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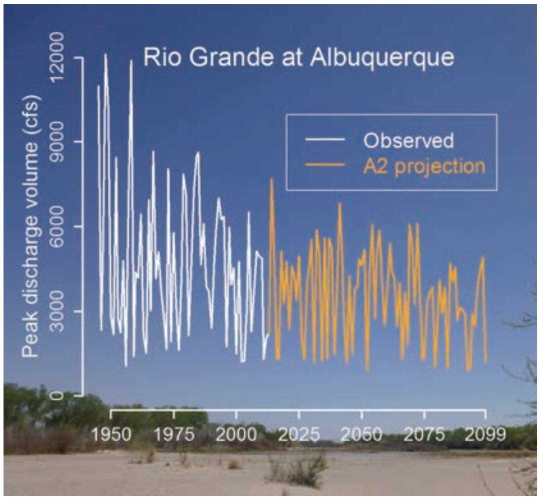
Climate Change and Wildfire Effects in Aridland Riparian Ecosystems: An Examination of Current and Future Conditions

D. Max Smith and Deborah M. Finch



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Abstract

Aridland riparian ecosystems are limited, the climate is changing, and further hydrological change is likely in the American Southwest. To protect riparian ecosystems and organisms, we need to understand how they are affected by disturbance processes and stressors such as fire, drought, and non-native plant invasions. Riparian vegetation is critically important as foraging, resting, migrating, and breeding habitat to birds and other animal species in the southwestern United States. Fremont cottonwood (*Populus fremontii*), Arizona sycamore (*Platanus wrightii*), and other woody species provide birds with nesting sites and foraging opportunities, some of which are absent or rare in adjacent plant communities. The structurally diverse, species-rich vegetation along many southwestern streams supports high densities of territories and nest sites for a variety of birds including several species of high conservation priority. Survival and reproduction of woody riparian plants is largely determined by periodic floods and droughts. As in other regions, rivers and streams of the American Southwest have been heavily altered by human activity, resulting in significant changes to disturbance regimes. Hydrological models, incorporating greenhouse gas emission scenarios, project that these changes will be exacerbated by climate change. In this report, we review the ecohydrology of southwestern streams and share results from our study sites along the Middle Rio Grande to describe effects of hydrological changes, wildfire, and invasions on plant communities and riparian-nesting birds. We also examine climate change projections and output from population models to gauge the future of aridland riparian ecosystems in an increasingly arid Southwest.

Keywords: breeding birds, climate change, Middle Rio Grande, riparian, woody vegetation, wildfire

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Cover photos: Clockwise from the top, Tonto Creek near Roosevelt, AZ; a wildfire in the Middle Rio Grande riparian forest; Mourning Dove (*Zenaida macroura*) nestlings; the Rio Grande stream bed near Albuquerque, NM. All photos by D.M. Smith.

Contents

Introduction

A frequently discussed function of aridland riparian ecosystems is the contribution of woody riparian plants to breeding bird habitat. Fremont cottonwood (*Populus fremontii*), Arizona sycamore (*Platanus wrightii*), and other species provide birds with nesting sites and foraging opportunities, some of which are absent or rare in adjacent plant communities (fig. 1); (Bock and Bock 1984; Carothers et al. 1974; Hunter et al. 1987). The structurally diverse, species-rich vegetation along many southwestern streams supports high densities of territories and nest sites for a variety of birds including several species of high conservation priority (fig. 2; Smith and Finch 2014; Smith et al. 2009a; Stoleson and Finch 2003).

Figure 1—The mixture of native and nonnative woody vegetation along the San Juan River has greater structural diversity than the adjacent plant community.

Survival and reproduction of woody riparian plants is largely determined by periodic disturbances such as flood and drought. As in other regions, streams of the American Southwest have been heavily affected by human activity, resulting in significant changes to disturbance regimes (Meritt and Poff 2010; Shafroth et al. 2002). Hydrological models, incorporating greenhouse gas emission scenarios, predict that these changes will be exacerbated by climate change (Seager et al. 2013). Given the limited extent of aridland riparian ecosystems and likelihood of further hydrological change, an understanding of current and future effects of disturbance processes on populations of riparian plants is needed to protect breeding bird communities in the American Southwest. In this report, we review the ecohydrology of southwestern streams and share results from our studies at the Middle Rio Grande to describe effects of hydrological changes and wildfire on woody plants and riparian-nesting birds. We also examine climate change projections and output from population models to gauge the future of aridland riparian ecosystems in an increasingly arid Southwest.

Figure 2—The riparian vegetation along the upper Gila River in southwestern New Mexico has high species richness of woody plants and extremely high densities of nesting birds including the Federally endangered Southwestern Willow Flycatcher (*Empidonax traillii extimus*) and the Federally threatened Yellow-Billed Cuckoo (*Coccyzus americanus*).

Chapter 1. Dynamics of Aridland Riparian Ecosystems

Purpose and Methodology

To maintain the function of aridland riparian ecosystems, an understanding of the dynamics of streamside plant and animal communities is needed. In this chapter, we examine current variation in hydrological characteristics among sites distributed across the American Southwest, review the ecohydrology of riparian plant species, and describe use of riparian vegetation by breeding birds.

For the purposes of this report, we focus on semiarid-to-arid portions of Utah, Colorado, New Mexico, and Arizona that are within the Colorado River and Rio Grande basins. We reviewed peer-reviewed studies of riparian vegetation and riparian-nesting birds conducted along streams in this region. We also summarized data from gauge sites along nine streams in the Colorado River basin and two in the Rio Grande basin to illustrate hydrological differences among these streams (table 1). We classified the stream gauge sites into two geographical groups. Six of the sites, referred to hereafter as "Rocky Mountain sites," were along streams headwatered in the Rocky Mountains of Wyoming and Colorado. Five sites, referred to as "Central Highland sites," were headwatered in the Central Highland ranges of Arizona and New Mexico (fig. 3). The Rocky Mountain stream sites are on the Colorado River, the Green River, the Gunnison River, The San Juan River, the Rio Chama, and the Rio Grande. The Central Highland stream sites are on the Gila River, the Salt River, the San Francisco River, Tonto Creek, and the Verde River. Gauge sites were between 600 and 2,000 m in elevation. Based on long-term temperature and precipitation records, the Rocky Mountain stream gauge sites were cooler and drier than the Central Highland sites (table 1).

We examined patterns of hydrologic variables that characterize streams and affect survival and reproduction of riparian vegetation. These variables, defined here, were:

- *annual discharge* (in million cubic meters) = the total volume of water measured at a stream gauge site each year;
- *mean daily discharge* (in cubic meters per second) = the mean discharge volume for each day of the year;
- *peak discharge magnitude* (in cubic meters per second) = the maximum mean daily discharge value measured each year; and
- *peak discharge date* = the day of each year that the peak discharge occurred.

We obtained discharge data for the period of 1/1/1960 to 12/31/2011 recorded at the 11 stream gauge sites by the U.S. Geological Survey and made available online by the National Water Information System Database (http://waterdata.usgs.gov/nwis). We estimated the annual discharge volume and determined the magnitude and date of the peak discharge at each stream site for each year of the historical period.

Table 1—Geographic and climatic characteristics of the 11 stream gauge sites examined in this chapter. Precipitation and temperature data were obtained from weather stations near stream gauge sites, made available the Western Regional Climate Center (http://www.wrcc.dri.edu/Climsum.html).

Gauge site	Gauge number	Basin	Headwaters	Elevation (m)	Mean annual precipitation (cm)	Mean min temperature (C)	Mean max temperature (C)
Colorado River at Cameo	09095500	Colorado River	Rocky Mountains	1,467	25.1	4.9	19.6
Gunnison River at Grand Junction	09152500	Colorado River	Rocky Mountains	1,411	22.4	4.7	18.9
Green River at Greendale	09234500	Colorado River	Rocky Mountains	1,705	30.2	–1.4	15.3
San Juan River at Bluff	09379500	Colorado River	Rocky Mountains	1,234	19.8	3.9	21.3
Rio Grande at Otowi	08313000	Rio Grande	Rocky Mountains	1,673	25.1	1.4	20.3
Rio Chama at Abiquiu	08286500	Rio Grande	Rocky Mountains	1,914	24.9	2.9	18.2
Salt River at Roosevelt	09498500	Colorado River	Central Highlands	664	33.8	12.7	27.2
Verde River above Horseshoe Dam	09508500	Colorado River	Central Highlands	618	36.1	13.5	29.3
San Francisco River at Clifton	09444500	Colorado River	Central Highlands	1,047	34.0	11.0	27.2
Gila River at Gila	09430500	Colorado River	Central Highlands	1,419	36.6	3.1	23.6
Tonto Creek near Roosevelt	09499000	Colorado River	Central Highlands	769	44.5	9.4	27.5

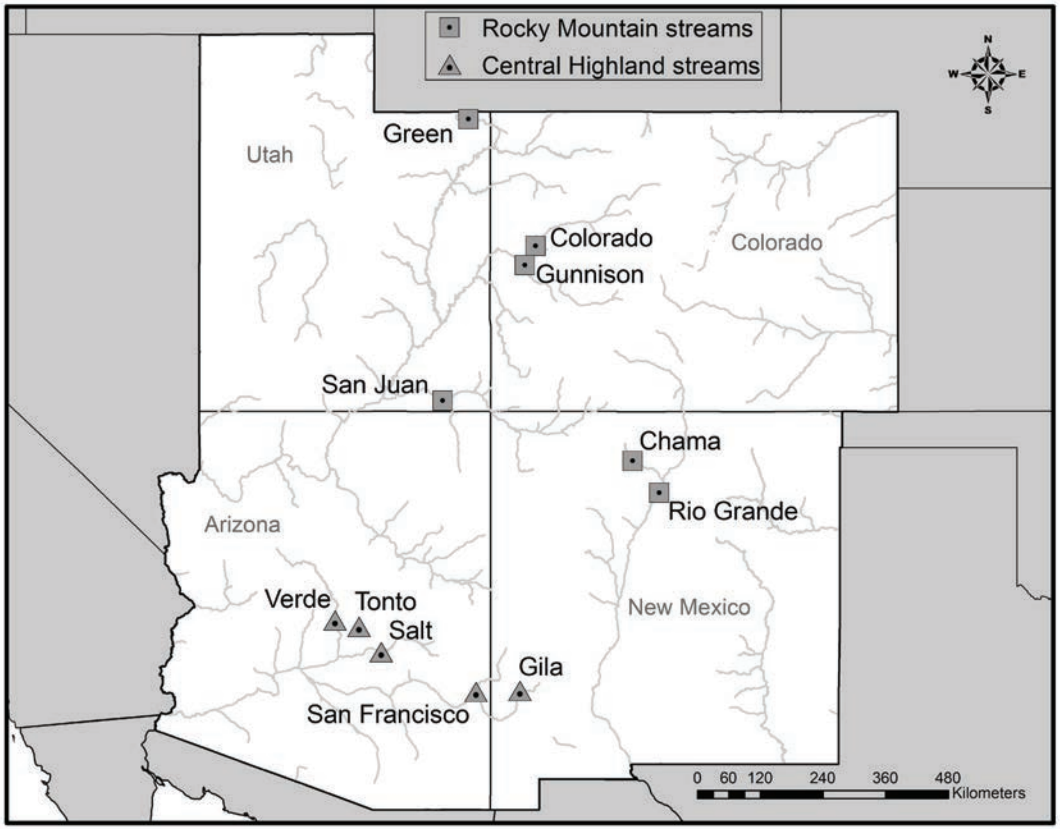
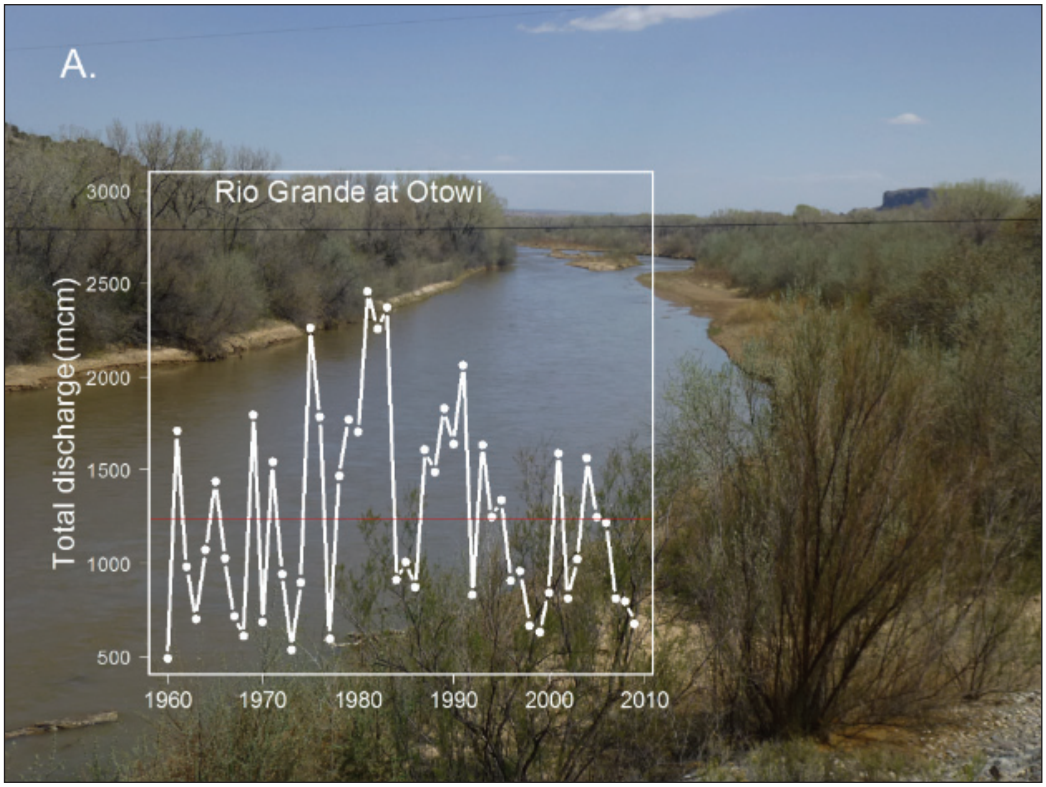


Figure 3—The gauge sites examined were along streams headwatered in the Rocky Mountains and Central Highland ranges.

Characteristics of Southwestern Streams

Hydrologic Patterns

Our analysis of stream gauge data shows that there are consistent differences in characteristics between Rocky Mountain sites and the Central Highland sites. Of the sites examined, mean annual discharge and mean peak discharge were generally greater for Rocky Mountain sites, but maximum peak discharges were greater at most of the Central Highland sites (figs. 4 and 5, table 2). Timing of peak discharge was more consistent at Rocky Mountain sites, with peaks that generally occurred in May or June (table 2). On average, peak discharge at Central Highland sites occurred in February or March (table2), but these peaks also occurred during other times of the year (figs. 5 and 6). These differences in discharge patterns result from a number of factors. With their high elevation (>3,000 m) and latitudinal position, the Rocky Mountain ranges accumulate heavy snowpacks during the winter months, which contribute to the peak discharges that predictably occur during the spring and summer in the Southwest. The Central Highland ranges are lower in elevation and latitude than the Rocky Mountain headwaters, so a combination of snowmelt and rain contributes to peak discharges (Neary et al. 2012; Webb et al. 2007). Central Highland ranges intercept extremely large amounts of rainfall from winter Pacific frontal storms, fall tropical storms, and summer monsoons (Stromberg et al. 2007).



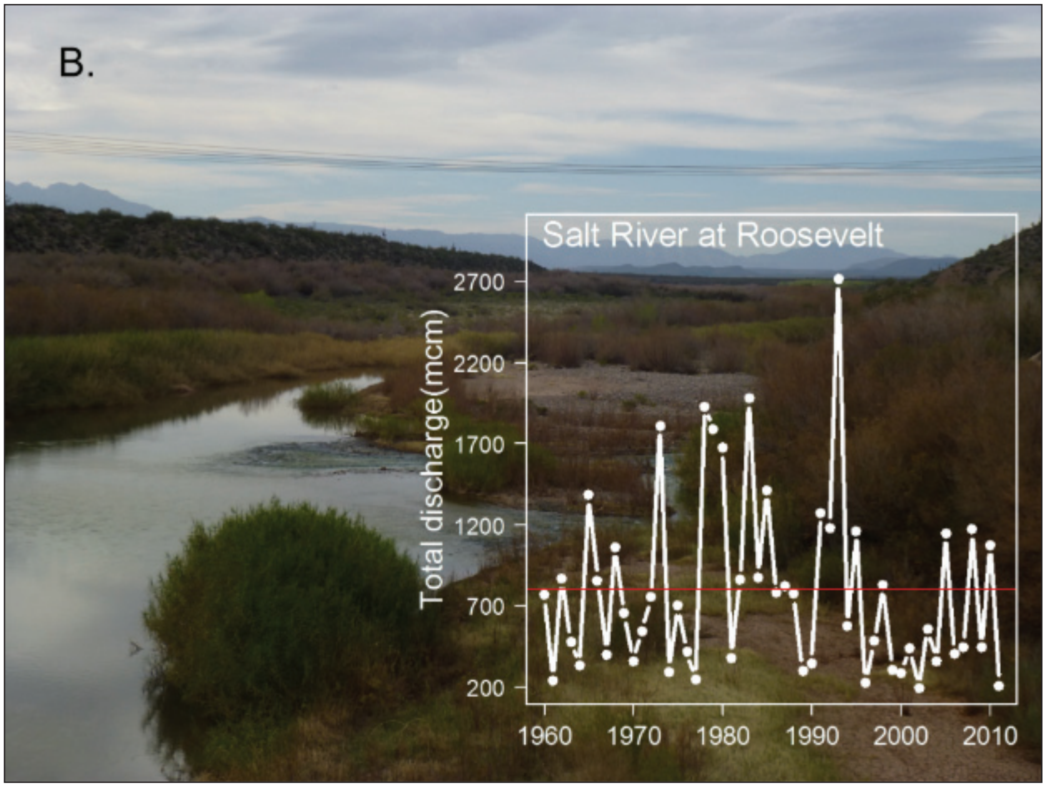
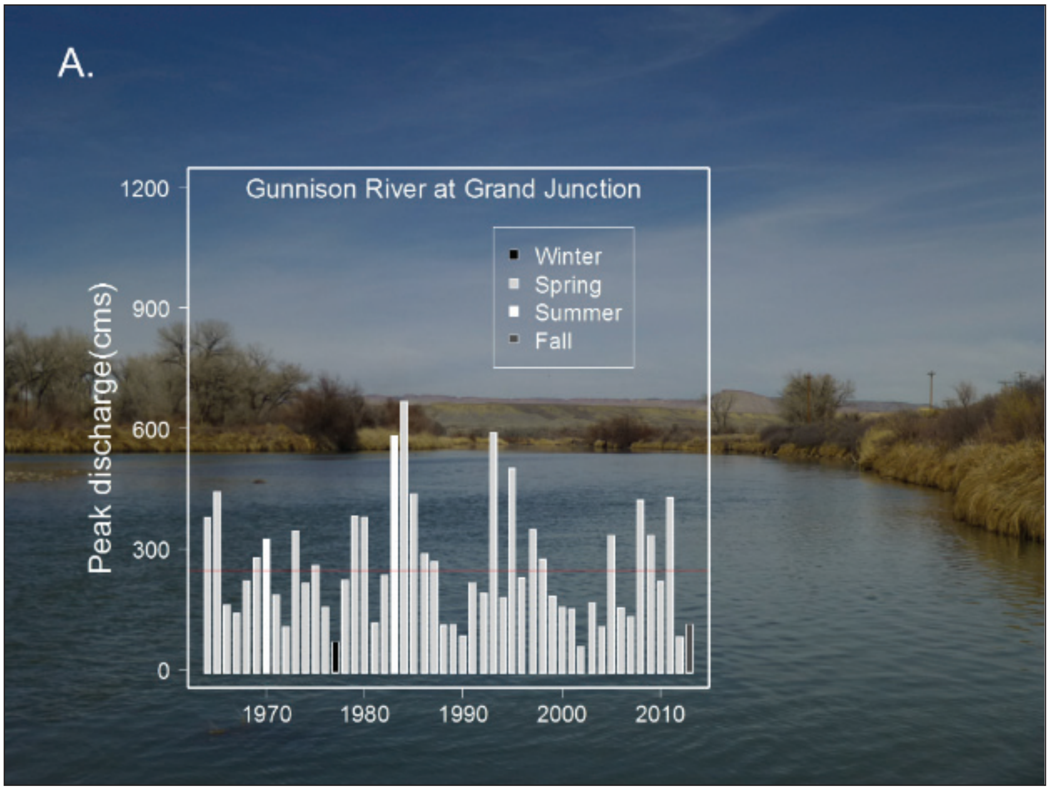


Figure 4—Annual discharge measured at stream gauge sites at the Rio Grande at Otowi, NM, (A) and Salt River at Roosevelt, AZ (B). The range of annual discharge is greater for the Salt River and similar Central Highland streams than for the Rio Grande and other Rocky Mountain streams. The red line indicates the long-term average.



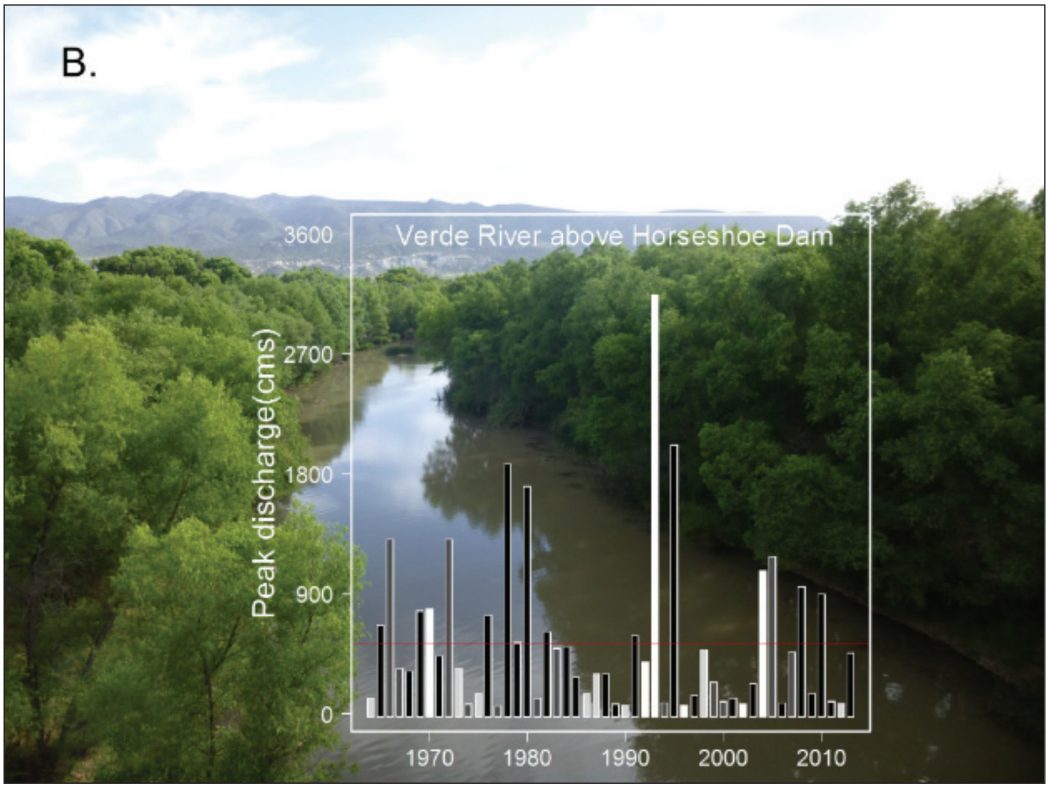


Figure 5—Magnitude and timing of peak discharges measured at stream gauge sites at the Gunnison River at Grand Junction, CO, (A) and the Verde River above Horseshoe Dam, AZ (B). Maximum peak discharges are greater at the Verde and other Central Highland streams than at the Gunnison and other Rocky Mountain streams. Timing of peak discharge varies among the streams as well. The red line indicates the long-term average.

Table 2—Streamflow characteristics recorded at stream gauge sites used in this analysis. Data were obtained from the National Water Information System Database (http://waterdata.usgs.gov/nwis).

Gauge site	Mean annual discharge 1960–2011 (million cubic meters)	Maximum peak discharge 1960–2011 (cms)	Peak discharge month
Colorado River at Cameo	3394.0	1076.0	May
Gunnison River at Grand Junction	2194.1	656.9	May
Green River at Greendale	1741.0	365.2	June
San Juan River at Bluff	1799.2	982.5	June
Rio Grande at Otowi	1235.1	339.8	May
Rio Chama at Abiquiu	414.3	183.4	May
Salt River at Roosevelt	797.1	2525.8	March
Verde River above Horseshoe Dam	538.2	3114.8	March
San Francisco River at Clifton	211.9	1478.1	March
Gila River at Gila	157.9	662.6	March
Tonto Creek near Roosevelt	151.1	1039.2	March

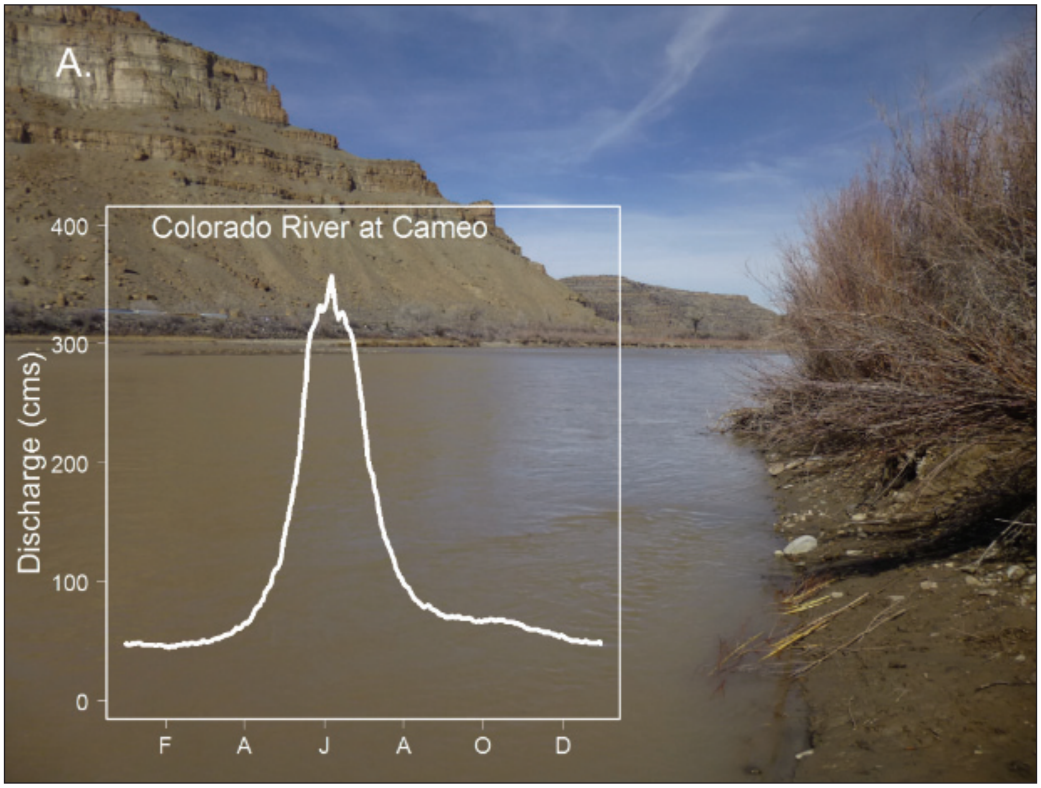
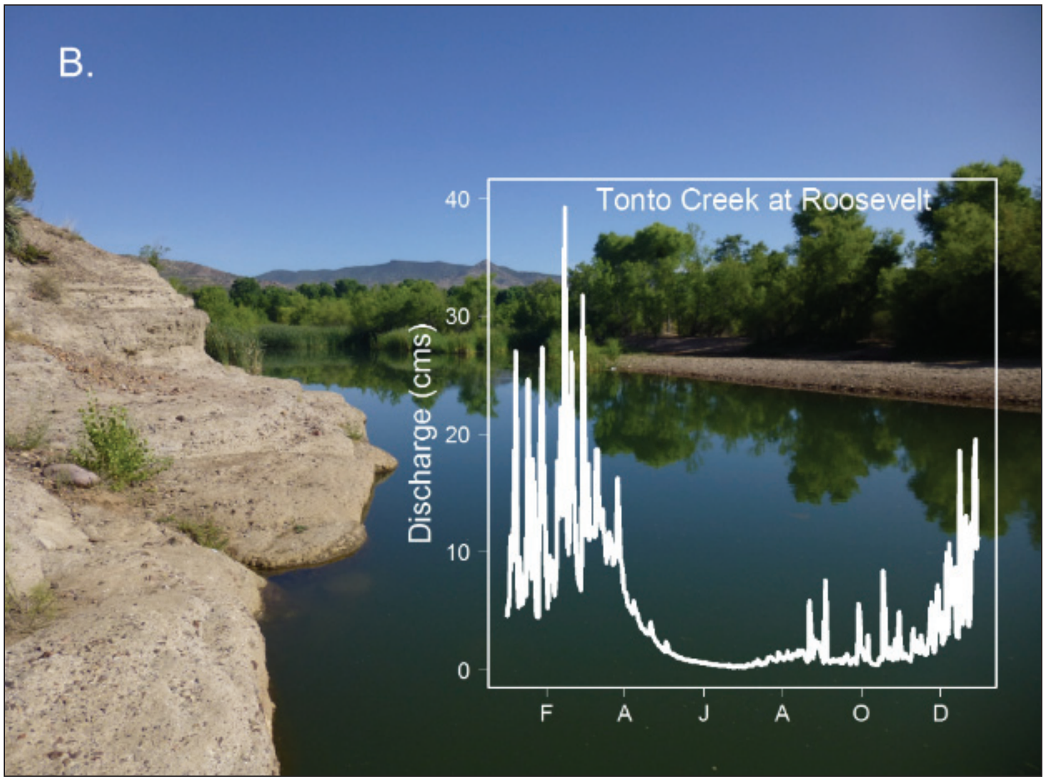


Figure 6—Mean daily discharge for each day of the year from 1960 to 2011 at the Colorado River at Cameo, CO, (A) and Tonto Creek at Roosevelt, AZ (B). Peak discharges of the Colorado and other Rocky Mountain streams typically occur in spring. Peak discharges of Tonto Creek and other Central Highland streams can occur from late summer through early spring.



Given the localized nature of these storms and annual variation in their occurrence, there is substantial variation in peak discharge at Central Highland streams (Stromberg et al. 2007).

The differences in peak discharge are reflected in the phenology of woody vegetation. At Rocky Mountain streams, cottonwoods and willows release seeds during the spring and summer when snowmelt-driven floods typically subside (Cooper et al. 1999; Molles et al. 1998). At Central Highland streams, cottonwood and willow seed dispersal occurs during the late winter and early spring, typically coinciding with the drawdown of high flows that result from the combination of snowmelt and rain (Beauchamp and Stromberg 2007).

Anthropogenic Changes

In recent centuries, hydrology, geomorphology, and ecology of streams have been affected by factors that include changes in climate and anthropogenic land use (Scurlock 1998; Stromberg et al. 2010a). Among the most significant of the latter is regulation of streams to control flood risk and provide water for agricultural, industrial, and municipal purposes. This regulation includes the construction of dams and reservoirs, channelization of streams, withdrawal of groundwater, and diversion of surface flows (fig. 7; Phillips et al. 2011; Summit 2013). Peak discharge magnitude, timing, and duration are now altered from historical conditions at many streams and some sections that once had perennial flows now run dry apart from periods of heavy runoff (White and Stromberg 2009). Other sections are inundated by reservoir pools behind dams while

Figure 7—Anthropogenic influences on streams include (A) diversion of surface flows, (B) groundwater pumping, (C) channelization, and (D) construction of large dams and reservoirs.

below many dams, magnitude and timing of peak flows have been altered. In addition, sediment accumulates upstream from dams, while sediment-poor water causes degradation below dams, disconnecting floodplains and increasing the depth to groundwater (Novack 2006).

The level of regulation differs among streams' gauge sites we examined. Dams were constructed upstream from all of the Rocky Mountain gauge sites, with the largest dams and reservoirs (those exceeding 1 million acre feet capacity) upstream from the gauges on the Green and San Juan rivers. Large dams and reservoirs are located downstream from these sites as well, with major dams on the Colorado River below its junctions with the Green, San Juan, and Gunnison Rivers (Summit 2013). There are also large dams and reservoirs downstream from the gauge sites on the Rio Chama and Rio Grande (Phillips et al. 2011). In addition to dams, trans-basin diversions lie upstream from several Rocky Mountain stream sites. These include the Colorado-Big Thompson Project, which transfers flows from the upper Colorado River over the Continental Divide to eastern Colorado, augmenting flow in the Big Thompson River (Dewine and Cooper 2007). The San Juan-Chama Project transfers flows from the San Juan River to the Rio Chama for municipal and agricultural use in central New Mexico (Flanigan and Haas 2008). There are only minor dams and diversions above the Central Highland stream gauge sites examined, so the streams are largely unregulated at these locations. Large dams, reservoirs, and diversion projects are located downstream from the gauge sites examined on the Verde River, Salt River, and Gila River (Webb et al. 2007). Though flows are generally perennial at each of the Central Highland stream sites, dams and diversions prevent the lower Salt River from flowing to its confluence with the Gila River apart from periods of heavy precipitation (White and Stromberg 2009). Likewise, the lower Gila River rarely reaches its confluence with the Colorado River in Arizona (Summitt 2013).

Ecohydrology of Woody Vegetation

Opportunities for reproduction of woody plants are limited along aridland streams (Bock and Bock 1989; Cooper et al. 1999). Pioneering species such as cottonwoods, willows (*Salix* spp.), and saltcedars (*Tamarix* spp.) have short-lived seeds that will not establish unless they settle on damp and exposed substrates (fig. 8). Other riparian trees, including boxelders (*Acer negundo*) and Russian olives (*Elaeagnus angustifolia*), have large seeds with long viability periods and the ability to establish in shaded sites with ground cover, though damp conditions are required to induce germination (fig. 9; Dewine and Cooper 2007; Katz and Shafroth 2003).

Figure 8—The small seeds dispersed by (A) Rio Grande cottonwood, (B) Goodding's willow, and (C) saltcedar are transported by wind and water and require exposed, damp surfaces for germination.

Figure 9—The larger seeds of (A) boxelder and (B) Russian olive are dispersed by wind and by animals. These seeds can germinate on covered surfaces if moisture is present.

Along aridland streams, peak discharges can produce floods that create conditions required for germination of woody plants. Periods of heavy precipitation or snowmelt result in flows that scour vegetation and litter, re-route stream channels, and deposit sediment (fig. 10). In the wake of these floods lie sites that are devoid of competing vegetation and ideal for germination of pioneering species (fig. 11; Auble and Scott 1998). As flood waters recede, they leave behind soil moisture that triggers germination and promotes seedling survival (Bhattacharjee et al. 2008). Seed dispersal of cottonwood, willow, and saltcedar typically coincides with the drawdown of spring floods, when these sites are left exposed (Braatne et al. 1996; Sher et al. 2002). Following establishment, phreatophytic species such as Fremont cottonwood and Goodding's willow (*Salix gooddingii*) require a connection between their roots and the groundwater table to ensure growth and survival (Busch et al. 1992; Snyder and Williams 2000). At many streams, high peak discharges are needed to recharge aquifers and maintain this connection (Stromberg 2001). Reduction in magnitude of peak discharge can reduce recharge rates, causing dieback and drought mortality (fig. 12). Through these influences on reproduction and survival of woody vegetation, stream characteristics such as magnitude and timing of peak discharge exert great control over the composition of riparian ecosystems (Brand et al. 2008; Merritt and Bateman 2012).

Figure 10—High flows as seen here on the Animas River in northwestern New Mexico are important for providing germination sites and recharging groundwater aquifers, thereby promoting reproduction and survival of woody riparian vegetation.

Figure 11—These cottonwood (yellow arrow) and saltcedar (red arrow) seedlings established following the drawdown of high springtime flows near the active channel of the Middle Rio Grande.

Figure 12—During periods of low flow, woody riparian plants are vulnerable to drought stress and mortality if depth to groundwater exceeds root lengths, as shown by dead shrub in lower left of figure.

Current State of Riparian Ecosystems

Changes in discharge brought upon by large-scale regulation have had varying effects on woody riparian vegetation. Dams limit the extent of flooding and germination within the floodplain and lower the water table in areas away from the active channel, thereby increasing mortality of established trees (Coble and Kolb 2013; Dewine and Cooper 2007; Molles et al. 1998). Changes to peak discharge timing also limit reproduction of species such as cottonwoods and willows (Cooper et al. 1999; Mortenson and Weisberg 2010). Along many regulated southwestern streams, conditions have become more suitable for nonnative species, such as saltcedar and Russian olive, that have longer seed dispersal periods, longer seed viability, and greater resistance to drought than certain native species (Birken and Cooper 2006; Busch and Smith 1995; Mortenson and Weisberg 2010).

Following the alteration of streamflow and floodplain dynamics, wildfire emerged as a significant disturbance agent at the end of the 20th century. Following the reduction in frequency and magnitude of floods, litter and debris accumulated in the forest understory. This accumulation, along with increased density of native and nonnative vegetation, resulted in fuel loads, fire size, and fire intensity that are greater than existed prior to stream modification (Bêche et al. 2005; Stuever et al. 1995). Wildfire is thought to be a historically rare occurrence in riparian zones and little is known about its effects on riparian organisms (Bock and Block 2005). Information from postfire sites is therefore needed to project long-term changes in aridland riparian ecosystems.

Use of Riparian Vegetation by Breeding Birds

The trees and shrubs growing along aridland streams are used by scores of bird species during the nesting season. Both native and nonnative plants serve as nest sites and foraging substrates for birds occupying multiple niches (Smith and Finch 2014). Though riparian-nesting birds use a variety of plants for nesting and foraging, results from studies along southwestern streams highlight differences among woody species in the resources they provide.

Cottonwoods, Goodding’s willow, boxelder, and Arizona sycamore are riparian trees that are frequently used as nest sites for birds along aridland streams (fig. 13). Under typical conditions, cottonwoods and sycamores are the tallest species in alluvial stretches, capable of forming stands supporting greater densities and richness of birds than other vegetation types (Carothers et al. 1974; Merritt and Bateman 2012; Powell and Steidl 2001; Strong and Bock 1990). Cottonwoods and sycamores are used by a greater number of bird species than other trees because they have large branches to support heavy nests, they have dead and decaying limbs that are excavated by woodpeckers for food and nest sites, and they feature substantial canopies used by foliage gleaners (Bock and Bock 1984; Hunter et al. 1987; Smith and Finch 2014; Stoleson et al. 2000).

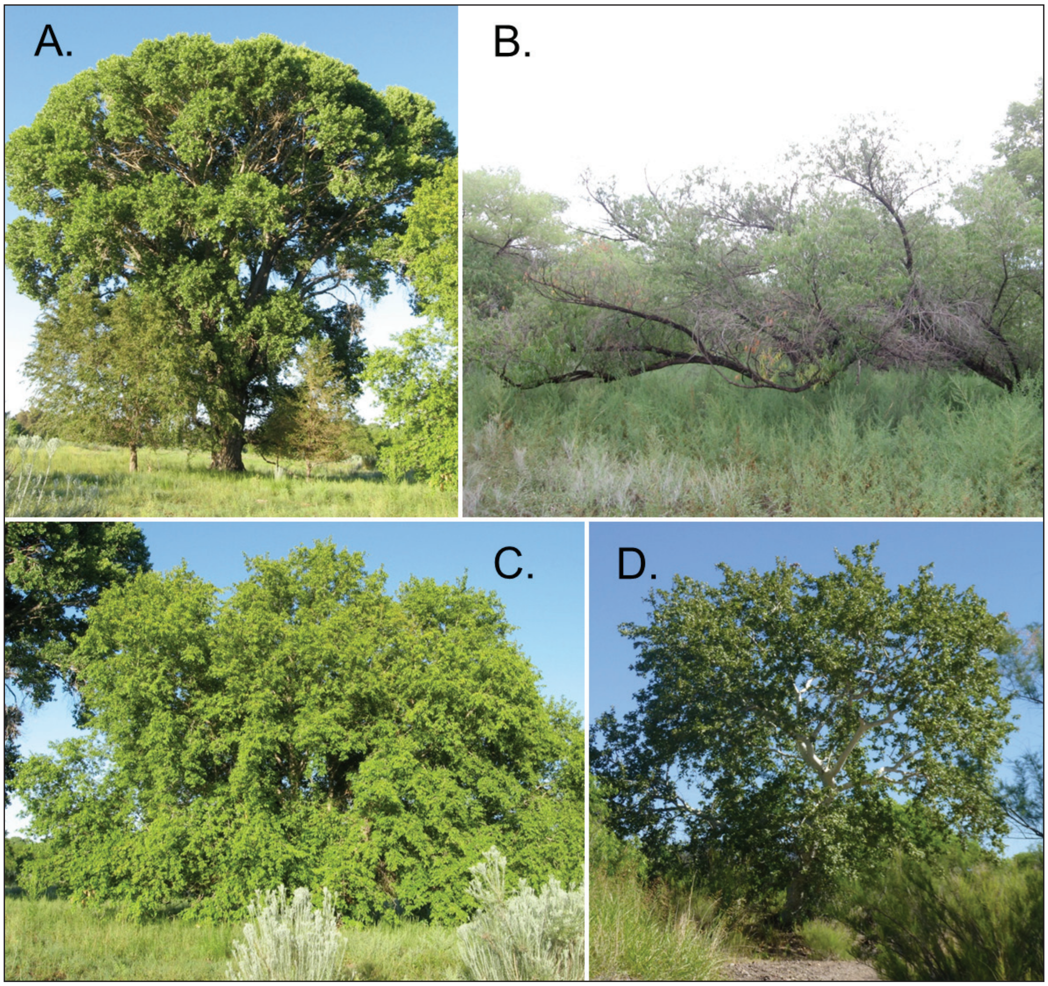


Figure 13—Native trees of aridland riparian ecosystems include (A) cottonwood, (B) Goodding's willow, (C) boxelder, and (D) Arizona sycamore.

Range-restricted species including Common Black-Hawk (*Buteogallus anthracinus*), Violet-crowned Hummingbird (*Amazilia violiceps*), and Gila Woodpecker (*Melanerpes uropygialis*) construct most, if not all of their nests in cottonwoods and sycamores at their riparian breeding grounds (Hunter et al. 1987; Smith and Finch 2014; Wethington 2002). In addition to nest sites, cottonwoods maintain populations of arthropods, such as cicadas (*Tibicen marginatus)* and floodplain crickets (*Gryllus alogus*), by providing food, water, and oviposition sites (Sabo et al. 2008; Smith et al. 2006a). These arthropods are critical sources of food for adult and juvenile birds (Rosenberg et al. 1982), making cottonwoods a key component of the riparian food web (fig. 14).

Fewer birds are known to nest in Goodding's willow and boxelder as compared to cottonwoods and sycamores, but these trees are important to certain species and the breeding bird community as a whole. Mature boxelder and Goodding's willow trees are excavated by woodpeckers and they are used as nest plants by the range-restricted, secondary cavity-nesting Lucy's Warbler (*Oreothlypis luciae*; Stoleson et al. 2000).

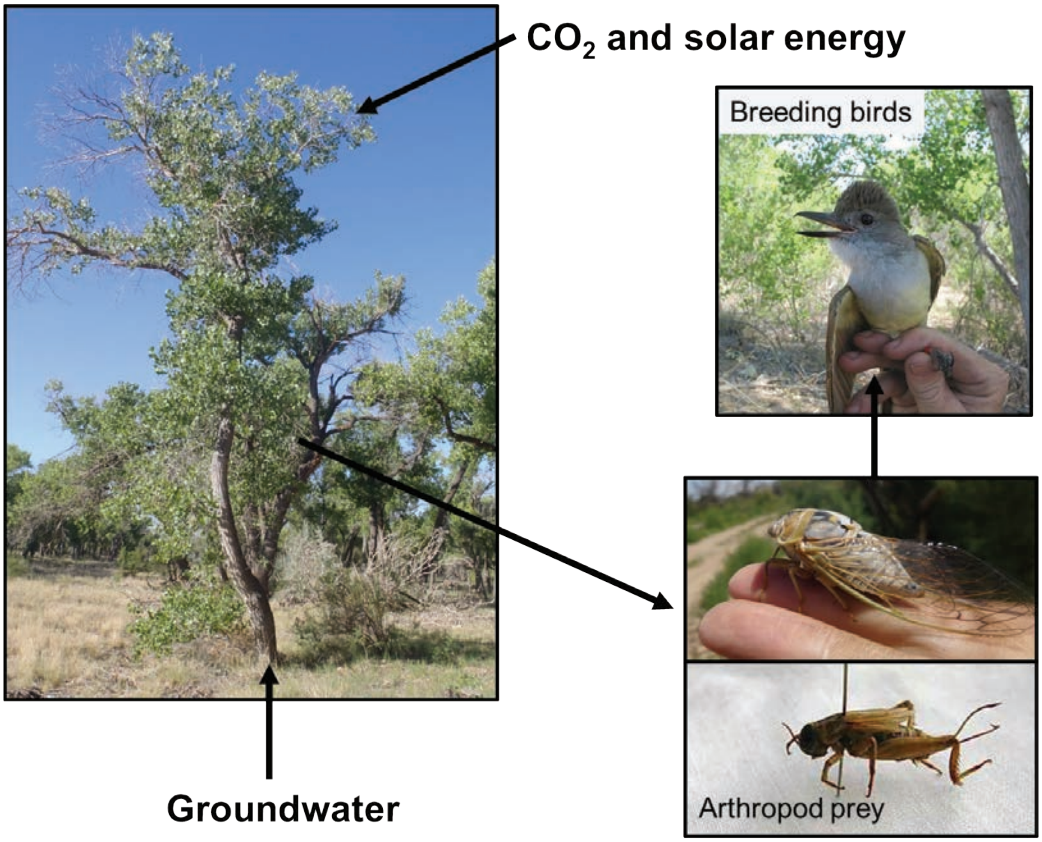


Figure 14—Floodplain cottonwoods take in groundwater, CO_2, and solar energy to create carbohydrates. They also store water in the roots and transport tissue. Cicadas obtain food and water from the roots as nymphs and the branches as adults; floodplain crickets obtain food and water from green leaves that are blown off by the wind. These arthropods are in turn consumed by breeding birds and other insectivores.

The trunks of Goodding's willows often grow horizontally, making them ideal nest sites for Mourning Doves (*Zenaida macroura*) and other large understory-nesters (Smith et al. 2012). Along with cottonwood, the presence of Goodding's willow increases the abundance of breeding birds in riparian patches (Brand et al. 2008). Boxelder is as an important habitat component for birds along the upper Gila River because it is preferred as a nest plant over other trees by the Federally endangered Southwestern Willow Flycatcher (*Empidonax traillii extimus*; Stoleson and Finch 2003), and is also used by the Federally threatened Yellow-Billed Cuckoo (*Coccyzus americanus*). In addition, Willow Flycatcher nests in boxelder are less vulnerable to Brown-headed Cowbird (*Molothrus ater*) parasitism than nests in other species (Brodhead et al. 2007). Where it co-occurs with larger trees, boxelder forms a woody subcanopy layer that is absent in many western riparian forests, effectively increasing potential nesting opportunities (Knopf and Olson 1984; Smith and Finch 2014).

Russian olive and saltcedar are widespread nonnative species that form a considerable component of riparian-nesting bird habitat in the Southwest (Friedman et al. 2005). These species can grow as small trees (fig. 15) but are typically not large enough to support cavity and canopy-nesting birds (Smith and Finch 2014; Stoleson and Finch 2001). Russian olive and saltcedar do, however, provide nest sites for shrub and subcanopy-nesting birds including Willow Flycatcher and Yellow-billed Cuckoo (Brown 1992; Stoleson and Finch 2001). Densities of some breeding birds are greater in stands of saltcedar than in native stands (Brand et al. 2009). For some species, including Black-chinned Hummingbird (*Archilochus alexandri*) and Willow Flycatcher, rates of nesting success in these plants is equal to or exceeds rates measured in native vegetation (Smith et al. 2009b; Sogge et al. 2008). In some areas, however, Willow Flycatchers, Bell's Vireo, and possibly other species, have greater rates of cowbird parasitism in nonnative vegetation than in native vegetation (Brand et al. 2009; Stoleson and Finch 2001). In addition to providing nest sites, nonnative woody plants contain densities of arthropod prey similar to those of native riparian plants (Durst et al. 2008; Mund-Meyerson 1991). Russian olive and saltcedar can therefore provide foraging opportunities for certain foraging guilds, but birds in canopy gleaning, bark gleaning, and excavating guilds require larger trees such as cottonwoods and sycamores (Bock and Bock 1984; Ellis 1995).

Results from the studies above have shown that composition of riparian vegetation influences habitat suitability for certain riparian-nesting birds. Changes in hydrological conditions and disturbance regimes can therefore influence not only woody plants, but also their associated animal communities.

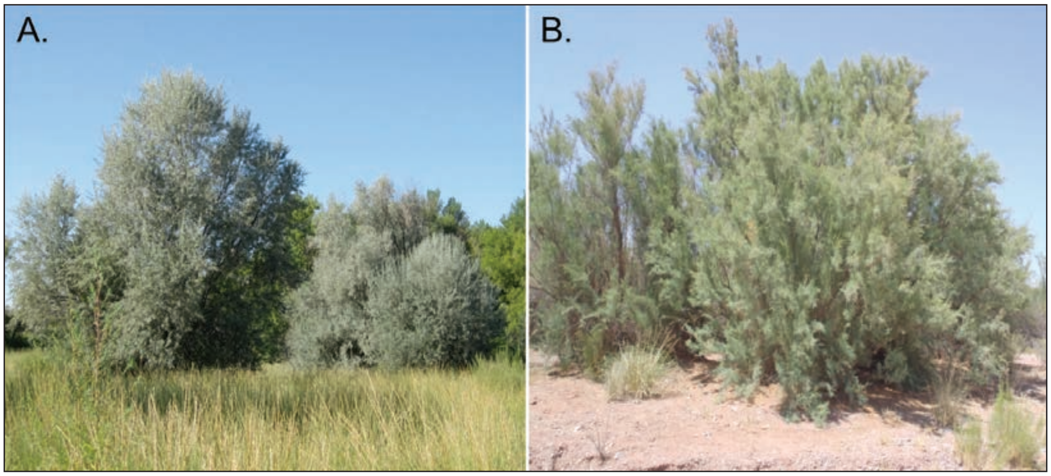


Figure 15—(A) Russian olive and (B) saltcedar grow as small trees in aridland floodplain, but they typically do not grow large enough to support the nests of canopy- and cavity-nesting birds. These birds prefer trees that support nest heights of 10 m or taller.

Chapter 2. Response of Woody Riparian Plants to Wildfire

Introduction: Wildfire and Woody Vegetation

Wildfire influences the composition and structure of woody plant communities, primarily as an agent of mortality. When the aboveground tissues of trees and shrubs are killed by wildfire (hereafter referred to as "topkilled"), ecosystems are affected by the loss of services including canopy shading and vertical transport of water and nutrients (Smith et al. 2006a; Whelan 1995). These losses in turn alter the availability of resources to animal communities. Fire also creates snags and woody debris, which provide habitat for terrestrial and aquatic animals and help to control hydrologic and geomorphic processes (Brown 2002; Minckley and Rinne 1985; Smith et al. 2012; Thomas et al. 1979).

Woody plants use a variety of mechanisms to regenerate after fire. Though fire can result in topkill of broadleaved trees and shrubs, individuals of certain species recover vegetatively by producing basal sprouts or root suckers from underground buds (Kramer and Kozlowski 1979). If these sprouts experience conditions ideal for growth and survival, top-killed stands can replace themselves after fire (Bond and Midgley 2001; Cocking et al. 2014; Keyser et al. 2005). In addition, certain species can rapidly reestablish a forest canopy with epicormic sprouts that arise from tree boles and branches (Clarke et al. 2012). Woody species can also recover from fire through seedling establishment. For some species in frequently burned vegetation types, fire stimulates seed dispersal by opening serotinous cones and, through mechanisms such as scarification and soil warming, encourages germination (Bonnet et al. 2005; Enright et al. 1996; Keeley 1987; Lotan 1976;).

On stream sections such as the Middle Rio Grande in central New Mexico, fires often burn with high severity (fig. 16) fueled by litter, debris, and vegetation that has accumulated in the absence of high-magnitude floods (Drus 2013; Ellis 2001; Stuever 1997). Few studies have examined response of woody plants in riparian areas to wildfire relative to other southwestern vegetation types. However, it is widely perceived that on regulated streams, native woody plants do not recover from fire as well as nonnative species such as Russian olive and saltcedar, which resprout more vigorously and are more flexible in their germination requirements (Busch 1995). To address the consequences of a changing disturbance regime, a systematic understanding of the effects of wildfire on structure and composition of riparian ecosystems is needed. In this section, we report wildfire effects measured along the Middle Rio Grande in central New Mexico and review results from previous studies at this and other streams.

Figure 16—The trees in this photo, taken along the Middle Rio Grande 2 weeks after a high-severity wildfire, were burned and topkilled.

Postfire Dynamics Along the Middle Rio Grande

Study Area and Methods

The Middle Rio Grande is the section of the Rio Grande that flows through central New Mexico between Cochiti Dam and Elephant Butte Reservoir (fig. 17). Alterations to the river's hydrology have occurred over several centuries, but large-scale changes began in the 1920s when, to increase agriculture production and reduce flood damage, agencies constructed a network of diversion dams, levees, irrigation canals, and drains (Phillips et al. 2011; Scurlock 1998). The levees currently prevent the river from meandering across its natural floodplain. The stream bank has been stabilized by vegetation and structures, known as jetty jacks that limit the movement of the active channel within the floodway. A canopy of mature Rio Grande cottonwoods (*Populus deltoides* ssp. *wislizenii*) established in the 1950s when alluvium was stabilized by jetty jacks and colonized by seedlings (Crawford et al. 1993). The river became fully regulated by the closure of Cochiti Dam in 1973. Since completion of the dam, peak flood volume has been reduced and the stream channel has been narrowed and incised to disconnect the floodplain from streamflow in many locations (Novack 2006).

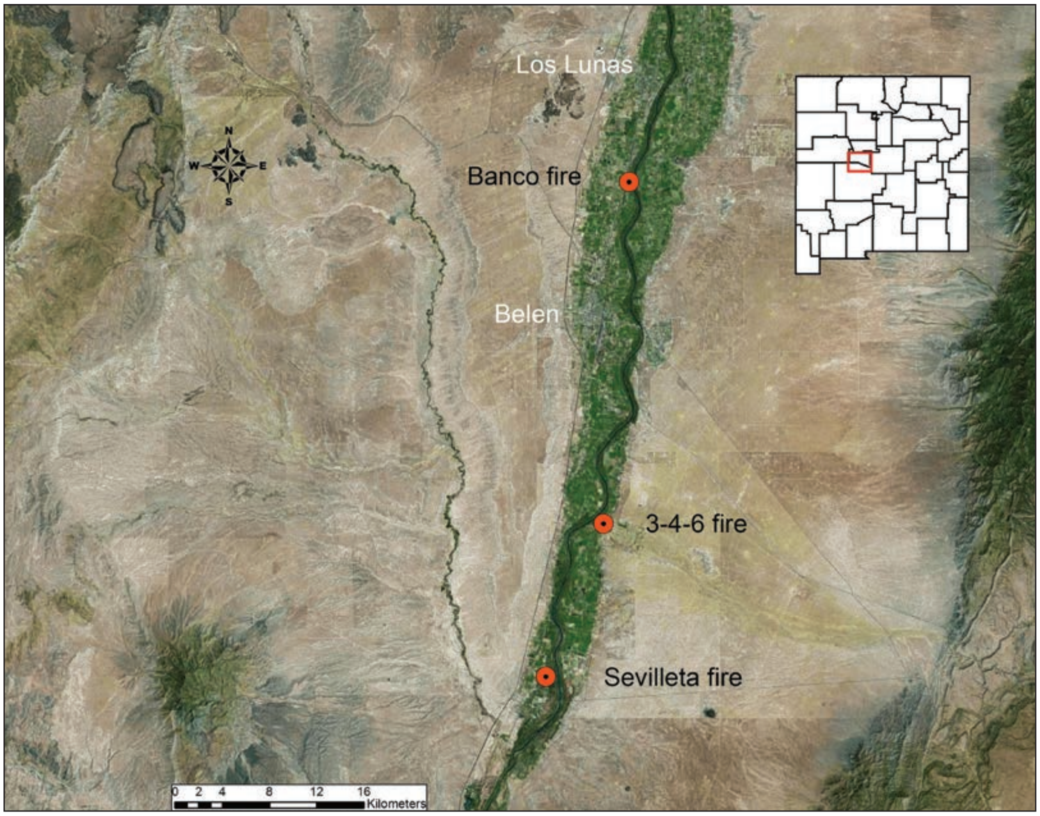


Figure 17—Our Middle Rio Grande study area contained the Banco wildfire site in the north section and the 3-4-6 and Sevilleta wildfire sites in the south section.

During years of low discharge, the active channel runs dry in certain areas where water is diverted for agriculture. In wet years with heavy runoff, flood pulses, released from Cochiti Dam, inundate some portions of the floodway, but floods lack the shear force needed to scour away woody vegetation. In addition, sediment transport is often confined to the active channel, where any woody plants that are established post-flood are vulnerable to subsequent high flows (Howe and Knopf 1991; Molles et al. 1998). As a result, cottonwoods, which once occurred in varied stage classes scattered across the Middle Rio Grande floodplain, are largely confined to dense, senescent stands located between levees and the stream channel (Whitney 1996).

As the role of flooding has diminished along the Middle Rio Grande, wildfire has become an increasingly common disturbance (Stuever et al. 1995; Williams et al. 2007). To measure postfire recovery of riparian trees, we collected data in two sections of riparian forest that differ in characteristics of hydrology and wildfire. The north section, located between the towns of Los Lunas and Belen in Valencia County, is larger than the south section and is bordered to the east by the active stream channel and to the west by agricultural fields and residential areas (fig. 17). The south section is south of Belen in Socorro County in a more rural area and is bordered to the west by agricultural fields. Based on our observations during the high runoff years of 2005 and 2008, the discharge threshold above which flooding occurs is lower at the north section than at the south section (table 3; Smith et al. 2009a). In addition, the typical depth to

Table 3—Section-specific variables at our Middle Rio Grande study area. Bankfull discharge threshold refers to discharge volume required for some flooding to occur, drought discharge refers to discharge volume below which drought mortality is observed, and floodway inundation threshold refers to discharge volume required for floods to inundate the entire area between the stream channel and the levees.

Variable	Section	
	North	South
Total area (ha)	411	321
Bankfull discharge threshold (cms)	142	198
Drought discharge threshold (cms)	42	57
Floodway inundation threshold (cms)	170	227
Area burned 2002–2012 (ha)	3	253

groundwater, which influences cottonwood drought mortality, is shallower in the north than in the south (Smith et al. 2009a). These sections also differ in the amount of area typically burned by wildfire (table 3). Though fewer fires per year are ignited in the rural south section, average fire size is larger because of longer firefighting response times than in the north section (Williams et al. 2007).

In 2013, we visited sites burned by three wildfires. The Sevilleta Fire was the largest of the three, burning 154 ha of the forest on the west side of the river on April 2 and 3, 2011. The 3-4-6 Fire was intermediate in size, burning 38.5 ha on June 26, 2011, and the Banco Fire was the smallest, burning 0.5 ha on March 26, 2008. The Sevilleta and 3-4-6 sites were located south of Belen in Socorro County and the Banco site was south of Los Lunas and north of Belen in Valencia County. We selected these sites because they were the most recent fires to occur in our study area during our 2013 sampling period. Prior to burning, each site consisted of a cottonwood canopy and an understory composed of native and nonnative vegetation, with Russian olive and saltcedar the most abundant woody species (Smith et al. 2007).

At each site, we established at least one transect that extended from the levee road to the river channel. We established a sampling point every 40 m along each transect, until we reached the stream channel or left the burned area. We marked a 12.6-m radius sampling plot at each point. Within the sampling plot, we measured bole condition (live or topkilled), diameter at breast height (d.b.h.), and sprouting status of each tree greater than 5 cm d.b.h. We assigned each tree a basal sprout status based on condition of trees and postfire sprouts. The statuses were: “bole live, no sprout,” “bole dead with live sprout,” “bole dead with no sprout,” and “bole dead with dead sprout.” Sprouting trees typically had one basal sprout clump, so we did not count the number of basal sprouts per tree. In addition to basal sprouts, we checked every tree for epicormic sprouts along the bole and in the canopy. At half of the sampling points, we measured abundance of regenerating shoots within a 5-m radius subplot centered on the sampling point. We identified the species of each shoot and determined whether it was a basal sprout, a root sucker, or a sapling that had germinated.

Topkill

The death of aboveground tissue is largely determined by the severity with which a woody plant is burned. If intensity of fire is too low to penetrate the outer layers of bark or scorch crown foliage, a woody plant may retain viable aboveground tissue. The ability to withstand fire, however, varies among size and species of woody plants (Gignoux et al. 1997; Hoffman and Solbrig 2003). Over 90 percent of cottonwoods, saltcedars, and Russian olives were topkilled at the sites we monitored (table 4). All of the cottonwoods examined at the 3-4-6 site and Banco site were topkilled. Ninety-five percent were topkilled at the Sevilleta Fire site (fig. 18), which appeared burned

Table 4—Woody plant species observed regenerating at post-wildfire fire sites in our Middle Rio Grande study area.

Species	Number of trees examined	Percent topkilled	Percent with basal sprouts	Percent with epicormic sprouts	Number of root suckers observed	Number of saplings observed
Rio Grande cottonwood	296	92	47	1.7	79	5
Saltcedar	96	94	90	0	371	0
Russian olive	52	93	80	0	4	0
Goodding’s willow	9	89	89	0	0	0
Screwbean mesquite	2	100	100	0	0	0
White mulberry	1	100	100	0	0	11
Siberian elm	4	25	0	0	2	2

Figure 18—The cottonwoods in this photo, taken at the Sevilleta site 2 years after the fire, were burned by a mixed-severity fire and topkilled, but they retained bark and fine branches. The understory vegetation includes native and nonnative woody plants that resprouted after the fire.

with less severity than the 3-4-6 site (fig. 19) and the Banco site. The sample sizes for Goodding's willow, screwbean mesquite (*Prosopis pubescens*), white mulberry (*Morus alba*), and Siberian elm (*Ulmus pumila*) were too small to compare rates of topkill among sites. Earlier studies along the Middle Rio Grande found that topkill of cottonwoods was 100 percent where trees are burned with high severity, 78 percent to 100 percent under moderate severity, and 52 percent to 70 percent under low severity (Ellis 2001; Stuever 1997). Topkill of cottonwoods ranged from 52 percent to 89 percent at four additional wildfire sites monitored by Johnson and Merritt (2009) along the Middle Rio Grande. Stromberg and Rychener (2010) estimated that 58 percent of cottonwoods were topkilled by fire along the unregulated San Pedro River. These estimates indicate that wildfire will result in topkill of 50 to 100 percent of cottonwoods, with the percentage largely determined by fire intensity, which is itself influenced by a number of factors (Ellis 2001).

Figure 19—The cottonwoods in this photo, taken at the 3-4-6 site 2 years after the fire, were burned by a high-severity fire and topkilled and quickly shed their bark.

Basal Sprouting

At our Middle Rio Grande study sites, basal sprouting was the primary recovery mechanism for cottonwoods, saltcedars, and Russian olives (figs. 20 and 21). Russian olives and saltcedars were more likely to have live basal sprouts than were cottonwoods (table 4). A higher percentage of cottonwoods had basal sprouts at the Banco site than at the 3-4-6 and Sevilleta sites (table 5). However, we examined a much larger sample of cottonwoods at the 3-4-6 and Sevilleta sites (n = 108 and 192, respectively) than at the Banco site (n = 4). Cottonwoods at the Sevilleta site were more likely to have live basal sprouts than cottonwoods at the 3-4-6 site, but there were similar percentages of trees with live or dead basal sprouts (58.5 percent at 3-4-6 and 63.5 percent at Sevilleta). Short-term survival of cottonwood basal sprouts was therefore greater at the Sevilleta fire (83.8 percent) than at the 3-4-6 fire (36 percent). Factors that varied between these sites, such as fire intensity, tree size, and season of fire, may explain this difference in short-term survival.

The percentage of cottonwoods with live basal sprouts at the Sevilleta site (53 percent) was within the range previously reported at fire sites to the south at Bosque Del Apache National Wildlife Refuge (45 percent and 61 percent; Ellis 2001) and to the north in Albuquerque (32 percent and 35 percent, Stuever 1997). The percentage with sprouts at the 3-4-6 site (21 percent) was lower than all sites.

Figure 20—Many top-killed trees, such as this cottonwood (right foreground) at the Sevilleta site, produced basal sprouts.

Figure 21—Russian olives (A) and saltcedars (B) recovered from wildfire via basal sprouting at the 3-4-6 and Sevilleta sites.

Table 5—Percentage of top-killed trees that had live basal sprouts in 2013 at three post-wildfire sites in our Middle Rio Grande study area.

	Site		
Species	**Banco**	**3-4-6**	**Sevilleta**
Rio Grande cottonwood	100	21	57
Saltcedar	96	85	na
Russian olive	86	76	75

This variation among sites may be explained by a number of variables including hydrological characteristics, such as depth to groundwater, flood frequency (Smith et al. 2009a), and fire intensity (Ellis 2001). Hydrological characteristics were similar between the Sevilleta and 3-4-6 sites, but fire intensity differed. Prior to the fire, the 3-4-6 fire had a higher density of large saltcedar, woody debris, and litter (Smith et al. 2007), which likely increased fire intensity relative to the Sevilleta site. The fires also burned during different times of the year, with the 3-4-6 burning in June and the Sevilleta burning in April. The trees at the 3-4-6 site were larger than those at the Sevilleta site, indicating that they were older. Large, older trees may allocate more resources to reproduction and less to sprouting, resulting in lower basal sprouting frequency and lower survival of basal sprouts (Hodgkinson 1998). Because basal sprout production was the primary mechanism of postfire recovery we observed, additional work is necessary to isolate the effects of hydrology, fire intensity, and tree age on basal sprouting by cottonwoods and other woody plants.

Epicormic Sprouting

Postfire epicormic sprouting has been observed in several broadleaved species, most notably among *Eucalyptus* species in Australia, but has been reported for fewer species in the United States (Meier et al. 2012). Five of the 192 top-killed cottonwoods examined at the Sevilleta site produced epicormic sprouts in the branches of their canopies. These trees retained intact bark and fine branches, indicating that they were burned with low severity. Two of the epicormic sprouting cottonwoods also produced basal sprouts (fig. 22). Cottonwoods burned in the Banco and 3-4-6 fires produced basal sprouts and root suckers, but not epicormic sprouts, probably because individuals were burned with greater severity than those at the Sevilleta Fire. The higher-severity fires likely destroyed aboveground meristematic tissues or depleted stores of non-structural carbohydrates, preventing trees at these sites from producing epicormic sprouts (Clarke et al. 2012). Stromberg and Rychener (2010) noted epicormic sprouting among Fremont cottonwoods along the San Pedro River, where, as at our Sevilleta site, most of the postfire sprouts were basal and a small percentage were epicormic. We do not know if the epicormic sprouts appeared in 2011 or 2012 at the Sevilleta site, but the amount of time between top-kill and epicormic sprouting should be measured so that removal of snags for salvage or firewood can be delayed long enough to prevent loss of canopy that may be restored by epicormic sprouting. Additional monitoring is also needed to identify characteristics of cottonwoods that produce epicormic sprouts and determine if sprouts on cottonwoods become reproductively mature faster than basal sprouts. If epicormic sprouts produce seeds within a few years of fire, these trees may aid the recovery of cottonwood forests.

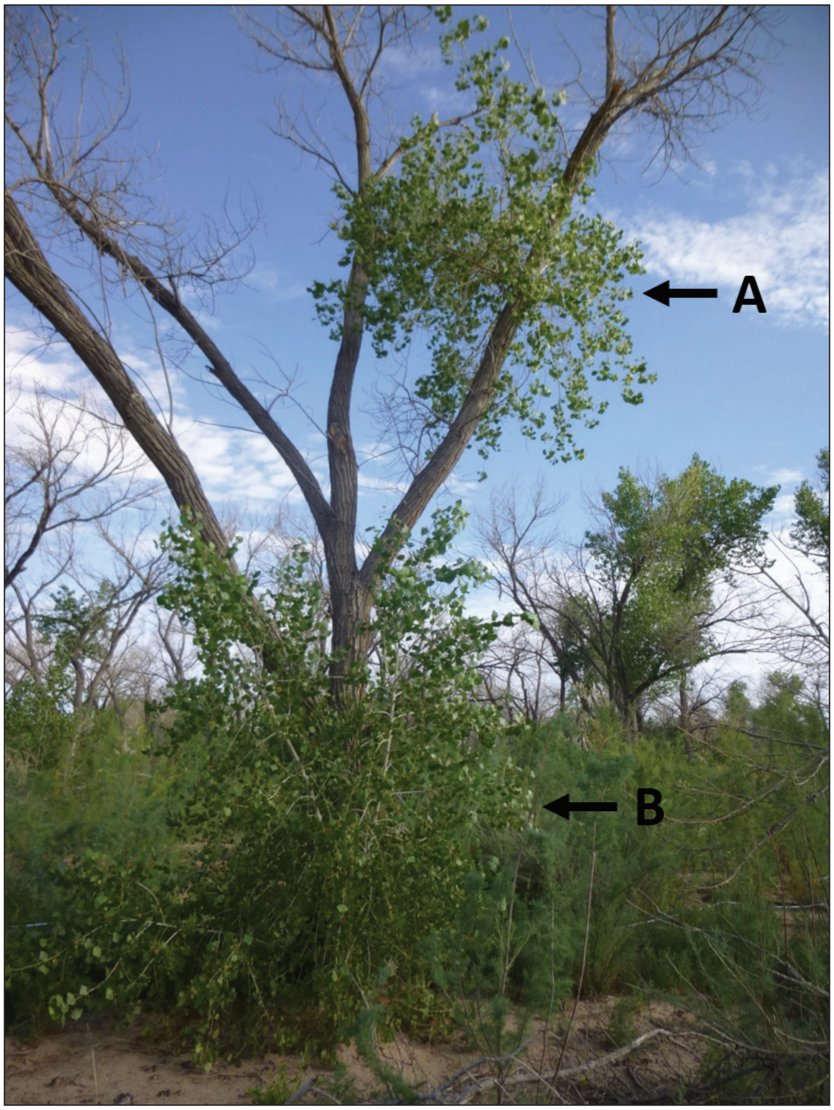


Figure 22—This cottonwood was topkilled by the Sevilleta fire in 2011, but had produced (A) epicormic sprouts and (B) basal sprouts by the time this photo was taken in 2013.

Root Suckering

Root suckers were present at each post-wildfire site (fig. 23). Overall, root sucker abundance was greatest for saltcedar, intermediate for cottonwood, and lowest for Russian olive (table 4). Cottonwood root sucker density was greatest at the Banco site, saltcedar root sucker density was greatest at the 3-4-6 site, and Russian olive root suckers were present only at the 3-4-6 site (table 6). Root sucker production is often an effective postfire regeneration mechanism because underground meristematic tissues and carbohydrate reserves are protected from fire and can produce suckers that will outgrow competing seedlings (Bond and Midgley 2001; Keyser et al. 2005). Unlike basal sprouts, which emerge in the first months after a fire, root suckering may continue in subsequent seasons and may be stimulated by precipitation or flood (Ellis 2001).

Figure 23—In addition to basal sprouts, we found root suckers produced by cottonwoods, saltcedars, and other woody species at post-wildfire sites.

Table 6—Density (number per m^2) of root suckers at three post-wildfire sites in our Middle Rio Grande study area.

	Site		
Species	**Banco**	**3-4-6**	**Sevilleta**
Rio Grande cottonwood	0.09	0.01	0.004
Saltcedar	0.02	0.08	0.01
Russian olive	0	0.01	0

Long-term studies, however, are needed to verify these suggestions. We found root suckers at each site, but basal sprout density was greater than root sucker density at two of the three sites. Root sucker density was greatest at the mesic Banco site, where ideal hydrological conditions may have encouraged continuous sprouting and growth in the 5 years after the fire occurred. Ellis (2001) also found greater numbers of cottonwood root suckers at a mesic wildfire site that burned with mixed intensity. Closer monitoring of additional wildfire sites is needed to better understand the phenology of root sucker production and survival.

Postfire Germination

The Banco site, which was burned in March of 2008 and partially flooded in June of that year, was the only location where we observed postfire germination of woody plants in either section. We found cottonwood saplings in a sampling plot approximately 10 m from the active channel. These saplings were growing in a band parallel to the active channel and were more than 5 m away from the nearest adult cottonwoods. During an earlier survey in June of 2008, 2 months after the burn, we observed cottonwood seeds settling in moist soil at this site (fig. 24). At this time, a flood pulse had been released from Cochiti Reservoir north of Albuquerque (fig. 25). The source of seeds germinating in the post-wildfire site was an adjacent unburned stand of mature cottonwoods. The only other woody species that had apparently germinated at this site were white mulberry (*Morus alba*) and Siberian elm (*Ulmus pumila*; table 4).

Figure 24—We observed postfire germination of cottonwoods at the Banco wildfire site after cottonwood seeds settled in moist soil 3 months after the fire in 2008 (A). We found cottonwood saplings in the same area in 2009 (B) and in 2012 (C).

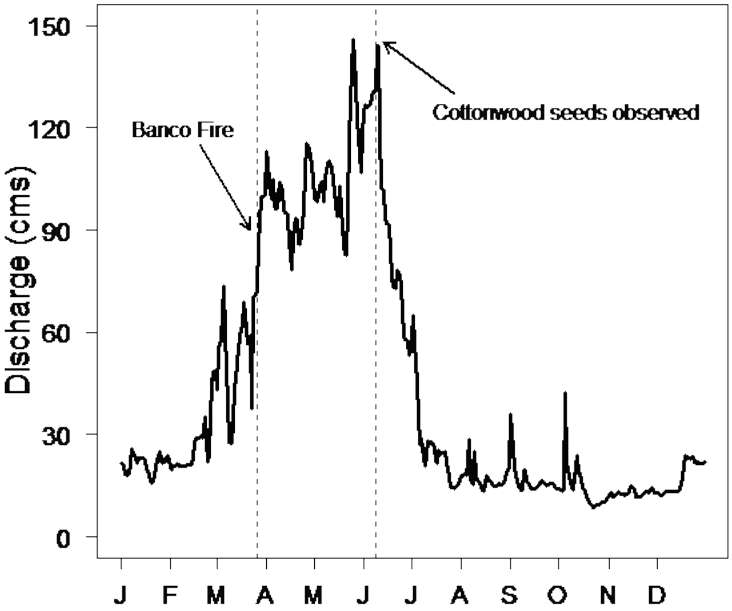


Figure 25—Discharge of the Middle Rio Grande, measured at the Albuquerque gauge site, exceeded a flood threshold after the Banco Fire occurred, coinciding with the dispersal of cottonwood seeds we observed at the site in 2008. Data were obtained from the USGS National Water Information System database (http://waterdata.usgs.gov/nwis).

We have never observed cottonwood germination in unburned areas of the north and south study sections. In this portion of the Middle Rio Grande, the continuous canopy, dense understory, and dampened flood pulse prevent seedling establishment (Howe and Knopf 1991; Molles et al. 1998). Ellis (2001) also observed saplings in post-wildfire sites along the Middle Rio Grande that were flooded within 2 years of being burned. Our observations of saplings in post-wildfire sites indicate that fire stimulates cottonwood recruitment by opening the canopy and removing vegetation and litter from the soil surface. This disturbance creates sites where, if flooded, germination will occur. A combination of fire and low-magnitude flooding can therefore act as a replacement for high-magnitude flooding and stimulate cottonwood germination along this regulated stream.

Summary: Impacts of Wildfire on Aridland Riparian Ecosystems

We demonstrated that cottonwoods are vulnerable to topkill but can regenerate through multiple mechanisms under ideal conditions. In most respects, however, nonnative Russian olives and saltcedars were superior to cottonwood in their ability to recover from fire. In xeric portions of the forest, such as the Sevilleta and 3-4-6 sites, low basal sprouting and root suckering rates of cottonwood will result in much of the area currently occupied by cottonwoods being replaced by saltcedar and Russian olive. In mesic areas, such as the Banco site, cottonwoods may increase in number because of heavy root suckering and postfire germination. Mesic areas like this are rare in our study reach (Molles et al. 1998), however, and continued occurrence of wildfire will likely reduce the extent of cottonwood stands in xeric locations. Wildfire may accelerate a conversion from native to nonnative cover along regulated sections of streams, but effects could differ along unregulated sections where timing and magnitude of peak discharges are not as heavily altered.

Chapter 3. Use of Woody Plants by Riparian-Nesting Birds in Unburned Plots and Post-Wildfire Sites

Introduction: Effects of Wildfire on Breeding Birds

With their dependence on vegetation for nesting and foraging sites, and relative ease of monitoring, breeding birds are an ideal group through which we can evaluate the effects of disturbances on animal communities. Wildfire disturbance in particular has a profound influence on breeding bird habitat in the western United States. In ponderosa pine (*Pinus ponderosa*) forests, for example, wildfire changes the composition and structure of woody vegetation by increasing snag density, opening space in the forest canopy, and stimulating growth of shrubs (Chambers and Mast 2005; Hutto et al. 2008; Kotliar et al. 2007). These changes benefit snag- and shrub-associates and aerial insectivores, but reduce the abundance of birds that require vegetation in the canopy and litter on the forest floor (Saab and Powell 2005).

In the American Southwest, wildfire effects have been documented for birds nesting in coniferous forests, grasslands, and shrublands, but few studies have been conducted in floodplain riparian forests (Bock and Block 2005). As we have shown in the previous chapter, wildfire has the potential to dramatically influence the composition and structure of riparian forests. These changes undoubtedly affect riparian vertebrate communities, such as breeding birds, but the nature of these effects remains largely unknown.

The riparian forest along the Middle Rio Grande in central New Mexico forms an extensive zone of productive habitat for riparian-nesting birds (Farley et al. 1994; Smith et al. 2009a). Though short-term effects of fire have been described for arthropod communities and some birds in this forest (Bess et al. 2002; Smith et al. 2006a,b, 2007, 2012), less is known about response to fire by the greater bird community. A deeper understanding of responses to fire in floodplain systems is necessary to help managers identify habitat features and sensitive species that require protection under the current disturbance regime. In this exploratory analysis, we document the use of nest plants by riparian-nesting birds in unburned and post-wildfire sites to determine which species may be positively or negatively affected by fire-induced changes.

Nest-Plant Use in Unburned and Postfire Riparian Sites

Study Sites and Methods

We searched for nests along the Middle Rio Grande in central New Mexico as part of a wider study examining effects of fuel reduction and wildfire on riparian organisms. For this comparison, we analyzed data from nine unburned plots and six post-wildfire sites (fig. 26). Ten of the sites were on land managed by the Middle Rio Grande Conservancy District and two were on land managed by the U.S. Fish and Wildlife Service.

Figure 26—We searched for nests in (A) unburned plots, in (B) post-wildfire sites including the Chavez wildfire site shown here 1 year postburn, and in (C) 2 years postburn.

Our unburned plot data were collected from control plots or fuel reduction plots prior to their treatments. Control and fuel reduction plots were clustered in three blocks, with plots as close to one another in each block as possible to ensure that vegetation was similar prior to treatments (Smith et al. 2007). Post-wildfire data were collected in study sites that were established after portions of the forest were accidentally burned (Smith et al. 2007, 2012). Mean plot size was 22.9 ha for unburned plots and 18.4 ha for post-wildfire sites (table 7).

Table 7—Characteristics of plots searched for nests along the Middle Rio Grande.

Plot	Type	Year burned	Years searched	Area searched (ha)	Ownership[a]
Middle 1	Unburned	NA	2000–2003	19.4	MRGCD
Middle 2	Unburned	NA	2000–2004	29.2	MRGCD
Middle 3	Unburned	NA	2000–2003	13.2	MRGCD
Middle 4[b]	Unburned/Wildfire	2002	2000–2008	9.4	MRGCD
Middle 5	Wildfire	2000	2000–2003	22.3	MRGCD
Middle 6	Wildfire	2000	2000–2008	31.4	MRGCD
Middle 7[b]	Unburned	NA	2002–2008	35.0	MRGCD
San Francisco	Wildfire	2003	2005–2008	19.0	MRGCD
South 1	Unburned	NA	2000–2004	28.9	MRGCD
South 2	Unburned	NA	2000–2002	15.6	MRGCD
South 3	Unburned	NA	2000–2002	26.7	USFWS
South 4	Unburned	NA	2000–2002	15.5	USFWS
South 5	Wildfire	1996	2000–2007	22.1	USFWS
South 6	Wildfire	1996	2000–2006	6.1	USFWS

[a] Ownership: MRGCD = Middle Rio Grande Conservancy District, USFWS = U.S. Fish and Wildlife Service.

[b] Middle 4 was initially established as a control plot but was monitored as a wildfire site after burning in 2002. Middle 7 was established in 2002 as a control plot replacement for Middle 4.

Crews visited unburned plots and wildfire sites daily to systematically search for nests of all species encountered. Nest searches were conducted from late April through August each year from 2000 to 2008. At least once per week, crew members walked throughout each plot to locate nests by following adults carrying material or food, incidentally flushing adults from nests, or listening for begging sounds of nestlings. We recorded nest locations with a handheld GPS and revisited each nest when it was no longer active to record nest plant species and condition (live or snag) and measure nest height, using a clinometer where necessary. We described nest substrate as "fallen debris" if the nest was constructed on fallen trunks or branches.

We compared nest substrate use among birds grouped into three nesting guilds. We assigned species that excavate their own nest cavities into the excavator guilds. Species that do not excavate their own nest cavities were assigned to the secondary cavity-nesting guild. We separated open-nesting species into two guilds based on their nest placement within riparian forest strata. If a species' mean nest height was lower than 10 m and minimum nest height was lower than 3 m, we assigned that species to the subcanopy/shrub guild. We assigned species to the canopy guild if their mean nest height was higher than 10 m and minimum nest height was higher than 3 m.

General Nest Plant Use

We summarized data from 1,321 nests of 39 landbird species (table 8). In unburned plots, 40 percent of nests were constructed in live cottonwoods, with another 40 percent in the two most abundant nonnative trees, Russian olive and saltcedar (fig. 27). Following wildfire, cottonwood and Russian olive were used for smaller percentages of nests, but saltcedar use was similar between unburned plots and post-wildfire sites (fig. 27). Wildfire appeared to increase the use of cottonwood snags, but it did not affect use of Russian olive snags or saltcedar snags (fig. 27).

Specific Nest-Plant Use

In post-wildfire sites, the species that nested in snags most frequently was Western Kingbird (scientific names in table 8), which constructed 65 percent of its open-cup nests in the leafless canopies of postfire cottonwood snags (fig. 28). This was also the only canopy-nesting species to use snags in unburned plots. Kingbirds prefer to nest in exposed sites, from which they can see potential competitors or predators and actively defend their territories (Gamble and Bergin 2012), making cottonwood snags an ideal substrate for this species. The frequent use of nest sites with high exposure to the sun by Western Kingbird indicates physiological adaptations to heat by adults, eggs, and nestlings. Another four canopy-nesting species and six subcanopy/shrub species nested in post-wildfire snags, showing that snags are a useful resource for these guilds following fire.

Table 8—Bird species monitored at unburned (UB) and post-wildfire (WF) sites.

Species	Nest guild	Nests monitored number		Cottonwood use		Russian olive use		Saltcedar use		Snags use		Fallen debris use	
		UB	WF	UB	WF	UB	WF	UB	WF	UB	WF	UB	WF
				Percent									
Cooper's Hawk *Accipiter cooperii*	Canopy	18	4	100	100	0	0	0	0	0	0	0	0
Swainson's Hawk *Buteo swainsoni*	Canopy	4	3	100	67	0	0	0	0	0	33	0	0
Great Horned Owl *Bubo virginianus*	Canopy	3	3	100	67	0	0	0	0	0	33	0	0
Western Wood-Pewee *Contopus sordidulus*	Canopy	7	10	100	50	0	0	0	0	0	50	0	0
Western Kingbird *Tyrannus verticalis*	Canopy	8	53	63	23	13	0	0	0	13	76	0	0
American Crow *Corvus brachyrhynchos*	Canopy	1	0	100	—	0	—	0	—	0	—	0	—
Common Raven *Corvus corax*	Canopy	2	2	100	50	0	0	0	0	0	50	0	0
Summer Tanager *Piranga rubra*	Canopy	13	6	77	100	8	0	0	0	0	0	0	0
Ladder-backed Woodpecker *Picoides scalaris*	Excavator	3	1	67	0	0	0	0	0	33	100	0	0
Downy Woodpecker *Picoides pubescens*	Excavator	28	7	86	43	0	0	0	0	14	57	0	0
Hairy Woodpecker *Picoides villosus*	Excavator	7	5	71	20	0	0	0	0	29	80	0	0
Northern Flicker *Colaptes auratus*	Excavator	17	24	59	21	0	0	0	0	41	79	0	0
Black-capped Chickadee *Poecile atricapillus*	Excavator	5	0	100	—	0	—	0	—	0	—	0	—
Barn Owl *Tyto alba*	Secondary cavity	0	1	—	0	—	0	—	0	—	100	—	0
American Kestrel *Falco sparverius*	Secondary cavity	4	7	75	0	0	0	0	0	25	100	0	0
Ash-throated Flycatcher *Myiarchus cinerascens*	Secondary cavity	28	13	61	15	4	0	0	0	36	85	0	0
White-breasted Nuthatch *Sitta carolinensis*	Secondary cavity	4	7	50	0	0	0	0	0	50	100	0	0
Bewick's Wren *Thryomanes bewickii*	Secondary cavity	28	13	46	15	4	0	4	0	32	85	4	0
Eastern Bluebird *Sialia sialis*	Secondary cavity	0	1	—	0	—	0	—	0	—	100	0	0

Table 8—(Continues).

Species	Nest guild	Nests monitored number		Cottonwood use		Russian olive use		Saltcedar use		Snags use		Fallen debris use	
		UB	WF	UB	WF	UB	WF	UB	WF	UB	WF	UB	WF
				Percent									
European Starling *Sturnus vulgaris*	Secondary cavity	6	4	67	0	0	0	0	0	33	100	0	0
Lucy's Warbler *Oreothlypis luciae*	Secondary cavity	1	0	100	—	0	—	0	—	0	—	0	—
Mourning Dove *Zenaida macroura*	Subcanopy/shrub	128	235	17	19	31	15	24	22	11	15	10	23
Greater Roadrunner *Geococcyx californianus*	Subcanopy/shrub	2	1	0	0	0	100	100	0	0	0	0	0
Black-chinned Hummingbird *Archilochus alexandri*	Subcanopy/shrub	256	111	32	22	40	16	20	16	3	9	0	1
Bushtit *Psaltriparus minimus*	Subcanopy/shrub	2	1	100	100	0	0	0	0	0	0	0	0
American Robin *Turdus migratorius*	Subcanopy/shrub	1	2	100	100	0	0	0	0	0	0	0	0
Gray Catbird *Dumetella carolinensis*	Subcanopy/shrub	1	2	0	0	100	50	0	0	0	0	0	0
Northern Mockingbird *Mimus polyglottos*	Subcanopy/shrub	6	2	0	50	17	0	50	0	0	0	0	0
Phainopepla *Phainopepla nitens*	Subcanopy/shrub	27	2	74	100	0	0	22	0	0	0	0	0
Common Yellowthroat *Corvus corax*	Subcanopy/shrub	0	1	—	0	—	0	—	0	—	0	—	0
Yellow-breasted Chat *Icteria virens*	Subcanopy/shrub	5	26	0	0	20	27	60	23	0	0	0	0
Spotted Towhee *Pipilo maculatus*	Subcanopy/shrub	15	7	0	0	0	0	7	29	0	0	0	0
Black-headed Grosbeak *Pheucticus melanocephalus*	Subcanopy/shrub	27	22	26	5	19	9	48	50	4	9	0	0
Blue Grosbeak *Passerina caerulea*	Subcanopy/shrub	29	31	7	3	7	13	86	68	0	0	0	0
Lazuli Bunting *Passerina amoena*	Subcanopy/shrub	0	1	—	0	—	0	—	0	—	0	—	0
Indigo Bunting *Passerina cyanea*	Subcanopy/shrub	1	1	0	0	0	0	0	100	0	0	0	0
Bullock's Oriole *Icterus bullockii*	Subcanopy/shrub	1	10	100	80	0	0	0	0	0	20	0	0
House Finch *Haemorhous mexicanus*	Subcanopy/shrub	5	1	80	0	0	0	0	0	20	100	0	0
Lesser Goldfinch *Spinus psaltria*	Subcanopy/shrub	6	2	17	50	0	0	83	0	0	50	0	0

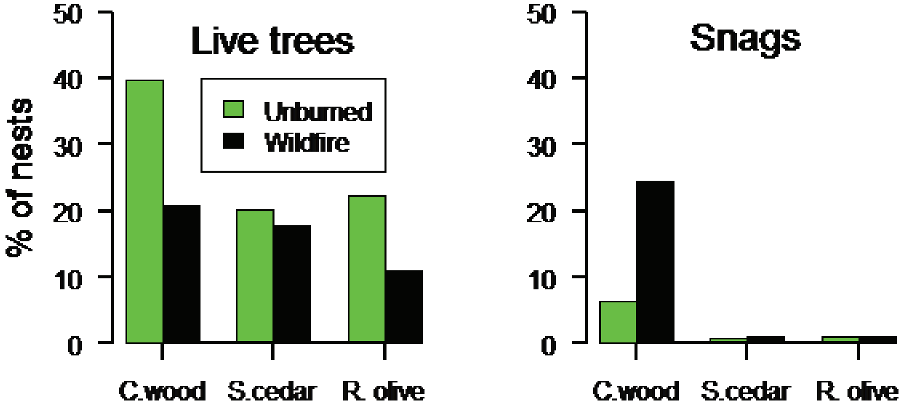


Figure 27—The percentage of nests found live cottonwoods was greater in unburned plots but the percentage in cottonwood snags was greater in wildfire sites. In addition, the percentage of nests found in Russian olive trees was greater in unburned plots.

Figure 28—Most of the Western Kingbird nests we observed were constructed in the canopies of cottonwood snags. This nest was constructed in a cottonwood topkilled by fire.

Snags have been identified as important nesting sites for cavity-nesting birds (Thomas et al. 1979). Though no cavity-nesting species responded to the postfire increase in snag density as strongly as Western Kingbird, the excavator and secondary cavity-nester guilds constructed a greater percentage of their nests in snags at post-wildfire sites than in unburned plots. Post-wildfire snags were used by all cavity-nesting species except for Black-capped Chickadee and Lucy's warbler, both of which forage by gleaning arthropods from live foliage (Foote et al. 2010; Johnson et al. 2012). Snag-associate species will likely occupy post-wildfire sites for 10 to 12 years after a fire, by which time most snags will have fallen to the ground (D.M. Smith, personal observation). In addition to snags, live cottonwoods were important nest sites for cavity-nesting birds. Unburned riparian plots had mature cottonwoods with dead branches and trunks that were frequently excavated as nest sites (fig. 29) and later used by secondary cavity nesters (Sedgwick 1997). By consuming dead wood and existing cavities, wildfire may actually reduce nest site availability for cavity-nesters in this forest.

Figure 29—In unburned plots, mature cottonwoods frequently had nesting cavities excavated in dead portions of trunks and branches.

Wildfire can reduce live canopy of Middle Rio Grande forest plots by up to 80 percent (Johnson and Merritt 2009). Cooper's Hawk, Summer Tanager, and other canopy-nesting birds require large cottonwoods or similar trees that support their nests (fig. 30) and meet their nest height preferences (Hunter et al. 1987; Smith and Finch 2014). The low percentages of canopy nests constructed in Russian olive and saltcedar in both unburned and post-wildfire plots (table 9) indicate that these trees will not provide adequate nests for most canopy nesters if cottonwood cover is reduced following fire. Canopy nesters that are unable to nest in snags will therefore avoid post-wildfire sites if there is no regeneration of the cottonwood canopy.

Figure 30—Cooper's Hawks constructed their large nests in the cottonwood canopy of unburned plots.

Table 9—Number of species and percentage of nests constructed in live trees, snags, and fallen debris in unburned (UB) and post-wildfire (WF) sites by birds in each nesting guild.

	Number of bird species		Cottonwood		Saltcedar		Russian olive		Snags		Fallen debris	
Guild	**UB**	**WF**	**UB**	**WF**	**UB**	**WF**	**UB**	**WF**	**UB**	**WF**	**UB**	**WF**
							Percent					
Canopy	8	7	89	40	0	0	4	0	2	59	0	0
Excavator	5	4	77	24	0	0	0	0	23	76	0	0
Secondary cavity	6	7	56	9	1	0	3	0	34	91	1	0
Subcanopy/shrub	16	19	28	19	27	24	30	15	5	11	3	12

Though wildfire removes canopy nest sites, abundance of understory nesting sites quickly increases after fire as a result of postfire sprouting and deposition of woody debris (Smith et al. 2012). Species in the shrub/subcanopy guild used cottonwood, Russian olive, and saltcedar with similar frequency in unburned plots but used saltcedar more frequently than cottonwood or Russian olive in post-wildfire sites (table 9). The higher use of saltcedar in post-wildfire sites occurred because vigorous sprouting of saltcedar produced dense stands that appealed to shrub-nesting birds (fig. 31). Shrub/subcanopy nesters used fallen debris more frequently in the post-wildfire sites, where top-killed trees begin to shed branches, trunks, and pieces of bark within the first years after fire (fig. 32). The species that used fallen debris most frequently was Mourning Dove, which constructed many nests on trunks, branches, and pieces of bark that had fallen from cottonwoods into resprouting trees and shrubs (Smith et al. 2012). Our observations of nest plant use suggest that Mourning Doves and other generalists appear to benefit the most from changes in forest structure that occur following wildfire. To fully understand the influence of wildfire on reproductive success, however, additional information is needed including effects of wildfire on food availability, thermal regimes, nest predation, and nest parasitism.

Figure 31—Blue Grosbeaks and other shrub-nesting birds constructed many nests in postfire saltcedar resprouts.

Figure 32—Within the first years following fire, top-killed cottonwood shed large pieces of bark and branches. The fallen debris was often suspended in resprouting vegetation, forming ideal nesting sites for Mourning Doves and other species. Nests were placed on top of pieces of debris and were obscured by the sprouted vegetation.

Summary: Impacts of Wildfire on Riparian-Nesting Birds

At the Middle Rio Grande and other streams, wildfire removes cottonwood canopy, creates snags and fallen debris, and induces resprouting of woody plants, especially saltcedar. These changes to forest structure and composition create nest sites for some bird species but also make the forest unsuitable for nesting by others. As postfire succession occurs, density of canopy-nesting birds will remain low if cottonwoods do not recover. In addition, cavity-nesting species will lose nest sites if snags are not replaced by mature trees. Projections of postfire cottonwood dynamics are therefore necessary to link characteristics of disturbances to habitat suitability for breeding birds.

Chapter 4. Climate change, Wildfire, and the Future of Aridland Riparian Ecosystems

Introduction: Climate Change and Hydrology of the American Southwest

Streamflows are susceptible to alterations resulting from anthropogenic climate change (fig. 33), the effects of which are expected to be especially severe in the American Southwest (Garfin et al. 2014; Seager et al. 2007). Under current and projected levels of CO_2 emissions, climatologists predict that increased temperatures and reduced snowpack will decrease the amount of snowmelt runoff that enters streams (Pierce et al. 2008; Seager and Vecchi 2010; Seager et al. 2013). Summer monsoons also cause flood events in many of the region's streams (Stromberg et al. 2010b). Changes in patterns of these storms are more difficult to model than changes in winter precipitation, but there is evidence that greenhouse warming will force monsoons to occur later in the year and with greater severity, further changing streamflow dynamics (Cook and Seager 2013; Serrat-Capdevila et al. 2013). Droughts, which are currently a fixture of the Southwest's climate, will continue to occur, but with increasing severity, resulting in further reductions in discharge volume (Cayan et al. 2010; Gutzler and Robbins 2011; Woodhouse et al. 2010). These changes will undoubtedly affect riparian ecosystems by limiting germination and increasing mortality of species dependent on floods and groundwater, to the benefit of generalist species, including nonnative trees

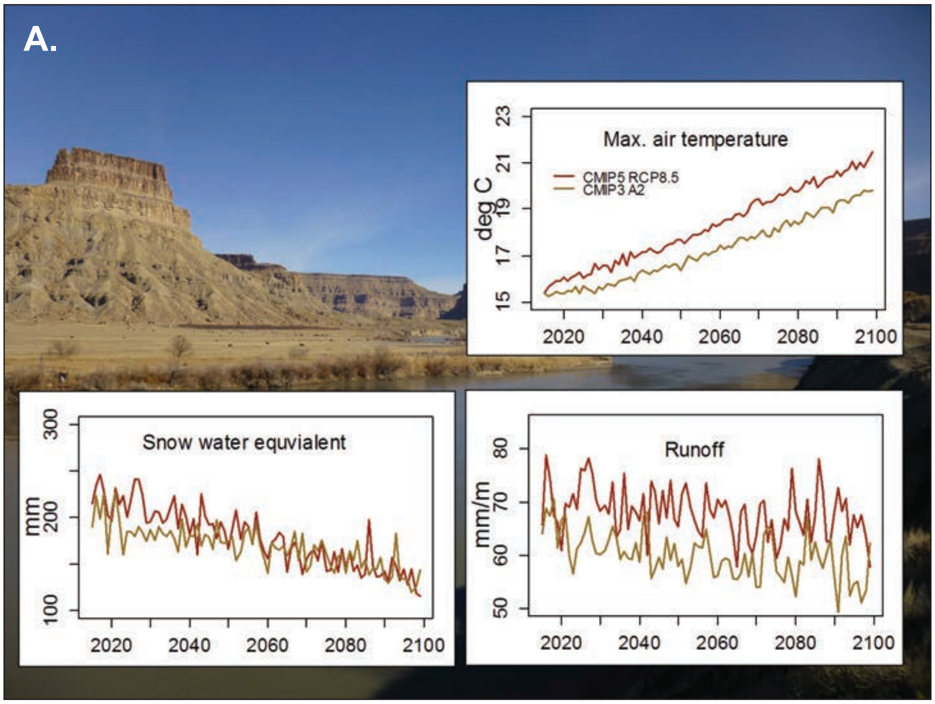


Figure 33—Mean projections from global climate models show increases in maximum air temperature and decreases in snow water equivalent and runoff for (A) the Colorado River Basin, (B) the Gila River Basin, and (C) the Rio Grande Basin (Reclamation 2016a,b).

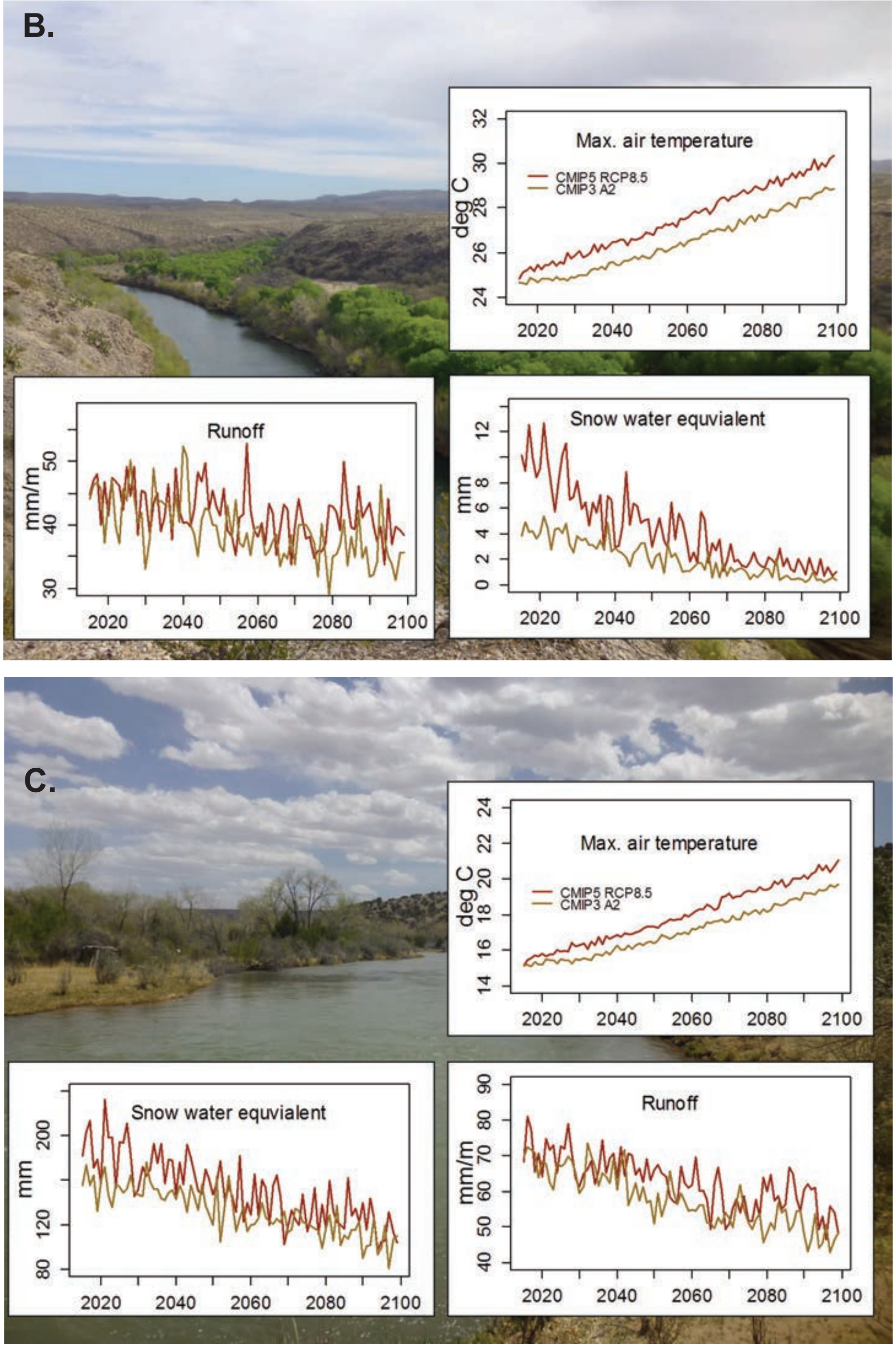


Figure 33—(Continued).

and shrubs, with characteristics such as seed viability and drought resistance that differ from native riparian obligates (Merritt and Poff 2010; Perry et al. 2012; Stromberg et al. 2010b). Because characteristics of discharge can determine rates of reproduction and survival of woody vegetation, we examined projected changes in peak magnitude and timing at the stream gauge sites described in Chapter 1. We then incorporated projections for the Rio Grande into a cottonwood population model. Using output from this model, we can predict the effects of climate change on riparian forest structure and breeding bird habitat.

Hydrological Projections

Methodology

We obtained projections of mean daily discharge for the stream gauge sites, made available by the U.S. Bureau of Reclamation (Miller et al. 2011). The projections use bias-corrected, spatial-downscaled precipitation and temperature data from the World Climate Research Programme Coupled Model Intercomparison Project phase 3 (CMIP3; Appendix A). We calculated averages from 36 projections of mean daily discharge from 15 global climate models run under the A2 emissions scenario for each of the 11 sites. We projected changes in magnitude and timing of peak discharge for the period of 2010 to 2099. To do this, we calculated the mean of each variable across the 36 model runs for the periods of 1980 to 2009, 2010 to 2039, 2040 to 2069, and 2070 to 2099. We then calculated the departure from the 1980 to 2009 mean for each of the three periods with future years. Using these values, we adjusted long-term observed means to reflect the projected changes.

Changes in Magnitude and Timing of Peak Discharge

Overall, models predicted greater changes in peak discharge date than in peak discharge magnitude (fig. 34). Though averaged model output predicted decreases in peak discharge magnitude at the Rio Grande, Rio Chama, and San Juan River, the Rio Grande was the only stream in which these decreases were projected by more than 50 percent of the model runs (table 10). At each Central Highlands stream, more than 50 percent of model runs predicted no change in peak discharge magnitude (table 10).

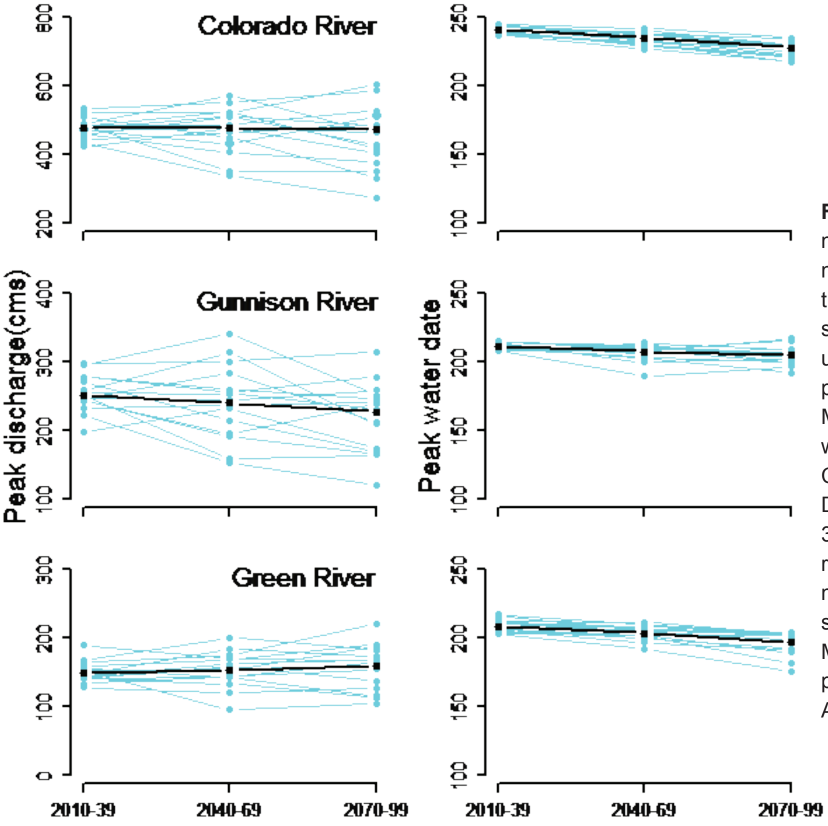
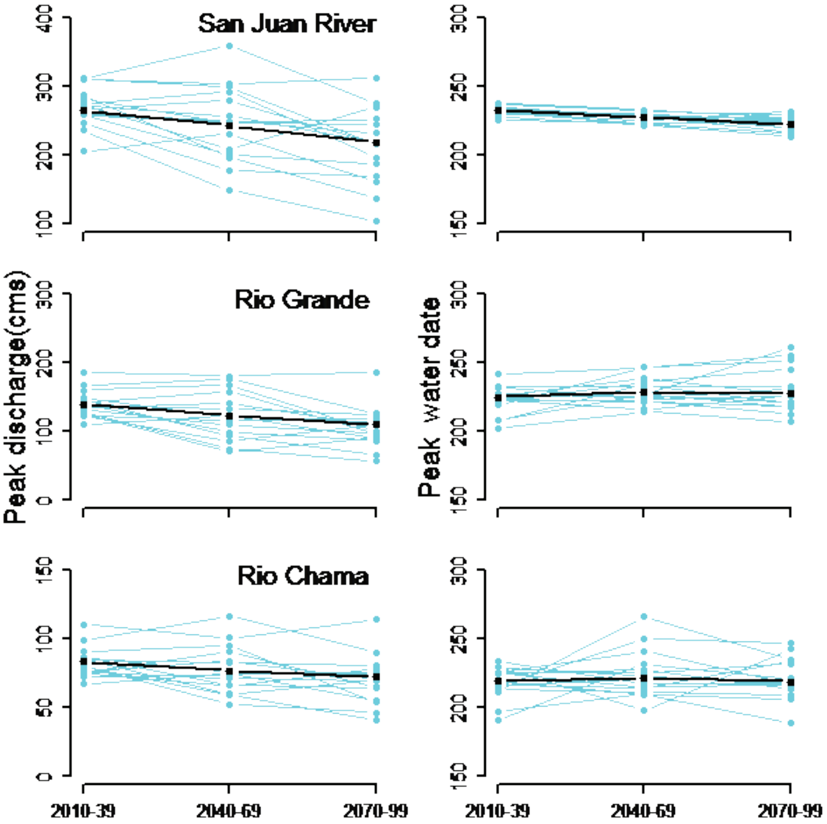


Figure 34—Projections of magnitude and timing of peak mean daily discharge under the A2 emissions scenario showed decreases in volume and increasingly early peak dates at several Rocky Mountain streams. Changes were less apparent at the Central Highland streams. Data are averaged for three 30-year periods. Black lines represent averages from 36 model runs; blue lines represent individual model runs. Methods for calculating these projections are described in Appendix A.



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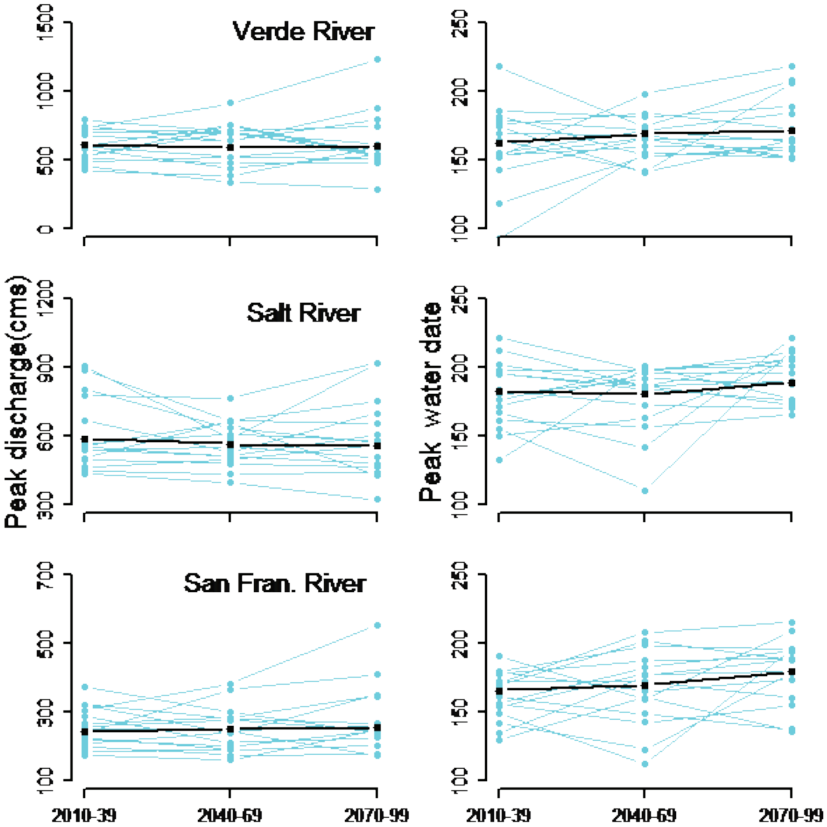

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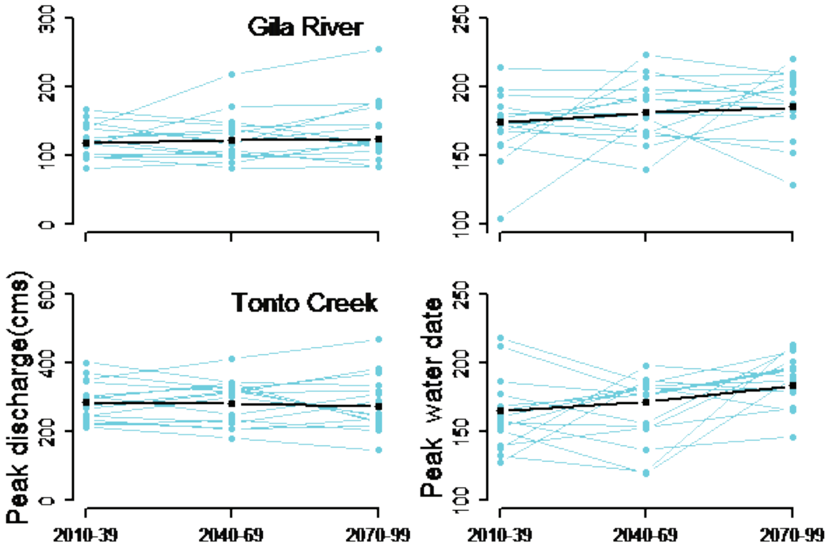

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Table 10—Percentage of 36 model projections indicating changes in peak discharge volume and date (ordinal day) of peak discharge at stream gauge sites.

	Peak discharge volume (m^3/second)			Peak discharge date (ordinal day)		
Gauge site	**Decrease**	**Increase**	**Null**	**Decrease**	**Increase**	**Null**
Colorado River at Cameo	28	14	58	100	0	0
Gunnison River at Grand Junction	33	3	64	83	0	17
Green River at Greendale	14	25	61	81	0	19
San Juan River at Bluff	47	3	50	94	0	6
Rio Grande at Otowi	53	3	44	36	6	58
Rio Chama at Abiquiu	25	3	72	31	0	69
Salt River at Roosevelt	14	8	78	8	3	89
Verde River above Horseshoe Dam	19	11	70	11	14	75
San Francisco River at Clifton	6	19	75	6	6	88
Gila River at Gila	5	17	78	8	8	84
Tonto Creek near Roosevelt	8	8	84	3	19	78

At least 80 percent of the model runs predicted an earlier peak discharge date at the Colorado River, Gunnison River, Green River, and San Juan River. Greater than 50 percent of model runs indicated no change in date for the Rio Grande basin streams and all of the Central Highland streams.

Peak discharges in Rocky Mountain streams typically result from snowpack runoff, which is strongly influenced by temperature. The predicted increase in temperatures would shrink the snowpack and accelerate snowmelt (Garfin et al. 2014; Pierce et al. 2008). As a result, peak discharges will on average occur earlier and with lower magnitude (Hurd and Coonrod 2008; Stewart et al. 2005). There is less model agreement on changes in amount of rainfall, which has a greater impact on timing and magnitude of peak discharge in the Central Highland streams (Ellis et al. 2008). Greenhouse warming may result in heavier storms occurring during the summer, fall, and winter periods, which would increase variability of peak discharges from year to year, especially at Central Highland streams, which flood readily when heavy rainfall occurs (Christensen et al. 2004; Hawkins et al. 2015). The models we examined did not show a directional change in peak discharge volume or timing at Central Highland streams, but variability in peak discharge volume will likely occur as a result of increased intensity of monsoons and Pacific frontal storms (Cook and Seager 2013; Garfin et al. 2014).

Modeling Changes in Cottonwood Populations

Stochastic Population Model Development

To make an initial assessment of how the above projected changes in hydrology will affect riparian ecosystems, we modified the stochastic cottonwood population model developed by Lytle and Merritt (2004) to predict future populations of cottonwoods at our study sections along the Middle Rio Grande (Appendix B). Lytle and Merritt (2004) used historical hydrograph data to simulate occurrence of floods and droughts, which are stochastic events that determine survival and reproduction. The original stage-structured model contained six classes: seedling, 1-year-old sapling, 2-year-old tree, 3-year-old tree, 4-year-old tree, non-reproductive adult (5- to 10-year-old tree), and reproductive adult (6 years or older). The model also contained stage-specific transition probabilities and a fecundity term, which was greater than 0 only in years when the hydrograph peaks above a flood threshold and the declining limb coincides with the seed dispersal period (fig. 35). Transition probabilities are based on survival, which is reduced when peak discharge volume is below a drought threshold (Lytle and Merritt 2004). Our modified model includes wildfire as an additional stochastic event, as well as the effects of regulation and climate change on cottonwood populations. We applied the model to our Middle Rio Grande study area by fitting it with vital rates, including top-kill and resprout probability, estimated from our observations and those from previous studies (Ellis 2001; Stuever 1997; Appendix B).

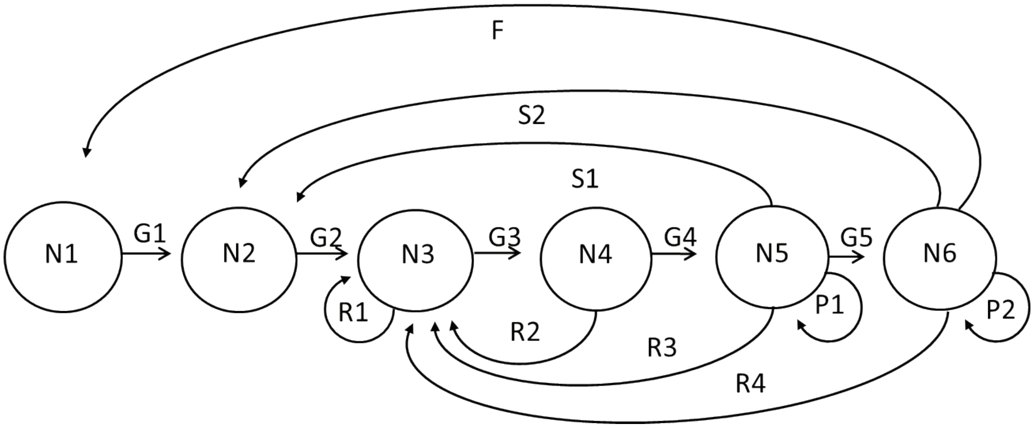

Figure 35—Life cycle model for cottonwoods along the Middle Rio Grande. Stages are seedlings (N1), 2- to 4-year-olds (N2-N4), non-reproductive adults (N5), and reproductive adults (N6). *N* is density, *G* is the probability of transition to the next stage, *P* is the probability of remaining in that stage, *R* is probability of retrogression from one stage to another as a result of postfire resprouting, *F* is fecundity through seedling germination, and *S* is fecundity through root suckering.

We used the model to project populations of trees over a 100-year period in each section, with flood, fire, and drought occurring as stochastic events. For each projection year, we adjusted magnitude and timing of peak discharge to reflect projected changes under the A1B, A2, and B1 greenhouse gas emission scenarios. We calculated 95 percent confidence intervals and range of stage class densities from 1,000 simulations under each emissions scenario at both sections of forest. We report changes in density of the youngest and oldest stage classes because mature cottonwoods are important nest sites for riparian-nesting birds and the presence of seedlings represents the occurrence of a reproductive event.

Cottonwood Projections

Our model predicted a sharp decline in density of mature cottonwoods under each emissions scenario in both sections (figs. 36 and 37). In most of the simulations (Appendix B), the decline in density was steeper for the south section where fire size was greater and flood frequency was lower than in the north section. There was a wider range of projected densities in the south section because of the greater number of seedling germination events that occurred following wildfire in the simulations. Range of projected seedling densities was also greater in the south section. Though typically

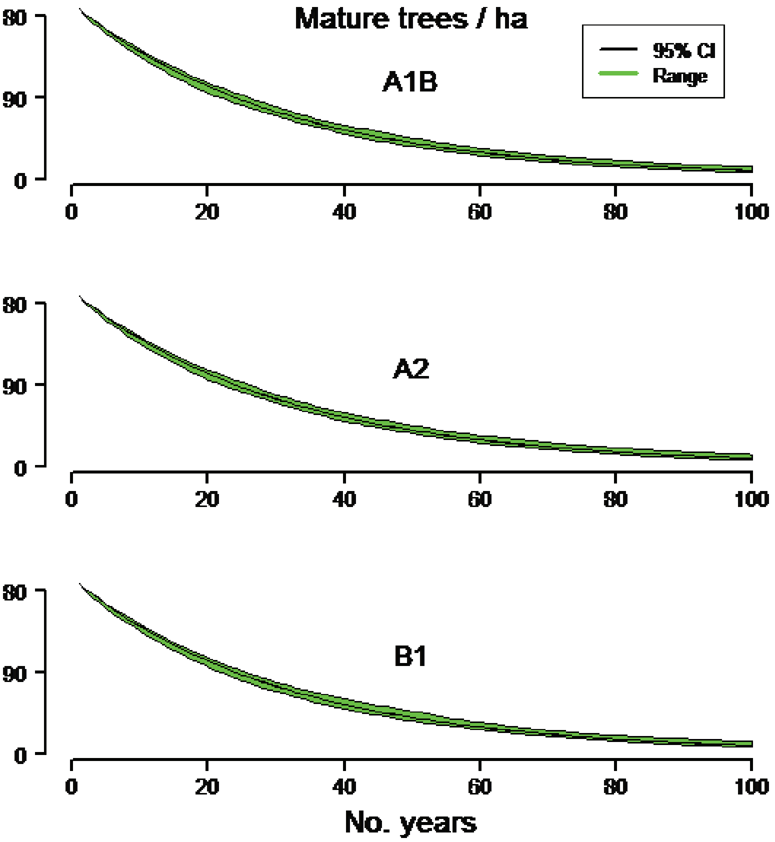

Figure 36—Density of mature cottonwoods decreased through the projection period, as shown by the 95 percent confidence intervals and ranges from simulated projections modeled under three emission scenarios in the north section of our Middle Rio Grande study area. The A1B scenario represented moderate increases in greenhouse gas emissions, A2 represented large increases, and B1 represented small increases.

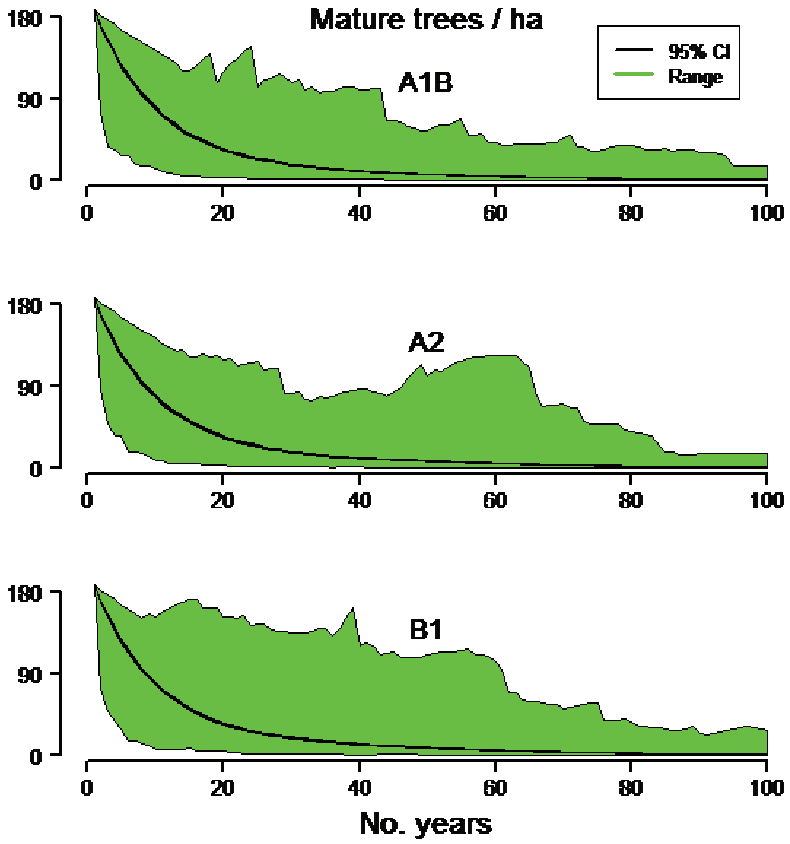

Figure 37—Density of mature cottonwoods decreased through the projection period under most simulations modeled under three emission scenarios in the south section of our Middle Rio Grande study area as shown by the 95 percent confidence intervals. The range of densities from simulated projections fluctuated because of occasional reproduction following wildfire and flood. The A1B scenario represented moderate increases in greenhouse gas emissions, A2 represented large increases, and B1 represented small increases.

rare, seedlings were present in at least one simulation of each projection year at the north section under all three emission scenarios (fig. 38). At the south section, the range of seedling densities decreased with time and, under the A2 emissions scenario, were absent after 80 years (fig. 39). These model results show that reproduction of cottonwoods will be possible under certain conditions at both sections, but decreases in discharge will make reproduction increasingly unlikely at the south section.

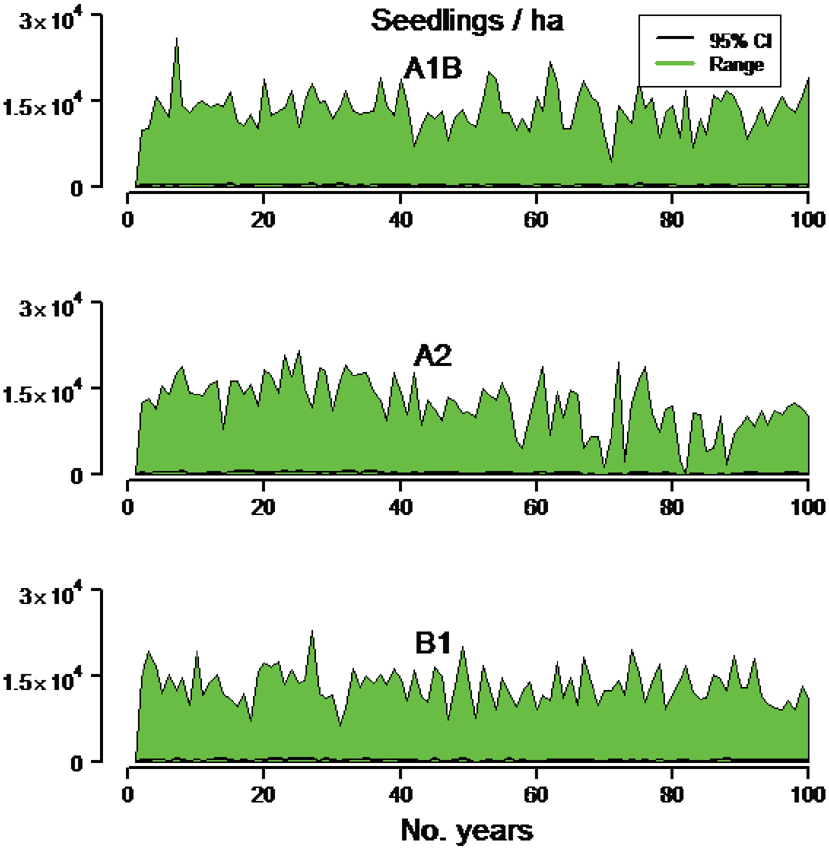

Figure 38—Density of cottonwood seedlings were similar under most simulations modeled under three emission scenarios in the north section of our Middle Rio Grande study area. The upper range of densities indicated that seedlings were present in at least one simulation in each projection year. The A1B scenario represented moderate increases in greenhouse gas emissions, A2 represented large increases, and B1 represented small increases.

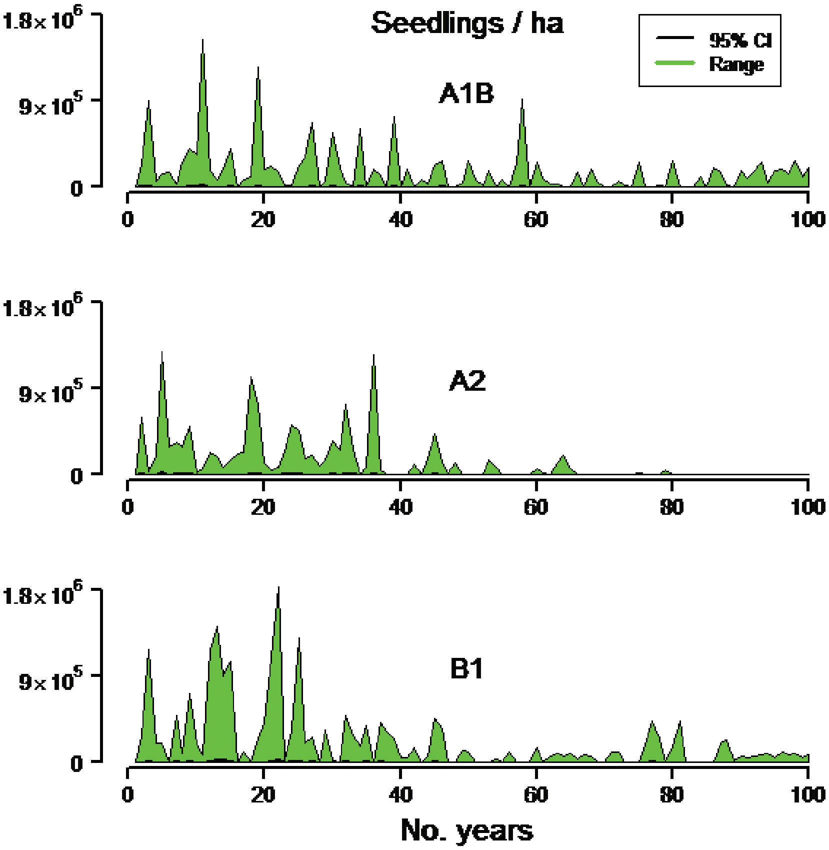

Figure 39—Projections of cottonwood seedling density differed among three emission scenarios in the south section of our Middle Rio Grande study area as shown by the 95 percent confidence intervals and range of densities. The upper ranges indicated that, as greenhouse warming increased, fewer seedlings were present in simulations, especially under the A2 scenario. The A1B scenario represented moderate increases in greenhouse gas emissions, A2 represented large increases, and B1 represented small increases. The range of densities was greater in the south section than in the north.

Management Implications

Our model output supports the contention that Middle Rio Grande cottonwood forests are in decline and will be largely replaced by other woody species by the end of this century (Howe and Knopf 1991; Molles et al. 1998). The decline in cottonwood density will be more rapid in the south section of our study area, which had lower flood probability and higher wildfire probability than the north section. Nonnative woody species, such as Russian olive and saltcedar, are present throughout our study area and will likely increase in abundance as cottonwood declines. Replacement of cottonwood by Russian olive and saltcedar will change the structure of the Middle Rio Grande riparian forest by increasing the density of low-stature vegetation and decreasing canopy height. Riparian-nesting birds will be affected as a result, with canopy-nesting birds including Cooper's Hawk and Summer Tanager immediately affected by loss of nest sites, followed by cavity nesters and snag-associates including Western Kingbird and Lucy's Warbler. Loss of cottonwoods would profoundly change the unique composition of riparian-nesting birds at the Middle Rio Grande and other streams, as most of the remaining birds will be widespread and generalist species such as Spotted Towhee and Blue Grosbeak (Smith and Finch 2014).

One of our most significant observations was the germination of cottonwoods following high severity fire and low-magnitude flooding in 2008. During years of heavy snowpack, managers can plan releases from Cochiti Dam to induce germination of cottonwoods in portions of the forest that have been burned or otherwise disturbed. Based on our observations from 2008, peak flows at Albuquerque should exceed 140 cms (5,000 cfs) and the hydrograph peak should occur in late May or early June. Additional monitoring of fire-and-flood events will be necessary to refine prescriptions for cottonwood germination and identify germination requirements for additional native species.

Though cottonwoods are a major element of this forest, multi-species or multi-guild models should be implemented to fully anticipate changes in forest structure and composition at this and other streams (Merritt et al. 2010). Arizona sycamore is not as sensitive to fire as cottonwood (Bock and Bock 2015), so changes in disturbance regimes will have effects that differ from the Rio Grande where this species is a component of the forest canopy. Additional information about the response to flood, fire, and drought by native and nonnative species is needed to better predict changes in the quality of habitat for riparian-nesting birds.

We found that there is substantial variation among southwestern streams in their natural hydrology, extent of regulation, and vulnerability to climate change. For this reason, projections of cottonwood populations must incorporate data specific to individual stream sites or groups of similar sites. In addition to considering climate change effects, models should also consider how hydrological patterns will be influenced by changes in water use by agricultural, municipal, and industrial sectors. Following the example of carbon emission scenarios, multiple water use scenarios should be developed to reflect the changes in the human landscape of the Southwest. Output from these models can then be used to set standards for surface flows necessary to maintain aridland riparian ecosystems in the region.

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Appendix A—Hydrological Projections

We obtained projections of mean daily discharge for each of these sites, which were made available by the USBOR (http://gis.usbr.gov/Streamflow_Projections/). These were made to project unregulated inflow to reservoirs from 1960 to 2099 (Miller et al. 2011). The projections use bias-corrected spatial downscaled precipitation and temperature data from the World Climate Research Programme Coupled Model Intercomparison Project phase 3 (Meehl et al. 2007). The precipitation and temperature data were incorporated into a National Weather Service River Forecasting System (NWS RFS) model, along with evapotranspiration demand, which was estimated using the Variable Infiltration and Capacity model, to produce hydrological projections (Miller et al. 2011). The projections, produced by the NWS RFS model, were bias-corrected by ensuring that means from the NWS RFS models, forced with observed temperature and precipitation data, had the same average as models forced with projected data across the same period. We calculated averages from 36 projections of mean daily discharge from 15 global circulation models (table A1) run under the A2 emissions scenario for each of the 11 stream sites. We did not use these projections to estimate future daily discharge because the amount of water that will be diverted for municipal, agricultural, and industrial use in the future is unknown. We instead examined changes to the volume of annual discharge, magnitude of peak discharge, and timing of peak discharge. We projected estimates of these variables for each year from 1980 to 2099. We used Akaike's Information Criteria (AIC) to determine whether changes in these variables over the

Table A1—Models used to project streamflow at gauge sites.

Model name	Modeling center or group	Number of projections
bccr_bcm2	Bjerknes Centre for Climate Research, Norway	1
cccma_cgcm3	Canadian Centre for Climate Modelling & Analysis	5
cnrm_cm3	Météo-France / Centre National de Recherches Météorologiques	1
csiro_mk3	CSIRO Atmospheric Research, Australia	1
gfdl_cm2	U.S. Dept. of Commerce / NOAA / Geophysical Fluid Dynamics Laboratory	2
giss_model_e_r	NASA / Goddard Institute for Space Studies	1
inmcm3_0.1	Institute for Numerical Mathematics, Russia	1
ipsl_cm4.1	Institut Pierre Simon Laplace, France	1
miroc3_2_medres	Center for Climate System Research (The University of Tokyo), National Institute for Environmental Studies, and Frontier Research Center for Global Change (JAMSTEC)	3
miub_echo_g	Meteorological Institute of the University of Bonn	3
mpi_echam5	Max Planck Institute for Meteorology, Germany	3
mri_cgcm2_3_2a.1	Meteorological Research Institute, Japan	5
ncar_ccsm3_0.1	National Center for Atmospheric Research	4
ncar_pcm1.1	National Center for Atmospheric Research	4
ukmo_hadcm3.1	Hadley Centre for Climate Prediction and Research / Met Office, UK	1

projection period were best described by regression models representing increases or decreases, or by intercept-only null models representing no change (Burnham and Anderson 2002). We analyzed output from each climate model at each gauge site by choosing the regression model with the lowest AIC value as the best representation of changes in annual discharge, peak discharge, and date of peak discharge. We determined the percentage of climate models predicting increases, decreases, or no change in each variable. If output from greater than 50 percent of the climate models fell into one of the above categories, we interpreted results as a predicted change or no change. If output from greater than 50 percent did not fall into any of the categories, we interpreted results as too uncertain to draw conclusions. To display changes in these rates, we calculated the mean of each variable across the 36 model runs for the periods of 1980 to 2009, 2010 to 2039, 2040 to 2069, and 2070 to 2099. We then calculated the departure from the 1980 to 2009 mean for each of the three future periods. We also estimated CVs of discharge volume, peak discharge magnitude, and peak discharge timing to evaluate the likelihood that variability of these characteristics will change in the future.

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Appendix B—Application of the Cottonwood Population Model

We compiled data from our study and others to modify the Lytle and Merritt (2004) stochastic cottonwood population model. We constructed the model, imported data into the model, and analyzed the output using a framework scripted in R statistical software (R Development Core Team 2011). We estimated cottonwood density in unburned stands using point center-quarter data collected by Johnson and Merritt (2009) at sample points spaced at 40-m intervals along five transects within or near our study sections. To include rates of postfire sprouting and germination of cottonwoods, we used the rates we estimated in the previous chapter, as well as rates from older burns along the Middle Rio Grande (Ellis 2001; Smith et al. 2009; Stuever 1997). We estimated the area of forest annually burned in each section by obtaining dates and locations of fires from New Mexico State Forestry and the Valencia and Socorro County fire departments for the 10-year period of 2002 to 2011. We visited each burn to mark the boundaries with GPS and we used Google Earth Pro to measure the area within the perimeter to the nearest 0.5 ha.

The original model contained six stage classes: seedling, 1-year-old sapling, 2-year-old tree, 3-year-old tree, 4-year-old tree, non-reproductive adult (5- to 10-year-old tree), and reproductive adult (6 years or older). The model also contained stage-specific transition probabilities and a fecundity term, which was non-zero only in years when floods of specified magnitude, duration, and time of year occurred. Density dependence is incorporated into the model through stage-specific self-thinning rates (Lytle and Merritt 2004). We added wildfire disturbance to the model by including two stage transitions (fig. B1). R_1 through R_4 represent the probability of basal resprouting after fire. We built this transition into the model as a retrogression to stage three because, at the end of the first postfire growing season, basal resprouts resemble 2-year-old cottonwood trees. The second transition we added was root-suckering, represented by S_1 and S_2.

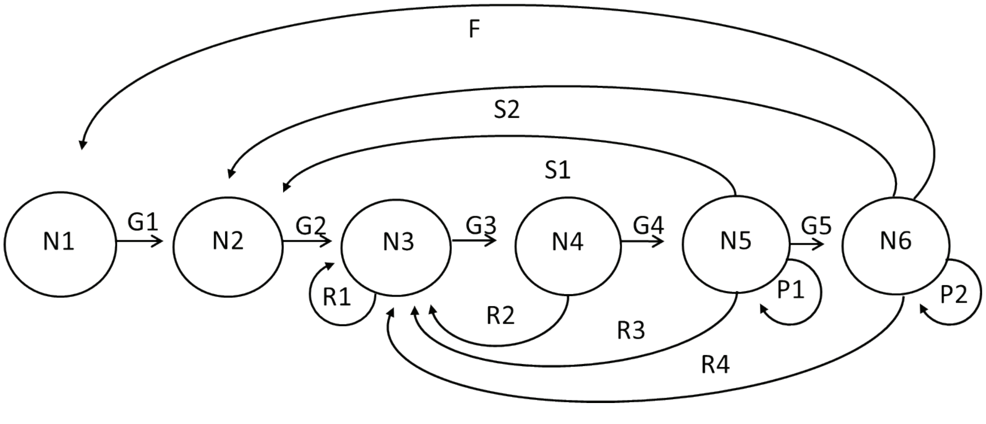

Figure B1—Life cycle model for cottonwoods along the Middle Rio Grande. Stages are seedlings (N1), 2- to 4-year-olds (N2-N4), non-reproductive adults (N5), and reproductive adults (N6). *N* is density, *G* is the probability of transition to the next stage, *P* is the probability of remaining in that stage, *R* is probability of retrogression from one stage to another as a result of postfire resprouting, *F* is fecundity through seedling germination, and *S* is fecundity through root suckering.

This transition is a fecundity term in which stage classes five and six produce cloned offspring in the second age class after fire. We assume that subadult and mature cottonwoods are the only stage classes with root structures capable of producing suckers. Root suckers begin in stage class two instead of one because they grow more rapidly than germinated seedlings. We found that basal sprout and root sucker production varied considerably among studies and study sites, so we programmed the model to sample rates of percent of trees producing basal sprouts and the number of root suckers per tree from the range of these values observed in our study and previous studies (table B1).

Table B1—Stage-specific variables that were components of population parameters for cottonwoods along the Middle Rio Grande in central New Mexico.

		Cottonwood stage classes, 1 = youngest stage, 6 = oldest stage					
Variable	**Description**	**1**	**2**	**3**	**4**	**5**	**6**
$N_i(0)$	Initial population density (cottonwoods/ha)	0	0	0	0	0	188
b_i	Self-thinning rate	0.029	0.10	0.91	0.66	0.2	–
a_i	Baseline transition probability	1	1	1	1	0.06	0.03
Fl_i	Flood mortality in area flooded	0.97	0.33	0.02	0.02	0.0	0.0
$HSwf_i$	Top-kill rate in area burned with high severity	1	1	1	1	1	1
$MSwf_i$	Top-kill rate in area burned with medium severity	1	1	0.96	0.95	0.75	0.85
LSwfi	Top-kill rate in area burned with low severity	1	0.9	0.8	0.7	0.4	0.6
Dr_i	Drought mortality in drought year	0.49	0.16	0.083	0.05	0.01	0.01
BS_i	Range of basal sprout rate for top-killed trees	0	0	0	0.33–1	0.33–1	0.33–1
RS_i	Range of root suckers produced per tree top-killed by high severity fire	0	0	0	0	0–6	0–6

We expressed the life cycle model as a projection matrix multiplied by the density of cottonwoods in each stage class (fig. B2). In the model code, the seedling fecundity value (F)—the number of germinated seeds that enter a study section during the growing season—is zero unless certain conditions are met. The model assumes that cottonwood seeds are present every year because seeds are transported throughout the forest by wind and water. Because of the current lack of scouring flow at our study sections, germination of these seeds only occurs in areas where litter and vegetation are removed by high-severity wildfire no more than 3 years before a flood (Ellis 2001). For each year the model was run, we calculated seedlings per section (F) using the following equation:

$$F_{(t)} = K_{(seed)}*Flood_{(t)}*Fire_{(t)} * Decl_{(t)}$$

where $K_{(seed)}$ is the maximum seedling abundance, $Flood_{(t)}$ and $Fire_{(t)}$ are the proportions of the floodway that were burned by high-severity fire during the last 3 years and flooded prior to the end of the current seed dispersal period, and $Decl_{(t)}$ is a function

$$N(t+1) = \begin{bmatrix} 0 & 0 & 0 & 0 & 0 & F \\ G1 & 0 & 0 & 0 & S1 & S2 \\ 0 & G2 & R1 & R2 & R3 & R4 \\ 0 & 0 & G3 & 0 & 0 & 0 \\ 0 & 0 & 0 & G4 & P1 & 0 \\ 0 & 0 & 0 & 0 & G5 & P2 \end{bmatrix} \quad X \quad \begin{bmatrix} N1 \\ N2 \\ N3 \\ N4 \\ N5 \\ N6 \end{bmatrix}$$

Figure B2—Projection matrix for cottonwoods along the Middle Rio Grande, New Mexico. Stages are seedlings, 2- to 4-year-olds, non-reproductive adults, and reproductive adults. *N* is density, *G* is the probability of transition to the next stage, *P* is the probability of remaining in that stage, *R* is probability of retrogression from one stage to another as a result of post-fire resprouting, *F* is fecundity through seedling germination, and *S* is fecundity through root suckering.

that determines the number of seedlings that germinate and survive the first growing season, based on timing of peak discharge and drawdown rate (Lytle and Merritt 2004). We selected flood thresholds based on our observations of flood extent and stream gage measurements from 2003 to 2013. We did not collect data on age class distributions at our study section, so we set an initial age distribution of entirely mature trees to reflect conditions reported in many areas (Howe and Knopf 1991). This distribution, however, is not necessarily reflective of the entire study area.

We used the model to project populations of trees over a 100-year period in each section, with flood, fire, and drought occurring as stochastic events that can occur during each time step of 1 year. Spatial domain of the model included the north and south study sections. During each time step, a section could experience drought, and all or parts of a section could experience flooding, or neither drought nor flooding could occur. To assign flood and drought status to years of each section's model projections, we used streamflow values from U.S. Geological Survey discharge data for the Middle Rio Grande at Albuquerque (USGS gauge number 08330000). We randomly sampled, with replacement, peak mean daily discharge from each year between 1975 (the year following the completion of Cochiti Dam) and 2012. We then adjusted the sampled magnitude and timing of peak discharge to reflect the projected change under three CMIP3 carbon emission scenarios. These scenarios were A1B, which represented moderate increases in emissions; A2, which represented large increases; and B1, which represented small increases.

For a given projection year, the section experienced drought if the sampled discharge value was below the section's drought threshold. The section would experience complete flooding if the peak discharge was above the threshold for the floodway to be fully inundated. Partial flooding of the section occurred if the sampled discharge was greater than the bankfull discharge threshold (threshold above which portions of the floodway are inundated), but below the full floodway inundation threshold. In this situation, we calculated the percent difference between the sampled discharge and the full inundation threshold to calculate the percentage of the section that was flooded that year.

In addition to hydrological status, we assigned a wildfire status to each section for each year of the projection period. In the model, portions of a section could be burned by wildfire, but when a given percentage of a section was burned, it could not be burned again until 5 years had passed, allowing fuel to reaccumulate. For each projection year, we first sampled a percentage of area burned, with replacement, from the 10-year historical record of the section. If a percentage greater than 0 was selected, we then determined what proportions of this percentage would be burned with high, medium, and light severity. We randomly generated three values between 0 and 1 that summed to 1. We multiplied the percent of area burned value by each random proportion to arrive at the percentage of the section to be burned by each severity class. Next, we multiplied the area available to be burned (this is the area within the section that had not been burned during the last 5 years) by the three severity percentages to produce the number of hectares that would be burned with high, medium, and light severity that year. We assumed that, during projection year one, neither section had burned during the previous 5 years. During each projection year, F is greater than 0 only in the portions of a section where high-severity fire occurred during that year or during the previous 2 years and flooding occurred during the current year. In addition to fecundity, the values of other stage-specific variables were influenced by flood, fire, and drought (table B2). During each projection year, the model code calculates the percentage of trees that will be affected by drought, flood, and fire. Transitions and stage abundances are then determined for the year. We conducted 1,000 simulated projections for each section and calculated 95 percent confidence intervals and range for stage-class densities during each year of the projection.

Table B2—Parameters used in projection models for populations of cottonwoods along the Middle Rio Grande in central New Mexico.

Parameter	Description	Components
F	Fecundity: Number of seedlings established in year t	Flood area, flood timing, stage height decline, high-severity wildfire area
G	Stage-specific progression probability	Stage-specific transition rate, self-thinning rate, flood mortality, wildfire mortality, drought mortality
P	Probability of remaining in stage class 5 or 6	Stage-specific transition rate, flood mortality, wildfire mortality, drought mortality
R	Probability of retrogression from stages 3–6 to stage class 3 after topkill by wildfire	Wildfire area, fire severity, top-kill rate, basal sprout rate
S	Number of root suckers produced by adult trees	Wildfire area, fire severity, root-suckering rate

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