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Fire Ecology and Management in Lowland Riparian Ecosystems of the Southwestern United States and Northern Mexico

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Abstract

Lowland riparian ecosystems, defined as those occurring at elevations at or below 5,000 feet (1,564 meters), constitute a small fraction of total land area in the southwestern United States and northern Mexico, yet they are extremely important to human livelihoods and biotic communities. In the hotter and drier conditions projected under ongoing climate change, riparian ecosystems are increasingly critical to the well-being of humans and wildlife. Riparian areas have been modified in various ways and to a large extent through human endeavor to utilize resources more predictably. These alterations often interfere with multiple and complex ecological processes, making riparian areas vulnerable to disturbance and change. Few naturally functioning riparian areas remain in the region, and those that do are imperiled by climate change, groundwater pumping, land use, and altered disturbance regimes. Some evidence suggests that fire regimes are changing in southwestern riparian zones; wildfires may be increasing in frequency and severity. This literature review summarizes and synthesizes the state of the knowledge of wildfire and prescribed fire effects on physical processes and vegetation, and postfire rehabilitation. Changing fire regimes are likely to have drastic and potentially irreversible effects on regional biodiversity and ecosystem function. However, options are available for managing riparian ecosystems that could make them more resilient to fire and climate change. This study is intended to inform management and identify gaps in systematically reviewed literature.

Keywords: riparian ecosystem, fire, fuel, restoration, management, plant, hydrology

Cover: Clockwise, from upper left: (A) Beaver dam, Bill Williams River National Wildlife Refuge, Mohave County, Arizona, summer 2012 (photo: Amanda D. Webb). (B) Santa Elena Canyon on the border between the United States and Mexico in Big Bend National Park, Brewster County, Texas, February 2016 (photo: Amanda D. Webb). (C) A prescribed fire burning giant reed in Big Bend National Park (photo: National Park Service). (D) Cottonwoods at San Pedro Riparian National Conservation Area, Cochise County, Arizona, circa 2003 (photo: Amanda D. Webb, Bureau of Land Management).

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Acknowledgments

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Executive Summary

Riparian ecosystems in the southwestern United States and northern Mexico provide abundant resources for human and ecological communities in desert ecoregions. Within the last 100 to 150 years, riparian areas have been modified extensively by modern human uses, including direct alterations of rivers by dams, diversions, and clearing floodplains for agricultural purposes (Chaney et al. 1990; Glenn and Nagler 2005; Krueper 1993; Stromberg et al. 2003; Webb and Leake 2006). These modifications have altered the complex ecological and hydrological processes of these systems, resulting in significant changes in the physical environment and biota (Jemison 2003; Nagler et al. 2011; Webb and Leake 2006). For this report, riparian areas are defined as ecosystems wherein soil moisture, landform, and vegetation are influenced strongly by the perennial, intermittent, or ephemeral flow of surface water and subsurface water associated with nearby water bodies, such as rivers, streams, springs, and cienegas (a wet meadow type of wetland).

Significant alteration of hydrological regimes is often associated with successful nonnative plant colonization and naturalization (Glenn and Nagler 2005; Nagler et al. 2011; Poff et al. 2011; Ringold et al. 2008; Webb and Leake 2006). Altered hydrological regimes and sedimentation processes as a result of river impoundment can reduce or eliminate germination and recruitment of cottonwood while allowing germination and recruitment of tamarisk (also known as saltcedar) to continue or increase (see table A.1 in Appendix A for list of plant species mentioned in text) (Beauchamp and Stromberg 2007; Glenn and Nagler 2005; Merritt and Poff 2010; Mortenson and Weisberg 2010; Ringold et al. 2008; Webb and Leake 2006). Tamarisk has become widespread along many river reaches (Busch and Smith 1993; Drus 2013; Glenn and Nagler 2005), and it is a highly combustible nonnative plant.

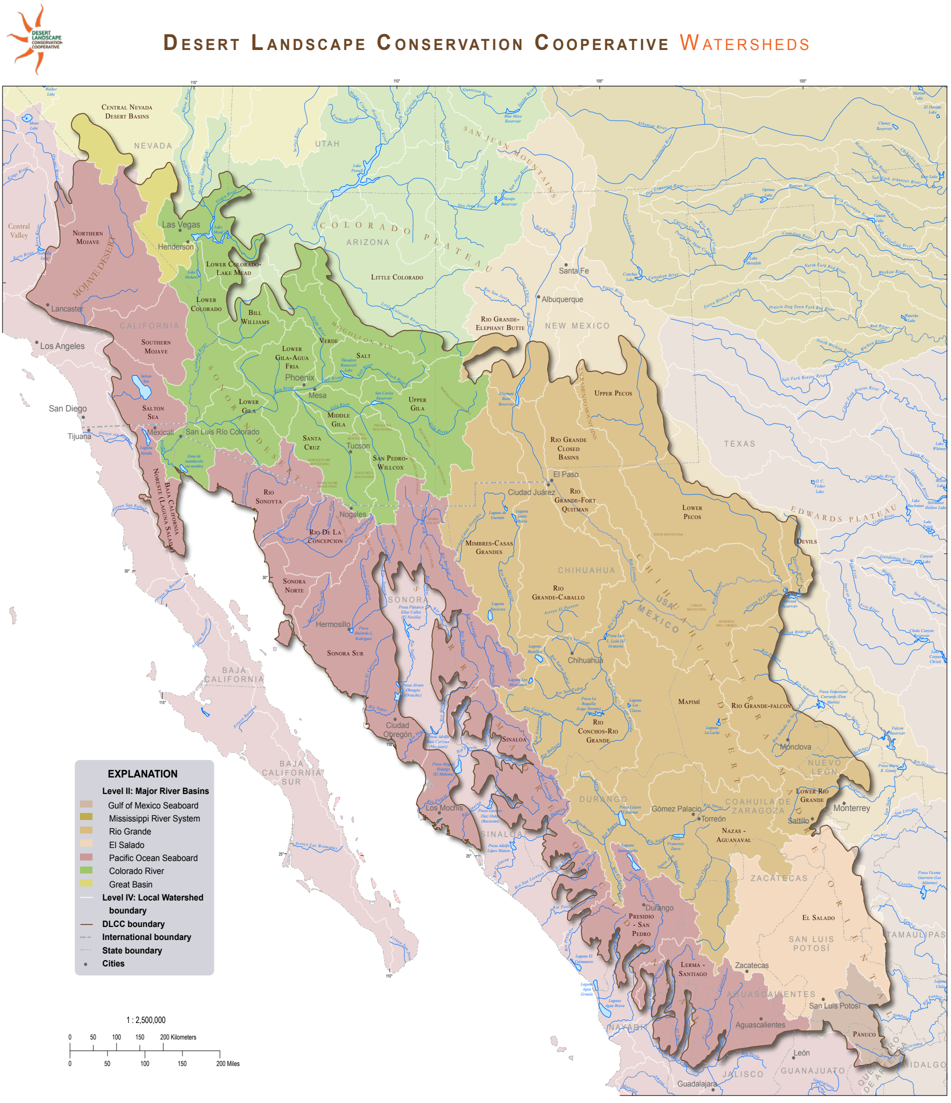
A small but growing body of literature suggests that fire frequency and fire severity are increasing in many southwestern riparian ecosystems (Busch 1995; Busch and Smith 1993; Drus 2013; Friggens et al. 2013; Jemison 2003; Parker et al. 2005; Smith et al. 2009; Stromberg and Rychener 2010). This is especially evident where hydrological processes that support native trees have been disrupted and nonnative plants make up a significant portion of the vegetation community. For example, where tamarisk has become the dominant vegetation type in many lowland riparian systems, its highly combustible fine fuels and adaptations to high frequency fire are driving changes in riparian ecology and fire regimes. However, there is limited research on fire impacts in southwestern riparian zones. Responses of riparian vegetation to fire are not well documented, yet this information is important for ecosystem planning and management (Stromberg et al. 2009).

Managers, researchers, regional coordinators, and others working with the Desert Landscape Conservation Cooperative (Desert LCC) noted that while fire has been studied extensively at watershed scales, in montane systems, and in grasslands, there is relatively little information about fire in riparian ecosystems, and especially at lower elevations. Desert LCC was a self-directed, non-regulatory partnership addressing the conservation challenges posed by climate change and other landscape-scale ecosystem stressors in the southwestern United States and northern Mexico (Desert LCC n.d.). Desert LCC partners identified the need for an assessment of the state of the knowledge of fire in lowland riparian ecosystems in the region. To help address this priority, the Desert LCC partnered with the University of Arizona and the USDA Forest Service to produce and publish this report. This report summarizes a review of literature concerning wildfires and prescribed fires in lowland riparian ecosystems to help inform management, restoration, and post-fire rehabilitation, and to identify gaps in available science.

The literature review was conducted systematically using keywords developed with the help of Desert LCC partners and subject matter experts. Literature was searched for in four databases, and different types of literature were reviewed, such as conference proceedings, technical reports, field reports, journal articles, theses, and dissertations. The literature included in this report meets the following criteria: It 1) addresses the direct effects of fire on the stream corridor in lowland (versus montane) riparian ecosystems, including plants and physical characteristics; 2) focuses on lowland streams occurring at elevations at or below 5,000 feet (1,564 meters); and 3) includes project areas within the Desert LCC region (fig. 1). This water-limited region includes the Mojave, Sonoran, and Chihuahuan Deserts, and neighboring semiarid highlands. In this report, this study area is referred to as the Warm Deserts of North America (WDNA). For comparison, some literature is included from areas outside the WDNA, such as arid regions of southern California, Australia, and southern Africa.

The literature search identified 67 sources meeting these criteria, including 40 observational studies and experiments. Details on literature sources are presented in tabular form in Appendices B and C, which include literature source type, subject, area of interest, and fire effects and fire properties studied for all fire-related literature that met the criteria for inclusion in this review. Section 4 contains summaries of literature found and is organized by subject, plant community, and physical ecosystem characteristics.

In general, current literature suggests that the frequency and severity of lowland riparian fires are increasing, although the extent and magnitude are unknown. Several key reinforcing factors appear to be driving this change. Dams, diversions, and other human impacts, and the



Desert LCC Lambert Azimuthal Equal Area projection
World Geodetic System Datum of 1984

River basin boundaries from the North American Environmental Atlas by the Commission for Environmental Cooperation, 2010
Hydrography, roads, populated places, and political boundaries from the National Atlas of the United States, 2004
Natural Earth 1:10m Physical Labels (3.0.0) derived from Patterson's Physical Map of the World, 2008

Figure 1—The Desert Landscape Conservation Cooperative geography, referred to in this report as Warm Deserts of North America (WDNA), encompasses the Mojave, Sonoran, and Chihuahuan Deserts, and parts of several major watersheds (source: Desert LCC n.d.).

consequences of climate change, have significantly altered the hydrology of many streams. Such changes have led to the demise of many native riparian plant communities, while contributing to increases in the extent and distribution of tamarisk, and possibly other nonnative plants that are characterized by different fire histories and tolerances. Increases in fire frequency and severity contribute to the decline of native riparian plants while, at least in some cases, contributing to further increases in the extent and distribution of nonnative plants. Some studies point to instances where the invasion of tamarisk into riparian areas formerly dominated by native deciduous woodland communities is changing the fire regime to the further detriment of native species, such as cottonwood, willow, and mesquite. Although the effects of other nonnative species (e.g., Russian olive, giant reed, common reed, Siberian elm, Johnsongrass) on fire characteristics are not well documented, anecdotal evidence combined with studies from other regions of North America suggests that these species could be similarly impacting riparian fire behavior and fire regimes in the WDNA.

Many native plant species can recover post-fire, but conditions often favor nonnative species. Certain kinds of fires and fire regimes are interacting with altered flow regimes to accelerate the replacement of native plants with nonnative plants.

Despite their ability to regenerate post-fire, Fremont and Rio Grande cottonwood are at serious risk of rapid decline due to the combined effects of changing hydrological regimes and increasing fire frequency and severity. Tamarisk is well adapted to short fire return intervals, and high fire severity further facilitates its presence in riparian ecosystems. More frequent and severe fires are likely to result in continued large-scale mortality of mature Fremont and Rio Grande cottonwood trees in their reproductive years. Where natural flow regimes are more intact and tamarisk is suppressed by native vegetation, native trees may be able to maintain their dominance in the post-fire landscape.

One of the key findings of this review is that fire regimes, fuel types, and post-fire outcomes are influenced strongly by hydrological regimes. Although fire is a natural disturbance process in riparian ecosystems that provides many important ecological benefits for biota, changing fire frequencies and severities in riparian ecosystems may be exacerbating undesirable changes in vegetation composition and structure. The results of the literature review underscore the importance of management and restoration strategies in reducing undesirable effects of fire on riparian ecosystems, but the long-term success of efforts will depend on how well they address the root causes of increasing fire frequency and severity. A priority is to identify and implement management actions that couple promoting natural fires and fire regimes that are not detrimental to native ecological communities, with management actions focused on hydrological processes. Management options include prescribed fire combined with other restoration actions, implementing environmental flows, actions that promote native vegetation, and the use of applied conservation genetics to inform preservation and restoration efforts. Best management practices will vary depending on site conditions, and incorporating local knowledge into the development of best practices is likely to be critical to success in many places.

An important conclusion of this literature review is that altered fire regimes are contributing to rapid ecological change along rivers in the southwestern United States. Flooding is generally considered the primary driver in riparian ecosystems, but in the absence of natural flooding, fire may play a more influential role in shaping riparian ecosystems. These two foundational changes in the dynamic nature of riparian ecosystems indicate that ecological communities are changing at many levels from processes, to plant structure and composition, and on up to higher trophic levels that include wildlife species of special concern. We expect these changes to be reflected in ecological communities as the distribution and relative abundance of many species are changing along with processes that provide their habitat. Riparian fire effects on ecological communities extend beyond the riparian corridor when riparian areas that acted formerly as fire breaks become channels for fire spread.

A holistic approach to riparian ecosystem management is essential to address the underlying causes of vegetation change which result in fires that threaten native ecological communities. Whether management goals are to safeguard high quality riparian areas, improve degraded systems, or prevent fire spread through these sensitive areas, the underlying principle explored here applies: Lowland riparian areas are conditioned by the interactions of hydrology, species composition, and fire disturbance. More information is needed on the fire ecology of riparian trees, prescribed fire, and post-fire rehabilitation. There is also a need for fuel and fire behavior models for the unique setting in riparian ecosystems, and specifically models for tamarisk stands. Including input from managers of riparian ecosystems at all stages of the research process is critical to ensure study results can be applied to decisionmaking and on-the-ground management actions.

Section 1 - Introduction

Riparian areas are among the most diverse and productive ecosystems in desert bioregions, supporting floras and faunas that are typically more biologically rich than adjacent upland ecosystems (Brown et al. 1977; Carothers et al. 1974; Chaney et al. 1990; Ffolliott et al. 2003; Johnson 1989; Krueper 1993; Ohmart and Anderson 1986). In the binational region of the southwestern United States and northern Mexico, rivers, streams, springs, and their associated biota have provided critical sustaining resources such as water, food, shelter, and transportation corridors to human, fish, and wildlife communities in deserts for thousands of years (Cordell 1984; Fish et al. 2006; Johnson 1989). Vegetation in riparian ecosystems provides breeding, migration, and wintering habitat for native arthropods, game species, birds, small mammals, reptiles, and amphibians. Ecosystem services for humans, such as recreational opportunities, pollinators, and firewood are also provided.

The word "riparian" has been defined for a variety of purposes, including legal determinations, mapping, and research across many scientific disciplines. Therefore, many definitions are in use, some with nuanced differences (Zaimes 2007). For this report, riparian areas are defined as ecosystems in which soil moisture, landform, and vegetation are influenced strongly by the perennial, intermittent, or ephemeral flow of surface water and subsurface water associated with nearby water bodies, such as rivers, streams, springs, and cienegas (a wet meadow type of wetland) (fig. 2). More specifically, this report focuses on lowland riparian ecosystems that occur at or below 5,000 feet (1,564

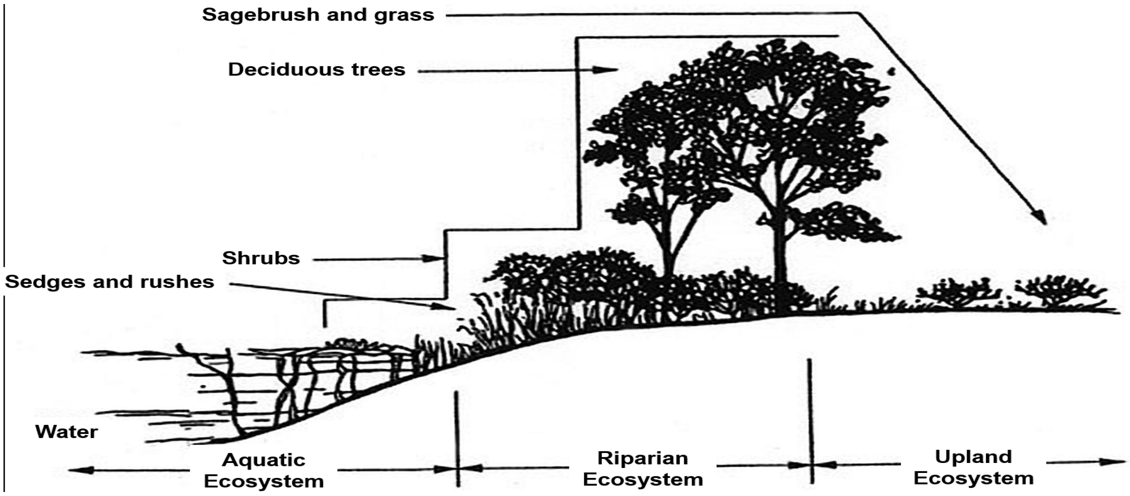


Figure 2—Schematic delineating upland, riparian, and aquatic areas. Reproduced from BLM (1991).

meters) in elevation in the southwestern United States and northern Mexico.

Riparian areas provide a wide range of ecosystem services. Vegetation reduces nonpoint source pollution in rivers by filtering out sediments, chemicals, and nutrients from water flowing over land after precipitation events (Zaimes 2007). The roots of streamside plants help prevent excessive erosion during high flow events, and the presence of vegetation in the areas surrounding the active channel helps slow water velocity when floods cause water levels to rise above river banks (NRCS 1997). Root systems promote groundwater infiltration (Ffolliott et al. 2003). High quality fish habitat depends on adequate water supply and suitable water temperatures for spawning and survival (Gresswell 1999). Riparian trees and shrubs shade surface water, preventing high temperatures from impacting water quality (NRCS 1997) and reducing water lost to evaporation. Riparian areas also provide many recreational opportunities for people, including hiking, swimming, bird and wildlife viewing, boating, horseback riding, fishing, hunting, and camping. The amount of groundwater recharge from stream channels varies across the landscape, but in some places, precipitation and flood flows in channels contribute significantly to the recharge of aquifers (Baillie et al. 2007; Newman et al. 2006; Plummer et al. 2004; Simpson et al. 2013).

Riparian corridors contribute significantly to regional biodiversity; for example, riparian species constitute 33 percent and 40 percent of the total native flora in the Sonoran and Madrean floristic provinces, respectively (McLaughlin 2003; Naiman et al. 1993). Eighty percent of vertebrate species in New Mexico and Arizona depend on riparian areas during at least half of their life cycles, and over half of these are riparian obligates (Chaney et al. 1990). Riparian ecosystems also provide habitat for many threatened or endangered species. Of the 40 Federally listed threatened or endangered wildlife species in New Mexico, including several endemic species, at least 70 percent require aquatic or riparian habitat, or both, to feed, reproduce, or carry out their life cycles (USFWS 2015). The number of species inhabiting riparian areas varies by taxonomic groups and protected area, but it is generally significant (table 1). Table A.1 in Appendix A lists all plant species mentioned in the text.

Biological diversity has multiple benefits beyond intrinsic value. Riparian zones provide food, water, and cover to diverse wildlife (Brown et al. 1997; Chaney et al. 1990; FFfolliott et al. 2003; Kreuper 1993), contributing to the allure of desert rivers as ecotourism destinations and thus supporting local economies. As observed in other ecosystems, high plant biodiversity may enhance riparian ecosystem services by aiding erosion control, promoting nutrient cycling, buffering nonnative species invasions, increasing diversity and abundance at higher trophic levels,

Table 1—Number of species in major taxonomic groups observed in four protected riparian areas in Arizona: Bill Williams River National Wildlife Refuge (BWRNWR) (USFWS 2013a); San Pedro Riparian National Conservation Area (SPRNCA) (BLM n.d.); San Bernardino National Wildlife Refuge (SBNWR) (USFWS 2013b); Leslie Canyon National Wildlife Refuge (LCNWR) (USFWS 2013b).

Taxon	BWRNWR	SPRNCA	SBNWR + LCNWR
Aquatic species	192 (23 dragonflies)	2 native fish	8 fish
Birds	355	350	335
Herptiles	41	>40	43 reptiles, 13 amphibians
Insects	23 dragonflies, 40 butterflies	-	500 bees, 75 butterflies
Mammals	57	>80	67
Total area (acres; hectares)	6,100; 2,468	57,000; 23,067	5,134; 2,078

and increasing overall ecosystem resilience (Balvanera et al. 2006; Folke et al. 2004; Naeem 2006).

Lowland riparian ecosystems in the southwestern United States have been modified extensively by human uses, most significantly in the last 100 to 150 years. These changes include increases in the number and size of dams, water diversions, channelization, grazing, forestry, clearing, groundwater pumping, establishment of agricultural fields in floodplains, introduction of nonnative plant species, urban expansion, and the extirpation of beavers (*Castor canadensis*) during the fur trapping era (Chaney et al. 1990; Krueper 1993; Petrakis et al. 2017; Poff et al. 2011; Stromberg et al. 2003; Webb and Leake 2006). In response to physical modifications and water use, the species composition and spatial extent of riparian plant communities have changed over time (Petrakis et al. 2017; Webb and Leake 2006); in many cases, native riparian vegetation has been eliminated altogether (Ohmart and Anderson 1986; Petrakis et al. 2017; Poff et al. 2011). Today, riparian ecosystems are threatened not only by the negative impacts of anthropogenic land and water management, but by the pervasive effects of changing climate on regional hydrology (Poff et al. 2011; Smith and Finch 2017). Since the 1960s, there have been increasing reports that fire also threatens these ecosystems (Poff et al. 2011).

Given their inherent value as reservoirs of biological diversity and ecosystem services, and the complex influences between humans and riparian ecosystems, the decline of riparian ecosystems is alarming, particularly in light of new challenges presented by climate change. Human-induced changes to riparian ecosystem characteristics can decrease ecosystem resilience to disturbance and environmental change (Seavy et al. 2009; Stromberg et al. 2012) and affect disturbance regimes

(Folke et al. 2004; Merritt and Poff 2010; Mortenson and Weisberg 2010).

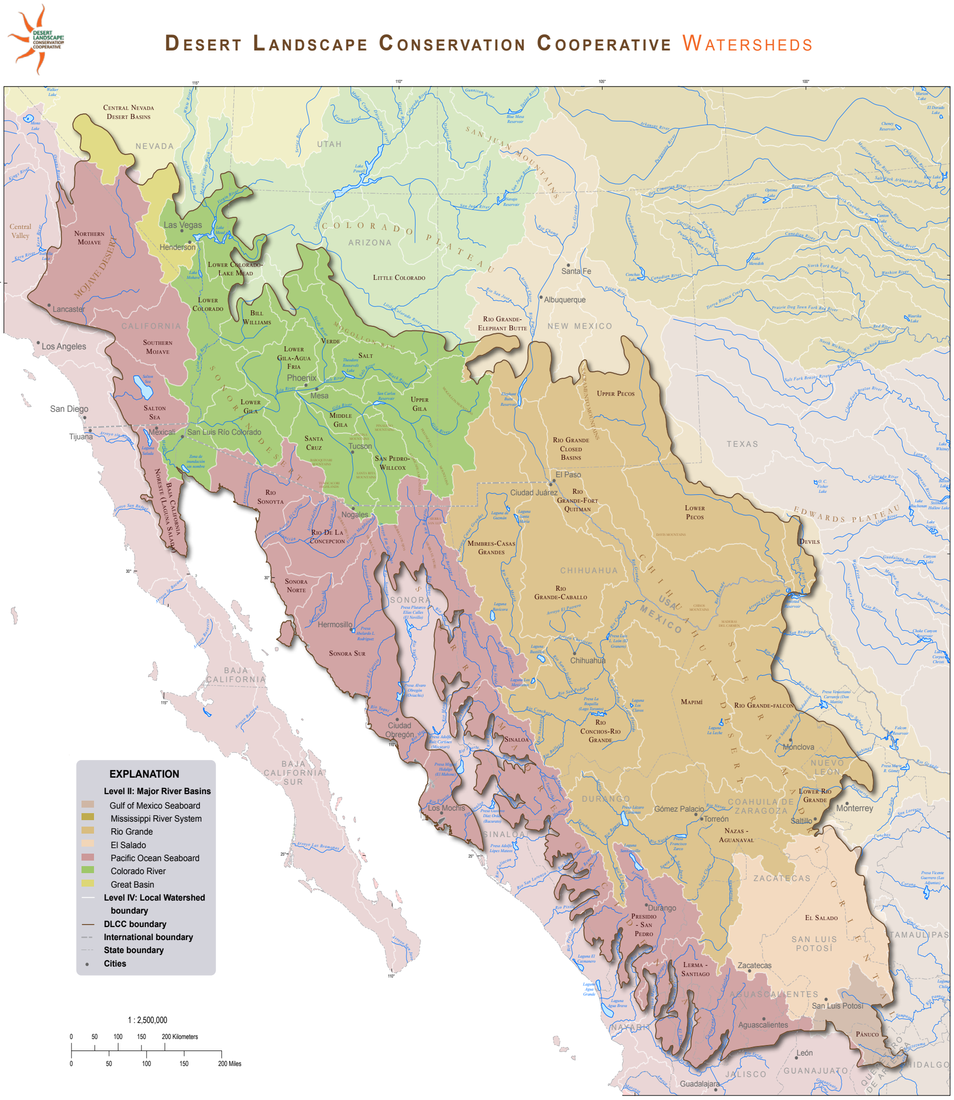
1.1 Purpose

Historically, some riparian ecosystems functioned as natural fire breaks, slowing or stopping fires moving across the broader landscape due to their higher fuel moistures relative to adjacent ecosystems, even during drought (Brooks and Minnich et al. 2006; Dwire and Koffman 2003; Friggens et al. 2013; Smith and Busch 1992). A small but growing body of literature suggests that fire frequency and severity are increasing in many riparian ecosystems where nonnative plants make up a significant portion of the vegetation community (fig. 3) (Busch 1995; Busch and Smith 1993; Drus 2013; Friggens et al. 2013; Jemison 2003; Parker et al. 2005; Smith et al. 2009; Stromberg and Rychener 2010). However, the extent and degree to which fire regimes are changing, and the effects of these changes on riparian ecosystem function and resilience, are poorly understood.

These observations highlight the need for increased understanding of fire in riparian ecosystems. Specifically, information about how riparian vegetation responds to fire is important for ecosystem planning and management (Stromberg et al. 2009). In this era of uncertainty about the ongoing effects of a changing climate, understanding these issues at a bioregional scale across jurisdictional and international boundaries is essential to inform sustainable natural resource management, policy, and decisionmaking.

Figure 3—A fire crew monitors a fire burning in standing dead tamarisk, a nonnative invasive plant (photo: U.S. Fish and Wildlife Service).

This study was developed in collaboration with managers, researchers, regional coordinators, and others partnering with the Desert Landscape Conservation Cooperative (Desert LCC), a self-directed, non-regulatory, regional partnership that addressed the conservation challenges posed by climate change and other landscape-scale ecosystem stressors (Desert LCC n.d.). Desert LCC partners had noted that although the ecological role of fire has been well studied in many montane forest and grassland systems, there was comparatively little information available about fire in lowland riparian ecosystems. In response, they formed a team focused on guiding research and improving understanding of fire in lowland riparian ecosystems. As part of a larger effort to achieve these goals, the team requested support for a literature review of fire effects in lowland riparian ecosystems of the Desert LCC



Desert LCC Lambert Azimuthal Equal Area projection
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River basin boundaries from the North American Environmental Atlas by the Commission for Environmental Cooperation, 2010
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Figure 4—The Desert Landscape Conservation Cooperative geography, referred to in this report as Warm Deserts of North America (WDNA), encompasses the Mojave, Sonoran, and Chihuahuan Deserts, and parts of several major watersheds (source: Desert LCC n.d.).

region. This geography is referred to in this report as Warm Deserts of North America (WDNA), a water-limited region that includes the Mojave, Sonoran, and Chihuahuan Deserts, and neighboring semiarid highlands (fig. 4). With funding from the U.S. Fish and Wildlife Service and Bureau of Reclamation, the Desert LCC partnered with the University of Arizona and the Forest Service to conduct this literature review.

As a product of these partnership efforts, this study is intended to inform management, restoration, and post-fire rehabilitation of riparian systems by summarizing and synthesizing the state of the knowledge of the direct effects of wildfire and prescribed fire on physical processes and vegetation in lowland riparian ecosystems, and to identify critical research needs. This systematic literature review answers the following questions:

1. What does the published literature suggest about patterns, drivers, and trends in fire severity and frequency?
2. How do fire effects vary among species, and what might these results mean for research and management?
3. What does the literature suggest about the overall impact of riparian fire on ecological communities?
4. What knowledge gaps exist in the current literature, and what further research is needed to improve understanding of riparian fire effects and management?

This report is designed to be relevant for a variety of users, including those concerned about watershed and river health or management; climate change; biodiversity in the southwestern United States and northern Mexico; the future of riparian and aquatic environments; and water-dependent communities, economies, agriculture, and cultural practices along lowland rivers. Water and land managers will find literature summaries in tabular form in Appendices B and C for quick reference; these summarize riparian fire literature on specific plant communities, the physical environment, and prescribed fire effects. Restoration project managers can use this report to inform on-the-ground project decisions, planning, and monitoring. For researchers, there are recommendations for future research, an extensive bibliography, and the literature assessments in tabular form in Appendices B and C.

Section 2 - Fire in Context: Lowland Riparian Ecosystems

> *"Historically, the river flowed perennially throughout most of its reaches except during severe drought periods. The main channel often shifted its location in the floodplain in response to changes in flow quantities and other natural features and occurrences. Riparian forests (bosques), wetlands, swamps, and marshes bordered the river along many of its reaches. Over time, sedentary farmers and residents have occupied the floodplains and confined the river to an unnaturally narrow corridor in order to maximize their use of the fertile and easily exploited lands found there. Unfortunately, the occupation and exploitation have led to the loss of many of the resources and values that were sought from the* [Middle Rio Grande]*..."*
>
> —Jemison (2003, p. 85)

The biological diversity of the Warm Deserts of North America (WDNA) reflects the diversity of ecosystems, which vary with landform, vegetation, elevation, latitude, and other factors (figs. 5–8). This is also true for riparian ecosystems, of which there are several different types. The diversity of riparian ecosystems is noteworthy in the context of fire because site-specific conditions affect fire behavior and subsequent fire effects.

Figure 5—This riparian area lies along an intermittent river reach at Bill Williams River National Wildlife Refuge on the eastern edge of the Mojave Desert; Mohave County, Arizona, summer 2012 (photo: Amanda D. Webb).

Figure 6—This riparian woodland in the Sonoran Desert contains mixed native species and tamarisk, Middle Gila River; Gila County, Arizona, fall 2007 (photo: Amanda D. Webb).

Figure 7—The San Pedro Riparian National Conservation Area lies in the transition between the Sonoran and Chihuahuan Deserts, just below 5,000 feet (1,564 meters) elevation, where mesquite (green) and Fremont cottonwood (gold) grow adjacent to grasslands along some parts of the river; Cochise County, Arizona, fall 2003 (photo: Amanda D. Webb).

Figure 8—The Rio Grande winds its way through the Chihuahuan Desert at Big Bend National Park, Brewster County, Texas (photo: Jeff Bennett, National Park Service).

Multiple frameworks exist for categorizing riparian ecosystems (Triepke 2013; USFWS 2009). Many of these systems are restricted to the United States. NatureServe (2015) is unique in that it maps ecological systems in both Mexico and the United States, making it useful for transboundary conservation efforts (table 2).

The impacts of human activity on riparian ecosystems, in the past and present, are an important aspect of the diversity of these ecosystems. Human actions are highly influential in shaping these environments, with both intended and unintended effects. Within the last 150 years, riparian areas have been modified extensively, including direct alterations of rivers (Chaney et al. 1990; Glenn and Nagler 2005; Kreuper 1993; Stromberg et al. 2003; Webb and Leake 2006). These alterations often involve the use of structures such as dams, levees, and channelized streambeds (Dreesen et al. 2002; Glenn and Nagler 2005; Petrakis et al. 2017; Stromberg et al. 2003). Nonnative plant colonization and naturalization are often associated with significant alteration of hydrological regimes, including flood control structures (Glenn and Nagler 2005; Nagler et al. 2011; Poff et al. 2011; Ringold et al. 2008; Webb and Leake 2006).

For example, altered hydrological regimes and sedimentation processes in rivers lead to greater dominance of tamarisk (also known as saltcedar) where cottonwood germination and recruitment opportunities decline (and are potentially eliminated), tamarisk germination and recruitment opportunities increase, or both (Beauchamp and Stromberg 2007; Glenn and Nagler 2005; Merritt and Poff 2010; Mortenson and Weisberg 2010; Ringold et al. 2008; Webb and Leake 2006). Tamarisk has become the dominant or codominant vegetation along many river reaches (Drus 2013; Glenn and Nagler 2005). High densities of tamarisk and other invasive woody plants such as Russian olive increase fuel loads and flammability, making riparian ecosystems more vulnerable to fire (Drus 2013; Ellis 2001; Glenn and Nagler 2005; Kaczynski and Cooper 2015).

Riparian areas have long been used for livestock forage, timber, and firewood (Bahre 1991). Many riparian forests were once harvested for timber, causing increased erosion and heavy sedimentation in streams (Periman and Kelly 2000). Historical overgrazing of livestock in and near riparian zones led to extensive loss of topsoil, soil compaction, channel downcutting, and other forms of erosion that contributed to the lowering of water tables (Bahre 1991; Chaney et al. 1990; Sayre 2011). Grazing management has improved over the years, although the effects of improper grazing have been long-lasting (Bahre 1991; Chaney et al. 1990), and livestock are still grazed in some riparian areas. Livestock grazing may favor tamarisk over cottonwoods and willows because the latter are more palatable (Racher and Britton 2003; Stromberg et al. 2010).

Table 2—NatureServe (2015) riparian ecological systems of the Warm Deserts of North America region. Ecological systems that were the focus of this literature review are shown with gray shading. See table A.1 for common names and botanical authority.

Name and code	**Physical setting**	**Hydrology**	**Common native species**	**States**[1]
Sonoran Fan Palm Oasis CES302.759	<2,953 feet (<900 meters); canyon waterways, fault lines	spring-fed; permanent subsurface water, rainfall with possible flooding	*Washingtonia filifera*, *Salix* spp., *Platanus racemosa*, *Populus fremontii*, *Prosopis glandulosa*, *Fraxinus velutina*	AZ, CA, MXBC, MXSO, NV
North American Warm Desert Riparian Mesquite Bosque CES302.752	<3,609 feet (<1,100 meters); valleys	perennial or intermittent streams; annual water table fluctuations	*Prosopis glandulosa*, *Prosopis velutina*, *Baccharis salicifolia*, *Pluchea sericea*, *Salix exigua*	AZ, CA, MXBC, MXCH, MXSO, NM, NV, TX
North American Warm Desert Riparian Woodland and Shrubland CES302.753	<3,937 feet (<1,200 meters); canyons, desert valleys	medium to large streams; perennial with high flow variability; periodic or annual flooding; snowmelt, rainfall, diffuse groundwater, springs	*Salix* spp., *Juglans major*, *Populus fremontii*, *Fraxinus velutina*, *Platanus racemosa*, *Celtis laevigata* var. *reticulata*	AZ, CA, MXBC, MXCH, MXSO, NM, NV, TX
Western Great Plains Riparian CES303.956	elevation not specified; plains, floodplains, ravines, braided streambeds	small to medium rivers and streams; snowmelt, springs, summer rains, groundwater	*Populus deltoides*, *Salix* spp., *Celtis laevigata*, *Juniperus* spp., *Forestiera pubescens*, *Prosopis glandulosa*, *Panicum* spp.	NM, TX
North American Warm Desert Lower Montane Riparian Woodland and Shrubland CES302.748	3,609–5,906 feet (1,100–1,800 meters); canyons	perennial or intermittent; periodic flooding; bedrock aquifers, snowmelt, rainfall	*Populus fremontii*, *Populus deltoides* ssp. *wislizeni*, *Acer negundo*, *Platanus wrightii*, *Fraxinus velutina*, *Salix exigua*, *Baccharis salicifolia*	AZ, CA, MXBC, MXBS, MXCH, MXSO, NM, NV, TX
North American Warm Desert Cienega CES302.747	<6,562 feet (<2,000 meters); floodplains, valley floors	spring-fed, permanently saturated soils; springs; maybe flooded by overbank flows if a stream is nearby	*Juncus* spp., *Carex* spp., *Sporobolus* spp., *Distichilis* spp., *Salix gooddingii*, *Populus fremontii*, *Fraxinus velutina*	AZ, CA, MXCH, MXSO, NM, NV, TX
North American Warm Desert Wash CES302.755	elevation not specified; washes, arroyos	intermittently flooded; sheet flow, gulley flow from rainfall	*Baccharis sarothroides*, *Chilopsis linearis*, *Acacia greggii*, *Prosopis* spp., *Rhus microphylla*, *Prunus fasciculata*	AZ, CA, MXBC, MXCH, MXSO, NM, NV, TX
North American Arid West Emergent Marsh CES300.729	elevation not specified; depression such as around slow streams and rivers, lakes, ponds	frequent or continual inundation	*Scirpus* spp., *Typha* spp., *Juncus* spp., *Polygonum* spp., *Phalaris* spp.	AZ, CA, MXBC, MXCH, MXSO, NM, NV, TX, UT

Name and code	Physical setting	Hydrology	Common native species	States[1]
Inter-Mountain Basins Wash CES304.781	elevation not specified; streambeds and streambanks	intermittently flooded	*Ericameria nauseosa, Sarcobatus vermiculatus, Artemisia* spp., *Prosopis* spp., *Acacia greggii*	AZ, CA, NV, UT
Rocky Mountain Lower Montane-Foothill Riparian Woodland and Shrubland CES306.821	2,953–9,186 feet (900–2,800 meters); flood zones, streambanks, floodplains, islands, irrigation ditches	perennially wet; annual or episodic flooding	*Acer negundo*, *Populus* spp., *Juniperus scopulorum, Rhus trilobata, Salix* spp., *Prunus virginiana*	AZ, NM, UT
Great Basin Foothill and Lower Montane Riparian Woodland and Shrubland CES304.045	4,000–7,000 feet (1,220–2,135 meters); canyons, draws, floodplains, narrow valleys	perennial or intermittent streams; high flow variability; annual or periodic flooding; snowmelt, springs, bedrock aquifers, rainfall	*Populus* spp., *Salix* spp., *Abies concolor, Alnus incana, Artemisia cana, Cornus sericea, Carex* spp., *Juncus* spp.	CA, NV, UT
Rocky Mountain Subalpine-Montane Riparian Shrubland CES306.832	4,920–11,407 feet (1,500–3,477 meters); narrow to wide floodplains and valley bottoms; low-gradient; isolated hillslope springs	streams; isolated springs in uplands; snowmelt; isolated springs in uplands	*Salix* spp., *Alnus incana, Betula* spp., *Cornus sericea*	AZ, NM, NV, UT
Rocky Mountain Subalpine-Montane Riparian Woodland CES306.833	4,920–10,830 feet (1,500–3,300 meters); narrow canyons and valleys; river and stream floodplains and terraces	high water table, periodic flooding; snowmelt	*Abies* spp., *Picea* spp., *Pseudotsuga menziesii, Populus tremuloides, Juniperus scopulum*	AZ, NM, NV, UT

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[1] AZ = Arizona; CA = California; MXBC = Baja California, Mexico; MXCH = Chihuahua, Mexico; MXSO = Sonora, Mexico; NM = New Mexico; NV = Nevada; TX = Texas; UT = Utah.

In many places, riparian environments have been drained or cleared to develop agricultural lands, resulting in impacts to groundwater aquifers and their connections to river surface waters (Dreesen et al. 2002; Friggens et al. 2013; Jemison 2003; Kreuper 1993). Over the last 100 years, groundwater pumping to support agriculture and the expansion of southwestern cities has accelerated to a rate far exceeding recharge in many aquifers (Guido 2008). As water tables have dropped in response to pumping, groundwater discharge to river surface water has decreased or disappeared in many river reaches. Consequently, perennial flow has shifted to intermittent or ephemeral flow, and typical riparian vegetation is often lost (Ffolliott et al. 2003; Friggens et al. 2013; USGS 2003; Webb and Leake 2006).

The net effect of these human modifications has been alteration of the complex ecological and hydrological processes of these systems, accompanied by subsequent changes in the physical environment and biota (fig. 9) (Jemison 2003; Nagler et al. 2011; Petrakis et al. 2017; Webb and Leake 2006). Analysis of historical photographs suggests that vegetative, hydrological, and morphological characteristics of riparian ecosystems have changed greatly since the mid- to late 1800s (Webb and Leake 2006). Although some changes have been documented, overall we know very little about the characteristics of regional riparian ecosystems before this time period.

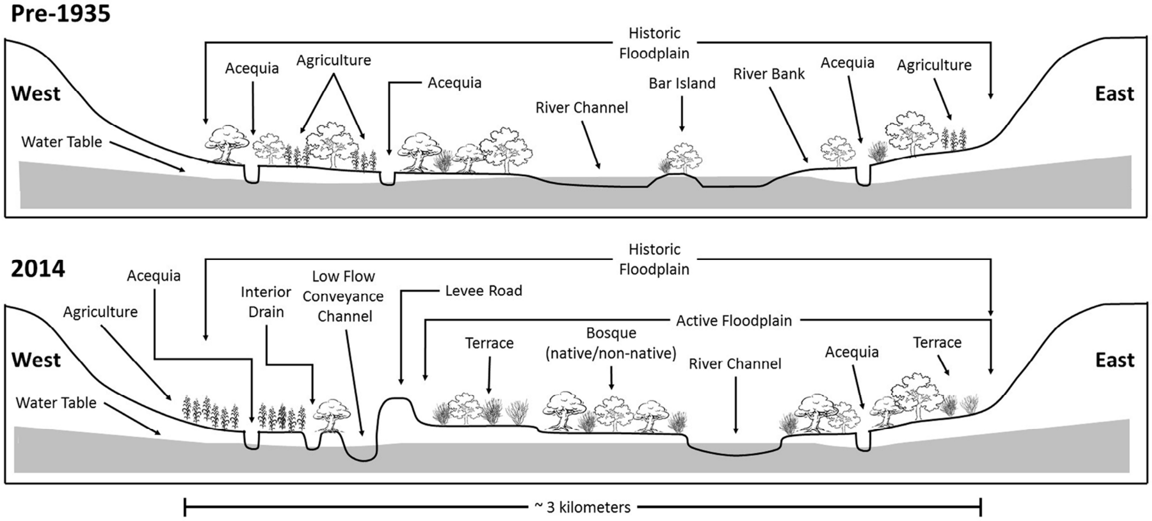

Figure 9—Generalized comparison of the historical and contemporary floodplain along the San Acacia Reach of the Middle Rio Grande in New Mexico before the construction of the San Acacia Diversion Dam in 1935 (top) and in 2014 (bottom). Note the narrowing of the active floodplain and river channel due to an increase in agriculture in the floodplain and streamflow management structures installed during this time period. Figure reproduced from Petrakis et al. (2017) under the Creative Commons Attribution License (CC by 4.0).

The southwestern United States is experiencing rapid environmental change due to documented climate change, particularly the recent trend toward hotter temperatures and increasing drought (Garfin et al. 2013; Seager et al. 2007). Riparian ecosystems and species in the southwestern United States are especially vulnerable because warming trends are projected to increase aridity in a region already water limited. These changes could potentially dewater channels on which riparian flora and fauna depend and increase the risk of other stressors such as fire and the spread of nonnative plants (Capon et al. 2013; Friggens et al. 2013; Smith and Finch 2015, 2017). In addition to climate variation, anthropogenic impacts on riparian hydrology, vegetation, and stream morphology may alter fire properties, increasing the likelihood of high severity wildfires (Dwire and Kauffman 2003) and potentially reducing the capacity of riparian ecosystems to adapt to climate change (Capon et al. 2013).

Section 3 - Methods

Our objectives for this literature review were to:

1. search and select literature,
2. record metadata on selected literature,
3. analyze studies investigating fire effects on vegetation and the physical environment,
4. summarize the state of the knowledge of fire in riparian ecosystems, and
5. synthesize results.

3.1 Literature Search and Selection

In the fall and winter of 2014–2015, we systematically searched two academic databases available through University of Arizona libraries: Web of Science (https://login.webofknowledge.com) and Proquest Dissertations and Theses (https://www.proquest.com/libraries/academic/dissertations-theses/pqdtglobal.html).

To determine whether literature was relevant to the scope of the project, we screened search results by reviewing titles, keywords, and abstracts. Previous consultation with the team had identified the following criteria to filter literature for summary and synthesis: The literature 1) addresses the direct effects of fire on stream corridor (versus upland) including vegetation and the physical environment, 2) focuses on lowland (versus montane) streams, defined as occurring at elevations at or below 5,000 feet (1,564 meters), and 3) includes project areas within the WDNA (see figure 4 in section 1). We included some literature from areas outside the WDNA, such as arid regions of southern California, Australia, or southern Africa, for comparison and to highlight potential knowledge gaps. All forms of literature were to be considered for the review, including peer-reviewed publications, technical reports, investigations, field reports, and qualitative descriptions.

After the literature searches in Web of Science and Proquest, we compiled a list of plant species mentioned in the literature on fire effects. We presented this species list to the members of the Desert LCC riparian fire team, who suggested additional high priority plant species.

We incorporated these prioritized taxa along with some common riparian species into literature searches using two Federal databases, the Fire Effects Information System (FEIS) (https://www.feis-crs.org/feis/) and Treesearch (https://www.fs.usda.gov/treesearch). We searched for fire effects information on several taxa (table 3).

Table 3—List of plant species and genera targeted in systematic literature search (USDA PLANTS Database, https://plants.usda.gov).

Common name	**Scientific name**
Arizona sycamore	*Platanus wrightii* S. Watson
Arizona walnut	*Juglans major* (Torr.) A. Heller
arrowweed	*Pluchea sericea* (Nutt.) Coville
boxelder	*Acer negundo* L.
cattail	*Typha* spp. L.
common reed	*Phragmites australis* (Cav.) Trin. ex Steud.
desert willow	*Chilopsis linearis* (Cav.) Sweet
Fremont cottonwood	*Populus fremontii* S. Wats.
giant reed	*Arundo donax* L.
Goodding's willow	*Salix gooddingii* C.R. Ball
Johnsongrass	*Sorghum halepense* (L.) Pers.
mesquite	*Prosopis* spp. L.
mule-fat	*Baccharis salicifolia* (Ruiz & Pav.) Pers.
netleaf hackberry	*Celtis laevigata* Willd. var. *reticulata* (Torr.) L.D. Benson
Rio Grande cottonwood	*Populus deltoides* W. Bartram ex Marshall ssp. *wislizeni* (S. Watson) Eckenwalder
Russian olive	*Elaeagnus angustifolia* L.
Siberian elm	*Ulmus pumilla* L.
tamarisk	*Tamarix* spp. L.
white mulberry	*Morus alba* L.
willow baccharis	*Baccharis salicina* Torr. & A. Gray

We included pertinent literature generated by Desert LCC-funded projects as of January 2017 although this material was not detected via other literature searches. We supplemented the literature found from the database searches with additional sources suggested by members of the Desert LCC team, through conversations with other experts, and by reviewing works cited in fire effects literature. We included all the fire-related literature found from all databases and by other means in the subsequent analysis.

3.2 Collecting Metadata

We collected metadata from each source, including literature source type, subject, and geographic areas of interest. Where relevant, we also collected metadata about which species and physical factors were discussed or investigated, and whether the study involved wildfire or prescribed fire. We used these metadata to document the fire literature found during the literature search and identify patterns in the literature. In instances where papers were published from previously completed dissertations or theses, and where both the publication and the dissertation or thesis were detected in the literature search, we recorded the metadata from the study only once with preference given to published sources.

To assess the state of the science on fire in lowland riparian ecosystems, we collected additional metadata on which aspects of fire were investigated in experiments and observational studies. For each of these literature sources, we recorded one or more of the following fire-specific subjects that were investigated: fire behavior, fire frequency, fire severity, fire size, occurrence, plant mortality, and postfire regeneration. We also recorded which fire studies included hydrological factors in the analysis.

We coded literature source types with one or more of the following options: observational study, experiment, model, review, technical report, nontechnical report, field report, remote sensing, case study, and meta-analysis. We coded literature type according to where the source of information about fire was found, which did not always reflect the source type or types of the entire literature source. For example, if a source, such as a dissertation, included a literature review concerning fire effects on plants, but the primary experimental work from the dissertation was not focused on fire, then we coded the literature source type as “Review” instead of “Experiment.”

We coded literature subject types with one or more of the following options:

- fire and native plants,
- fire and nonnative plants,
- fire and both native and nonnative plants,
- vegetation structure only,
- fire and the abiotic environment,
- postfire rehabilitation.

We selected only one of the following options for coding fire type: wildfire only, prescribed fire only, both wildfire and prescribed fire, or not specified.

3.3 Summarizing Results

We summarized literature on fire effects on physical processes and vegetation, fire as a restoration tool in riparian ecosystems, and postfire rehabilitation. We read and reviewed papers identified through the literature search, and summarized relevant information in a thematic narrative. The narrative was supplemented with two tables of literature source metadata (Appendices B and C).

3.4 Synthesis

After reviewing and analyzing the literature, we synthesized the information about fire in riparian ecosystems. We incorporated recommendations from Smith (2015) for developing fire syntheses that are useful to managers. Our synthesis focused on addressing the following questions:

1. What does the literature suggest about the overall impact of riparian fire on ecological communities?
2. What does the literature suggest about patterns, drivers, and trends in fire severity and frequency?
3. How do fire effects vary among species, and what might these results mean for research and management?
4. What knowledge gaps exist in the current literature, and what further research is needed to improve understanding of riparian fire effects and management?

Section 4 - Results: Summaries of Literature Reviewed

In this section, we summarize study results and other information from 67 resources that describe direct effects of fire on the physical environment in the stream and floodplain corridor, and associated riparian vegetation along lowland streams. We found few published sources regarding lowland riparian fire before the 1990s, with an increasing number of publications in later years (fig. 10).

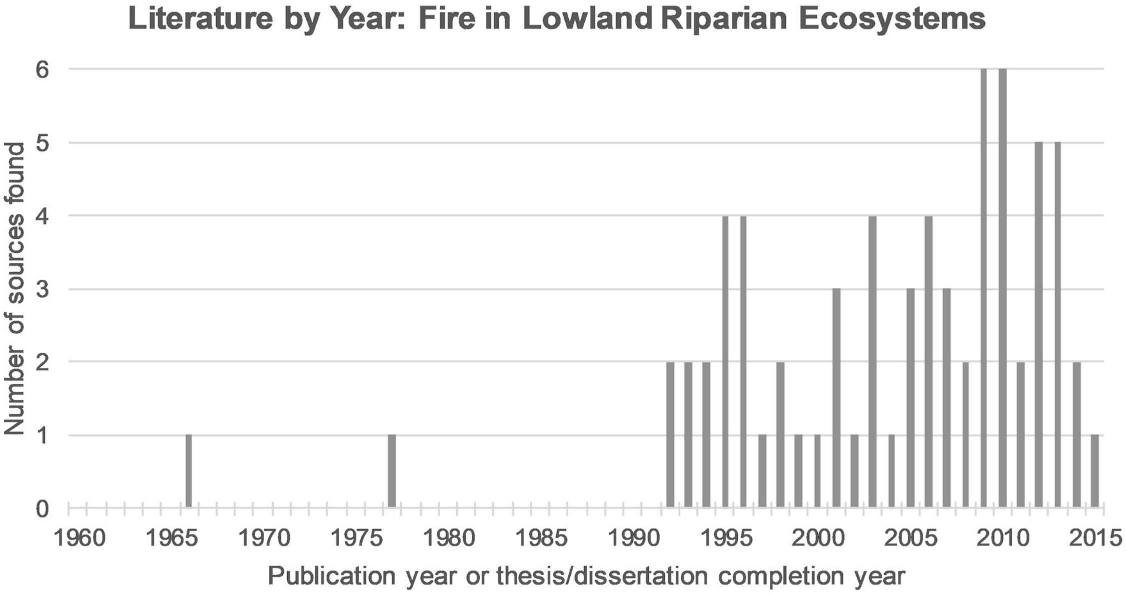

Figure 10—Number of sources found from 1960 through 2015 (compiled November 2016). The number of publications, dissertations, and theses on fire in lowland riparian ecosystems has increased dramatically since the early 1990s.

Literature sources and metadata are provided in Appendices B and C. Appendix B lists all fire-related literature sources by geographic area of interest and includes information about source subject and fire types discussed. Appendix C lists all fire-related observational studies and experiments included in the review and states source subject, fire type, and which of the following aspects of fire were investigated: fire behavior, fire frequency, fire severity, fire size, occurrence, plant mortality, and regeneration. Appendix C also indicates which studies include hydrological factors in the analysis.

4.1 Fire Behavior, Regimes, and Effects on Native Plants and Plant Communities

The literature summaries begin with fire effects on native riparian plants and plant communities. This section is organized by plant species or riparian plant communities wherein the species in the subsection title can dominate or codominate vegetation. All native species and communities with fire literature detected through our literature search are included.

4.1.1 Cottonwood-Willow Riparian Woodland Communities

Fremont cottonwood, Rio Grande cottonwood, and Goodding's willow are prominent species in riparian ecosystems of the Warm Deserts of North America (WDNA). Fremont cottonwood occurs in all U.S. States within the WDNA (NRCS 2018). Fremont cottonwoods also occur in much of northwestern Mexico, including Chihuahua, Sonora, Durango, and Baja California (NRCS 2018; SEINet n.d.). Rio Grande cottonwood occurs in the Rio Grande drainage in western Texas, New Mexico, Arizona, eastern Utah, and western Colorado (Eckenwalder 1977) as well as in Mexico. Goodding's willow is found in all U.S. States within the WDNA (Reed 1993) and in northern Mexico (Kearney and Peebles 1979).

Where present, mature cottonwoods and willows generally constitute the upper canopy layer in riparian vegetation structure (fig. 11). Below this canopy, bare ground, additional woody species, and herbaceous vegetation may be present in the understory to varying degrees,

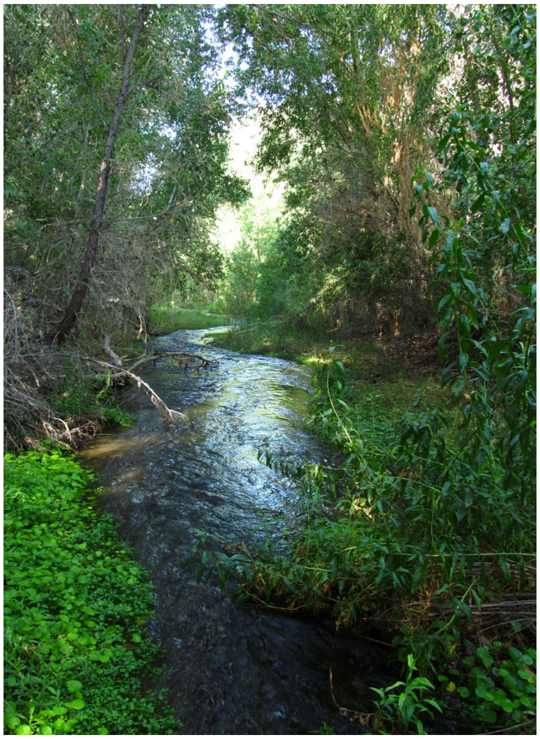

Figure 11—Fremont cottonwoods and Goodding's willows form the upper canopy along the river at Bill Williams River National Wildlife Refuge; Mohave County, Arizona, summer 2012 (photo: Amanda D. Webb).

depending on site conditions and hydrological regimes. These understory conditions also affect fire regimes (LANDFIRE 2008c).

Ordination analysis on the Colorado and Bill Williams Rivers in western Arizona showed fire to be one of the primary factors determining cottonwood-willow vegetation communities, along with moisture, salinity, and community maturity (Busch and Smith 1995). In another study at higher elevations in the Transverse Ranges in California, fire influenced plant community composition, as did elevation, stream power, and valley width (Bendix 1994).

Historical fire regimes in cottonwood-willow riparian woodlands are not well known (Friggens et al. 2013; Smith and Finch 2015). It is likely that in many cases, riparian areas slowed or stopped fires that started in the uplands because the riparian zone was cooler and live fuel moistures were higher, even during dry times (Brook and Minnich 2006; Dwire and Kofffman 2003). Conversely, on the San Pedro River, increasing fire frequency may be shifting vegetation away from native woody species and toward more herbaceous plant communities, which could be more similar to historical or prehistoric conditions (Stromberg et al. 2009).

However, fire regimes today are highly variable in riparian ecosystems throughout the WDNA. LANDFIRE's models for Biophysical Settings (BpS) indicate that fire regimes in cottonwood-willow communities fall into two categories: Fire Regime Group I (≤35-year return interval, low and mixed severity) and Fire Regime Group V (>200-year fire return interval, any severity) (LANDFIRE 2008a) (table 4). Another LANDFIRE source, the Rapid Assessment Condition Model, puts the fire regime in cottonwood-willow habitats in New Mexico and parts of Arizona in Fire Regime Group III (35- to 200-year frequency, low and mixed severity). Longer fire return intervals and lower fire severity may reflect the minimal buildup of fine fuels and litter where flooding scours the understory of the riparian deciduous woodland in the southwestern United States (LANDFIRE 2008c). These resources offer divergent classification of fire regime groups for cottonwood-willow communities. A possible reason for this may be that fire regimes are highly variable across different types of lowland riparian ecosystems. Additionally, the inherently dynamic nature of these ecosystems may make fire regimes hard to characterize.

Table 4—Selected LANDFIRE 2008 (LF 1.1.0) Biophysical Settings (BpS) models and descriptions for desert riparian ecosystems. Source: https://www.landfire.gov/national_veg_models_op2.php.

National BpS model, name, **FRG**[1]	**BpS group model, name** **(model source)**	**Geographic range**[2] **and biophysical site description**
1311551 North American Warm Desert Riparian Systems **FRG** I (≤35-year return interval, low and mixed severity)	252 Cottonwood-Goodings Willow-Mesquite-1 (1311551)	Perennial and somewhat intermittent warm desert (Mojave and Sonoran Deserts) drainages in southern CA, NV, AZ, and southwestern UT. Riparian systems occur primarily along perennial streams and rivers along the Colorado, Salt, Virgin, Muddy, and Mojave River corridors adjacent to low elevation shrublands. Elevation is typically below 4,000 feet (1,219 meters). When mesquite bosque is the dominant type outside of perennial waterways, it is also found at elevations lower than 3,609 ft (1,100 m) along intermittent streams or in valley bottoms along playa edges with a perched water table. The vegetation is a mix of riparian woodlands, shrublands, and grasslands. Vegetation is very patchy in rivers with active flood regimes.
1311552 North American Warm Desert Riparian Systems - Stringers **FRG** V (>200-year fire return interval, any severity)	253 Goodings Willow-Velvet Ash-Acacia-5 (1311552)	Intermittent to dry warm desert (Mojave and Sonoran Deserts) drainages with mostly subsurface flow in southern CA, NV, AZ, and southwestern UT. Narrow riparian systems occur primarily along low elevation shrublands (creosote bush, blackbrush, and paloverde matrix vegetation) and in canyons and washes or as spring brooks. Elevation is typically below 4,000 feet (1,219 meters). Examples of intermittent streams can be found in the Amargosa Gorge, Whitewater River, Andreas Canyon, Paiute Creek, and Palm Canyon. Oasis woodlands occur in isolated stands such as the Palm Canyon, Thousand Palms, and Twentynine-palms oases.
1411551 North American Warm Desert Riparian Systems **FRG** I (≤35-year return interval, low and mixed severity)	252 Cottonwood-Goodings Willow-Mesquite-1 (1311551)	Perennial and somewhat intermittent warm desert (Mojave and Sonoran Deserts) drainages in southern CA, NV, AZ, and southwestern UT. Riparian systems occur primarily along perennial streams and rivers along the Colorado, Salt, Virgin, Muddy, and Mojave River corridors adjacent to low elevation shrublands. Elevation is typically below 4,000 ft (1,219 m). When mesquite bosque is the dominant type outside of perennial waterways, it is also found at elevations lower than 3,609 feet (1,100 meters) along intermittent streams or in valley bottoms along playa edges with a perched water table. The vegetation is a mix of riparian woodlands, shrublands, and grasslands. Vegetation is very patchy in rivers with active flood regimes.
1411552 North American Warm Desert Riparian Systems - Stringers **FRG** V (>200-year fire return interval, any severity)	253 Goodings Willow-Velvet Ash-Acacia-5 (1311552)	Intermittent to dry warm desert (Mojave and Sonoran Deserts) drainages with mostly subsurface flow in southern CA, NV, AZ, and southwestern UT. Narrow riparian systems occur primarily along low elevation shrublands (creosote bush, blackbrush, and paloverde matrix vegetation) and in canyons, washes, or as spring brooks. Elevation is typically below 4,000 feet (1,219 meters). Examples of intermittent streams can be found in the Amargosa Gorge, Whitewater River, Andreas Canyon, Paiute Creek, and Palm Canyon. Oasis woodlands occur in isolated stands such as the Palm Canyon, Thousand Palms, and Twentynine-palms oases.

National BpS model, name, FRG[1]	BpS group model, name (model source)	Geographic range[2] and biophysical site description
1511551 North American Warm Desert Riparian Systems **FRG** I (≤35-year return interval, low and mixed severity)	252 Cottonwood-Goodings Willow-Mesquite-1 (1311551)	Perennial and intermittent desert drainages in central and southwestern AZ. Riparian systems occur primarily along perennial streams and rivers along the Lower Colorado, Lower Salt, Lower Verde, Lower Gila, Big Sandy, Bill Williams, Santa Maria, Hassayampa, and Lower Santa Cruz River corridors adjacent to Sonoran Desert Scrub. The vegetation is a diverse mosaic of riparian forests, shrublands, streamside marshes, and barren alluvial surfaces.
1511552 North American Warm Desert Riparian Systems - Stringers **FRG** V (>200-year fire return interval, any severity)	253 Goodings Willow-Velvet Ash-Acacia-5 (1311552)	Intermittent to dry warm desert (Mojave and Sonoran Deserts) drainages with mostly subsurface flow in southern CA, NV, AZ, and southwestern UT. Narrow riparian systems occur primarily along low elevation shrublands (creosote, blackbrush, and paloverde matrix vegetation) and in canyons and washes or as spring brooks. Elevation is typically below 4,000 feet. Examples of intermittent streams can be found in the Amargosa Gorge, Whitewater River, Andreas Canyon, Paiute Creek, and Palm Canyon. Oasis woodlands occur in isolated stands such as the Palm Canyon, Thousand Palms, and Twentynine-palms oases.
2511550 North American Warm Desert Riparian Systems **FRG** V (>200-year fire return interval, any severity)	251 Alkali Sacaton-Vine Mesquite-5 (2511550)	Tularosa Basin, Jornada Basin, south of Interstate 10, northwest of Lordsburg, southeastern NM, and extending into southeastern AZ, in localized settings. Generally associated with flats, swales, and bottomlands. Desert grassland with extensive clayey and alkaline bottomland plains and intermittently flooded swales.
2611550 North American Warm Desert Riparian Systems not specified	254 Desert Willow-Arizona Sycamore-5 (2611550)	Southern AZ and NM, Trans-Pecos TX, and adjacent Mexico. Consists of mid- to low elevation (3,609–5,906 feet; 1,100–1,800 meters) riparian corridors along perennial and seasonally intermittent streams. The vegetation is a mix of riparian woodlands and shrublands. Vegetation is dependent on annual or periodic flooding and associated sediment scour or annual rise in the water table for growth and reproduction.
2611620 Western Great Plains Floodplain Systems **FRG** I (≤35-year return interval, low and mixed severity)	261 Eastern Cottonwood-Sandbar Willow-Green Ash-3 (3111620)	Floodplains of medium and large rivers of the Western Great Plains. Dominant communities within this system range from floodplain forests to wet meadows to gravel/sand flats; however, they are linked by underlying soils and the flooding regime.

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[1] Fire Regime Group (FRG) characterizes the presumed historical fire regimes within landscapes based on interactions between vegetation dynamics, fire spread, fire effects, and spatial context.

[2] AZ = Arizona; CA = California; NV = Nevada; NM = New Mexico; TX = Texas; UT = Utah.

Nonnative plant species can increase fire frequency and intensity in riparian ecosystems when they make up a significant portion of the cottonwood-willow woodland community, as has been documented with tamarisk (also known as saltcedar) (Busch 1995; Busch and Smith 1993; Drus 2013; Jemison 2003; Parker et al. 2005). The return interval in the cottonwood-willow woodland depends on characteristics of the understory, with intervals as low as 15 years in areas where tamarisk is present (Drus 2013; LANDFIRE 2008c; Racher and Britton 2003). Altered streamflow regimes are prominent among the other anthropogenic influences, such as fire suppression and stream channel modifications, that have increased the spread of invasive species (Friggens et al. 2013; Merritt and Poff 2010; Ringold et al. 2008; Smith et al. 2009; Stromberg and Rychener 2010). Encroachment into these communities by nonnative plant species such as tamarisk, Russian olive, and giant reed alters fuel loads and structure.

Relatively little information on fire in cottonwood-willow woodlands is available except as influenced by the presence of tamarisk (outlined in the nonnative vegetation section of this report). When heat is applied, tamarisk foliage loses water and ignites more quickly than Fremont cottonwood, Goodding's willow, and coyote willows; however, tamarisk is not necessarily consumed more quickly than native species (Drus 2013). Foliar architecture is a major factor influencing the differences in flammability between these species (Drus 2013).

Post-fire mortality rates of native trees can increase dramatically in the presence of tamarisk. An assessment of 30 sites (Drus 2013) revealed that higher mortality and consumption of Fremont cottonwoods and willows were associated with increasing pre-fire tamarisk cover; higher fire intensity where tamarisk was present resulted in more tissue damage, and when pre-fire tamarisk cover exceeded 50 percent, 100 percent of native trees had total fine material consumption (fig. 12). In this study, mortality was defined as trees that did not resprout after fire, or resprouted and later died.

Most studies of fire effects on Rio Grande cottonwood in the WDNA have been conducted on the Middle Rio Grande in New Mexico. Rio Grande cottonwood mortality is influenced by fire severity, with more trees surviving low severity fires (as determined by surface fuel consumption) than moderate severity fire (Stuever 1997). No trees survive high severity fires, and few or no trees survive moderate severity fires (Ellis 2001; Stuever 1997). High severity fires may be less pervasive in areas with natural flooding regimes, where floods scour litter and woody debris from the riparian corridor forest floor and prevent accumulation (Ellis 2001). In these studies, mortality was defined by complete top-kill immediately after fire and subsequent resprout survival (Ellis 2001), and complete top-kill (Stuever 1997), respectively.

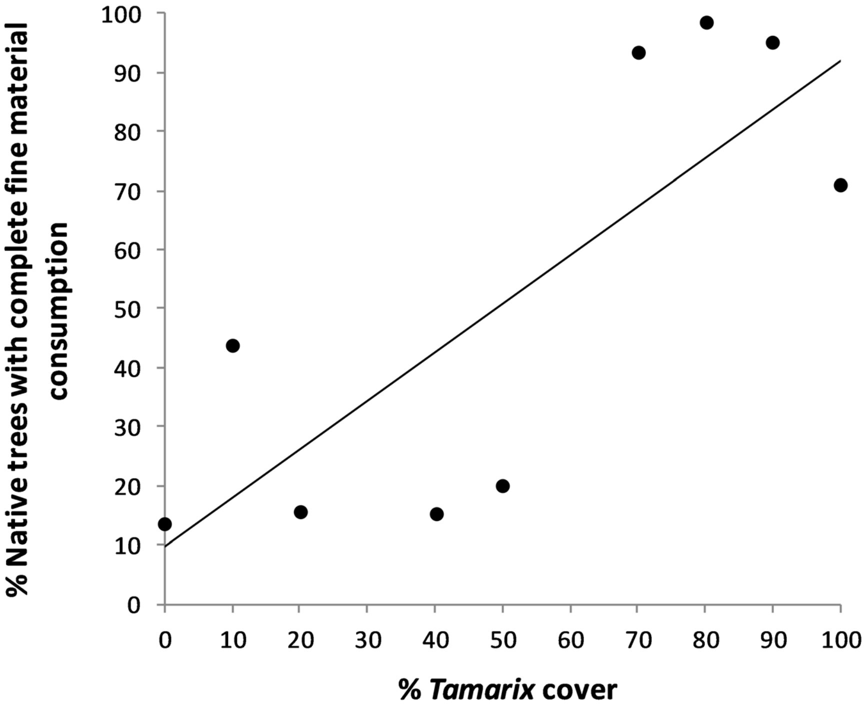


Figure 12—Relationship between pre-fire tamarisk cover and percentage of native trees with total consumption of fine material (1 hour fuels). Spearman's ρ (correlation) = 0.77; N = 9. Reproduced from Drus (2013) with author's permission.

A model by Smith and Finch (2015) projected cottonwood population density at two sites on the Middle Rio Grande over 100 years based on historical hydrological, wildfire, and drought data. Rio Grande cottonwood population sensitivities differed between study sites with respect to changes in wildfire, flood, and drought. The two sites also differed in flow and channel morphology such that between 1975 and 2012, the north section flooded in 2 years (5 percent of the years) and the south flooded in 14 years (38 percent). These results suggest that Rio Grande cottonwood may decline under current conditions of flow-regulated streams, even without factoring in possible climate change scenarios (Smith and Finch 2015). Others have come to similar conclusions about Rio Grande cottonwood populations on the Middle Rio Grande (Howe and Knopf 1991). Growth of nonnative woody species in this system, including tamarisk and Russian olive, is expected to increase with loss of the upper canopy cover that cottonwoods provide (Howe and Kopf 1991; Smith and Finch 2015). The rate of cottonwood density decline increased when climate change was included in the models (Smith and Finch 2015).

On the undammed and perennial upper San Pedro River in Arizona, Fremont cottonwood mortality (as calculated from two population growth equations using stem densities measured pre- and post-fire) was lower than in some previous studies examining cottonwood mortality in flow-regulated systems (Stromberg and Rychener 2010). Factors affecting cottonwood mortality vary (table 5), and cottonwood mortality from fire may be lower in the presence of perennial water and shallow depth to groundwater (Stromberg and Rychener 2010).

Table 5—Documented factors affecting fire-related cottonwood mortality. Researchers defined mortality in the following ways: Trees did not resprout after fire or resprouted and later died (Drus 2013); complete top-kill immediately after fire and resprout survival (Ellis 2001); resprout survival of trees labeled as top-killed as determined by no green leaves in the canopy (Smith et al. 2009); calculated from two population growth equations using stem densities measured pre- and post-fire (Stromberg and Rychener 2010); complete top-kill (Stuever 1997).

Factor studied	**Species**[1]	**Source**
Presence of surface water	FC	Stromberg and Rychener 2010
Flooding characteristics (experimental or natural)	RGC	Ellis 2001
Fire severity class (litter consumption)	RGC	Ellis 2001; Stuever 1997
Fire severity class (presence of canopy scorch)	FC	Stromberg and Rychener 2010
Fire severity (% woody debris consumption)	RGC	Ellis 2001
Fire severity (basal charring more or less than halfway around the base of the main stem)	FC	Stromberg and Rychener 2010
Fire severity (basal charring completely encircling tree)	RGC	Stuever 1997
Tamarisk cover (%)	*Populus* spp.	Drus 2013
Tree diameter (diameter at breast height)	RGC, FC, RGC, respectively	Ellis 2001; Stromberg and Rychener 2010; Stuever 1997
Time since fire	RGC	Ellis 2001; Smith 2009; Stromberg and Rychener 2010

[1] FC = Fremont cottonwood; RGC = Rio Grande cottonwood.

In the spring complex of Ash Meadows, Nevada, likelihood of high burn severity increased in areas with dense, homogeneous vegetation (Sunderman and Weisberg 2012). High burn severity was less likely in low-lying areas prone to seasonal inundation and near surface water (Sunderman and Weisberg 2012).

Many native woody species resprout after fire, and species may resprout at different rates within a given set of environmental conditions (fig. 13). Even in areas of high fire severity, top-killed Rio

Grande cottonwoods can resprout (Ellis 2001; Smith and Finch 2017; Smith et al. 2009). Rio Grande cottonwoods have been observed with basal resprouts, epicormic resprouts, and root suckers post-fire (Smith and Finch 2017). Goodding's willows resprout prolifically post-fire (Reed 1993; Smith and Busch 1992; Stromberg and Rychener 2010), even when totally consumed aboveground (Ellis 2001). However, the effect of fire intensity and severity on the response of Rio Grande cottonwood, Fremont cottonwood, and Goodding's willow is unclear, with existing studies showing contrasting effects of fire severity on resprouting (Ellis 2001; Stromberg and Rychener 2010; Stuever 1997).

Figure 13—Arrowweed and coyote willow resprouted in spring after the Three Slashes Fire, which burned in August and September 2011, Cibola National Wildlife Refuge, California and Arizona (photo: Matthew Grabau, used with permission).

Many native woody riparian species can resprout following fire (table 6). However, existing information suggests that resprout mortality of Rio Grande cottonwood tends to increase with time since fire (Ellis 2001; Smith et al. 2009). Regardless of time since fire, resprout mortality (defined in this cited study as mortality of resprouts on trees considered top-killed as determined by the absence of green leaves in the canopy) increases during hot, dry periods and with lowering water tables, as was observed with cottonwoods along the Middle Rio Grande (Smith et al. 2009). These factors contribute to conditions wherein post-fire resprouting tamarisk becomes more dense and can replace cottonwoods (Smith et al. 2009). Additional research is needed to better understand environmental factors and management actions that influence resprout survivability of cottonwoods and willows over multiple years post-fire (Ellis 2001; Smith and Finch 2017; Smith et al. 2009; Stuever 1997).

Table 6—Native riparian-associated species that have been observed resprouting post-fire.

Scientific name	Common name	Source
Alnus rhombifolia	white alder	Bendix and Cowell 2013
Amorpha fruticosa	desert indigobush	Ellis 2001
Baccharis salicifolia	mule-fat	Coffman et al. 2010; Ellis 2001
Chilopsis linearis	desert willow	Bock and Bock 2014
Forestiera neomexicana	New Mexico olive	Ellis 2001
Fraxinus velutina	velvet ash	Bock and Bock 2014; Sunderman 2009
Juglans major	Arizona walnut	Bock and Bock 2014
Platanus wrightii	Arizona sycamore	Bock and Bock 2014
Pluchea sericea	arrowweed	Busch and Smith 1993
Populus balsamifera subsp. *trichocarpa*	black cottonwood	Coffman et al. 2010
Populus deltoides subsp. *wislizeni*	Rio Grande cottonwood	Ellis 2001; Smith and Finch 2015, 2017; Smith et al. 2009; Stuever 1997
Populus fremontii	Fremont cottonwood	Bock and Bock 2014; Stromberg and Rychener 2010
Prosopis glandulosa	honey mesquite	Sunderman 2009
Prosopis pubescens	screwbean mesquite	Smith and Finch 2015; Sunderman 2009
Prosopis velutina	velvet mesquite	Stromberg and Rychener 2010
Quercus agrifolia var. *frutescens*	live oak	Bendix and Cowell 1993
Salix exigua	coyote willow	Coffman et al. 2010
Salix gooddingii	Goodding's willow	Ellis 2001; Smith and Finch 2015; Smith et al. 2009; Stromberg and Rychener 2010; Stuever 1997
Salix laevigata	red willow	Coffman et al. 2010
Salix lasiolepis	arroyo willow	Coffman et al. 2010
Typha domingensis	southern cattail	Glenn et al. 2013; Mexicano et al. 2013

Post-fire conditions and physiological mechanisms affecting water potential and hydraulic efficiency may help explain differences in recovery success among species (Smith and Busch 1992). For example, tamarisk and arrowweed have higher water use efficiency (that is, greater carbon fixation potential per unit water utilized) and a greater tolerance of saline conditions after fire than Goodding's willow and Fremont cottonwood. In addition, arrowweed is a clonal native shrub that can reproduce via underground rhizomes (Vogl and McHargue 1966). These mechanisms may contribute to an increase in the dominance of tamarisk and arrowweed after fire in a cottonwood-willow woodland (Busch and Smith 1993).

Flooding is an important form of disturbance in riparian ecosystems and is necessary for providing habitat for cottonwood and willow germination and recruitment (Karrenberg et al. 2002; Mahoney and Rood 1998; Smith and Finch 2015; Stromberg 1998). Cottonwoods and willows typically occur where flooding occurs, in active channels or in the floodplain. On flow-regulated rivers where high magnitude flooding rarely occurs, fires followed by low magnitude flooding can potentially stimulate cottonwood germination by clearing soils of litter and vegetation, and opening the woodland canopy (Smith and Finch 2015, 2017). For example, along the Middle Rio Grande, cottonwood germination has been observed after fire in areas where post-fire flooding occurred (Ellis 2001; Smith and Finch 2015, 2017). Cottonwood seedlings were found in areas that flooded within 2 years post-fire (Ellis 2001). In another study, where an area that burned in a spring 2008 wildfire flooded partially the following June, cottonwood seeds were observed in moist soil 2 months after the fire (Smith and Finch 2017), and saplings were observed there during data collection in 2013 (Smith and Finch 2015). Saplings of Siberian elm and white mulberry, both nonnative species, were also observed in this area (Smith and Finch 2017).

Fire can increase opportunities for native tree regeneration in some riparian environments, but its influence is spatially variable and not well understood across broad geographic regions (Bendix and Cowell 2013). Where regeneration occurs, resprouts and seedlings are vulnerable to subsequent and recurring fires (Brooks and Minnich 2006; Nagler et al. 2005). Fire can initiate shifts in population size structure of riparian trees (Stromberg and Rychener 2010). Stem diameter may influence survival of Fremont and Rio Grande cottonwood during and following fire, with smaller trees having higher probability of mortality than larger trees (Stromberg and Rychener 2010; Stuever 1997); this may also be the case with Goodding's willow (Stromberg and Rychener 2010). Although larger Rio Grande cottonwoods are less likely to be top-killed, of those that are, smaller individuals are more likely to resprout post-fire (Ellis 2001). In Fremont cottonwoods, the influence of size on resprout rate can be minimal (Stromberg and Rychener 2010). However, physiological factors affecting resprout rate are not well understood (Stuever 1997). Severe fires can induce uniform changes in structure and composition across the entire riparian zone by initiating widespread mortality (Bendix and Cowell 2010b).

In an analysis of 12 years of fire data from the Colorado River and Bill Williams River floodplains, Fremont cottonwood was almost absent from burned plots, including those that had been dominated by cottonwoods and willows before the fire, although willows persisted (Busch 1995). Arrowweed abundance and tamarisk cover increased post-fire, whether dominated by cottonwoods and willows, tamarisk, or mesquite before the fire (Busch 1995). Pre-fire communities dominated by cottonwoods and willow or mesquites shifted toward greater abundance of arrowweed, and burned areas were associated with the presence of tamarisk (Busch 1995).

In southern California, fire caused extensive mortality of white alder, the dominant pre-fire riparian tree, and led to dominance by Fremont cottonwood and coast live oak; both species resprouted post-fire more than white alder (Bendix and Cowell 2013). An ordination study in the Transverse Ranges in southern California showed a vegetation community characterized by mule-fat and red willow to be associated with riparian areas that had recently burned; a community characterized by Fremont cottonwood and white alder was associated with higher elevations and longer time since burning (Bendix 1994).

4.1.2 Mesquite Bosques

Multiple mesquite species occur within the WDNA. The literature reviewed for this report focused primarily on three species: screwbean mesquite, honey mesquite, and velvet mesquite. Although mesquites are present in multiple plant community types within the WDNA, they can form a unique type known as a mesquite bosque along lowland rivers (fig. 14). These woodlands are dominated by mesquite and can include a mix of other woody trees and shrubs, including cottonwood, saltbush, and arrowweed (LANDFIRE 2008b). Tamarisk has heavily invaded many mesquite bosques throughout the region (LANDFIRE 2008b).

Figure 14—A mesquite bosque along the river in the San Pedro Riparian National Conservation Area fills in low-lying areas adjacent to golden Fremont cottonwoods that line the river channel; Cochise County, Arizona, fall 2003 (photo: Amanda D. Webb, Bureau of Land Management).

Screwbean mesquite is associated primarily with riparian woodlands (Meyer 2005). It occurs in lowland floodplains throughout the southwestern United States (Kearney and Peebles 1979) and in the Mexican States of Baja California, Sonora, and Chihuahua (Little 1979). Honey mesquite occurs throughout much of the southwestern United States and into the plains to the east, as well as in Mexico (Martin and Hutchins 1980). In Mexico, honey mesquite occurs in the States of Chihuahua, Sonora, and Baja California, and as far south as Sinaloa (SEINet n.d.). Though occurring primarily in riparian areas, honey mesquite also extends into upland habitats throughout much of its range (Steinberg 2001). Velvet mesquite occurs in Arizona to western Texas (Martin and Hutchins 1980), and south to Michoacán, including Baja California (Little 1979). Velvet mesquite is associated with both riparian and upland habitats (Uchytil 1990).

Published information on fire behavior in mesquite bosques is sparse. A study of fire effects on vegetation in the desert springs ecosystem of Ash Meadows National Wildlife Refuge in Nevada found that screwbean mesquite was more likely to die (top-killed and not resprouting) from fire than honey mesquite (fig. 15) (Sunderman 2009). Both species were observed to resprout post-fire, but regeneration from seed in burned areas was more common with screwbean mesquite (Sunderman 2009). Velvet ash appeared to have low seed recruitment in burned areas (Sunderman 2009). Fire damage to screwbean mesquite and honey mesquite was less likely in sandier soils, and damage to honey mesquite and velvet ash was less likely where soil conductivity was lower and topographic curvature was greater (Sunderman 2009). Research from other parts of the United

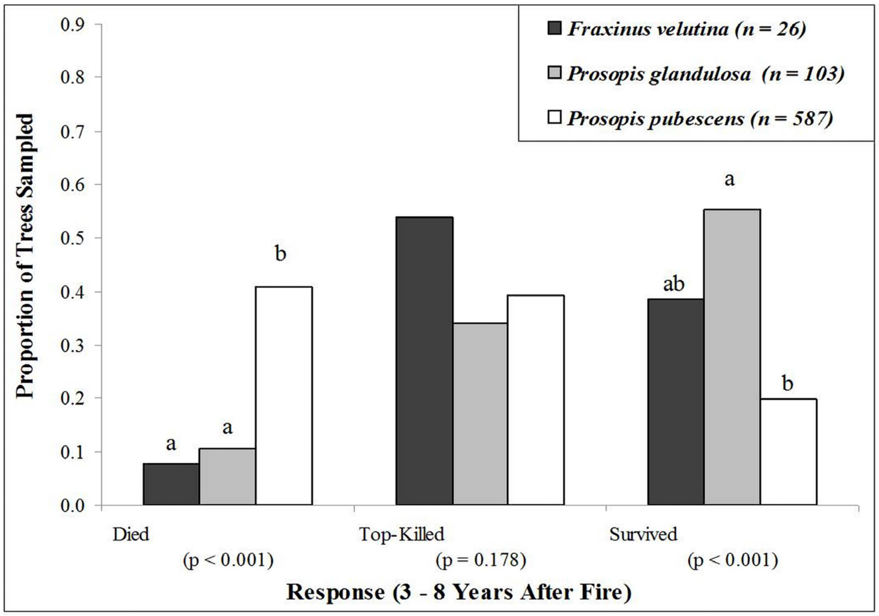


Figure 15—Response of velvet ash (black), honey mesquite (medium gray), and screwbean mesquite (white) to fire. Columns show the proportion of each species that died (was top-killed and did not resprout), was top-killed and resprouted, and survived with minimal damage above-ground. Lowercase letters indicate significant differences among species within a group of columns for a given response. Marascuilo multiple comparisons (α = 0.05). Reproduced from Sunderman (2009) with author’s permission.

States suggests that screwbean mesquite is likely to decline in response to fire (Meyer 2005).

Historical fire regimes in mesquite bosques are not well documented, but it is likely that fires were less severe than the fires common today, as decades of diminished flood disturbance and fire suppression have contributed to high fuel loads (Friggens et al. 2013). The Rapid Assessment Reference Condition Model for the mesquite bosque vegetation type in Arizona and New Mexico indicates Fire Regime Group III (35- to 200-year frequency, low and mixed severity) (LANDFIRE 2008b) with a 45-year mean fire interval and high variation as affected by flood regimes, fire regimes of adjacent vegetation types, herbivory, drought, and anthropogenic ignitions (LANDFIRE 2008b). When surrounded by desert shrub vegetation, mesquite bosques can have fire intervals closer to 100 to 500 years (LANDFIRE 2008b). Tamarisk invasion into mesquite bosques is common (LANDFIRE 2008b), and can reduce return intervals (increasing fire frequency) where tamarisk makes up a significant proportion of fuels (Busch 1995).

In an assessment of pre-fire and post-fire vegetation communities and burn patterns using data from a 12-year period in the Colorado River floodplain, less area was burned in communities dominated by either screwbean mesquite or honey mesquite than was expected given its relative proportion of the total study area, and extent of fires was associated strongly with areas dominated by tamarisk (Busch 1995). Following fire, tamarisk came to dominate areas formerly dominated by mesquite, and abundance of the native shrub arrowweed increased. Fire reduced cover of honey mesquite, although it was more frequently detected in areas with pre-fire mesquite dominance than in other tamarisk- or cottonwood- and willow-dominated communities. Frequency and cover of screwbean mesquite were low throughout all community types (Busch 1995).

Much of the information on fire effects on velvet mesquite has been derived from studies in rangeland ecosystems (Uchytil 1990), and relatively little information is available about effects of fire on velvet mesquite, specifically in lowland riparian ecosystems. Stromberg and Rychener (2010) studied fire effects on woody riparian species along the San Pedro River in southeastern Arizona. Data collected before and 2 months after a fire indicated that velvet mesquite basal area had declined 95 percent and stem density had declined 83 percent (Stromberg and Rychener 2010). However, comparison of results from sites that had burned 2 to 3 years previously and similar unburned sites showed decreased live basal area in burned compared to unburned areas, but higher stem densities, suggesting substantial resprouting by velvet mesquite in the time since the fire (Stromberg and Rychener 2010).

4.1.3 Southern Cattail

Southern cattail occurs in wetland and riparian ecosystems throughout much of the southern United States, including all States in the WDNA (USDA 2015), and throughout Mexico (Kearney and Peebles 1960; SEINet n.d.). Published information on southern cattail and fire (fig. 16) comes primarily from studies conducted at Cienega de Santa Clara in Sonora, where southern cattail is the dominant vegetation. Compared to nonfire years, fire in the marsh led to an increase in the Normalized Difference Vegetation Index (Mexicano et al. 2013), and in evapotranspiration (Glenn et al. 2013). Fires burned off accumulated thatch, and cattail responded by putting on new growth (Glenn et al. 2013; Mexicano et al. 2013).

Figure 16—Cattails emerge from the aquatic environment at Bill Williams River National Wildlife Refuge; Mohave County, Arizona, summer 2012. Vegetation transitions to mesquite and tamarisk growing in adjacent areas upslope (photo: Amanda D. Webb).

4.1.4 California Fan Palm Communities

Fan palm groves occur in the southern California Mojave Desert and south into San Pedro Martir, and Sierra Juarez in Mexico (Vogl 1967). Many populations in the United States occur along the San Andreas Fault (Vogl and McHargue 1966) within the WDNA. A handful of other populations exist in Nevada (Cornett 1983) and Arizona.

California fan palm occurs in desert oases with year-round moisture availability (Howard 1992), and in alkaline soils near springs and streams below 3,937 feet (1,200 meters) (Anderson 2000), where it is the dominant canopy species (Szaro 1989). Other woody species that can occur in these communities include tamarisk (Brooks and Minnich 2006; Vogl and McHargue 1966), coyote willow, mesquite, and Fremont cottonwood (Vogl and McHargue 1966).

The presence of water provides for ample fine fuels (Howard 1992) including cattails, reeds, and grasses (Vogl and McHargue 1966). When leaves of a fan palm die, they desiccate and persist along the trunk in what is known as shag (fig. 17) (Turner et al. 1995), which is highly flammable (Vogl and McHargue 1966). With these fuels present, lightning can result in fire even during a heavy winter rain (Vogl and McHargue 1966).

Figure 17—California fan palms in Joshua Tree National Park occur in oases where groundwater emerges along geological fault lines; Riverside County, California, spring 2005 (photo: Amanda D. Webb).

California fan palm usually survives fire (Howard 1992). The trunk resists igniting and flaming, and charring increases resistance to future fires (Vogl and McHargue 1966). Fire clears the understory of competition for resources and facilitates palm recruitment (Brooks and Minnich 2006; Vogl and McHargue 1966). As in other communities, tamarisk in California fan palm communities responds positively to fire, and where present may take advantage of these conditions more rapidly than palms (Brooks and Minnich 2006). Tamarisk structure provides ample ladder fuels and increases incidence of palm crown fires (Brooks and Minnich 2006).

Native Americans are thought to have burned oases approximately every 4 years (Vogl and McHargue 1966), but nonanthropogenic fire regimes are largely unknown. Signs of fire are common in oases along the San Andreas Fault (Vogl and McHargue 1966).

4.2 Fire Behavior, Regimes, and Effects on Nonnative Plants and Plant Communities

This section is organized by plant species or riparian plant communities wherein the species in the subsection title can dominate or codominate vegetation. We have included all nonnative species and communities with fire literature detected through our literature search.

4.2.1 Tamarisk Communities

Several species in the genus Tamarix have expanded their ranges over much of western North America in recent decades, with some of the most pervasive occurrences within the WDNA (Zouhar 2003). Tamarisk abundance varies depending on site conditions, and swaths of dense monotypic tamarisk thickets are common. In contrast, there are some locations where tamarisk makes up only a small part of the vegetation community (Stromberg and Rychener 2010).

Tamarisk is a large shrub that grows up to 26 feet (8–9 meters) tall (Zouhar 2003) with a dense vertical canopy composed of both live and desiccated fine fuels as well as dead branches. In flammability experiments, tamarisk foliage lost water more quickly than cottonwoods and willows (fig. 18). In one study, live tamarisk was more flammable than desiccated foliage (Drus 2013), although another study found desiccated fuels, rather than green foliage, to be the more influential factor in tamarisk volatility (Racher 2003).

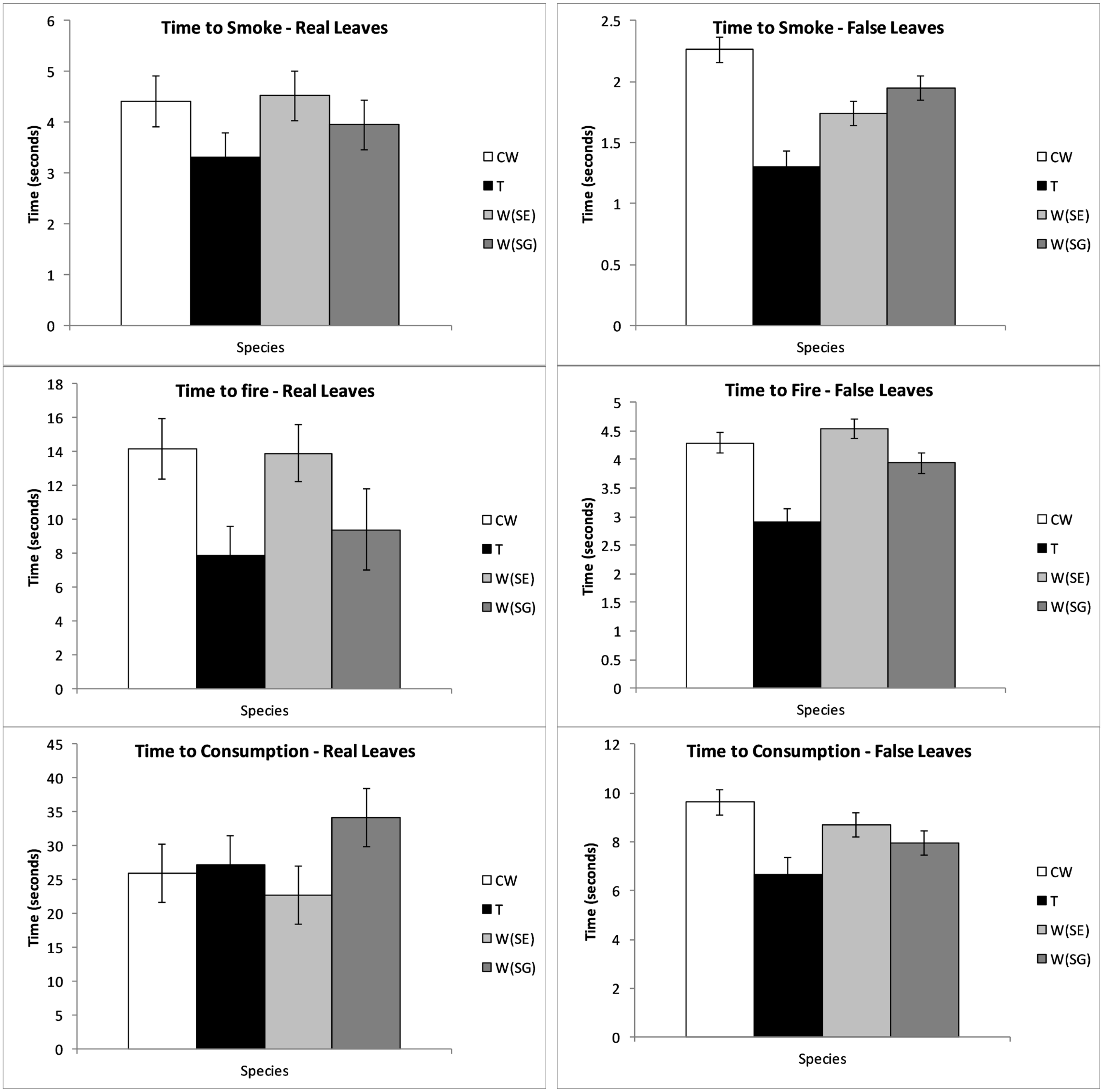


Figure 18—Amount of time it took during laboratory experiments for fresh, real leaves (a–c) and leaf analogs (d–f) to smoke, catch on fire, and be consumed at 1202 °F (650 °C). N = 30 for real leaves; N = 72 for false leaves. CW indicates Fremont cottonwood, W(SE) coyote willow, W(SG) Goodding's willow, and T tamarisk. Error bars show ± standard error. Lowercase letters indicate significant differences among species ($p \leq 0.05$). Reproduced from Drus (2013) with author's permission.

When stands of tamarisk burn, they can contribute to extreme fire behavior, and consequently, dangerous firefighting conditions, often resulting in fires that are very difficult to control (Racher 2003; Racher and Mitchell 1999). Areas dominated by tamarisk that have not burned recently can sustain crown fires with extreme fire behavior over a broad range of conditions and easily shoot fire brands over 492 feet (150 meters) from the flaming front (Racher 2003).

Wildfire can kill aboveground tamarisk tissue (Ellis 2001; Stromberg and Rychener 2010). Laboratory tests on tamarisk seeds collected near Blythe, California examined the survival of seeds and seedlings under different temperatures (121, 149, 177, and 204 °C [250, 300, 350, and 400 °F] for seeds and seedlings, 93 °C [200 °F] for seedlings) applied at varying durations (1, 2, and 5 minutes) (Ohrtman et al. 2012). Seed survival decreased with increasing temperature and longer duration, and survival was higher for seeds that had been sown in moist soils compared to dry soils. Seedling survival decreased with increasing temperature, longer duration, and seedling age, with 1-day-old seedlings being more heat tolerant than 5-day-old seedlings at most temperatures and durations.

Tamarisk is capable of resprouting under a variety of post-fire conditions, including within cottonwood- and willow-dominated systems (Stromberg and Rychener 2010), and in areas of high fire severity (Ellis 2001). Tamarisk is more successful than Goodding's willow and Fremont cottonwood at sprouting and surviving in dry and saline conditions (Busch and Smith 1993). For example, tamarisk had more efficient mechanisms for dealing with water stress than Goodding's willow in burned vegetation along the Colorado River and Bill Williams River floodplains (Smith and Busch 1992). Water use efficiency was higher in tamarisk and arrowweed than in Goodding's willow and Fremont cottonwood, which appears to give tamarisk and arrowweed, both halophytic species, an advantage in post-fire vegetation communities (Smith and Busch 1992). Cottonwood resprouts have a better chance of survival where there is shallower depth to groundwater, whereas tamarisk resprouts are successful in drier soil conditions (Smith et al. 2009). Over time, dense tamarisk can replace cottonwood-willow woodlands where cottonwood-willow habitat is degraded (Smith et al. 2009).

Some studies suggest that relative resprouting rates may vary among tamarisk and native species due in part to fire severity. Resprouting occurred in 53 percent of tamarisk individuals in an area of high fire severity, and in 55 percent of individuals in an area of predominantly low and moderate fire severity (Ellis 2001), a smaller difference than has been reported for some native species.

Tamarisk's stress response to fire can result in increased flowering and seed production (Friggens et al. 2013; Racher and Britton 2003).

Tamarisk resprouts may take as little as 3 months to flower, whereas cottonwoods can take as long as 8 years (Smith et al. 2006b). These reproductive traits help tamarisk colonize quickly in areas that have recently burned (Friggens et al. 2013).

The presence of tamarisk in a riparian ecosystem can have a significant influence on post-fire vegetation communities. Tamarisk fires benefit tamarisk. Tamarisk is adapted to short fire return intervals, resprouts after high severity fire, and in dry or saline soil conditions, can easily recover from fire more quickly than cottonwood and willow (Busch 1995; Busch and Smith 1993; Drus 2013; Ellis 2001; Nagler et al. 2011; Smith and Busch 1992). It can also come to dominate vegetation post-fire that was dominated by mesquite pre-fire (Busch 1995). Post-fire vegetation community shifts toward halophytic species (tamarisk and arrowweed) can be long-lived (Busch 1995).

After assessing 30 sites, Drus (2013) found that mortality (defined as trees that did not resprout after fire or resprouted and later died) and consumption of cottonwoods and willows were associated with increasing pre-fire tamarisk cover. This was due to higher fire intensity where tamarisk is present, resulting in more tissue damage (Drus 2013). When pre-fire tamarisk cover was greater than 50 percent, cottonwood mortality reached 100 percent (fig. 19) (Drus 2013).

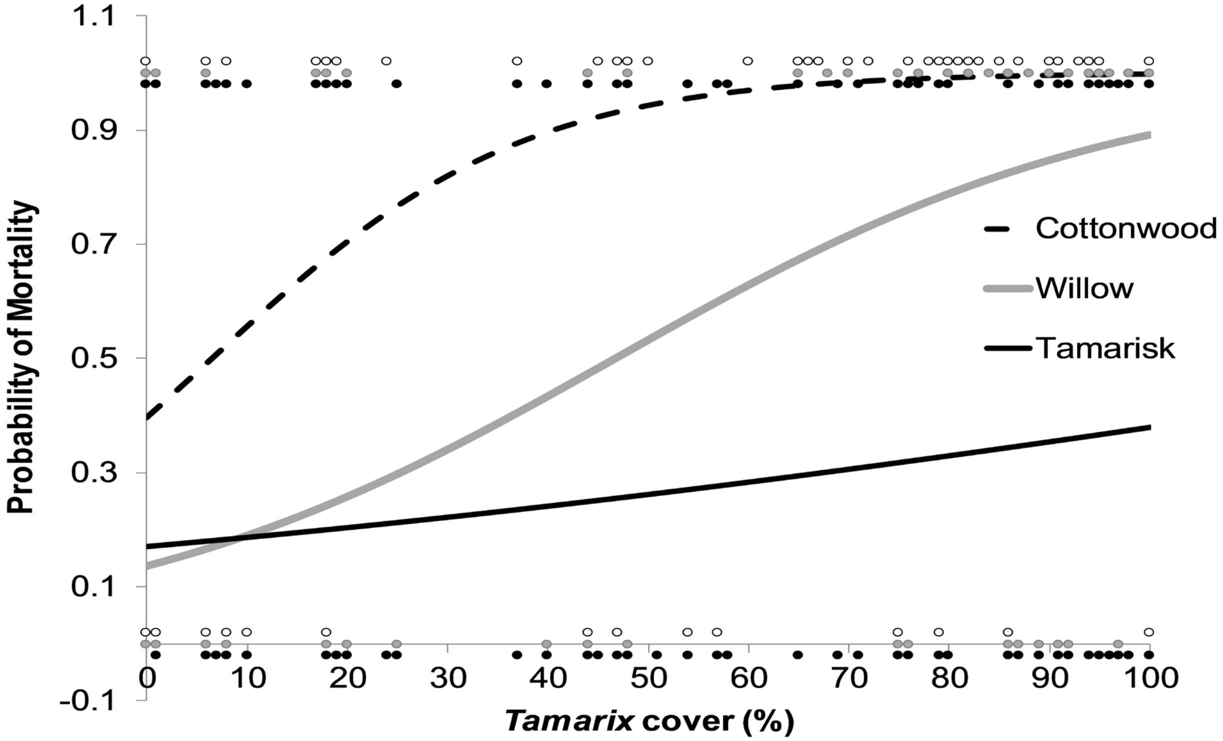


Figure 19—Probability of fire-related mortality of cottonwood (dashed line), willow (gray line), and tamarisk (black line) as a function of tamarisk cover. Points arranged horizontally from 0 (bottom) and 1 (top) along the y-axis represent live (0) and dead (1) individuals. The white points represent cottonwood, gray points are willow, and black points are tamarisk. The points are offset to prevent overlapping. Logistic regression, N = 98 transects. Reproduced from Drus (2013) with author's permission.

A team mapping vegetation along a reach of the Middle Rio Grande in 1995 found evidence of 31 fires that had occurred since a previous assessment in 1984. Where fire had burned, coyote willow was the first woody species to appear, then tamarisk, followed by Russian olive, and last cottonwood (Mount et al. 1996). In another study, burned plots had higher tamarisk density than unburned plots (Smith et al. 2009).

Fire appears to be a significant factor driving vegetation change along the Middle Rio Grande (Mount et al. 1996). Others' preliminary observations on the Rio Grande support the inference that fire is facilitating tamarisk expansion into areas formerly dominated by native trees and shrubs, particularly where groundwater levels have dropped (Smith et al. 2009). Fire is triggering shifts in vegetation away from native species-dominated systems to increased abundance of nonnative species; declining water tables and drought associated with climate change are likely to be creating conditions that favor recovering tamarisk over native trees (Smith et al. 2009).

However, fires do not always favor tamarisk. On the free-flowing San Pedro River in an area dominated by cottonwoods and willows, a high tamarisk mortality rate (calculated from coupled population growth equations using stem densities measured pre- and post-fire) and other factors led to a reduction in tamarisk abundance (Stromberg and Rychener 2010). Where natural flow regimes are more intact and tamarisk is suppressed by native vegetation, native trees may be able to maintain their dominance in the post-fire landscape (Stromberg and Rychener 2010).

Tamarisk invasion in lowland riparian ecosystems in the southwestern United States has been described as having anomalous effects on fire regimes (Busch and Smith 1993; Drus 2013). Tamarisk can increase fire frequency to the point that it prevents full maturation of native woody species, including mesquite, cottonwood, and willow (Busch 1994). The invasion of tamarisk in these ecosystems increases both fire frequency and the likelihood that fire will spread across the riparian zone (Drus 2013).

From 1981 through 1992 on the lower Colorado River, between Davis Dam and the U.S.-Mexican border, 37 percent of riparian vegetation in the study area burned (183 fires burning 16,300 hectares [40,300 acres]), which translates to a fire rotation of about every 35 years (Busch 1995). A subsequent analysis found no significant difference between the total area of all fires combined and the area burned that was dominated by tamarisk before the fire, while the areas of cottonwood-willow and mesquite were each proportionately less than the total area burned. The study also detected an increase in the number of and size of fires over the 12-year period (Busch 1995).

4.2.2 Russian Olive Communities

Russian olive is a woody species that occurs along rivers in the WDNA, and neighboring regions, and as far north as southern Canada (DiTomaso and Healy 2003; Nagler et al. 2011). However, its distribution, abundance, and status within plant communities (e.g., dominant, codominant, understory) are not well documented in the western States within the WDNA (Nagler et al. 2011). One study estimated Russian olive to be present along 7 percent of perennial stream length in Arizona, and 17.2 percent in xeric regions across the western United States (not including New Mexico) (Ringold et al. 2008). Most of the fire-related research we found on this species took place along the Middle Rio Grande.

Russian olive resprouts vigorously post-fire (Brock 1998; Caplan 2002; Parker et al. 2005; Shafroth et al. 2010). Basal resprouting and root suckering have been observed after fire (Smith and Finch 2017). On the Middle Rio Grande, an overall higher percentage of Russian olive individuals resprouted at three burned sites compared to Rio Grande cottonwoods (Smith and Finch 2015, 2017). However, cottonwoods had much higher resprouting rates than Russian olive in mesic areas, and lower resprouting rates in more xeric parts of the riparian woodland (Smith and Finch 2015). Dense thickets of Russian olive along the Middle Rio Grande have been described as fire prone and increasing the risk of catastrophic fire (Caplan 2002). However, there is a lack of literature on Russian olive regarding fire adaptations, fire behavior, and response (Zouhar 2005).

4.2.3 Siberian Elm

Siberian elm inhabits riparian areas and other ecosystems in the southwestern and central United States (Parker et al. 2005), and parts of northern Mexico (SEINet n.d.). Siberian elm resprouts rapidly post-fire, and information concerning its effects on fire frequency and severity is lacking (Parker et al. 2005). Root suckers of Siberian elm have been observed post-fire, and the species may be able to reproduce by seed in burned areas (Smith and Finch 2017).

4.2.4 Giant Reed Communities

Giant reed is found in Arizona, Nevada, New Mexico, California, and other regions of the United States, as well as northern Mexico (DiTomaso and Healy 2003). Giant reed is known for growing in dense monocultural patches of varying size, and in some places, co-occurring with native woody species (Coffman et al. 2010; DiTomaso and Healy 2003).

Giant reed is flammable (fig. 20) and adapted to periodic fires. As it fills in the riparian corridor, it makes native vegetation more susceptible to fire (DiTomaso and Healy 2003). The 2003 Simi/Verdale wildfire burned across the Santa Clara River, California in vegetation of mixed native woody plants (red willow, arroyo willow, coyote willow, black cottonwood, and mule-fat) and giant reed (Coffman et al. 2010). Giant reed appeared to facilitate the spread of fire from one side of the river to the other, where it continued to burn thousands of acres (Coffman et al. 2010).

Figure 20—Giant reed is burned during a prescribed fire in Big Bend National Park, Brewster County, Texas (photo: National Park Service).

Giant reed recovered more rapidly than native woody species and by 1 year post-fire, native riparian species were reduced to 0.4 percent relative cover of total vegetation compared to 25.0 percent pre-fire (Coffman et al. 2010). Productivity of giant reed that was many times higher post-fire than native species may have been influenced in part by increased soil nutrients around giant reed plants post-fire, whereas nutrient levels around burned native plants did not increase significantly with fire (fig. 21) (Coffman et al. 2010).

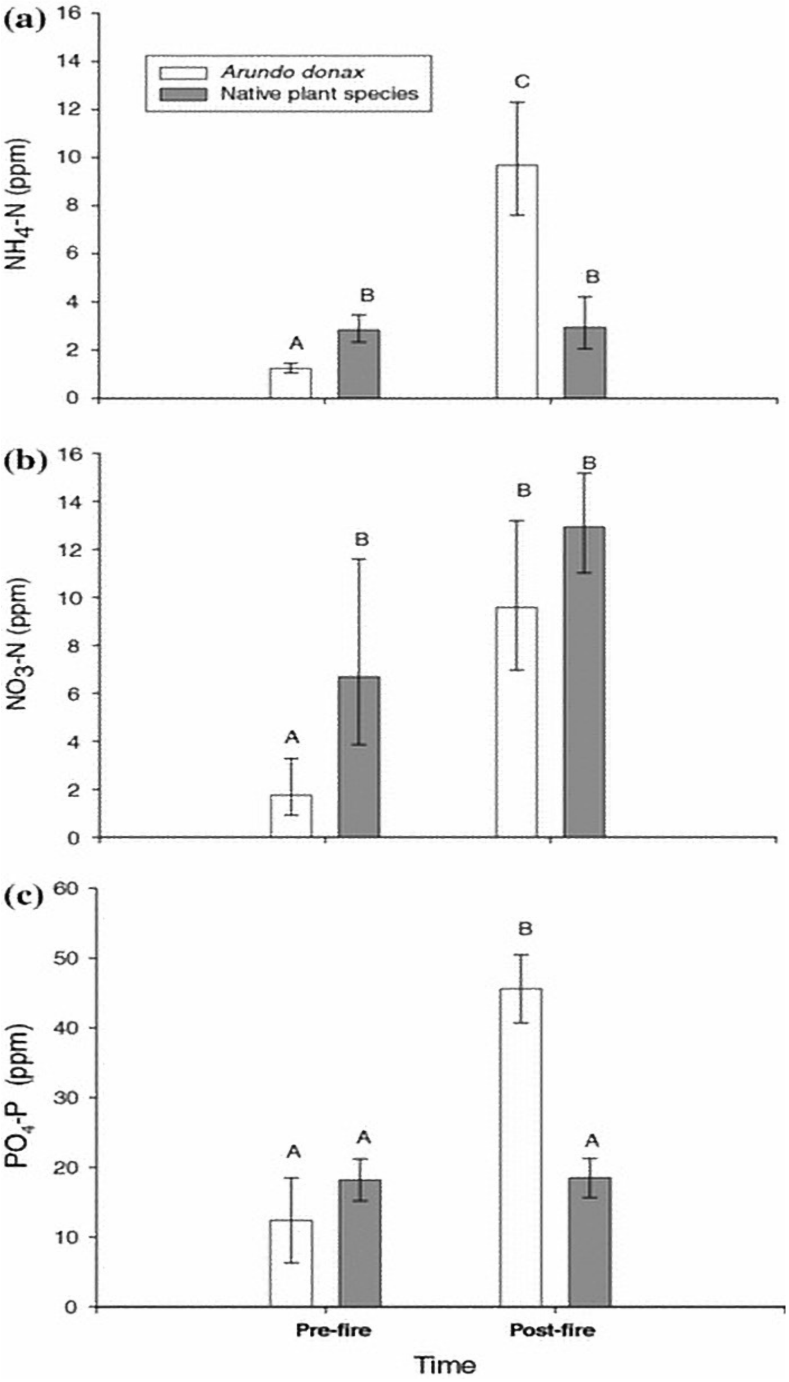


Figure 21—Comparison of pre-fire and post-fire mean nutrient levels in soil adjacent to giant reed and native woody plants. PO4 = phosphate; NO3 = nitrate; NH4 = ammonium. Letters denote results of post-hoc hypothesis tests (comparison of means) with significance at α < 0.05. Reproduced from Coffman et al. (2010) under the Creative Commons Attribution Noncommercial License, published by Springer Nature.

4.2.5 Common Reed

Observations and herbarium collections accessible through the Southwest Environmental Information Network (SEINet) indicate occurrences of common reed in all U.S. States within the WDNA and in parts of Mexico including Sonora and Baja California (fig. 22). Native and nonnative genotypes of this species are present within the WDNA (Meyerson et al. 2010). Common reed can be a desirable species in areas where it is native, but it can also be weedy and invasive (DiTomaso and Healy 2003).

Figure 22—Common reed surrounds open waters of the Rio Grande in Big Bend National Park along the U.S. border with Mexico; Brewster County, Texas, February 2016 (photo: Amanda D. Webb).

The search for this taxon in databases, including the FEIS, returned no studies on the relationship between common reed and fire in the WDNA. Gucker (2008) surmises from existing literature that common reed stands are highly productive and flammable, and facilitate fire spread.

4.2.6 Johnsongrass

Observations and herbarium collections accessible through the Southwest Environmental Information Network indicate that Johnsongrass occurs in all U.S. States within the WDNA and several States in Mexico including Sonora, Chihuahua, and Baja California and extending as far south as Aguascalientes (SEINet n.d.).

The only source found on this species was a review in the FEIS. However, of the studies reviewed for this FEIS summary, none contained information on the relationship between fire and Johnsongrass within the WDNA.

Howard (2004) surmises from existing literature that Johnsongrass can increase fine fuel loads and continuity, and may be able to survive severe fire due to the deep rhizomes from which it sprouts, although this adaptation may not be present in all populations. A laboratory study investigating the influence of heat applied for 120 seconds on Johnsongrass seed germination found no change in germination between the control, 200 °F (93 °C), and 400 °F (204 °C) treatment groups, but reduced germination at 600 °F (316 °C), and germination failure at 800 °F (427 °C) and 1000 °F (538 °C) (Mitchell and Dabbert 2000).

4.3 Fire Effects on Abiotic Ecosystem Components and Processes

Physical aspects of the environment, such as hydrology, stream morphology, water quality, and soil characteristics, are the foundation for riparian plant communities and other biota. Fire can alter these conditions and processes directly and indirectly (DeBano and Neary 1996). Because of their influence on vegetation structure and composition, fire effects on abiotic factors are important for predicting post-fire plant population trajectories (Smith et al. 2009).

Of 67 literature sources addressing the direct effects of fire on lowland riparian ecosystems, 14 sources address fire effects on abiotic ecosystem components of lowland riparian systems in the southwestern United States, and 1 source discusses these effects in Mexico.

4.3.1 Hydrology and Geomorphology

The range of geomorphological and hydrological processes understood to be influenced by fire has increased in recent decades (Shakesby and Doerr 2006). Wildfire is a major driver of geomorphological change and its effects can extend well beyond the fire perimeter, particularly in other portions of the channel corridor (Moody and Martin 2009). Channel systems respond to the effects of wildfire on vegetation and soil in complex ways, and understanding fluvial response to fire is important for improving prediction of problems associated with post-fire flooding, channel instability, and related water supply implications (Shakesby and Doerr 2006). However, the number and types of investigations examining wildfire effects on these processes in lowland riparian ecosystems of the desert Southwest and Mexico are few compared to studies in the region's montane ecosystems. Studies of fire effects at the catchment scale appear more common than fire effects in riparian ecosystems.

An important impact of wildfire in riparian ecosystems is its effect on erosion and stream morphology. Post-fire effects on stream channels involve complex feedback mechanisms, with changes occurring over short and long intervals (Shakesby and Doerr 2006). In a meta-analysis of post-fire erosion and sediment movement within 2 years of a wildfire (data collected over an 80-year period across the western United States, including montane systems), approximately 25 percent of post-fire coarse-grained sediment yield came from hillslope erosion, while 75 percent originated within channels (Moody and Martin 2009). Channels are areas where sediments are stored, and post-fire floods can mobilize these sediments (Moody and Martin 2009; Shakesby and Doerr 2006). The magnitude of erosion and sediment transport depends on the timing and intensity of post-fire rainfall and subsequent flooding, and sediment yields are determined more by sediment availability than by soil erodibility or slope (Moody and Martin 2009).

Infiltration, evapotranspiration, and soil moisture are often affected by fire, especially when high intensity combustion sends a persistent heat pulse deep into the soil profile. Fire changes rates of water infiltration into soil (Shakesby and Doerr 2006), most notably when hydrophobic conditions are created, although this does not appear to have been studied in riparian areas of the southwestern United States. In a review of other existing literature, Shakesby and Doerr (2006) found that a reduced infiltration rate is most commonly reported and sometimes associated with higher burn severity, such as with hydrophobicity. In some environments, fire can decrease soil moisture in upper soil layers by altering conditions that influence evapotranspiration (Busch and Smith 1993). Wildfire led to an increase in evapotranspiration in a wetland environment dominated by southern cattail in the Colorado River delta (Glenn et al. 2013). Post-fire decreases of water content in upper soil layers may be less severe in areas with shallow water tables, which may allow for moisture replenishment from groundwater (Busch and Smith 1993).

4.3.2 Water Quality

Water quality is important for sustaining aquatic organisms and humans. Fires that kill or consume riparian trees can reduce shade cover over streams, leading to increased water temperature and a decrease in dissolved oxygen (Corbin 2012; DeBano and Neary 1996).

The input of ash, debris, and eroding sediments after a fire decreases water quality (Corbin 2012). Post-fire erosion in the stream channel and surrounding areas greatly affects stream sedimentation levels (Desilets 2007; Moody and Martin 2009). During selected rain events in the 18 months following the 2003 Aspen Fire in Arizona, much of the suspended sediment (fine to coarse) in Sabino Creek came from hillslopes, and this was largely attributed to ash and fine sediments from destabilized soils from higher regions of the steep watershed after the wildfire (Desilets 2007).

4.3.3 Woody Debris

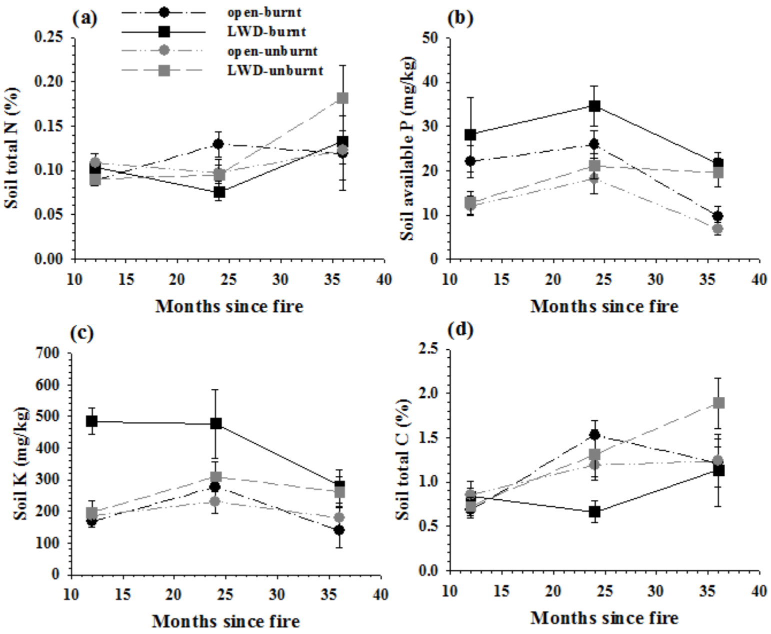
Large woody debris influences fire effects on abiotic processes in several ways, and debris interaction with fire, vegetation, and fluvial processes can create ecological feedback loops (Bendix and Cowell 2010a).

The presence and extent of large woody debris depend on woody plant mortality, and disturbances such as floods and fires can be sizable sources of woody debris (Bendix and Cowell 2010a; DeBano and Neary 1996).

Woody debris plays an important role in riparian and aquatic ecosystems. Piles of woody debris deposited during flood events in stream channels and floodplains increase hydraulic roughness and act to deflect or block streamflows (Bendix and Cowell 2010a; DeBano and Neary 1996). Debris slows stream velocity, or redirects streamflows, and thereby greatly affects channel morphology and sediment deposition (Bendix and Cowell 2010a; DeBano and Neary 1996). Sediment deposited due to the presence of woody debris can provide habitat for riparian plants and therefore influences species composition and distribution (Bendix and Cowell 2010a). The presence of woody debris can also help stabilize streams following disturbance (DeBano and Neary 1996). A lack of woody debris can accelerate sedimentation and removal of nutrients as water carries more sediment through the system with potential negative impacts to water quality (DeBano and Neary 1996). In a study in South Africa, soil-available phosphorus, soil exchangeable potassium, and soil pH were higher in burned quadrats that included large woody debris piles compared to burned quadrats that did not have piles, indicating woody debris can influence the mosaic of post-fire soil conditions and communities of recovering vegetation (fig. 23) (Pettit and Naiman 2007b).

Large woody debris deposited by floods before burning can also be a significant source of concentrated fuels, increasing the risk of more frequent fires and higher severity fires, and increasing fire residence times (Pettit and Naiman 2007b). Therefore, the presence of large woody debris

Figure 23—Soil N (nitrogen), P (phosphorus), K (potassium), and C (carbon) levels at 1, 2, and 3 years after fire in burned and unburned quadrats where large woody (LW) debris piles were present and absent (open). Error bars show ± standard error. Sabie River, Kruger National Park, South Africa, 2003–2005. Reproduced from Pettit and Naiman (2007b) with permission from the publisher, John Wiley and Sons. © 2007 by the Ecological Society of America.



has the potential to increase risk of tree mortality during fire (Bendix and Cowell 2010a; Ellis 2001). Post-fire woody debris in the form of snagfall also contributes to fuel loads (Bendix and Cowell 2010a).

The amount and timing of input of woody debris into waterways from post-fire snagfall depend on two variables: 1) subsequent flood events, and 2) species composition of snags (Bendix and Cowell 2010a). Flooding can increase rates of snagfall when flowing water fells snags (Bendix and Cowell 2010a). Independent of flooding, however, there is evidence that various tree species remain standing for different lengths of time after burning, with some species falling shortly after a fire and others standing for years or decades. As a consequence, recruitment of woody debris may occur in multiple pulses post-fire (Bendix and Cowell 2010a). Two years after the Sevilleta Fire (April 2011) and 3-4-6 Fire (June 2011) along the Middle Rio Grande, Russian olive branches and snags were more likely to fall than cottonwoods, and the type of woody debris accumulation differed among species and between the two sites (fig. 24) (Smith and Finch 2015).

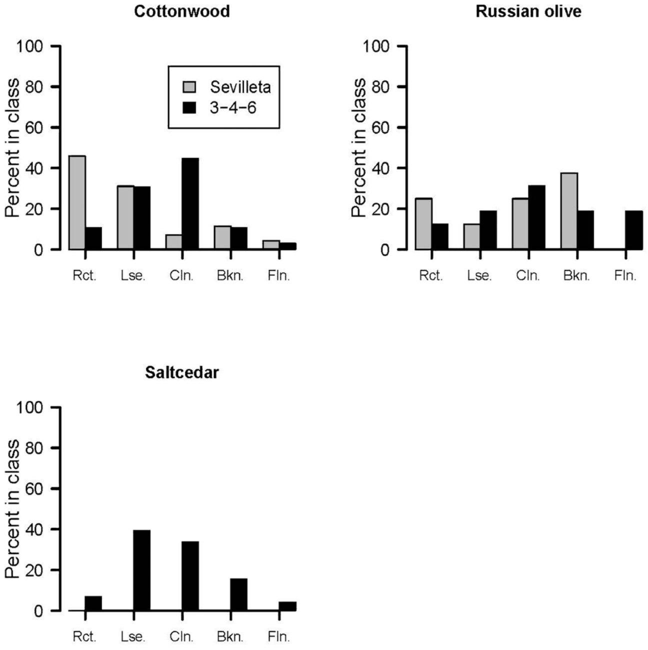


Figure 24—Percentage of snags in each of four decay classes at two wildfire sites. Classes are recent snag (rct.), loose-bark snag (lse.), clean snag (cln.), broken snag (bkn.; bole broken above diameter at breast height [d.b.h.]), and fallen snag (fln.; uprooted or broken below d.b.h.). Figure reproduced from Smith and Finch (2015) with permission.

4.3.4 Soils

By consuming vegetation and organic matter, fire alters nutrient levels and nutrient availability in soils. Fire may result in increases of many nutrients (Busch and Smith 1993; Pettit and Naiman 2007b). Plant cover in recovering vegetation can be higher in areas with higher available phosphorus post-fire (Pettit and Naiman 2007b). However, pH may increase with fire and subsequently alter which nutrients are available for uptake by vegetation (Busch and Smith 1993; Pettit and Naiman 2007b). Higher salinity and boron levels may be detrimental to the recovery of certain plant species after fire, and may give tamarisk an advantage over some native species (Busch and Smith 1993). On the Santa Clara River in California, an analysis of soil nutrients pre- and post-fire near giant reed and native species showed that nutrient levels increased with fire around giant reed and did not increase around native vegetation (Coffman 2010).

How long nutrients persist in a given place may depend on post-fire precipitation and subsequent erosion and flooding as water can transport nutrients (Busch and Smith 1993). Fire can increase soil erodibility by decreasing vegetation and litter cover (Shakesby and Doerr 2006), although this does not appear to have been documented in riparian areas within the target geography of this report. The likelihood of nutrients being transported by water is diminished in riparian ecosystems with infrequent flooding (Busch and Smith 1993).

4.4 Prescribed Fire, Fuel Treatments, and Post-fire Rehabilitation

There is limited information on prescribed fire and its utility as a restoration tool (fig. 25), and fire effects in areas of fuel reduction treatments. Tamarisk was the only taxon represented in the prescribed fire literature except for one paper on a laboratory experiment investigating fire effects on Johnsongrass seeds (Mitchell and Dabbert 2000). All study areas from prescribed fire literature were in the United States.

Prescribed fire effects on tamarisk are variable (Racher 2003). Prescribed fires in tamarisk can be intense and erratic, and can become uncontrollable (Jorgenson 1996; Racher and Mitchell 1999). In one experiment used to develop burning prescription recommendations for tamarisk, fire brands carried more than 492 feet (150 meters) (Racher 2003). Tamarisk stands that have not burned in several years exhibit extreme fire behavior across a wide spectrum of weather conditions (Delwiche 2009). In contrast, under certain conditions, tamarisk can be

Figure 25—A fire crew member uses a pneumatic torch to start a prescribed fire in Bosque del Apache National Wildlife Refuge, Socorro County, New Mexico (photo: U.S. Fish and Wildlife Service).

nearly impossible to burn, such as when the fuel has a high live moisture content (Jorgenson 1996). Another factor that may influence tamarisk fire behavior is time since last burn (Racher et al. 2001). To increase tamarisk consumption, one option is to cut down 20 to 25 percent of large trees 1 to 2 months before burning to create dry ground fuels that preheat tamarisk, and then burn it with a head fire instead of a backing fire to gain more heat (Jorgenson 1996).

As a stand-alone treatment, prescribed fire is not effective at eliminating tamarisk due to the species' adaptive response to fire, but it can be useful for reducing fuel loading, or when paired with other treatments. Prescribed fire may be effective in controlling tamarisk when applied multiple times over several years (Racher and Britton 2003), in combination with herbicide treatments (Jorgenson 1996), or when combined with mechanical removal treatments and herbicide (Finch and Dold 2008). Lovich et al. (1994) reported varying degrees of success combining cut-stump herbicide application and prescribed burning followed by additional herbicide applications. On the Middle Rio Grande, mechanical removal of nonnative plants followed by prescribed fire was more effective at reducing ground fuels than mechanical removal alone or followed up with planting of native species (Bateman et al. 2012).

Prescribed fire experiments by Racher and colleagues on the Pecos River floodplain in New Mexico found that tamarisk mortality varied greatly within treatment areas, with no consistent mortality associated with phenological stage (Delwiche 2009). Mortality was low or nonexistent in areas that had burned 5 years earlier compared to areas that were not previously burned, where mortality averaged 30 percent (Delwiche 2009).

Although prescribed burning may not be effective for eliminating tamarisk, it can be a low-cost tool for preventing development of dense monotypic stands, which can increase fire risk and hazards (Racher 2003). Using prescribed fire to reduce biomass can open the canopy to allow easier access for subsequent herbicide applications (Jorgenson 1996; Lovich et al. 1994). Using fire in this way can also slow tamarisk spread, promote herbaceous vegetation by opening the canopy, and potentially promote diversity (Racher et al. 2001). Such treatments may reduce the potential for costly wildfires and property damage (Racher et al. 2001).

In a study examining revegetation success following tamarisk control along the Colorado River and the Middle Rio Grande, the method of tamarisk removal did have a significant effect on restoration outcomes (Bay and Sher 2008). Removal methods were burning, chemical application, mechanical clearing, or root-plowing, or some combination of these. Instead of removal methods, factors that were correlated with greater tamarisk reduction were those related to water availability, including higher precipitation, flooding since the time of tamarisk removal, and closer proximity to perennial water.

Laboratory tests were conducted on tamarisk seeds collected near Blythe, California to determine survival of seeds and seedlings under different temperatures applied at varying durations (Ohrtman et al. 2012). Results suggest that prescribed fire could be used to reduce tamarisk seedling recruitment if temperature and duration thresholds can be reached in the riparian environment.

Although tamarisk can create intense and unpredictable fires, sometimes tamarisk will not carry prescribed fire. In January 1996 at a site along the Colorado River in California, tamarisk apparently had such a high live foliar moisture content that it would not burn despite abundant dead foliage from the dormant plant (Jorgenson 1996). At a different site in California, an unsuccessful October prescribed fire was followed up with a 20 percent reduction of standing tamarisk by chainsaw and then burned again in December (Jorgenson 1996). The cut woody material dead on the ground dried and then burned, creating conditions that helped carry fire into the stand of tamarisk (Jorgenson 1996).

Restoration experiments were conducted along the Middle Rio Grande (Anjozian 2008; Finch and Dold 2008). The experimental sites were characterized by overstories of Rio Grande cottonwood, understories of woody native and nonnative species, and high fuel loads. Treatments included mechanical removal of live, dead, and downed nonnative woody plants (fig. 26), mechanical removal combined with herbicide treatments and prescribed fire, and mechanical removal combined with herbicide and native plant revegetation. Treatments resulted in reduced ground fuels, fewer exotic woody plants, and more open understories. Among the conclusions was that nonnative woody plant removal reduces fire risk, particularly the risk of high intensity fires, and so aids in the preservation of native riparian trees (Finch and Dold 2008).

Figure 26—Fuel reduction treatments are carried out along the Middle Rio Grande in central New Mexico (photo: USDA Forest Service, Rocky Mountain Research Station).

In a laboratory experiment, germination rate of Johnsongrass seeds was reduced when temperature applied for 120 seconds was increased from 400 °F (204 °C) to 600 °F (316 °C), and germination was eliminated when temperature increased from 600 °F to 800 °F (427 °C) (Mitchell and Dabbert 2000). The investigators surmised that prescribed fire produces sufficient heat to kill Johnsongrass seeds on the soil surface.

Depending on how fuel treatments are executed, these actions may not always be successful in reducing fire risk to native trees. At sites along the Middle Rio Grande, standing tamarisk fuels were cut, chipped, and distributed on the forest floor (Johnson and Merritt 2009). The authors found no evidence that this fuel reduction technique reduced cottonwood mortality (as determined by no live canopy) during subsequent wildfires, and argued that fuels were not reduced but rather were simply altered in structure and distribution, including piled chips around the bases of trees. Study results suggested that fuel load, rather than fuel structure, was more influential in cottonwood bole mortality (Johnson and Merritt 2009).

Post-fire rehabilitation can be difficult in riparian ecosystems. Prescribed fire can be used at sites where enhanced nutrient dynamics are desired, but not only are native species more likely to be injured in high intensity fires than nonnative species, they can also be negatively impacted where burning tamarisk results in salt-laden ash falling from burned vegetation and being added to the soil (Shafroth et al. 2010). Factors that can limit success include nitrogen limitation, salinity, hummocky microrelief, compaction from livestock trampling, and tamarisk leaf litter (Lair 2006). Mature, dense stands of tamarisk can negatively impact the arbuscular mycorrhizae which are symbiotic to species that may be used in revegetation efforts (Lair 2006).

High post-fire soil salinity levels can be improved by mechanically creating microtopographic relief patterns on soil surfaces and by using soil amendments (Shafroth et al. 2010). The effectiveness of amendments is constrained by the cost of application rates, because to achieve maximum efficacy, these products need to be incorporated into soils using tillage or irrigation, which is often not feasible (Shafroth et al. 2010).

Section 5 - Discussion: Literature Review Synthesis

> *"In its natural role, fire is not a disturbance that impacts systems; rather it is an ecological process that is as much a part of the environment as precipitation, wind, flooding, soil development, erosion, predation, herbivory, carbon and nutrient cycling, and energy flow. Fire resets vegetation trajectories, sets up and maintains a dynamic mosaic of different vegetation structures and compositions, and reduces fuel accumulations. Humans have often disrupted these processes, and the result can be that fire behavior and effects are outside of the range of natural variation. At that point, fire is considered an exogenous disturbance factor."*
>
> —Sugihara et al. (2006, p. 62)

Fire is a natural ecosystem process that provides many important ecological benefits to native biota. However, in the Warm Deserts of North America (WDNA), where lowland riparian ecosystems have undergone major changes in hydrology and vegetation composition and structure, increasingly frequent and severe fires threaten native biotic communities, and ecosystem integrity and function (fig. 27).

Figure 27—Ash covers the ground in this high severity burn area along the Middle Rio Grande in central New Mexico (photo: USDA Forest Service, Rocky Mountain Research Station).

Riparian ecosystems are dynamic, and site-specific conditions can change abruptly over time and space. Post-fire outcomes vary accordingly (Dwire and Kauffman 2003) and are influenced by multiple interacting factors, including:

- Pre-fire and post-fire streamflow regimes (Ellis 2001; Glenn and Nagler 2005)
- Channel morphology (Bendix and Cowell 2010a)
- Soil characteristics (Busch and Smith 1993; Coffman et al. 2010; Pettit and Naiman 2007b; Sunderman 2009)
- Depth to groundwater (Busch and Smith 1993; Smith et al. 2009)
- Climate (Shakesby and Doerr 2006; Smith et al. 2009)
- Post-fire precipitation characteristics (Smith et al. 2009)
- Pre-fire vegetation community composition or structure, or both (Busch 1995; Coffman et al. 2010; Drus 2013; Glenn and Nagler 2005; Stromberg and Rychener 2010; Stuever 1997)
- Recovery mechanisms of individual taxa (Busch and Smith 1993; Coffman et al. 2010; Glenn and Nagler 2005; Smith and Busch 1992)

5.1 What Are the Primary Patterns, Drivers, and Trends in Fire Severity and Frequency?

Land use, water use, and climate are primary drivers of riparian area condition. The effects of these on hydrological and geomorphological conditions constrain what vegetation and fuel types can occur at a single point in space and time. In the long term, climate and macrotopography determine the distribution of biomes; in the short term, land and water management can have immediate effects on riparian ecosystems that people can control. People can address climate change issues through water and land management, by reducing greenhouse gas emissions, and through climate change adaptation actions (Garfin et al. 2013).

Information on historical fire regimes is very limited, and the full variability of historical fire regimes within and among lowland riparian areas is not well understood. However, there is much to be learned from the patterns and trends observed over the last 100 years of human activity and ecosystem change in riparian areas. Given that lowland riparian ecosystems are heterogeneous in the region and fire regimes are highly variable today, it is likely that they also varied historically.

5.1.1 It Is Difficult to Generalize About Historical Fire Regimes Across the Southwestern United States and Northern Mexico

The diversity of riparian ecosystems and their dynamic nature, combined with limited historical knowledge and centuries of ecosystem changes from nonindigenous peoples, make it difficult to generalize about historical, or “natural,” fire regimes across such a large geographic region.

Historical photos show that considerable vegetation change has occurred in many riparian ecosystems since the mid- to late 1800s (Webb and Leake 2006). Some evidence suggests increasing woody vegetation in areas that were previously herbaceous marsh-like environments, while other evidence suggests declines in riparian forests (Webb and Leake 2006). These disparities could explain some disagreement in the literature about how riparian fire regimes have changed over time. Though there is agreement that fire frequency and severity have been increasing in recent decades, there are disparities regarding historical conditions and fire regimes, and whether current shifts in fire regimes are moving further away from, or closer to, historical conditions.

Fire is considered by some to have been uncommon in riparian ecosystems historically or prehistorically with subsequently increasing fire intensity (Dwire and Koffman 2003; Friggens et al. 2013; Smith and Busch 1992). On the lower Colorado River, the Gila River in central Arizona, and the Mojave River in California, several reaches that formerly supported riparian woodlands no longer do so, and woody vegetation is dominated in many places by tamarisk (also known as saltcedar) (Webb and Leake 2006), leading to increased fire frequency and severity.

In other areas, such as the San Pedro River in Arizona, it has been suggested that increasing fire frequency in this system in the late 1900s and early 2000s may be shifting vegetation trajectories back toward a more herbaceous system, which could be more similar to historical or prehistoric conditions (Stromberg et al. 2009). The occurrence of extensive stands of riparian woodlands may be a more recent development in areas that were formerly barren or dominated by herbaceous vegetation, mainly due to channel downcutting (Webb and Leake 2006). An analysis of riparian vegetation and environmental changes along the San Pedro between 1935 and 2003 indicates that the area of vegetation dominated by Fremont cottonwood and Goodding's willow has been increasing (Stromberg et al. 2010).

However, inferences about changes in vegetation and fire regimes over time are limited due to lack of historical or prehistoric data. In one study that analyzed historical photographs to understand riparian vegetation change over time along the San Pedro River, the photographic record begins in the mid- to late 1800s (Webb and Leake 2006). Another study analyzed change on the San Pedro between 1935 and 2003 (Stromberg et al. 2010). Riparian areas had already been through a period of rapid ecological change due to natural resource uses that new settlers introduced. For example, Spanish explorers brought livestock with them in the 1500s, settlers cut timber for hundreds of years, and in the 1800s, mills along the San Pedro used river water to process ore from mines elsewhere in the valley, and fur trappers hunted beavers until they were extirpated (Fish et al. 2006; Sayre 2011). Changes that had already occurred along the San Pedro River prior to these study dates had very likely triggered significant alterations to riparian ecosystem structure and function relative to what was common when the predominant cultures were indigenous peoples (fig. 28). These changes had a profound influence in shaping the floodplain, vegetation, and fire regimes of the San Pedro riparian zone and surrounding uplands (Fish et al. 2006; Sayre 2011).

Figure 28—Beaver dams like this one at Bill Williams River National Wildlife Refuge had become uncommon in the southwestern United States by 1900; Mohave County, Arizona, summer 2012 (photo: Amanda D. Webb).

5.1.2 Fire Frequency and Severity Are Increasing in Some Riparian Ecosystems

Fire frequency and severity are increasing in at least some riparian ecosystems due to altered hydrological regimes, drought, and their influences on native and nonnative plants. In today's riparian ecosystems, fires can potentially encourage growth of native vegetation by providing seeds access to mineral soil, removing competing vegetation, and releasing nutrients (fig. 29) (Brooks and Minnich 2006; Busch and Smith 1993; Ellis 2001; Friggens et al. 2014; Pettit and Naiman 2007b; Smith and Finch 2015; Sunderman 2009; Vogl and McHargue 1966). However, conditions that lead to more frequent and more severe fires (fig. 30) typically result in widespread top-kill of native trees (Drus 2013; Ellis 2001; Stuever 1997), and change plant successional status dramatically or trigger a rapid conversion to new vegetation types (Busch 1995; Friggens et al. 2013).

Fire regimes in riparian vegetation communities dominated by native species vary greatly across the diverse riparian environments of the WDNA. LANDFIRE's BpS models indicate that lowland riparian fire regimes fall into three different fire regime categories (see table 4 in section 4): Fire Regime Group I (≤35-year return interval, low and mixed severity), Fire Regime Group III (35- to 200-year frequency, low and

Figure 29—Fire burning broken branches and flood-deposited debris on the ground along the lower San Pedro River during the 2015 Malpais Hill Fire; Pinal County, Arizona (photo: Jeff Lark, The Nature Conservancy, used with permission).

Figure 30—Standing dead tamarisk burns at Bosque del Apache National Wildlife Refuge; Socorro County, New Mexico (photo: U.S. Fish and Wildlife Service).

mixed severity), and Fire Regime Group V (>200-year fire return interval, any severity) (LANDFIRE 2008a).

Fire frequency is increasing in at least some lowland riparian ecosystems in response to drought, close proximity to human activity that leads to anthropogenic ignitions, and the presence of tamarisk (Busch 1995; Drus 2013; Parker et al. 2005; Stuever et al. 1995; Sunderman and Weisberg 2012; Weisenborn 1996). Where it occurs, tamarisk invasion can reduce fire intervals from more than 200 years, or 35 years or more, to 15 years (Drus 2013; LANDFIRE 2008c).

In some lowland riparian ecosystems, fire severity is also increasing due to altered flood disturbance, high fuel loads, flammability, and continuous homogenous fuel structure (Drus 2013; Ellis 2001; Friggens et al. 2013; Glenn and Nagler 2005; Johnson and Merritt 2009). Although increases in fuel loads and continuity in some places may not be due solely to nonnative plants, increases in fuel loads, flammability, and continuous fuel structures resulting from nonnative plant invasions in riparian woodlands and bosques are the primary contributing factors to increasing fire severity. For example, tamarisk creates more intense and erratic fire behavior than native trees, and fire effects are more severe where it is present (Drus 2013; Racher 2003). By influencing the spatial distribution of snags, woody debris, and litter, burn intensity affects abiotic conditions and subsequent vegetation and other fuels (Bendix and Cowell 2010a; DeBano and Neary 1996; Friggens et al. 2014; Smith and Finch 2015).

Both fire frequency and severity are increasing in some riparian systems due to the growing extent of nonnative plants that alter fuel properties (increased flammability and continuity), including increased fuel loading (increased biomass). Whether caused by drought, land use, or human engineering, changes in the underlying physical environment, such as surface flows, groundwater characteristics, and channel and floodplain morphology, can lead to reduced habitat quality and regeneration opportunities for native trees. This results in more opportunities for tamarisk to spread (Busch 1995; Busch and Smith 1993; Drus 2013; Friggens et al. 2013; Jemison 2003; Merritt and Poff 2010; Mortenson and Weisberg 2010; Nagler et al. 2011; Parker et al. 2005; Smith and Busch 1992; Smith et al. 2009; Webb and Leake 2006). By altering riparian vegetation composition and structure, fuel characteristics are changed, resulting in divergent fire behavior.

Altered fuel complexes is one pathway by which hydrological regimes indirectly affect fire behavior and regimes. For example, tamarisk has come to dominate in many areas, often where streams have been dammed, diverted, and channelized (Busch and Smith 1995; Levine and Stromberg 2001; Merritt and Poff 2010; Ringold et al. 2008; Smith et al. 1998). Channelized streams provide better habitat conditions for tamarisk over cottonwoods and willows, and vegetation dominated by tamarisk promotes stream incision, contributing to dropping water tables (Busch and Smith 1995; Glenn and Nagler 2005). An analysis of nonnative plant species occurrences in the western United States found that tamarisk and Russian olive are especially prevalent on large streams and are associated with dams, disturbance to the riparian corridor, and in the case of Russian olive, disturbances elsewhere in the watershed (Ringold et al. 2008). However, the prevalence of Russian olive in the region may have as much or more to do with factors other than flow regulation (Mortenson and Weisberg 2010), which highlights the importance of looking at species individually instead of assuming that flow regulation is the main factor influencing the spread of all nonnative plants. This distinction is important for assessing restoration potential (Merritt and Poff 2010; Mortenson and Weisberg 2010). Although we found limited information about many nonnative species (Russian olive, giant reed, common reed, Siberian elm, Johnsongrass), anecdotal evidence combined with studies from other regions of North America suggests that these species could have impacts on riparian fire behavior and fire regimes in the WDNA that would negatively affect cottonwood-willow woodlands (Caplan 2002; Coffman et al. 2010; DiTomaso and Healy 2003; Gucker 2008; Howard 2004; Parker et al. 2005; Zouhar 2005).

Depth to groundwater is a major factor influencing vegetation type and extent in riparian ecosystems (Busch and Smith 1995; Glenn and Nagler 2005; Jemison 2003; Lite and Stromberg 2005). Where groundwater levels have dropped, tamarisk has a competitive advantage over cottonwoods (Lite and Stromberg 2005). In a study on the San Pedro River in Arizona, Fremont cottonwoods and Goodding's willows of diverse age classes dominated over tamarisk under natural flow regimes where surface water flowed more than 76 percent of the time and depth to groundwater averaged less than 8.5 feet (2.6 meters) and fluctuated less than 1.6 feet (0.5 meter) during the year (Lite and Stromberg 2005). Areas where groundwater was deeper and surface flows were more intermittent had higher tamarisk cover and less cottonwood and willow cover (fig. 31) (Lite and Stromberg 2005).

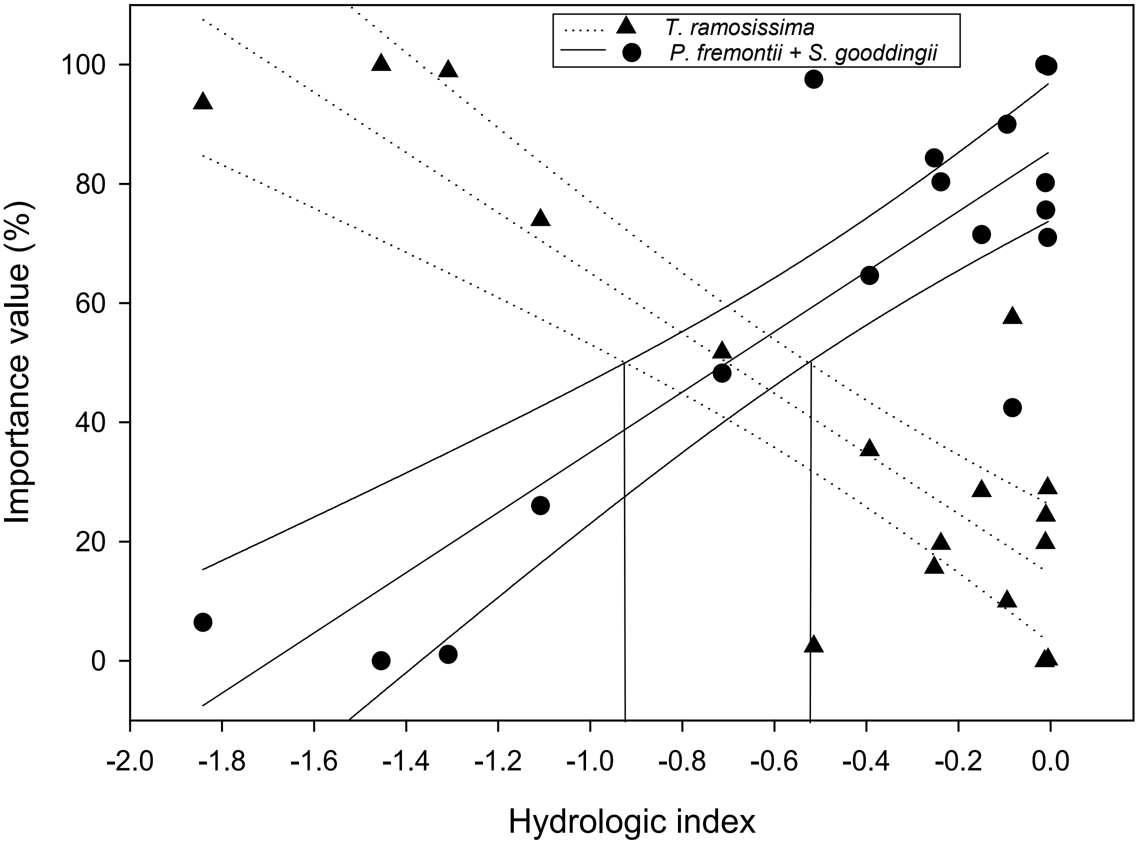


Figure 31—Hydrological thresholds for maintaining Fremont cottonwood-Goodding's willow stands on the San Pedro River in Arizona, based on data collected over a 2-year period (regression lines with 95-percent confidence intervals). Importance values summarize relative abundances of tamarisk, and Fremont cottonwood and Goodding's willow combined. Higher importance values indicate higher relative abundances. Wetter conditions are represented by higher hydrologic index values. Figure reprinted from Lite and Stromberg (2005) with permission from Elsevier.

Altered flood disturbance affects fire frequency and severity through multiple pathways (fig. 32). Where the natural flood regime has been disrupted and flows are less variable, tamarisk is able to colonize areas along the stream channel and throughout the floodplain more successfully while these conditions prevent cottonwood recruitment, changing fuel load properties (Levine and Stromberg 2001; Lite and Stromberg 2005; Merritt and Poff 2010; Nagler et al. 2011; Stromberg 2001). The results of several studies suggest that where soil and hydrological conditions are sufficient to allow cottonwoods to germinate and grow, cottonwoods can outcompete tamarisk (Bay and Sher 2008; Bunting et al. 2011; Levine and Stromberg 2001; Sher and Marshall 2003; Sher et al. 2002).

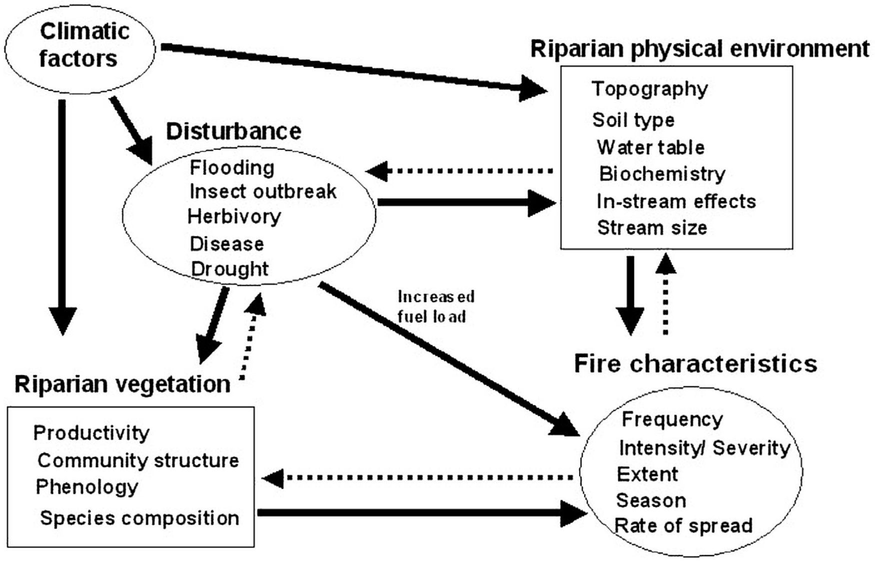


Figure 32—Conceptual representation of fire processes in riparian ecosystems. Climate strongly influences fire characteristics and the physical environment, which then determine fire effects on vegetation. In turn, vegetation affects fire characteristics and the physical environment. Nonfire disturbances can increase fuel loads, which then affect fire characteristics. Figure reprinted from Pettit and Naiman (2007a) with permission from Springer Nature Customer Service Centre GmbH.

A lack of overbank flooding reduces moisture that helps break down ground fuels and can lead to a widespread accumulation of fallen branches that would otherwise be redistributed by flood waters (Ellis 2001; Friggens et al. 2013; Glenn and Nagler 2005). Furthermore, artificial flooding implemented by inundating riparian areas may not serve adequately as a surrogate for overbank flooding with respect to its effects on fuel accumulations (Ellis 2001). Along the Middle Rio Grande, fire severity was lower in areas that received extensive flooding during years with high river flow compared to areas that did not receive regular flooding from the river but were inundated as part of an experiment (Ellis 2001). Areas with artificial flooding had a higher buildup of organic debris that contributed to higher fire severity with widespread mortality of Rio Grande cottonwoods (Ellis 2001).

5.1.3. Climate Change Will Continue to Affect Riparian Ecosystems in Variable and Complex Ways

Combined with other stressors associated with human modification of riparian ecosystems, climate change is expected to have profound and wide-ranging effects in different settings, and impacts are likely to vary throughout the WDNA (Friggens et al. 2013, 2014; Seavey et al. 2009; Smith and Finch 2015). Many components of riparian ecosystems are at risk of negative impacts, including physical processes, hydrology and streamflows, vegetation, fire regimes, and wildlife (Friggens et al. 2013, 2014; Seavy et al. 2009; Smith and Finch 2015).

Anthropogenic greenhouse gas emissions have been causing rapid changes in climate, especially since the mid-20th century (IPCC 2014). In the desert ecoregions of the southwestern United States and northern Mexico, observed maximum temperatures increased 2.7 to 3.1 °F (1.5–1.7 °C) during the 20th century; minimum temperatures have increased even more, 3.6 to 5.4 °F (2.0–3.0 °C) (Garfin et al. 2013). Under a high emissions scenario (CMIP3 A2), temperatures in the Southwest region are expected to rise another 2.2 to 3.9 °F (1.2–2.2 °C) by 2050 and 5.0 to 9.0 °F (2.8–5.0 °C) by 2099 (Garfin et al. 2013). Changes in precipitation are more difficult to predict than temperature, but projections suggest decreasing precipitation in the southern part of the southwestern United States and slight increases or little change in precipitation in northern parts of the southwestern United States (Garfin et al. 2013). Droughts and heat waves are expected to be more frequent, more intense, and of longer duration (Garfin et al. 2013).

Total streamflow has diminished in recent years in major southwestern river basins, and hotter conditions and greater precipitation variability in the future are expected to contribute to continuing decreases in annual discharge (Garfin et al. 2013; Overpeck et al. 2010; Smith and Finch 2016, 2017; Udall and Overpeck 2017; Vano et al. 2014). Riparian ecosystems are vulnerable not only to local changes in climate, but to those that occur upstream in the channel and watershed (Friggens et al. 2013; Smith and Finch 2015; Udall and Overpeck 2017). Earlier snowmelt and reduced snowpack in upper watersheds, owing to drought or climate change, can affect soil moisture and flood timing relative to plant phenology downstream, reducing opportunities for plant germination and seedling survival (Finch et al. 2012; Garfin et al. 2013; Smith and Finch 2016).

Changes in winter precipitation in higher elevations will affect both base flows and floods downstream in lowland riparian ecosystems. For example, warming in montane areas is expected to increase the rate of snowmelt and lead to an increase in severe flooding events along the Middle Rio Grande (Friggens et al. 2013). Research on the Colorado River shows that approximately one-third of recent reduction in annual streamflow is due to unprecedented high temperatures throughout the basin; any future increases in precipitation may be overwhelmed by increasing temperature with respect to their effects on streamflow (Udall and Overpeck 2017).

Lowland riparian flora and fauna are sensitive to changes in streamflow and groundwater conditions (Friggens and Finch 2015; Garfin et al. 2013; Smith and Finch 2015, 2016). Climate change, combined with anthropogenic land and water use paradigms, is reducing riparian habitat quality for many riparian species, including threatened and endangered wildlife species and the native riparian plants they depend on (Friggens and Finch 2015; Friggens et al. 2014; Smith and Finch 2015, 2016).

Climate change is expected to affect riparian vegetation in ways that will increase fire risk. For instance, climate change is expected to help facilitate continued spread of nonnative species (Smith and Finch 2017). However, this may not occur because of an increase in habitat that is suitable climatically; there is already a large area of land that is currently suitable for tamarisk, and that is not expected to change significantly by the year 2100 (Bradley et al. 2009). Instead, the continued spread of nonnative species may have more to do with climate change effects on native species. Climate change could alter competitive relationships between native species, and tamarisk and Russian olive directly and indirectly (Nagler et al. 2011). For example, flood timing could diverge from spring seed releases of cottonwoods and willows and reduce reproductive success, making way for tamarisk and Russian olive to increase (Friggens et al. 2013; Smith and Finch 2017). Prolonged droughts increase fire risk by decreasing water availability and reducing fuel moisture (Pettit and Naiman 2007a).

On the Middle Rio Grande, the interactive effects of drought, altered hydrological regimes, and climate change are shifting vegetation communities toward greater abundance of tamarisk and Russian olive (Smith and Finch 2015, 2017). In some places, climate change in combination with other stressors is expected to reduce or even eliminate the conditions needed by Rio Grande cottonwoods to reproduce within 80 years, and greatly reduce the density of mature trees over the next 100 years (Smith and Finch 2017). Projections along the Middle Rio Grande show that burn probability and fire intensity are expected to increase over time (Friggens et al. 2014). As human populations increase, there will be additional pressures on natural systems (Friggens et al. 2014), including potential increases in anthropogenic ignition rates (Stuever et al. 1995).

Climate variability influences riparian vegetation via various pathways, including evapotranspiration, intensity and timing of storms, and annual precipitation rates (Crimmins 2007; Friggens et al. 2014; Nichols 2007; Smith and Finch 2015, 2016). Oscillatory modes of the climate system, such as the Pacific Decadal Oscillation, El Niño Southern Oscillation, and Atlantic Multidecadal Oscillation, have been shown to influence climate in southwestern North America via teleconnections to terrestrial climate (Woodhouse et al. 2009). One study along the Verde River in Arizona suggests that densities of cottonwood-willow stands and channel morphology were influenced by flood magnitudes that were correlated with low frequency variability in the Pacific Decadal Oscillation (Lopez et al. 2003).

5.2 How Do Fire Effects Vary Among Species, and What Might These Results Mean for Research, Restoration, and Management?

Riparian plant response to fire varies by species, fire intensity, site-specific conditions (Dwire and Kauffman 2003; Ellis 2001; Friggens et al. 2013), and time since last fire (Sunderman 2009). Influential factors include pre-fire and post-fire streamflow regimes (Ellis 2001; Glenn and Nagler 2005), soil characteristics (Busch and Smith 1993; Coffman et al. 2010; Pettit and Naiman 2007a,b; Sunderman 2009), depth to groundwater (Busch and Smith 1993; Smith et al. 2009), and post-fire precipitation characteristics (Smith et al. 2009).

5.2.1. Many Native Species Can Recover Post-fire, but Conditions Often Favor Nonnative Species

Several native species have been observed resprouting after fire-induced top-kill, and a few have also been observed regenerating by seed after a burn (table 7). These results suggest that these species possess adaptations that allow them to recover after a fire. However, native trees may regenerate by seed only where post-fire flooding occurs; where no flooding occurs, recurring fires are likely to eliminate cottonwoods and willows. Frequent and intense fires interact with altered hydrological regimes to contribute to the replacement of cottonwood-willow woodlands with tamarisk and other nonnative plants (fig. 33). These fires may be expediting vegetation shifts toward nonnative species resulting from climate change and management impacts on hydrological regimes in a manner similar to that observed in other systems (Falk 2017). For instance, fires that occur in tamarisk-dominated systems may reinforce the dominance of tamarisk (Drus 2013; Nagler et al. 2011).

Two graminoid species, native southern cattails and nonnative giant reed, are documented to have a positive response to fire (Coffman et al. 2010; Glenn et al. 2013; Mexicano et al. 2013). Two other native riparian

Table 7—Postfire regeneration response of seven native woody species observed in the literature included in this review. "Yes" indicates that the regeneration response is possible for each species listed. The actual regeneration outcome that occurs after a fire, however, depends on site-specific conditions. Questions marks indicate that no evidence was found to support a yes or no conclusion.

Scientific name	Common name	Resprouts?	Resprouts after high severity fire?	Regenerates by seed?
Fraxinus velutina	velvet ash	Yes	?	Yes
Populus deltoides subsp. *wislizenii*	Rio Grande cottonwood	Yes	Yes	Yes
Populus fremontii	Fremont cottonwood	Yes	Yes	?
Prosopis glandulosa	honey mesquite	Yes	?	Yes
Prosopis pubescens	screwbean mesquite	Yes	?	Yes
Prosopis velutina	velvet mesquite	Yes	?	?
Salix gooddingii	Goodding's willow	Yes	Yes	?

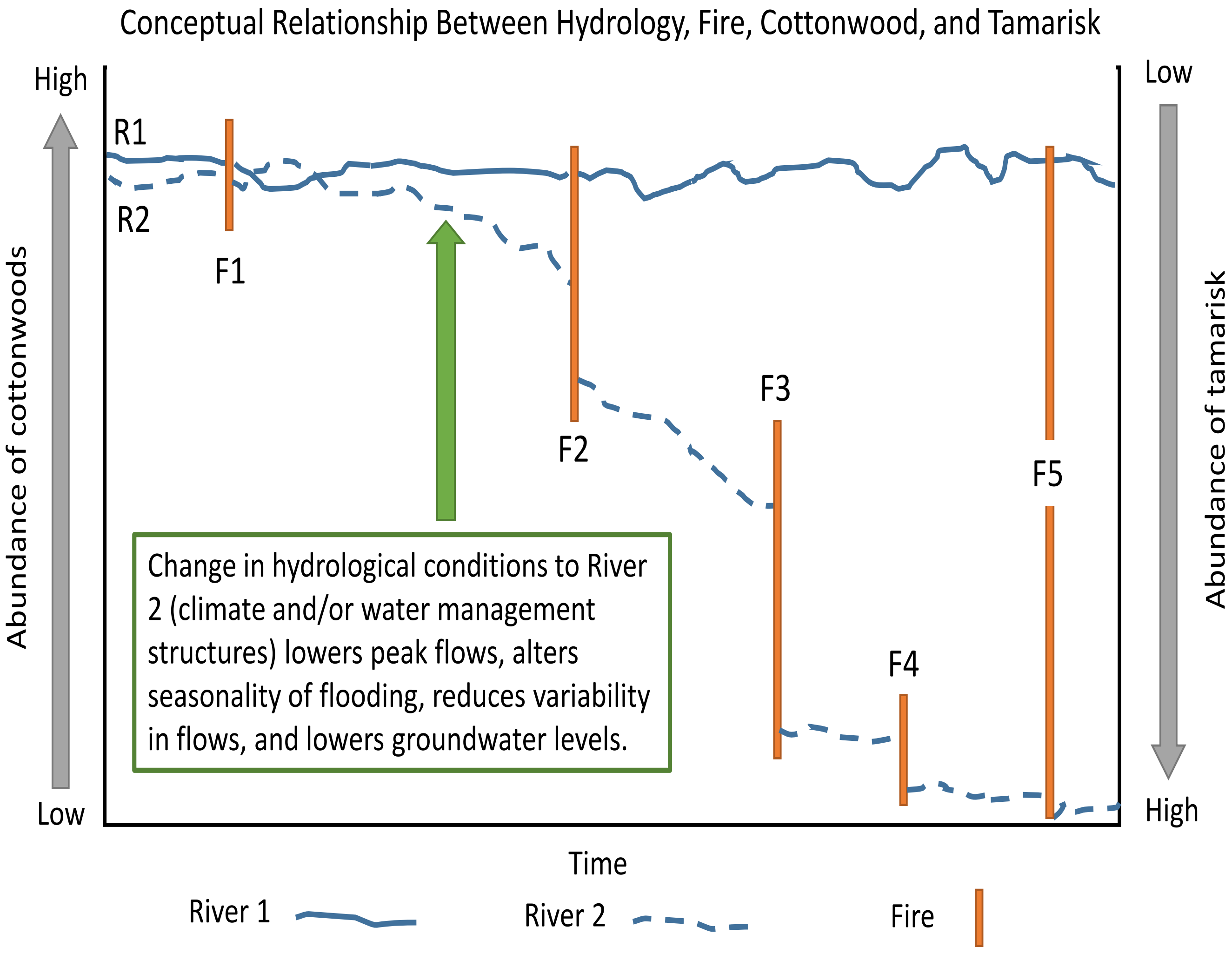


Figure 33—Conceptual diagram depicting the hypothetical relationship between hydrology, fire, cottonwood, and tamarisk for two rivers. At the beginning of the time period, River 1 (R1) and River 2 (R2) have similar hydrological regimes with variable stream flows and seasonal flooding. Fires 1, 2, and 5 collectively represent Fire Regime Group I (≤35-year return interval, low and mixed severity) (LANDFIRE 2008a), which occurs on River 1 without significant effects on the relative abundances of cottonwood and tamarisk because habitat requirements for cottonwood are maintained. River 2 undergoes hydrological alteration between Fires 1 and 2 that reduces peak flows and variability of flows, and lowers groundwater levels. This reduces habitat quality for cottonwoods, which helps facilitate the spread of tamarisk. When Fire 2 occurs under drier conditions on River 2, tamarisk recovers from fire more quickly than cottonwoods, and some cottonwoods do not survive. By the time Fire 3 occurs on River 2, tamarisk cover has reached 50 percent, increasing fire intensity and leading to widespread mortality of cottonwood. There is a rapid, abrupt, and widespread shift toward tamarisk dominance. Tamarisk is adapted to short fire return intervals, and fires become more frequent as tamarisk cover increases (Fire 4), eventually occurring every 10 to 15 years. Cottonwood cannot survive the combination of drier conditions, more severe fires, and more frequent fires, and is eliminated from River 2.

species, California fan palm and arrowweed, benefit from fire (Brooks and Minnich, 2006; Busch and Smith 1993; Glenn et al. 2013; Mexicano et al. 2013; Vogl and McHargue 1966). California fan palm, being heat resistant, suffers relatively little mortality during fires and can use burned areas for germination (Vogl and McHargue 1966).

5.2.2 Increasing Fire Frequency and Severity Are Hastening the Already Rapid Decline of Cottonwoods

Cottonwood habitat and populations have been in decline since at least the early 1960s due to habitat alteration and nonnative plant invasions (Howe and Knopf 1991; Pendleton et al. 2011). Despite their ability to regenerate post-fire, Fremont and Rio Grande cottonwood are declining rapidly due to the combined effects of changing hydrological regimes and increasing fire frequency and severity. Along the Middle Rio Grande, other woody species that are more drought and fire tolerant, such as tamarisk and Russian olive, are expected to replace Rio Grande cottonwoods by the end of the 21st century (Smith and Finch 2017). Cottonwood population models suggest that the impacts of fire and flooding have variable effects on populations, depending on site conditions (fig. 34) (Smith and Finch 2015, 2017).

Effects of higher burn frequency include reduced seedling survival for Fremont cottonwoods (Brooks and Minnich 2006; Nagler et al. 2005), and change in habitat structure and species composition (Brooks and Minnich 2006; Busch 1995). Frequent fires have been implicated in the demise of young Fremont cottonwood trees where flooding provided adequate germination sites (Nagler et al. 2005).

More frequent and severe fires are likely to continue to result in large-scale mortality (including top-killed trees that do not resprout, and those that resprout initially but do not survive) of Fremont and Rio Grande cottonwood trees in their reproductive years (fig. 35). Tamarisk, which thrives in a broad range of conditions, is more successful at resprouting and growing biomass post-fire than native Fremont cottonwoods and Goodding's willows, in part because it uses water more efficiently than these species and is better adapted to saline environments (Busch and Smith 1993). Lowering water tables and streams that have been channelized often result in these kinds of conditions, reducing soil moisture availability and thus changing the environment in favor of tamarisk (Glenn and Nagler 2005; Nagler et al. 2011; Smith and Busch 1992; Stromberg et al. 2003; Webb and Leake 2006).

Where natural flow regimes are more intact and tamarisk is suppressed by native vegetation, native trees may be able to maintain their dominance after fire (Stromberg and Rychener 2010). Factors affecting

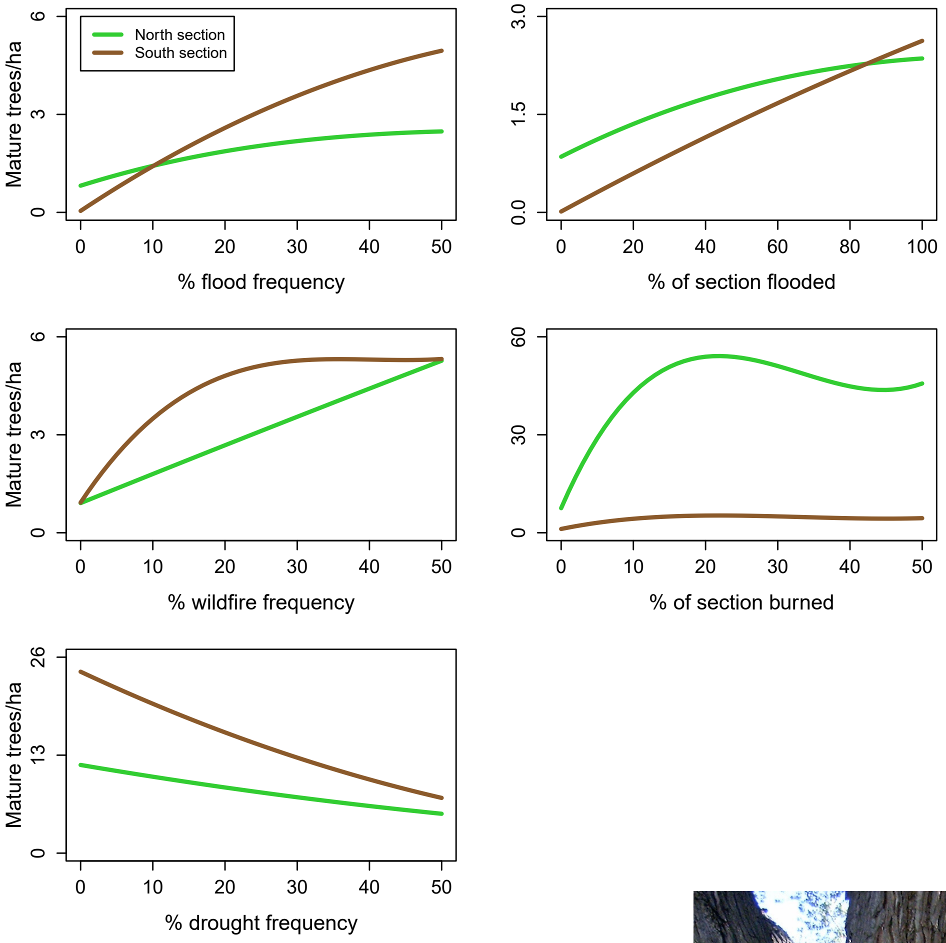


Figure 34—Sensitivity of mature cottonwood tree density to changes in flood frequency, area flooded, wildfire frequency, area burned, and drought frequency at two study sections along the Middle Rio Grande in central New Mexico. Each point represents the mean density of trees at the end of 200-year projections simulated 1,000 times. Figure reproduced from Smith and Finch (2015) with permission.

Figure 35—Very large cottonwoods like this are increasingly rare; San Pedro Riparian National Conservation Area, Cochise County, Arizona, circa 2005 (photo: Chris Hartman, used with permission).

resprouting and resprout survival were not identified sufficiently to determine best practices for supporting cottonwood regeneration post-fire, but given the species' dependence on surface and groundwater regimes, as well as soils and geomorphology, it is likely that these are limiting factors to long-term cottonwood resprout survival and competitive success.

5.2.3 Options for Addressing Undesirable Effects of Fire Through Management and Restoration

Our review suggests that fire regimes, fuel types, and post-fire outcomes are influenced by hydrological regimes. We recommend implementing management actions that promote fires and fire regimes that are not detrimental to biodiversity, water quality, and hydrological processes. These include prescribed fire and other treatments to reduce fuel loads presented by nonnative species, implementing environmental flows, actions that promote native vegetation, and the use of applied conservation genetics to inform preservation and restoration efforts. Best management practices will vary depending on site conditions.

Preserving cottonwoods may require a mix of active interventions in some systems. Ultimately, cottonwood persistence will depend on habitat quality, particularly key hydrological and geomorphological conditions, and maintaining genetic characteristics of remaining populations. Carefully designed fire and fuels management have the potential to reduce the negative impacts of anomalous fire on these species. Mature reproductive cottonwoods may need protection from high severity fire to minimize top-kill that results in loss of reproductive individuals until new stands have a chance to recruit. Ideally, areas where native trees are regenerating (fig. 36) should be protected from fire until juveniles have a chance to develop some fire resistance (Nagler et al. 2005). Observations of Rio Grande cottonwoods regenerating by seed post-fire when hydrological conditions were sufficient (Ellis 2001; Smith and Finch 2015) are further evidence that

Figure 36—Fremont cottonwood saplings line the river at Bill Williams River National Wildlife Refuge; Mohave County, Arizona, summer 2012 (photo: Amanda D. Webb).

prescribed fire could be used in combination with other treatments to serve as a restoration tool.

One key approach for minimizing fire frequency and severity where cottonwood-willow woodlands are a value at risk is to implement environmental flows. According to the Brisbane Declaration (2007), "environmental flows" can be defined as "the quantity, quality, and timing of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods that depend on these ecosystems . . . " Returning flow regimes that mimic the natural range of variability for undammed streams, which should be informed by the life histories of both desirable and undesirable plant species (Merritt and Poff 2010; Mortenson and Weisberg 2010), would have multiple benefits for addressing the root causes of the region's increased risk for high frequency and high severity fires. Benefits would include:

- Increased water availability for native vegetation (Busch and Smith 1995; Nagler et al. 2005)
- Promoting flood scour that clears fuels on the ground and discourages recruitment of herbaceous and shrubby ladder fuels (Levine and Stromberg 2001)
- Promoting floods that reallocate sediments and nutrients in a natural way to support native species (Busch and Smith 1995; Levine and Stromberg 2001)
- Promoting geomorphological stream and floodplain characteristics that support native, diverse riparian vegetation (Naiman et al. 1993)
- Safeguarding and promoting groundwater available to riparian vegetation (Busch and Smith 1995)
- Promoting flood disturbance that creates germination sites for cottonwood and willow (Merritt and Poff 2010; Nagler et al. 2005)

It may also be important to use other means of promoting beneficial fire and reducing risk of elevated fire frequencies and severities that are detrimental to native riparian trees. Fuel treatments, including prescribed fire, may be needed in riparian areas, and possibly also in adjacent uplands, to protect intact cottonwood-willow woodlands and other desirable native riparian communities in the short term. Prescribed fire in combination with other control techniques, such as removal or herbicide use, where practical, is an option in some circumstances and can be applied strategically across watersheds to protect the most vulnerable riparian areas (Sidman et al. 2015). It may even be possible to stimulate germination of cottonwoods and other native trees by using prescribed fire (Smith and Finch 2017).

Recent projections suggest that habitat is available for the tamarisk leaf beetle (*Diorhabda carinulata*) throughout much of the WDNA, and these insects continue to spread (Coulson et al. 2016). Research from the Humboldt River Basin in the Great Basin Desert of Nevada suggests that fire risk increases in the short term post-herbivory as dead leaves persist on branches, but then diminishes over time as tamarisk uses up energy reserves over repeat defoliations, and eventually ceases to put on new leaves (Drus et al. 2012). Prescribed burning at high intensities, especially in the summer, can be effective at killing tamarisk at a time when plant starch reserves are low, making tamarisk beetle herbivory and prescribed fire potentially more effective when used together (Brooks et al. 2008; Drus et al. 2014). Moderate increases in fire rate of spread and maximum temperature were found post-herbivory where fuels were desiccated, but herbivory effects on fire behavior were minimal compared to weather and fuel conditions (Drus et al. 2012). These findings suggest that tamarisk is highly flammable regardless of herbivory or the similar fuel conditions that result from treating tamarisk with herbicide (Drus et al. 2012). Planting native trees post-treatment has the potential to restore habitat for species that could be affected by the post-herbivory die-back of tamarisk (Coulson et al. 2016).

Restoring riparian ecosystems has great potential for helping ecological communities adapt to climate change (Seavy et al. 2009). Although riparian ecosystems are vulnerable to climate change (Capon et al. 2013; Friggens et al. 2013; Garfin et al. 2013; Overpeck et al. 2010; Smith and Finch 2015, 2016; Vano et al. 2014), they also have high adaptive capacity (Capon et al. 2013), making them naturally resilient to impacts from climate change (Seavy et al. 2009). Where riparian areas are intact and functional, they help wildlife by providing thermal refugia and habitat connectivity; they also connect terrestrial and aquatic ecosystems (Seavy et al. 2009). Restoring riparian ecosystems in an era of climate change will require addressing climate impacts specific to local conditions and using local knowledge to help craft adaptation strategies (Seavy et al. 2009).

Restoration success must be measured differently depending on restoration goals (Bateman et al. 2012; Gann et al. 2018). Restoration treatments may have unforeseen secondary effects on the ecosystem and should be evaluated before treatment. Different restoration goals may call for different treatment methods. Restoration design requires an ecosystem perspective that considers multiple taxa (Mortenson and Weisberg 2010). Restoration planning best practices include developing overarching ecological objectives and developing monitoring plans that allow for adapting treatments and management based on observed effects (Bateman et al. 2012; Rieger et al. 2014).

Genetic diversity of cottonwoods and other native riparian species in dwindling habitats could be a concern for climate change adaptation and ecosystem management in lowland riparian ecosystems due to widespread habitat conversion and degradation, increasing distribution and abundance of nonnative plants, and increasing fire-related mortality. Loss of genetic diversity reduces the likelihood that plant genotypes capable of surviving and reproducing in changing environments will exist to pass on traits that otherwise may be selected for, and reduces the likelihood that these species will persist (Richards et al. 2016). Genetic variation is one key to species adaptations to novel climate regimes (Falk and Millar 2016).

Many plant traits related to tolerances of environmental conditions are heritable and differ between populations from areas with different environmental conditions. A common garden experiment revealed that populations of Fremont cottonwood and Goodding's willow from warmer regions had higher annual net primary productivity than populations from cooler regions, when grown under extremely warm conditions (Grady et al. 2011). Other heritable traits include morphological and physiological traits associated with coping with water stress (Rowland et al. 2001), salt tolerance (Rowland et al. 2004), and possibly reproductive phenology (Cushman et al. 2014; Karrenberg et al. 2002). Genetic differentiation among Fremont cottonwood populations from different regions increases as a function of differences in winter and spring precipitation (Cushman et al. 2014). In a common garden experiment, individuals of plains cottonwood from populations of different regions showed variation in fall leaf phenology associated with difference in latitude (Friedman et al. 2012). Insights from these types of studies may help managers anticipate future impacts to these species due to climate change and adapt management strategies accordingly (Stockwell et al. 2016).

Mid-size to large streams (5th-, 6th-, and 7th-order streams) (fig. 37) facilitate gene flow of Fremont cottonwoods while resistance to gene flow increases greatly and nonlinearly in smaller streams. Therefore, river connectivity in lowland riparian ecosystems is particularly important for the persistence of this species, and restoration should include efforts to maintain or restore gene flow (Cushman et al. 2014).

Figure 37—Large streams like the Rio Grande in Big Bend National Park, Brewster County, Texas are especially important for facilitating cottonwood gene flow (photo: Amanda D. Webb).

5.3 What is the Overall Impact of Riparian Fire on Ecological Communities?

Increasing fire frequency and severity are contributing to rapid ecological change along rivers in the southwestern United States. Flooding is typically the primary disturbance process in riparian ecosystems, but in the absence of natural flooding, the influence of fire as a disturbance process is increasing. These two foundational changes in the dynamic nature of riparian ecosystems indicate that ecological communities are reorganizing at many levels. We expect these changes will be reflected in communities as many species are eliminated along with processes that modulate their habitat.

5.3.1 Changing Fire Regimes Threaten Biodiversity

Historically, mesic conditions and the variable patchiness of vegetation in riparian woodlands very likely limited wildfire size and intensity (Dwire and Kauffman 2003; Lambert et al. 2010). Currently, riparian areas are experiencing fires of a different nature: more likely to spread; more intense; and more affected by relatively continuous, uniform fuel loads compared to patchy vegetation (fig. 38). By influencing the spatial distribution of organic material and the composition and structure of vegetation, burn severity affects resource availability for wildlife (Friggens et al. 2014). Fire can reset habitat succession or result in habitat conversion in riparian woodlands; over time, more frequent and severe fires are expected to have negative impacts on mature riparian woodlands, with subsequent consequences for wildlife, including carnivores (Friggens et al. 2013). Contemporary fires often spread into riparian areas from adjacent upland vegetation (Bock and Bock 2014; Coffman et al. 2010), but fires can also be started in the riparian zone.

Figure 38—This trail through a dense tamarisk stand in Bill Williams River National Wildlife Refuge, Mohave County, Arizona, illustrates the dense fuel structure both horizontally across individuals and vertically from the ground up through the canopy (photo: Amanda D. Webb).

Fremont and Rio Grande cottonwood in the WDNA are at increased risk of rapid decline due to reduced habitat quality for reproduction and survival, and the added stressors of increasing nonnative species and increasing fire frequency and severity. Losing cottonwoods would send cascading consequences through riparian biotic communities. Naturally variable flooding regimes create episodic recruitment opportunities for cottonwoods and willows, resulting in a mosaic of cottonwood-willow forest patches of varying age and structure (Stromberg 1993). These patches help support diverse ecological communities (Naiman et al. 1993; Stromberg 1993).

The presence of native woody species increases wildlife diversity and supports unique faunal assemblages (Friggens et al. 2014; Glenn and Nagler 2005; Smith et al. 2006a; Szaro 1980). Pollinators are an important contributor to biological diversity of southwestern riparian ecosystems, and can be adversely affected by displacement of cottonwoods by nonnative plants (Pendleton et al. 2011). On the Middle Rio Grande, proximity to cottonwoods and percentage of cottonwood canopy cover were the most influential variables sustaining higher densities of ground-emerging cicadas (*Tibicen dealbatus*), an important food source for other wildlife (Smith et al. 2006a). The threatened western yellow-billed cuckoo (*Coccyzus americanus*) (fig. 39) is an example of a riparian species in the southwestern United States that uses cottonwoods and willows for foraging and nesting habitat, and cicadas are an important part of its diet (USFWS 2014). Several other species are at risk due to reduced native riparian woodlands, including southwestern willow flycatchers (*Empidonax traillii extimus*), and Chiricahua leopard frogs (*Lithobates chiricahuensis*) (Coulson et al. 2016; USFWS 2002, 2012).

Figure 39—A western yellow-billed cuckoo perches in a Fremont cottonwood tree in Bill Williams River National Wildlife Refuge; Mohave County, Arizona, summer 2012 (photo: Amanda D. Webb).

The impacts of changing water availability and the spread of nonnative plants have reduced cottonwood-willow habitat. Habitat fragmentation reduces genetic diversity in plant populations (Aguilar et al. 2008; Richards et al. 2016). Low genetic diversity reduces populations' ability to adapt to rapidly changing environmental conditions (Huenneke 1991; Richards et al. 2016). Because cottonwood-willow woodlands are rare and fragmented, even a single large-scale event, such as a drought or fire, could reduce population size and degrade genetic diversity (Gitlin et al. 2006, 2009).

Eroding genetic diversity of cottonwoods and willows could have far-reaching ecological consequences for biotic communities and ecosystem processes. Interactions between plants and soil biota result in plant-soil feedbacks that vary with plant genotype and can affect soil chemistry (Fischer et al. 2014). Fischer and colleagues (2014, p. 10) proposed that "the selective environment (positive and negative) imposed by soil communities may be fundamental in determining plant genotype success, which may in-turn affect soil communities …" Cottonwood genetic diversity promotes arthropod community diversity (Ferrier et al. 2012) and may have similar effects on other levels of community diversity, thus exerting influence on ecosystem dynamics (Whitham et al. 2003).

5.3.2 Riparian Fire Effects on Ecological Communities Extend Beyond the Riparian Corridor

Implications of changing fire regimes in riparian ecosystems may extend well beyond the stream corridor (fig. 40). While we typically characterize the uplands as having significant influence on riparian ecosystems downslope and downstream, in the case of fire, this influence may be reciprocal. Riparian ecosystem changes due to altered hydrological regimes and nonnative plants may switch riparian zones from relatively moist, verdant fire breaks into zones that include fine ladder fuels or monotypic stands of fine fuels consisting of dense thickets of tamarisk, giant reed, or other species (Brooks and Minnich 2006; Coffman et al. 2010; Dwire et al. 2003). For example, an area dominated by giant reed carried fire through a broad riparian zone to the other side of the river, where the fire continued to consume thousands of acres of shrubland on the other side (Coffman et al. 2010). In addition, the invasion of tamarisk in these ecosystems increases both fire frequency and the likelihood that fire would be carried across the riparian zone (Drus 2013).

Fire frequency is increasing in the Sonoran and Mojave Deserts, largely due to the spread of nonnative grasses (Brooks and Chambers 2011). As nonnative plants, water use, and land use promote fire spread and increase fire frequency in riparian zones, and nonnative grasses

Figure 40—Lowland riparian ecosystems are affected by watershed conditions upslope. Changing fire regimes in riparian ecosystems could have impacts on adjacent uplands; San Pedro Riparian National Conservation Area, Cochise County, Arizona, fall 2003 (photo: Amanda D. Webb, Bureau of Land Management).

increase fire frequency across the desert, it is likely that large areas of bottomlands are already developing different fire regimes as well.

5.4 What Knowledge Gaps Exist in the Current Literature, and What Further Research Is Needed to Improve Understanding of Riparian Fire Effects and Management?

In this section, we outline knowledge gaps based on the results of our systematic literature search and review. Some sources of information may exist that our methods did not detect, or that we were unable to find through conversations with people working in riparian ecosystems in the WDNA. Nonetheless, our review indicates numerous knowledge gaps that, if addressed, would be key to taking informed action to improve fire management in riparian ecosystems, and support continued research. Development of new research should include input from fire managers, riparian ecosystem managers, and others faced with similar resource management challenges and goals related to riparian fire and ecosystem resilience. Translational ecology provides a framework for this knowledge codevelopment and exchange that ensures research results can be used to make management decisions (Enquist et al. 2017; Meadow et al. 2015; Wall et al. 2017).

5.4.1 Fire Ecology of Riparian Trees and Fire Effects

Among the most pressing needs is an understanding of how soil and hydrological conditions (surface and groundwater) affect post-fire recovery of vegetation, both native and nonnative species. Research is needed in this area to inform the conservation, restoration, and rehabilitation of native plant communities in the context of fire risk. Many native woody species resprout after fire-induced top-kill. These results suggest that fire *per se* does not act alone in influencing the failure of these species to recolonize burned areas. Rather, post-fire conditions in the context of pre-fire habitat quality, hydrology, and streamflow variability are likely to be the primary factors influencing species response, with fire intensity also exerting a strong influence.

More information on multiple interacting factors affecting native plant species recovery after fire is needed, including groundwater, surface flows, soils, geomorphology, and fire severity (fig. 41). In particular, more research

Figure 41—Goodding's willows resprouted in spring after the Three Slashes Fire, which burned in August and September 2011; Cibola National Wildlife Refuge, California and Arizona (photo: Matthew Grabau, used with permission).

is needed to better understand environmental factors and management actions that influence resprout survivability, reproduction, and recruitment over multiple years post-fire (Ellis 2001; Smith and Finch 2017; Smith et al. 2009; Stuever 1997).

5.4.2 Prescribed Fire and Post-fire Rehabilitation

> *"Saltcedar [tamarisk] reduction may yield an interaction of both positive and negative impacts resulting from biological, fire, or herbicide application, requiring site-specific evaluation for restoration potential… Different methods of achieving desirable growth medium conditions need testing through varied techniques of seedbed preparation to enhance microenvironmental conditions in the root zone of planted species, including saltcedar leaf litter dispersal or incorporation, improved contact of seeds with mineral soil, salinity reduction in surface soil layers, mycorrhizal fungi inoculation, and manipulation of soil nitrogen dynamics."*
>
> *—Lair (2006, p. 12)*

Limited information is available on prescribed fire and its utility as a restoration tool, and fire effects in areas of fuel reduction treatments. Basic but comprehensive information on prescribed fire effects in riparian zones is essential to design effective treatments. Fuel treatments, including prescribed fire, may be needed to protect intact cottonwood-willow woodlands and other desirable native riparian communities. Fire may be a tool for increasing structural diversity of vegetation to support wildlife diversity, or to remove litter and downed timber in areas where natural flooding does not occur to redistribute these materials and promote decomposition. There is little to no information to help inform management thresholds when management goals may be achieved through burning. We also found no information about fire management in the wildland-urban interface in desert riparian ecosystems.

Very little information is available about post-fire rehabilitation, which is needed to better inform restoration and efforts to mitigate the negative impacts of increasing fire frequency and severity. Existing restoration literature that is not related to fire may have useful insights for dealing with conditions that can result from both fire and nonfire environmental factors, such as increased salinity and soil moisture limitations (e.g., Bunting et al. 2011).

Figure 42—This wildfire produced tall flames and a churning column of smoke along the Middle Rio Grande near Bosque del Apache National Wildlife Refuge; Socorro County, New Mexico, September 2017 (photo: Donald A. Falk).

5.4.3 Fire Behavior

Very little information is available about fire behavior in lowland riparian ecosystems. Existing fire behavior information focuses on tamarisk (Racher 2003; Racher and Mitchell1999), with one study comparing tamarisk with cottonwoods and willows (Drus 2013). In preparation for this report, we had difficulty finding photos of fires burning in riparian ecosystems (fig. 42).

The development of fuel and fire behavior models for tamarisk stands would be useful to fire managers. More could be done to increase access to information on fire behavior through empirical observations and fire behavior models. Though fire managers may observe fire behavior during incidents, these observations are rarely recorded and made available to the larger riparian management community. More extensive use of Fire Behavior Monitors during riparian burning could add to the body of knowledge on fire behavior. Furthermore, landscape fire behavior models, such as FARSITE (Finney and Andrews 1999), could be refined for modeling the unique fuels and setting of riparian areas.

5.4.4 Fire and the Physical Environment

The literature in our review suggests complex links between hydrological regimes, channel and floodplain dynamics, vegetation structure and composition, and riparian fire regimes. Each of these components of the system, and their interactions, is affected by management and climate. Research is needed that explores these links and that can be used to inform mitigation, climate change adaptation, and water and land management. For example, there is a need for science that can improve understanding of changing fire regimes in the context of climate warming, and project how they will affect vegetation, wildlife, and ecosystem services over the next 20 to 100 years.

There is an overall lack of information on fire effects on hydrological regimes, stream geomorphology, soil hydrological properties, and water quality. Studies and information from higher elevations, or other parts of the world, could potentially apply in lowland riparian ecosystems, but a review of that literature is outside the purview of this report.

5.4.5 Information About Plant Species Not Found in the Literature

We did not detect literature sources for several species for which we searched that are known to occur in these ecosystems. We found little or no research on fire behavior and effects on important riparian tree species, including Arizona sycamore, Arizona walnut, boxelder, narrowleaf cottonwood, netleaf hackberry, and desert willow. Plant species that are important to pollinators were also missing, such as mule-fat and willow baccharis.

The most studied native species is Rio Grande cottonwood, with *Populus* being the most studied native genus. Although some information is available on mesquites where they occur in riparian ecosystems, it is a relatively small body of work given the ecological importance of mesquite bosques.

The most studied of all the genera for which we found information is *Tamarix*. All other nonnative plants were much less represented in the literature. To improve understanding about the influence of nonnative plants on fire regimes and fire effects on nonnative plants, more information is needed on other species, such as Russian olive, giant reed, and common reed.

5.4.6 Riparian Fire in Mexico

Although the geographic scope of this review was designed to include literature from Mexico, only two sources were found with project boundaries in Mexico. Both were from the Cienega de Santa Clara (Glenn et al. 2013; Mexicano et al. 2013) near the U.S. border. This lack of documentation does not necessarily indicate a lack of management effort or investigations into the role of fire in Mexico's lowland riparian areas, but only limited access to this literature. Our standard literature search may not have captured the state of the knowledge of fire effects in riparian ecosystems in Mexico accurately, or it may be that few observations are being documented.

Figure 43—The Rio Grande in Big Bend National Park in Brewster County, Texas delineates land in Mexico (at left) and the United States (at right). The yellowish grass growing along the river is common reed (photo: Amanda D. Webb).

Differing land and water use paradigms between the United States and Mexico can provide valuable contrasting information about the effects of different cultural practices on natural resources (fig. 43). To help fill this information gap, we encourage collaborative research and climate adaptation planning across the United States-Mexico border. Bilingual translation of studies from the United States is likely to help Mexican colleagues, and vice versa.

5.4.7 Accessible and Comprehensive Source of Spatially Explicit Data on Fires in Riparian Ecosystems

We are unaware of the existence of any publicly accessible database containing spatially explicit data for the full range of fires that have significant impacts on riparian ecosystems. For many researchers who study fire, Monitoring Trends in Burn Severity (MTBS) is an invaluable database for accessing maps of fire extent and burn severity for large fires that have occurred anywhere in the United States since 1984 (Eidenshink et al. 2007). In the western United States, all fires that are at least 1,000 acres (405 hectares) in size are included in the database.

The size threshold in MTBS excludes many riparian fires. Riparian ecosystems are small features of the landscape compared to most other ecosystems in the WDNA, making small events relatively significant. Riparian fire ecology is understudied, and conditions within riparian ecosystems vary greatly over short time periods and distances, so the MTBS size threshold excludes information that may be important locally for informing management, or in aggregate for determining regional trends and patterns in riparian fire severity and effects.

In this review, there are several studies on fires much smaller than the MTBS size threshold (e.g., Bendix 1994; Busch 1995; Smith and Finch 2015; Stromberg et al. 2009), suggesting that information from smaller events is considered by the research community to be valuable for understanding riparian fire ecology. The lack of free and accessible spatial data for smaller fires may help explain why riparian ecosystems are underrepresented in the fire literature, and why there is a dearth of spatially explicit studies examining fire effects in riparian ecosystems.

5.4.8 Solutions to Issues Arising from Differing Methods and Approaches to Fire Research, Management, and Writing

A formal meta-analysis of studies across the WDNA would be valuable, with a focus on fire effects on cottonwoods and other keystone native species. At present, however, many of the data in existing studies are not comparable. Researchers used different methods for describing fire severity, mortality, and regeneration, making it difficult to pool these data and analyze them to determine fire effects patterns across a geographic area, or within a species, beyond a single river system.

Managers, practitioners, and researchers could benefit from agreeing on standard methods to use in fire research and monitoring in riparian ecosystems. Doing so could expand understanding of species' response to fire and fire properties (behavior, frequency, severity) in riparian ecosystems across the region. This may be particularly useful for landscape-scale or regional conservation and planning efforts where standardized metrics can facilitate tracking of progress toward conservation goals over time. Existing fire monitoring resources include the Fire Monitoring Handbook (NPS 2003), the Fire Effects Monitoring Guide (USFWS 2005), and FIREMON: Fire Effects Monitoring and Inventory System (Lutes et al. 2006).

Another practice that could help improve understanding of fire in riparian ecosystems would be for authors to provide more specific information about the fires they studied. For example, several sources reviewed for this report discussed fire, but did not state explicitly whether they meant wildfire, prescribed fire, or both. Authors commonly discuss the effects of specific fires, but sometimes do not provide the official incident names of the fires. Knowing the fire name would allow readers to look up more specifics about the fire, such as where it started, fire behavior, and the weather during that time. Fire effects are also not reported consistently, complicating evaluation of post-fire recovery. Explicit reporting of fire type, behavior, first-order fire effects, and incident name and date could help improve analysis and understanding of fire. Similarly, the use of standard descriptors for fire behavior would unify understanding of fire effects.

5.4.9 Other Management and Research Questions

How do riparian ecosystems affect fire spread within watersheds, and can they be managed as natural fire breaks for the benefit of both ecological communities and infrastructure? Where have people managed riparian areas as fire breaks for promoting mixed severity burn mosaics? Where possible, managing riparian ecosystems for this benefit could be a climate change and wildfire regime change adaptation strategy that could be useful in landscape-scale or watershed planning efforts. Such a strategy could potentially reduce fire size and spread in the case of large, severe, or otherwise undesirable fires. Riparian areas could be managed to promote patchiness of the nonriparian burn area toward mixed severity mosaics. This could help conserve local genetic resources in existing biota on the other side of a river and provide opportunities for repopulation, or possibly reintroduction. In some areas, it may be desirable to manage riparian ecosystems to facilitate fire spread, such as in a wetland-type ecosystem dominated by native herbaceous vegetation, especially if occurring in a grassland or other fire-dependent ecosystem. Either way, such efforts could be beneficial to upland fire management, water supply for humans and ecosystems, and work toward conservation goals in riparian ecosystems and watersheds.

Fire effects have been studied extensively at the watershed scale, but little information is available on the impacts specific to riparian burning. How do impacts from erosion, sediment transport, deposition, sediment yield after fire of different severities, and the subsequent effects on water quality, aquatic species, and riparian species, differ based on fire size and relative extent in riparian and upland areas? How do these impacts differ specifically between fires that burn mainly riparian vegetation versus fires that burn a significant portion of the uplands? What methods of restoration, and what suites of species, can be used to promote riparian ecosystems that are resilient to fire?

5.5 Conclusions

A holistic approach to riparian ecosystem management is essential to address the underlying causes of vegetation change that result in fires that threaten native ecological communities. Whether management goals are to safeguard high quality riparian areas, improve degraded systems, or prevent fire spread through these sensitive areas, the underlying principle explored here applies: Lowland riparian areas are conditioned by the interactions of hydrology, species composition, and fire disturbance. Lowland riparian areas do not exist in isolation, but are connected strongly to adjacent uplands and larger watershed dynamics.

Fire ecology and fire management should be included in discussions about ecosystem resilience and desired future conditions. Existing riparian fire literature suggests that while nonnative plants can alter fire behavior and regimes, hydrological conditions that promote high-risk fuel loading is a major underlying cause of more frequent and intense fire in lowland riparian ecosystems. More information is needed on the fire ecology of riparian trees, prescribed fire, and post-fire rehabilitation. There is also a need for fuel and fire behavior models for the unique setting in riparian ecosystems, and specifically models for tamarisk stands. To help improve our understanding of riparian fire ecology and management, we encourage interdisciplinary research that explores fire effects in the context of direct drivers of riparian ecosystem condition including hydrological regimes, channel and floodplain geomorphology, abiotic processes, riparian plant succession, vegetation and fuel types, climate, and watershed condition. Ideally, this research should be conceived and carried out in a way that is meaningful across systems and across the WDNA. Including input from managers of riparian ecosystems at all stages of the research process is critical for ensuring that study results can be applied to decisionmaking and on-the-ground management actions.

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Appendix A - Names of Plants Included in This Report

Table A.1— Table of plant species names included in this report. Source: USDA PLANTS Database, https://plants.usda.gov.

Common name	Scientific name
Arizona sycamore	*Platanus wrightii* S. Watson
Arizona walnut	*Juglans major* (Torr.) A. Heller
arroyo willow	*Salix lasiolepis* Benth.
arrowweed	*Pluchea sericea* (Nutt.) Coville
birch	*Betula* spp. L.
black cottonwood	*Populus balsamifera* L. subsp. *trichocarpa* (Torr. & A.Gray ex Hook.) Brayshaw
blackbrush	*Coleogyne ramosissima* Torr.
boxelder	*Acer negundo* L.
bulrush	*Scirpus* spp. L.
California fan palm	*Washingtonia filifera* (Linden ex André) H. Wendl.
California sycamore	*Platanus racemosa* Nutt.
canarygrass	*Phalaris* spp. L.
catclaw acacia	*Acacia greggii* (A. Gray)
cattail	*Typha* spp. L.
chokecherry	*Prunus virginiana* L.
coast live oak	*Quercus agrifolia* Née
common reed	*Phragmites australis* (Cav.) Trin. ex Steud.
cottonwood spp.	*Populus* spp. L.
coyote willow	*Salix exigua* Nutt.
creosote bush	*Larrea* spp. Cav.
desert almond	*Prunus fasciculata* (Torr.) A. Gray
desert willow	*Chilopsis linearis* (Cav.) Sweet
desertbroom	*Baccharis sarothroides* A. Gray
dropseed	*Sporobolus* spp. R. Br.
fan palm	*Chamaerops* spp. L.
fir spp.	*Abies* spp. Mill.
Fremont cottonwood	*Populus fremontii* S. Wats.
giant reed	*Arundo donax* L.
Goodding's willow	*Salix gooddingii* C.R. Ball
gray alder	*Alnus incana* (L.) Moench
greasewood	*Sarcobatus vermiculatus* (Hook.) Torr.
honey mesquite	*Prosopis glandulosa* Torr.
Johnsongrass	*Sorghum halepense* (L.) Pers.
juniper spp.	*Juniperus* spp. L.
knotweed	*Polygonum* spp. L.
littleleaf sumac	*Rhus microphylla* Engelm. ex A. Gray

Appendix A continued.

Common name	**Scientific name**
mesquite spp.	*Prosopis* spp. L.
mule-fat	*Baccharis salicifolia* (Ruiz & Pav.) Pers.
narrowleaf cottonwood	*Populus angustifolia* James
netleaf hackberry	*Celtis laevigata* Willd. var. *reticulata* (Torr.) L.D. Benson
paloverde	*Parkinsonia* spp. L.
panicgrass	*Panicum* spp. L.
plains cottonwood	*Populus deltoides* Bartram ex Marshall subsp. *monilifera* (Aiton) Eckenwalder
quaking aspen	*Populus tremuloides* Michx.
red willow	*Salix laevigata* Bebb
redosier dogwood	*Cornus sericea* L.
Rio Grande cottonwood	*Populus deltoides* W. Bartram ex Marshall subsp. *wislizeni* (S. Watson) Eckenwalder
Rocky Mountain Douglas-fir	*Pseudotsuga menziesii* (Mirb.) Franco var. glauca (Beissn.) Franco
Rocky Mountain juniper	*Juniperus scopulorum* Sarg.
rubber rabbitbrush	*Ericameria nauseosa* (Pall. ex Pursh) G.L. Nesom & Baird
rush	*Juncus* spp. L.
Russian olive	*Elaeagnus angustifolia* L.
sagebrush	*Artemisia* spp. L.
saltbush	*Atriplex* spp. L.
saltgrass	*Distichilis* spp. Raf.
screwbean mesquite	*Prosopis pubescens* Benth.
sedge	*Carex* spp. L.
Siberian elm	*Ulmus pumilla* L.
silver sagebrush	*Artemisia cana* Pursh
skunkbush sumac	*Rhus trilobata* Nutt.
southern cattail	*Typha domingensis* Pers.
spruce spp.	*Picea* spp. A. Dietr.
stretchberry	*Forestiera pubescens* Nutt.
tamarisk (saltcedar)	*Tamarix* spp. L.
velvet ash	*Fraxinus velutina* Torr.
velvet mesquite	*Prosopis velutina* Wooton
white alder	*Alnus rhombifolia* Nutt.
white fir	*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.
white mulberry	*Morus alba* L.
willow spp.	*Salix* spp.
willow baccharis	*Baccharis salicina* Torr. & A. Gray

Appendix B - Literature Sources Listed by Area of Interest

The following information was compiled for each paper related to fire effects along with author, title, publication date, and source.

Literature source type (can be more than one)
A = meta-analysis
C = case study
E = experiment
F = field report
M = model
N = nontechnical report
O = observational study
R = review
S = remote sensing
T = technical report

Study subject (can be more than one)
A = fire and the abiotic environment
B = fire and both native and nonnative plants
N = fire and native plants
O = fire and nonnative plants
R = postfire rehabilitation
S = vegetation structure only (no discrimination between native and nonnative species or between species)

Fire type
B = both wildfire and prescribed fire
N = not specified
P = prescribed fire only
W = wildfire only

Appendix B continued.

Area of interest	Title	Citation	Source	Subject	Fire
Appleton-Whittel Research Ranch, southeastern Arizona	Effects of wildfire on riparian trees in southeastern Arizona	Bock and Bock 2014	O	N	W
Ash Meadows, NV	Fire patterns and post-fire vegetation response in a Mojave Desert spring ecosystem	Sunderman 2009	O	A, B	W
Ash Meadows, NV	Remote sensing approaches for reconstructing fire perimeters and burn severity mosaics in desert spring ecosystems	Sunderman and Weisberg 2011	O, S	B	W
Ash Meadows, NV	Predictive modelling of burn probability and burn severity in a desert spring ecosystem	Sunderman and Weisberg 2012	O, S	A, B	W
Australia; Mediterranean Basin; South Africa	Wildfire as a hydrological and geomorphological agent	Shakesby and Doerr 2006	R	A	W
California	Southeastern deserts bioregion	Brooks and Minnich 2006	R	B	N
California	The use of prescribed fire and mechanical removal as means of control of tamarisk trees	Jorgenson 1996	T	O	P
California	Tamarisk control on public lands in the desert of southern California: Two case studies	Lovich et al. 1994	C, T	O	P
Cienega de Santa Clara, Mexico	Evapotranspiration and water balance of an anthropogenic coastal desert wetland: Responses to fire, inflows and salinities	Glenn et al. 2013	O, S	N	W
Cienega de Santa Clara, Mexico	Vegetation dymanics in response to water inflow rates and fire in a brackish *Typha domingensis* Pers. marsh in the delta of the Colorado River, Mexico	Mexicano et al. 2013	O, S	N	W
Colorado River Delta	Regeneration of native trees in the presence of invasive salt-cedar in the Colorado River delta, Mexico	Nagler et al. 2005	O, S	B	W
General	*Sorghum halepense*	Howard 2004	R	O	B
General	Fire in saltcedar ecosystems	Racher and Britton 2003	R	O	B
General	*Salix gooddingii*	Reed 1993	R	N	B
General	Saltcedar and Russian olive control demonstration act science assessment	Shafroth et al. 2010	R, T	O, R	B
General	Saltcedar impacts on salinity, water, fire frequency, and flooding	Weisenborn 1996	R	O	W
General	*Elaeagnus angustifolia*	Zouhar 2005	R	O	B
General	*Tamarix* spp.	Zouhar 2003	R	O	B

Appendix B continued.

Area of interest	Title	Citation	Source	Subject	Fire
Laboratory (seeds from a commercial vendor)	Potential fire effects on seed germination of four herbaceous species	Mitchell and Dabbert 2000	E	O	P
Laboratory (seeds from near Blythe, CA)	Fire as a tool for controlling *Tamarix* spp. seedlings	Ohrtman et al. 2012	E	O	P
Lake Mead National Recreation Area; Lower Humboldt River Basin, NV	The effect of leaf beetle herbivory on the fire behaviour of tamarisk (*Tamarix ramosissima* Lebed.)	Drus et al. 2012	E	O	P
Lower Colorado River	Effects of fire on water and salinity relations of riparian woody taxa	Busch and Smith 1993	O	A, B	N
Lower Colorado River	Fire in a riparian shrub community: Postburn water relations in the *Tamarix–Salix* association along the lower Colorado River	Smith and Busch 1992	O	B	N
Lower Colorado River; Bill Williams River, AZ	Effects of fire on southwestern riparian plant community structure	Busch 1995	O	B	W
Lower Colorado River; Bill Williams River, AZ	Mechanisms associated with decline of woody species in riparian ecosystems of the southwestern US	Busch and Smith 1995	O	A, B	N
Lower Colorado River; Rio Grande	Revegetation strategies and technologies for restoration of aridic saltcedar (*Tamarix* spp.) infestation sites	Lair 2006	T	R	B
Madrean Province	Effects of fire on riparian systems	DeBano and Neary 1996	R	A	B
Matador WMA Lake Meredith, TX; Red Bluff Ranch near Roswell, NM	Management of saltcedar in eastern New Mexico and Texas	Racher and Mitchell 1999	F	O	P
Middle Rio Grande, NM	Riparian forest restoration: Conflicting goals, tradeoffs, and measures of success	Bateman et al. 2012	E	B	P
Middle Rio Grande, NM	Controlling Russian olives within cottonwood gallery forests along the Middle Rio Grande floodplain (New Mexico)	Caplan 2002	F, R	O	W
Middle Rio Grande, NM	Short-term response of woody plants to fire in a Rio Grande riparian forest, Central New Mexico, USA	Ellis 2001	E, O	B	W
Middle Rio Grande, NM	Middle Rio Grande Basin research report 2008	Finch and Dold 2008	N	O	P
Middle Rio Grande, NM	Vulnerability of species to climate change in the Southwest: Terrestrial species of the Middle Rio Grande	Friggens et al. 2013	R	B	B
Middle Rio Grande, NM	Vulnerability of riparian obligate species to the interactive effect of fire, climate and hydrological change	Friggens et al. 2014	M, R	A, B	B

Appendix B continued.

Area of interest	Title	Citation	Source	Subject	Fire
Middle Rio Grande; NM	Relationships between hydrology, exotic plants, and fuel loads in the Middle Rio Grande	Jemison 2003	E	A, B	B
Middle Rio Grande; NM	The effects of wildfire on native tree species in the Middle Rio Grande bosques of New Mexico	Johnson and Merritt 2009	E, O	B	W
Middle Rio Grande; NM	Riparian habitat change along the Isleta-Belen reach of the Rio Grande	Mount et al. 1996	O	B	N
Middle Rio Grande; NM	Modeling woody plant regeneration and woody debris accumulation under future streamflow and wildfire scenarios in semi-arid riparian ecosystems	Smith and Finch 2015	O, M	A, B	W
Middle Rio Grande; NM	Climate change and wildfire effects in aridland riparian ecosystems: An examination of current and future conditions	Smith and Finch 2017	O, M	A, B	W
Middle Rio Grande; NM	Wildfire, exotic vegetation, and breeding bird habitat in the Rio Grande bosque	Smith et al. 2006	O	B	W
Middle Rio Grande; NM	Post-fire recovery of riparian vegetation during a period of water scarcity in the southwestern USA	Smith et al. 2009	O	B	W
Middle Rio Grande; NM	Fire-induced mortality of Rio Grande cottonwood	Stuever 1997	O, M	N	W
Middle Rio Grande; NM	Initial assessment of the role of fire in the Middle Rio Grande bosque	Stuever et al. 1995	O	B	W
New Mexico	Strategy for long-term management of exotic trees in riparian areas for New Mexico's five river systems, 2005–2014	Parker et al. 2005	T	O	B
Pecos River, NM	Saltcedar: Is burning an option?	Delwiche 2009	E, T	O	P
Pecos River, NM	Prescribed burning prescriptions for saltcedar in New Mexico	Racher et al. 2001	F	O	P
Pecos River, NM; Arroyo del Macho, NM	Prescription development for burning two volatile fuel types	Racher 2003	E	O	P
Piru and Sespe watersheds, CA	Among-site variation in riparian vegetation of the southern California Transverse Ranges	Bendix 1994	O	N	N
Protrero John Creek, Piedra Blanca Creek, CA	Fire, floods and woody debris: Interactions between biotic and geomorphic processes	Bendix and Cowell 2010a	O	A	W
Protrero John Creek, Piedra Blanca Creek, CA	Impacts of wildfire on composition and structure of riparian forests in southern California	Bendix and Cowell 2010b	O	N	W

Appendix B continued.

Area of interest	Title	Citation	Source	Subject	Fire
Protrero John Creek, Piedra Blanca Creek, CA	Disturbance and riparian tree establishment in the Sespe Wilderness, California, USA	Bendix and Cowell 2013	O	N	W
Sabie River, South Africa	Postfire response of flood-regenerating riparian vegetation in a semi-arid landscape	Pettit and Naiman 2007	O	A, B	W
Sabino Creek, AZ	Flood processes in semi-arid streams: Sediment transport, flood routing, and groundwater-surface water interactions	Desilets 2007	O	A	W
San Andreas Fault region, CA	Vegetation of California fan palm oases on the San Andreas Fault	Vogl and McHargue 1966	R	N	N
San Pedro River, AZ	Effects of fire on riparian forests along a free-flowing dryland river	Stromberg and Rychener 2010	O	B	W
San Pedro River, AZ	Return of fire to a free-flowing desert river: Effects on vegetation	Stromberg et al. 2009	O	S	W
San Pedro River, AZ	A century of riparian forest expansion following extreme disturbance: Spatio-temporal change in *Populus*/*Salix*/*Tamarix* forests along the Upper San Pedro River, Arizona, USA	Stromberg et al. 2010	O	B	N
San Pedro River, AZ; Santa Cruz River, AZ	Ground-water surface-water interactions and long-term change in riverine riparian vegetation in the southwestern United States	Webb and Leake 2006	C	B	W
Santa Clara River, CA	Wildfire promotes dominance of invasive giant reed (*Arundo donax*) in riparian ecosystems	Coffman et al. 2010	O	B	W
Southwestern United States	Success of active revegetation after *Tamarix* removal in riparian ecosystems of the southwestern United States: A quantitative assessment of past restoration projects	Bay and Sher 2008	O	O	P
Southwestern United States	Invasion, ecology and management of *Elaeagnus angustifolia* (Russian olive) in the southwestern United States	Brock 1998	R	O	P
Southwestern United States	Impact, biology, and ecology of saltcedar (*Tamarix* spp.) in the southwestern United States	Di Tomaso 1998	R	A, B	B
Southwestern United States	Tamarisk (*Tamarix* spp.) and desert riparian ecosystem change	Drus 2013	E, O	B	B
Southwestern United States	The development and perpetuation of the permanent tamarisk type in the phreatophyte zone of the Southwest	Horton 1977	R	O	N

Appendix B continued.

Area of interest	Title	Citation	Source	Subject	Fire
Western United States	Comparative ecophysiology of *Tamarix ramosissima* and native trees in western US riparian zones	Glenn and Nagler 2005	R	B	W
Western United States	Synthesis of sediment yields after wildland fire in different rainfall regimes in the western United States	Moody and Martin 2009	A, R	A	W
Western United States	Distribution and abundance of saltcedar and Russian olive in the western United States	Nagler et al. 2011	R	B	W

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Appendix C—Science Assessment of Fire in Lowland Riparian Ecosystems

The following information was compiled for each paper related to fire effects along with author, title, publication date, and source.

Study subject (can be more than one)
A = fire and the abiotic environment
B = fire and both native and nonnative plants
N = fire and native plants
O = fire and nonnative plants
R = postfire rehabilitation
S = vegetation structure only (no discrimination between native and nonnative or between species)

Fire type
B = both wildfire and prescribed fire
N = not specified
P = prescribed fire only
W = wildfire only

Appendix C continued.

Table C.1—Assessment of fire properties, effects, and hydrological influences represented in literature on fire effects in lowland riparian ecosystems of the Warm Deserts of North America; includes literature categorized as an experiment or observational study in this review.

Citation	Subject	Fire	Behavior	Frequency	Severity	Fire size	Occurrence	Mortality	Regeneration	Hydrology
Bateman et al. 2012	B, R	P					X			
Bay and Sher 2008	O	P					X		X	
Bendix 1994	N	N					X			
Bendix and Cowell 2010a	A	W					X			X
Bendix and Cowell 2010b	N	W					X	X	X	X
Bendix and Cowell 2013	N	W					X		X	X
Bock and Bock 2014	N	W		X				X	X	
Busch 1995	B	W		X		X	X		X	
Busch and Smith 1993	A, B	N							X	X
Busch and Smith 1995	A, B	N					X			X
Coffman et al. 2010	B	W					X		X	
Delwiche 2009	O	P	X					X		
Desilets 2007	A	W					X			X
Drus 2013	B	B	X	X	X	X		X	X	
Drus et al. 2012	O	P	X							
Ellis 2001	B	W			X		X	X	X	X
Friggens et al. 2014	A, B	B	X				X			
Glenn et al. 2013	N	W					X			X
Jemison 2003	B	P					X			X
Johnson and Merritt 2009	B	W						X	X	
Mexicano et al. 2013	N	W		X			X		X	X
Mitchell and Dabbert 2000	O	P	X						X	
Moody and Martin 2009	A	W								X

Appendix C continued.

Citation	Subject	Fire	Behavior	Frequency	Severity	Fire size	Occurrence	Mortality	Regeneration	Hydrology
Mount et al. 1996	B	N					X		X	
Nagler et al. 2005	B	W						X	X	X
Ohrtman et al. 2012	O	P	X					X		
Pettit and Naiman 2007	A, B	W							X	X
Racher 2003	O	P	X							
Smith and Busch 1992	B	N							X	X
Smith and Finch 2015	A, B	W		X	X			X	X	X
Smith and Finch 2017	A, B	W			X			X	X	X
Smith et al. 2006	B	W					X	X	X	
Smith et al. 2009	B	W					X	X	X	X
Stromberg and Rychener 2010	B	W						X	X	X
Stromberg et al. 2009	S	W								X
Stromberg et al. 2010	B	N		X			X			X
Stuever 1997	N	W			X			X	X	
Stuever et al. 1995	B	W		X		X	X			
Sunderman 2009	A, B	W		X	X			X	X	X
Sunderman and Weisberg 2011	B	W		X	X	X	X			
Sunderman and Weisberg 2012	A, B	W		X	X					X

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