WATER RESOURCES RESEARCH, VOL. 49, 1–18, doi:10.1002/wrcr.20256, 2013

Coupled hydrogeomorphic and woody-seedling responses to controlled flood releases in a dryland river

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Received 6 September 2012; revised 11 April 2013; accepted 14 April 2013.

[1] Interactions among flow, geomorphic processes, and riparian vegetation can strongly influence both channel form and vegetation communities. To investigate such interactions, we took advantage of a series of dam-managed flood releases that were designed in part to maintain a native riparian woodland system on a sand-bed, dryland river, the Bill Williams River, Arizona, USA. Our resulting multiyear flow experiment examined differential mortality among native and nonnative riparian seedlings, associated flood hydraulics and geomorphic changes, and the temporal evolution of feedbacks among vegetation, channel form, and hydraulics. We found that floods produced geomorphic and vegetation responses that varied with distance downstream of a dam, with scour and associated seedling mortality closer to the dam and aggradation and burial-induced mortality in a downstream reach. We also observed significantly greater mortality among nonnative tamarisk (*Tamarix*) seedlings than among native willow (*Salix gooddingii)* seedlings, reflecting the greater first-year growth of willow relative to tamarisk. When vegetation was small early in our study period, the effects of vegetation on flood hydraulics and on mediating flood-induced channel change were minimal. Vegetation growth in subsequent years resulted in stronger feedbacks, such that vegetation's stabilizing effect on bars and its drag effect on flow progressively increased, muting the geomorphic effects of a larger flood release. These observations suggest that the effectiveness of floods in producing geomorphic and ecological changes varies not only as a function of flood magnitude and duration, but also of antecedent vegetation density and size.

Citation: Wilcox, A. C., and P. B. Shafroth (2013), Coupled hydrogeomorphic and woody-seedling responses to controlled flood releases in a dryland river, *Water Resour. Res.*, *49*, doi:10.1002/wrcr.20256.

1. Introduction

[2] Riparian vegetation and morphodynamics can be tightly coupled along river corridors. Vegetation produces both below-ground and above-ground effects that strengthen banks, stabilize bars, trap sediment, and alter local hydraulics [*Petryk and Bosmajian*, 1975; *Hey and Thorne*, 1986; *Nepf*, 1999; *Micheli and Kirchner*, 2002; *Green*, 2005; *Corenblit et al*., 2007; *Schnauder and Moggridge*, 2009; *Rominger et al*., 2010; *Sandercock and Hooke*, 2010; *Dean and Schmidt*, 2011; *Edmaier et al*., 2011]. These effects can in turn influence channel pattern by, for example, promoting the evolution of anabranching [*Tooth and Nanson*, 2000] or single-thread, meandering channels instead of braided channels [*Williams*, 1978; *Gran and Paola*, 2001; *Tal and Paola*, 2007; *Braudrick et al*., 2009; *Davies and Gibling*, 2011]. Flow and sediment regimes, water availability, and channel morphology can strongly influence distribution patterns and population dynamics of woody riparian vegetation [*Hupp and Osterkamp*, 1996; *Scott et al*., 1996; *Cooper et al*., 2003; *Stella et al*., 2011]. Many investigations into hydrogeomorphic effects on riparian vegetation have focused on the establishment of pioneer riparian trees and the role of site creation, soil moisture, and the timing of floods compared to seed-dispersal timing [*Scott et al*., 1996; *Mahoney and Rood*, 1998; *Cooper et al*., 1999; *Stella et al*., 2006; *Asaeda et al*., 2011]. A more limited number of studies have examined how the subsequent fate of seedlings established within or near the active channel is affected by flood-induced scour, burial, or breakage [*Stromberg et al*., 1993; *Johnson*, 1994; *Auble and Scott*, 1998; *Friedman and Auble*, 1999; *Johnson*, 2000; *Levine and Stromberg*, 2001; *Dixon et al*., 2002; *Polzin and Rood*, 2006; *Asaeda and Rajapakse*, 2008].

[3] Feedbacks between morphodynamics and vegetation may be strongly time dependent, as a result of changes in vegetation characteristics, flow variability, and other factors. As vegetation grows with time since establishment, plants both exert a greater influence on physical processes and become more resilient to them. Increases in plant diameter and height produce greater vegetation drag [*Freeman et al*., 2000], reducing the proportion of total shear stress applied to grains on the bed and thereby reducing the

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0043-1397/13/10.1002/wrcr.20256

erosional effects of floods of a given magnitude. Time since establishment can also influence the strength of roots, their cohesive effect, and their resilience to scour [*Edmaier et al*., 2011]. The shifting strength and direction of feedbacks once vegetation becomes well established produces a "ratchet effect" whereby such vegetation becomes progressively more difficult to remove as it grows [*Tal et al*., 2004]. Seasonal variations (presence or absence of leaves) may also alter vegetation drag by increasing or reducing frontal area [*Freeman et al*., 2000]. A conceptual model proposed by *Corenblit et al*. [2007] suggests four stages in the temporal evolution of feedbacks: an initial period following a channel-setting flood in which geomorphic processes dominate; a second stage in which pioneer vegetation is recruited on bare surfaces; a third stage in which ecogeomorphic feedbacks are strongest; and a fourth stage in which vegetation is mature and dense enough that it strongly controls channel planform and is insensitive to all but the largest floods.

[4] Efforts to restore riparian zones and manage nonnative riparian vegetation would benefit from improved understanding of hydrogeomorphic effects on vegetation. Multiple species and hybrids of tamarisk (*Tamarix* spp., aka saltcedar), shrubs and small trees native to Eurasia, have become the dominant riparian vegetation along many rivers in the southwestern United States, often replacing native cottonwood-willow (*Populus-Salix*) woodlands [*Friedman et al*., 2005; *Nagler et al*., 2011]. Tamarisk invasions have altered riparian habitat [*Shafroth et al*., 2005] and contributed to reductions in channel width, increased sediment storage, and other geomorphic changes along western rivers [*Graf*, 1978; *Hereford*, 1984; *Allred and Schmidt*, 1999; *Grams and Schmidt*, 2002; *Birken and Cooper*, 2006; *Dean and Schmidt*, 2011]. Millions of dollars have been spent on controlling tamarisk by chemical, mechanical, and biological control methods [*Shafroth et al*., 2005], sometimes with unintended geomorphic consequences such as increased erosion [*Vincent et al*., 2009].

[5] Shifts in the magnitude and timing of peak flows and other anthropogenic alterations of flow regimes can favor tamarisk over native pioneer species [*Stromberg*, 2001; *Stromberg et al*., 2007; *Merritt and Poff*, 2010]. Comparisons of tamarisk and native pioneer trees (cottonwood and/or willow) at the seedling stage have found that tamarisk is more vulnerable to the effects of burial [*Levine and Stromberg*, 2001] and inundation [*Gladwin and Roelle*, 1998] and that cottonwood seedlings can outcompete tamarisk seedlings when grown in mixtures [*Sher et al*., 2000, 2002]. Tamarisk recolonizes rapidly following floods [*Shafroth et al*., 2005], however, and beyond the seedling stage, tamarisk becomes highly resilient to hydrogeomorphic forces as well as to drought [*Everitt*, 1980; *Cleverly et al*., 1997]. Identification of flows and fluvial processes that adversely affect tamarisk relative to native taxa could assist efforts to prescribe flows downstream of dams for managing tamarisk.

[6] Although to our knowledge, managed flood releases have not been used as an explicit tool for tamarisk control, manipulation of flow releases from dams by mimicking natural flows has been implemented to achieve other ecosystem objectives [*Poff et al*., 1997; *Arthington et al*., 2006; *Merritt et al*., 2010; *Konrad et al*., 2012]. Tools are lacking, however, for providing managers with site-specific, quantitative information about the magnitude, duration, frequency, rate of change, and timing of flows sufficient to achieve such objectives [*Rood et al*., 2005; *Palmer and Bernhardt*, 2006]. One avenue for developing the information to address these questions, and more generally to develop new insights into relationships between hydrogeomorphic processes and ecosystem functions, is conducting large-scale flow experiments [*Konrad et al*., 2011].

[7] Here we report on a series of dam-managed flood releases on a dryland river that we used as a multiyear flow experiment to investigate geomorphic changes, flood hydraulics, woody seedlings, and their interactions. At the outset of our study, we used a planned flood release to test the hypothesis that this event would cause greater mortality among a cohort of small (1 year old) nonnative tamarisk seedlings than among native willow (*Salix gooddingii*) seedlings, as well as to evaluate associated flood hydraulics and geomorphic changes. Our study evolved and broadened in subsequent years, as additional managed floods that varied in magnitude and duration were released in our study system, into an investigation of the evolving nature and strength of vegetation-hydrogeomorphic feedbacks as a result of spatial and temporal variations in coupled geomorphic and vegetation responses to controlled floods. We apply these inquiries to develop guidance regarding the effectiveness of flow prescriptions in dammed rivers, where high-flow releases are small compared to predam floods, in achieving downstream geomorphic and ecological objectives.

2. Study Area

[8] The Bill Williams River (BWR) historically flowed 65 km from the confluence of the Santa Maria River and Big Sandy River into the Colorado River in western Arizona, USA (Figure 1), draining 13,800 km^2 and alternating between canyon and alluvial valley reaches. Because the hydrology of the BWR is influenced by wetter conditions in its mountainous headwaters, where average annual precipitation exceeds 40 cm yr^{-1}, and arid conditions in the lower basin (12 cm yr^{-1} precipitation) [*Shafroth and Beauchamp*, 2006], we characterize the river as "dryland" rather than arid or semiarid. Both the upstream and downstream limits of the BWR are currently submerged within reservoirs. Alamo Dam, a U.S. Army Corps of Engineers flood-control facility that was completed in 1968 and impounds Alamo Lake, now forms the upstream limit of the BWR. At its downstream end, the BWR flows into Lake Havasu, an impoundment on the Colorado River that is the source for the Central Arizona Project Aqueduct and the Colorado River Aqueduct, which supply water to several large cities in the southwestern United States.

[9] Alamo Dam has substantially reduced peak flows in and sediment supply to the BWR (Figure 2). For example, the ratio of the postdam 2 year flood (Q_2) to the pre-dam Q_2 is 0.04, based on Log Pearson III analysis of peak-flow data from the U.S. Geological Survey BWR below Alamo Dam, AZ gauge (#09426000). This ratio, a metric known as Q^* [*Magilligan and Nislow*, 2005; *Schmidt and Wilcock*, 2008], provides an indication of the extent to which a dam

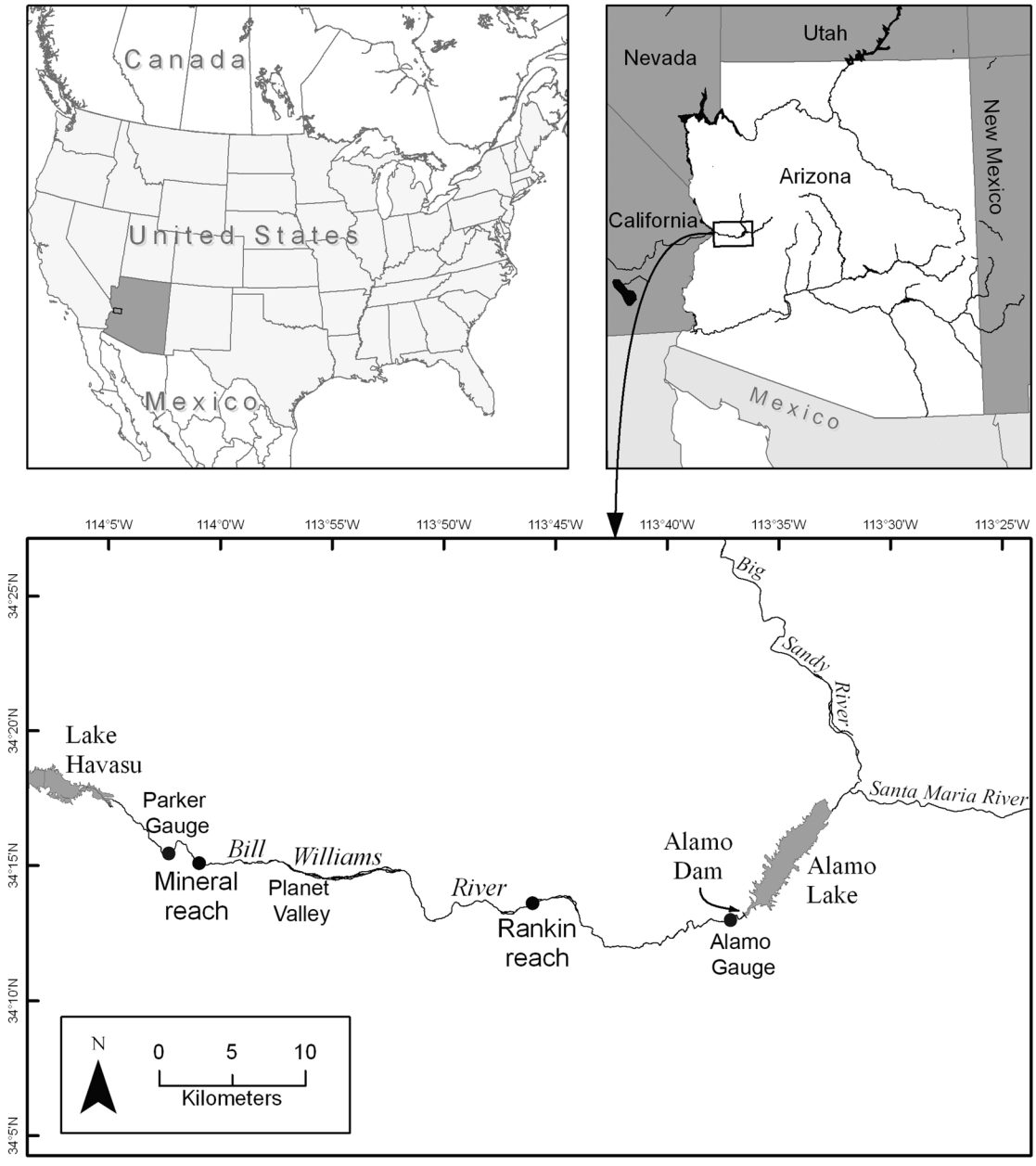

Figure 1. Bill Williams River in western Arizona, USA. The two study reaches identified in the bottom plot, Rankin and Mineral, are 18 and 48 km downstream of Alamo Dam, respectively (flow is from right to left).

has reduced transport capacity; the 0.04 value for the BWR is indicative of extreme peak flow reduction. The upper 85% of the basin's drainage area is effectively disconnected from the BWR by Alamo Dam, blocking the supply of bed-material from the upper basin. Further, there are no perennial tributaries downstream of Alamo Dam.

[10] The BWR's alluvial valleys, the largest of which is the 13 km long Planet Valley, exert a strong control on the routing of both flow and sediment through the BWR, causing gains and losses of surface flow and storing large volumes of sediment. The alluvial aquifer in Planet Valley (Figure 1) acts as a sponge, such that all of the river's base flow typically infiltrates at the upstream end of the valley and emerges at the downstream end, where valley width and depth to bedrock decline [*Jackson and Summers*, 1988; *House et al*., 2006]. Planet Valley and its antecedent water-table elevation also influence routing of high flows down the BWR [*Shafroth et al*., 2010; *Simpson et al*., 2013], as discussed further below.

[11] The severe reduction of both transport capacity and sediment supply in the BWR has been accompanied by the spread of tamarisk and other floodplain vegetation as well as by channel narrowing [*Shafroth et al*., 2002]. This channel narrowing trend started in the 1950s, before Alamo Dam was built [*Shafroth et al*., 2002], likely as a result of regional climatic shifts [*Sheppard et al*., 2002] that reduced peak flows along many rivers in the southwestern United States [*Hereford*, 1984].

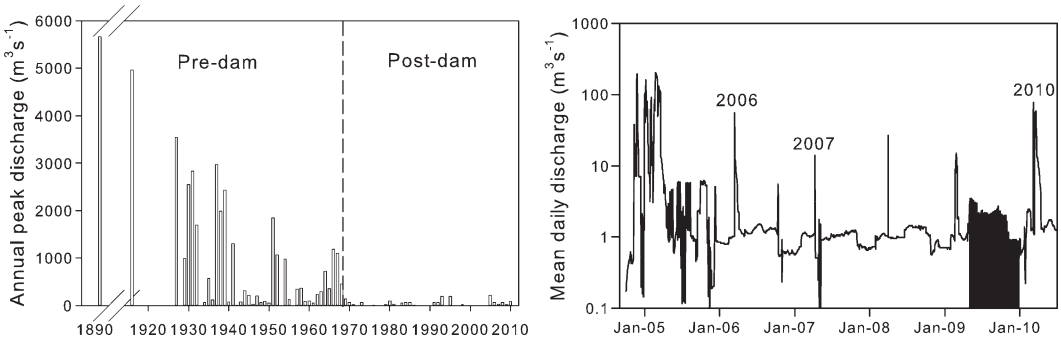

Figure 2. Flow data for the Bill Williams River below Alamo Dam gauge. Left plot shows annual peak discharges for the period of record and illustrates the postdam reduction of high flows. Right plot shows mean daily flows from late 2004 to mid-2010, with annotations of flood events around which our data collection was completed; note difference in *y* axis scale compared to left plot.

[12] Whereas many river corridors in the southwestern United States are dominated by nonnative tamarisk, the BWR has a diverse riparian flora that includes tamarisk but also Goodding's willow (*S. gooddingii*), Fremont cottonwood (*Populus fremontii*), seep willow (*Baccharis salcifolia*), arrowweed (*Pluchea sericea*), mesquite (*Prosopis* spp.), and cattail (*Typha* spp.). Plant species richness is lower in the BWR than in its unregulated upstream tributary, the Santa Maria River, however, likely as a result of flood reduction [*Stromberg et al*., 2012]. In an effort to sustain the native riparian woodland habitat in the BWR, flow management at Alamo Dam has been guided in recent years by collaborative efforts between the Army Corps of Engineers and other stakeholders [*Shafroth and Beauchamp*, 2006; *Shafroth et al*., 2010] and has followed the Environmentally Sustainable Water Management framework [*Richter et al*., 2003; *Konrad et al*., 2012]. Environmental flow releases have included base flows designed to provide summer and fall irrigation for cottonwoods and willows as well as flood releases, as water availability allows, to promote cottonwood and willow recruitment. Dam operations for environmental purposes on the BWR are facilitated because competing water uses such as hydropower production or irrigation are absent or limited, because the downstream floodplain is sparsely populated, and because water released from Alamo Dam is delivered to and impounded by Lake Havasu (Figure 1). Communication among scientists and managers has allowed scientists to provide input into the design of flow releases and to capitalize on planned flood releases for data collection. These factors combine to create a unique field laboratory.

[13] This study investigates a series of floods released from Alamo Dam into the BWR (Table 1, Figure 2). During the winter of 2004 and 2005, high runoff associated with El Niño caused multiple high-flow events in the range of the dam's maximum outlet capacity (approximately $200\,m^3s^{-1}$). These events, although they occurred before the data collection effort described below, were significant to our study because they scoured vegetation from low-elevation bars and created bare surfaces for seedling establishment. The falling limb of the last of these events, in spring 2005, was managed to promote seedling recruitment by drawing down flow releases from Alamo Dam at a rate of approximately $0.5\,m^3s^{-1}day^{-1}$, subsequently resulting in the widespread establishment of riparian seedling patches initially codominated by tamarisk and willow, some of which we subsequently monitored. In March 2006, a controlled flood was released from Alamo Dam in which discharge was ramped up to an instantaneous peak of $69\,m^3s^{-1}$, maintained at that peak for 7.5 h, and then dropped and held at $56\,m^3s^{-1}$ for 2 days, followed by a gradual drawdown of approximately $1\,m^3s^{-1}day^{-1}$. Smaller pulse flow releases occurred in 2007 and 2008 (Table 1). In 2010, another El Niño year in which inflows to Alamo Lake were large, the highest-magnitude, longest-duration flood since 2005 was released (Table 1).

[14] Flows at the downstream end of the BWR were lower than those released from Alamo Dam during these floods as a result of infiltration and associated flow attenuation within Planet Valley and other alluvial valleys. Flows measured downstream of Planet, at the BWR near Parker, AZ gauge (# 09426620), were most similar to those measured at Alamo when antecedent water-table levels in Planet Valley were high and flood durations were longer

Table 1. Summary of High-Flow Characteristics for 2005–2010 Flood Releases From Alamo Dam on Bill Williams River, AZ, Based on Measurements at the Alamo Gauge, Immediately Downstream of Alamo Dam (USGS #09426000), and the Parker Gauge, 50 km Downstream (BWR near Parker, AZ, # 09426620)

Date[a]	USGS Gauge	Peak Magnitude (m^3s^{-1})	Duration (h)
21 Feb 2005	Alamo	205	Multiday[b]
	Parker	208	
14 Mar 2006	Alamo	69	7.5[c]
	Parker	66	
9 Apr 2007	Alamo	29	16
	Parker	5.1	
31 Mar 2008	Alamo	65	8
	Parker	1.6	
7 Mar 2010	Alamo	85	36[d]
	Parker	63	

[a]Date of initial flood release from Alamo Dam, passage of peak at Parker gauge may differ.

[b]The peak magnitude reported for this event is a daily average. Flows remained $>100\,m^3s^{-1}$ for approximately 3 weeks during this event.

[c]$69\,m^3s^{-1}$ maintained for 7.5 h, followed by 2 days at $56\,m^3s^{-1}$.

[d]$85\,m^3s^{-1}$ maintained for 36 h, followed by $56\,m^3s^{-1}$ for 5 days.

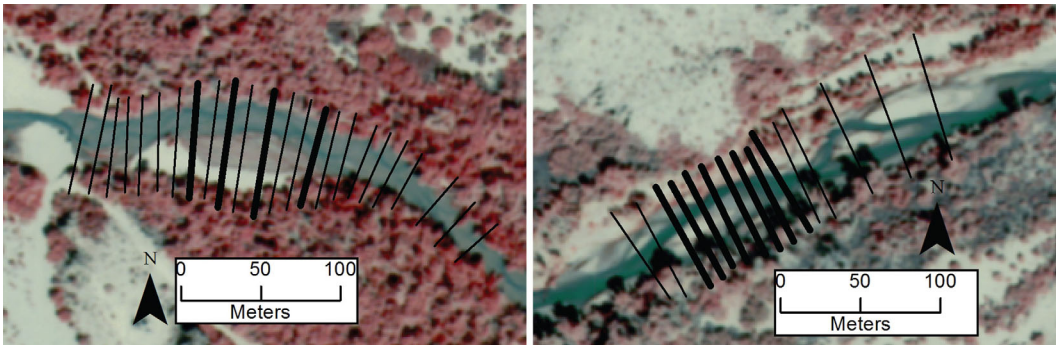

Figure 3. (left) Mineral (right) and Rankin study reaches (September 2005 aerial photographs), showing data collection transects. Topography was measured along all transects; grain size and vegetation data were collected along transects denoted with thicker line (grain size was also measured at upstream- and downstream-most transects). Flow is from right to left.

(e.g., 2005, 2006, 2010). In contrast, when antecedent water levels were low and/or flood durations were short (2007 and 2008), flood peaks were substantially attenuated in downstream portions of the BWR. For example, peak flows measured at Parker were 19% and 2% of the upstream peaks in 2007 and 2008, respectively (Table 1).

[15] The magnitude and duration of the releases were constrained by water availability, dam release capacity, and concerns over potential impacts to other downstream land and water management interests. Consequently, the events were small compared to historic floods on the BWR (Figure 2). These events were substantial, however, when considered within the context of the postdam hydrologic regime; the 2005 event was the largest since dam construction. The observed flood releases were timed to overlap with the seed-release period of willow and cottonwood, although tamarisk also releases seed concurrently with willow on this river system [*Shafroth et al*., 1998].

3. Methods

[16] We conducted a multiyear field campaign that combined measurements of vegetation, topography, bed sediment, and flood hydraulics in two study reaches (Figures 1 and 3). The first reach is 18 km downstream from Alamo Dam and is referred to hereafter as the Rankin reach (after a nearby ranch). The second reach is 48 km downstream from the dam and is referred to as the Mineral reach (after a nearby ephemeral tributary, Mineral Wash). These reaches were selected for detailed study because of (1) the presence of bars with cohorts of seedlings that established on bare surfaces in 2005, as indicated by our observations during multiple field visits between November 2004 and February 2006, (2) differences in flow, sediment supply, and bed-material size, as a result of relative positions downstream from Alamo Dam, allowing assessment of spatial variations in ecogeomorphic interactions, and (3) access; these are among a small number of reaches of the BWR with road access.

[17] Our field measurements focused on the 2006, 2007, and 2010 flood releases and their effects. Data collection among years varied as a function of the amount of advance notice we had for floods, personnel and equipment availability, and study objectives. Our most intensive data collection effort surrounded the 2006 event, which we used to investigate (1) differential seedling mortality among willow and tamarisk (the most frequently occurring species in our study plots) and associated changes in the size distribution of seedlings, (2) spatial variability in flood-induced changes in channel morphology and grain size, in relation both to the presence of vegetation and proximity of study reaches to Alamo Dam, and (3) flood hydraulics. Elements of these analyses were continued for the 2007 and 2010 event, which, in combination with the 2006 flood, were used to study the evolution of vegetation-hydrogeomorphic feedbacks over a multiyear, multiflood period. Whereas the 2006 and 2007 flood releases were planned well enough in advance to permit detailed preflood measurements, the 2010 flood was an El Niño-driven event with minimal advance notice.

3.1. Vegetation Sampling

[18] Starting in late February 2006, we sampled woody seedlings in 1 m^2 plots within belt transects aligned perpendicularly to the long axis of the sample bars. These transects align with the geomorphic surveys of cross-section topography and bed-material texture described below. At Mineral the plots were arrayed along four transects, spaced 20–30 m apart, ranging in width from 10 to 24 m, and distributed along one vegetated bar with a surface area of approximately 1500 m^2, as measured in ArcGIS (Figure 3). At Rankin the plots were arrayed along eight transects, spaced 10 m apart, ranging in width from 2 to 14 m, and distributed along two vegetated bars with surface areas of approximately 500 m^2 and 300 m^2 (Figure 3).

[19] At the time of our first vegetation sampling effort, all woody plants in our study plots were less than 1 year old seedlings that established in association with the 2005 floods. Ninety percent of all measured woody seedlings were tamarisk, 9% were Goodding's willow, and $< 1\%$ were arrowweed or seep willow. Herbaceous vegetation cover in our plots was $5.9 \pm 9.8\%$ at Mineral and $16.0 \pm 18.3\%$ at Rankin (mean $\pm$ standard deviation). Sixty-one and 38 plots at Mineral and Rankin, respectively, contained seedlings of either tamarisk or willow at the beginning of our field study. Of these 99 plots, 48 contained both species, 28 contained willow but not tamarisk, and 23 contained tamarisk but not willow. The elevations above the base-flow water surface of these groups of plots were

similar: 0.21 ± 0.07 m (mean ± standard deviation) for mixed tamarisk and willow plots, 0.19 ± 0.07 m for willow-only plots, and 0.25 ± 0.08 m for tamarisk-only plots.

[20] Within each plot, the diameter at the ground surface, total height, and species identity of every woody seedling were recorded, and stem density (number of woody plants per square meter). The same variables were measured in all plots ten weeks later (in early May 2006) following the March 2006 pulse flood.

[21] In April 2007, we remeasured the same variables in all plots immediately before that year's flood release. Several days later, after the 2007 flood recession, we resurveyed stem density in all plots at Rankin; plant size was not resurveyed because the short time since the previous surveys precluded likely plant growth. At Mineral, because of flood attenuation and the previous year's aggradation, the vegetated study plots were not inundated by the 2007 event, so we did not complete postflood vegetation surveys. Field measurements associated with the 2010 event (described below) did not include vegetation surveys because of time and personnel limitations.

[22] We calculated seedling mortality as the change in the number of live stems in each plot (postflood minus preflood stem density) associated with the 2006 and 2007 events. We equate stem-density reductions to mortality based on the associated geomorphic mechanisms, as discussed below, and the absence of standing dead stems in our plots. We then completed several tests of the significance of these changes as a function of species (tamarisk versus willow), reach (Rankin versus Mineral), year (2006 versus 2007), and antecedent plant conditions. First, we completed a two-way ANOVA on species and reach for the 2006 data only (the test on differences between reaches was restricted to 2006 because we did not measure density changes in 2007 at Mineral). Second, using the Rankin data only, we completed a two-way ANOVA with species and year as factors. Third, to test the effects of drag associated with seedlings on observed stem-density differences, we added antecedent vegetation density [*Nepf*, 1999] as a covariate. Vegetation density, the projected plant area per unit volume (m^{-1}), can be approximated, treating plants as cylinders, as stem density times average stem diameter. In our case we calculated vegetation density for each plot as: $(\text{stem density})_{\text{tamarisk}} \times (\text{average diameter})_{\text{tamarisk}} + (\text{stem density})_{\text{willow}} \times (\text{average diameter})_{\text{willow}}$. For plots with other woody seedlings (e.g., seep willow, arrowweed), this equation was modified accordingly. This approach neglects details of how plant architecture and flexibility influence drag but provides a reasonable approximation of vegetation drag [*Nepf*, 1999; also see *Kean and Smith*, 2005].

[23] Stem-density-difference values included zeros (no change in density) and negative values, where stem density increased as a result of addition of flood-trained stems from upstream of the plot and/or burial of the main stem, leaving multiple secondary stems protruding from the ground surface. Because these negative values resulted from our methods rather than from real increases in the numbers of plants, we set them to zero for statistical tests. We applied a $\log_{10}(x+1)$ transformation to the density-difference values to better satisfy normality and homoscedasticity assumptions.

[24] To compare the size distribution of seedlings before and after the 2006 flood and to test for significant differences, we performed two-sample Kolmogorov-Smirnov (K-S) tests on diameter and height. The K-S test is suited to comparing distributions and, as a nonparametric test, does not require normally distributed data. Quantile-quantile plots (not shown here) confirmed that seedling size for the stems sampled was not normally distributed.

3.2. Geomorphic Change Surveys

[25] Topographic surveys of the Mineral and Rankin reaches were completed in 2006, 2007, and 2010 to measure flood-induced topographic changes and channel evolution. At the outset of our field campaign, in February 2006, we surveyed 23 and 15 cross sections in the Mineral and Rankin reaches, respectively, a subset of which contained the plots in which vegetation was sampled, as well as thalweg profiles. In May 2006, both reaches were resurveyed. We used Trimble Real Time Kinematic (RTK) GPS units for these surveys. To characterize topographic change associated with the March 2006 event, we calculated differences in the average May (postflood) versus February (preflood) bed elevation for each cross section. Calculations of elevation differences were performed for all portions of the bed inundated by the 2006 event (according to water surface elevation surveys at the flood peak) and for subsets of transects along our vegetated study bars. To gain additional insight into scour and fill dynamics, we deployed scour chains longitudinally along vegetation study bars in the two study reaches in February 2006 [see for example *Powell et al.*, 2006, for details on methods of interpreting scour chains].

[26] We resurveyed topography along varying subsets of our cross sections, as well as longitudinal profiles, in subsequent years. Following the 2007 high-flow release, we resurveyed six cross sections in the Rankin reach using a Pentax PCS300 total station. Because of the substantial attenuation of the 2007 flood (Table 1), we did not resurvey topography at Mineral after this event. Before and after the 2010 flood, 13 and 15 cross sections were resurveyed in the Mineral and Rankin reaches, respectively, using RTK-GPS.

[27] Bed sediment samples were collected before and after the 2006 flood to measure flood-induced changes in grain size, inter-reach differences, and the mediating effect of vegetation on grain size changes. We used a "can on a stick" bulk sampler [*Edwards and Glysson*, 1999], which was suitable for the sand- and fine-gravel-dominated sediments in our study reaches, that penetrated to a depth of 7 cm and collected 0.4–1 kg of sediment per sample. Between 80 and 93 samples were collected in each reach during each sampling period at evenly spaced intervals along selected cross sections, some of which overlapped with our vegetation transects. All samples were dried and sieved at 1/2–φ intervals (from 0.063 to 32 mm) to determine grain size distributions.

[28] For statistical comparisons, we composited grain size samples either by cross section, for unvegetated cross sections, or into unvegetated and vegetated portions of those cross sections that contained vegetation plots. This resulted in 56 composite samples (10 in Mineral, 18 in Rankin, measured both before and after the 2006 event). For each of these, we calculated the D_{50} and the fraction of the composited sample mass within five grain size categories: >8mm ($\varphi<-3$), 2–8 mm ($-3<\varphi<-1$), 1–2 mm

($-1<\varphi<0$), 0.5–1 mm ($0<\varphi<1$), and 0.0625–0.5 mm ($1<\varphi<4$); the categories that encompass a smaller range of φ classes are those in which bed materials are most prevalent on the BWR. We first tested whether the 2006 flood produced a change in grain size distributions using two-way ANOVAs, with D_{50} and fraction of sample mass within grain-size categories as response variables and time (preflood versus postflood) and reach (Rankin versus Mineral) as factors. We then calculated the postflood minus preflood difference in grain size (for D_{50} and within each of the size categories), and using these differences as response variables, we applied a linear model with reach and vegetation (vegetation versus no vegetation along transects) as factors. We also represented vegetation as a continuous variable, using the average preflood stem density along the transect (0 for unvegetated transects) as a covariate, but this did not change results compared to using vegetation as a categorical variable. Tests for autocorrelation of grain size responses among transects, using the acf function in R [*R Development Core Team*, 2012], showed that autocorrelation was not significant for lag = 1 (i.e. among adjacent transects) and therefore did not need to be accounted for in subsequent significance testing.

3.3. Flood Hydraulics Measurements

[29] In addition to the surveys described above of preflood and postflood ecogeomorphic characteristics, we also measured several components of high-flow hydraulics during our study floods. To determine the arrival time of floods released from Alamo Dam at Rankin, we deployed pressure transducers (in 2006 and 2010); in 2007 we visually recorded the arrival of the flood pulse at Rankin. These data were used to calculate the reach-average velocity for the 18 km between Alamo Dam and Rankin. Flood timing and duration at Mineral were largely inferred from the BWR near Parker gauge (<2 km downstream).

[30] We also measured local velocities during the 2006 and 2007 floods along our study transects, including measurements within vegetation patches, at positions near (2 m from) the edge of patches, and at positions further away from (>4 m) patches. In 2006, we completed 15 velocity measurements ($n = 7$ within, 4 near, 4 away from vegetation patches) at Mineral and Rankin, and in 2007, we completed 18 velocity measurement, 6 in each location type, at Rankin. We used a SonTek FlowTracker acoustic Doppler velocimeter, at 0.4 times the flow depth and recording for 40 seconds. Velocity samples were limited to wadeable areas; depths at measurement positions were 0.50 ± 0.27 m (range of 0.2–1.2 m). Velocities did not vary significantly with the flow depth of the measurement position ($p = 0.09$). To evaluate the effects of measurement proximity to vegetation patches on velocity, we performed two-way analysis of variance, where year (2006 and 2007) and location (in, near, and away from vegetation patches) were factors, and a Tukey HSD multiple comparison test was applied.

4. Results

[31] We observed differences in vegetation and geomorphic responses to flood releases both temporally, between the 2006, 2007, and 2010 events, and spatially, between our upstream (Rankin) and downstream (Mineral) study reaches. Evolution of vegetation and morphology are evident in repeat photographs of the study reaches (e.g., Figures 4 and 5) and are illustrated by the data presented below on seedling mortality, shifts in seedling size distributions, topographic and textural changes, and flood hydraulics.

4.1. Vegetation Responses to Controlled Floods

[32] Both the 2006 and 2007 floods caused substantial seedling mortality. Stem density reductions associated with the 2006 flood differed significantly between species (Figure 6) and reaches (Table 2).

[33] Preflood stem densities of tamarisk seedlings exceeded willow densities in both study reaches (Figure 6; $F_{1,143} = 96$, $p<0.001$), but tamarisk experienced significantly greater flood mortality. The 2006 event produced 85% reductions in tamarisk density in both reaches, compared to reductions of 26% and 64% for willow seedlings in Mineral and Rankin, respectively. Tamarisk was eliminated from 33% of plots that originally contained tamarisk at Mineral and 30% at Rankin postflood, whereas willow was eliminated from 17% of plots that originally contained willow at Mineral and 23% at Rankin. Flood-induced seedling mortality was limited to low-elevation bars in the active channel (<1m above the thalweg); no changes were observed on higher vegetated surfaces.

[34] Measurements completed before the 2007 event indicated differences in vegetation evolution since 2006 between reaches. At Rankin, stem densities of both willow and tamarisk had increased since the post-2006 event measurements (Figure 6), suggesting that some new colonization and resprouting had occurred in this reach. Whereas willow densities exceeded pre-2006 levels (at Rankin), tamarisk densities did not. At Mineral, in contrast, stem densities showed little change since the post-2006 measurements, suggesting that new colonization or resprouting had been minimal.

[35] Stem-density reductions associated with the 2007 event were significantly lower than in 2006 (Table 2). As in 2006, flood-induced mortality was greater for tamarisk than willow (Table 2, Figure 6), which continued the trend of an increase in the ratio of willow to tamarisk, from 0.07 at the beginning of the study period (pre-2006) to 0.55 after the 2007 event. The 2007 event did not inundate study plots in the lower (Mineral) reach, and field observations confirmed that plant mortality was minimal at Mineral.

[36] For both the 2006 and 2007 events, antecedent vegetation density (projected plant area per unit volume) had highly significant effects on flood-induced mortality (Table 2). The mean ($\pm$ standard deviation) of antecedent vegetation density was $0.1 \pm 0.09\,\mathrm{m}^{-1}$ at Rankin in 2006, $0.06 \pm 0.05\,\mathrm{m}^{-1}$ at Mineral in 2006, and $0.12 \pm 0.08\,\mathrm{m}^{-1}$ at Rankin in 2007.

[37] The average diameter and height of willow seedlings was 2–5 times greater than tamarisk during all field measurement periods (Figure 7). These seedlings were recruited during the 2004–2005 floods and thus show very high growth rates. By the time of the 2007 event, vegetation that had established in 2005 and survived the 2006 event had grown sufficiently (Figures 4c, 5c, and 7) to

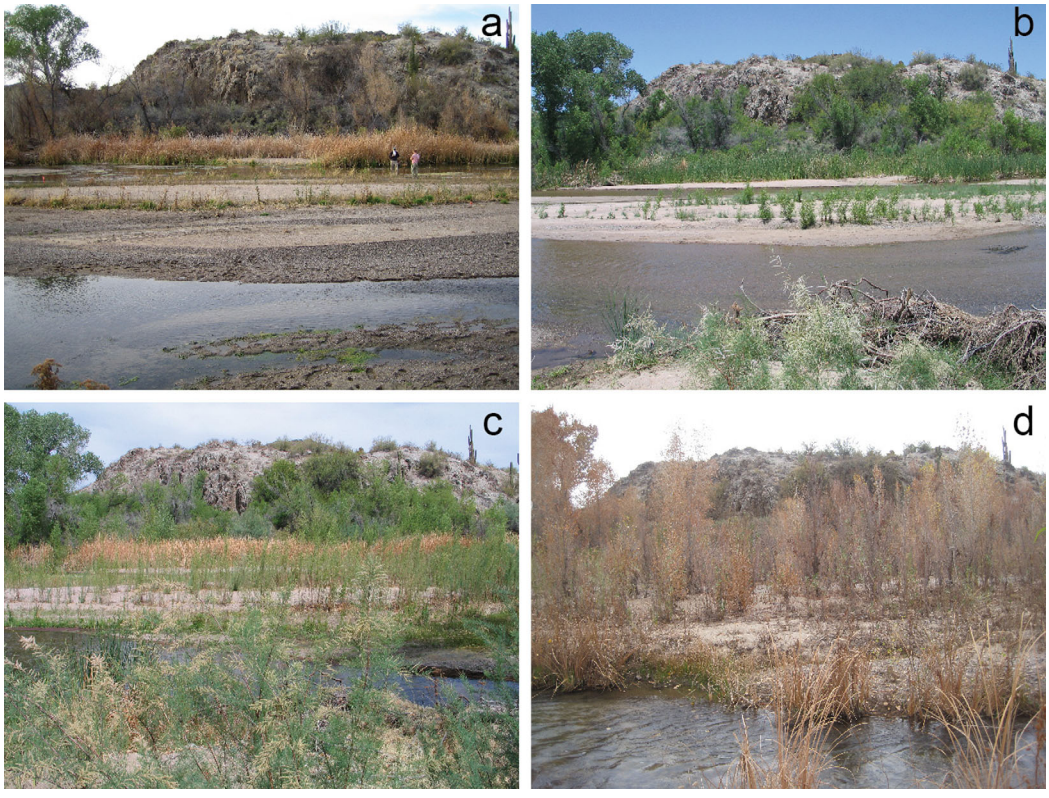

Figure 4. Repeat photographs of central portion of Rankin study reach, from left bank looking to north: (a) February 2006, (b) May 2006, (c) April 2007, (d) December 2010. Sequence illustrates lateral movement of bars, growth of seedlings that have survived floods, resilience of larger woody plants to floods, and presence of cattail (along far bank).

mediate the geomorphic effects of that event and to in turn influence flood-induced seedling mortality. This dynamic, whereby seedlings surviving one flood grow and produce greater drag in subsequent floods, continued in the 2010 event, as discussed further below.

[38] Comparison of histograms of plant height between survey periods illustrates flood effects on vegetation structure, whereby the greatest mortality occurred among seedlings <40 cm in height and <4 mm in ground diameter, as evidenced by a dramatic reduction in the numbers of those smaller plants (Figure 8). In contrast, plants >70 cm in height and >6 mm in diameter showed little change as a result of flooding. Two sample K-S tests show that the shift in seedling size distributions associated with truncation of smaller sizes was significant for the 2006 flood ($p<0.001$ for both diameter and height; $D=0.47$ for diameter and $D=0.49$ for height, where D is a test statistic describing the maximum distance between the preflood and postflood distribution functions). These results suggest a threshold of plant resistance to flooding for these events and antecedent conditions.

4.2. Flood-Induced Geomorphic Changes

[39] Despite similar bed gradients (0.0028 in Rankin, 0.0026 in Mineral), the two study reaches experienced different geomorphic responses to floods, which in turn contributed to the different vegetation responses described above. In the reach closer to the dam (Rankin), the 2006 flood caused scour and shifting of bars and changes in the position of the base-flow channel. Along our 15 cross sections, local erosion and deposition offset each other laterally, such that the average of postflood minus preflood elevations was small ($\Delta Z = 0.09 \pm 0.04$ m; range = 0.02–0.14 m). Of our two vegetation sample bars in the Rankin reach, the upstream one that supported the majority of seedlings surveyed preflood was scoured and trimmed ($\Delta Z = -0.09 \pm 0.09$ m for vegetation transects along this bar, e.g., cross section 90 in Figure 9) such that only a narrow line of vegetation remained. On the downstream bar, net elevation change was limited ($\Delta Z = 0.05 \pm 0.09$ m for vegetation transects along this bar, e.g., cross sections 30 and 50 in Figure 9), even in areas with substantial reductions in stem density. Scour chains that were recovered indicated mean scour depths of approximately 0.3 m on the bar surface and subsequent fill. Some of the scour chains could not be relocated, possibly as a result of deeper scour.

[40] In the Mineral reach, we observed differences in topographic response between vegetated and unvegetated areas of the bed. In February 2006, our sample bar had two distinct levels: a lower, vegetated surface bordering the channel, and a higher unvegetated surface (e.g., cross sections 70, 90, and 110 in Figure 9). Surveys after the 2006 flood showed aggradation on the lower, vegetated surface of the bar ($\Delta Z = 0.20 \pm 0.11$ m for vegetated portions of eight cross sections; range = 0.07–0.33 m). This aggradation buried smaller seedlings and produced the mortality

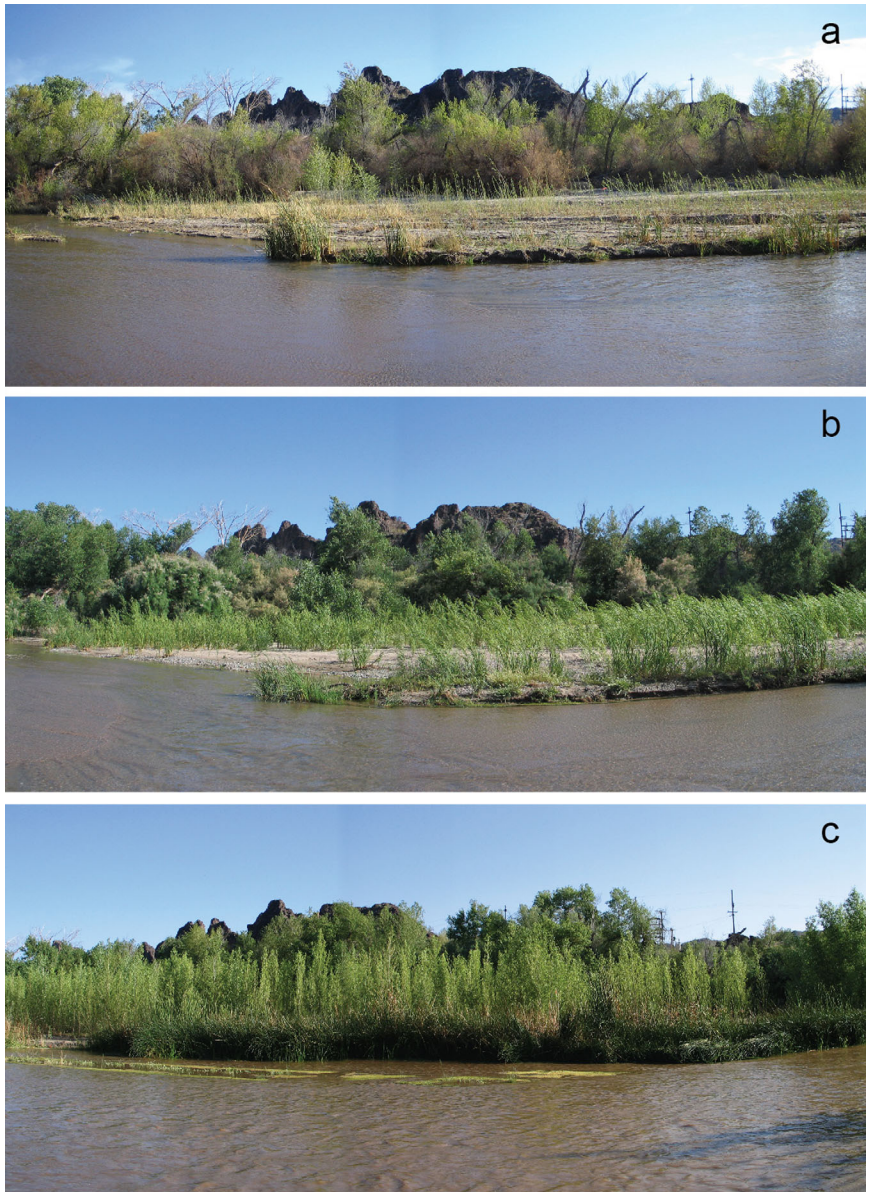

Figure 5. Repeat photographs of vegetation study bar in Mineral reach, looking from channel toward river left: (a) February 2006, where mix of 1 year old tamarisk and willow seedlings are visible, (b) May 2006, where willow seedlings now dominate as a result of flood-induced burial of the smaller tamarisk, (c) April 2007, where willow seedlings that survived the 2006 flood have grown and stabilized the bar.

described above. The upper surface of the bar changed little, such that the vegetated and unvegetated surfaces had similar postflood elevations but were perched higher above the low-flow channel (Figure 9). This difference in relative elevation was exacerbated by incision of the low-flow channel bed adjacent to the vegetated sample bar ($\Delta Z = -0.13 \pm 0.03$ m for the same eight cross sections noted above). Aggradation of the vegetated portion of the bar and channel incision were offsetting in terms of average elevation change in the Mineral reach ($\Delta Z = -0.06 \pm 0.04$ m for 24 cross sections).

[41] The 2007 flood produced only limited geomorphic changes in the Rankin reach ($\Delta Z = 0.04 \pm 0.04$ m, for the average of post-2007 cross-section measurements minus May 2006 surveys). As noted above, the 2007 event had almost entirely attenuated by the time it arrived at Mineral and therefore did not alter channel morphology there.

[42] The 2010 flood, despite being the largest of the 2006–2010 period (Table 1), produced little topographic change. Cross-section surveys documented small amounts of localized scour and fill (e.g., Figure 9) but no reach-wide patterns of net topographic change ($\Delta Z = 0.02 \pm 0.06$ m, for the average of post-2010 minus pre-2010 cross-section elevations in Rankin). In both reaches the larger magnitude of the 2010 flood was offset by the increases in vegetation size and density, limiting its geomorphic effectiveness.

[43] An additional source of geomorphic change during the latter years of our study period was the expansion of beaver, especially in the Rankin reach. Beaver activity

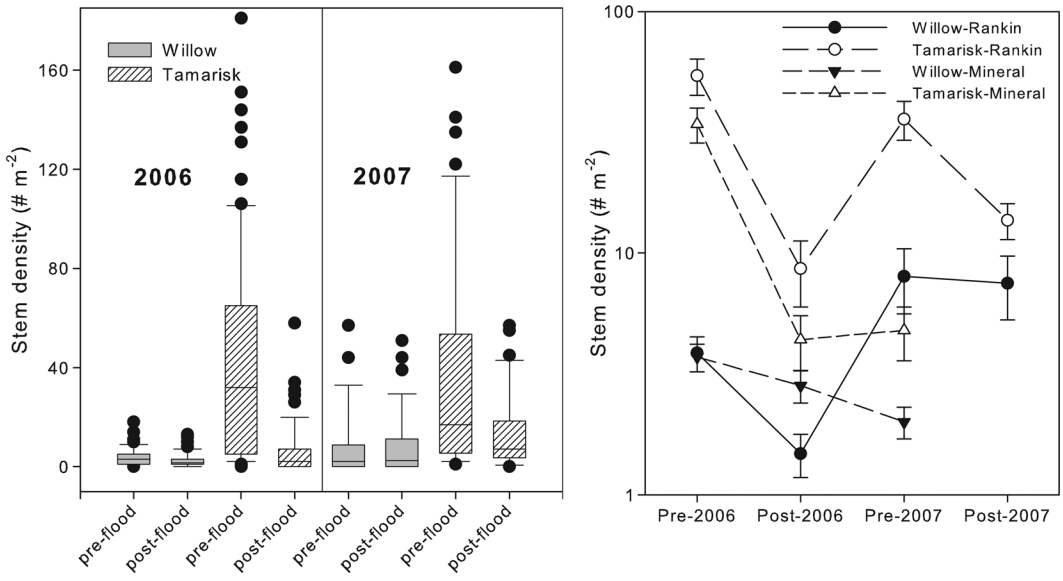

Figure 6. Stem densities of willow and tamarisk before and after 2006 and 2007 floods. Left plot shows distributions of measured densities, with 2006 data from Rankin and Mineral combined and 2007 data from Rankin only (boxes bound 25th and 75th percentiles, solid lines in boxes illustrate medians, and whiskers bound 10th and 90th percentiles). Right plot shows mean and standard error of stem densities, differentiated among species and reaches. Densities for post-2007 in Mineral were not measured because the 2007 event did not inundate vegetation plots in that reach.

produced variability in longitudinal profiles, with dams and impoundments producing up to 1.2 m high steps in the profile by 2010 in Rankin (Figure 10). In both reaches, thalwegs were more entrenched relative to surrounding bars by 2010 compared to February 2006, in part as a result of beaver activity (Figures 9 and 10).

[44] Neither reach experienced changes in active channel width in any of the floods. Bank erosion potential is limited in these reaches by vegetation, including armoring of steeper banks by mats of arrowweed and energy dissipation by dense, near-bank stands of cattail. Early in the study period the effects of these species were confined to the banks (in the case of arrowweed) and within a few meters of the banks (in the case of cattail; e.g., Figure 4), although cattail had expanded further into the channel by 2010.

[45] With respect to bed-material size, bed sediments at Rankin are coarser than at Mineral throughout their size distributions (Figure 11). At the beginning of the study period D_{50} values averaged over each reach were 2.7 and 0.7 mm in Rankin and Mineral, respectively. The Rankin reach includes many gravel lenses and patches, whereas bed sediments were more unimodal in the coarse sand range in Mineral (Figure 11). Initial (preflood) differences in D_{50} between vegetated and unvegetated portions of transects were not significant ($F_{1,25} = 0.4$, $p = 0.5$), although fractions of finer sediments (0.0625–0.5 mm) were higher in vegetated transects. The 2006 event did not produce significant changes in grain sizes, in either D_{50} or fraction of sample mass within different size categories. Tests of how postflood minus preflood difference in grain size varied among reaches and as a

Table 2. Results of Statistical Tests of Flood-Induced Reductions in Stem Density as a Function of Species (Tamarisk, Willow), Reach (Rankin, Mineral), Year (2006, 2007), and Antecedent Vegetation Density (Stem Density × Stem Diameter)

Model	Factor	Degrees of Freedom	Sum of Squares	Mean Square	*F* Value	*P* Value
2006[a]	Species	1	79.4	79.4	138.6	<0.0001
	Reach	1	5.4	5.4	9.4	0.003
	Vegetation density	1	43.2	43.2	75.4	<0.0001
	Species×vegetation density	1	14.4	14.4	25.4	<0.0001
	Residuals	101	57.9	0.57		
Rankin[b]	Species	1	35.4	35.4	30.9	<0.0001
	Year	1	7.2	7.2	6.3	0.014
	Vegetation density	1	22.7	22.7	19.8	<0.0001
	Residuals	73	83.6	1.1		

[a]Species×reach, reach×vegetation density interactions not significant.
[b]Model testing for effects of year (2006 versus 2007) applied to Rankin only, because 2007 event did not inundate plots at Mineral. Two-way interactions not significant.

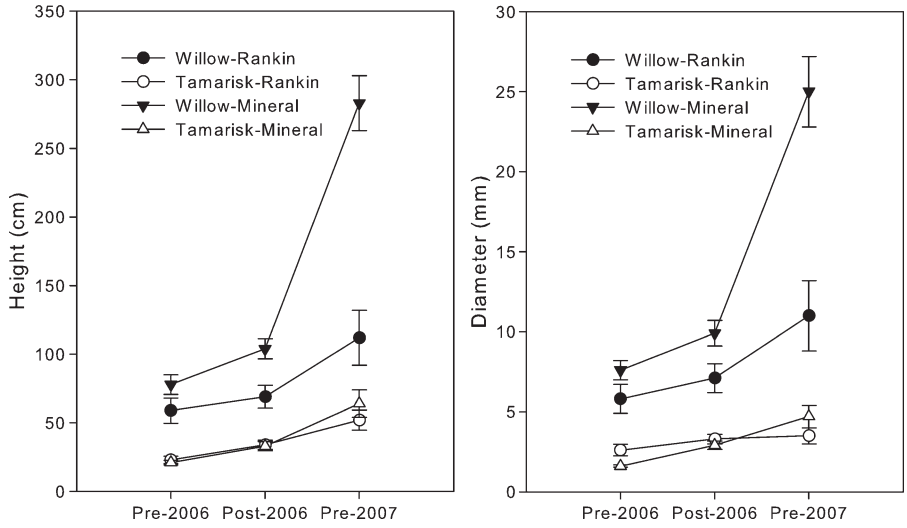

Figure 7. Seedling height and diameter (mean and standard error) measured in February 2006 (preflood), May 2006 (postflood) and April 2007 (preflood), for willow and tamarisk, in Rankin and Mineral study reaches. Because vegetation measurements following the 2007 event were taken within 1 week of the preflood measurements, a time period in which minimal plant growth would have occurred, diameter and height were not measured after the 2007 event.

function of vegetation found that these factors were not significant for D_{50} and most size categories. The postflood minus preflood fraction within the fine tail of grain size distributions (0.0625–0.5 mm) did show significant variation between reaches ($F_{1,25} = 7.1$, $p = 0.013$) and between vegetated and unvegetated

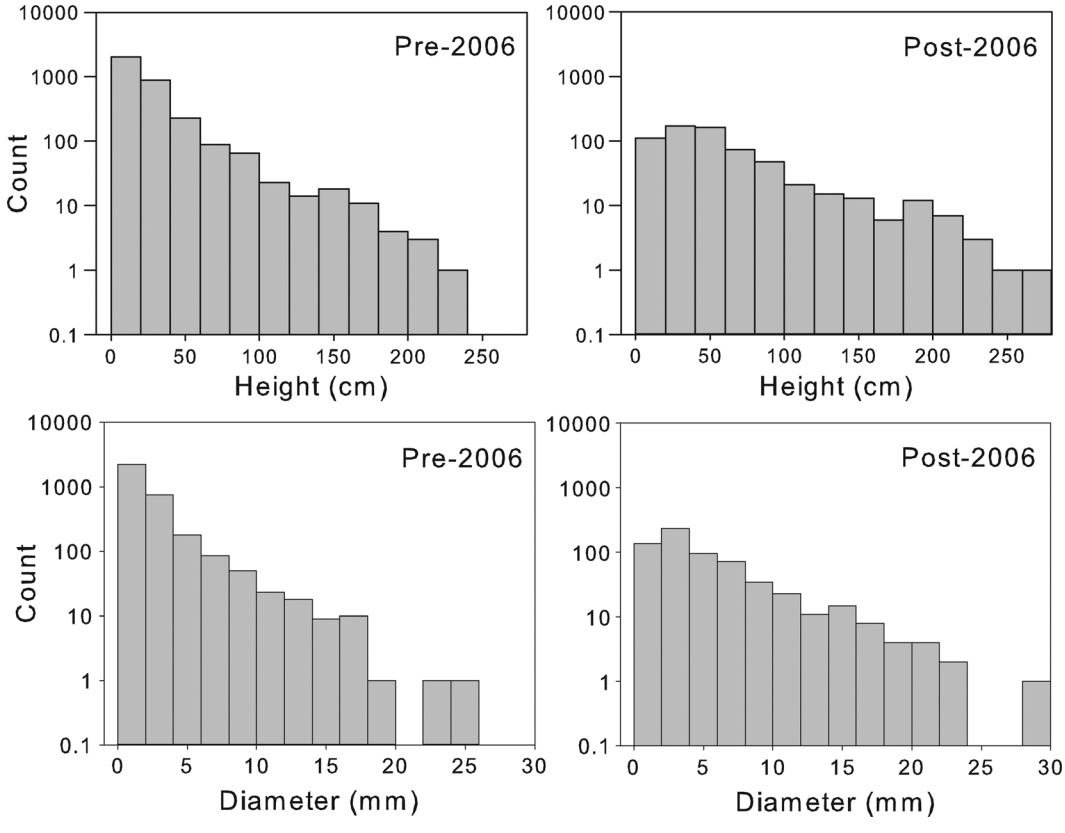

Figure 8. Histograms of plant heights (top) and diameters (bottom) measured in February and May 2006, before and after the 2006 flood, with reaches (Mineral and Rankin) and species (tamarisk and willow) combined. Data are based on repeat measurements of the same study plots. The histograms illustrate size-dependent mortality during the 2006 flood, primarily occurring among plants <40 cm in height and <4 mm diameter.

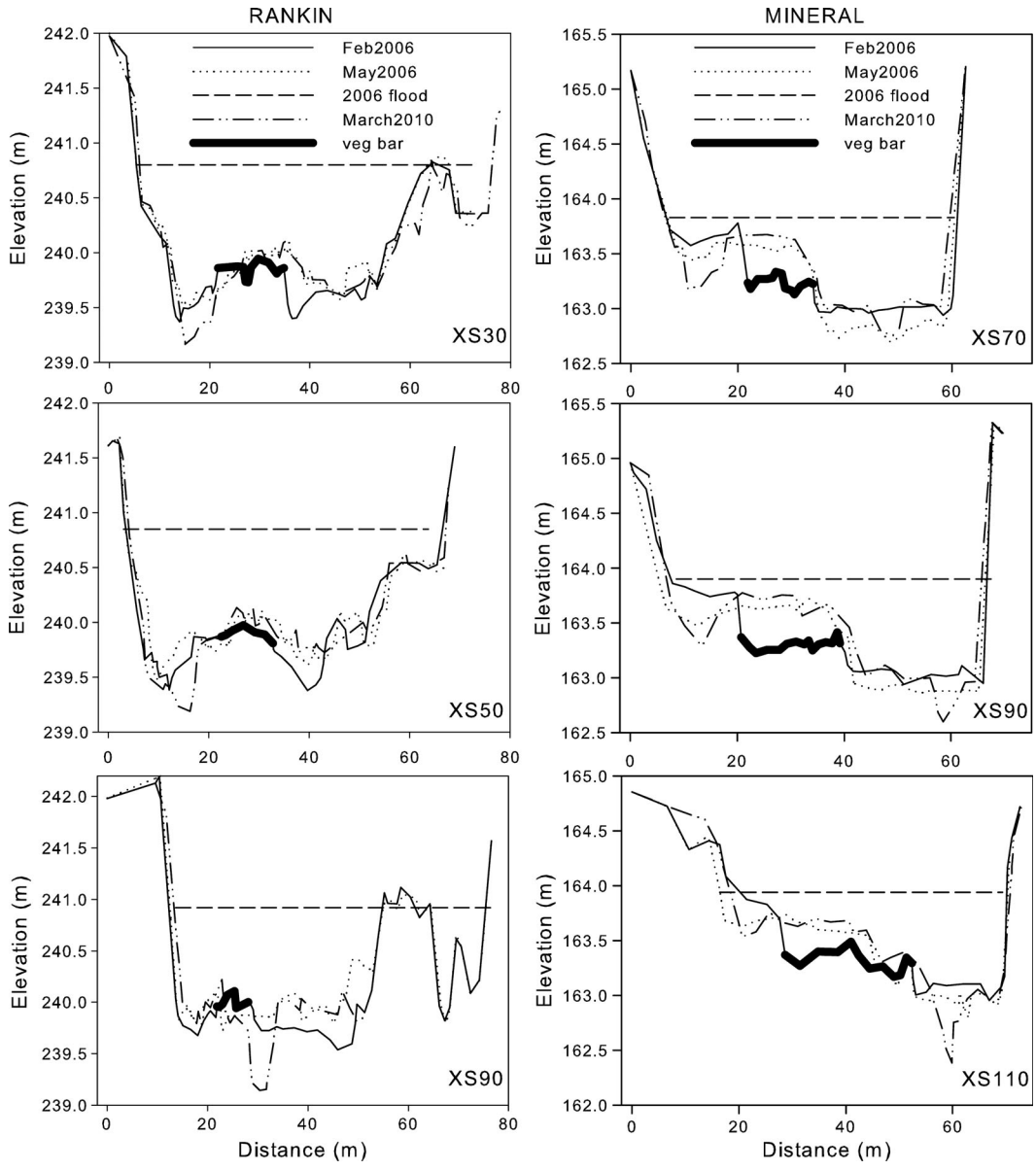

Figure 9. Changes in morphology across the study period for representative cross sections at Rankin (XS 30, 50, 90; left) and Mineral (XS 70, 90, 110; right). Cross-section numbers refer to their distance in meters from downstream end of each reach (also see Figures 3 and 10). Water surface elevation for 2006 flood and location of vegetation transects within cross sections (veg bar) also shown. The Rankin cross sections show lateral shifting, erosion, and deposition resulting from the 2006 flood and subsequent (March 2010) beaver-influenced morphologic adjustments. The Mineral cross sections show vegetation-influenced aggradation and leveling of the bar (on the left side of the cross sections) resulting from the 2006 flood, as well as increased elevation differences with time between the bar and the base-flow channel (on the right side of the cross sections).

transects, with the reduction being greater in unvegetated than in vegetated areas ($F_{1,25} = 9.5$, $p = 0.005$).

4.3. Flood Hydraulics

[46] Velocity data collected at multiple scales provide evidence of vegetation effects on flow conditions during BWR floods. Velocities averaged over the 18 km reach between Alamo Dam and our upstream reach for the 2006, 2007, and 2010 events, based on the time between flow releases at the dam and arrival at Rankin, were 0.8, 0.6, and 0.5 m s^{-1}, respectively (Figure 12). These velocities do not show the expected relationship with Q (i.e., hydraulic geometry would suggest a power-law relationship with a positive exponent). We attribute this discrepancy to the increasing height and density of both woody seedlings and cattail, from 2006 when plants were small and sparse to 2010 when vegetation was dense and tall. Although we did not quantitatively survey vegetation in the 18 km reach between Alamo Dam and the Rankin reach, field and aerial photo observations of that reach show a similar pattern to

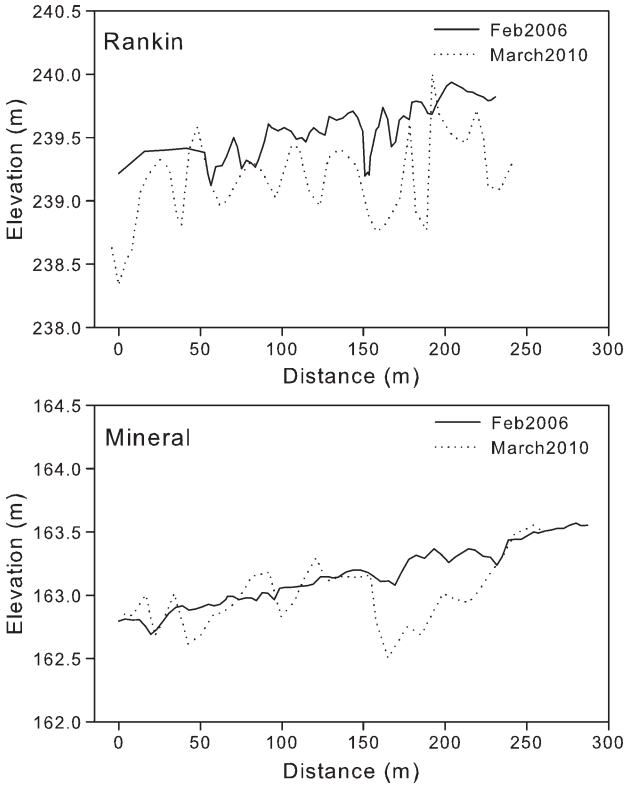

Figure 10. Longitudinal profiles of (top) Rankin and (bottom) Mineral study reaches at start (February 2006) and end (March 2010) of study period, where 0 on the x axis is the downstream end of each study reach.

our Rankin reach: a steady increase in the size and density of both woody plants and cattail from 2006 to 2010. Vegetation dynamics also contribute to variations in the lag between peaks of the different flood events at the Alamo and Parker gauges. However, flood travel times and attenuation between Alamo and Parker are affected by not only changes in vegetation density and size, but also by surface water-groundwater interactions.

[47] Velocity data collected in and around vegetation patches showed a significant two-way interaction between year and measurement location ($F_{2,27} = 6.7$, $p = 0.004$), indicating that the magnitude of the location effect (within, near, or away from vegetation patches) depended upon year. Pairwise comparisons among factors showed that in 2006, differences were not significant at $\alpha = 0.05$ among velocities within, near, and away from vegetation patches ($\bar{u} = 0.90 \pm 0.35$ m s^{-1}, 0.87 ± 0.27 m s^{-1}, and 0.74 ± 0.12 m s^{-1} within these three groups, respectively). In 2007, however, velocities were significantly lower (at $\alpha = 0.05$) within vegetation patches ($\bar{u} = 0.67 \pm 0.26$ m s^{-1}) than in positions near vegetation ($\bar{u} = 1.0 \pm 0.22$ m s^{-1}) and away from vegetation ($\bar{u} = 1.3 \pm 0.13$ m s^{-1}).

5. Discussion

5.1. Ecogeomorphic Feedbacks

[48] Our observations illustrate an evolution of the strength and nature of vegetation-morphodynamic feedbacks between 2005 and 2010. Early in the study period, one-way effects of hydrogeomorphic processes on vegetation were evident. The 2005 floods reset channel form, scoured vegetation from the active channel, and resulted in recruitment of a cohort of new pioneer vegetation. The 2006 flood, although much smaller than the 2005 floods, encountered a river system with only small seedlings in the active channel (Figures 4a and 5a). As a result, the 2006 flood was geomorphically effective, exceeding sediment transport thresholds, producing associated channel change by scour or aggradation, and causing vegetation mortality. Evidence of vegetation feedbacks was mixed for the 2006 event. On the one hand, antecedent vegetation density strongly influenced seedling mortality, and aggradation occurred in vegetated portions of the Mineral reach but not in unvegetated areas. On the other hand, vegetation effects on local (ADV-measured) velocities and grain size changes were not significant for this event.

[49] The survival of some seedlings in the 2006 event and their subsequent growth, combined with new recruitment, strengthened vegetation effects on channel morphology and later flows. Although the small size of the 2007 flood makes it difficult to tease out influences of hydrogeomorphic processes on vegetation and vice versa for this event, our observations of flood-induced mortality of small seedlings (as in 2006) but persistence of larger plants, limited geomorphic change, and significant vegetation effects on local velocities are suggestive

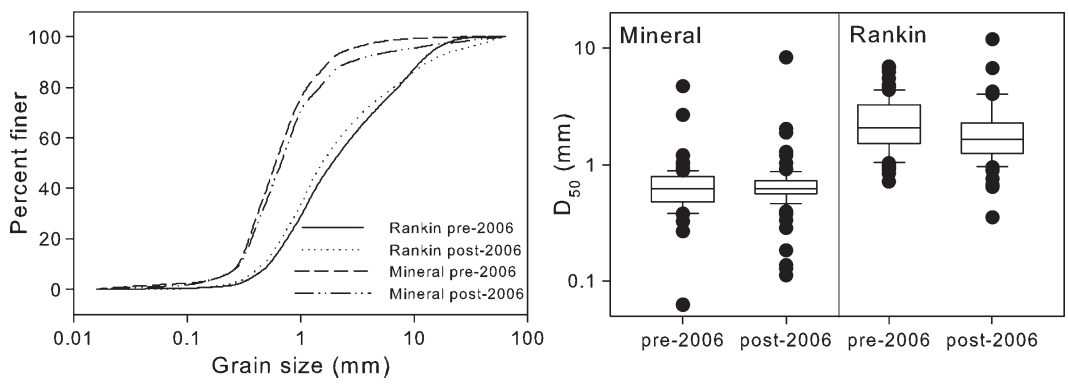

Figure 11. Grain size data from Rankin and Mineral before and after 2006 flood: left plot shows size distributions of composited bulk samples, and right plot shows median grain size (D_{50}) for all samples.

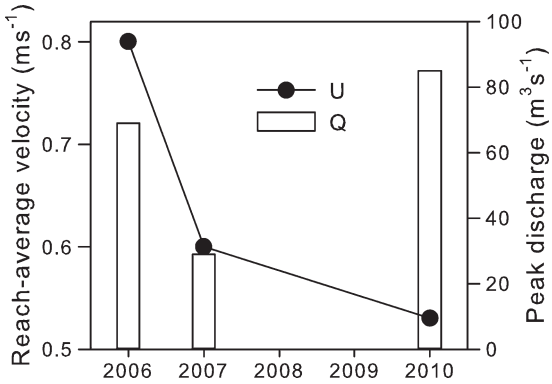

Figure 12. Velocities averaged over the 18 km reach from Alamo Dam to the Rankin reach, based on the time between flow releases at Alamo (from USGS gauge data) and their arrival at Rankin (from pressure transducer and/or visual records); right axis shows corresponding peak discharges released at Alamo. The pattern reflects increasing vegetation drag.

of increasingly bidirectional vegetation-hydrogeomorphic effects. By the time of the 2010 flood, vegetation was large and dense enough to have been relatively insensitive to scour and burial. The competitive advantage shown by willow in the 2006 and 2007 events persisted; the willow that recruited in 2005 and survived the 2006 event were many meters high as of late 2010 (e.g., Figure 4d). Moreover, cattail density greatly increased along the BWR after 2007. The net effect of increases in cattail density and larger woody seedlings was to produce substantial drag that mediated the geomorphic effectiveness of the 2010 event, which caused minimal topographic change despite its 30% larger magnitude than the 2006 flood.

[50] *Corenblit et al*. [2007]'s conceptual model describing the temporal evolution of feedbacks between hydrogeomorphic processes and riparian vegetation applies well to the BWR. The 2005 floods triggered a "geomorphic phase" in which the effect of vegetation on channel form was limited. The period surrounding the 2006 floods corresponded to a "pioneer phase"; pioneer vegetation was present but small enough to have limited feedback effects. Seedlings that survived the 2006 event then shifted the system toward an "ecogeomorphic" phase by the time the 2007 flood occurred, with more bidirectional vegetation-hydrogeomorphic effects. Observations of the 2010 flood suggest that by then the system had moved toward a phase in which vegetation strongly influenced physical processes, analogous to the fourth stage described by *Corenblit et al*. [2007]. Transitions among these phases may be especially rapid in the BWR, where dam-released base flows contribute to high growth rates of riparian vegetation.

[51] The trend of increasing biotic influences in the absence of large floods has not been restricted to vegetation; beaver activity has also expanded substantially in the BWR [*Andersen and Shafroth*, 2010; *Andersen et al*., 2011]. These conditions will likely persist until a large flood (analogous to the 2005 floods) can scour cattail and other vegetation, remove beaver dams, induce avulsion, or otherwise reset the system. The observed cattail encroachment and expansion of beaver highlight several other complexities of ecogeomorphic feedbacks, including the influence of shifts in the composition of the vegetation community and the potentially significant role of nonwoody (but dense and stiff) vegetation.

[52] Our observations also illustrate how feedbacks can vary spatially. The mechanism of stem-density reductions differed longitudinally (between reaches) as a result of geomorphic processes (aggradation-induced burial versus scour) and flood attenuation, helping explain differences in response between sites. In the Mineral reach, aggradation of the vegetation study bar produced by the 2006 flood increased the elevation difference between the bar and the base-flow channel, perching the bar above the water surface elevation of the 2007 event. Seedlings that survived the 2006 aggradation were those that were tall enough not to have been buried; although we expected resprouting of buried seedlings, we did not observe this. In contrast, in the Rankin reach, erosion of the bed and low-elevation bars removed seedlings. This highlights the legacy effects of geomorphic changes caused during one flood on the morphodynamics of subsequent events.

[53] The effects of the flows we observed were limited in lateral extent, such that flow forces exceeded thresholds for bed erosion and seedling mortality within the active channel but did not erode banks. Banks along our study reaches, although rich in sand and therefore lacking the cohesion of more clay-rich banks, are armored by vegetation that produces drag and focuses flow energy along the bed of the active channel. This vegetation growth along banks, which has been facilitated by dam-related flow reductions, shifts the threshold for bank erosion and channel widening to larger flood events and results in a scaled-down, equilibrium channel form compared to the predam condition. Other studies from the region have evaluated relationships between vegetation, bank strength, and floods. In Canyon de Chelly, AZ, an undammed system where tamarisk removal occurred, the presence of clay in streambanks limited flood-induced widening [*Jaeger and Wohl*, 2011]. In contrast, flooding that followed tamarisk removal along the Rio Puerco, NM produced massive bank erosion [*Vincent et al*., 2009].

[54] Surface water-groundwater exchange dynamics also produce spatial variations in morphodynamics and their feedbacks with vegetation [e.g., *Webb and Leake*, 2006]. Discontinuities in surface flows during base-flow conditions are common features in arid and semiarid rivers [*Davies et al*., 2009], but gain or loss of surface water to groundwater can also alter high flows and associated sediment dynamics. The attenuation of high flows by Planet Valley, and associated flow differences between the upstream (Alamo) and downstream (Parker) gauges in the BWR (Table 1), illustrate linkages among routing of high flows, antecedent water-table levels, and flood duration. The resulting variations in geomorphic and vegetation changes among our upstream and downstream reaches highlight how releases from dams can have vastly different downstream effects as a result of flow losses.

5.2. Implications for Flow Management

[55] Our study is representative of the type of large-scale flow experiments discussed by *Konrad et al*. [2011],

encompassing both manipulative elements, in which we provided input to the nature and timing of flow releases and measured specific biological and physical responses, and mensurative elements, in which we measured system responses over a longer time period reflecting the effects of a suite of high and low flows. The results presented above show that sequenced flood pulses favor native willow seedlings over nonnative tamarisk by causing differential levels of mortality and thus increasing the relative density of willow over tamarisk. This complements findings elsewhere that increases in relative density of cottonwood seedlings over tamarisk seedlings provide cottonwood with a competitive advantage [*Sher et al*., 2000, 2002] and that cottonwood seedlings can outcompete tamarisk seedlings following mechanical manipulations [*Bhattacharjee et al*., 2009]. This flood-induced change in seedling composition, combined with the much larger size of the willow plants, increases the likelihood that willow will outcompete tamarisk and dominate these sites, provided water availability remains high [*Stromberg et al*., 2007].

[56] This result suggests that under certain conditions, flow releases that scour or bury tamarisk seedlings before they are well established can be an effective tool to provide native taxa with a competitive advantage over tamarisk. Such scour-oriented flow releases, combined with flow management intended to promote establishment of native vegetation [*Mahoney and Rood*, 1998], could be an addition to the suite of tools used to manage tamarisk and restore native riparian vegetation [*Rood et al*., 2005; *Shafroth et al*., 2005, 2008; *Merritt et al*., 2010]. Flow releases to scour tamarisk would have a limited time window before tamarisk became large enough to withstand scour. For example, floods in ephemeral channels in semiarid southeast Spain that killed herbaceous vegetation and small shrubs did not cause mortality of tamarisk and other trees [*Sandercock and Hooke*, 2010], in agreement with other studies documenting the resilience of tamarisk to floods [e.g., *Graf*, 1978]. Our data on the change in the size distribution of plants caused by floods suggest that tamarisk are especially susceptible to scour at heights <40 cm in height and <4 mm in diameter, although this threshold of plant height versus scour potential will vary as a function of flood discharge and local hydraulic conditions, including drag associated with larger plants.

[57] In "labile" [*Church*, 2006] sand-bed rivers such as the BWR, where thresholds for mobilization of bed material and sediment transport are easily exceeded, even small managed flood releases can achieve geomorphic work of erosion, deposition, and morphologic change in the active channel. Because these processes set the template for many ecosystem processes, this geomorphic work can therefore make managed floods ecologically effective along low-elevation bars in the active channel. In gravel-bed rivers, in contrast, where bed materials are close to mobilization thresholds during bankfull events, larger floods are typically needed to affect channel change [e.g., *Church*, 2006] and/or fully mobilize the bed [*May et al*., 2009].

[58] Beyond the active channel, small managed floods are unlikely to cause geomorphic and vegetation changes in rivers such as the BWR because of both limited lateral inundation extents and the ratchet effect of vegetation along channel banks. Managed flood releases are therefore only effective for achieving ecosystem goals in the scaled-down context of the active channel rather than the entire valley bottom [*Stillwater Sciences*, 2002]. Using flows to achieve objectives such as floodplain scour, triggering avulsions, and other forms of channel reorganization that occur under reference conditions [*Trush et al*., 2000] is not likely to be feasible in most dammed rivers. This limitation has implications for vegetation community structure on floodplains [*Lytle and Merritt*, 2004].

[59] Our results illustrate how the geomorphic and ecological effectiveness of a given flood event is dependent not only on the flood magnitude and duration, but also on the recent sequencing of high-flow events. Although literature on geomorphic hydrology recognizes the importance of recent flood history [e.g., *Kochel*, 1988], discussions of the attributes of flow regimes that influence aquatic ecosystems [e.g., *Poff et al*., 1997] typically do not address sequencing. Sequencing of flood pulses can influence the establishment dynamics and evolution of vegetation communities. Releasing pulses in consecutive years, even if they are relatively small, can provide a competitive advantage to natives and prevent choking of the active channel with vegetation. In contrast, periods of extended base flow without floods, especially in systems where base flow is elevated above natural levels by dam releases, can allow vegetation encroachment in the active channel [*Shafroth et al*., 2002]. High base-flow releases designed to benefit target native species (e.g., cottonwood and willow) can have unintended consequences by benefitting nontarget species (e.g., beaver and cattail).

[60] Flood releases can have substantial downstream variations in effectiveness. Such longitudinal variations have been previously considered with respect to the effects of tributary inputs on water and sediment, which can reduce imbalances in sediment supply and transport capacity caused by dams [*Schmidt and Wilcock*, 2008]. Groundwater dynamics and vegetation characteristics can also greatly influence the downstream effects of a given flood release, as discussed above, highlighting a challenge of prescribing a single flow out of a dam for a long reach of river and the need for variability in flow prescriptions.

[61] An open question in the BWR and other dammed rivers is how reductions in sediment supply (1) influence vegetation and its feedbacks with morphodynamics and (2) should be accounted for in planning managed flood releases. The influence of sediment supply on fluvial processes is a fundamental tenet of fluvial geomorphology [*Parker*, 2004; *Schmidt and Wilcock*, 2008] that has been underlined by research on controlled flood releases on the Colorado River through Grand Canyon [*Hazel et al*., 2006; *Wright et al*., 2008; *Melis et al*., 2012]. Manifestations of reduced supply in dammed rivers, such as coarsening of bed material and incision, are well documented [e.g., *Williams and Wolman*, 1984], but their effects on vegetation are not. Dam-induced coarsening could influence vegetation both by altering the capacity of substrates to retain moisture and by increasing the critical shear stress of bed materials, thus reducing the frequency of bed (and seedling) scour. Vegetation, in turn, by altering drag and sediment deposition, can mediate relationships between sediment supply, flow, and bed material size. In the BWR, effects of supply limitation are evident immediately

downstream of Alamo Dam, where the channel is coarse (gravel-cobble, compared to sand in upstream reaches) and incised several meters below its floodplain, but the downstream extent and ecosystem implications of such changes are uncertain [*Dekker*, 2012].

6. Conclusions

[62] We observed spatial variations in mechanisms of seedling mortality and temporal variations in the strength and direction of vegetation-morphodynamic feedbacks and associated responses to managed floods. Our results illustrate how the effect of floods or other components of hydrologic regimes on riparian vegetation are mediated by geomorphic processes on both a reach scale, where coupled vegetation and geomorphic characteristics influence scour and deposition, and on a basin scale, where spatial variations of flow and sediment supply may have important influences on morphologic and vegetation responses. Our finding that controlled flood releases caused differential mortality of native willow versus nonnative tamarisk illustrates the potential to manage streamflow to influence riparian vegetation dynamics, including establishment and mortality of native versus alien species. Our investigations suggest that in sand-bed rivers, even small flood releases can affect ecogeomorphic change, albeit at a reduced scale compared to larger natural floods. Our observations also show that the geomorphic and ecological effectiveness of flow releases varies longitudinally, with distance downstream, and as a function of antecedent conditions. Dam-released floods can provide both a means of achieving downstream ecosystem objectives and of conducting experiments to develop quantitative insights into relationships between morphodynamics and ecosystem processes.

[63] **Acknowledgments.** We thank Paul Kinzel, James Johnsen, Tom Bates, Doug Andersen, Donna Shorrock, Vanessa Beauchamp, Stan Culling, John Hickey, and Robert Pleszewski for field assistance; Jill Majerus and Matthew Gilbert for sediment sieving; and Brian Steele, Jonathan Nelson and Gregor Auble for guidance. We also thank Anne Lightbody, John Stella, and three anonymous reviewers for comments that greatly improved the manuscript. This work was funded by US Army Corps of Engineers, US Geological Survey, US Fish & Wildlife Service and National Science Foundation (EAR-1025076). A portion of this research was performed while the lead author held a National Research Council Postdoctoral Research Associateship at the U.S. Geological Survey's Geomorphology and Sediment Transport Laboratory in Golden, CO. Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

References

Allred, T. M., and J. C. Schmidt (1999), Channel narrowing by vertical accretion along the Green River near Green River, *Utah, Geol. Soc. Am. Bull.*, *111*(12), 1757–1772.

Andersen, D. C., and P. B. Shafroth (2010), Beaver dams, hydrological thresholds, and controlled floods as a management tool in a desert riverine ecosystem, Bill Williams River, Arizona, *Ecohydrology*, *3*(3), 325–338, doi:10.1002/eco.113.

Andersen, D. C., P. B. Shafroth, C. Pritekel, and M. O'Neill (2011), Managed flood effects on beaver pond habitat in a desert riverine ecosystem, *Bill Williams River, Arizona USA, Wetlands*, *31*(2), 195–206, doi:10.1007/s13157-011-0154-y.

Arthington, A. H., S. E. Bunn, N. L. Poff, and R. J. Naiman (2006), The challenge of providing environmental flow rules to sustain river ecosystems, *Ecol. Appl.*, *16*(4), 1311–1318.

Asaeda, T., and L. Rajapakse (2008), Effects of spates of different magnitudes on a Phragmites japonica population on a sandbar of a frequently disturbed river, *River Res. Appl.*, *24*(9), 1310–1324, doi:10.1002/rra.1128.

Asaeda, T., P. I. A. Gomes, K. Sakamoto, and M. H. Rashid (2011), Tree colonization trends on a sediment bar after a major flood, *River Res. Appl.*, *27*(8), 976–984, doi:10.1002/rra.1372.

Auble, G. T., and M. L. Scott (1998), Fluvial disturbance patches and cottonwood recruitment along the upper Missouri River, *Montana, Wetlands*, *18*(4), 546–556.

Bhattacharjee, J., J. Taylor, L. Smith, and D. Haukos (2009), Seedling competition between native cottonwood and exotic saltcedar: Implications for restoration, *Biol. Invasions*, *11*(8), 1777–1787, doi:10.1007/s10530-008-9357-4.

Birken, A. S., and D. J. Cooper (2006), Processes of Tamarix invasion and floodplain development along the lower Green River, *Utah, Ecol. Appl.*, *16*(3), 1103–1120.

Braudrick, C. A., W. E. Dietrich, G. T. Leverich, and L. S. Sklar (2009), Experimental evidence for the conditions necessary to sustain meandering in coarse-bedded rivers, *Proc. Natl. Acad. Sci.*, *106*(40), 16,936–16,941, doi:10.1073/pnas.0909417106.

Church, M. (2006), Bed material transport and the morphology of alluvial river channels, *Annu. Rev. Earth Planet. Sci.*, *34*, 325–354.

Cleverly, J. R., S. D. Smith, A. Sala, and D. A. Devitt (1997), Invasive capacity of Tamarix ramosissima in a Mojave Desert floodplain: The role of drought, *Oecologia*, *111*(1), 12–18, doi:10.1007/s004420050202.

Cooper, D. J., D. M. Merritt, D. C. Andersen, and R. A. Chimner (1999), Factors controlling the establishment of Fremont cottonwood seedlings on the upper Green River, *USA, Regul. Rivers Res. Manage.*, *15*(5), 419–440.

Cooper, D. J., D. C. Andersen, and R. A. Chimner (2003), Multiple pathways for woody plant establishment on floodplains at local to regional scales, *J. Ecol.*, *91*(2), 182–196.

Corenblit, D., E. Tabacchi, J. Steiger, and A. M. Gurnell (2007), Reciprocal interactions and adjustments between fluvial landforms and vegetation dynamics in river corridors: A review of complementary approaches, *Earth Sci. Rev.*, *84*(1-2), 56–86, doi:10.1016/j.earscirev.2007.05.004.

Davies, B. R., M. C. Thoms, K. F. Walker, J. H. O'Keefe, and J. A. Gore (2009), Dryland rivers: Their ecology, conservation and management, in *The Rivers Handbook: Hydrological and Ecological Principles*, vol. *2*, edited by P. Calow and G. E. Petts, pp. 484–511, Blackwell Sci., Oxford, U. K.

Davies, N. S., and M. R. Gibling (2011), Evolution of fixed-channel alluvial plains in response to Carboniferous vegetation, *Nat. Geosci.*, *4*(9), 629–633, doi:10.1038/ngeo1237.

Dean, D. J., and J. C. Schmidt (2011), The role of feedback mechanisms in historic channel changes of the lower Rio Grande in the Big Bend region, *Geomorphology*, *126*, 333–349.

Dekker, F. J. (2012), Sediment dynamics in a dryland river: Grain-size variations, erosion rates, sediment mixing, and dam effects, MSc. thesis, 78 pp., Univ. of Mont., Missoula, Mont.

Dixon, M. D., M. G. Turner, and C. F. Jin (2002), Riparian tree seedling distribution on Wisconsin River sandbars: Controls at different spatial scales, *Ecol. Monogr.*, *72*(4), 465–485.

Edmaier, K., P. Burlando, and P. Perona (2011), Mechanisms of vegetation uprooting by flow in alluvial non-cohesive sediment, *Hydrol. Earth Syst. Sci.*, *15*(5), 1615–1627, doi:10.5194/hess-15–1615-2011.

Edwards, T. K., and G. D. Glysson (1999), Field methods for measurement of fluvial sediment, in *Techniques of Water-Resources Investigations, Book 3*, Chap. C2, pp. 1–89, U.S. Geol. Surv., Reston, Va.

Everitt, B. (1980), Ecology of saltcedar—A plea for research, *Environ. Geol.*, *3*(2), 77–84, doi:10.1007/bf02473474.

Freeman, G. E., W. J. Rahmeyer, and R. R. Copeland (2000), Determination of resistance due to shrubs and woody vegetation, *Tech. Rep. ERDC/CHL*; TR-00–25, 62 pp., U.S. Army Corps of Eng., Washington, D. C.

Friedman, J. M., and G. T. Auble (1999), Mortality of riparian box elder from sediment mobilization and extended inundation, *Regul. Rivers Res. Manage.*, *15*(5), 463–476.

Friedman, J. M., G. T. Auble, P. B. Shafroth, M. L. Scott, M. F. Merigliano, M. D. Preehling, and E. K. Griffin (2005), Dominance of non-native riparian trees in western USA, *Biol. Invasions*, *7*(4), 747–751, doi:10.1007/s10530-004-5849-z.

Gladwin, D., and J. Roelle (1998), Survival of plains cottonwood (Populus deltoides subsp.Monilifera) and saltcedar (Tamarix ramosissima) seedlings in response to flooding, *Wetlands*, *18*(4), 669–674, doi:10.1007/bf03161681.

Graf, W. L. (1978), Fluvial adjustments to the spread of tamarisk in the Colorado Plateau region, *Geol. Soc. Am. Bull.*, *89*(10), 1491–1501.

Grams, P. E., and J. C. Schmidt (2002), Streamfow regulation and multilevel flood plain formation: Channel narrowing on the aggrading Green River in the eastern Unita Mountains, *Colorado and Utah, Geomorphology*, *44*(3–4), 337–360.

Gran, K., and C. Paola (2001), Riparian vegetation controls on braided stream dynamics, *Water Resour. Res.*, *37*(12), 3275–3283.

Green, J. C. (2005), Modelling flow resistance in vegetated streams: Review and development of new theory, *Hydrol. Process.*, *19*(6), 1245–1259.

Hazel, J. E., D. J. Topping, J. C. Schmidt, and M. Kaplinski (2006), Influence of a dam on fine-sediment storage in a canyon river, *J. Geophys. Res.*, *111*, F01025, doi:10.1029/2004jf000193.

Hereford, R. (1984), Climate and ephemeral-stream processes: Twentieth-century geomorphology and alluvial stratigraphy of the Little Colorado River, *Arizona, Geol. Soc. Am. Bull.*, *95*(6), 654–668.

Hey, R. D., and C. R. Thorne (1986), Stable channels with mobile gravel beds, *J. Hydraul. Eng.*, *112*(8), 671–689.

House, P. K., P. B. Shafroth, and V. B. Beauchamp (2006), *Hydrology and fluvial geomorphology, in Defining Ecosystem Flow Requirements for the Bill Williams River, Arizona*, edited by P. B. Shafroth and V. B. Beauchamp, pp. 9–30, U.S. Geol. Surv., Reston, Va.

Hupp, C. R., and W. R. Osterkamp (1996), Riparian vegetation and fluvial geomorphic processes, *Geomorphology*, *14*(4), 277–295.

Jackson, W., and P. Summers (1988), *Bill Williams River Field Assessment: Hydrology, Hydrogeology, and Geomorphology*, U.S. Bur. of Land Manage., Denver, Colo.

Jaeger, K. L., and E. Wohl (2011), Channel response in a semiarid stream to removal of tamarisk and Russian olive, *Water Resour. Res.*, *47*, W02536, doi:10.1029/2009wr008741.

Johnson, W. C. (1994), Woodland expansion in the Platte River, Nebraska: Patterns and causes, *Ecol. Monogr.*, *64*(1), 45–84.

Johnson, W. C. (2000), Tree recruitment and survival in rivers: Influence of hydrological processes, *Hydrol. Process.*, *14*(16–17), 3051–3074.

Kean, J. W., and J. D. Smith (2005), Generation and verification of theoretical rating curves in the Whitewater River basin, Kansas, *J. Geophys. Res.*, *110*, F04012, doi:10.1029/2004jf000250.

Kochel, R. C. (1988), Geomorphic impact of large floods: Review and new perspectives on magnitude and frequency, in *Flood Geomorphology*, edited by V. R. Baker, R. C. Kochel, and P. C. Patton, pp. 169–188, John Wiley, New York.

Konrad, C. P., et al. (2011), Large-scale flow experiments for managing river systems, *BioScience*, *61*(12), 948–959, doi:10.1525/bio.2011.61.12.5.

Konrad, C. P., A. Warner, and J. V. Higgins (2012), Evaluating dam reoperation for freshwater conservation in the Sustainable Rivers Project, *River Res. Appl.*, *28*(6), 777–792, doi:10.1002/rra.1524.

Levine, C. M., and J. C. Stromberg (2001), Effects of flooding on native and exotic plant seedlings: Implications for restoring south-western riparian forests by manipulating water and sediment flows, *J. Arid Environ.*, *49*(1), 111–131.

Lytle, D. A., and D. M. Merritt (2004), Hydrologic regimes and riparian forests: A structured population model for cottonwood, *Ecology*, *85*(9), 2493–2503.

Magilligan, F. J., and K. H. Nislow (2005), Changes in hydrologic regime by dams, *Geomorphology*, *71*(1-2), 61–78, doi:10.1016/j.geomorph.2004.08.017.

Mahoney, J. M., and S. B. Rood (1998), Streamflow, requirements for cottonwood seedling recruitment—An interactive model, *Wetlands*, *18*(4), 634–645.

May, C. L., B. Pryor, T. E. Lisle, and M. Lang (2009), Coupling hydrodynamic modeling and empirical measures of bed mobility to predict the risk of scour and fill of salmon redds in a large regulated river, *Water Resour. Res.*, *45*, W05402, doi:10.1029/2007wr006498.

Melis, T. S., J. Korman, and T. A. Kennedy (2012), Abiotic & biotic responses of the Colorado River to controlled floods at Glen Canyon Dam, *Arizona, USA, River Res. Appl.*, *28*(6), 764–776, doi:10.1002/rra.1503.

Merritt, D. M., and N. L. Poff (2010), Shifting dominance of riparian Populus and Tamarix along gradients of flow alteration in western North American rivers, *Ecol. Appl.*, *20*(1), 135–152.

Merritt, D. M., M. L. Scott, N. L. Poff, G. T. Auble, and D. A. Lytle (2010), Theory, methods and tools for determining environmental flows for riparian vegetation: Riparian vegetation-flow response guilds, *Freshwater Biol.*, *55*(1), 206–225, doi:10.1111/j.1365–2427.2009.02206.x.

Micheli, E. R., and J. W. Kirchner (2002), Effects of wet meadow riparian vegetation on streambank erosion. 2. Measurements of vegetated bank strength and consequences for failure mechanics, *Earth Surf. Process. Landforms*, *27*(7), 687–697, doi:10.1002/esp.340.

Nagler, P. L., E. P. Glenn, C. S. Jarnevich, and P. B. Shafroth (2011), Distribution and abundance of saltcedar and Russian olive in the western United States, *Crit. Rev. Plant Sci.*, *30*(6), 508–523, doi:10.1080/07352689.2011.615689.

Nepf, H. M. (1999), Drag, turbulence, and diffusion in flow through emergent vegetation, *Water Resour. Res.*, *35*(2), 479–489.

Palmer, M. A., and E. S. Bernhardt (2006), Hydroecology and river restoration: Ripe for research and synthesis, *Water Resour. Res.*, *42*, W03s07, doi:10.1029/2005wr004354.

Parker, G. (2004), *1D Sediment Transport Morphodynamics with Applications to Rivers and Turbidity Currents, e-book*, Saint Anthony Falls Lab., Univ. of Minn., Minneapolis. http://hydrolab.illinois.edu/people/parkerg//morphodynamics_e-book.htm.

Petryk, S., and G. I. Bosmajian (1975), Analysis of flow through vegetation, *J. Hydraul. Div.*, *101*(HY7), 871–884.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg (1997), The natural flow regime: A paradigm for river conservation and restoration, *BioScience*, *47*(11), 769–784.

Polzin, M. L., and S. B. Rood (2006), Effective disturbance: Seedling safe sites and patch recruitment of riparian cottonwoods after a major flood of a mountain river, *Wetlands*, *26*(4), 965–980.

Powell, D. M., R. Brazier, J. Wainwright, A. Parsons, and M. Nichols (2006), Spatial patterns of scour and fill in dryland sand bed streams, *Water Resour. Res.*, *42*, W08412, doi:10.1029/2005wr004516.

R Development Core Team (2012), *R: A Language and Environment for Statistical Computing*, edited by R Found. for Stat. Comput., Vienna, Austria.

Richter, B. D., R. Mathews, and R. Wigington (2003), Ecologically sustainable water management: Managing river flows for ecological integrity, *Ecol. Appl.*, *13*(1), 206–224.

Rominger, J. T., A. F. Lightbody, and H. M. Nepf (2010), Effects of added vegetation on sand bar stability and stream hydrodynamics, *J. Hydraul. Eng.*, *136*(12), 994–1002.

Rood, S. B., G. M. Samuelson, J. H. Braatne, C. R. Gourley, F. M. R. Hughes, and J. M. Mahoney (2005), Managing river flows to restore floodplain forests, *Frontiers Ecol. Environ.*, *3*(4), 193–201.

Sandercock, P. J., and J. M. Hooke (2010), Assessment of vegetation effects on hydraulics and of feedbacks on plant survival and zonation in ephemeral channels, *Hydrol. Process.*, *24*, 695–713, doi:10.1002/hyp.7508.

Schmidt, J. C., and P. R. Wilcock (2008), Metrics for assessing the downstream effects of dams, *Water Resour. Res.*, *44*, W04404, doi:10.1029/2006wr005092.

Schnauder, I., and H. Moggridge (2009), Vegetation and hydraulic-morphological interactions at the individual plant, *patch and channel scale*, Aquat. Sci., *71*(3), 318–330, doi:10.1007/s00027-009-9202-6.

Scott, M. L., J. M. Friedman, and G. T. Auble (1996), Fluvial process and the establishment of bottomland trees, *Geomorphology*, *14*(4), 327–339.

Shafroth, P. B., and V. B. Beauchamp (Eds.) (2006), *Defining Ecosystem Flow Requirements for the Bill Williams River, Arizona*, 135 pp., U.S. Department of Interior, U.S. Geol. Surv., Reston, Va.

Shafroth, P. B., G. T. Auble, J. C. Stromberg, and D. T. Patten (1998), Establishment of woody riparian vegetation in relation to annual patterns of streamflow, *Bill Williams River, Arizona, Wetlands*, *18*(4), 577–590.

Shafroth, P. B., J. C. Stromberg, and D. T. Patten (2002), Riparian vegetation response to altered disturbance and stress regimes, *Ecol. Appl.*, *12*(1), 107–123.

Shafroth, P. B., J. R. Cleverly, T. L. Dudley, J. P. Taylor, C. Van Riper, E. P. Weeks, and J. N. Stuart (2005), Control of Tamarix in the Western United States: Implications for water salvage, wildlife use, and riparian restoration, *Environ. Manage.*, *35*(3), 231–246, doi:10.1007/s00267-004-0099-5.

Shafroth, P. B., V. B. Beauchamp, M. K. Briggs, K. Lair, M. L. Scott, and A. A. Sher (2008), Planning riparian restoration in the context of Tamarix control in western North America, *Restor. Ecol.*, *16*(1), 97–112.

Shafroth, P. B., A. C. Wilcox, D. A. Lytle, J. T. Hickey, D. C. Andersen, V. B. Beauchamp, A. Hautzinger, L. E. McMullen, and A. Warner (2010), Ecosystem effects of environmental flows: Modelling and experimental floods in a dryland river, *Freshwater Biol.*, *55*, 68–85.

Sheppard, P. R., A. C. Comrie, G. D. Packin, K. Angersbach, and M. K. Hughes (2002), The climate of the US Southwest, *Clim. Res.*, *21*, 219–238.

Sher, A. A., D. L. Marshall, and S. A. Gilbert (2000), Competition between native Populus deltoides and invasive Tamarix ramosissima and the implications for reestablishing flooding disturbance, *Conserv. Biol.*, *14*(6), 1744–1754.

Sher, A. A., D. L. Marshall, and J. P. Taylor (2002), Establishment patterns of native Populus and Salix in the presence of invasive nonnative Tamarix, *Ecol. Appl.*, *12*(3), 760–772.

Simpson, S. C., T. Meixner, and J. F. Hogan (2013), The role of flood size and duration on streamflow and riparian groundwater composition in a semi-arid basin, *J. Hydrol.*, *488*, 126–135.

Stella, J. C., J. J. Battles, B. K. Orr, and J. R. McBride (2006), Synchrony of seed dispersal, hydrology and local climate in a semi-arid river reach in California, *Ecosystems*, *9*(7), 1200–1214, doi:10.1007/s10021-005-0138-y.

Stella, J. C., M. K. Hayden, J. J. Battles, H. Piegav, S. Dufour, and A. K. Fremier (2011), The role of abandoned channels as refugia for sustaining pioneer riparian forest ecosystems, *Ecosystems*, *14*, 776–790.

Stillwater Sciences (2002), Merced River Corridor Restoration Plan, prepared by Stillwater Sciences, Berkeley, Calif. for CALFED Bay-Delta Prog., Sacramento, Calif.

Stromberg, J. C. (2001), Restoration of riparian vegetation in the southwestern United States: Importance of flow regimes and fluvial dynamism, *J. Arid Environ.*, *49*(1), 17–34, doi:10.1006/jare.2001.0833.

Stromberg, J. C., B. D. Richter, and D. T. Patten (1993), Response of a Sonoran riparian forest to a 10-year return flood, *Great Basin Nat.*, *53*, 118–130.

Stromberg, J. C., S. J. Lite, R. Marler, C. Paradzick, P. B. Shafroth, D. Shorrock, J. M. White, and M. S. White (2007), Altered stream-flow regimes and invasive plant species: The Tamarix case, *Global Ecol. Biogeogr.*, *16*(3), 381–393, doi:10.1111/j.1466–8238.2007.00297.x.

Stromberg, J. C., P. B. Shafroth, and A. F. Hazelton (2012), Legacies of flood reduction on a dryland river, *River Res. Appl.*, *28*(2), 143–159, doi:10.1002/rra.1449.

Tal, M., and C. Paola (2007), Dynamic single-thread channels maintained by the interaction of flow and vegetation, *Geology*, *35*(4), 347–350, doi:10.1130/g23260a.1.

Tal, M., K. Gran, A. B. Murray, C. Paola, and D. M. Hicks (2004), Riparian vegetation as a primary control on channel characteristics in multi-thread rivers, in *Riparian Vegetation and Fluvial Geomorphology*, edited by S. J. Bennett and A. Simon, pp. 43–58, AGU, Washington, D. C.

Tooth, S., and G. C. Nanson (2000), The role of vegetation in the formation of anabranching channels in an ephemeral river, Northern plains, arid central Australia, *Hydrol. Process.*, *14*(16–17), 3099–3117.

Trush, W. J., S. M. McBain, and L. B. Leopold (2000), Attributes of an alluvial river and their relation to water policy and management, *Proc. Natl Acad. Sci. USA.*, *97*(22), 11,858–11,863.

Vincent, K., J. Friedman, and E. Griffin (2009), Erosional consequence of saltcedar control, *Environ. Manage.*, *44*(2), 218–227.

Webb, R. H., and S. A. Leake (2006), Ground-water surface-water interactions and long-term change in riverine riparian vegetation in the southwestern United States, *J. Hydrol.*, *320*(3–4), 302–323, doi:10.1016/j.jhydrol.2005.07.022.

Williams, G. P. (1978), *The case of the shrinking channels—The North Platte and Platte Rivers in Nebraska*, U.S. Geol. Surv. Circ. 781, USGS, Arlington, Va.

Williams, G. P., and M. G. Wolman (1984), *Downstream effects of dams on alluvial rivers*, 83 pp., U.S. Geol. Surv. Prof. Pap. 1286, USGS, Washington D. C.

Wright, S. A., J. C. Schmidt, T. S. Melis, D. J. Topping, and D. M. Rubin (2008), Is there enough sand? Evaluating the fate of Grand Canyon sandbars, *GSA Today*, *18*(8), 4–10, doi:10.1130/GSATG12A.1.