SRP13548

Journal of Arid Environments (2001) **49**: 17–34
doi:10.1006/jare.2001.0833, available online at http://www.idealibrary.com on IDEAL®

Restoration of riparian vegetation in the south-western United States: importance of flow regimes and fluvial dynamism

Juliet C. Stromberg

Department of Plant Biology, Arizona State University, Tempe AZ 85287-1601, U.S.A.

Riparian ecosystems in the south-western United States have undergone extensive physical and biological changes, due, in part, to alteration of natural flow regimes and suppression of fluvial processes. Many riparian ecosystem restoration projects are achieving success because they recognize the importance of restoring the hydrologic regime. In other words, these projects are restoring flows of water and sediment in sufficient quantities and with appropriate temporal and spatial patterns. Other projects have proceeded without recognition of the need to incorporate environmental stream flow requirements into management plans. To increase success rate of riparian ecosystem restoration, this paper describes some changes that have occurred within riparian ecosystems of the southwestern United States, reviews the role of stream flow regimes in structuring riparian plant communities, and assesses various ways in which riparian plant communities can be restored by naturalizing ecological processes.

© 2001 Academic Press

Keywords: restoration; riparian ecosystem; hydrology; stream flow regime; arid and semi-arid; *Populus*

Introduction

Restoration of riparian ecosystems has become a major enterprise in developed nations. Many projects have been implemented in the western United States that have a goal of reversing the riparian loss and degradation that streams have undergone over the past century (Goodwin *et al.*, 1997). Some of these restoration efforts have achieved greater success than others. To improve success rate, greater information sharing is needed on the ecology of riparian ecosystems, causes of degradation, and strengths and limitations of various restoration approaches. To this end, this paper describes some changes that have occurred within riparian ecosystems of the south-western United States, reviews the role of key environmental factors that structure riparian plant communities, and assesses various ways in which riparian vegetation can be restored by naturalizing ecological processes.

Riparian ecosystem losses and the need for restoration

The physical settings and dynamic processes of streams in the south-western United States have been changed dramatically over the past century. The Colorado River, the

0140-1963/01/090017 + 18 $35.00/0 © 2001 Academic Press

SRP13549

largest stream in the region, has been transformed from a free-flowing, dynamic waterway to a string of storage pools. Its stream flow has been greatly reduced and only rarely reaches the Colorado River delta. Multiple impoundments also disrupt patterns of water and sediment flow on major and minor tributaries of the Colorado River (Graf, 1999). Diversion structures, including Granite Reef Dam on the Salt River, and New Waddell Dam on Arizona's Agua Fria River, completely reroute stream flow into canals, leaving dry river beds below (Graf, 1982; Springer *et al.*, 1999). Other streams, such as the Santa Cruz in Arizona, have been dewatered as a result of ground-water pumping. Ground-water mining, stream flow diversion, and damming in combination have caused ground-water tables along a portion of the Gila River to plummet by 200 m in the last century (Judd *et al.*, 1971). Concentrations of dissolved solids (salinity levels) have increased in many streams, a result of high rates of evaporation from expansive, shallow impoundments, and run-off from agricultural fields. Vast areas of flood plains have been converted to agricultural fields and urban settlements, necessitating construction of dikes and levees to constrain flood waters.

These physical changes have led to biological losses and various types of biological change (Rood & Mahoney, 1990; Patten, 1998; Johnson, 1998). Along some reaches in which floods have been suppressed but stream flows not diverted, riparian biomass has increased. For example, riparian vegetation has increased significantly along the Bill Williams River, a small tributary to the Colorado, since closure of Alamo Dam in 1968. Most of the new vegetation, however, is composed of the exotic woody plant *Tamarix ramosissima* and the dense riparian forests have become fire-prone (Busch, 1995; Shafroth, 1999). Along the Colorado River, narrow strips of marshland and riparian forest have developed in areas of the narrow Glen Marble and Grand Canyon reaches that formerly were scoured by annual floods (Stevens *et al.*, 1994). Along much of the lower Colorado River and major tributaries, however, riparian vegetation has declined and the vegetation mosaic in the flood plain has simplified. Much of the lower Colorado River flood plain is now vegetated by *Tamarix ramosissima* and *Pluchea sericea* (Ohmart *et al.*, 1988). Both are reproductively opportunistic, stress-tolerant shrubs, that can withstand saline soils, drought, and repeated burning. Near the delta of the Colorado River, the cumulative effects of river use have nearly obliterated the most extensive wetlands in the south-western United States and northern Mexico, leaving only a few scattered ecological remnants amidst an expanse of dry, salty, unvegetated land (Glenn *et al.*, 1992, 1996; Briggs & Cornelius, 1998).

Generally, in arid regions, riparian species diversity is lower along streams that do not flow perennially (Tabacchi *et al.*, 1996). As streams are dewatered, species decline in number because the wetland to upland species continuum is truncated: obligate wetland species that depend upon shallow water tables or perennial stream flows decline, while upland species that grow on drier portions of the flood plain may be unaffected or expand into areas once occupied by riparian obligates (Stromberg *et al.*, 1996). Lowered ground water levels have killed phreatophytic riparian plants along the Carmel River in California, Coal Creek in Colorado, and the Santa Cruz and Gila Rivers in Arizona, among others (Bryan, 1928; Rea, 1983; Groeneveld & Griepentrog, 1985; Scott *et al.*, 1999). Fremont cottonwood (*Populus fremontii*)-Godding willow (*Salix gooddingii*) forest associations, dependent on ample amounts of fresh water and specific flooding patterns, have become globally imperiled (Fenner *et al.*, 1985; Busch & Smith, 1995). Riverine marshlands have become rare (Hendrickson & Minckley, 1984).

Declines in species diversity also can occur if flood disturbance becomes infrequent or if the temporal and spatial diversity of flood disturbance declines (Pollock *et al.*, 1998). Flood suppression may be a key reason why plant diversity is low in the dense thickets of *Tamarix* that develop on flow-regulated streams (Brock, 1994). Although these declines have been attributed to changes produced by *Tamarix*, the ultimate cause may be river damming and flood suppression. Plant biodiversity levels can be equally high in

SRP13550

understories of the patchy forests of *Populus* and *Tamarix* that occur along free-flowing streams (Stromberg, 1998*a*).

The combined effects of physical and vegetational changes in riparian habitats has endangered many riparian-dependent animal species, including south-western willow flycatcher (*Empidonax traillii extimus*), Yuma clapper rail (*Rallus longirostris yumanensis*), and least Bell's vireo (*Vireo bellii pusillus*). Large percentages of native fish species have become endangered or extinct (Minckley & Deacon, 1991; Richter *et al.*, 1997*a*). Restoration of habitat for endangered species has been one impetus for restoration projects in the south-western U.S.A. (Kus, 1998). Other restoration goals expressed by managers of Nature Conservancy preserves, National Wildlife Refuges, flood control districts, and other sites, include increasing the abundance of rare native riparian vegetation types, shifting dominance from exotic plant communities to native, improving wildlife habitat and recreational amenities, and reducing the size of flood peaks that have increased due to activities such as road construction and urbanization.

Approaches to riparian ecosystem recovery

Success rates for effecting desired ecosystem changes vary widely. Our ability to restore riparian ecosystems sometimes falls short of our desire, despite some assertions to the contrary (Hey & Phillipi, 1999). Sometimes, efforts fail because of an inability to implement or sustain necessary changes. In other cases, the failure lies with inadequate scientific recommendations. Too often, restoration efforts fail because the underlying factors that caused the riparian degradation are not addressed (Briggs, 1996; Briggs *et al.*, 1994; Hobbs & Norton, 1996; Alpert *et al.*, 1999). For example, a decade ago in the south-western United States, riparian restoration was synonymous with cottonwood pole planting. Many of these planting efforts failed, because no-one asked, 'What factors caused the loss of the cottonwoods?'. Today, removal of exotic plant species seemingly has become a restoration panacea, but again, few are asking 'What factors have allowed for establishment of the exotics?'. Additions or removals of plants can sometimes suffice as the sole restoration effort. More frequently, though, there is a need to 'dig deeper' to identify and reverse the root causes of ecosystem degradation (Wissmar & Beschta, 1998). Because we do not yet have all the answers, restoration projects should be designed as science-based experiments or at least integrate some degree of experimentation. Adaptive research and management policies should be integral to the process.

Primary root causes of riparian loss and degradation are alteration of herbivory regimes, disruption of hydrologic regimes, and direct conversion to irrigated cropland and urban areas. Restoration measures to reverse impacts of livestock grazing generally require exclusion of the grazing animals or large declines in their stocking rates in riparian zones as well as upland sites throughout the watershed (Williams *et al.*, 1997). Active intervention may be required to reverse long-term effects of overgrazing, such as compacted soils and lowered water-tables. With respect to direct disruptions of hydrologic regimes, there is growing recognition that restoring natural flow regimes, including natural patterns of flood disturbance and fluvial dynamism, is critical to restoration success (Briggs, 1996; Middleton, 1999; Graf *et al.*, 2001). Projects with the greatest chance for sustainable improvement encompass an 'ecosystem management agenda' of restoring ecological processes, as a basis for restoring ecosystem complexity (Whisenant, 1999). Active restoration techniques such as planting trees or shaping land forms sometimes are used to jumpstart the recovery but the ultimate goal is to allow process restoration to naturally drive ecosystem recovery (Kauffman *et al.*, 1997).

One such ecosystem management effort is underway in California. Los Angeles Department of Water and Power, in conjunction with Mono County, is involved in a multi-year effort to restore the Owens River gorge. Restoration of flood pulses and

SRP13551

base flows has allowed for re-establishment of riparian vegetation as well as redevelopment of aquatic micro-habitat features such as pools and runs (Hill & Platts, 1998). In Utah, the U.S. Bureau of Reclamation, through the Utah Reclamation Mitigation and Conservation Commission (URMCC), is restoring a portion of the Provo River, as mitigation for riparian degradation resulting from Federal reclamation projects. The Provo River Restoration Project seeks to modify the geomorphology and hydrology of the river to create a more naturally functioning riparian system. The project operates according to an 'ecosystem restoration standard' that integrates multiple aspects of the environment (URMCC, 1997). Unfortunately, full restoration of water and sediment flow is prevented by a new, upstream dam.

Another case in point is the Truckee River in Nevada. The U.S. Fish and Wildlife Service began managing Stampede Reservoir along the Truckee River for occasional spring flood release, to stimulate spawning of endangered fish species including the cui-ui (*Chasmistes cujus*). An ancillary benefit was establishment of *Populus fremontii* seedlings, which also depend upon appropriately timed floods. Gourley (1998) states, 'A whole array of ecosystem components may begin to recover when restoring a basic ecosystem process', such as the natural flow regime of a river. The chances of meeting the needs of many native species and providing sustainable ecosystem improvement are increased if we take an ecosystem approach that restores hydrologic and geomorphic processes, including flood disturbance (Bayley, 1991; Stanford *et al.*, 1996). However, the Truckee River ecosystem and many others remain under threat due to continued demands on stream water, barriers imposed by dams, and lack of restoration of the full complexity of the flood regime.

Approaches that are less self-sustaining and more contingent on human intervention continue to be undertaken. For example, a $90 million effort is underway to restore Sonoran riparian vegetation to a dewatered section of Phoenix's Salt River. The Rio Salado project is a joint effort by the City of Phoenix, U.S. Army Corps of Engineers, and other groups, and is funded largely by Federal monies. Goals include creation of wildlife habitat and passive recreational opportunities: indirect benefits including economic revitalization of neighborhoods along the river corridor also are anticipated. While the outcome of this effort remains to be seen, the project is conceptually flawed as a true ecosystem restoration effort because it involves artificial approaches that do not address the root causes of ecosystem degradation. There are no plans to release water or sediment from upstream dams or to recharge the alluvial aquifer. Rather, the plan calls for releasing pumped ground water into excavated surface channels and planting 75,000 contract-grown trees. The rehabilitation efforts probably will result in creation of an urban park with some wildlife habitat value. It remains to be seen how biodiversity, bioproductivity, and ecosystem resilience will compare to that of free-flowing streams, in the absence of a restored flood regime and ground water source, and without connectivity to high-quality riparian ecosystems.

Natural stream flow regimes

As a broad restoration guide, restoration of natural stream flow regimes is fundamental (Poff *et al.*, 1997). The natural flow regime hypothesis states that riverine species require the temporal and spatial pattern of stream flow to which they have adapted over time. Thus, to maintain native riparian ecosystems, one must retain regional flow regimes. However, the specifics of implementation are complicated by many factors. For one, the context within which natural flows exert influence has changed. Because so many factors that influence riparian structure and function have been altered, full restoration of natural flows may not be the preferred option in all circumstances. For example, very large scouring floods may not be desirable if a stream reach has become detached from potential post-flood recolonization sources, as might occur if the habitat

SRP13552

has become fragmented. Analogously, prairie restorationists are finding that it is not always appropriate to restore natural fire regimes to tiny patches of prairie, because the net effect can be a reduction in species diversity (Collins *et al.*, 1998).

Another complication arises because of the high degree of temporal hydrologic variability in arid regions. Which time-period defines historic norms? How much can we deviate from these norms before we alter biotic components in unacceptable ways (Richter *et al.*, 1997*b*)? Will ecosystems be sufficiently restored if we allow for the 25-year return flood but not the 100-year return flood? What if we increase flood frequency to twice per year rather than three times per year? Which flow regime components are most ecologically sensitive or influential? A small deviation in ground water depths or flood timing may cause greater change than, for example, a small deviation in flood magnitude. What are the consequences of restoring flood waters but not associated sediments? Such questions demand research. In restoration contexts, these questions could be tested using an adaptive management approach until uncertainty levels decline.

There is a compelling reason to study the relationships between stream flows and riparian ecosystem structure and function: competing pressures on water resources. While riparian restoration efforts are on the increase, so too are water development and extraction pressures on streams and aquifers. We are in the midst of a water crisis, regionally and globally (Duda & El-Ashry, 2000). Although per capita municipal water consumption has decreased in the United States, overall human population size is up and so is total water use. Desert cities, including Las Vegas and Phoenix, are among the most rapidly growing in the United States, putting increasing pressure on limited water supplies. People even argue over the rights to water run-off from irrigated lawns (Oad & DiSpigno, 1997). Irrigated agriculture continues to use the largest share of the water resources in the south-western United States, with most going for production of livestock feed. Electrical power grids are at their limit, a consequence in part of years of rejecting investment in solar and other renewable power sources; thus, pressures are high to manage streams for hydro-power production. Where there is competition for a limited resource, such as water, one must be able to justify the ecological importance of virtually every drop.

Research into environmental water needs intensified in the 1970s and 1980s, triggered by fisheries ecologists who wished to quantify in-stream flow needs for declining populations of fish (Maddock, 1999). Ecologists soon realized that in addition to in-channel flow needs, there was a need for over-bank flows and sub-surface (groundwater) flows to maintain the riparian vegetation that helps to shape aquatic habitat features (Hill *et al.*, 1991). Studies to determine the water needs of riparian vegetation have since branched in many directions.

One branch includes transpiration studies, which seek to determine how much water a plant population or assemblage needs by asking how much it uses. Techniques ranging from canopy-scale micrometeorological measurements of heat flux to plant-scale measurements of sap flow and heat-pulse velocity, have revealed a wide range of responses to the question of 'how much water does a patch of riparian vegetation require?'. Transpiration rates vary with stand composition, biomass structure, and plant age. Along Arizona's San Pedro River, transpiration rates among *Populus fremontii* patches ranged from 3 to 6 mm per day (Schaeffer *et al.*, 2000). Young, densely-foliated stands transpired more water than did older stands, although individually, old *P. fremontii* trees transpired more water than young ones. Transpiration rates also vary with water availability (Devitt *et al.*, 1997). *Populus* trees along the San Pedro River transpired twice as much water when growing at sites with shallow *vs.* deep ground water (Goodrich *et al.*, 2000). Flood-inundated *Tamarix ramosissima* along New Mexico's Rio Grande transpired twice as much water as did those that were not flooded (Cleverly *et al.*, 2000). Bioproductivity and transpiration rates of other riparian plant species, including *Prosopis velutina* and *Sporobolus wrightii,* vary depending on the quantity of

SRP13553

summer rains (Scott *et al.*, 2000*a*). With sufficient data on transpiration rates by plant assemblage and with vegetation maps, one can estimate total water use rates for long river reaches (O'Keefe & Davies, 1991; Goodrich *et al.*, 2000). With a knowledge of the water requirements for high-quality states of each assemblage, one can estimate the amount of water necessary to restore or maintain ecosystem integrity.

In addition to knowing quantities of water needed, it is important to know how water should be distributed in time and space. A plant's requirement for a particular spatial and temporal distribution of water can be determined by examining how population demographic processes such as recruitment, growth, and mortality vary in relation to hydrologic factors such as flood timing or ground water depth. One can then define thresholds and quantify relationships between plant processes and hydrologic conditions. Spatial and temporal components of water needs also can be determined by analysing water samples for stable isotopes of oxygen and hydrogen. This can help to identify the source and depth of water used by the plant, be it the flood plain aquifer, regional aquifer, stream channel, rain water, or some combination thereof.

Water requirements can not be individually quantified for all species in riparian ecosystems (Sparks, 1995). Riparian ecosystems are 'hot-spots' of biodiversity and hundreds of different plant species can occur along perennial and intermittent streams (Naiman *et al.*, 1993; Brown, 1994; Wolden *et al.*, 1994; Tabacchi *et al.*, 1996; Dixon & Johnson, 1999). However, it may be possible to classify species into guilds, select indicator or focal species within each guild, and thereby quantify the water needs for a large number of species. Each focal species could define 'different spatial and compositional attributes that must be present in a landscape and their appropriate management regimes' (Lambeck, 1997).

Salix gooddingii and *Populus fremontii* are representative of a guild of Sonoran riparian species that depend on shallow ground water and periodic winter/spring flooding. Dense productive stands of these trees occur where ground water averages less than about three meters deep (Busch *et al.*, 1992; Stromberg *et al.*, 1996; Smith *et al.*, 1998). Of the two species, *P. fremontii* seems to be able to utilize water from unsaturated soil layers to a greater degree then *S. gooddingii*, a more strict phreatophyte (Snyder & Williams, 2000). Both are quite sensitive to drought (Rood *et al.*, 2000; Leffler *et al.*, 2000). Annual growth rate of *Populus* and *Salix* species declines in years with low stream flows and deep ground-water tables, with seasonal or annual declines of 1 m having been observed to kill adults and juveniles (Stromberg & Patten, 1995; Willms *et al.*, 1998; Scott *et al.*, 1999; Scott *et al.*, 2000*b*; Shafroth *et al.*, 2000). Other woody pioneer plants of low-elevation south-western streams that appear to depend on similarly shallow ground-water include *Baccharis salicifolia*, *Salix exigua*, and *Salix bonplandiana* (Gary, 1963).

Continued establishment of *Populus fremontii* and *Salix gooddingii* depends on periodic occurrence of years with appropriately timed flood flows, high growing-season stream flows, and very shallow water tables (Everitt, 1995). Along free-flowing streams in the Sonoran Desert, regeneration floods occur about once every 5–10 years (Stromberg, 1998*b*; Stromberg *et al.*, 1991). Recruitment patterns are irregular, varying to a large degree with frequencies of El Niño years (Webb & Betancourt, 1992; Swetnam & Betancourt, 1998). In recent decades, there has been a high frequency of years with abundant winter rains and floods and consequently many new cohorts of *Populus fremontii* and *Salix gooddingii* have established.

After a flood pulse, survivorship of *Populus* and *Salix* seedlings depends on their ability to maintain root contact with the declining water table. Yearlings are able to tap water at a depth of about a metre or so by summers' end (Stromberg *et al.*, 1991; Mahoney & Rood, 1998). Along the Green River in Colorado, however, some *P. deltoides* subsp. *wislizenii* did not become phreatophytic until they were several years old (Cooper *et al.*, 1999). Between-site differences in ground water requirements may arise due to differences in climate (rainfall, temperature, evaporative stresses),

SRP13554

soil types and stratigraphy (soil water holding capacity, clay lenses and perched water layers) and other hydrologic modifiers (inundation frequencies).

Mahoney & Rood (1998) have developed a model, coined the recruitment box, that bounds the requirements for *Populus* seedling establishment with respect to post-flood recession rate, river stage, and flood timing. Recruitment also depends on high-energy floods to perform the geomorphic 'work' necessary to create recruitment sites and reinitiate community succession. Small winter or spring floods result in only a small amount of sediment movement, erosion, and deposition, and produce only small recruitment bands (Stromberg *et al.*, 1993*a*). Large, long winter floods, such as occurred during the 1992–1993 El Niño rainy season, form new channels and widen existing channels (Huckleberry, 1994). Extensive stands of *Populus* and *Salix* can establish on the expanses of bare mineral soil (Stromberg, 1997).

The period of water recession following the winter flood peaks can extend well into summer. This creates recruitment opportunities for species with a range of temporal regeneration niches, including spring-seeding *Populus* and *Salix* and summer-seeding *Baccharis salicifolia*, *Tessaria sericea*, and *Tamarix* spp. Late-summer monsoon floods also directly create recruitment opportunities. Sonoran panic-grass (*Panicum sonorum*), for example, depends on summer floods to complete its life-cycle. Once common along the lower Colorado River, the plant has been locally extirpated, due partly to suppression of summer floods (Nabhan, 1985). Many other plants also belong in this guild of summer-flood dependent pioneer species.

Prosopis velutina, *Prosopis glandulosa*, *Prosopis pubsecens*, *Sporobolus wrightii* and *Sporobolus airoides* are among a guild of seral, warm-season species that establish in response to summer rains or summer floods. They are deep-rooted, facultative phreatophytes that can utilize deep ground-water and shallow soil moisture. In wetter parts of their range, they can survive solely on precipitation (Scott *et al.*, 2000*a*; Tiller, unpublished data). Biomass structure varies greatly for *P. velutina* depending on ground-water availability. Canopy heights of 4 m are typical for *P. velutina* trees in Sonoran desert uplands. Larger size is attained for trees growing along ephemeral stream courses that receive periodic flooding. Trees reach greatest heights (up to 12 m) when growing along rivers where ground-water is at a depth of less than about 10 m (Stromberg *et al.*, 1992; Stromberg *et al.*, 1993*b*). Small, low-energy floods that deposit sediment around pioneer plants serve to create aggraded fluvial surfaces upon which these seral species establish.

Natural flood regimes help to maintain high biodiversity not only by creating a diversity of temporal regeneration niches but also by creating micro-habitats that vary spatially in depth to the water table, light availability, and soil properties. For example, flood deposits vary in depth, texture, and nutrient content, and support different assemblages of plants (Marks, 1950). Deep deposits of sand and gravel, too dry for marsh plants or for *Populus-Salix* seedlings, favor a guild of drought-tolerant shrubs including *Atriplex linearis*, *Bebbia juncea*, *Chrysothamnus nauseosus*, *Hymenoclea monogyra*, *Petalonyx thurberi* and herbs including *Cleome lutea*, *Dicoria cansescens*, *Eriogonum spp.*, *Euphorbia hyssopifolia*, *Heterotheca psammophila*, *Polanisia dodecandra*, *Sporobolus contractus*, *Sporobolus cryptandrus*, and *Tiquilia plicata* (Rea, 1983; Wolden *et al.*, 1994; Stromberg *et al.*, 1996, 1997). Nutrient-rich soils with high content of silt and clay favour *Lycium andersonii*, *Lycium fremontii*, *Viguiera dentata*, *Panicum obtusum*, *Plueraphis mutica*, *Sporobolus wrightii*, *Sporobolus airoides*, *Ziziphus obtusifolia* and many other species. Salty areas in flood plains support a guild of halophytes including *Allenrolfea occidentalis*, *Atriplex lentiformis*, *Distichlis spicata*, and *Suaeda torreyana*.

Floods also influence biodiversity patterns by causing river channels to relocate and meander, creating abandoned channels and backwater depressions, and inducing channel widening and subsequent re-narrowing. Areas with standing water or near-surface water tables, such as might develop after floods erode terraces or form off-channel depressions, support marshlands vegetated by species including *Juncus articulatus*,

SRP13555

Juncus torreyi, *Typha domingensis*, and *Scirpus americanus* (Grace, 1989; Stromberg *et al.*, 1996, 1997). Beavers, keystone species in riparian ecosystems, also engineer the marshy conditions favoured by these obligate and facultative wetland plants.

Summer and winter floods maintain high productivity rates of riparian forests by providing water for ground-water recharge, wetting flood plain soils, depositing nutrients, and flushing salts. Growth rate of the phreatophytic tree *Platanus wrightii*, for example, increases not only with size of the winter floods (a primary source of ground-water recharge), but also with the frequency of small summer floods, which increase nutrient availability (Grimm & Fisher, 1986; Stromberg, 2001). Several plant species, including *Populus fremontii* and *Salix gooddingii*, do not tolerate high concentrations of salts (Shafroth *et al.*, 1995; Glenn *et al.*, 1998) and decline in germination rate and growth-rate if salt concentrations are not flushed by flood flows. Large floods also function to fire-proof riparian ecosystems (Ellis, 2001). Without floods, plant debris and litter accumulates, plant water content decreases, and fires become larger and more frequent (Busch, 1995). Fires favour species that are clonal or readily resprout from the root crown, such as *Tessaria sericea*, *Chloracantha spinosa* and *Tamarix ramosissima*, *Populus fremontii*, in contrast, is readily killed by summer burns.

Restoration constraints and compromises

Ultimately, full restoration of riparian ecosystems hinges on removing impediments to the natural flows of water and sediments (Schmidt *et al.*, 1998). There are cases in which flow regimes have been fully restored, in response to changing societal goals. In central Arizona, for example, a decision was made to decommission the hydro-power dam on Fossil Creek and restore full flows to the stream. Benefits from restoring downstream aquatic and riparian habitat were believed to outweigh the small loss of hydro-power production and loss of habitat developed above the dam. There are other cases where full naturalization of fluvial processes is not desired by all stakeholders. In such cases, how do we make compromises between water needs of the riparian and aquatic ecosystems and direct human water demands? Can we maintain or restore ecosystem integrity while accommodating some degree of water extraction, hydro-power production, flood control, and/or flood plain agriculture (Schmidt *et al.*, 1998)? Generally, management emphasis on the production of commodities requires that one accept ecological costs of reduced site potential and functional abilities of the riparian ecosystem. However, there are many changes that can be made to restore a greater degree of riparian ecosystem structure and function. Some changes are described below, organized by ecological stress factors.

Loss of stream flows or declines in ground water

There are several sustainable solutions for restoring stream flows and raising water tables to levels that allow for recovery of hydrophytic and mesophytic vegetation types such as *Populus-Salix* forests or riverine marshlands, while also allowing for water extraction for human consumption. Water can be stored in aquifers rather than reservoirs, municipal water can be recycled and released into stream channels, stream channels rather than canals can be used for water delivery, efficiency of municipal, agricultural, and industrial water-use can be increased, and extraction demands can be reduced. Ultimately, integrated, watershed-based approaches to water management are needed to reverse adverse effects of ground water mining and surface water diversions. All water users, municipal, agricultural, or industrial, need to work together and address water overdraft problems.

In the arid south-west, where open water evaporation rates are greater than 2·7 m year^{-1}, it seems more ecologically advantageous to store water in aquifers than in

SRP13556

surface impoundments. A case in point involves the Agua Fria River in central Arizona. Lake Pleasant Reservoir, behind New Waddell Dam, stores Colorado River water that is delivered through Central Arizona Project canals and pipelines. The reservoir loses more water to evaporation per year than arrives from the in-flows of the Agua Fria River itself. The reach downstream of the dam is completely dewatered. A modeling study showed that a several mile stream reach below the dam could be used as a conduit for delivery of water to a ground water recharge and recovery site (Springer *et al.*, 1999). If Central Arizona Project water was released from the dam, the shallow bedrock layer would allow water in the aquifer to rise to levels that would sustain riverine marshland, *Populus-Salix* forest and *Prosopis* woodland. No more water would be released to the atmosphere through evapotranspiration than if the water were stored in the reservoir, and there would be substantial increases in riparian habitat.

Many cities, such as Nogales, Arizona and Phoenix, Arizona are recycling water by releasing treated municipal effluent into stream channels. With increased planning efforts, more riparian corridors could benefit from such a process. For example, a recent decision in Pima County, Arizona allows the county to buy reclaimed water (municipal effluent) for riparian restoration projects. Projects that secure endorsement by the U.S. Fish and Wildlife Service will be eligible for a portion of a 5000 acre-foot pool for each of the first 5 years of conservation efforts. A key concern is where to utilize the water to maximize its habitat value. Regional planning efforts are underway to identify sites that would maximize the environmental benefits of reclaimed water. Hydrogeologic studies can identify sites where shallow water tables exist or are likely to develop, and thus sites able to support phreatophytic riparian vegetation. Ecological studies can identify sites that are connected to high quality patches of riparian vegetation and thus more likely to have value as wildlife habitat.

Agricultural return flows constitute yet another source of water for riparian restoration efforts. Return flows are being considered as a water source to maintain cottonwood-willow habitat in the Limnitrophe area of the Lower Colorado River, to allow for survivorship of plants that established after the 1992–1993 winter floods (LCRBR, 2000). Elsewhere in the lower Colorado River flood plain, agricultural return flows have been used to increase survivorship of riparian trees and shrubs planted as part of revegetation efforts (Briggs & Cornelius, 1998). When using such flows to maintain or restore riparian habitat, it seems prudent to restrict or minimize use of biocides and fertilizers on farm fields, periodically flush soils to reduce concentrations of salts, and provide a year-round water source. Multiple drainage ditches could be created to simulate the multi-channeled pattern of desert streams and provide the dense strips of riparian forests intermingled with forest edges required by some bird species.

Daily fluctuations in water levels

A reach of the Salt River below Phoenix Arizona has been revitalized by the daily release of over 100 million gallons of treated municipal water into the channel (Rea, 1983). Water level in the channel, however, varies over the course of a day with the water use patterns of urbanites. To minimize diel fluctuations, restoration efforts are underway through the Tres Rios project. Water will be released into side basins and wetland treatment cells and then into the stream channel. Research and adaptive management are needed to determine whether there also is a need to restore seasonal fluctuations in water level.

Stream flows also can vary below hydro-power dams. Below Glen Canyon Dam on the Colorado River, the river stage once fluctuated up to 5 or 7 m day^{-1}, in response to diel cycles of hydro-power demands. To minimize adverse impacts, fluctuations presently are limited to about 1 m day^{-1}, which constrains the ability of Glen Canyon Dam

SRP13557

to fully respond to daily power demands. In the long-term, efforts are needed to shift to more sustainable energy sources such as solar power and thereby eliminate the need for peaking power from hydro-power dams.

Physical barriers

Dikes, levees, and other structures separate channels from flood plains on many streams. To remedy this situation on California's Cosumnes River, the Nature Conservancy has implemented a large-scale restoration project that entails breaching levees and allowing flood waters free access to abandoned farm land (Reiner, 2000). Potential for success was demonstrated by a natural breaching of the levee about 10 years ago, which was followed by development of a diverse riparian forest. Restoration efforts also have been made in urban areas to reconnect rivers and flood plains (Riley, 1998; de Waal *et al.*, 1998). There are opportunities to remove physical barriers and restore some degree of channel-flood plain connectivity along many desert streams, including the Colorado and Rio Grande. By allowing periodic flows onto flood plains, one provides the water, nutrients, sediments, and plant propagules to restore and sustain productive, diverse riparian forests (Molles *et al.*, 1998). Such efforts may require purchase of agricultural land for conservation purposes, or restrictions on home development in flood plains. Benefits accrue, however, in the form of downstream flood reduction (as the flood waters spread laterally across the flood plain), aquifer recharge, and biodiversity increases.

High concentrations of dissolved solids

Primary causes of the increased salinity of Colorado River water, as for many other streams, are use of flood plain lands for irrigated agriculture and storage of water in shallow reservoirs (Briggs & Cornelius, 1998). Water chemistry can be improved to levels that support a wide range of riparian plant species, including fresh-water dependent *Populus* and *Salix*, by reducing amounts of irrigated flood plain lands, increasing efficiency of irrigation systems, growing low-water-use crops, and/or reducing use of agricultural fertilizers. Reducing reliance on reservoirs for water storage will result in less evaporation of water and thus lower salinity levels. Released of flood pulses from dams can flush salts that have accumulated in flood plain soils.

Flood flow alteration

On many streams in western United States and Canada, recruitment rates of *Populus* and *Salix* species have declined because floods have been suppressed or altered in ways that do not meet their regeneration requirements. To remedy this situation in Alberta, Canada, scientists and Environmental Advisory Committees have provided information that has led to changed operation of some dams (Rood *et al.*, 1995, 1998; Mahoney & Rood, 1998). St. Mary and Oldman Rivers are managed for delivery of summer irrigation water and still flood fairly regularly in wet years. Rates of river meandering and channel realignment are relatively unaltered, and so are the processes that create 'nursery bars' for establishment of *Populus* seedlings. Changes have been made, however, such that flood waters now recede slowly enough to allow for high survival of the seedlings. Ecological models call for the stream stage to drop less than four cm per day, allowing seedling roots to maintain contact with moist soil. Another part of the agreement calls for an increase in summer base flow levels, thereby reducing risk of tree death from drought.

SRP13558

Such operating agreements could be put in place for the many flow-regulated streams in the south-western United States, including the Verde, Gila, and Colorado Rivers. Large flows are released from water-storage dams on these streams during occasional wet years, but the water is not released so as to optimize its environmental benefits. Sometimes, releases fortuitously meet the regeneration needs of riparian plants. El Niño weather patterns in 1978, 1983 and 1993 variously allowed for establishment of *Populus* and *Salix* trees along portions of the lower Gila and Colorado; reservoirs were filled to levels that necessitated large releases during winter and spring (Rea, 1983; Briggs & Cornelius, 1998; Zamora-Arroyo *et al.*, 2001). With operating agreements in place, dam managers could be prepared to release flows during wet years in ways that mimic natural hydrographs and favor establishment of native species adapted to natural flow patterns. To keep the trees alive, maintenance flows would have to be secured.

Often, there are constraints on the degree to which natural flood regimes can be restored. The Bill Williams River in western Arizona is regulated by Alamo Dam, which was built to minimize flood pulses into the Colorado River. Over the past 25 years, size and frequency of winter and summer flood peaks in the Bill Williams River have decreased. The U.S. Fish and Wildlife Service, Army Corps of Engineers, and university scientists have worked together to develop a flow-release plan. The plan calls for release of high base flows and periodic flood flows to improve the quality of riparian habitat in the below-dam wildlife refuge, while also maintaining flood control and recreational and wildlife benefits in Alamo Lake. However, there are physical constraints and design features that restrict the size of flood pulses that can be released from the dam (Shafroth *et al.*, 2001). Without large scouring floods, dense post-dam vegetation (much of which is exotic *Tamarix*) will remain fire-prone. Another anticipated consequence of the lack of large scouring floods is a decline in rates of establishment of pioneering *Populus* and *Salix*. These rates are predicted to decline despite the release of appropriately timed spring flows (Shafroth *et al.*, 1998). Solutions being pursued at the Bill Williams National Wildlife Refuge are intervention in the form of small-scale clearing of *Tamarix*, to be followed by *Populus-Salix* regeneration flows.

Actions that mimic fluvial processes, such as flood plain clearing followed by flood irrigation, also are being undertaken at other sites where natural processes cannot be fully restored (Friedman *et al.*, 1995). For example, at Bosque del Apache Wildlife Refuge, as on much of New Mexico's highly regulated Rio Grande, *Tamarix* has become dominant. Lowered water tables, increased river salinity, and lack of winter/spring floods have all contributed to decline of *Populus* forests, while past flood plain clearing and at least one appropriately timed summer flood allowed for influx of *Tamarix* (Everitt, 1998). To restore native forests, Refuge managers have mimicked the effects of large floods by using bulldozers, herbicides, and fire to clear extensive stands of *Tamarix* at a cost of \$750–1300 ha^{-1} (Taylor & McDaniel, 1998; Taylor *et al.*, 1999). They then released water onto the bare flood plains in spring with a seasonal timing that mimicked the natural flood hydrograph of the Rio Grande. This allowed for establishment of a diverse assemblage of plant species. Flood pulsing followed by seeding of native species also can be undertaken on agricultural fields, as has been done at places along the lower Colorado River. Restoration of agricultural fields ultimately may prove more cost-effective than restoration of *Tamarix* lands. Although land costs may be high, water delivery structures already are in place and vegetation clearing costs are negligible.

Loss of sediment

Dams and reservoirs impede downstream flows of sediment. Reduced transport of silt and clay may reduce biodiversity below dams, given that abundance of herbaceous

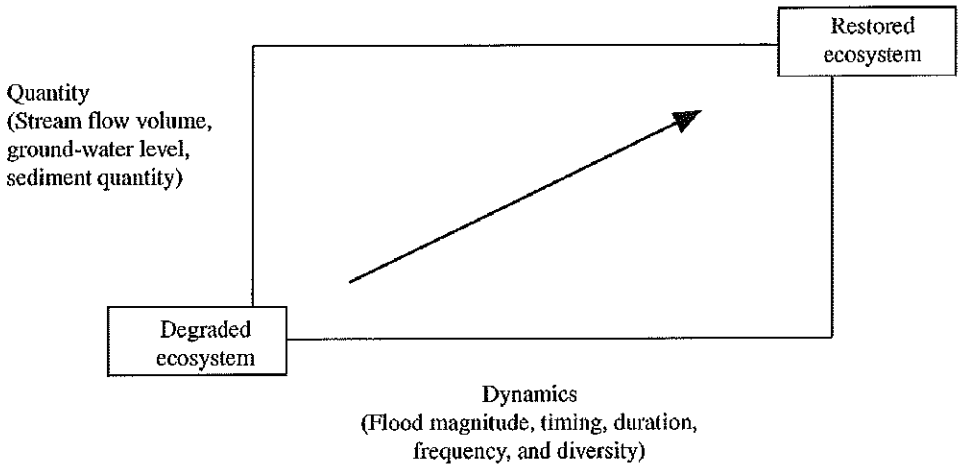

Figure 1. Conceptual diagram of flow regime changes needed for riparian ecosystem restoration.

riparian species tends to increase on fine soil textures (Stromberg, 1998*a*; Jansson *et al.*, 2000*a*). Biodiversity also declines as dynamic fluvial processes, such as channel migration, erosion, and sedimentation, become static (Shields *et al.* 2000) and spatial and temporal heterogeneity decline (Pollock *et al.*, 1998). Orthophosphates typically adheres to clays and silts and thus decline in below-dam systems as fine sediments are deposited in reservoirs, contributing to downstream productivity declines.

Sediment and nutrients can be restored to some below-dam reaches by adding sediment bypass structures (Schmidt *et al.*, 1998). Other barriers imposed by dams, such as restricted dispersal of plant propagules (Jansson *et al.*, 2000*b*), are more difficult to remedy, short of de-commissioning dams. Given such limitations, there is a need to assess economic and environmental costs and benefits of all dams in the south-western United States, as a basis for deciding which warrant removal or breaching (Shuman, 1995; Born *et al.*, 1998). At some sites, dam removal provides substantial ecological benefit, while minimally reducing the production of 'goods' (Wunderlich *et al.*, 1994).

Watershed alterations

Full restoration of riparian ecosystems depends on restoration of upland ecosystems throughout the watershed. Long-term overgrazing, fuel-wood harvest, and urbanization have, in places, reduced plant cover and soil in the uplands. These land-uses can result in flashier streams characterized by larger flood peaks and smaller base flows. In some areas, fire suppression has resulted in greater tree densities, higher transpiration rates, and smaller stream flows (Covington *et al.* 1997). Watershed restoration will require a mix of passive measures, such as restoring natural fire regimes and grazing regimes, and active measures. Controlled burns may be necessary for restoring structure and function to upland forests.

Conclusions

There is growing recognition of the important influence exerted by stream flows and fluvial processes on riparian vegetation structure and composition. Many riparian restoration projects are on a trajectory towards success because restoration managers

SRP13560

recognize the importance of restoring stream flow regimes, in other words, of restoring flows of water and sediment in sufficient quantities and with appropriate temporal and spatial patterns (Fig. 1). Substantial changes need to be made in the way we manage streams and water resources in the south-western United States. These changes include naturalizing flood pulses to restore fluvial dynamism, allowing rivers access to their flood plains and letting them be their own 'managers', recharging ground-water, reducing rates of water extraction, reducing dependence on hydro-power, and removing selected dams. Without such changes, we acquiesce to continued biodiversity losses and reduced function of riparian ecosystems. As Stanford *et al.* (1996) state, stream restoration calls for 'instituting a management belief system that relies upon natural habitat restoration' as opposed to artificial techniques such as plant propagation or installation of artificial in-stream structures (river engineering). Changes need to be made throughout society so that we become more efficient and less wasteful in our use of water and power. Scientists can help to insure that we restore desired conditions by focusing research endeavors on the water needs of aquatic and riparian ecosystems.

The comments and criticisms from Mike Scott, Duncan Patten, and Ed Glenn are greatly appreciated.

References

Alpert, P., Griggs, F.T. & Peterson, D.R. (1999). Riparian forest restoration along large rivers: initial results from the Sacramento River Project. *Restoration Ecology*, **7**: 360–368.

Bayley, P.B. (1991). The flood-pulse advantage and the restoration of river-floodplain systems. *Regulated Rivers: Research and Management*, **6**: 75–86.

Born, S.M., Genskow, K.D., Filbert, T.L., Hernandez-Mora, N., Keefer, M.L. & White, K.A. (1998). Socioeconomic and institutional dimensions of dam removals: the Wisconsin experience. *Environmental Management*, **22**: 359–370.

Briggs, M.K. (1996). *Riparian Ecosystem Recovery in Arid Lands: Strategies and References*. Tucson, Arizona: University of Arizona Press, 159 pp.

Briggs, M.K. & Cornelius, S. (1998). Opportunities for ecological improvement along the lower Colorado River and delta. *Wetlands*, **18**: 513–529.

Briggs, M.K., Roundy, B.A. & Shaw, W.W. (1994). Trial and error: assessing the effectiveness of riparian revegetation in Arizona. *Restoration & Management Notes*, **12**: 160–167.

Brock, J.H. (1994). *Tamarix* spp. (Salt cedar), an invasive exotic woody plant in arid and semi-arid riparian habitats of western U.S.A. *In*: de Waal, L.C., Child, L.E., Wade, P.M. & Brock, J.H. (Eds), *Ecology and Management of Invasive Riverside Plants*. pp. 27–44. West Sussex, England: John Wiley and Sons Ltd. 217 pp.

Brown, D.E. (Ed.), (1994). *Biotic Communities: South-western United States and North-western Mexico*. Salt Lake City: University of Utah Press. 342 pp.

Bryan, K. (1928). Change in plant association by change in ground water level. *Ecology*, **9**: 474–478.

Busch, D.E. (1995). Effects of fire on southwestern riparian plant community structure. *Southwestern Naturalist*, **40**: 259–267.

Busch, D.E. & Smith, S.D. (1995). Mechanisms associated with decline of woody species in riparian ecosystems of the Southwestern U.S. *Ecological Monographs*, **65**: 347–370.

Busch, D.E., Ingraham, N.L. & Smith, S.D. (1992). Water uptake in woody riparian phreatophytes of the Southwestern United States — a stable isotope study. *Ecological Applications*, **2**: 450–459.

Cleverly, J.R., Thibault, J.R., Pearce, J. & Dahm, C.N. (2000). Evapotranspiration from Middle Rio Grande (New Mexico) riparian ecosystems: the roles of flooding and community composition. *Ecological Society of America 85th Annual Meeting Abstracts*: 76.

Collins, S.L., Knapp, A.K., Briggs, J.M., Blair, J.M. & Steinauer, E.M. (1998). Modulation of diversity by grazing and mowing in native tallgrass prairie. *Science*, **280**: 745–747.

SRP13561

Cooper, D.J., Merritt, D.M., Anderson, D.C. & Chimner, R.A. (1999). Factors controlling the establishment of Fremont cottonwood seedlings on the Upper Green River, USA. *Regulated Rivers: Research and Management*, **15**: 419–440.

Covington, W.W., Fule, P.Z., Moore, M.M., Hart, S.C., Kolb, T.E., Mast, J.N., Sackett, S.S. & Wagner, M.R. (1997). Restoring ecosystem health in Ponderosa pine forests of the Southwest. *Journal of Forestry*, **95**: 23–29.

Devitt, D.A., Piorkowski, J.M., Smith, S.D., Cleverly, J.R. & Sala, A. (1997). Plant water relations of *Tamarix ramosissima* in response to the imposition and alleviation of soil moisture stress. *Journal of Arid Environments*, **36**: 527–540.

deWaal, L.C., Large, A.R.G. & Wade, P.M. (1998). *Rehabilitation of Rivers: Principles and Implementation*. New York: John Wiley and Sons. 331 pp.

Dixon, M.D. & Johnson, W.C. (1999). Riparian vegetation along the middle Snake River, Idaho: zonation, geographical trends, and historical changes. *Great Basin Naturalist*, **59**: 18–34.

Duda, A.M. & El-Ashry, M.T. (2000). Addressing the global water and environment crises through integrated approaches to the management of land, water and ecological resources. *Water International*, **25**: 115–126.

Ellis, L.M. (2001). Short-term response of woody plants to fire in a Rio Grande riparian forest, Central New Mexico, USA. *Biological Conservation*, **97**: 159–170.

Everitt, B.L. (1995). Hydrologic factors in regeneration of Fremont cottonwood along the Fremont River, Utah. Natural and Anthropogenic Influences in Fluvial Geomorphology. *Geophysical Monographs*, **89**: 197–208.

Everitt, B.L. (1998). Chronology of the spread of tamarisk in the central Rio Grande. *Wetlands*, **18**: 658–668.

Fenner, P., Brady, W.W. & Patton, D.R. (1985). Effects of regulated water flows on regeneration of Fremont cottonwood. *Journal of Range Management*, **38**: 135–138.

Friedman, J.M., Scott, M.L. & Lewis, W.M. (1995). Restoration of riparian forest using irrigation, artificial disturbance, and natural seedfall. *Environmental Management*, **19**: 547–557.

Gary, H.L. (1963). Root distribution of five-stamen tamarisk, seepwillow, and arrowweed. *Forest Science*, **9**: 311–314.

Glenn, E.P., Felger, R.S., A. Burquez, A. & Turner, D.S. (1992). Cienega de Santa Clara: endangered wetland in the Colorado River Delta, Sonora, Mexico. *Natural Resources Journal*, **32**: 817–824.

Glenn, E.P., Lee, C., Felger, R. & Zengel, S. (1996). Effects of water management on the wetlands of the Colorado River delta, Mexico. *Conservation Biology*, **10**: 1175–1186.

Glenn, E., Tanner, R., Mendez, S., Kehret, T., Moore, D., Garcia, J. & Valdes, C. (1998). Growth rates, salt tolerance and water use characteristics of native and invasive riparian plants from the delta of the Colorado River, Mexico. *Journal of Arid Environments*, **40**: 281–294.

Goodrich, D.C., Scott, R., Qi, J., Goff, B., Unkrich, C.L., Moran, M.S., Williams D., Schaeffer, S., Snyder, K., MacNish, R., Maddock, T., Pool, D., Chehbouni, A., Cooper, D., Eichinger, W., Shuttleworth, W., Kerr, Y., Marsett, W. & Ni, W. (2000). Seasonal estimates of riparian evapotranspiration using remote and in-situ measurements. *Journal of Agricultural and Forest Meteorology*, **105**: 281–309.

Goodwin, C.N., Hawkins, C.P. & Kershner, J.L. (1997). Riparian restoration in the Western United States: overview and perspective. *Restoration Ecology*, **5(4S)**: 4–14.

Gourley, C.R. (1998). Restoration of the lower Truckee River ecosystem: challenges and opportunities. *Journal of Land, Resources, & Environmental Law*, **18**: 113–121.

Grace, J.B. (1989). Effects of water depth on *Typha latifolia* and *Typha domingensis*. *American Journal of Botany*, **76**: 762–768.

Graf, W.L. (1982). Tamarisk and river-channel management. *Environmental Management*, **6**: 283–296.

Graf, W.L. (1999). Dam nation: a geographic census of American dams and their large-scale hydrologic impacts. *Water Resources Research*, **35**: 1305–1311.

Graf, W., Stromberg, J.C. & Valentine, B. (2001). Rivers, dams, and willow flycatchers: a summary of their science and policy connections. *Geomorphology*, In press.

Grimm, N.B. & Fisher, S.G. (1986). Nitrogen limitation potential of Arizona streams and rivers. *Journal of the Arizona Nevada Academy of Science*, **21**: 31–43.

Groeneveld, D.P. & Griepentrog, T.E. (1985). Interdependence of ground water, riparian vegetation, and streambank stability: a case study. *United States Forest Service General Technical Report*, **RM-120**: 44–48.

SRP13562

Hendrickson, D.A. & Minckley, W.S. (1984). Ciénegas-vanishing climax communities of the American Southwest. *Desert Plants*, **6**: 131–175.

Hey, D.L. & Philippi, N.S. (1999). *A Case for Wetland Restoration.* New York: John Wiley & Sons. 215 pp.

Hill, M.T. & Platts, W.S. (1998). Ecosystem restoration: a case study in the Owens River Gorge, California. *Fisheries*, **23**: 18–27.

Hill, M.T., Platts, W.S. & Beschta, R.L. (1991). Ecological and geomorphological concepts for instream and our-of-channel flow requirements. *Rivers*, **2**: 198–210.

Hobbs, R.J. & Norton, D.A. (1996). Towards a conceptual framework for restoration ecology. *Restoration Ecology*, **4**: 93–110.

Huckleberry, G. (1994). Contrasting channel response to floods on the middle Gila River, Arizona. *Geology*, **22**: 1083–1086.

Jansson, R., Nilsson, C., Dynesius, M. & Andersson, E. (2000*a*). Effects of river regulation on river-margin vegetation: a comparison of eight boreal rivers. *Ecological Applications*, **10**: 203–224.

Jansson, R., Nilsson, C. & Renofalt, B. (2000*b*). Fragmentation of riparian floras in rivers with multiple dams. *Ecology*, **81**: 899–903.

Johnson, W.C. (1998). Adjustment of riparian vegetation to river regulation in the Great Plains, USA. *Wetlands*, **18**: 608–618.

Judd J.B., Laughlin, J.M., Guenther, H.R. & Handergrade, R. (1971). The lethal decline of mesquite on the Casa Grande National Monument. *Great Basin Naturalist*, **31**: 153–159.

Kauffman, J.B., Beschta, R.L., Otting, N. & Lytjen, D. (1997). An ecological perspective of riparian and stream restoration in the western United States. *Fisheries*, **22**: 12–24.

Kus, B.E. (1998). Use of restored riparian habitat by the endangered least Bell's vireo. *Restoration Ecology*, **6**: 75–82.

Lambeck, R.J. (1997). Focal species: A multi-species umbrella for nature conservation. *Conservation Biology*, **11**: 849–856.

LCRBR Lower Colorado Region, Bureau of Reclamation. (2000). Long term restoration program for the historical Southwestern Willow Flycatcher (*Empidonax traillii extimus*) habitat along the Lower Colorado River, http: //www.lc.usbr.gov/~g2000.

Leffler, A.J., L.E. England, & J. Naito. (2000). Vulnerability of Fremont cottonwood (*Populus fremontii* Wats.) individuals to xylem cavitation. *Western North American Naturalist*, **60**: 204–210.

Maddock, I. (1999). The importance of physical habitat assessment for evaluating river health. *Freshwater Biology*, **41**: 373–391.

Mahoney, J.M. & Rood, S.B. (1998). Streamflow requirements for cottonwood seedling recruitment-an integrative model. *Wetlands*, **18**: 634–645.

Marks, J.B. (1950). Vegetation and soil relations in the Lower Colorado Desert. *Ecology*, **31**: 176–193.

Middleton, B. (1999). *Wetland Restoration: Flood Pulsing and Disturbance Dynamics.* New York: John Wiley & Sons, 388 pp.

Minckley, W.L. & Deacon, J.E. (Eds), (1991). *Battle Against Extinction: Native Fish Management in the American West.* Tucson, Arizona: University of Arizona Press, 517 pp.

Molles, M.C., Crawford, C.S., Ellis, L.M., Valett, H.M. & Dahm, C.N. (1998). Managed flooding for riparian ecosystem restoration — managed flooding reorganizes riparian forest ecosystems along the middle Rio Grande in New Mexico. *BioScience*, **48**: 749–756.

Nabhan, G.P. (1985). *Gathering the Desert.* Tucson, Arizona: University of Arizona Press, 232 pp.

Naiman, R.J., DeCamps, H. & Pollock, M. (1993). The role of riparian corridors in maintaining regional biodiversity. *Ecological Applications*, **3**: 209–212.

Oad, R. & DiSpigno, M. (1997). Water rights to return flow from urban landscape irrigation. *Journal of Irrigation and Drainage Engineering-ASCE*, **123**: 293–299.

Ohmart, R.D., Anderson, B.W. & Hunter, W.C. (1988). The ecology of the lower Colorado River from Davis Dam to the Mexico-United States international boundary: a community profile. *United States Fish and Wildlife Service Biological Report*, **85(7.19)**: 1–296.

O'Keefe, J. & Davies, B. (1991). Conservation and management of rivers of the Kruger National Park: suggested methods for calculating instream flow needs. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **1**: 55–71.

SRP13563

Patten, D.T. (1998). Riparian ecosystems of semi-arid North America: diversity and human impacts. *Wetlands*, **18**: 498–512.

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

Pollock, M.M., Naiman, R.J. & Hanley, T.A. (1998). Plant species richness in riparian wetlands-a test of biodiversity theory. *Ecology*, **79**: 94–105.

Rea, A.M. (1983). *Once a River–Bird Life and Habitat Changes on the Middle Gila.* Tucson, Arizona: University of Arizona Press, 285 pp.

Reiner, R.J. (2000). Restoring floodplain forest and annual grasslands by incorporating natural disturbance regimes at two sites in California. *Ecological Society of America 85th Annual Meeting Abstracts*: 30–31.

Richter, B.D., Mendelson, M.A. & Master, L.L. (1997*a*). Threats to imperiled freshwater fauna. *Conservation Biology*, **11**: 1081–1093.

Richter, B.D, Baumgartner, J.V., Wigington, R. & Braun, D.P. (1997*b*). How much water does a river need? *Freshwater Biology*, **37**: 231–249.

Riley, A.L. (1998). *Restoring Streams in Cities: a Guide for Planners, Policymakers, and Citizens.* Washington DC: Island Press, 443 pp.

Rood, S.B. & Mahoney, J.M. (1990). Collapse of riparian poplar forests downstream from dams in western prairies: probable causes and prospects for mitigation. *Environmental Management*, **14**: 451–464.

Rood, S.B., Mahoney, J.M., Reid, D.E. & Zilm, L. (1995). Instream flows and the decline of riparian cottonwoods along the St. Mary River, Alberta. *Canadian Journal of Botany*, **73**: 1250–1260.

Rood S.B, Kalischuk, A.R. & Mahoney, J.M. (1998). Initial cottonwood seedling recruitment following the flood of the century of the Oldman River, Alberta, Canada. *Wetlands*, **18**: 557–570.

Rood, S.B., Patino, S., Coombs. K. & Tyree, M.T. (2000). Branch sacrifice: cavitation-associated drought adaptation of riparian cottonwoods. *Trees-Structure and Function*, **14**: 248–257.

Schaeffer, S.M., Williams, D.G. & Goodrich, D.C. (2000). Transpiration of cottonwood/willow forest estimated from sap flux. *Journal of Agricultural and Forest Meteorology*, **105**: 257–270.

Schmidt, J.C., Webb, R.H., Valdez, R.A., Marzolf, G.R. & Stevens, L.E. (1998). Science and values in river restoration in the Grand Canyon. *BioScience*, **48**: 735–747.

Scott, R.L., Shuttleworth, W.J., Goodrich, D.C. & Maddock, T. (2000*a*). The water use of two dominant vegetation communities in a semiarid riparian ecosystem. *Journal of Agricultural and Forest Meteorology*, **105**: 241–256.

Scott, M.L., Lines, G.C. & Auble, G.T. (2000*b*). Channel incision and patterns of cottonwood stress and mortality along the Mojave River, California. *Journal of Arid Environments*, **44**: 399–414.

Scott, M.L., Shafroth, P.B. & Auble, G.T. (1999). Responses of riparian cottonwoods to alluvial water table declines. *Environmental Management*, **23**: 347–358.

Shafroth, P.B., Friedman J.M. & Ischinger, L.S. (1995). Effects of salinity on establishment of *Populus fremontii* (cottonwood) and *Tamarix ramosissima* (saltcedar) in Southwestern United States. *Great Basin Naturalist*, **55**: 58–65.

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

Shafroth, P.B., Stromberg, J.C. & Patten, D.T. (2000). Woody riparian vegetation response to different alluvial water-table regimes. *Western North American Naturalist*, **60**: 66–76.

Shafroth, P.B., Stromberg, J.C. & Patten, D.T. (2001). Riparian vegetation response to altered disturbance and stress regimes. *Ecological Applications*, In press.

Shields, F.D., Simon, A. & Steffen, L.J. (2000). Reservoir effects on downstream channel migration. *Environmental Conservation*, **27**: 54–66.

Shuman, J.R. (1995). Environmental considerations for assessing dam removal alternatives for river restoration. *Regulated Rivers-Research and Management*, **11**: 249–261.

Smith, S.D., Devitt, D.A., Sala, A., Cleverly, J.R. & Busch, D.E. (1998). Water relations of riparian plants from warm desert regions. *Wetlands*, **18**: 687–695.

Snyder, K.A. & Williams, D.G. (2000). Water sources used by riparian trees varies among stream types on the San Pedro River. *Journal of Agricultural and Forest Meteorology*, **105**: 227–240.

SRP13564

Sparks, R.E. (1995). Need for ecosystem management of large rivers and their floodplains. *BioScience*, **45**: 168–182.

Springer, A.E., Wright, J.M., Shafroth, P.B., Stromberg, J.C. & Patten, D.T. (1999). Coupling ground-water and riparian vegetation models to simulate riparian vegetation changes due to a reservoir release. *Water Resources Research*, **35**: 3621–3630.

Stanford, J.A., Ward, J.V., Liss, W.J., Frissell, C.A., Williams, R. N., Lichatowich, J.A. & Coutant, C.C. (1996). A general protocol for restoration of regulated rivers. *Regulated Rivers: Research and Management*, **12**: 391–413.

Stevens, L.E., Schmidt, J.C., Ayers, T.J. & Brown, B.T. (1994). Flow regulation, geomorphology, and Colorado River marsh development in the Grand Canyon, Arizona, *Ecological Applications*, **5**: 1025–1039.

Stromberg, J.C. (1997). Growth and survivorship of Fremont cottonwood, Goodding willow, and salt cedar seedlings after large floods in central Arizona. *Great Basin Naturalist*, **57**: 198–208.

Stromberg, J. (1998*b*). Dynamics of Fremont cottonwood (*Populus fremontii*) and saltcedar (*Tamarix chinensis*) populations along the San Pedro River, Arizona. *Journal of Arid Environments*, **40**: 133–155.

Stromberg, J.C. (1998*a*). Functional equivalency of saltcedar (*Tamarix chinensis*) and Fremont cottonwood (*Populus fremontii*) along a free-flowing river. *Wetlands*, **18**: 675–686.

Stromberg, J.C. (2001). Influence of stream flow regime and temperature on growth rate of the riparian tree, *Platanus wrightii*, in Arizona. *Freshwater Biology*, **46**: 227–240.

Stromberg, J.C. & Patten, D.T. (1995). Instream flow and cottonwood growth in the eastern Sierra Nevada of California USA. *Regulated Rivers*, **12**: 1–12.

Stromberg, J.C., Patten, D.T. & Richter, B.D. 1991. Flood flows and dynamics of Sonoran riparian forests. *Rivers*, **2**: 221–235.

Stromberg, J.C., Tress, J.A., Wilkins, S.D. & Clark, S. (1992). Response of velvet mesquite to groundwater decline. *Journal of Arid Environments*, **23**: 45–58.

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

Stromberg, J.C., Wilkins, S.D. & Tress, J.A. (1993*b*). Vegetation-hydrology models as management tools for velvet mesquite (*Prosopis velutina*) riparian ecosystems. *Ecological Applications*, **3**: 307–314.

Stromberg, J.C., Tiller, R, & Richter, B.D. (1996). Effects of groundwater decline on riparian vegetation of semiarid regions: the San Pedro River, Arizona, USA. *Ecological Applications*, **6**: 113–131.

Stromberg, J.C., Fry, J. & Patten, D.T. (1997). Marsh development after large floods in an alluvial, arid-land river. *Wetlands*, **17**: 292–300.

Swetnam, T.W. & Betancourt, J.L. (1998). Mesoscale disturbance and ecological response to decadal climatic variability in the American Southwest. *Journal of Climate*, **11**: 3128–3147.

Tabacchi, E., Planty-Tabacchi, A., Salinas, M.J. & DeCamps, H. (1996). Landscape structure and diversity in riparian plant communities: a longitudinal comparative study. *Regulated Rivers: Research and Management*, **12:** 367–390.

Taylor J.P. & McDaniel, K.C. (1998). Restoration of saltcedar (*Tamarix* sp.)-infested floodplains on the Bosque del Apache National Wildlife Refuge. *Weed Technology*, **12**: 345–352.

Taylor, J.P., Wester, D.B. & Smith, L.M. (1999). Soil disturbance, flood management, and riparian woody plant establishment in the Rio Grande floodplain. *Wetlands*, **19**: 372–382.

URMCC (Utah Reclamation Mitigation and Conservation Commission). (1997). Provo River Restoration Project: Final Environmental Impact Statement.

Webb, R.H. & . Betancourt, J.L. (1992). Climatic variability and flood frequency of the Santa Cruz River, Pima County, Arizona. *U. S. Geological Survey Water Supply Paper*, **2379**: 1–40.

Whisenant, S.G. (1999). *Repairing Damaged Wildlands: a Process-Orientated, Landscape-Scale Approach*. Cambridge, UK: Cambridge University Press, 312 pp.

Williams, J.E., Wood, C.A. & Dombeck, M.P. (Eds), 1997. *Watershed Restoration: Principles and Practices*. Bethesda, Maryland: American Fisheries Society, 559 pp.

Willms, J., Rood, S.B., Willms, W. & Tyree, M. (1998). Branch growth of riparian cottonwoods: a hydrologically sensitive dendrochronological tool. *Trees*, **12**: 215–223.

Wissmar, R.C. & Beschta, R.L. (1998). Restoration and management of riparian ecosystems: a catchment perspective. *Freshwater Biology*, **40**: 571–585.

Wolden, L., Stromberg, J.C. & Patten, D.T. (1994). Flora and vegetation of the Hassayampa River Preserve. *Journal of the Arizona Nevada Academy of Science*, **28**: 76–111.

SRP13565

Wunderlich R.C., Winter, B.D. & Meyer, J.H. (1994). Restoration of the Elwha River ecosystem. *Fisheries*, **19**: 11–19.

Zamora-Arroyo, F., Nagler, P., Briggs, M., Radtke, D., Rodriquez, H., Garcia, J., Valdes, C., Huete, A. & Glenn, E. (2001). Regeneration of native trees in response to flood releases from the United States into the delta of the Colorado River, Mexico. *Journal of Arid Environments*, **49**: 49–64. doi:10.1006/jare.2001.0835.