

HYDROLOGIC ALTERATION AND ECOSYSTEM CHANGE IN SOUTHWESTERN
RIPARIAN FORESTS:
RESTORATION INSIGHTS LEARNED FROM EXPERIMENTAL FLOODING ON
THE BILL WILLIAMS RIVER,
ARIZONA

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Abstract

Historically gallery forests of *Salix gooddingii* (Goodding willow) and *Populus fremontii* (Fremont cottonwood) covered thousands of kilometers of riparian habitat in the southwestern US and northern Mexico. Much of this forest cover has been lost due to the unfavorable abiotic conditions caused by hydrologic and land use changes. Dams and ground-water pumping change flood regime attributes, such that post-dam hydrographs are characterized by lower base and peak flows and changes in the seasonality of events. The hydrological and geomorphological processes changed by dams are significant drivers of riparian forest succession, in that they create and destroy alluvial habitat and control water availability for plants. Specifically, the successful recruitment and survival of *S. gooddingii* and *P. fremontii* requires periods of inundation and flood recession to regulate water table depths and other habitat requirements. Concurrent with these changes, the invasive species *Tamarix ramosissima* (tamarisk) has invaded degraded riparian sites due to its greater drought tolerance and wider range of environmental requirements compared to native species. This paper reviews the biological and physical processes involved with riparian ecosystem change in the Southwest, culminating with a review of ecosystem change and environmental flow experimentation on the Bill Williams River, Arizona

I: The Physical and Hydrological Processes of Riparian Zones

Introduction

In their natural state, rivers perform many functions critical to humans and to natural ecosystems; among these important functions are flood mitigation, water purification, and habitat provision (Table 1). As rivers continue to be utilized for a myriad of purposes (*e.g.*, municipal/irrigation water supply, recreation, transportation, and hydropower generation), the potential for providing ecosystem services and supporting ecological functions has become threatened. River functions are particularly significant from both ecological and economic perspectives in that they are often irreplaceable by technology or too expensive to replicate. For instance, the economic value of ecosystem services is estimated at \$20,000 per hectare for wetlands alone (Postel and Richter 2003).

Recognition of the importance of river ecosystem services such as those described above is the impetus for many managers and river scientists who advocate for ecological restoration in riparian systems. In the western and southwestern United States, flow regulation resulting from large dams and groundwater pumping have led to deeper water tables and other hydrological changes which have caused rampant failures of riparian tree regeneration (Busch and Smith 1995). Concurrent with these changes, the exotic species tamarisk (*Tamarix spp.*) has invaded and (in many contexts) replaced native vegetation (Bendix and Hupp 2000) and (Glenn and Nagler 2005). In this paper I will discuss the mechanisms associated with these changes and present a case of riparian change and subsequent experimentation with environmental flow prescriptions on the Bill Williams River, Arizona.

The Fluvial Setting

Different landforms within a river corridor tend to support distinct suites of vegetation, according to their tolerance of flooding and the physical characteristics (*e.g.*, soil type, groundwater depth, etc.) of each site. Accordingly, the effects of floods on woody riparian species are both hydrological and geomorphological. Hydrological impacts to these species include mechanical damage, saturation and propagule transport, while geomorphological impacts relate to the creation/destruction of substrate. Potential effects to vegetation are influenced by the relative landscape position of each respective landform within a river corridor; according to descending elevation these include lower terraces, floodplains and the channel itself (Bendix and Hupp 1999) (Figure 1). These topographical features occupy the riparian zone –the terrestrial area of variable width which is adjacent to and influenced by a perennial or ephemeral body of water (Helms 1998).

Floodplains occupy the largest amount of area in riparian zones and can be defined hydrologically based on flood return intervals (usually, 1-3 years) or morphologically as the alluvial surface constructed by the river under current environmental conditions. These definitions are also complementary with the concept of bank-full discharges –the flow volume necessary for water to access the floodplain. Bank full discharges are also the flows which actually create the channel and subsequently deposit the sediment necessary for floodplain creation. Terraces are the highest elevation landform in the riparian zone and are the remnants of old floodplains that persist, following channel migration (Bendix and Hupp 1999). Because they are disconnected from stream processes, due to their higher relative elevation, terrace soils are coarser and

better drained than floodplains so they support more xeric species compared to those in the floodplain.

As flood discharges move across this elevation-gradient (channel-floodplain-terrace), soils become saturated and inundated differently based on their elevation above the channel. This leads to patterns of species composition and structure based on tolerance to water availability; for instance, during a drought or anaerobic respiration during periods of inundation. The hydro-period is considered to be the single most critical regulator of species composition and the functioning of wetlands such as riparian forests. The hydro-period is defined as the frequency, duration, depth, and seasonality of flooding. The duration of flooding can significantly influence the selection of species, such that the composition of floodplain communities become limited to species that are adapted to long periods of inundation and also resilient to damage from scour or burial (Brinson and Verhoeven 1999).

Fluvial Geomorphology

Erosion and sediment deposition are the two fundamental mechanisms that regulate the characteristics of a stream channel and subsequent vegetation communities. The shape of the channel combined with sediment attributes (*e.g.*, sediment loads and particle sizes) and flow characteristics -the timing, frequency, and magnitude of floods- regulate the trajectory of a river system's physical and ecological functioning.

The concept of dynamic equilibrium has been used to describe the stability of a stream and its ability to maintain its current form with repetitive cycles of deposition and erosion. Dynamic equilibrium conditions exist when the average channel size, shape, and sediment characteristics remain constant over a period of years. Average rates of

sediment erosion and deposition rates are also equivalent during dynamic equilibrium, a situation that results in stability in the channel form and stream gradient (Briggs 1996). This characteristic (continuously changing but stable landforms) is ubiquitous in river systems globally and contributes to the productivity and diversity of riparian vegetation communities.

Channel widening is an influential physical process that changes the gradient of a river and leads to fluctuations in channel shape and pattern. Widening usually occurs during periods of high-energy erosive flows, either during a single flood or during a series of smaller floods. Sediment-laden high-volume discharges tend to remove stream-side vegetation and dislodge channel-embedded woody debris, resulting in bank collapses and channel widening (Briggs 1996). Although it depends on where the widening occurs and the geology of the basin, if widening continues, portions of the channel will straighten and the gradient will increase. This ultimately leads to a deeper channel, producing a situation which requires higher magnitude flows and longer periods of time to breach the channel banks and access the floodplain. Alternatively, lower energy flows promote channel narrowing as deposited fine sediments accrete. Narrow channels require less energy to breach banks and access floodplains (Briggs 1996). Accordingly, narrow channels are considered more desirable in terms of native riparian forest sustainability; however this can vary by region and river type.

Straight channels facilitate higher water velocity and sediment loads than those that meander. Central to the straightening process is the ability for flows to break through meander bends. Straightening can occur at a variety of spatial scales ranging from stream reach to basin-wide, however the extent to which this occurs depends on the stream's

antecedent condition (*e.g.*, sinuosity, gradient, etc.) and the characteristics of the flood, including discharge magnitude and water velocity. River reaches with straight channels are generally spatially limited to streams with high gradients, but this can also ensue following anthropogenic land-use changes such as livestock grazing (Briggs 1996). Independent of causality, straightened channels promote higher velocity flows that have the potential to cause bank erosion and limit floodplains to the margins of the stream banks (Rosgen 1994).

Although they also contribute to incision, bank collapses are part of the natural evolution of streams and are considered desirable at appropriate frequencies and intensities. Actively eroding banks create and maintain diverse habitats and moderate changes in channel morphology and pattern. Erosion from headwater areas provide streams with sediment sources that are stored as channel bed materials or suspended material in the channel. This sediment aids in the creation of riparian channel landforms and leads to riparian habitats that support diverse wetland plant communities (*e.g.*, *Carex spp.* and *Juncus spp.*) that are adapted to continuous periods of inundation. Stored sediment in the channel bed also makes up the substrate required by aquatic macroinvertebrate guilds such as filter feeders, shredders, scrapers, and predators. These coarse grained materials promote oxygen exchange, provide aquatic invertebrates with protection from predators, serve as attachment sites for filter feeders and provide a food source for periphyton. Alternatively, when sediment supplies from bank collapses are too large (or when particles are dominated by fine versus coarse material), aquatic habitats are at risk of becoming buried or otherwise damaged (Florsheim et al. 2008).

Bar formation is a *depositional* process that has a significant effect on channel pattern; this landform also serves as the substrate for seedling establishment. Bars are in-channel alluvial features formed by the longitudinal movement of sediments; particularly, sand and gravel. As flows decrease in competence (*i.e.*, the largest-sized particle movable by a particular flow), sediments accumulate into strips of land called bars. The most prevalent type of bar is a point bar (Figure 2), which forms when sediments are carried from the convex side (fastest moving water) of a meander to the concave side (slowest moving water), here sediments accrete to form the bar (Brinson and Verhoeven 1999). Mid-channel bars (Figure 2) develop when in-channel sediment loads decrease in competence and deposit within the channel. Diversion of flow around the resulting bar then causes more deposition, eventually leading to island formation (Ligon et al. 1995). The repetition of these processes form meandering (laterally moving) sinuous stream profiles, which are dynamic and resilient to floods of varying magnitudes (Briggs 1996).

The last geomorphological characteristic of channels to discuss is the longitudinal profile of a stream. The longitudinal profile is the plane, or aerial view along the length of a stream. Longitudinal profiles are strongly influenced by changes in sediment loads and the resulting erosion or deposition. As described above, when the stream gradient increases, its sinuosity decreases due to the widening and straightening of the channel (Briggs 1996). The repetition of geomorphological process interacts with the heterogeneity of floodplain topography and fluctuations in flow and sediment, leading to a variety of longitudinal profiles within a single river. Several of these broad profile-types include meandering with wide floodplains (E, C, and B), straight channels with high

gradients (Aa+, A, F, and G) and braided (multi-channel) rivers (Figure 3) (Rosgen 1994).

Dams and Consequences to Flood Regimes

Flood regime characteristics include the timing (*i.e.*, season), frequency, magnitude, and duration of flood events. The dynamics of these attributes influence sediment movement and ultimately the size, shape, and function of a stream channel, but large dams have altered flood and channel attributes significantly. On a global basis, 60% of the 227 largest rivers have been altered by dams (Postel and Richter 2003) for various purposes including agricultural production, municipal and industrial water supplies, flood control, and hydro-electric power generation. Most dams in the U.S. were constructed during a period of time when comprehensive environmental assessments were not required, thus many of the resulting morphological, hydrological, and ecological effects were not fully anticipated (Rood et al. 2005).

A reduction in the variability of historic or *natural* hydrographs is among the most observable changes in post-dam hydrographs. The Glenn Canyon Dam on the lower Colorado River (Figure 4) exemplifies this trend. Large dams such as this produce more uniformity in flood discharge volumes, usually characterized by lower base and peak flows and changes in the seasonality of flood events. In the Southwest, a major consequence of reduced flood variability and magnitude has been loss of sediment which settles out in reservoirs before flows are released (Rood et al 2005). Likewise, dams alter chemical attributes (*e.g.*, dissolved O₂), which affect ecosystem processes, particularly, decomposition, and the sustainability of aquatic micro- and macro-invertebrates

populations. In both channel and riparian environments, dams also serve as physical migration barriers to native fishes, macroinvertebrates, and riparian trees such as cottonwood and willow (Konrad et al. 2012).

Impoundments alter the hydrology and associated geomorphology of rivers such that their erosive power increases (*e.g.*, due to greater channel water capacity and sediment loads) and leads to greater channel incision and water table decline (Bendix and Hupp 2000). Depending on the type of river and dam, major geomorphic consequences may include incision or aggradation, change in channel pattern, the changes in stream bed particle size, channel widening or narrowing, increased or decreased lateral channel migration, and bank collapse (Ligon et al. 1995). The geomorphological, hydrological and biological functions in riparian ecosystems are inextricably linked. Specifically, the transport of sediment, organic debris, and nutrients are the fundamental mechanisms which drive woody plant community development (Konrad et al. 2012). Recognition of these linkages has led researchers and river managers to embrace the notion that flood characteristics exert significant controls on the ecological trajectory of riparian forest communities (Table 2).

II: Riparian Forest Ecology and Ecosystem Change

***Populus fremontii*-*Salix gooddingii* Ecosystems**

Gallery forests of Goodding willow (*Salix gooddingii*) and Fremont cottonwood (*Populus fremontii*) historically covered thousands of kilometers of low elevation (<1250 m in Arizona) rivers in the southwestern U.S. and northern Mexico. According to Swift (1984) as cited in Horton et al. 2001, today these forests are among the most threatened ecosystem types in the U.S., and have lost more than 80% of their historical cover. The magnitude of native riparian forest loss across the southwest in general may not be represented in these claims, however. This estimate (>80% loss in native forest cover) comes from a single study focusing on highly regulated reach of the lower Colorado River rather than an integrated assessments based on different hydrological scenarios (Webb et al. 2007) across the range of cottonwood willow forests in the southwestern U.S.

Despite disagreements about the extent of native riparian forest change (Webb et al. 2007), hydrological modifications combined with unsustainable land use practices (e.g., forest clearing, and intensive livestock grazing) *have* changed the composition and structure of these forests throughout their range to favor more xeric species. Both willow and cottonwood are considered phreatophytic (plants which require alluvial ground water and the associated capillary fringe) and are very drought intolerant (Horton et al. 2001). The physiological requirements of these native species combined with the alteration of riparian habitats have facilitated species composition shifts to the invasive species *Tamarix spp.* (tamarisk or salt cedar), which is comparatively very drought tolerant.

Synergies associated with novel hydrology, tamarisk and native forest decline will be explored in further detail later in the paper.

Linkages between flood regimes and native seedling survival and establishment were proposed by Mahoney and Rood (1998) in the *recruitment box model* (Figure 5). According to the model, seed release and germination are cued to stream flows' seasonal patterns and discharge levels. In an unaltered flood regime, seed release occurs on the recession limb of summer floods and the successful establishment of riparian seedlings becomes limited to locations where flow drawdown rates do not exceed 2.5 cm day^{-1} . Over time as channels adjust and migrate, progressive linear arcuate bands of trees (*i.e.*, "isochrones"; Stromberg 1993) establish parallel to the channel, where the youngest cohorts establish closest to the channel and the oldest cohorts furthest (up to 200 m). Progressively, water tables become deeper further from the channel and require longer roots (maximum range of 2-4 m for cottonwoods and willows) to access water (Stromberg 1993, Glenn and Nagler 2001).

Mature cottonwoods and willows can produce up to 25 million seeds per year, so their populations are not constrained by propagule amount. Seeds are released in early spring (March-April) and require moist mineral soil at elevations of 0.6-2.0 m above the late summer stage (to reduce late summer flood scour) for successful establishment. Seeds are only viable for 1-5 weeks and usually germinate within 24-48 hours (Stromberg 1993).

Once successfully established, seedling root growth rates range from $0.6\text{-}1.3 \text{ cm day}^{-1}$; this leads to 72-162 cm of root growth during their first growing season. First year seedlings are able to survive relatively slow water declines ($2\text{-}4 \text{ cm/day}$), suggesting that

seedlings can access moisture from the capillary fringe above the water table (Horton and Clark 2001). Mature cottonwoods and willows can eventually access groundwater of depths fluctuating between 2-3 meters (Glenn and Nagler 2005) and 3-4 meters (Stromberg 1993). However, initial site characteristics exert significant controls on root morphology and the physiological condition of seedlings, which influences riparian forest succession over time.

The over-story species diversity in cottonwood-willow forests is generally low due to the disturbance regime in riparian floodplain forests. Bendix and Hupp (2001) propose that cottonwood-willow forests (and other flood-adapted associations) are held in successional abeyance because development of latter successional stages is delayed as long as the disturbance regime remains in place. Where dams and other alterations do not substantially alter hydrological processes, water tables are generally higher than adjacent uplands and disturbance is ubiquitous from seasonal flooding. This disturbance regime produces temporal and spatial heterogeneity in water table depths and inundation frequencies, which contributes to high *understory species* diversity. Several of the general plant-type associations present in cottonwood-willow understories include Sonoran interior marshlands, Sonoran riparian scrublands, and Sonoran riparian mesquite forests (Stromberg 1993).

Forest structure is highly variable in cottonwood-willow forests and can range from dense to open, with tree densities and basal areas ranging from 50 – 800 trees ha⁻¹ and 18 m² ha⁻¹, respectively. Age class and structural forest diversity in the over-story of cottonwood-willow stands can be high even when species diversity is low. Cottonwoods and willows can establish in pure or mixed stands as well as with a suite of other species

(Stromberg 1993) ranging from riparian obligates to xeric shrubs and trees such as *Prosopis* spp. (mesquite) (Table 3).

Tamarix Eco-physiology and Invasion Trends

Tamarix ramosissima (salt cedar, tamarisk) is an exotic invasive species introduced into the southwestern and western U.S. from central Asia during the mid-1880s through the 1930s. It was initially planted for flood control along railroad rights of way and as a shade tree along canals, riparian areas, and in human dominated landscapes; it continues to be planted in northern Mexico (Shafroth et al. 2007). Other invasive taxa present in these regions include *T. chinensis* (five-stamen tamarisk) *T. parviflora* (small flower tamarisk), and *T. gallica* (French tamarisk).

Currently tamarisk occupies 404,685 to 647,497 hectares from northern Mexico to central Montana and from central Kansas to central California (Shafroth et al. 2005) (Figure 6). Concurrent with its introduction, environmental stresses including groundwater pumping, dam construction and other channel and flow manipulations have facilitated tamarisk's ability to become a dominant or sub-dominant species in many riparian, wetland, xeric, and halophytic plant associations in the Southwest (on the Colorado, Rio Grande, and Pecos Rivers), the West, and in northern Mexico (Stromberg et al 2007). Tamarisk has competitive adaptations including salinity tolerance, drought tolerance, and post-fire re-sprouting abilities, all of which have led to its rapid spread of approximately 50 km/year. Along regulated southwestern rivers, these factors have led to the replacement of stressed cottonwood-willow assemblages (Glenn and Nagler 2005).

Tamarisk is a ruderal, woody shrub or small tree that reaches sexual maturity after only one year. Individuals can produce more than 500,000 small wind- and water-

dispersed seeds (Glenn and Nagler 2005). Although *Fremont cottonwood* and *Goodding willow* also produce similar sizes and large quantities of seed, tamarisk continues to do so for much later into the season than natives. Seeds are produced bi-modally, starting in April and peaking in mid-June, continuing during a second window from August to October (Horton and Clark 2001). Once established, tamarisk can form dense stands, aided by rapid growth rates (3-4 m/year) and higher leaf area indices (a measure of photosynthetic capacity) compared to cottonwood and willow (2-4 and 1-2, respectively). Although, structurally it can occur in extensive, dense monotypic stands, it can also be found in small patches or as a minor presence within a mostly native vegetation mosaic. For instance, (Shafroth et al 2005). Stromberg et al. (2007) found that differences in native versus tamarisk forest cover were attributed to the ephemeral versus perennial nature of flow and to differences between regulated versus unregulated river reaches; the former of each was dominated by tamarisk and the latter by native species.

Like *Populus fremontii* and *Salix gooddingii*, tamarisk requires moist mineral soil and low vegetative cover (to minimize competition) for establishment. Once established however, tamarisk seedlings are able to survive water stress better than *P. fremontii* and *S. gooddingii* (Horton and Clark 2001). In contrast, tamarisk seedlings also experience higher mortality rates in the presence of late spring floods because the only seedbeds available at dispersal are subject to further inundation and sedimentation/scour.

Levine and Stromberg (2001) found that *Tamarix ramosissima* seedlings were less well adapted than *Populus fremontii* in the presence of sedimentation rates exceeding 1 cm per year; annual sedimentation rates on many southwestern rivers range from 1-10 cm. Only when *Tamarix* seedlings reached 5 weeks of age and a height range of 4-6 cm

were they able to survive in higher numbers and out-perform *P. fremontii* and *S. gooddingii* seedlings. Significantly, the two native species both out-performed *Tamarix* at younger and smaller stages, suggesting that native species may have a competitive advantage over *Tamarix* in natural flow conditions. Factors including soil moisture, soil salinity, nutrient-levels, and light-levels also likely affected survival in this laboratory experiment (Levine and Stromberg 2001).

Tamarisk usually occurs on mesic sites such as river bottomlands or reservoir margins. However, it can also grow in upland communities with “*xero-riparian*” genera [e.g., *Prosopis spp.* (mesquite) and *Atriplex spp.* (saltbush)], and on de-watered riparian sites which have exceeded the environmental tolerances of cottonwood and willow (Stromberg et al 2007). Mature tamarisks have extensive roots which are able to reach deep water tables of 4-6 meters, which are quite deep relative to those cottonwoods and willows whose root depths range between 2-3 meters (Stromberg 1993) and 3-4 meters (Glenn and Nagler 2005). Tamarisk also appears to have high water use efficiency, as evidenced by its high stomatal resistance (*i.e.*, ability to retain water versus transpire water vapor), which leads to a low risk of xylem cavitation under both water and salinity stress (Glenn and Nagler 2005). When water is more available, it displays high evapotranspiration rates (0.7 to 3.4 m/year) and may lower water tables at the detriment of native species (Shafroth et al 2005, Busch and Smith 1995) and human water supplies. However, the degree to which *Tamarix* alters ground water tables through evapotranspiration has also been debated (Glen and Nagler 2005, Stromberg et al 2005) and Horton and Clark 2001).

It is a common misconception that tamarisk produces saline soils as a competitive strategy, through exudation of NaCl through its leaf litter. Glenn and Nagler (2005) noted that it is a confusion of cause and effect; it is more likely that tamarisk is a facultative halophyte that is merely tolerant of elevated salinities in floodplain soils. Novel flood regimes characterized by longer flood return intervals and lower magnitude flows often fail to remove salts which have risen to the soil surface via capillary action. Tamarisk and other halophytes, most notably, salt grass (*Distichlis spicata*), come to dominate these sites until flows of sufficient magnitude reduce salinities to levels more conducive for pherathophytes. In the absence of sufficient flood magnitudes, this causes a negative feedback loop in which tamarisk populations increases at the expense non-halophytic species such as cottonwood-willow communities.

Fire is an additional factor that aids in the spread of tamarisk. Wildfires have increased in southwestern riparian zones concurrent with a trend of drying and biological invasions. Fire return intervals in riparian zones have historically been very low due to high moisture availability, the presence of standing water in addition to low fuel loadings owed to frequent flood flows. Fire frequencies and extents have progressively increased in Southwestern riparian areas, but increased fire frequencies in riparian zones does not necessarily lead to an immediate or irreplaceable loss of native vegetation. Indeed, both cottonwood and willow have re-sprouting abilities that can serve to mitigate community shifts and other negative impacts. However, tamarisk also has post-fire stump re-sprouting abilities which seem to surpass those of native vegetation. Furthermore, tamarisk generates higher fuel loads with more flammable needle litter in greater amounts than cottonwoods or willows. The lack of scouring floods combined with the fuel

characteristics and re-sprouting ability of tamarisk puts native forests at further risk of being replaced by tamarisk.

Approaches to Ecological Restoration in Southwestern Riparian Areas

As indicated multiple times in this paper, altered hydrological and geomorphological conditions are recurring mechanisms implicated in the decline of Southwestern riparian forests. Forest succession in these ecosystems is influenced by life history attributes (*e.g.*, reproductive phenology) and environmental tolerances to drought, inundation and scour, such that recurring flood disturbance perpetuates woody pioneer species (Stromberg et al 2007b). Recognition of the synergy that exists between these physical processes and native riparian vegetation life-history traits can guide land and water managers' decisions so that treatments produce the desired composition, structure and functions in the ecosystem being restored.

Methods of restoring ecosystem indicators in riparian systems occurred as early as the late 1940s, with efforts initially focused on maintaining minimum in-stream flows designed to sustain sport-fish populations. These classifications have given way to more complex methods. Today in the field of "environmental flows" (*i.e.*, flows designed to produce explicit ecological responses), there are over 200 methods that can be grouped into hydrological rules, hydrological rating methods, habitat simulation methods, and holistic methods (Arthington et al. 2006).

Beechie et al. (2010) describe the method of process-based restoration, a paradigm that is gaining popularity in river restoration. Process based restoration addresses the root causes of degradation by re-establishing the historic rate and magnitude of physical, chemical, and biological processes that drive habitat conditions

and system dynamics. It attempts to improve ecological trajectories in a way such that minimal management actions are required for recovery so that the system is able to respond favorably to future events without intervention. Using a process-based approach also avoids the possibility of creating habitat conditions that a site cannot support because the processes themselves are driving habitat creation and change. Beechie et al. (2010) provides the following set of principles to aid in the restoration of river ecosystems using this approach: (1) target the root cause of ecosystem change, (2) tailor restoration actions to local conditions, (3) match the scale of restoration to that of physical and biological processes, and (4) be explicit about expected outcomes.

Large flow experiments are a type of process-based restoration where ecological benefits are provided to riparian ecosystems by focusing on hydrological regimes. Large flow experiments are broadly defined as scenarios in which field observations and analyses are used to test hypotheses based on physical and biological responses to different flow release characteristics. Flow experiments can be split into mensurative or manipulative experiments. Mensurative experiments are those in which ecosystem response data are collected based on observed (not set by the investigator) flow characteristics, whereas manipulative experiments utilize post-flood data which are collected based on a flood prescription designed by the investigator. Manipulative experiments may be of better comparative utility because the design allows the effects related to stream flow to be disentangled from other factors (Konrad et al. 2011).

Arthington et al. (2006) further advocate developing regional standards for environmental flows (Table 4) because using more robust indicators could aid in diagnosing cause-effect relationships as part of an adaptive management framework. Effects could then be

distinguished among those caused by flow modification per se and changes in land use or other confounding factors.

The Sustainable Rivers Project (SRP) is a program launched in 2002 by the Nature Conservancy (TNC), which utilizes process-based restoration through dam-reoperation and collaborative environmental management. TNC has worked with the U.S. Army Corps of Engineers (the Corps) to modify dam operations in terms of environmental flow prescriptions that quantify the magnitude, duration, frequency, and seasonal timing of reservoir releases to achieve specific ecological outcomes (Konrad 2010). As of 2009, SRP has worked with the Corps to develop flow prescriptions on five demonstration sites which include: Green River, Kentucky; Savannah River, Georgia and South Carolina; Bill Williams River, Arizona; Big Cypress Creek, Texas; and Middle Fork Willamette River, Oregon. The project has also been applied internationally to rivers in China and Costa Rica

Prescriptions for each river system are developed during expert workshops based on the best available data and simulation models such as the Corps' HEC-RAS or HEC-EFM Models (Arthington et al 2009). The prescriptions developed in these workshops address riverine, floodplain and estuarine ecosystem components through a range of flow conditions including floods, high-flow pulses, base flows, and extreme low flows (Konrad 2010). The ecological outcomes of these treatments are then evaluated through collaborative partnerships of federal and state agencies, universities, and NGOs (Konrad 2009). Konrad et al. (2012) subsequently designed a set of potential environmental indicators (Table 5) which managers can use to assess operational changes such as those being explored in the SRP; potential riparian vegetation indicators include (1) the areal

cover of floodplain vegetation (forest, marsh, etc.), (2) native riparian species density, richness, and age structure, and (3) density and age structure comparisons with invasive or upland species.

III: The Bill Williams River

Site Description

The Bill Williams River (BWR; Figure 7) drains 12,300 km² (Bush and Smith 1995) to 13,700 km² (Shafroth et al 2002) in the Basin and Range Physiographic Province at the northern junction of the Sonoran and Mohave Deserts (Stromberg et al. 2012). The region is hot and arid, with median annual temperatures of 21.6° C (Stromberg et al 2010) and precipitation ranging from 22 to 45 cm annually. The headwaters of the river are located in west-central Arizona's highlands at an elevation of 1,829m. The BWR then flows for approximately 61.5 km at a gradient of .002-.004 m (Shafroth et al 2002), until it converges with the Colorado River, draining into Lake Havasau at an elevation of 137 m (Stromberg et al 1998).

The BWR is primarily a sand bed river and has segments of both perennial and ephemeral stream flow, which is characteristic of braided rivers throughout the southwestern US. (Shafroth et al 2010). Channel and floodplain sediments are composed mostly (81%) of coarse sediments such as sand (67%), which require flows of 35 m³ s⁻¹ and larger to mobilize (Shafroth et al. 2002). The principal tributaries to the BWR are the Big Sandy and Santa Maria Rivers (Figure 7), which drain 3937 km² and 7278 km², respectively. Stream flow inputs are derived from a bimodal precipitation regime characterized by frontal winter rain storms (with smaller relative inputs of high elevation snow) and late summer monsoon rainstorms (Shafroth et. al. 2010).

Relative to the Lower Colorado River (which the BWR is a tributary to), the BWR is generally unaffected by anthropogenic channelization or incision. Thus, the floodplains are still subject to periodic flooding (Bush and Smith 1995). The Bill

Williams's only impoundment is the Alamo Dam (Figure 8), which is located 6.5 km downstream from the junction with the Santa Maria and Big Sandy Rivers. It was constructed in 1968 primarily for flood control and to minimize water fluctuations in the Colorado River (Shafroth et al 2012). Alamo Dam is approximately 86 m high (Webb et al. 2007) and its reservoir has a storage capacity of $1233 \times 10^6 \text{ m}^3$ (Shafroth et al. 2002); the maximum release capacity at the base of the dam is approximately $198 \text{ m}^3 \text{ s}^{-1}$ (Webb et al. 2007).

Planet Basin is an additional regulator of stream flow and is located approximately 39 km downstream of Alamo Dam. Planet Basin is a 9.5 km long reach with wide permeable channels and deep alluvium (Figure 7), which (despite being natural) decreases peak flow magnitudes and alters the base flow downstream for approximately 17.7 km (Shafroth et al. 1998). The topography along the BWR is influenced by the Buckskin and Bill Williams Mountains where it flows through a series of narrow canyons and wide alluvial basins. Human use is minimal downstream of the Alamo Dam, with agriculture being limited to a single cotton farm along 2 km of the river. Grazing currently only occurs in a small area within Planet Basin (Shafroth et al. 2002).

The forested communities along the BWR are dominated by several woody pioneer species, typical of southwestern riparian ecosystems. These include Fremont cottonwood (*Populus fremontii*), Goodding willow (*Salix gooddingii*), tamarisk (*Tamarix ramosissima*), seep willow (*Baccharis salicifolia*), mesquite (*Prosopis spp.*), arrowweed (*Tessaria sericea*) and burro-brush (*Hymenoclea monogyra*); adjacent upland areas are dominated by Sonoran Desert Scrub communities (Shafroth et al 2002).

The Bill Williams River National Wildlife refuge is located on the lower third of the BWR, where the U.S. Fish and Wildlife Service manage approximately 19 km of the riverside habitat. This area comprises 930 ha of riparian forest and 202 ha of cattail (*Typha spp.*) marshes. The reserve boasts more than 343 species of bird (2 of which are endangered species -the southwestern willow flycatcher (*Empidonax traillii extimus*) and the Yuma clapper rail (*Rallus longirostris yumanensis*)), 56 species of mammal, and 28 reptiles and amphibians (Webb et al. 2007).

Flood Regime and Channel Modification

The primary effects of dams on rivers are changes in the pattern of stream-flow, progressively leading to physical habitat modification followed by shifts in the structure and composition of riparian vegetation. Accordingly, it is crucial to establish the degree and type of hydrological modification to illuminate the mechanisms contributing to structural and compositional shifts in physical habitats and biological communities.

Shafroth et al (1998) examined differences in median, mean and maximum annual peak flows on the BWR, comparing pre- and post-dam periods of 1941-1968 and 1969-1996, respectively (Figure 9). Annual peak flows have decreased from 416, 223 and 1943 m³ s⁻¹, respectively in the pre-dam period to 44, 18 and 198 m³ s⁻¹, respectively in the post-dam period. Ten-year peak flood magnitudes decreased from 1397 to 148 m³ s⁻¹ in the post-dam era (1968-1997), while mean annual stream flow magnitudes increased from 2.6 to 4.4 m³ s⁻¹; these increases were attributed to higher precipitation rates in the post-dam era. Summer (May-September) flow magnitudes also increased (from <1.5 to 2.5 m³ s⁻¹, due mostly to dam operation), while winter and spring flows decreased in the

post-dam era. Lastly, peak instantaneous flows have decreased by approximately 77% from 238.1 to 183.9 m³ s⁻¹ in the post-dam period (Shafroth et al. 2002) (Table 6).

Shafroth et al (2002) found significant correlations between May-September flood flow magnitudes and intermittency and channel width; channels were wider when flood magnitudes were higher, and summer flows were lower where flow was ephemeral versus perennial. Examination of historical aerial photographs revealed that channels became wide in 1953 following discharges of 1,843 and 1,062 m³ ha⁻¹ in 1951 and 1952, respectively. These channels narrowed slightly from 1953-1966 and widened again in 1966-1967 in response to 1186 and 1101 m³ sec⁻¹ flows. Ultimately, dam-induced hydrological changes led to a 77% (average of 111m) increase in channel narrowing, the majority of which occurred in the late 1970s and 1980s (Shafroth et al. 2002).

Pre- and Post-dam Forest Composition and Structure

Shafroth et al. (2002) evaluated historical photographs of the BWR and the unregulated Santa Maria River to assess changes to forest structure in the post-dam period. Because the Santa Maria River is a tributary to the BWR and has similar topographical influences and hydrology, it was chosen as a reference for which to compare post-dam vegetation structure and composition along the BWR. Pre-dam stands of riparian forests on the BWR were less expansive, smaller, and less dense than those in the post-dam era. Between 1953 and 1996, the total amount of floodplain area occupied by vegetation increased by 62% from 315 to 509 ha, whereas the cover in terrace environments during the same period increased by 23% from 542 to 667 ha. Above average precipitation in the late 1970s and 1980s contributed to an increase in cumulative

summer flows (Shafroth et al. 2002). Accordingly, during this period of time, seasonal and annual droughts were reduced on ephemeral stretches where drought stress would have limited vegetative cover. Peak flood magnitudes also decreased in the post-dam period, due in-part to the maximum release capacity ($198 \text{ m}^3 \text{ sec}^{-1}$) at the base of Alamo Dam. Decreased flood magnitudes have minimized the risk of scour and anaerobic stress from inundation. Thus, vegetative cover has expanded onto former channel environments where alluvial water table depths remained relatively shallow ($<2.8\text{m}$) and within the thresholds necessary for cottonwood and willows (Busch and Smith 1995).

According to Shafroth et al. (2002), on the BWR stem basal area is the highest in stands of *P. fremontii* and *S. gooddingii* (Table 7). *Populus-Salix* forest types currently account for 53% of the total basal area ($45 \pm 7 \text{ m}^2 \text{ ha}^{-1}$) compared to *Tamarix* forest types ($24 \pm 4 \text{ m}^2 \text{ ha}^{-1}$) in the post-dam period (Figure 10). Trends were similar on the Santa Maria River, but *Tamarix* accounts for less of the total basal area (29%) relative to *Populus-Salix* forest types (7 ± 2 vs. $24 \pm 6 \text{ m}^2 \text{ ha}^{-1}$, respectively). On the BWR, the live basal area ($12.5 \pm 4.0 \text{ m}^2 \text{ ha}^{-1}$) of all species (*P. fremontii*, *S. gooddingii* and *T. ramosissima*) is concentrated low in the bottomland ($<3\text{m}$ above the thalweg), with the remainder ($1.8 \pm 0.5 \text{ m}^2 \text{ ha}^{-1}$) located above this elevation. Relative differences of landscape position and the concentration of basal area were less pronounced in pre-dam reference sites.

Stem density comparisons between the Bill Williams and Santa Maria Rivers (*i.e.*, “pre- and post-dam”, respectively) revealed that a higher proportion of stems (of all 3 species) was concentrated $<3 \text{ m}$ above the thalweg ($14,839 \pm 5,118 \text{ stems/ha}$ vs. $5,816 \pm 1,971 \text{ stems/ha}$), with smaller increases of stem density $>3 \text{ m}$ above the thalweg in the

post-dam period $-12,898 \pm 4,576$ stems/ha vs. 1743 ± 708 stems/ha (Stromberg et al 2002). Currently, stem density is highest for *Tamarix* on the BWR (501 ± 86 stems/100m²) compared to native *Populus-Salix* types (351 ± 96 stems/100m²). The same trend (208 ± 34 and 126 ± 26 stems/100m²) was observed on the unregulated Santa Maria River, although stem density is much lower overall (Figure 10) (Stromberg et al 2012).

Herbaceous and woody species richness of all plant-types, herbaceous and perennial annuals, trees and shrubs, wetland and non-wetland species, and exotics and natives, is higher for *most* categories along the unregulated Santa Maria River versus the BWR (Figures 11-13). The diversity of annuals was two times greater on the Santa Maria River versus the BWR; annuals were also the plant category with the greatest differences between sites. A possible explanation for the disparity of annual diversity between sites is that the successful establishment of annuals on the BWR has been reduced from increased forest cover, which inhibits the colonization of annuals due to a lack of recruitment sites and adequate sunlight. Herbaceous perennials and woody species did not differ significantly among sites or season of investigation (spring vs. fall). Both wetland and non-wetland plant groups were markedly richer in the Santa Maria relative to the Bill Williams River. Lastly, both rivers had similar percentages of exotic species - 30% for the BWR and 28% on the Santa Maria- but the number of exotic species encountered was greater at the Santa Maria River.

Soil electroconductivity (a measure of soil salinity) was not significantly correlated with species richness in any of these categories nor was soil texture. Rather, overall lower richness in all plant types along the BWR was attributed to increases in woody species recruitment. The expansion of woody species (due to reduced flood

magnitudes, as discussed above) increased canopy cover resulting in decreased light levels and thick litter layers which can limit the germination success and survivorship of small-seeded colonizing species. Thus, forest biomass structure (as opposed to forest composition) seems to be the mechanism which has reduced species richness in forest understories and other niches where herbaceous species are found along the BWR (Stromberg et al 2012).

Modelling Ecosystem Responses to Environmental Flows

Shafroth et al. (1998) used seed dispersal phenology combined with physical variables such as flow attributes, soil texture, and soil electroconductivity to develop a model of germination. This was then applied to illuminate the combination of optimal flood attributes (*e.g.*, rate of recession flows) and reproductive phonological characteristics for the germination success of native riparian trees. High flow releases in January ($60\text{--}70\text{ m}^3\text{ s}^{-1}$), February ($145\text{ m}^3\text{ s}^{-1}$) and March ($198\text{ m}^3\text{ s}^{-1}$) were executed in 1993, gradually receding from 21 to $11\text{ m}^3\text{ s}^{-1}$ in April until they reached a low of $5\text{ m}^3\text{ s}^{-1}$ in October. In 1995, this was repeated but flows receded to $1.1\text{ m}^3\text{ s}^{-1}$ immediately following the March high flows to simulate drought stress.

In 1997 *P. fremontii* began dispersing seed on downstream reaches of the BWR from February 19 to April 13 and from February 26 to April 27 on upstream reaches. *S. gooddingii* began dispersing seed on March 22 and March 29 (on down and upstream sites, respectively), continuing through June 20 at both sites. *T. ramosissima* dispersal began March 29 and April 6 (at up and downstream sites, respectively) and continued

dispersing after October 10 (Figure 14), at which seed dispersal was no longer monitored (Shafroth et al. 1998).

The conclusions drawn from this model were similar to those of the Mahoney and Rood (1998) recruitment box model (Figure 5) in that germination locations were a function of seed dispersal and moist alluvial sediment. According to the recruitment box model, the maximum survivable water level decline for *Populus* seedlings, is $<2.5 \text{ cm day}^{-1}$, however it has also been documented that seedlings can survive recession rates of up to 10 cm day^{-1} with reduced growth rates (Mahoney and Rood 1991, 1992; Shafroth et al 1998). The Shafroth et al. (1998) model revealed successful germination using gradual flood recession rates, where the 1993 and 1995 *Populus* cohorts survived $1.2\text{-}4.4 \text{ cm day}^{-1}$ and $2.8\text{-}4.2 \text{ cm day}^{-1}$ recession rates, respectively.

With regard to the spatial heterogeneity in seedling establishment patterns, the model also corroborates previous investigations which have suggested that the optimal recruitment zone for *Populus* seedlings is 60-150 cm above the annual low water level (Mahoney and Rood 1998; Shafroth et al 1998). This elevational range reduces the risk of *seedling scour* from high magnitude flows and minimizes *seedling desiccation* caused by the deep water table depths and low soil moisture levels associated with periods of low flow. For all species, the Shafroth et al. (1998) model was 95% accurate in predicting germination failure, whereas germination success was predicted with 9-19% and 20-42% accuracy, respectively for the 1993 and 1995 cohorts. Live basal area, maximum annual ground water levels, and maximum flow recession rates were the variables that best differentiated between germination success and failure.

A model was also developed by Shafroth et al. (2010) to guide environmental flow prescriptions on the BWR as part of the TNC's Sustainable Rivers Project. During a 3-day workshop in 2005, 50 scientists and managers developed a conceptual hydrology-ecology model (Figure 15) for aquatic macroinvertebrates, fish, and riparian plants to assign a set of flow requirements for the BWR -based on the timing, magnitude, duration, frequency and the rate of change for particular ecological processes. The conceptual model was then integrated with (1) physical system models and ecological responses and (2) empirical relationships between flow and ecological responses to develop the Ecosystem Functions Model. This model uses riparian tree seedling establishment requirements to produce spatially explicit predictions of seedling recruitment locations in a geographical information system; it also incorporates the effects of small experimental floods on the comparative mortality of native and exotic riparian trees.

Between November 2004 and March 2005, a series of experimental flows was released from the base of Alamo Dam, ranging from 150 and 204 m³ sec⁻¹. In March 2006 and March 2007 flows of 69 m³ sec⁻¹ for 48 hours and 29 m³sec⁻¹ for 16 hours were released to promote riparian seedling recruitment. Although these events were small compared to historic releases on the BWR (flood return intervals of these flows were 1.7, 1.5 and 3 years, respectively), the floods resulted in substantial geomorphic changes, including scour and deposition of bars, removal of beaver dams, conversion of lotic to lentic habitats, and exposing bare substrates. Using the US Corps of Engineers HEC-EFM Model combined with published data related to seedling phenology, Shafroth et al. (2010) determined that the maximum flow recession rates for *P. fremontii*, and *T*

ramosissima were 6 cm day⁻¹ and 4 cm day⁻¹ for *S. gooddingii*, all based on 7-day flow releases.

T. ramosissima seedling density was much higher than that of *S. gooddingii* prior to the 2006 flood releases. However, the diameter and height of *Salix* seedlings was greater than *Tamarix*. After the floods, there was a greater reduction in *Tamarix* compared to *Salix*, suggesting that floods can lend a competitive edge to native species. At the regional scale, the findings of this research have contributed to a better understanding of the empirical relationships between flow regimes and ecological responses so that future releases can be conducted in a way that promotes specific ecological outcomes (Shafroth et al. 2010).

Further evidence of the physiological requirements of riparian seedlings was provided by Shafroth et al (2000), who investigated the comparative survival and community composition/structure of *P. fremontii*, *S. gooddingii* and *Tamarix* spp. Three sites were compared along the BWR from 1995 to 1997 and stem density and basal area was assessed based on the responses of riparian seedlings to increasing ground water depth. The site with greatest change in ground water depth (dropping from 0.86 m in 1995 to 1.97 m in 1996) corresponded to 92-100% mortality in *Populus-Salix* types and 0-13% mortality in *Tamarix*. The site with the deepest overall groundwater (2.55 m in 1996) but the least change between years (2.0 m in 1995), displayed lower *Populus-Salix* mortality and increases in basal area.

Shafroth et al.'s (2000) study suggests that changes in water depth are comparatively more important to survivorship and growth patterns of native species than the absolute depth of the water table. *Tamarix* has greater water use efficiency than either

Populus or *Salix* and can maintain high rates of photosynthesis at low water potentials, whereas native species are susceptible to xylem cavitation in stressed conditions. *Populus* and *Salix* also reduce leaf area in response to drought conditions, while *Tamarix* maintains high leaf area (Shafroth et al 2000). This allows *Tamarix* -but not *Populus* and *Salix*- to continue to allocate resources to root elongation and thus prevent drought-induced mortality.

Physiological Responses to Alluvial Water Availability

As already discussed, water availability significantly influences plant-level physiological functioning, and this regulates the structure and function at the population and community levels. Both *Populus* and *Salix* are known to experience drought-induced xylem cavitation and overall reduced physiological condition resulting from decreased moisture availability (Horton et al. 2003). Specifically, the reduction in xylem hydraulic conductivity in obligate phreatophytes progressively leads to tissue desiccation, loss of turgor, stomatal closure, reduced photosynthesis, and branch and crown mortality (Horton et al. 2001).

Horton et al (2001) investigated the physiological effects of water stress among *Populus fremontii*, *Salix gooddingii*, and *Tamarix chinensis* along ephemeral reaches of the BWR and the unregulated Hassayampa River. They explored how increases in groundwater depth affect stomatal conductance, net photosynthesis, and canopy dieback, illuminating different relationships to groundwater depth among species and study sites. Maximum groundwater depths on the Hassayampa River were greater than those on the

BWR (7 m and 4 m, respectively) and resulted in different physiological responses in all species, with natives being relatively more sensitive to water stress at both rivers.

At the Hassayampa River site, Horton et al. (2001) observed reduced predawn water potentials and leaf gas exchange rates for *P. fremontii* and *S. gooddingii* at groundwater levels exceeding 3 m. As an adaptive strategy to improve whole-plant water relations, there was evidence of canopy and branch dieback in these species but it was insufficient to prevent mortality where groundwater depths exceeded 3 m. Along the BWR, both *P. fremontii* and *S. gooddingii* experienced canopy dieback and mortality when ground water depths increased above a threshold level of 2.5-3.0 m. However (presumably due to relatively shallower maximum groundwater depths on the BWR), this did not adversely affect photosynthetic rates nor did it result in tree mortality. *T. chinensis* did not experience reductions in predawn water potentials, or leaf gas exchange rates on the BWR, so canopy dieback and tree mortality was not observed for this species. Reductions in predawn water potentials and leaf gas exchange we observed on the Hassayampa River, but these responses did not result in canopy dieback or tree mortality (Horton et al. 2001).

Horton et al. (2003) found similar trends in physiological condition among *Populus fremontii*, *Salix gooddingii*, and *Tamarix spp.* based on depth to groundwater and the comparative responses in water potential, leaf gas exchange rates, and vapor pressure deficits in these species. During the summer of 1997, mean stream flows along an 8.5 km reach of the BWR (immediately upstream of Lake Havasu) were 0.14, 0.10, 0.03 and 0.00 m³ sec⁻¹ (from May to August, respectively); these flows corresponded to increasing depths to ground water throughout the summer.

Decreased stomatal conductance was observed in *S. gooddingii* with increasing depth to groundwater however it did not correspond to decreased carbon gain. *P. fremontii* experienced less sensitivity to water table declines, where water availability only decreased in August when inflows ceased. Leaf gas exchange rates in *P. fremontii* appeared to be more sensitive to water stress than *S. gooddingii*, as evidenced by high leaf-to-air vapor pressure deficits which resulted in decreased stomatal conductance and reduced photosynthesis where ground water was deep. In contrast, *Tamarix spp.* was the least sensitive to water stress and showed no reduction in water availability over the range of hydrological conditions investigated. Furthermore, *Tamarix spp.* had the highest photosynthetic rate of the three species, which contributed to higher water-use efficiency.

While community-level mortality responses were not documented in the Horton et al. (2003) study, it can be inferred from these data that water stress is a major mechanism contributing to structural and compositional shifts in these riparian forests. The findings from these studies further support claims that *Tamarix* is more tolerant of flow perturbations, giving it a competitive advantage with regard to drought tolerance.

Conclusions

This review underscores the importance of integrated approaches to contend with the legacy effects of human caused hydrological modifications such as the impoundment and diversion of flow and groundwater pumping. Multiple lines of evidence presented here suggest that native riparian forest sustainability in the Southwest can be partially supported under more natural flood regimes. The basis of this claim is grounded on woody riparian species' adaptations to historical flood regime attributes –the frequency, timing, magnitude and duration of flow events. Specifically, when flows remain in sync with the reproductive phenology of riparian woody species (e.g., the timing of seed production and dispersal), physical habitat conditions are produced that support both their establishment and continued presence on the landscape (Shafroth et al. 2010, Horton and Clark 2001). The ecophysiological mechanisms of decline (e.g., xylem cavitation, sodium toxicity) can be mitigated by returning flows to a more historic pattern.

Restoration Using Tamarix Control

When historical hydrological functions are restored, native species appear to be more competitive with the invasive taxa *Tamarix spp.* The spread and dominance of tamarisk in southwestern riparian forests seems to be due to novel habitat conditions rather than inherent superiority based on propagule abundance. It is highly unlikely that *Tamarix* will be eradicated though either flow management or direct control, however populations can be reduced so that it is a component rather than a dominant species in these ecosystems. If this is accomplished, riparian ecosystem services and the perpetuation of native plant communities can be sustained.

Although attempts were not documented on the Bill Williams River to directly control *Tamarix spp.*, other researchers ((Bhattacharjee et al. 2006, Taylor and McDaniel 1998, Taylor et al. 2006) have shown that riparian communities can be successfully shifted from exotic to native species dominance using direct control combined with flow restoration. On the Bosque del. Apache National Wildlife Refuge along the Middle Rio Grande River Valley in central New Mexico, *Tamarix* removal was combined with irrigation (through levee management) and native plantings in an attempt to improve the status of native riparian establishment and succession. Restoration treatments initiated in the late 1980s incorporated a combination of chemical, and mechanical control measures to remove *Tamarix*. Mechanical methods used bulldozers to remove the root crowns from the soil, resulting in mortality rates of 97-99% at a cost of \$1500-\$1700/ha. Foliar herbicide treatments (using .25% *Imazapar* at the rate of 140L ha⁻¹) resulted in 60-80%, 90-95%, and 93-95% mortality rates at costs of \$4000-\$6200/ha, \$400-\$450/ha, \$240-\$280/ha for small scale, helicopter, and fixed wing aircraft applications, respectively (Shafroth et al 2005; Taylor and McDaniel 1998). Follow-up treatments for all methods were also necessary and are reflected in the costs and mortality rates presented.

In scenarios where seed sources of native species are lacking, supplemental plantings will be required as well. This was the case in the Taylor and McDaniel (1998) study where a large wildfire killed the majority of over-story cottonwood trees. Pole stock was planted for *Populus fremontii* at a cost of \$958-1332/ha, however this practice was abandoned due to its high cost and replaced with seedling planting at a cost of \$3.75 per planting or \$896/ha. Bhattacharjee et al. (2006) supplemented seed sources in a different investigation at the Bosque del. Apache National Wildlife Refuge by placing

seed-bearing cottonwood branches at intervals of 5-7 meters along the water's edge as prevailing winds dispersed the seeds.

Ten years after restoration treatments, the sites at the Bosque were re-visited and the ecosystem was assessed for changes resulting from *Tamarix* removal and native plantings. *Populus* leaf volume increased from 25.5 to 63.3% of the community and its over-story canopy cover was >70%. Taylor and McDaniel (2006) suggested that low initial densities of *Populus* seedlings allowed a few individuals to grow rapidly and dominate the over-story. While *Tamarix* remained common on these sites (41% of the total species composition), its over-story coverage was minimal (<10%) and its height was less than half that of *Populus*. This suggests that beyond the establishment phase, *Populus* is more competitive than *Tamarix*, given that the hydrological requirements of *Populus* (e.g., depth to groundwater of <3 m) are maintained. Healthy *Populus* seedlings (i.e., those not constrained by hydrology) can decrease the growth and survival of *Tamarix* seedlings through competition, relegating the remaining population to the forest under-story (Shafroth et al 2005).

An additional method of *Tamarix* control is through biological control with the Tamarisk leaf beetle (*Diorhabda elongata*), which originates from the native range of *Tamarix* in northwestern China. During the late 1960s, the US Department of Agriculture Research Service initiated a biological control research program to determine the suitability and potential non-target impacts of releasing the beetle. This resulted in the first introduction of the species to the Humboldt River in Nevada during the summer of 2001 (Hudgeons et al 2007).

Initial releases of *D. elongata* in Nevada resulted in the complete defoliation of nearly 20,000 *Tamarix spp.*, but new growth was present within several weeks. Despite this recovery, live tissue volumes decreased and resulted in decreased evapotranspiration, increased wildlife use and increased biodiversity. *D. elongata* is also becoming established across eastern Utah and defoliation is expected to occur annually in the future. Other releases in Wyoming, California, and Colorado have resulted in only moderate success. No releases have been shown to be successful south of 38° N, as *D. elongata* requires day lengths in excess of 14.5 hours which do not occur in southern latitudes; other biotypes adapted to these lower latitudes are currently being explored (Dudley 2005; Dennison et al. 2009).

Whether *D. elongata* will prove to be an effective mitigation tactic for *Tamarix* invasions has yet to be determined and will depend upon rigorous spatially explicit monitoring programs with inter-state, inter-region and inter-agency coordination. While *Tamarix* control measures do contribute to increased performance in native species, it should be repeated that those responses are dependent upon sufficient seed sources and favorable hydrological conditions for native species. If these additional requirements are not satisfied, *Tamarix* control, regardless of the method employed, will likely be ineffective.

An additional factor to take into consideration with regard to *Tamarix* control is the effect on wildlife populations, such as those of the federally endangered bird, the southwestern willow flycatcher (*Empidonax traillii extimus*). The southwestern willow flycatcher now uses *Tamarix* patches as breeding habitat and it has been documented that the productivity and survivorship of this species are similar among native and *Tamarix*

forest types (Sogee et al. 2003; Shafroth et al 2005). However, in critical habitats where the southwestern willow flycatcher has actively bred, control efforts have been curtailed. Habitat use in *Tamarix* may have been necessitated by the degradation of its primary habitat, cottonwood-willow forests. Factors such as patch size, vegetation phenology and structure, microclimatic conditions, proximity to open water, and rainfall patterns also influence the degree to which this species successfully utilizes non-native habitat types and should be considered prior to initiating a control program (Shafroth et al. 2005).

Lessons from the Bill Williams River

The Bill Williams River presented a unique opportunity to observe the mechanisms of riparian decline and to prepare prescriptions to restore hydrological functions for the support of native riparian forests. The BWR is highly unique in the southwestern US with regard to the small degree of alteration and its use by various interests. Compared to many large rivers in the Southwest, land ownership patterns along the BWR are dominated by public agencies and water rights are not legally tied to agricultural industries. This allowed experimental stream-flow prescriptions to be employed without the constraints of competing water users such as hydropower, agricultural, municipal, and industrial interests which are ubiquitous in other locations. Additionally, with the exception of the Alamo Dam, most of the BWR is unconstrained by dams and water diversions. Discharge levels and their associated physical and ecological responses could be correlated to flow release treatments from Alamo Dam and were not confounded by multiple control structures or impoundments.

Results of the Shafroth et al. (1998) and (2010) modelling studies corroborated the utility of the concepts presented in the Mahoney and Rood (1998) Recruitment Box model. It is well recognized that there are significant correlations between flow characteristics and the effects on physical habitats and riparian forest recruitment and succession; however quantitative estimates of these requirements vary significantly spatially and temporally. The Shafroth et al. (2010) model, for example suggested that flood recession rates on the BWR should not exceed 6 cm day^{-1} for *P. fremontii*, or 4 cm day^{-1} for *S. gooddingii*, while the Shafroth et al. (1998) model provided a range of $1.2\text{-}4.4 \text{ cm day}^{-1}$ and $2.8\text{-}4.2 \text{ cm day}^{-1}$ for *P. fremontii* at two different sites. More conservative rates of 2 cm day^{-1} were also provided by Mahoney and Rood (1998), which illustrates the need for additional studies of seedling requirements in different climatic and geomorphic contexts.

In order for restoration attempts in degraded riparian sites to be successful, more coordinated and strategic efforts are needed in the future. Collaborative partnerships such as those described for the Sustainable Rivers Project on the Bill Williams River will be required to fine tune objectives and to balance the needs of ecosystems with those of society. Relatively modest flow management programs can mitigate undesirable responses in riparian forests from flow perturbation with minimal cost or infrastructure when executed effectively. To achieve this success, flow-ecological response models, rigorous monitoring programs and a commitment to managing flows for ecological purposes will be required. This paradigm is gaining support in the scientific and management communities, however more research and experience is also needed to fine-tune the social and policy processes at play in adaptive management programs. These

needs are becoming even more pertinent as unknown patterns of drying and warming from climate change ensue with the concurrent increase in human demands for water, land and other natural resources.

Tables and Figures

Table 1: Ecosystem services provided by rivers, wetlands and other fresh water ecosystems (Postel and Richter 2003)

Ecosystem Service	Example/Benefits
Water Supply	>99% of municipal, industrial, and irrigation water comes from rivers and fresh water systems
Food Supply	Fish, waterfowl, clam, and mussels provide food to humans and wildlife
Water Purification	Wetlands filter and transform nutrients and plants assimilate excess nitrate and ammonium
Flood Mitigation	Flood waters are stored in floodplains and wetlands after heavy precipitation
Drought Mitigation	Waters stored in floodplains and wetlands are slowly released as seasonal shortages occur
Habitat Provision	Rivers, streams, floodplains and wetlands provide shelter, food, and breeding grounds for various wildlife
Soil Fertility	Soil flora and fauna transform and utilize flood-deposited nutrients for use by plants
Nutrient Delivery	Floods carry nutrient from off-site to both floodplains and estuaries
Salinity Regulation	Floods mix salt and fresh water in estuaries, and remove accumulated salts in floodplains
Aesthetic/Cultural Values	Rivers are repositories of scenic beauty and solitude and enhances the quality of human life
Recreation	Swimming, fishing, boating, hunting, wildlife viewing, etc.
Biodiversity	High numbers of niches promotes coexistence of diverse assemblages of plants, animals, and other biota

Table 2: Examples of riparian forest alteration and structural and ecological consequences (Brimson and Verhoeven 1999)

Alteration	Structural Change	Consequences
Impoundment and Upstream Effects	Loss of forest biomass and increase in water depth	Reduction in NPP, nutrient cycling, and other processes. Restricts migration
Impoundment and Downstream Effects	Increase in channel capacity, reduction in sediment supply, moderated flows	Increases scour, restricts migration through fragmentation.
Channelization	Increase in channel gradient, leading to decreased sinuosity	Reduces over-bank flows, reduces/eliminates floodplain plant recruitment area.
Channelization	Increase in channel capacity	Decreases overbank flows, leading to reductions in exchange of organisms, nutrients, and sediments between the channel and floodplain
Channel Restriction from Levees	Restricts floodplain water storage	Increase scour and deposition on remaining floodplain; reduces/eliminates floodplain for channel-dwellers
Livestock Grazing	Changes age structure of forest; stream banks deteriorate	Reduces NPP and sediment retention
Timber Harvest	Reduces biomass stocks and nutrient availability	Alters forest and trophic structure

Table 3: *Salix gooddingii*-*Populus fremontii* forest structure and reproductive characteristics (Stromberg 1993)

Forest Structure, Composition and Form Characteristics	Reproductive Characteristics and Requirements
Tree Density 50-800 trees/hectare	Seed Production 25,000,000 seeds/year
Maximum Size (height x diameter) 30m x 3m	Age to Sexual Maturity 5 to 15 years
	Seed Dispersal Period March-May
Basal Area 18 m ³ /hectare	Seed Viability Period 1-5 weeks
Canopy Cover/ Forest Structure >85%/ open, park-like or dense	Seed Germination Period 24-48 hours
Life Span 100-150 years	Age to Sexual Maturity 5 to 15 years
Co-occurring Woody Species <i>Fraxinus pennsylvanica</i> (green ash) <i>Prosopis pubescens</i> (screw bean mesquite) <i>Prosopis glandulosa</i> (western honey mesquite) <i>Prosopis velutina</i> (velvet mesquite) <i>Sambucus Mexicana</i> (elderberry) <i>Rhamnus spp.</i> (buckthorn) <i>Tamarix spp.</i> (tamarisk, <i>Tamarix</i>)	Seedling Growth Rates 6-13 mm/day
	Optimum Flood Recession for Survival <2.5 mm/day
	Soil Moisture Requirements for Survival >10%

Table 4: Steps in developing regional environmental flow standards (Arthington et al. 2006)

Step	Explanation
1. Develop classification for reference streams	Classification based on measures of flow magnitude, the frequency, timing, and duration of floods
2. Develop frequency distributions for each flow variable in each class	Using statistical modelling combined with gage data, drainage basin and climate data, estimate the flow metrics relevant to regional classifications
3. Compare frequency distributions from flow-modified streams with reference conditions in the same class	Flow modification must be defined in terms of deviation from reference conditions for each stream class to
4. Develop flow-response relationships for ecological health data from reference and flow modified streams for each flow variable	Develop empirical response curves for each ecological indicator and correlate to stream class and relevant flow characteristics

Table 5: Potential indicators for evaluating for prescriptions in dam re-operation (Konrad et al. 2012)

Indicators			
Floodplain Vegetation	Fish	Invertebrate	Physical Habitat
Floodplain forest (or other wetland habitat) area	Relative abundances of flow-sensitive species	Proportion of native scrapers and filter feeders	Total are and spatial distribution of riparian habitat types (pools, riffles, backwaters, side channels, oxbows, sand bars, lotic/lentic nreaches, marshes
Native riparian species density, diversity age structure or density and age structure of native riparian compared to invasive or native upland species	Flow-dependent life stage metrics	Flow-dependent life stage metrics	
	Proportion of community that is native	Extent and biomass of invasive species	

Table 6: Changes in flow characteristics on the Bill Williams River due to the Alamo Dam (Shafroth et al 1998) and (Shafroth et al. 2002)

Flow Metric (m³ sec⁻¹)	1943-1968	1969-1996
Mean Peak Flow	416	44
Median Peak Flow	223	18
Maximum Peak Flow	1943	198
	Pre-dam	1968-1987
1968-1987 10-Year Peak	1397	148
	Pre-dam	Post-dam
May-September Cumulative Flows	1.5	2.5
	Pre-dam	Post-dam
Peak Instantaneous Flow	238.1	183.9

Table 7: Extent and structure of four riparian vegetation patch-types along the Bill Williams and Santa Maria Rivers (Shafroth et al. 2002)

Variable	Patch Type	Bill Williams River	Santa Maria River
Basal area (m ² /ha)	Populus/Salix	45.3 ± 7.1	24.1 ± 6.1
	Tamarix	23.9 ± 3.7	7.5 ± 2.3
Stem Density (1000 stems/ha)	Populus/Salix	21.3 ± 7.4	16.1 ± 3.9
	Tamarix	25.1 ± 3.9	13.7 ± 2.2
Extent (percentage of transect)	Populus/Salix	6.5 ± 2.5	1.6 ± .08
	Tamarix	30.6 ± 9.5	34.2 ± 8.3

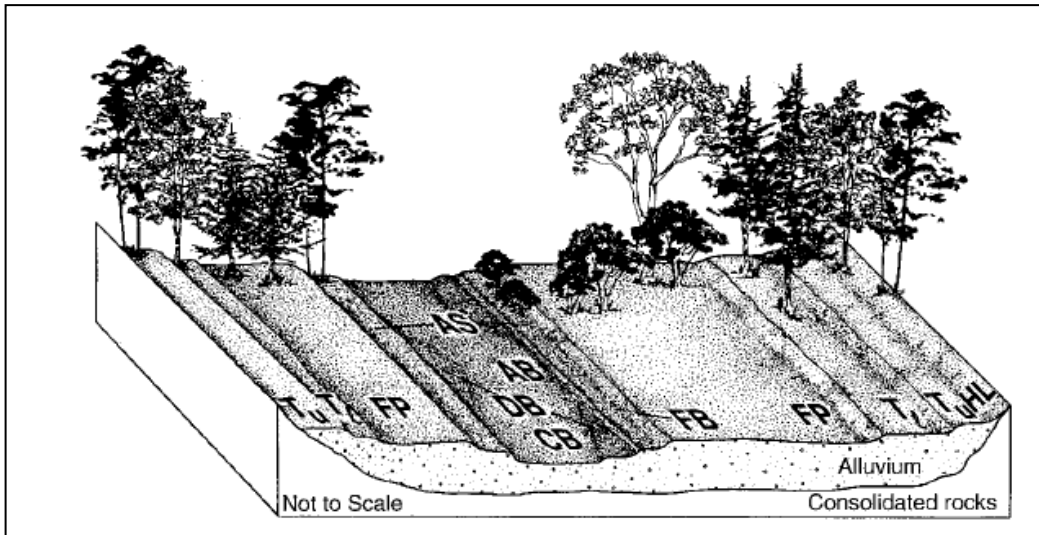


Figure 1: Diagram of fluvial landforms including channel bank (CB), depositional bar (DB), active channel bank (AB), active channel shelf (AS), floodplain bank (FB), floodplain (FP), low terrace (T1), upper terrace (T2), and hillslope (HL) (Bendix and Hupp 1999)

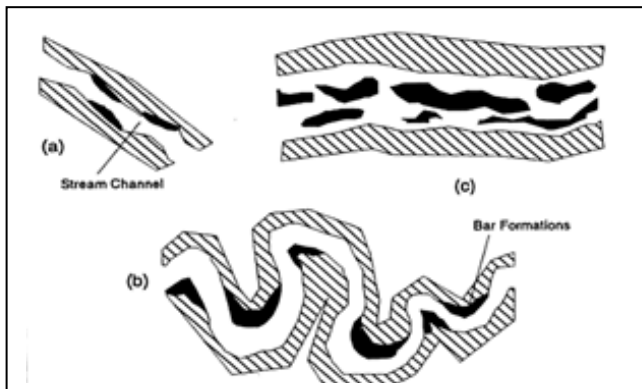


Figure 2: Example of channel patterns and bar-types; (a) straight with alternating bars, (b) meandering with point bars, (c) braided with mid-channel bars (Briggs 1996)

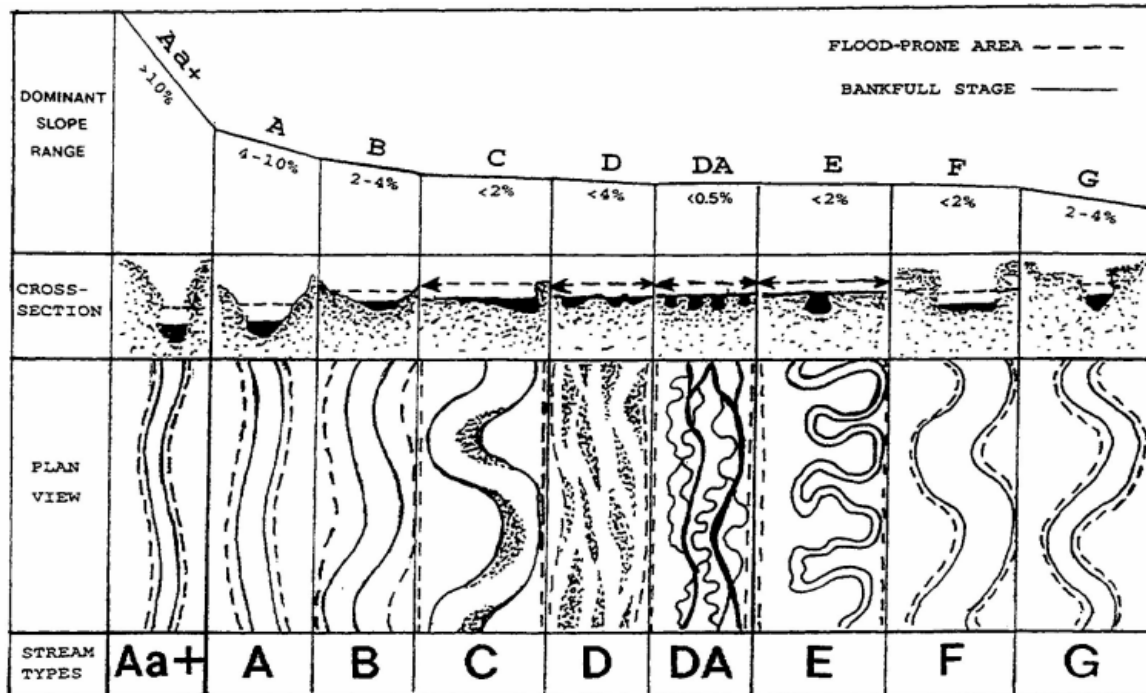


Figure 3: Rosgen (1994) stream classifications. The dashed line indicates the extent of the floodplain, which is influenced by the cross-sectional shape of the channel. Types E, B, and C are highly stable systems with wide floodplains, with F and G being incised channels with limited floodplain access. These stream-forms can occur simultaneously within a river, but are generally consistent at the stream-reach scale; specific stream-types are dependent on gradient, slope, land-use and the geomorphic features of the basin.

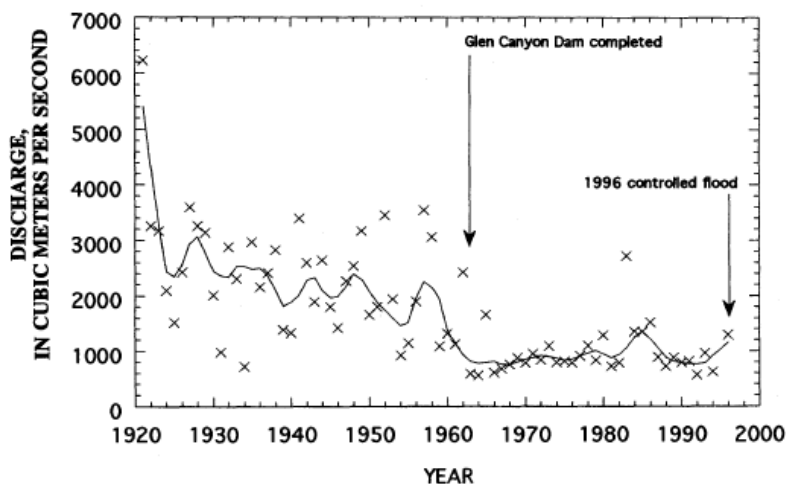


Figure 4: Pre and post dam hydrograph for the lower Colorado River at Glenn Canyon Dam (Schmidt et al. 1998)

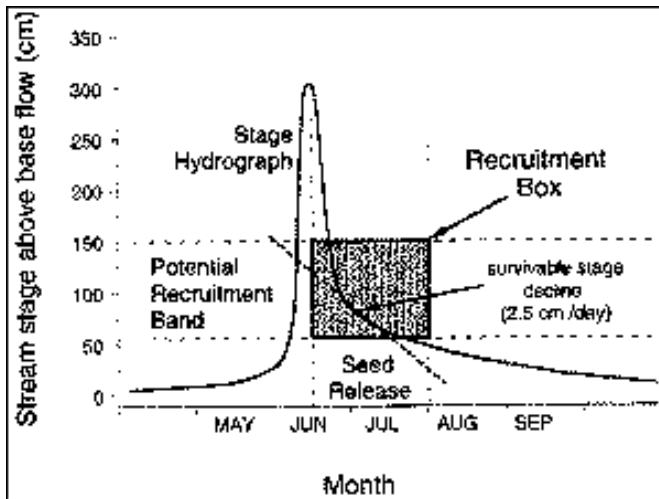


Figure 5: The *Recruitment Box Model*, representing the phenology and hydrograph that satisfies seedling establishment requirements (Mahoney and Rood 1998)

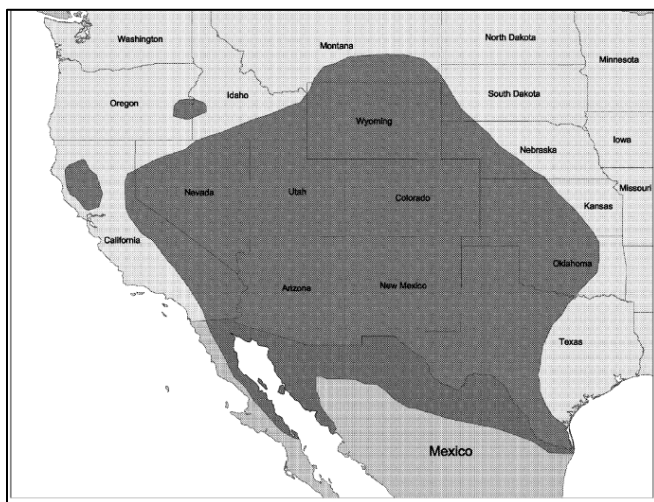


Figure 6: Current distribution of *Tamarix ramosissima* in the southwestern US and northern Mexico (Glenn and Nagler 2005)

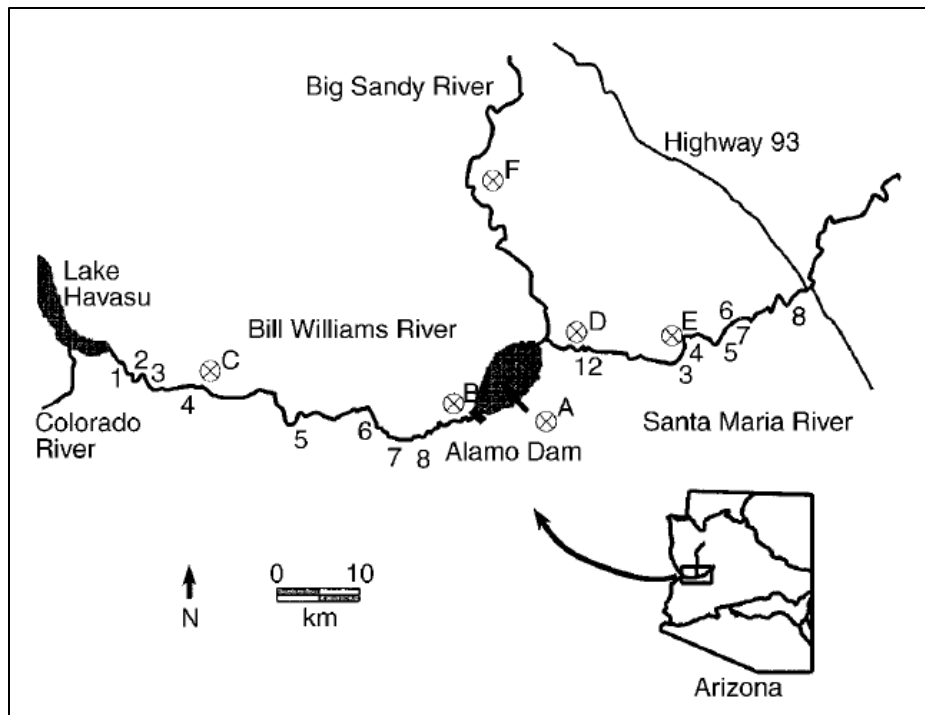


Figure 7: Bill Williams River Map (Shafroth et al. 2002)



Figure 8: Alamo Dam and Lake on the Bill Williams River (Bill Williams River Steering Committee 2014)

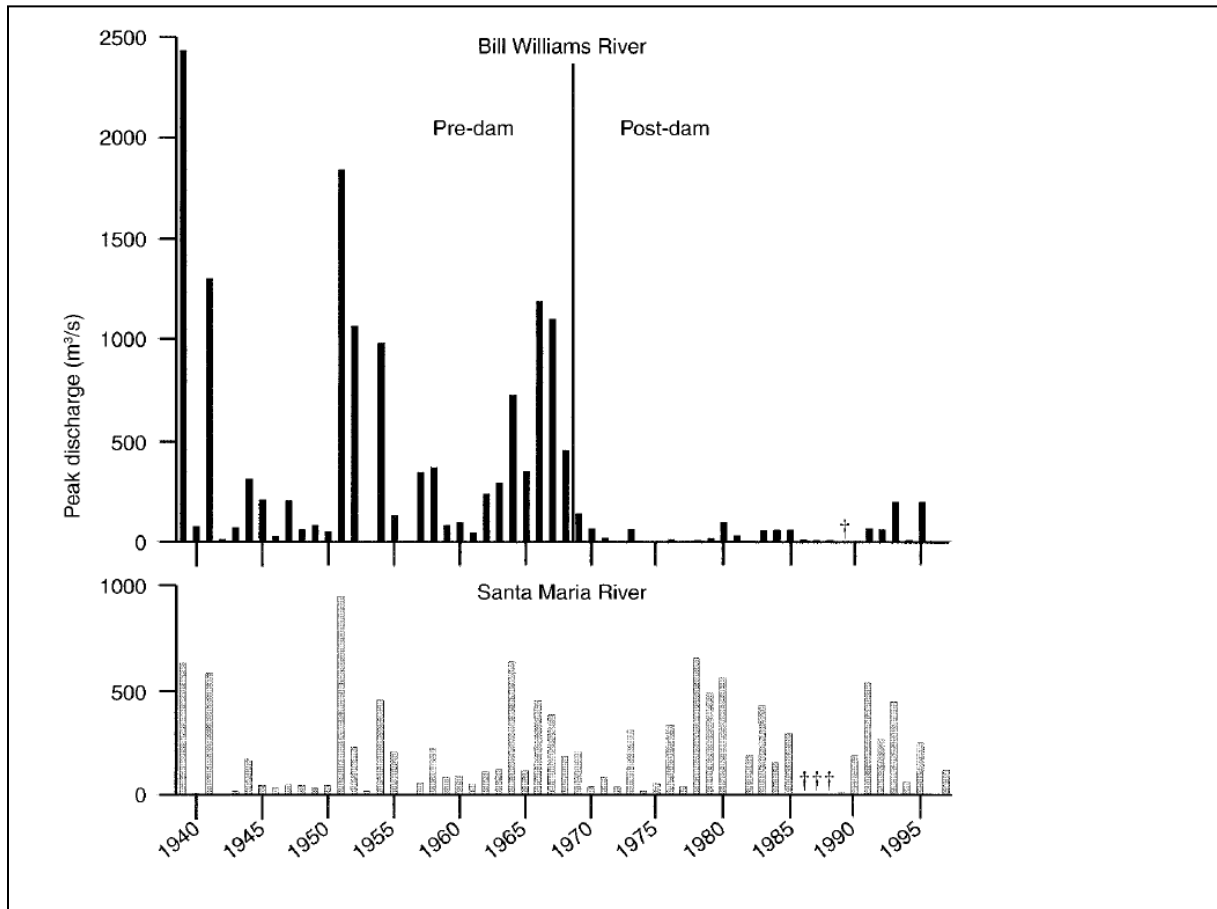


Figure 9: Annual flood series (1940-1995), Bill Williams and Santa Maria Rivers (Shafroth et al. 2002)

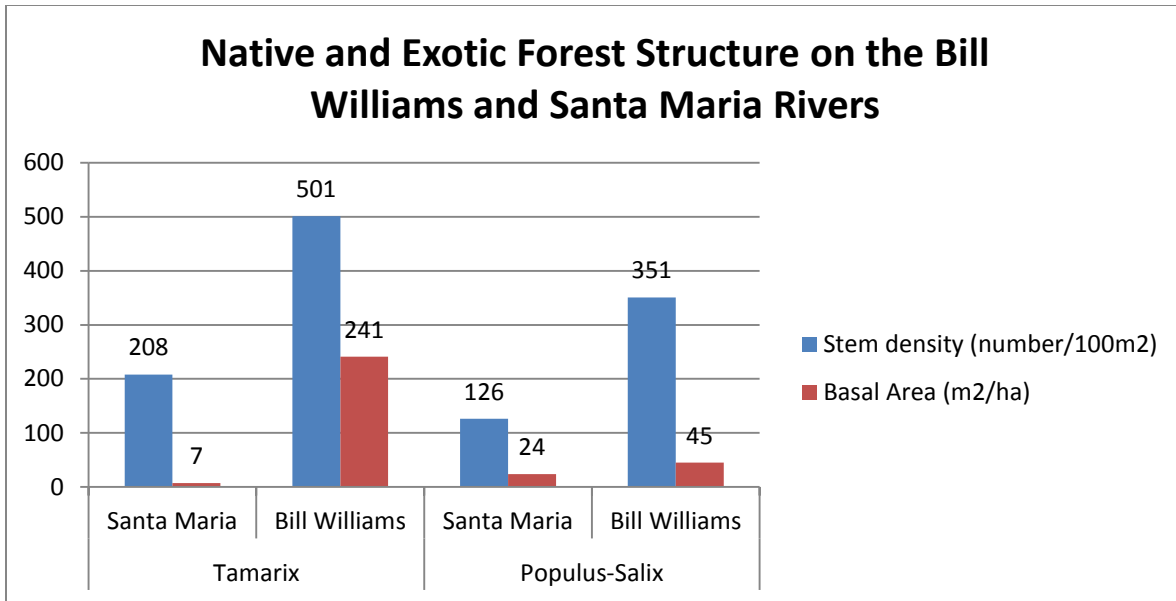


Figure 10: Comparisons of *Populus-Salix* and *Tamarix* basal area and stem density on the Bill Williams and Santa Maria Rivers (Shafroth et al. 2002)

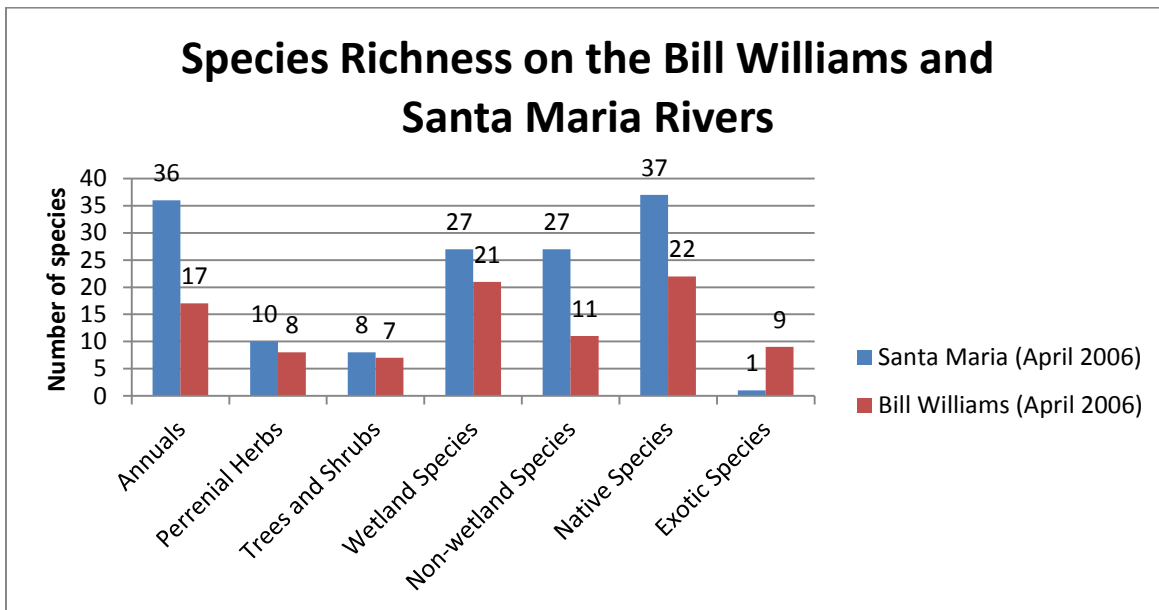


Figure 11: April 2006 plant species richness comparisons between the Bill Williams and Santa Maria Rivers (Shafroth et al 2012)

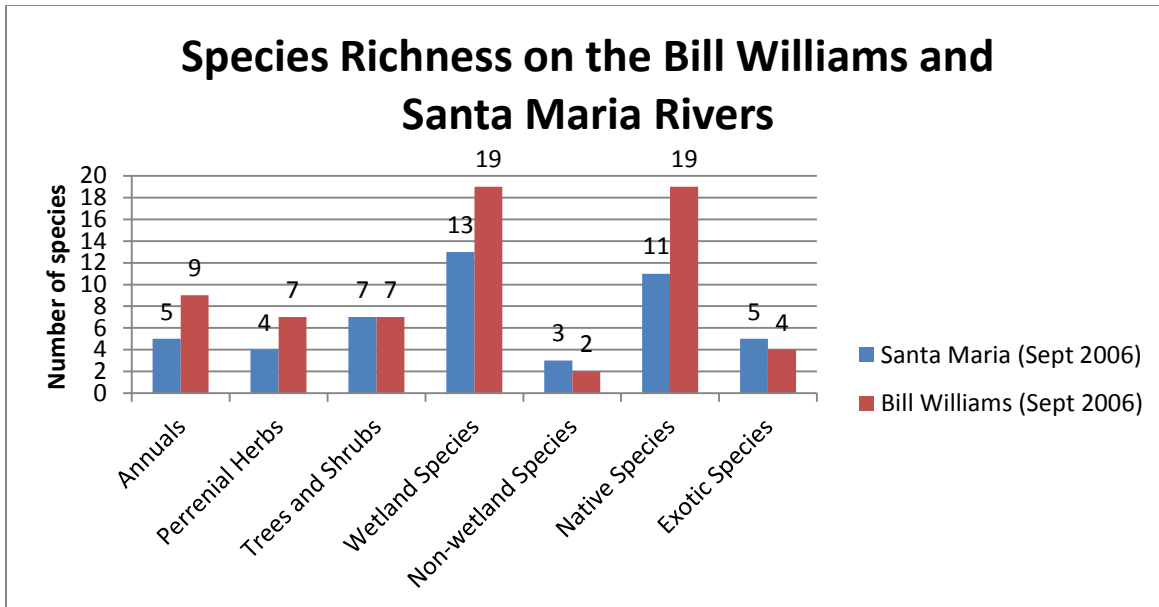


Figure 12: September 2006 plant species richness comparisons between the Bill Williams and Santa Maria Rivers (Shafroth et al 2012)

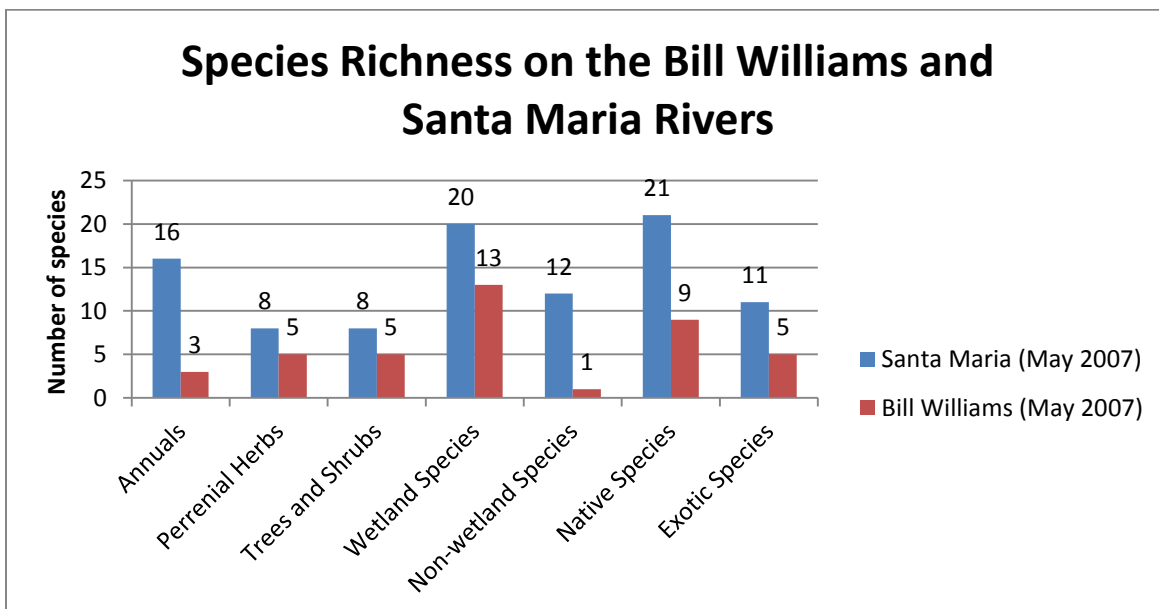


Figure 13: May 2007 plant species richness comparisons between the Bill Williams and Santa Maria Rivers (Shafroth et al 2012)

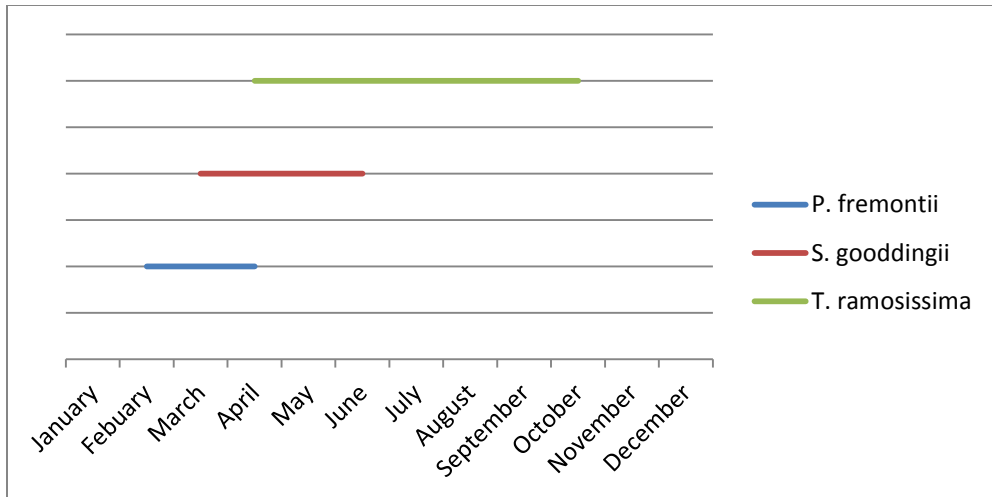


Figure 14: Seed dispersal periods of *Populus fremontii*, *Salix gooddingii*, and *Tamarix ramosissima* during 1997 on the Bill Williams River (Shafroth et al 1998)

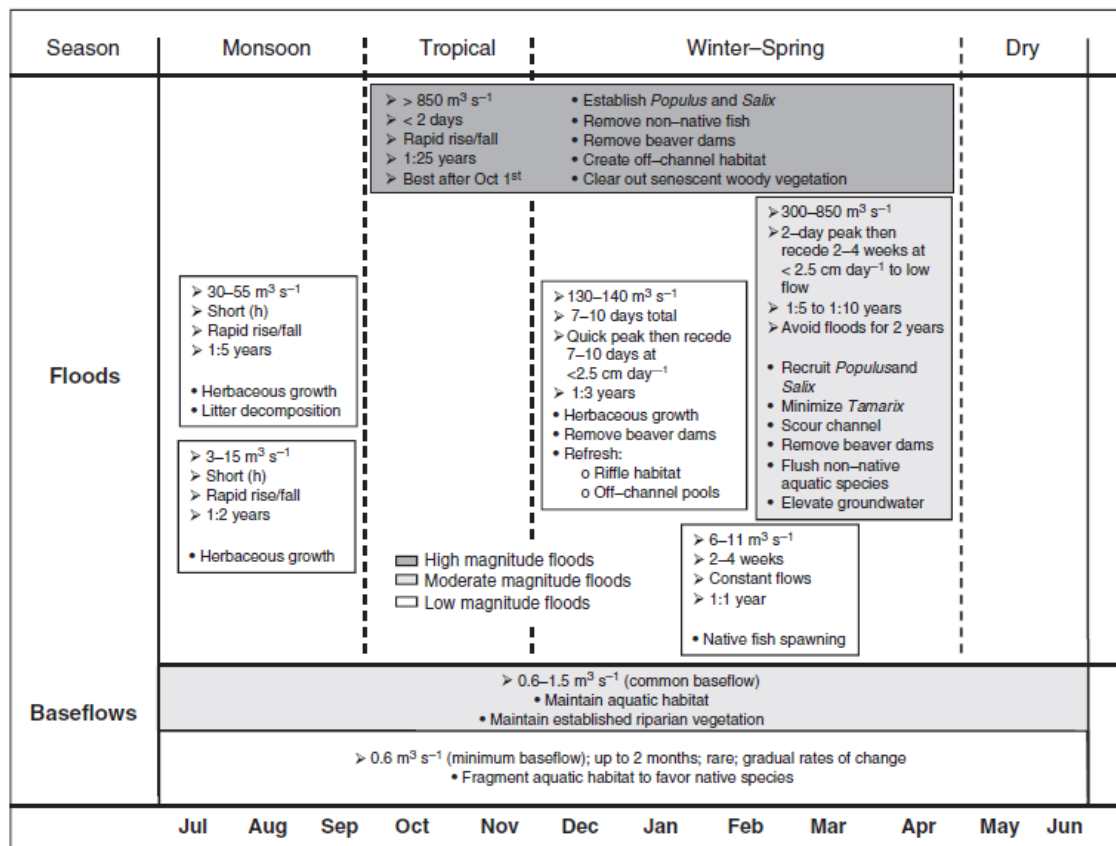


Figure 15: Flow requirements for the Bill Williams River, as developed by scientists and resource managers at a 2-day workshop. Arrows indicate flow characteristics and bullets indicate ecological functions associated with flows (Shafroth et al. 2010).

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