with natives, exploits farming operations.

(7) Urban encroachment.

Current Ideas and Actions for Riparian Management

In 1987 the Sacramento River National Wildlife Refuge was created by the U.S. Congress. Early-on willing sellers of flood-prone agricultural land adjacent to the river came forward to sell their marginally productive farmland for conservation purposes (Reiner and Griggs 1989). The properties were not only flood prone, but the soils were heterogeneous with areas of channel deposits (sand and gravel) mixed with the more productive silt and clay loams. This resulted in soils less than ideal for agricultural production and consequently of low economic return. Flooding and heterogeneous soil patterns were ideal for restoration of structurally diverse riparian vegetation that would function as habitat for numerous characteristic riparian wildlife species, as well as specific target species (sidebar 1) (Griggs 1993, 1994).

Sidebar 1—Focal Bird Species That Guide Riparian Restoration Design

Riparian Obligate Species	Riparian Associate Species
Yellow warbler	Lazuli bunting
Common yellowthroat	Nuttall's woodpecker
Yellow-breasted chat	Ash-throated flycatcher
Song sparrow	Bewick's wren
Black-headed grosbeak	Spotted towhee
Blue grosbeak	
Warbling vireo	
Yellow-billed cuckoo	
Southwestern willow flycatcher	

Horticultural Restoration

Horticultural restoration involved planting native trees and shrubs that are characteristically found growing on the alluvial soils along the channel and on the floodplain (Hujik and Griggs 1994a,b) (sidebar 2). These native trees and shrubs (and understory) would form the type of structure that target wildlife required (Griggs 2009).

Sidebar 2—List of Species Used in Central Valley Riparian Restoration Projects

Trees
Fremont cottonwood *Populus fremontii*
Valley oak *Quercus lobata*
Red willow *Salix laevigata*
Black willow *Salix gooddingii*

Large Shrubs
Arroyo willow *Salix lasiolepis*
Box elder *Acer negundo*
Oregon ash *Fraxinus latifolia*
Elderberry *Sambucus mexicanus*
Buttonbush *Cephalanthus occidentalis*

Flexible-stemmed Shrubs
Coyote brush *Baccharis pilularis*
Mulefat *Baccharis salicifolia*
Rose *Rosa intermontana*
Sandbar willow *Salix exigua*
Quailbush Atriplex *lentiformis*

Vines (Lianas)
Grape *Vitus californicus*
Pipevine *Aristolochia californica*
Clematis *Clematis ligustifolia*
Blackberry *Rubus ursinus*
Poison oak *Toxicodendron diversilobum*

Herbaceous Grasses
Creeping rye *Leymus triticoides*
Blue rye *Elymus glaucus*
Meadow barley *Hordeum brachyantherum*
Purple needlegrass *Nassella puchra*
Squirrel-tail *Elymus elymoides*

Broadleaves
Mugwort *Artemisia douglasiana*
Gumplant *Grindelia camporum*
Evening primrose *Oenothera*
Goldenrod *Euthamia occidentalis*

The vegetation that has grown in the restoration plantings has performed well (Alpert et al. 1999; Griggs and Golet 2002; Griggs and Peterson 1997; Griggs et al. 1993). Results of increasing wildlife use of the restoration plantings have been impressive (Borders et al. 2006; Golet et al. 2008, 2009; Williams 2010). Monitoring of bird populations was the most intensive and shows how different species colonized the sites over a decade. Valley elderberry longhorn beetle (listed as threatened) also moved into the planted elderberry shrubs. Special bird species—such as yellow-billed-cuckoo—nested in the new plantings, and least Bell's vireo nested in a restoration planting for the first time in 60 years in the Central Valley (Howell et al. 2010).

As of this writing, approximately 9,000 acres have been planted by The Nature Conservancy, River Partners, Department of Water Resources, U.S. Army Corps of Engineers, California State Parks, and Resource Conservation Districts since 1989 in the Sacramento Valley, all on public lands (Refuges and Preserves). An additional 2,500 acres have been planted in the San Joaquin Valley. Funding came from Federal and State grants and contracts.

Levee Set-Backs

The levees that protect several cities and much farmland from large winter flooding were designed over 100 hundred years ago under very different flow and sediment transport conditions than today. Hydraulic mining in the foothills of the northern Sierra Nevada in the late 1800s generated sediment that filled canyons and when in the Central Valley it filled channels causing flooding. Levee placement adjacent to the channel allowed the much deeper flows to scour and to move sediments through the system. Unfortunately, today the sediment has been flushed away and the artificially deep flows of "hungry water" from below dams are now eroding the levees at their bases.

One solution to this situation is to build new levees set back from the edge of the channel that would allow the channel to meander across the new floodplain that will be created. This would take away the intense scouring of the levees and provide acreage for riparian vegetation restoration. Benefits for flood maintenance would be the better absorption of peak flows, the reduction of maintenance costs, and the risk of levee failure (Greco and Larsen 2014; Larsen et al. 2006). The set-back area would become restored riparian vegetation to function as high quality wildlife habitat and an important means of replenishing local groundwater. See (www.multibenefitprojects.org), NGO web site.

Restoring River Processes

Ecological Flows

The ecological flows model (The Nature Conservancy et al. 2008) identifies the timing for dam releases to create flows large enough to affect sediment/scour processes and encourage recruitment of seedlings of Fremont cottonwoods and support the life histories of Chinook salmon, steelhead, green sturgeon, bank swallow, and western pond turtle.

Applying this hydrologic model to releases from a reservoir should mimic the natural hydrograph and encourage seedling establishment of cottonwood and willows. This has been carried out with success on the Truckee River through and downstream of Reno, Nevada (Rood et al. 2003). One issue that restricts this approach in the Central Valley has been dollar cost, or lost opportunity, from releasing water from dams that could have been used to grow crops.

Williams et al. (2009) have recently developed the Floodplain Activation Flow (FAF) calculation for regulated rivers to quantify the area of floodplain covered by the smallest flow (2-year recurrence) that will initiate biological activity. In this example, the 2-year recurrence flooding-interval was selected as necessary for juvenile salmon to move onto the floodplain. The flood must occur during March-May with a minimum duration of 7 days. The FAF calculation estimates area of floodplain that will be inundated at specific flows for a specific reach of a river. The manager can define quantified ranges for the criteria, allowing for better planning of the hydrological/biological objectives for any project on the floodplain.

Managed Hydrology in Farm Fields to Benefit Juvenile Rearing of Salmonids

Salmon and steelhead in their juvenile stage forage on the floodplain for invertebrates during spring floods—the same spring floods that prepare the seedbeds for the cottonwoods and willows. Due to dams and the altered flows, floodplains rarely flood today. Fish hatcheries raise juvenile salmon in concrete runways and feed them artificial food, releasing them into the river in April or truck them to the SF estuary and release them to the delight of invasive striped bass.

An alternative method for raising the juvenile salmon is to manage the hydrograph on an artificially flooded farm field (that functions as a pond) in the spring (February and March). The fish forage on native invertebrates in the pond water and rapidly increase in size and weight (Sommer et al. 2003). After 4 to 6 weeks in the pond, the fish are released into the river in March or April (Jeffres et al. 2008). This is a relatively new concept with only four seasons of demonstration projects of varying acreage. Results appear promising with significant growth (3 to 5 times weight gain) during their 4 to 6 weeks of residence on the flooded fields.

Managing Riparian Vegetation for Flood Flow Conveyance and Ecological Benefits (Flood Management)

Riparian vegetation in Central Valley today occurs between flood management levees. The California Department of Water Resources (CDWR) is legally charged with the management of the majority of levees and the floodway for the purpose of Public Safety (CDWR https://water.ca.gov/Programs/Flood-Management) and Greco and Larsen (2014) (fig. 28). Farmers working the lands between and outside the levees continue to try to optimize conditions for commercial production with little or no attention to river process or ecological quality. In many cases riparian vegetation is seen as a threat or nuisance. Thus, flood management and farming goals supersede the ecological goals of riparian ecology everywhere outside of the Federal refuges.

The challenge today is how to manage the landscape such that public safety, farming, and ecological riparian benefits are all realized for people and wildlife. The discussion below is an attempt to describe how ecological management of riparian vegetation can fit into the logic of flood managers that think in terms of risk and system reliability.

All of the flood-prone lands, floodways, and flood bypasses in California readily grow plants. These sites possess some of the most fertile soils on the planet, and when coupled with even a little moisture, allow vast number of species to grow. Whether those plants are useful for us and the ecosystem depends on how and what is managed. Native flood-adapted plants offer many advantages over exotic plants. (Micheli et al. 2004). They better support the historic ecology, are better adapted to the timing and processes needed to restore ecological functions, persist better under flood and drought and are therefore more predictable and reliable, and support native animals better than exotics.

Floodway Design for Flow Conveyance

Flood managers use two-dimensional hydraulic computer modeling for designing flows through the floodway. Recent work using two-dimensional computer models for flood flows shows that plants can be arranged to assist with managing flows to reduce flood risk. Until recently, flood modelers typically looked at vegetation as a problem or, at best, a necessary evil in managing floodways. Understanding of how plants can be used to help direct flood flows, reduce scour, and control erosion and deposition is now taking hold among flood modelers (Anderson et al. 2006; Stone et al. 2013). With this awareness comes the prospect of designing floodways that integrate ecological goals with risk reduction and economic goals and constraints (Fischenich 2006). Promoting native flood-adapted plants has the added advantage of assisting agriculture by reducing the seed banks of farm weeds and limiting the way weed seeds can be deposited on farm fields.

Managing Flood Risk With Riparian Vegetation

In flood management, the risk of property damage or injury of people has been paramount. Recently the idea that risk extends to the ecosystem has been brought into flood management discussions. In the Central Valley, the large number of threatened and endangered species that occupy floodplains and riparian lands means that flood managers must now consider the implications of their flood designs not only on traditional risk factors but also on features of the landscape that support the listed species. Assessment of risk has largely been accomplished using computer models. In these models, vegetation is associated with a roughness factor and contributes to defining the maximum water surface elevation during a flood event (Aberle and Jarvela 2013; Freeman et al. 2000). Modelers have typically adjusted the roughness factor to get model outputs to align with calibration points (water surface elevations known from previous flow events). This practice has led to vegetation removal justified by a need for smoother surfaces as depicted by the models. With the widespread adoption of two- and three-dimensional flood models (Wilson et al. 2006) we now have the ability to look beyond the simple application of roughness as a calibration parameter and consider roughness as a design feature (McKay and Fischenich 2011).

Emerging work is showing that native flood adapted plants can be equal to, or less rough than, bare soils (Kavvas et al. 2009). Two-dimensional modeling is also showing that a combination of roughness values can be used to our advantage in lowering flood risk (Soong and Hoffman 2002). Older models averaged the measurements of many parameters. Two-dimensional and three-dimensional modeling allows those parameters to be treated more independently with the result that we can now design flow paths within floodways that provide for ecosystem processes to be expressed while still managing flood risk at low levels.

With these new modeling capabilities, we can now discuss ecological risks, the role and placement of specific plant species, mixed age classes, mixed canopy structure, and other ecologically important attributes in the context of reducing risk to people and property (Fischenich and Copeland 2001). Using these tools we can visualize how complex roughness can be applied to protect sensitive areas, be they levees and homes or delicate habitats. Achieving the desired mix of roughness can be accomplished by placement of plants with appropriate physical structure. This dramatically changes the perspective of vegetation in flood management. Instead of being an impediment that increases flood risk, vegetation now becomes a valuable tool used to reduce and optimize risk exposure.

Improving Flood Management Reliability With Riparian Vegetation

In considering reliability we can again borrow from the world for flood management. Reliability is the potential for a particular feature or facility to provide a repeated outcome. Flood managers want levees and floodways to be reliable at conveying flows and keeping flood waters away from valued property and people. Floodways and levees undergo constant change that in turn shifts their reliability. Much of the change arises because of the ability for plants to colonize and grow in flood systems. Maintenance of levees and floodways is a constant challenge where floodway operators remove vegetation to maintain flow capacity (reduce roughness). Denuding these features provides temporary improvements in capacity but also primes the sites for recolonization and starts the cycle over. The maintenance needs and costs are considerable and maintenance programs are regularly facing shifting budgets and damage repair.

As noted above, succession in floodplain and riparian plant communities is marked by predictable changes in the density of plants, the number of stems per area, and height of the leaf canopy above the ground (Griggs 2009). The typical change in roughness peaks after colonization when vegetation communities are transitioning to longer lived, larger species and then declines as increasingly shade limits understory growth. Floodway maintenance schedules tend to remove vegetation at peak roughness, creating a disturbance that is somewhat unpredictable, leading to lower reliability in system performance.

Sidebar 3—Using Stand Design for Flood Risk Reduction

An alternative approach to periodic wholesale removal of high roughness vegetation is to deliberately manage to transition the plant community to mixed age class structures that include substantial percentages of older, lower roughness stands. This can be done by implementing a rotational thinning process. Two-dimensional flood modeling of Star Bend on the Feather River has illustrated the point. At this site, high roughness on the floodplain creates a threat of increasing channel scour along the left bank of the river channel where a water diversion facility is located. A rotational thinning option for the floodplain was modeled that removed a swath of plants longitudinally through the center of the floodplain at time zero. At 10-year intervals, additional thinning occurred along adjacent, parallel swaths. After four cuttings, the site had an age class distribution of 0-, 10-, 20-, 30-, and 40-year-old vegetation. Each cutting was sized to retain acceptable scour potential at the water diversion (fig. 31). At each thinning episode, a smaller swath than the previous cut was needed to maintain acceptable conveyance and roughness. At the 5th interval, it was estimated that normal tree fall and plant community dynamics would sustain the needed roughness distribution without further thinning.

While hypothetical, this exercise illustrates the potential for integrating plant stand management with risk reduction to provide a dependable, repeatable, reliable conveyance system. Rather than depend on regular infusions of funds for clearing plants, this option consistently reduced costs until a self-maintaining system emerged. At this point, the system is highly reliable, being self-regulating, and funding can be reduced to that needed for periodic assessment to ensure the stand retains its risk reduction capabilities. At the same time, the mixed age class structure increases the biodiversity and hydraulic complexity of the site, providing for a better mix of ecological process and function.

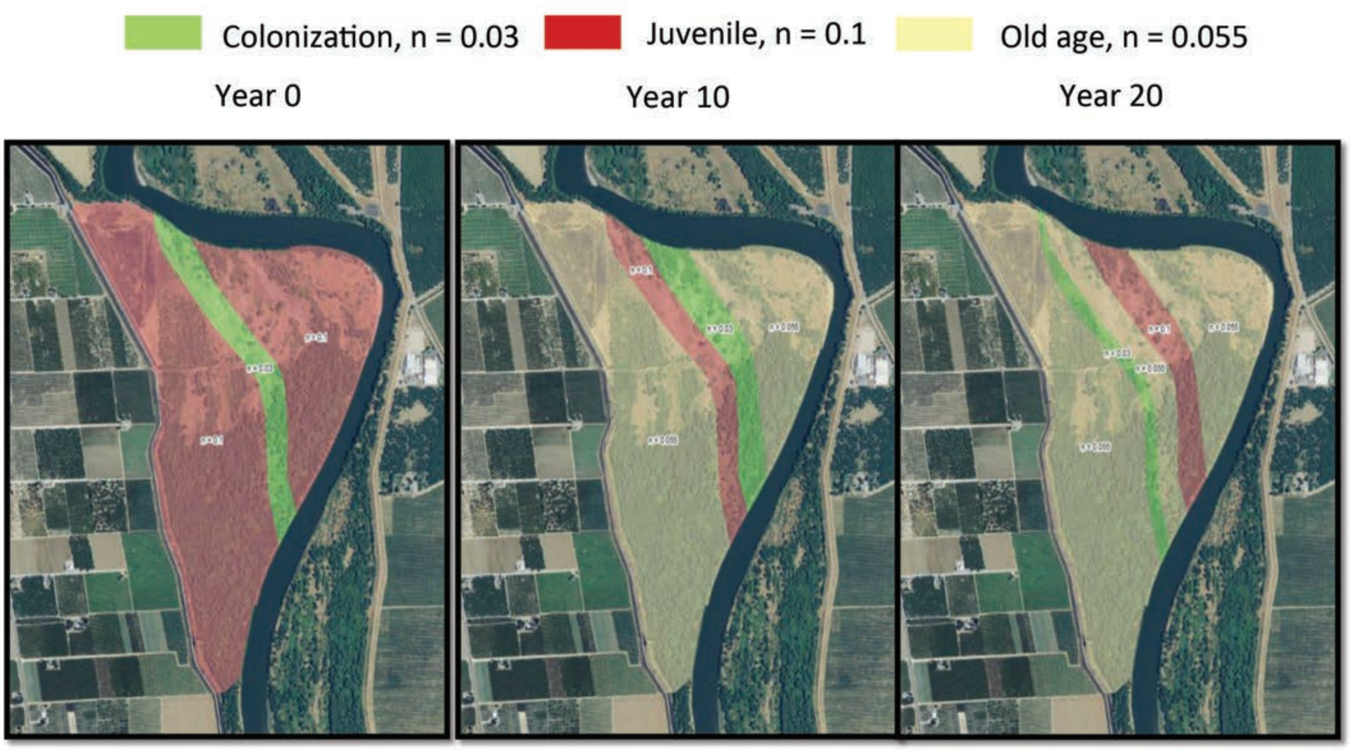

Figure 31—Rotational thinning that targets complex age class forest. Floodplain velocities change over time with vegetation growth and management resulting in higher/distributed velocities across floodplain and lower velocities (reduce erosion potential) through channel.

Monitor and Map

To achieve a greater expanse of Central Valley riparian habitat, it will be necessary to understand the quality and extent of existing habitat. Recent work has produced a fine-scale vegetation map of the riparian habitat of the Central Valley (California Department of Fish and Wildlife's VegCamp 2011). Conducted as part of the flood planning work for the Central Valley, the new vegetation maps provide data layers for GIS use, for the first time providing a valley-wide picture of where riparian vegetation flourishes. Now that a baseline has been established, changes in riparian vegetation communities can be tracked. Future monitoring will provide insights into how plants and animals are doing in the reconstituted riparian forests and whether these areas can produce the same scale of diversity and sustainability that once occurred.

Climate Change

The riparian landscape we experience going forward will not likely be the same as what we lost, or even what we have. Climate models predict a much hotter California in the future, especially in the San Joaquin Valley and Tulare Basin. Whether that heat comes with a shift in the amount of precipitation is unclear. But the models do predict more variation in precipitation and a loss of snowpack (State of California, Department of Water Resources 1993). This will likely result in more frequent flood flows of moderate scale and an increase in the frequency of occurrence of large damaging floods. The large dams and reservoirs provide a great deal of flood risk reduction for the modest events but are likely to operate much like a free-flowing river during large events, when the reservoirs are full and the dams must release everything coming at them. In these situations, we must anticipate where the flood flows will go and how the system can recover from damaging, high volume flows. One strategy is to put more flood capacity back on the floodplains, widen the bypasses, and to stabilize the most vulnerable flow paths using native floodplain adapted vegetation to maintain the floodways (Seavy et al. 2009). This is a strategy that merges flood risk management with conservation ecology in a sustainability context. Doing this would allow a revitalization of the Central Valley's riparian landscape, and the opportunity to take advantage of many contemporary and emerging tools in riparian assessment and flood modeling.

Conclusion

California's Central Valley once supported vast arrays of riparian and wetland habitats that consisted of complex topography and microhabitats. Today much of that ecosystem has been replaced with industrial agriculture and, more recently, with expanding urban landscapes. The combination of climate change and evolving social needs will continue to change the landscape structure of the Central Valley going forward. Riparian ecosystems can play an important part in providing a sustainable landscape, posing acceptable levels of risk from flood, drought, and other disruptions. To achieve this will require the combined efforts of land managers, conservationists, city and regional planners, and business interests all working to find common strategies that provide the desired mix of habitat features. New and emerging tools in flood modeling and conservation can be used to help visualize and design these future landscapes. Clear objectives will be needed to get the most value from these efforts.

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Chapter 9. Recreation Habitat Versus Ecological Habitat in Riparian Areas: Can We Manage for Both?

Bo Shelby and Doug Whittaker

Introduction

Yosemite Valley is a narrow, steep-walled canyon with fantastic natural areas, and its scenic and ecological wonders inspired the National Park concept and preservation ethic. But the Merced River through Yosemite Valley is one of the most heavily used non-urban riparian areas in the world. Over 4 million people visit each year, and 90 percent go to the box canyon where towering granite summits rise over 4,000 feet above the meandering river. About 8 miles long and less than one mile wide, the valley has a mix of riparian vegetation, open meadows, and forests. There are also roads, bridges, trails, acres of parking lots, campgrounds, four hotels or lodges, gift shops, and restaurants concentrated in the valley's eastern end. With over 600 campsites and 1,000 hotel units, and 7,000 vehicles and 20,000 people on busy days, the Valley's iconic resources are challenged by overuse and development.

The latest attempt to address these problems is the 2013 River Management Plan for the Merced, a designated part of the National Wild and Scenic Rivers System. Two previous plans were overturned by lawsuits brought by environmental groups, eventually leading to a settlement agreement filed with the Ninth Circuit Court that guided development of the third plan. The settlement identified many issues, including traffic and parking congestion, facility overdevelopment, attraction site crowding, and riverside impacts that degrade scenic, recreation, and ecological values. Because of our work on carrying capacities in Grand Canyon and other river corridors, we were identified in the settlement agreement to assist with the Park's planning effort.

Much of Yosemite Valley's 150 years of development has occurred within a half-mile of the river, resulting in many riparian impacts. At the largest scale, the 1879 breach of the El Capitan glacial moraine to drain seasonally wet meadows lowered the river bed by 5 feet and increased channel incision and subsequent erosion (Milestone 1978). As development increased, infrastructure was protected from flooding and lateral bank erosion through revetments (riprap), log jam removal, in-channel sediment mining, and riparian plantings. By the late 1970s, about 2.5 miles of the river's banks had been channelized, mostly near East Valley roads, hotels, and campgrounds. Infrastructure included as many as 38 bridges, some constricting the natural channel by 50 percent and collectively altering the river's natural dynamism (Cardno Entrix 2012; Milestone 1978). Despite narrowing at bridges and revetments, the overall result was increased channel widths and bank instability (Madej et al. 1994).

In addition to these larger-scale changes, the Merced's riparian areas received increasing levels of tourist activity. Hotels, cabins, campgrounds, and employee housing were often located within a hundred feet of the river or adjacent meadows, and networks of roads and trails provided access to riverside attractions. Infrastructure and access encouraged visitors to go to the river, and activities such as swimming, picnicking, and boating became important parts of the Yosemite Valley experience. Resultant impacts included spider webs of user-created trails in meadows and along the river, vegetation trampling along the banks, woody debris removal, and compaction/erosion around the roots of large trees (Madej et al. 1994). Coupled with larger-scale geomorphic changes that disconnected the river from meadows and riparian areas, these site-specific recreation impacts have become a major challenge for the Park Service.

Park scientists often see these impacts through a restoration lens. Geomorphologists contemplate removal of riprap and bridges as obvious solutions to large-scale impacts. Similarly, riparian ecologists see exclusions of recreation use as the preferred way to address site-specific impacts and restore native vegetation and riparian function. These kinds of restoration plans could radically alter the amount, location, and function of infrastructure that currently handles the volume of use. These plans could also prevent visitors from enjoying many sites along the river. This applies pressure on the Park Service's dual mandate to protect resources while allowing visitors to enjoy them.

As a practical matter, it is politically and administratively unrealistic to substantially reduce the hotels, campgrounds, and roads that accommodate millions of visitors. Few experts expect Yosemite Valley to return to mid-19th century levels of development and use. The question then shifts to finding an appropriate balance between infrastructure, use, and restoration in a high profile management plan for one of the world's best-known national parks. How do we address riparian impacts in such an environment?

The answer lies in several seemingly simple questions.

- What, exactly, are the impacts of concern?
- Where, when, and how do the impacts occur?
- To protect riparian values, can we get people to change the way they recreate, or must they be excluded to protect sensitive areas?

The remainder of this chapter considers riparian impacts on the Merced, how visitors perceive them, and whether visitors support management actions that might be used to reduce impacts or restore riparian conditions. Data come from a study of river use and visitors (Whittaker and Shelby 2012), as well as broad indices of ecological health in the riparian zone and adjacent meadows.

Yosemite Valley's Current Riparian Condition

Yosemite National Park has conducted several studies to document riparian ecological conditions (Cardno Entrix 2012). A comprehensive review of that work is beyond the scope of this chapter, but the plan focused on two primary indicators: (1) California Rapid Assessment Method (CRAM) scores, a multi-variable measure of riparian health; and (2) Largest Patch Index (LPI-5) scores for adjacent meadows, which measure fragmentation.

Riparian Health

The CRAM protocol was developed for assessing overall health in unconfined riverine riparian areas. It measures 14 variables scored on a scale from "A = best achievable" to "D = worst commonly observed" (Collins et al. 2008). These scores, in turn, are combined to create four overall attribute scores (ranging from "0 = poor" to "1 = best achievable"), or a single overall CRAM score (by evenly weighting each of the attribute scores, also ranging from 0 to 1). The scores can be used to compare areas, changes over time, or the contributions of different attributes to overall scores. This provides an overall indicator of riparian health and identifies areas with higher impacts that need more intensive study. The 14 variables were organized into four major categories, including buffer and landscape context, hydrology, physical structure, and biotic structure.

CRAM scores were developed for 81 alternating-bank assessment areas along a 16 km (9.9 mile) reach of the Merced River in Yosemite Valley (Cardno Entrix 2012). Overall scores ranged from 0.56 to 0.93 (median = 0.77), with generally lower scores in reaches adjacent to the higher density use and development in East Valley.

Several metrics showed good condition and varied little across the study area (including landscape connectivity, water source, plant layers, and percent invasives), while other metrics influenced differences in overall scores (buffer condition, channel stability due to revetments, buffer width, biotic structure, and topographic complexity). Areas with the lowest scores tended to have bank protection (riprap) or extensive bank erosion, narrow or absent riparian buffers, few different types of habitat features, lower topographic complexity, and greater human use. Conversely, areas with higher scores had wider buffers, minimal bank protection, greater structural complexity, and lower human use.

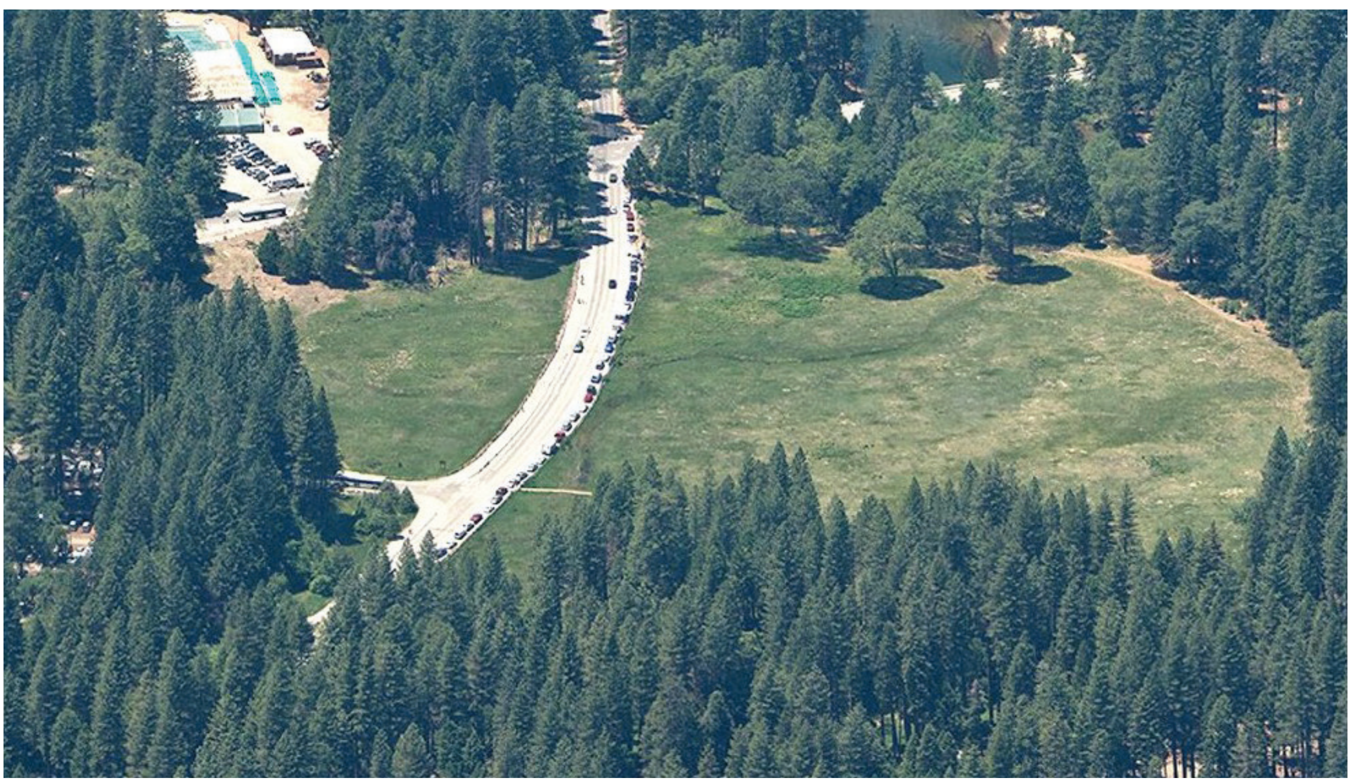
The assessment called attention to lower scores caused by channel stabilization around meander bends and near bridges, as well as bank erosion associated with human use from road turnouts, campgrounds, or hotels. Although CRAM scores are not designed to assess the relative influence of specific stressors (e.g., geomorphic issues from bridges and revetments vs. recreation impacts), the scores for Yosemite Valley suggest that recreation impacts in East Valley are part of the problem and need management attention.

Meadow Fragmentation

The seasonally wet meadows adjacent to the Merced River are also connected to the Valley's riparian values and threatened by use. Meadow function and health were assessed by a Largest Patch Index (LPI-5), which is the percent of a meadow in its five largest patches (adapted from McGarigal and Marks [1995] by Yosemite National Park staff). This measure is sensitive to sizes of intact areas and amounts of user-created trails, and it has been associated with impacts related to meadow hydrology, soil moisture, nonnative species, habitat quality, and barriers to small mammals (Forman 1995; Gaines et al. 2003; Holmquist 2004; Knight 2000; Leung et al. 2011a; Lindenmayer and Fischer 2006). In Yosemite Valley meadows, user-created trails and fragmentation impacts have been well documented and used to develop restoration priorities (Foin et al. 1977; Holmquist 2004; Holmquist and Schmidt-Gengenbach 2008; Leung et al. 2011b).

Stoneman Meadow in the heart of the East Valley provides an interesting example. A 1978 photo shows an extensive web of user-created trails (see fig. 32), a total of 3,170 meters or about 2 miles. The fragmentation index based on this photo is 40.40 percent (out of 100 percent possible), which met the National Park Service definition of a degraded meadow. The Park Service subsequently removed and restored most of these trails, then used barriers and walkways to funnel use to a single formal path across the meadow with a raised boardwalk surface. Over the next 30 years, the meadow recovered to the condition shown in figure 33. There was a 90 percent decrease in trails (to 327 meters), and the fragmentation index improved to 99 percent (nearly full restoration). Throughout this time, there had been no effort to limit numbers of visitors in the meadow, and from 1978 to 2011 the annual number of people visiting the Park increased by 54 percent. The two sets of trails are superimposed in figure 34. Ongoing monitoring continues to assess meadow condition, use levels, and visitor compliance with barriers and walkways to better understand relationships among these variables.

Figure 32—1978 photo of trails and road through Stoneman Meadow in East Yosemite Valley (Merced River is just outside the top of this photo). National Park Service photo.



F**igure 33**—2011 photo of restored Stoneman Meadow (main trail with boardwalk is visible near the T-junction). National Park Service photo.

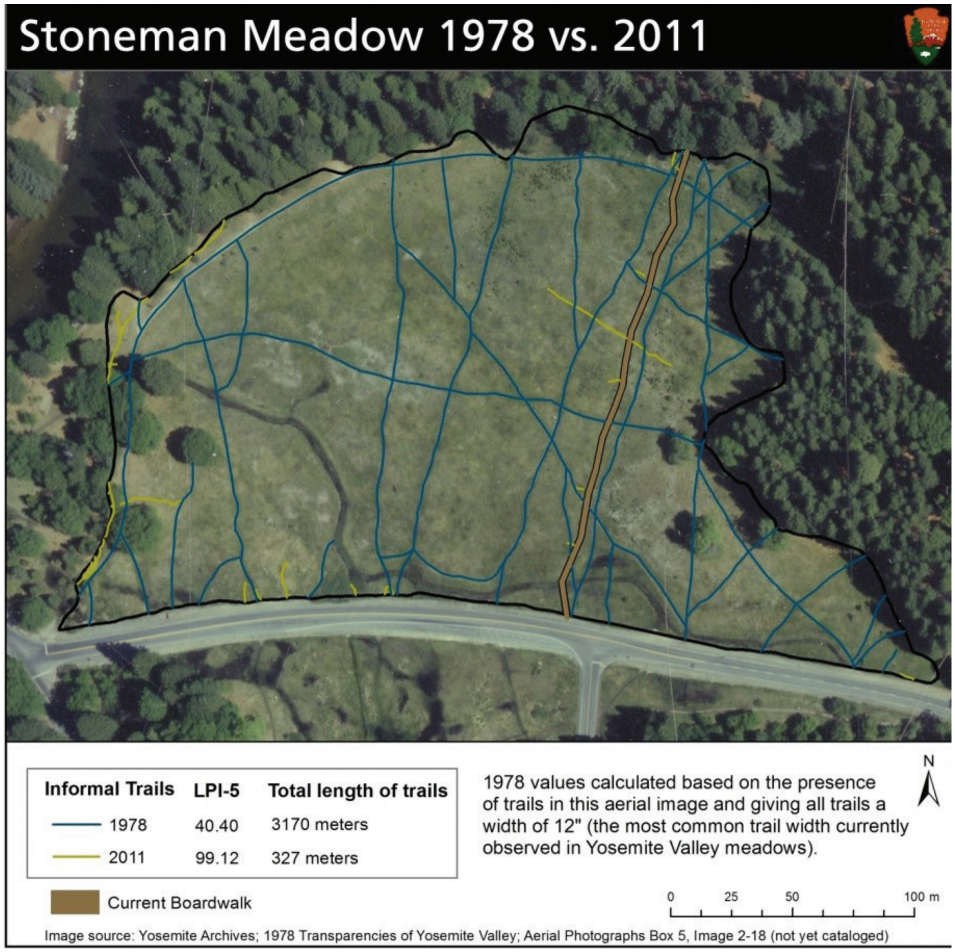


Figure 34—NPS comparison of trail lengths and meadow fragmentation values (LPI-5) for Stoneman Meadow in 1978 and 2011.

Recreation Impacts in Yosemite Valley's Riparian Areas

For the sake of discussion, the photo in figure 35 shows examples of typical riparian impacts caused by recreation use. Riparian impacts are complex, but Park ecologists assessing this site would identify trampling, loss of vegetation, soil compaction and erosion, and tree root exposure. If asked to evaluate these impacts, we suspect few ecologists would find them acceptable.

We were interested in how visitors perceive the same impacts. As part of a larger river study, we sampled 806 river users, including both floaters and shore users (Whittaker and Shelby 2012). The survey form included this photo (fig. 35), with the following instructions:

This river bank photo shows an area used by Park visitors along the Merced. National Park Service scientists evaluate river banks from an ecological perspective, but we are interested in how visitors perceive them. Please rate the acceptability of this river bank from your perspective.

Respondents answered on a 9-point acceptability scale; results are given in figure 36. Overall, 76 percent rated this river bank as acceptable, and an additional 13 percent rated it marginal; only 11 percent rated it unacceptable. This raises interesting questions about differences between scientists and recreation users, goals for preservation versus use, and definitions of acceptable impacts.

Figure 35—Photo of impacted riparian area along the Merced River in East Yosemite Valley, evaluated in survey of river users. Photo by Confluence Research and Consulting.

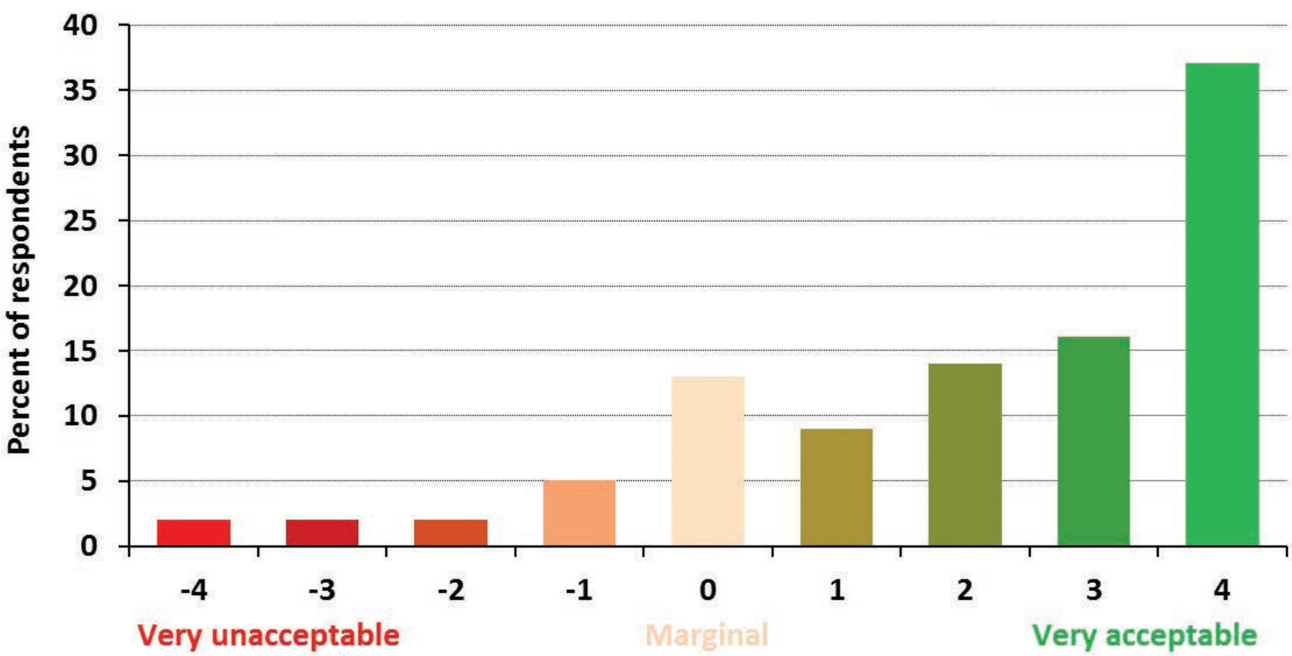


Figure 36—River users' ratings of riverbank conditions shown in the photo in figure 35. Data from Whittaker and Shelby 2012.

Where, When, and How Do Impacts Occur?

The impacts pictured in figure 35 occur along the Merced in Yosemite Valley, but they are not evenly distributed. They are found primarily in accessible places where trails, roads, and bridges approach the river, or where the river offers attractive features such as scenic views, beaches, or swimming holes. Reach-based riparian monitoring efforts (e.g., CRAM scores) help recognize the general problems of recreation impacts but are not specific enough to identify precise locations, types of impacts, or causes.

Ecologists hypothesized that riparian impacts were associated with boating use, a popular summer activity that occurs on a 2.5-mile reach adjacent to the highest use areas in East Valley. A raft rental and shuttle concession provides 150 to 200 boats on peak use days (with two to four people per boat). Private floaters in rafts, canoes, kayaks, and tubes may add 50 to 100 boats per day. The Merced River Plan included proposals to limit or eliminate river running, so our study observed boaters and shore users to document the way they used riparian zones.

Results showed that floaters mostly stayed on the water, in their boats or tubes. When they stopped on shore, it was primarily at open sandy beaches, where they generally stayed for short periods. Exceptions included the concession boating put-in (an eroded trail and unhardened, unvegetated bank next to a bridge) and trails leading from a sandy beach to a foot bridge commonly used for jumping (despite Park regulations that prohibit it). Floaters rarely stopped along vegetated riparian shorelines or steeper banks.

In contrast, land-based visitors utilized networks of user-created trails into the riparian zone, which connect parking lots, road pull-outs, campgrounds, and Valley developments with river overlooks, beaches, and other open shoreline areas. Shore users often set up on sand beaches, but they also dispersed along upland terraces, under the shade of larger trees, or even on riprapped banks. These locations tend to become clearings with minimal understory, possibly attracting more use because they look like designed recreation sites (similar to those in campgrounds).

Figure 37 illustrates riparian characteristics that are important for riverside recreation users. Observations from the river study indicate that river users are attracted to places like this, which we identified as "recreation habitat." Characteristics of recreation habitat include a convenient location with easy access, a sand beach, good places to sit, views of the river or valley, sun (for swimmers/sun-bathers), and shade (for sun protection). The recreation-related goals of visitors appear to be different from the ecological goals of scientists, so it makes sense that recreation habitat may be different from plant or animal habitat.

Are Users Willing to Change Their Impact-Related Behaviors?

Impacts depicted in the photos are seen differently by visitors versus ecologists. But do visitors care about ecological impacts, and are they willing to do anything differently to mitigate the effects of recreation use?

To measure support for management, study respondents were shown example photos of (1) split rail fencing and (2) a boardwalk and stairs. These are management actions the Park Service could use to reduce bank and meadow impacts associated with recreation use. Reasons and possible consequences were given, and visitors were asked to rate acceptability on a 9-point acceptability scale. The photos and questions are given below:

Figure 37—Visitors using riparian “recreation habitat” along the Merced River in East Yosemite Valley. Photo by Confluence Research and Consulting.

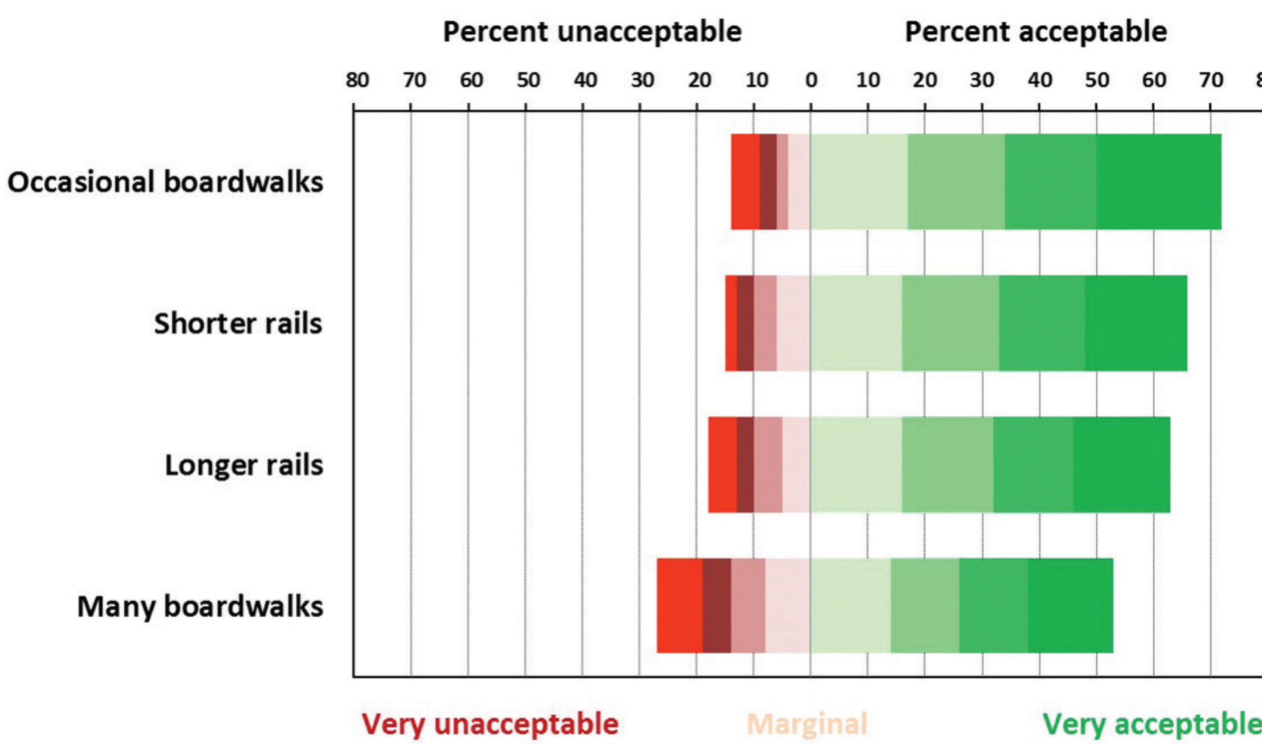


Figure 38—River users’ ratings of fencing and boardwalks. Data from Whittaker and Shelby 2012.

To reduce bank and meadow trampling along the river, the Park Service could close sensitive areas (see split rail fencing photo, fig. 40) and direct people toward areas that can withstand use (see boardwalk and stairs photo, fig. 41). However, these actions may decrease "naturalness," prevent access to some areas, or lead to congestion in other areas. Please rate the acceptability of the following actions.

- *Longer split rail fences (over 200 feet) to protect large areas from trampling, with short openings for river access.*
- *Shorter split rail fences (under 50 feet) to restore small sites with heavy trampling.*
- *Occasional boardwalks and stairs through meadows and sensitive areas to provide access to areas like beaches.*
- *Trail networks with many boardwalks and stairs directing use to less sensitive areas and discouraging off-trail use.*

Figure 38 shows acceptability ratings for the four fencing and boardwalk options. The two lower development options ("occasional boardwalks and stairs" and "short split rail fencing") were most acceptable (72 percent and 66 percent acceptable, respectively). Even the lowest-rated option ("trail networks with many boardwalks and stairs") was acceptable for 53 percent, marginal for 20 percent, and unacceptable for only 27 percent.

Respondents were also asked to evaluate more general approaches such as education and regulation, using a 5-point support-oppose scale. The three approaches were described as "education efforts that teach visitors to avoid sensitive areas," "close user-created trails that lead into sensitive areas," and "prohibit off-trail or off-beach use in sensitive areas."

Figure 39 shows widespread support for such actions to protect ecological or aesthetic values along the river. There was 81 percent support for education, 73 percent support for closing user-created trails, and 62 percent support for prohibiting off-trail/off-beach use in sensitive areas.

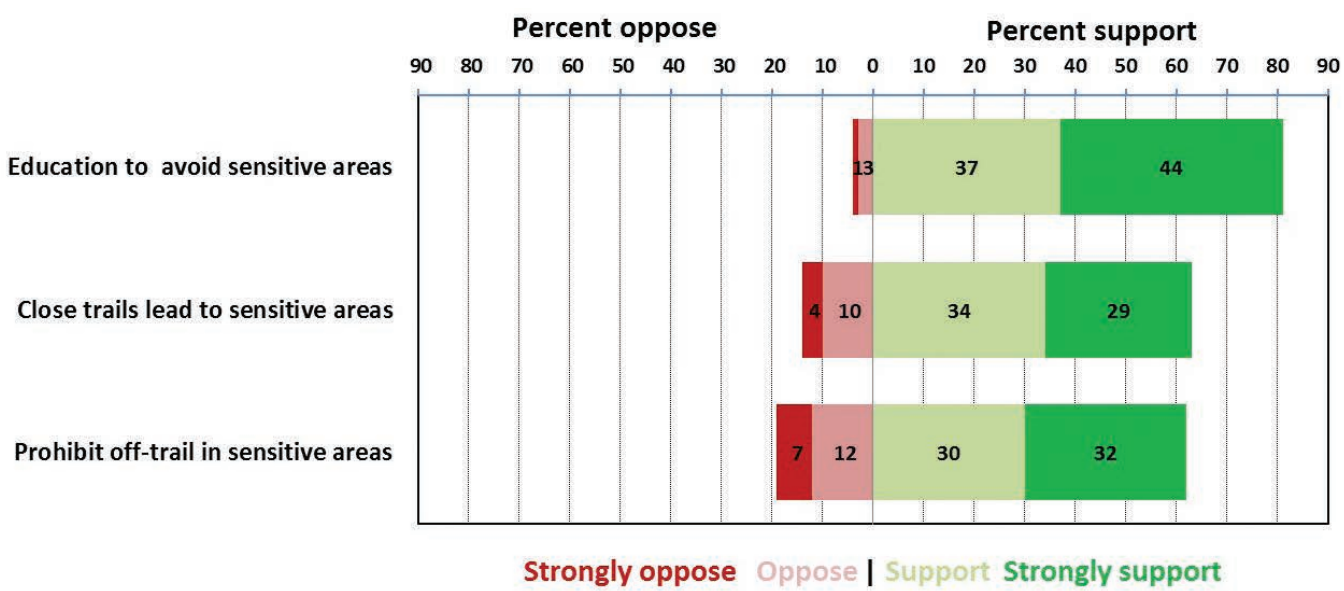


Figure 39—Support/opposition for education and regulations to address riparian impacts. Data from Whittaker and Shelby 2012.

Figure 40—Carefully designed recreation infrastructure can facilitate use and address riparian impacts.

Figure 41—Elevated light-penetrating boardwalks and stairs provide access and protect riparian vegetation on the Kenai River, Alaska; high densities of salmon anglers decimated banks before the advent of this simple but effective technology. Photo by Confluence Research and Consulting.

Management Implications Regarding Riparian Use Impacts

Data from the study indicate users do not commonly recognize riparian impacts, and they seek recreation habitat next to the river, where they contribute to these impacts. It may be difficult for the Park Service to keep visitors out of attractive recreation habitat, but the study shows broad public support for management actions to reduce impacts. These include defined trails, boardwalks, fences, and even closures, especially if applied judiciously.

The challenge is to organize use in riparian areas so that visitors can access attractive recreation habitat that is less susceptible to ecological impacts. A hypothetical example in figure 40 shows how carefully developed recreation infrastructure can direct human use to appropriate areas in the riparian zone. Trails are designed to keep use away from sensitive banks and riparian vegetation, while delivering people to more resistant areas that provide good recreation habitat next to the river (e.g., beaches and purposely chosen viewpoints). Where trails unavoidably cross or arrive at wet or sensitive areas, hardened surfaces can constrain use to a minimal footprint and ensure impacts do not exceed acceptable levels. Split rail fences at the back of the beach keep users off vegetated banks and direct them to trail access at either end.

Technological advances have improved the ecological benefits of hardened surfaces. For example, light-penetrating boardwalks and stairs (made of steel or aluminum, sometimes with fiberglass or composite grating) have become common on high density fishing streams in Alaska (see fig. 41). They allow diverse vegetation to grow underneath, preventing fragmentation-related impacts while providing anglers access to wading-based fishing areas. This is especially important on salmon streams, where month-long seasons can attract shoulder-to-shoulder densities commonly described as "combat fishing." Such pressure would otherwise decimate riparian vegetation (King and Hansen 2001; Larson and McCracken 1998; Lieptz 1994; O'Toole et al. 2009).

Data indicate users support this infrastructure and appreciate the benefits. A study of bank anglers on the Kenai River (Whittaker and Shelby 2010) showed that 71 percent support fencing and directed trail use (compared to 12 percent opposition), and 66 percent support additional fishing platforms (11 percent opposition). There are tradeoffs for primitiveness and visual quality, but most of the Kenai is not "wilderness-like" and has extensive development (e.g., private homes with docks, along with dozens of parks and campgrounds with boat launches and other facilities).

Conclusions

Differences between visitors and scientists or managers

The Merced River study echoes resource management literature that shows differences between visitors and scientists or managers for variables as diverse as depreciative behavior in developed campgrounds (Clark et al. 1971); acceptability of bare ground and fire ring impacts at wilderness campsites (Shelby et al. 1988); and evaluations of scenic beauty and recreation quality after different logging and silvicultural treatments (Brunson and Shelby 1992). Differences in perceptions of impacts can come from different goals, different specialized training in science or management, or specific responsibilities for protecting or enhancing resources. Differences do not mean that either perspective is somehow wrong, only that we need to recognize and work with those differences.

Recreation Habitat Is Different From Ecological Habitat

Recreation habitat is usually different from ecological habitat. This comes from the different goals and needs of recreation users in riparian zones, and it makes sense when we think of it this way. Resource managers often think carefully about providing for the needs of four-legged animals; the recreation habitat concept explicitly recognizes the need to do the same for two-legged ones. The recreation management field addresses the complexities of managing people in natural settings (Manfredo 2002; Manning 2011). While it is beyond the scope of this chapter to review this field, the conclusions listed here fit with literature that applies to riparian issues.

Users Endorse Ecological Goals

In spite of the visitor-scientist/manager differences listed above, visitors endorse ecological goals and express willingness to "be managed" to support them. It is possible to be cynical about this and assume that what people say is different from what they do (true, and well documented in the social psychology literature), or that even with majority support it only takes a few people behaving badly to defeat restoration efforts (also true, and well documented in the river management literature). But given this, how can our efforts be most effective?

Management Frameworks Can Help

Several decision-making frameworks have been developed by researchers to help planners and managers address recreation impacts or capacities, including Limits of Acceptable Change (LAC; Stankey et al. 1985); Carrying Capacity Assessment Process (C-CAP; Shelby and Heberlein 1986); and Visitor Experience and Resource Protection (VERP) (National Park Service 1997; Manning 2001). Each has differences in orientation, emphasis, terminology, and specific steps, but all were built on the same foundation of scientific and professional literature (Manning 2004), and they are more similar than different in their general approach. The concepts common to most include those discussed below.

Clear Management Objectives

The resource literature has many admonitions about the need for clear management objectives (Heberlein 1977; Hendee et al. 1977). At the broadest level, clarifying objectives may show how scientists' riparian restoration goals differ from visitors' recreation goals. As objectives are further specified, we can identify areas of convergence and areas of divergence, possibly identifying locations where ecological protection trumps recreation access, or vice versa. Clear objectives will help with effective management of riparian areas and the visitors who want to use them.

Precision About Recreation Needs

We need to be precise about recreation needs, recognizing that we cannot provide for all uses in every location. Helpful information includes desired activities, recreation opportunities to be provided, specific groupings of necessary attributes, sites available, locations where sites are needed, sites to be created, and sites that can be eliminated. Demand for and supplies of recreation resources are not evenly distributed. Demand is often concentrated in time (e.g., seasons of the year, days of the week, hours of the day) and space (e.g., at places with attractions). Supply of recreation sites is also uneven (e.g., camping beaches that are available only at low water, steep-walled sections of a canyon that have fewer campsites). Successful management, from ecological or social perspectives, needs to recognize (and when possible take advantage of) such distributions, and solutions are site-by-site as well as cumulative.

Greater Challenges in Lower Development Settings

It is often easier to manage people in front-country settings; development is done with people in mind, and users expect infrastructure, education, and regulations to direct and enforce appropriate behaviors. In backcountry settings, many people value the lack of management (e.g., the Wilderness Act specifies "primitive and unconfined recreation"), even though recreation habitat is more likely to conflict with ecological goals, and small numbers of well-intentioned users can produce unacceptable impacts.

Clarity About Impacts and Indicators

We need to be precise about impacts: what, where, and when. For example, on a cultural resources project, one stakeholder identified "impact to the entire river corridor," while another specified "visitors standing on the hearth" (but did not want to say where that was). Without judging the validity of either concern, we can manage visitors most effectively if we know exactly what and where the problems are, and how they are connected with visitor use. Defining specific, measurable indicators is central to recreation management frameworks. Yosemite managers found that broader CRAM scores and fragmentation indices identified impact issues, but solutions required specific information about site-level impacts and the recreation users causing them.

Specific Standards

Even with clear indicators, it can be challenging to specify when an impact goes from "acceptable" to "unacceptable." Reporting the level of a particular impact is descriptive; deciding whether that level is acceptable is evaluative. This important distinction has been made in the recreation management (Shelby and Heberlein 1986) and riparian ecology (Schmidt et al. 1998) literatures. All resource management decisions have an evaluative component, and most resource conflicts come from disagreements about evaluative judgements such as goals, objectives, or standards.

Diverse Management Approaches

Recreation management literature identifies four general categories of management actions for visitor impacts: (1) capital developments that build things or modify the environment; (2) education to modify human behavior indirectly through persuasion; (3) regulation to modify human behavior more directly; and (4) visitor capacities that limit use. Folded into these actions are technological, cognitive, and structural fixes (Heberlein 2012). For example, a light-penetrating walkway is a capital improvement using a technological fix that removes foot contact with the sensitive area, with a structural fix element that directs and concentrates use. It works only if it gets people where they want to go. Some impacts are unintentional, and education (a cognitive fix) might help. Each approach has complexities and tradeoffs.

How Are Impacts Related to Recreation Use?

The relationship between use and riparian impacts varies by the type and amount of impact:

- Some impacts are related to amount of use, and the relationship is linear (e.g., more vehicles take more space).
- Some impacts are related to amount of use, but the relationship is non-linear; pioneering use causes the most impact, and additional use has proportionally less impact (Cole 1987; Hammitt et al. 2015; Kuss et al. 1990).
- Some impacts are related to type of use rather than amount of use (e.g., just one person using poor camp or human waste practices can cause unacceptable impact).

Successful management actions need to consider these relationships. For example, impacts with linear relationships to amount of use can be effectively managed with a carrying capacity. In contrast, impacts related to type of use rather than amount of use do not respond to capacities but do respond to other management actions (e.g., requiring fire pans or human waste carry-out).

Education Is Not a Panacea; Persuasion Is Challenging

Education is sometimes seen as a panacea for human-caused impact problems As Roggenbuck (1992) states, “If people only understood the impacts they cause, we could get them to behave differently.” Education is also preferred by many managers because it is less obtrusive (Fish and Bury 1981), and users support the idea of “educating those other people who are causing the problem.” The complexities and pitfalls of education as a solution to environmental problems is well documented in Heberlein’s (2012) description of the “cognitive fix.” Education and enforcement may be important parts of any management program, but they require careful “engineering” and public support.

Managing for Ecological and Recreation Habitats

Managing for both ecological and recreation habitat is challenging. For example, higher intensity management (many boardwalks and fences in service of ecological habitat goals) may detract from naturalness (a recreation habitat goal). But in the long run, success with ecological habitat probably requires success with recreation habitat. We provide high quality experiences and nurture a stewardship ethic when people visit rivers and enjoy high quality riparian environments.

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Chapter 10. Intended Versus Unintended Effects During Riparian Restoration Create High Quality Recreation Habitat

Bo Shelby and Doug Whittaker

Introduction

Many of the same features that are good for riparian ecology are good for people. Varying flow regimes, diversity of plants and animals, dynamic geomorphology that produces open beaches and shaded/sheltered areas, and most importantly, the presence of water—all are attractive for recreation, so human use can be an explicit part of the restoration equation.

But when we consider restoration of riparian ecosystems, it is easy to overlook its effects on recreation. The "recreation habitat" concept was introduced in Chapter 9 in this volume. While that chapter explored ways to minimize recreation impacts on riparian habitat through management, this chapter asks whether restoration can create recreation habitat as part of its design. To balance restoration goals with goals for recreation resources, we need to recognize the activities people are trying to pursue, the riparian characteristics they seek, and how restoration choices may impact recreation habitat.

This chapter discusses a typology of anthropogenic effects on riparian habitat, with one dimension considering whether impacts are intentional or unintentional, and another considering whether impacts are positive versus negative. We then discuss information from users about recreation habitat needs, preferences, and decision-making, using camping as an example. This helps evaluate outcomes from restoration efforts in terms of specific recreation resources. A final section provides other examples, using photos with extended captions.

Descriptive and Evaluative Components

Recreation use typically creates changes in biological and physical characteristics. For example, walking across a meadow may crush plants or compact soils, and these changes can be described objectively. But an evaluative dimension is also needed: What are the goals for recreation opportunities and ecological conditions, and are the specific biophysical changes acceptable within those frameworks? An 18-inch-wide band of lost vegetation and compacted soil may be acceptable if defined as a trail, while a wider band or multiple bands may not. Recreation management frameworks distinguish between these kinds of descriptive and evaluative information, encouraging resource managers to explicitly define the conditions they are trying to provide or avoid (Graefe et al. 1990; Manning 2001; National Park Service 1997; Shelby and Heberlein 1986; Stankey et al. 1985).

Applying such a framework to riparian changes, the descriptive component focuses on scientific relationships between restoration actions and different conditions (e.g., measurable changes in species diversity, erosion rates, or percent of bare ground), independent of whether those conditions meet goals or standards. An evaluative dimension (e.g., standards for "acceptable" species composition, channel dynamism, or amount of bare ground) is considered separately to assess whether a restoration project accomplishes specific ecological goals or end-states.

This concept seems simple, but in practice often it is not. Although ecologists may share implicit restoration goals (e.g., increase species diversity or geomorphic dynamism), there are often tradeoffs among outcomes, especially when those outcomes are tangible and specific (Schmidt et al. 1998). In Grand Canyon, for example, operational changes for Glen Canyon Dam could partially restore flood flow regimes that benefit some species (e.g., humpback chub and other native fishes, sub-tropical song birds), but adversely affect others (e.g., trout, the eagles that feed on them, and other birds such as the southwestern willow flycatcher).

Clarifying Goals For Restoration

Aside from these specific examples, several different restoration endpoints are possible, some of which are difficult to define (Cole et al. 2008). A commonly expressed goal is to restore "natural" conditions, but increasing consensus in the scientific literature indicates such conditions are dynamic (Wu and Loucks 1995) and often conflated with current and/or historical human activities (Cole and Landres 1996). We can no longer assume ecosystems are clearly defined, balanced, or self-regulating, or that previous conditions are the most appropriate target for "active management" (Leopold et al. 1963). Cole et al. (2008) examine a range of possible restoration goals from an ecological perspective. They conclude that historical fidelity, biodiversity, species conservation, ecological integrity, and ecological resilience are distinct goals (albeit with some overlap) that may be more or less appropriate, realistic, or measurable in different restoration situations.

Given this, projects to recover "damaged" or "degraded" ecosystems would benefit from explicit definitions of those terms, along with specific goals for improvements. Even the term "restoration" implies a value judgment that the existing condition is "bad" and the restored condition will be "better." In some cases, restoration goals refer to some idealized equilibrium that may never have existed or cannot be replicated. Cole et al. (2008) argue that other terms such as "intervention" or "redirection" better describe intentional actions to alter an ecosystem's path. Regardless of the terminology, these different end-states are human-defined and could include recreation habitat goals.

Restoration can occur at different scales, ranging from specific sites to segments, the entire river, or the watershed. For example, restoration may include rehabilitating specific locations like camps or visitor attraction sites; improving the connectivity of meadows; altering dam-regulated flow regimes that have reduced flooding or sediment deposition; changing widespread grazing; altering trophic cascade effects from species removal; or introducing nonnative species such as tamarisk beetles to reduce an earlier invasive.

This chapter adopts a more flexible definition of riparian areas than some of the technical delineations described in other chapters. On a desert river like the Colorado in Grand Canyon, for example, users may camp above the narrow riparian zone defined by dam-altered hydrologic regimes, often because riparian changes such as tamarisk invasion or beach erosion have constrained camp-able areas. In contrast, wetter environments may encourage camping close to the river on sand or gravel bars that are free of insects or vegetation. For recreation users, the riparian area may include sand beaches, gravel bars, rock cliffs, upland benches, and associated vegetation ranging from grasses to shrubs to large trees. People seek recreation habitats that support their activities, and they may not care about or even recognize formal ecological definitions.

Riparian Changes Due to Human Activity

Many riparian changes are caused by human activities, ranging from dams, grazing, or urbanization, to restoration projects designed to mitigate these kinds of stressors. The typology presented in figure 42 evaluates two orthogonal dimensions: whether the effect was intentional or unintentional, and whether it was positive or negative. We have included several phrases that might be used to describe actions in each quadrant, illustrating the range of ways scientists, practitioners, policy makers, or stakeholders account for different outcomes.

The typology recognizes that some negative changes can occur through unintentional oversight or neglect, while other manipulations have well-understood negative consequences that were intentional but considered acceptable because of other benefits or tradeoffs. For example, good snail habitat on the dam face or in downstream fluctuation zones is a fortuitous unintentional effect of Glen Canyon Dam, just as shade and screening within campsites are unintentional positive effects of tamarisk introduction (although pruning may be required). One could discuss similar questions about different aspects of tamarisk beetle introduction.

In all these cases, changes caused by humans (either positive or negative) are forms of "riparian engineering"; this term recognizes that humans are part of the equation and are responsible for many outcomes, regardless of intentions. As we work to restore riparian

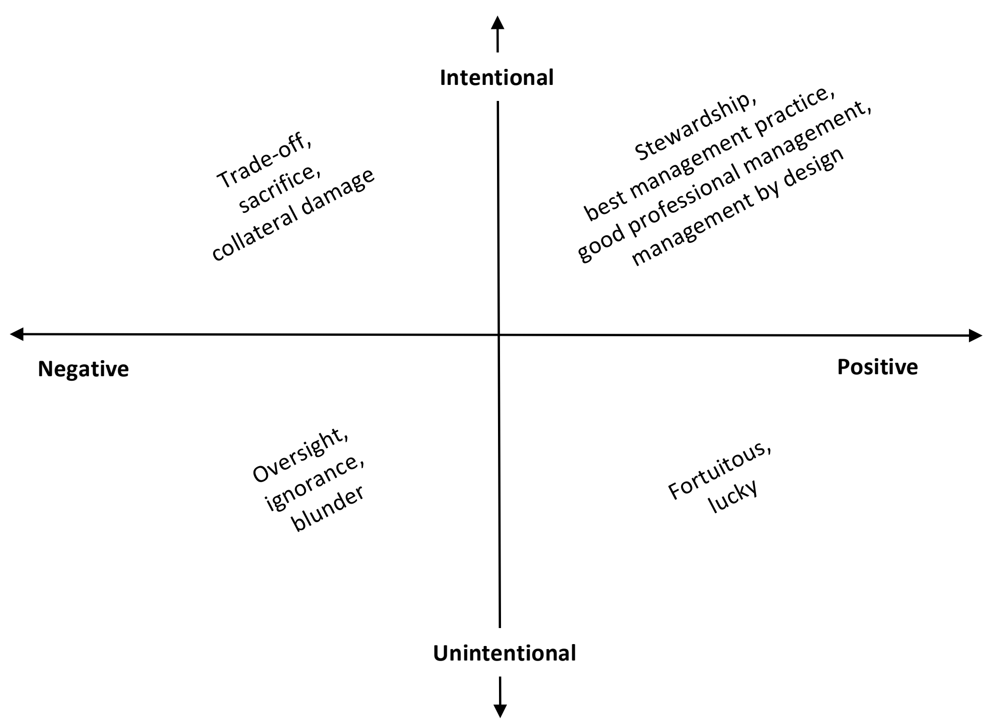

Figure 42—A typology of riparian effects due to human activity.

ecosystems from more negative states to more positive ones, we have the opportunity to consider, and perhaps intentionally improve, characteristics that benefit human activities like recreation. The result is recreation habitat "by design rather than by default" (Shelby and Shindler 1995; Shelby et al. 1992), with increased likelihood that outcomes will be intentional and positive rather than negative.

Creating High Quality Recreation Habitat: The Campsite Example

Campsites are an important riparian feature, particularly for rivers with multi-day trips. They are an excellent example of recreation habitat that can be affected by restoration projects on the orthogonal dimensions shown in figure 42.

Several studies have examined campsite choice in a range of settings to determine characteristics that are important to recreation users. A review developed three categories of campsite attributes (Brunson and Shelby 1990): *Necessity attributes* such as flat ground, nearby water, and shelter/shade are minimum requirements. *Experience attributes* such as screening from other sites, being out of sight/sound of other groups, or good scenery are secondary, although they enhance the overall experience. *Amenity attributes* are a final category of less-important characteristics that may be tiebreakers in the choice process; examples include the amount of bare ground or presence/size of fire rings.

To examine the choice model, Brunson and Shelby (1990) collected data from floaters on Oregon's Deschutes River, a desert canyon stream east of the Cascade Mountains. Summer flows are clear, cold, and fairly constant as a result of an upstream reservoir with power-generating and re-regulating dams. Camping most often occurs on a segment with several Class II-III rapids and a nationally known trout fishery that attracts highly specialized anglers.

As hypothesized, necessity attributes (flat ground, good boat tie-up, and shade) were rated most important. Experience attributes (good fishing water, screened site, and out of sight/sound of other sites) were less important and associated with desired experience outcomes such as "getting away from others." Amenity attributes (free of grazing, little bare ground, and distance from a railroad that runs along the river) were rated least important.

The study also explored alterations in the campsite choice process due to: differences in outcome goals (e.g., anglers care more about fishing water than non-anglers); the need to choose quickly because of competition from other users, fatigue, or threatening weather; and lack of information about sites that lie ahead (because non-motorized boaters approach sites sequentially and cannot go back upstream). Under ideal conditions, users go through the longer three-stage choice process, but constrained conditions may cut the process short any time minimum thresholds are met (similar to Simon's [1956] satisficing concept).

Campsite attribute needs and preferences suggest several priorities for restoration-related riparian engineering. Necessity attributes should be the first concern; the ability to access the site and tie-up, flat ground, and shade protection were most important to Deschutes River users. These attributes are clearly related to restoration options that affect the amount of beach, the type and size of vegetation, or availability of flat bench uplands.

A 1980s restoration project on the Deschutes provides an interesting example. Camps in Sections 1 and 2 are scarce because the left bank of the river is closed to camping by the Warm Springs Tribes. Many of the right-bank sites lack shade, a necessity attribute during hot dry summers. To create more and better camps, agencies and conservation groups cooperatively planted trees along river-accessible upland benches flat enough for camping. But with a long dry summer, growing trees required water. The solution was placing 5 gallon buckets with interpretive signs explaining the need to water trees until they developed deep roots. Tree growth was successful at several sites, although the choice of some tree species (e.g., Russian olive) introduced exotics with other restoration consequences, and cottonwoods created widow-maker hazards as they matured (L. Ripley 2016, personal communication). In short, there were some negative effects in spite of good intentions and a specific recreation habitat goal.

Riparian engineering may also affect experience attributes. Good fishing water was important for many Deschutes River floaters, and this is related to fish habitat goals for game and nongame species. Vegetation is another restoration variable that may provide screening within and between camp sites. Interestingly, amenity attributes such as bare ground and absence of grazing impacts were less important to Deschutes users, supporting the argument in our other chapter that recreation habitat may have dimensions that are different from biological habitat.

A similar campsite selection study of boaters in Hells Canyon on the Oregon/Idaho border showed similar results, with additional attributes related to daily flow fluctuations from power generation releases at Hells Canyon Dam (Shelby et al. 2002). Although floaters and power-boaters had some different needs for parking boats, both groups were concerned about getting stranded if water levels dropped over short periods (a negative outcome from an intentional change in flow regime).

Other Examples

To summarize, human-caused riparian restoration generally alters physical and biological ecosystems, probably changing characteristics that are important for recreation. This means that as we engineer physical or biological habitat, we are also engineering recreation habitat, either by default or by design. It makes sense to choose design, although we don’t always. Using photos with extended captions (figs. 43-58), the following section provides a variety of examples, ranging from site-specific to watershed and regional scales.

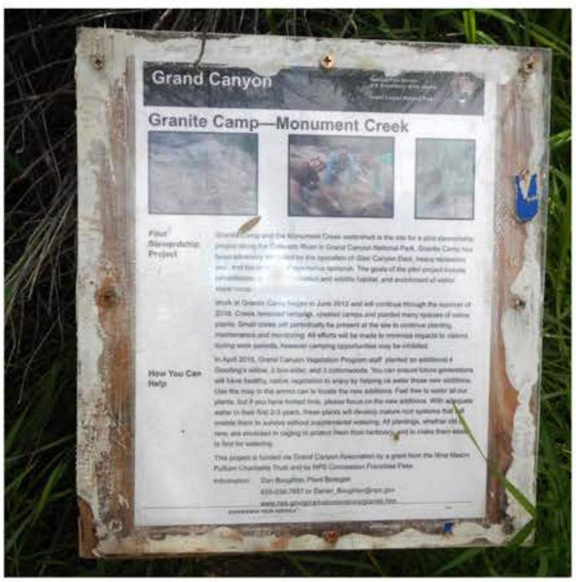

Figure 43—Granite Camp on the Colorado River (top) is a heavily-used "tamarisk camp," where trees have been pruned over the years to provide shade and screened sleeping areas. The restoration project described by this onsite sign (center left) is one of the few places in Grand Canyon where trees have been extensively planted and cultivated (center right, and in top photo) to replace tamarisk dying from introduced beetles (some trees have been cut and removed, bottom left). But it will be some time before these new trees reach the maturity of "old growth" tamarisk or replace its recreation benefits in terms of shade, shelter, or screening.

Figure 44—These tamarisk at Carbon Creek camp on the Colorado River are the best available shade and shelter for this kitchen setup, even though they are blackened and dying. In contrast to the cultivation at Granite Camp, most Grand Canyon camps (like this one) have no "engineered" vegetation to replace beetle-killed tamarisk. New vegetation may regrow spontaneously but may not provide good recreation habitat.

Figure 45—This Google Earth satellite photo shows the areas just above Phantom Ranch in Grand Canyon, with the Kaibab Trail and bridge crossing immediately downstream. There is high demand for Cremation Camp (on river left) because the 12-mile Inner Gorge area upstream has few camps per mile, and many parties want to camp near the Kaibab and Bright Angel Trails that provide access for passenger exchanges. In addition to resource protection goals, the site has been engineered to provide good recreation habitat for multiple groups. The "one party per camp" norm that exists in most of the Canyon is explicitly suspended here to increase capacity (insets show different landings that allow for camp-sharing).

Figure 46—This carefully engineered tent site at Crystal Camp in Grand Canyon provides excellent camping habitat. Data show visitors appreciate cues that identify where to camp and indicate that their activities are not causing unacceptable impacts. Such improvements protect the riparian area by keeping camps in the more resistant post-dam riparian zone and discouraging use in the more fragile native vegetation higher up the bank.

Figure 47—Funneling use to a single area that can then be hardened is an effective strategy to get visitors through sensitive riparian areas (Little Nankoweap camp on the Colorado River in Grand Canyon shown here). Top photos show steps from the boat landing to the camp area, soon after construction in 2005 (top left) and needing some work in 2016 (top right). Lower photos show the trail from camp up to Little Nankoweap Canyon and the Nankoweap granary, some of it in good shape (bottom left) and some not (bottom right).

Figure 48—Even in remote locations like the bottom of the Grand Canyon, riparian restoration may need to accommodate large amounts of recreation use, as suggested by parked boats at Deer Creek (top), Havasu (bottom left), and Elves' Chasm (bottom right). Almost all Grand Canyon trips stop at these popular attraction sites.

Figure 49—Good trail design provides hiking habitat by identifying routes that get people where they want to go (trail to upper Deer Creek in Grand Canyon shown here). At the same time, trails protect restoration projects by keeping vegetation loss and soil compaction at acceptable levels.

Figure 50—Techniques for identifying trails, keeping people on them, and minimizing impacts include those shown here: border rocks (top left), marking routes with cairns (top right and bottom left), and steps in steeper segments (bottom right). Photos are from Nankoweap and Carbon Creek areas in Grand Canyon.

Figure 51—This Google Earth satellite image of Nankoweap on the Colorado River in Grand Canyon shows where the Park Service developed and clearly marked a "best" trail for the popular hike to the Anasazi granary high on a cliff (lower inset), with a spectacular view down the Canyon (upper inset). Before this restoration, which included designating camps and day use areas, river runners created a web of dozens of trails.

Figure 52—Tamarisk has overtaken other riparian vegetation on many Southwestern rivers, creating dense stands that typically eliminate open space and flat ground needed for camping. Resource agencies have released tamarisk beetles to combat the species, a program claimed to be among the most carefully studied introductions ever. Although this is controversial, beetles have clearly reduced the amount of living tamarisk and probably its spread to other areas. But as shown in this photo from the Green River in Stillwater Canyon, impenetrable dead trunks and branches prevent recreation use and create fire hazards.

Figure 53—Piles of cut tamarisk (top photo) in Three Canyon camp at Trin Alcove Bend in the Green River's Labyrinth Canyon. Despite the extensive restoration effort, remaining stumps and uneven ground (bottom left) interfere with good camping habitat, and dense willow may quickly take over the area (bottom right). Some tent spots have been cleared in nearby oak uplands (center) and future high water may deposit enough sand to reduce the problem.

Figure 54—A Park Service/Canyon Country Youth Corps crew removes thickets of beetle-killed tamarisk at the confluence of the Green and Colorado Rivers (top left), part of what one crew-member described as a career goal to rid the area of this invasive. Trees up to 12–inches in diameter were cut with chainsaws (bottom left) and thrown in the river (top right and center right). Downstream, driftwood with saw-cut ends float on the river (center and bottom right), piles up on shore, or eventually winds up in Lake Powell. It is not clear whether this "new" form of driftwood is considered good, bad, or just a tradeoff of clearing invasive tamarisk, but regulations in some jurisdictions prevent river runners from using slash for firewood.

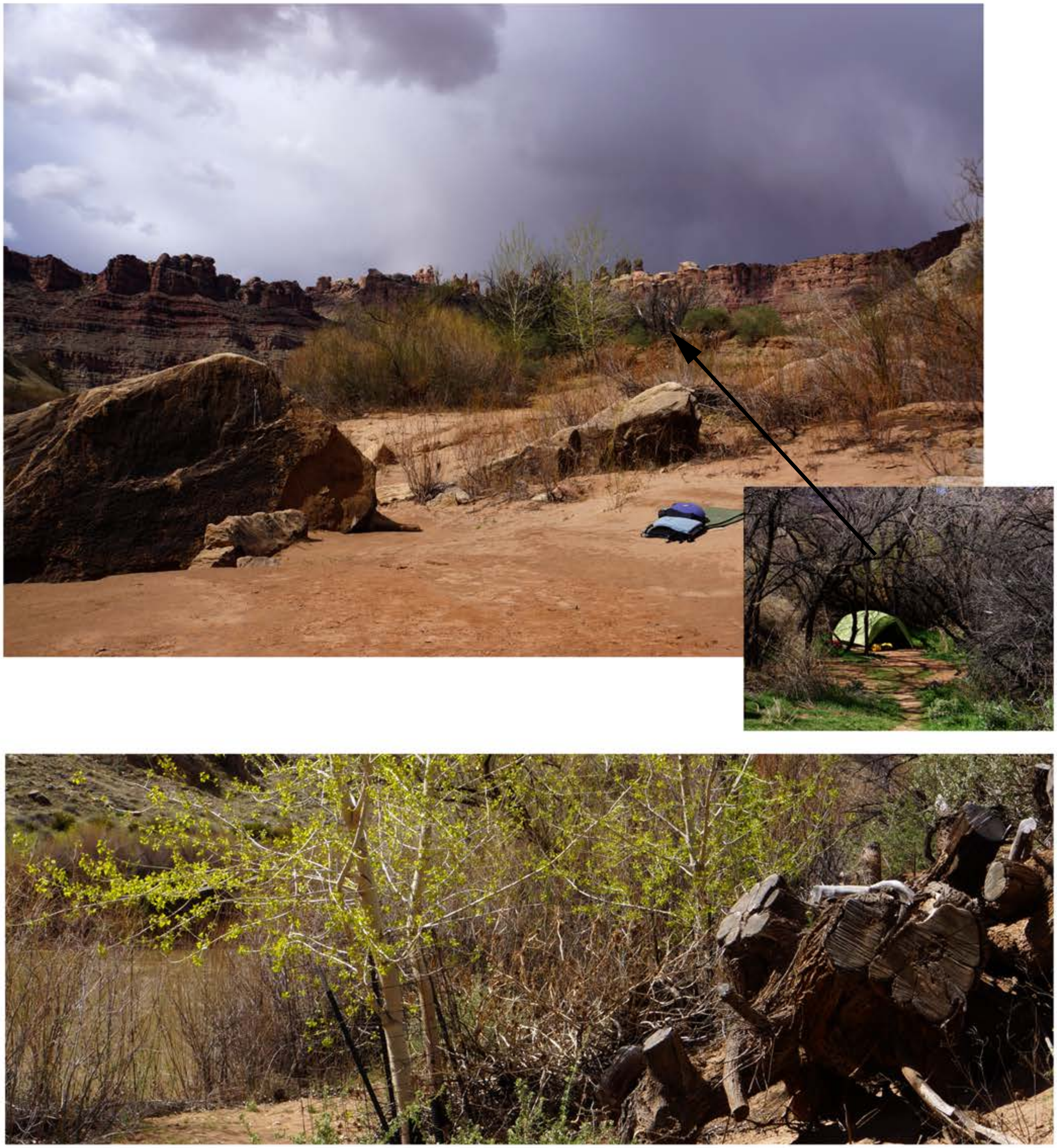

Figure 55—On a cold and stormy day at Spanish Bottom in Cataract Canyon below the Green/Colorado River confluence, even dead tamarisk with blackened trunks provide better camping habitat (inset) than a more aesthetically pleasing open beach (top photo) that was lashed by wind-driven sand and rain. Lower photo shows planted cottonwoods protected by staked wire cages; tamarisk stumps like those in the foreground remain after the trees have been cut and the slash removed. Although some cottonwoods were planted soon after beetle introduction, it will take many years before shade and shelter from these newer trees replace the "old growth" tamarisk (something this restoration project appears to recognize in its design).



Figure 56—A beach area recently protected from grazing on Oregon's John Day River (top photo, taken at a higher spring flow) shows willows establishing along the lower summer water line: willows may eventually take over or cut off access to the beach, decreasing recreation habitat. Mitigation for such effects can be included in riparian engineering during restoration projects. Pioneering restoration research has been done in the John Day and Deschutes River watersheds east of the Oregon Cascades. This work showed how riparian vegetation in historically grazed locations (center left) was markedly different from areas protected from grazing by roads or railroads (center right). Cottonwood plantings (background of lower right photo) are part of the effort, but a pleasant surprise has been dramatic recovery after removing livestock or simply installing watering tanks (on trailers with solar-powered pumps) higher in the riparian zone.

Figure 57—This whitewater park in Salida, Colorado, is a large-scale river restoration that includes extensive channel modifications as well as bank improvements. Specifically designed to provide hydraulic features for whitewater boating (ColoradoKayak.com photo, bottom left), the park creates diverse recreation habitats on and off the water, including pathways (top left), seating areas (top right and bottom right), and a "downtown restoration" setting for shops and restaurants (out of sight upstream on river right). Ironically, on-shore areas provide for much larger numbers of recreation-days, compared to boating and other water activities.

Figure 58—Signs marking restoration areas take many forms: short vs. long, less vs. more informative, friendly vs. more demanding instructions, and messages that sometimes appear incongruous (like "not a trail" with an obvious trail behind it, bottom right). The detailed sign at the Salida Whitewater Park (top) conveys an interdisciplinary message about habitat benefits for boaters, aquatic life, and riparian vegetation.

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