Spring 2011

DO INVASIVE RIPARIAN WOODY PLANTS AFFECT HYDROLOGY AND ECOSYSTEM PROCESSES?

Julie Huddle
University of Nebraska - Lincoln, jhuddle2@unl.edu

Tala Awada
University of Nebraska-Lincoln, tawada2@unl.edu

Derrel Martin
University of Nebraska-Lincoln, derrel.martin@unl.edu

Xinhua Zhou
University of Nebraska - Lincoln, xzhou2@unl.edu

Sue Ellen Pegg
University of Nebraska-Lincoln, spegg2@unl.edu

See next page for additional authors
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Authors

Julie Huddle, Tala Awada, Derrel Martin, Xinhua Zhou, Sue Ellen Pegg, and Scott Josiah

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Julie A. Huddle and Tala Awada

School of Natural Resources
Hardin Hall
University of Nebraska–Lincoln
Lincoln, NE 68583-0974
jhuddle2@unl.edu

Derrel L. Martin

Biological Systems Engineering
243 Chase Hall
University of Nebraska–Lincoln
Lincoln, NE 68583-0726

Xinhua Zhou and Sue Ellen Pegg

School of Natural Resources
Hardin Hall
University of Nebraska–Lincoln
Lincoln, NE 68583-0974

and

Scott J. Josiah

Nebraska Forest Service
103AA Entomology
University of Nebraska–Lincoln
Lincoln, NE 68583-0815

ABSTRACT—Political and socioeconomic pressures on riparian areas in semiarid regions of the Great Plains are growing as water resources become more limited. Management along waterways has altered stream ecology and hydrology in ways that encourage the invasion and expansion of native (e.g., *Juniperus virginiana*) and non-native (e.g., *Tamarix* sp. and *Elaeagnus angustifolia*) woody species. One management tool currently implemented to restore the hydrology or increase water yields along waterways in semiarid areas is the removal of vegetation or invasive species. How managers should respond to invasive woody plants to optimize hydrological functions without compromising other riparian ecosystem functions is still debatable. In this manuscript, we provide an overview of the ecological status and hydrological role of riparian vegetation in the northern Great Plains, with examples drawn from the region and other semiarid areas. Additionally, we present information compiled from published studies on water consumption of native and non-native species at both tree and stand levels, and we evaluate the ecohydrological outcomes from removal of invasive woody vegetation. Lastly, we consider the economic costs and benefits of woody species removal, and suggest considerations to help managers make decisions regarding woody species removal.

Key Words: ecohydrology, evapotranspiration, *Juniperus*, phreatophytes, *Populus*, riparian forests, *Tamarix*, woody species encroachment

INTRODUCTION

The word "riparian" is derived from the Latin word "riparious" which means "bank" of a stream, and refers to the land, flora, and fauna adjacent to or on the bank of a body of water (Ilhardt et al. 2000). Ilhardt et al. (2000) defined riparian areas functionally as "three-dimensional ecotones of interaction that include terrestrial and aquatic ecosystems, that extend down into the groundwater, up above the canopy, outward across the floodplain, up the near-slopes that drain to the water, laterally into the terrestrial ecosystem, and along the water course at a variable width." Thus, riparian areas encompass plant communities that are growing inside as well as outside the hydrological zones (Naiman and Décamps 1997). Such communities exert direct and indirect biological, physical, and chemical influence on, and are influenced by, an adjacent water body through both above- and below-ground interactions (Odum 1971). The ecosystem functions played by riparian areas as species conduit, barrier, energy source, energy sink, and habitat are well documented. These services result in terrestrial and aquatic biodiversity, corridors and habitats for wildlife, stream-bank stabilization, soil protection, water storage, groundwater recharge, mediation of seasonal water-level fluctuations, improved water quality, nutrient cycling, carbon sequestration, climate regulation, nonpoint pollution control, aesthetic, educational opportunities, as well as economically important products, biofuels, and water production (Haycock et al. 1997; Lynch and Tjaden 2000; Lee et al. 2003; Núñez et al. 2006; Sun et al. 2006; Rahel and Olden 2008).

In recent decades riparian areas have experienced changes in vegetative cover as invasive nonriparian woody species spread at the expense of ecologically important native riparian species (Cleverly et al. 2006). These changes reflect factors like natural species migration, altered management practices, damming, fire and flood control, climate change, and nitrogen depositions (Tabacchi et al. 2000). Ecological changes resulting from nonriparian woody species encroachment have substantially affected the hydrological cycle and water yields in several ecosystems. These ecohydrological changes are of particular concern in semiarid areas like the Great Plains (Stromberg et al. 1996; Warren et al. 1996; Heilman et al. 2009), where water for human use is in high demand, and where managers have been required to use water budgets as well as quantifying ecosystem responses to various water management scenarios (Dahm et al. 2002). Competing demands for water resources

and the role of riparian evapotranspiration in depleting watershed-level water budgets are prompting land managers and policy makers to seek better understanding of the ecological and socioeconomic roles of riparian systems so that water resource management can be improved. The objectives of this paper are therefore (1) to provide an overview on the current ecological status and hydrological role of riparian vegetation in the northern Great Plains, drawing examples from the region and other semi-arid areas; (2) to present published studies on water consumption in native and invasive woody species in various regions of the Great Plains and other semiarid areas; and (3) to assess the possible ecohydrological environmental effects and economic cost of the removal of invasive woody vegetation.

RIPARIAN VEGETATION IN THE NORTHERN GREAT PLAINS

Woody riparian vegetation in the Great Plains is dominated by the cottonwood species complexes that inhabit floodplain regions of sizeable rivers and streams, subirrigated valleys, and minor drainages as small groves or scattered individuals (Barker and Whitman 1988). Cottonwoods are pioneer species that produce short-lived seeds, which are carried over long distances by wind and require exposed sediments for seed germination and establishment (Taylor 2001). As rivers flood and meander, newly exposed moist substrate becomes available for cottonwood to colonize (Johnson et al. 1976; Friedman et al. 2006). In areas suitable for their establishment, cottonwoods can outcompete saltcedar simply by growing more quickly (Sher et al. 2000). Mature cottonwood trees depend on groundwater access to survive and are thus largely restricted to riparian areas supplied with alluvial groundwater originating from streamflow (Busch et al. 1992). In the northern Great Plains, cottonwood (*Populus deltoides*) usually dominates young stands. Over time, other, more shade-tolerant species such as green ash (*Fraxinus pennsylvanica*), boxelder (*Acer negundo*), American elm (*Ulmus americana*), hackberry (*Celtis occidentalis*), and peach-leaf willow (*Salix amygdaloides*), and occasionally bur oak (*Quercus macrocarpa*), catalpa (*Catalpa speciosa*), and walnut (*Juglans* spp.) regenerate beneath the cottonwoods as understory associates. Islands and stream banks in riparian areas are known to contain dense thickets of sandbar willow (*Salix exigua*) (Barker and Whitman 1988) and black willow (*Salix nigra*). The latter is often present in moist forest regions like those found in the southeastern portion of Nebraska (Rothenberger 1987). Roughleaf dogwood (*Cornus*

drummondii) is common in many lowland wooded areas, and honeylocust (*Gleditisia triacanthos*), which is adapted to a variety of environments, is especially tolerant of arid banks with southern exposures. Eastern redcedar (*Juniperus virginiana*) and chokecherry (*Prunus virginiana*) are widespread throughout the floodplain (Rothenberger 1987).

Cottonwood forest types along riparian areas are particularly vulnerable to floodplain degradation due to activities such as damming, livestock production, clearing land for agriculture, and human settlement and transportation corridors (Rood et al. 2003). An example of these changes took place along the Platte River in Nebraska, where, by 1900, pioneers, homesteaders, and Native Americans harvested large areas of woody vegetation along these riparian corridors, shifting this ecosystem from a broad region of heavily forested riparian woodlands to areas of herbaceous vegetative cover and agricultural fields (Johnson and Boettcher 2000; West and Ruark 2004).

Human activities have created conflict among stakeholders interested in the different environmental benefits and have impacted services provided by riparian areas (Auble et al. 1994). Damming, channelization, irrigation, and agricultural expansion along these waterways have aggravated the effects of other human disturbances by constraining natural water flow, abbreviating flooding and subsequent scouring, and augmenting water table variability. For example, damming in the upper Missouri River in North Dakota led to 77% and 98% reductions in average annual area eroded and deposited, respectively. Reduced river meandering decreased sediment loads, resulting in narrower and shallower channels. As a consequence, riparian forests composition along the Missouri River changed from being 47% dominated by young pioneer species of cottonwoods and willows prior to European settlement, to a forest where pioneer species accounted for only 6% of the species composition, an 87% drop, in 1979 (Johnson 1992). Intensive agricultural production during the 20th century developed in the Great Plains with little understanding of its impacts on natural resources including native grasslands, water, wildlife, and woodlands (Hughes 1994; Johnson 1997). Productive soils next to waterways were often cleared for row crop production, thus reducing buffer zones, and livestock overgrazing resulted in considerable losses in forest cover, increased soil erosion, and damage to linked plant, animal, soil, and water assets.

As human activities have altered river hydrology and reduced native riparian vegetation, many studies have documented increases in non-native woody invasive species, such as Russian olive (*Elaeagnus angustifolia*) and saltcedar (*Tamarix chinensis*), and native aggressive woody species, such as eastern redcedar (Carmen and Brotherson 1982; Barker and Whitman 1988; Johnson 1994, 1997; Glenn et al. 1998; Lesica and Miles 1999; Friedman et al. 2005, 2006). As invasive species have spread, ecohydrologic consequences, such as drops in the water table and reduced water yields, have been observed in riparian systems (Sala et al. 1996; Huxman et al. 2005; Pataki et al. 2005; Scott et al. 2006; Wilcox and Thurow 2006). Woody species encroachment and their potential impacts on water resources in semiarid regions of the Great Plains are addressed in the following sections.

WOODY SPECIES ENCROACHMENT

Altered hydrological regimes have been shown to change riparian community composition, structure, and function, and in many cases, to increase encroachment of native (nonriparian) aggressive and non-native woody species (Tabacchi et al. 1996; Graf 2001). In Arizona, Horton et al. (2001a, 2001b) observed that native riparian species like western cottonwood (*Populus fremontii*) and Goodding's willow (*Salix gooddingii*) died when they could no longer access groundwater because river modification and climatic factors reduced water availability and thus facilitated the expansion of saltcedar into these riparian areas. Similarly, western cottonwood stands were replaced by saltcedar in the absence of flood disturbance along the Lower Escalante River in Utah as cottonwood clones matured, thinned, and died (Irvine and West 1979). Along the Verde River in Arizona, saltcedar stand density was greater in reaches where water flow was regulated compared with reaches where water flow was unregulated; the opposite trend held true for mature (11–40 year) and old-growth (>40 year) cottonwood stands (Beauchamp and Stromberg 2007). Saltcedar and Russian olive replaced cottonwood stands as well along the Rio Grande in New Mexico and along the Marias River in Montana due to changes in river hydrology (Howe and Knopf 1991; Lesica and Miles 1999). Vegetation surveyed along the San Pedro River in Arizona was dominated by *T. ramosissima* where flow frequencies ranged between 40% and 60%, groundwater depth was between 3.2 m and 3.8 m, and groundwater depths fluctuated between 0.59 m and 0.75 m (Lite and Stromberg 2003). In contrast, *Populus fremontii* dominated sites where flow frequencies ranged between 73% and 78%, groundwater depth was between 2.4 m and 2.9 m, and groundwater depths fluctuations

lower than 0.48 m and 0.56 m. Along 20 reaches in four southwestern states (AZ, NM, UT, and CO), saltcedar is more abundant in reaches where hydrology has changed (Mortenson and Weisberg 2010). However, Russian olive abundance was uncorrelated with hydrologic changes.

Russian olive, saltcedar, and eastern redcedar have been widely planted throughout the United States since the 1900s. Both Russian olive and saltcedar produce large amounts of small seeds capable of germinating under a wide range of environmental conditions (Sala et al. 1996; Katz and Shafroth 2003). Eastern redcedar seeds are eaten and dispersed by birds (Lawson 1990). Friedman et al. (2005), in a study that examined non-native riparian tree species in the western United States, reported that saltcedar and Russian olive have become the third and fourth most frequently occurring woody riparian species in the region, with saltcedar dominating low elevation sites in the southwestern United States and Russian olive being abundant in the northern Great Plains. Saltcedar introduced in central Wyoming between 1936 and 1953 spread northward into the Yellowstone and Missouri rivers in Montana by 1951 and 1967 (Pearce and Smith 2007). The spread of *Tamarix* in the western United States is limited by its sensitivity to frost (Friedman et al. 2005), suggesting that its future spread could be impacted by changes in minimum temperatures. A mid- to late successional species, the native eastern redcedar (McKinley and Blair 2008) has also invaded as disturbance regimes and stream hydrology have changed. The expansion of invasive woody species is predicted to further alter the ecohydrology of streams with potentially adverse consequences on water budgets.

RIPARIAN VEGETATION AND WATER RESOURCES

Tabacchi et al. (2000) describe how riparian vegetation influences water availability in three ways. First, the physical structure of riparian vegetation largely determines the fate of sediment and nutrients from adjacent terrestrial ecosystems carried by runoff by altering overland flow rates. Second, both sediment retention and nutrient cycling within the riparian ecosystem control both the amount of nutrients that can reach the aquatic ecosystem and the stream water temperature due to shading. Organic debris from riparian vegetation can provide food and habitat for aquatic species. Thus, riparian ecosystems exert both physical and biological influences over the water quality of aquatic ecosystems. Third, processes such as soil evaporation, plant water consumption, and

foliage interception of precipitation affect the hydraulic conductivity between the terrestrial ecosystem and the aquatic ecosystem. In this review, we briefly address the physical and chemical roles of vegetation in riparian areas, emphasizing how vegetation affects hydrology through water consumption.

Water Flow and Physical Structure of Vegetation

The physical structure of vegetation has been shown to obstruct, facilitate, or divert water flow and therefore impact hydraulic connectivity and other large-scale water properties (Tabacchi et al. 2000), with vegetation responses and impacts varying between and within geographic regions and stream types. For example, Guillemette et al. (2005) in a review of 50 watershed basin studies concluded that the removal of over 50% of basin vegetation will result in flooding and erosion. On the other hand, pioneer species that colonize immediately following disturbance may increase heterogeneity of the water flow pattern while dense herbaceous cover may limit surface water infiltration and trap fine sediments that sustain moisture levels in the upper soil profile during dry periods. Johnson (1994) reported that flow reductions due to hydrological alterations along the braided Platte River in Nebraska initially favored the spread of cottonwood-willow forests. As cottonwood-willow forests spread, they transformed the river from a wide channel characterized by scattered patches of woody vegetation to a narrow, tree-lined channel that further altered water movement and yields.

Water Quality and Riparian Vegetation

It is well known that riparian vegetation filters and retains sediment and limits nonpoint source pollution from agricultural fertilizers and pesticides in cropland and rangeland areas, thus improving water quality in streams (Schultz et al. 1994; Schmitt et al. 1999; Dosskey et al. 2002; Richardson et al. 2007). Riparian vegetation also produces subsurface organic matter that fuels microbial denitrification and results in nitrogen losses from plant litter released into the atmosphere (Schade et al. 2001). However, litter of the invasive Russian olive or saltcedar contains a higher nitrogen content compared with that of cottonwood (Tibbets and Molles 2005). Elevated nitrogen levels in litter increase nitrification rates and change soil chemistry (Ehrenfeld 2003). Such alterations to the biogeochemical cycle are thought to limit the capacity of riparian vegetation to

absorb nutrients from neighboring agricultural land and to increase leaching, although the relationship between filtering capacity and leaching needs to be better tested in natural environments (Dosskey 2002). Ordination analysis of vegetation composition in rangelands located within 4 km of Utah Lake in Utah indicates that sites infested with saltcedar occur on dry saline sites dominated by annual species compared with uninfested sites (Carman and Brotherson 1982). Other studies indicate that high floodplain salinity levels can prevent restoration of riparian vegetation such as cottonwood forests in some river reaches (Shafroth et al. 2008). At the Bosque del Apache National Wildlife Refuge, species selected to restore sites after saltcedar removal are based on salinity levels, and cottonwood is planted when the soil salinity is low, ranging between 1.0 and 2.0 deciSiemen m^{-1} (Taylor and McDaniel 1998). Increased salt levels observed along the Colorado River in Mexico, associated with lack of overbank flooding, favors replacement of native forests with invasive species (Glenn et al. 1998). Thus, changes in soil biochemistry caused by leaching, vegetation changes, and nutrient cycling could have long-lasting effects on future riparian vegetation composition.

Water Consumption by Riparian Vegetation

Evapotranspiration, comprised of both transpiration (water loss) by plants and evaporation from the surfaces, is significantly influenced by vegetation structure, function, and composition, with subsequent impacts on the water balance and yields and on streamflow in riparian areas (Richardson et al. 2007). On a watershed scale, changes to vegetative cover can significantly alter the balance between evapotranspiration and streamflow (Hornbeck et al. 1993; Huxman et al. 2005). These alterations result from changes in (1) species composition and therefore transpiration rates (Table 1); (2) vegetation structure and architecture, thus the ratio of precipitation intercepted by vegetation and lost to evaporation (e.g., interception losses in temperate regions vary between 12% and 50% depending on species; Tabacchi et al. 2000); and (3) soil permeability (Huxman et al. 2005) and the resulting water infiltration due to root and litter characteristics. For example, the litter of species such as juniper and pine can trap moisture above the soil while leachates can create a hydrophobic soil layer that blocks water movement through dry soil (Madsen et al. 2008). Compared with upland sites, riparian phreatophytes use more water on an area basis. Although riparian vegetation covered about

8% of the Middle Rio Grande watershed in 1999 (Dahm et al. 2002), a water budget indicated that riparian evapotranspiration accounted for roughly 29% (5.3 $m^3 s^{-1}$ out of 18.5 $m^3 s^{-1}$) of the water lost (Cleverly et al. 2002).

Vegetation water balance depends on soil water availability, plant water uptake and storage, and evapotranspiration rates. Plants in riparian areas derive their water from rainfall, rivers and streams, groundwater, or a combination thereof. For example, in Utah, Dawson and Ehleringer (1991) reported that as *Populus angustifolia* trees mature they shift from using surface stream water to using deeper water feeding into the stream. Other species, like the Fremont cottonwood (*Populus fremontii*), utilize groundwater and stream water except during periods of low streamflow when they rely on growing-season rainfall (Busch et al. 1992; Leffler and Evans 1999). In contrast, perennial grassland communities and riparian shrubs were found to generally rely on a combination of current rainfall and water from the unsaturated soil profile.

As woody species invade and alter species composition, the source and amount of water uptake by plants are altered, affecting not only the site water balance but also the water available to native trees. For instance, phreatophyte trees that rely solely on groundwater in the semiarid areas of the Great Plains are highly susceptible to groundwater fluctuations caused by physical factors such as precipitation or biological factors such as competition from other species (Scott et al. 2000). Invasive woody species like saltcedar, Russian olive, and eastern redcedar, which use multiple water sources including the groundwater and the unsaturated soil profile, have been shown to have the capability to reduce the amount of water available for native species (Busch et al. 1992; Snyder and Williams 2000). The ability of native species in this case to display plasticity in water uptake becomes essential for survival as groundwater levels drop. Such plasticity is observed in western cottonwood, the dominant riparian species in the semiarid ecosystem along the San Pedro River in Arizona, which primarily uses groundwater as a source but was found to derive up to 33% of its transpiration water from precipitation during the rainy season (Snyder and Williams 2000). An associate of the western cottonwood, Goodding's willow, does not possess this plasticity and has been reported to exclusively use groundwater despite available water in unsaturated soils, thus restricting its ability to compete for water resources (Snyder and Williams 2000).

The temporal, spatial, and amount of water uptake by riparian vegetation is dependent on species, ecotype, and age and is influenced by biotic and abiotic factors.

TABLE 1

WATER USE OF MAJOR RIPARIAN TREE SPECIES IN THE GREAT PLAINS AS A FUNCTION
OF PRECIPITATION, LEAF AREA INDEX, DEPTH TO GROUNDWATER (DGW m), VOLUMETRIC
SOIL MOISTURE (%), OR VAPOR PRESSURE DEFICIT (VPD kPA)

Species	Location	Precipitation (mm)	Water use rates	Method	Reference
Elaeagnus angustifolia	Middle Rio Grande, NM		1,230 mm yr^{-1}	Eddy covariance	Dahm et al. 2002
Juniperus ashei	Uvalde County, TX 1991–1995	676 mm Mar.–Oct. 945 mm total yr	1.90 mm day^{-1} (Mar.-Oct.) (520 mm season[-1])	Bowen ratio energy balance	Dugas et al. 1998
J. deppeanna	Beaver Creek, AZ	553 mm yr^{-1}	432 mm yr^{-1} (1.21 mm day^{-1})	Water balance	Lane and Barnes 1987
J. osteosperma	Beaver Creek, AZ	441 mm yr^{-1}	414 mm yr^{-1} (1.13 mm day^{-1})	Water balance	Lane and Barnes 1987
J. osteosperma	Rush Valley, UT	239 mm yr^{-1}	0.85 mm day^{-1}	Eddy covariance Mar.–Oct. 2001	Leffler et al. 2002
J. osteosperma / Pinus monophylla	Los Alamos, NM Juniper leaf area index (LAI): 0.70	216 mm yr^{-1}	121 mm yr^{-1} *J. osteosperma* (0.33 mm day^{-1})	Water balance	Lane and Barnes 1987
	Pine LAI: 3.07		95 mm yr^{-1} *P. monophylla*		
J. osteosperma / Pinus monophylla	Los Alamos, NM Juniper LAI: 1.31	203 mm yr^{-1}	193 mm yr^{-1} *J. osteosperma* (0.53 mm day^{-1})	Water balance	Lane and Barnes 1987
	Pine LAI: 0.41		10 mm yr^{-1} *P. monophylla*		
J. osteosperma / Pinus monophylla	Los Alamos, NM Juniper LAI: 1.92	170 mm yr^{-1}	125 mm yr^{-1} *J. osteosperma* (0.34 mm day^{-1})	Water balance	Lane and Barnes 1987
	Pine LAI: 0.76		45 mm yr^{-1} *P. monophylla*		
J. virginiana One 15.2-cm-diameter tree	Odessa, NE 2004	844 mm yr^{-1}	1.0 mm day^{-1} April (62.3 L day^{-1} tree[-1]) 0.8 mm day^{-1} May-August (48.4 L day^{-1} tree[-1])	3 cm Granier sap flow probes	Landon et al. 2009
P. deltoides var. *wislizenii*	Middle Rio Grande, NM, Belen	Flooded site	980 mm yr^{-1} 2000 (2.70 mm day^{-1})	Eddy covariance	Dahm et al. 2002
P. deltoides 27 trees/ha with 80 trees/ha 9.0 m average distance between trees	Odessa, NE 2004	844 mm yr^{-1}	61-cm-diameter tree 0.26 mm day^{-1} April (16.5 kg day^{-1} tree[-1]) 3.78 mm day^{-1} May-August (240.5 kg day^{-1} tree[-1])	Granier sap flow 5 cm long probes	Landon et al. 2009
			22.9 cm diameter, tree 0.05 mm day^{-1} April (3.2 kg day^{-1} tree[-1]) 1.44 mm day^{-1} May-August (91.6 kg day^{-1} tree[-1])		
P. deltoides 3-year-old cuttings, 1,333 trees/ha	USDE Savanna River site, Aiken, SC	7.4 % soil water content	ST66: 24.0 mm mo[-1] (0.80 mm day^{-1}) S7C15: 33.4 mm mo[-1] (1.13 mm day^{-1})	Granier sap flow 2 cm long probes	Samuelson et al. 2007
P. deltoides 2,019 trees/ha 1-yr-old seedlings	Carswell Air Force Base, TX	800 mm yr^{-1} 2–3.4 m DGW 153 day season	255 mm yr^{-1} (0.70 mm day^{-1}) (8.2 kg day^{-1} tree[-1])	Granier sap flow	Vose et al. 2000
P. fremontii	Bill Williams and Colorado rivers, northwest Arizona Monthly measurements, 1989 and 1990	130 mm yr^{-1}	0.5 to 3.8 mmol m^{-2} s^{-1}, composite diurnal curve (360 mm yr^{-1}) (1.0 mm day^{-1}) (89 kg day^{-1} tree[-1])	Licor 1600 steady state porometer	Busch and Smith 1995
P. fremontii 3-year-old poles planted 4 m apart	Lower Colorado River, California, Cibola National Wildlife Refuge		1,200 mm yr^{-1} (3.3 mm day^{-1})	Time series MODIS satellite	Nagler et al. 2007
P. fremontii	Matheson Wetlands Reserve, Moab, UT 220 mm yr^{-1} annual ppt	Control site 0.1–2.5 m DGW	9.3 ± 0.2 mm day^{-1}	Granier sap flux 2 cm long probes 17 July–21 September	Pataki et al. 2005
		Saline site 0.7–2.7 m DGW	4.8 ± 0.1 mm day^{-1}		
P. fremontii	San Pedro River	247 mm yr^{-1}	1,271 mm yr^{-1} (3.5 mm day^{-1})	Bowens ratio	Scott et al. 2000
P. grandidentata 90-year-old trees in mixed forest	Michigan Biological Station, Pellston, MI July–August 1999 10:30–13:30 774 mm yr^{-1}	Soil moisture < 10%	0.09 mm hr^{-1} (3.12 kg hr^{-1} tree[-1])	Granier sap flux, 3 cm long probes, 4 trees	Bovard et al. 2005
		Soil moisture >15%	0.11 mm hr^{-1} (3.57 kg hr^{-1} tree[-1])		

TABLE 1 continued

Species	Location	Precipitation (mm)	Water use rates	Method	Reference
P. tremuloides 70-year-old stand	Prince Albert National Park, Saskatchewan, Canada	450 mm in 1994	280 mm yr^{-1} poplar trees (0.77 mm day^{-1})	Eddy flux tower	Black et al. 2006
P. tremuloides >80-year-old stand 21 cm dbh 980 trees/ha	Saskatchewan, Canada 123-day growing season	235 mm yr^{-1} 2001 286 mm yr^{-1} 2002	441 mm yr^{-1} (1.21 mm day^{-1}) (281 kg day^{-1} tree $^{-1}$) 323 mm yr^{-1} (0.88 mm day^{-1}) (206 kg day^{-1} tree $^{-1}$)	Eddy flux tower	Amiro et al. 2006
Populus tremuloides 70-year-old stand	Saskatchewan, Canada	488 mm yr^{-1}	120 mm yr^{-1} above hazelnut (0.33 mm day^{-1})	Eddy flux tower Late May-Oct.	Blanken and Black 2004
P. tremuloides, 150-year-old mixed stand wildfire chrono-sequence	Central Manitoba, Canada 439 mm mean annual precipitation	12-year-old, 6,800 trees/ha, 1.2 cm dbh	70% of surface conductance 8 mm yr^{-1} (0.53 kg day^{-1} tree^{-1}, growing season)	Kucera-type sap flow sensors for trees <4 cm dbh and Granier sap flow sensors for trees >4 cm dbh	Ewers et al. 2005
P. tremuloides, 9.0-17.8 m tall Comprised 13% total basal areal; *Abies balsamea* understory, 2.9-5.9 m tall	Chequamegon Ecosystem Atmosphere Study, Park Falls, WI	20-year-old, 3,500 trees/ha, 3.5 cm dbh 37-year-old, 4,200 trees/ha, 7.7 cm dbh	8 mm yr^{-1} (0.51 kg day^{-1} tree^{-1}, growing season) 37 mm yr^{-1} (2.6 kg day^{-1} tree^{-1}, growing season)	Kucera-type sap flow sensors for trees <4 cm dbh and Granier sap flow sensors for trees >4 cm dbh	Ewers et al. 2007
P. tremuloides	Central Saskatchewan, Canada VPD >1 kPa >100 W m^{-2} light	Parkland site 375 mm yr^{-1} Boreal site 462 mm yr^{-1}	4.8 mm day^{-1} 9.6 mm day^{-1}	Heat pulse sap flow	Hogg and Hurdle 1997
Populus/Salix	Middle Rio Grande, NM Upper San Pedro, AZ Lower Colorado, AZ		1,100–1,300 mm yr^{-1} (3.0–3.6 mm day^{-1})	MODIS and EVI data calibrated with eddy covariance	Nagler et al. 2005b
Prosopis velutina	Bosque del Apache National Wildlife Refuge, NM		400–1,100 mm yr^{-1} (1.1–3.0 mm day^{-1})	Scaled P-M for C/W and Bowen ratio MODIS and EVI	Goodrich et al. 2000; Nagler et al. 2005b
P. velutina	San Pedro Basin, southeastern Arizona	247 mm yr^{-1}	375 mm yr^{-1} (1.02 mm day^{-1})	Eddy covariance	Nagler et al. 2005
P. velutina	Upper San Pedro River, Arizona *P. velutina* savanna	6.4 m DGW 234 mm yr^{-1}	450 mm yr^{-1} (1.23 mm day^{-1})	Bowen ratio energy balance (May–September)	Scott et al. 2000
P. velutina	Upper San Pedro River, Arizona *P. velutina* savanna	253 mm yr^{-1} 2001 293 mm yr^{-1} 2000	485 mm yr^{-1} (2.3 mm day^{-1}) 401 mm yr^{-1} (1.23 mm day^{-1})	Eddy covariance (May–November)	Scott et al. 2004
P. velutina Semiarid lands	Floodplain terraces San Pedro River with *Sporobolus wrightii*	2.6 m DGW 343 mm yr^{-1}	407 mm yr^{-1} (1.11 mm day^{-1})	Eddy covariance, MODIS	Scott et al. 2006
Tamarix elongata 8-year-old stands	Shiyang River basin, northwest China	134 mm yr^{-1}	740 mm yr^{-1} (2.0 mm day^{-1})	Heat pulse sap flow	Qu et al. 2007
T. ramosissima	Bill Williams and Colorado rivers, northwest Arizona		1,220 mm yr^{-1} (3.34 mm day^{-1})	Steady state porometer, composite 1989–1990 diurnal curve	Busch and Smith 1995
T. ramosissima Nonflooded site	Middle Rio Grande, NM Sevilleta National Wildlife Refuge	3.4–3.7 m DGW	740 mm yr^{-1} 1999 (2.0 mm day^{-1}) 760 mm yr^{-1} 2000 (2.1 mm day^{-1})	Eddy covariance	Cleverly et al. 2002; Dahm et al. 2002
T. ramosissima Flooded site	Middle Rio Grande, NM Bosque del Apache National Wildlife Refuge	3.7–4.0m DGW	1,220 mm yr^{-1} 1999 (3.3 mm day^{-1}) 1,100 mm yr^{-1} 2000 (3.0 mm day^{-1})	Eddy covariance	Cleverly et al. 2002; Dahm et al. 2002
T. ramosissima	Middle Rio Grande, NM Upper San Pedro, AZ Lower Colorado, AZ		300–1,300 mm yr^{-1} (0.8–3.6 mm day^{-1})	MODIS and EVI data calibrated with covariance flux towers	Nagler et al. 2005b
Tamarix spp.	Chibola National Wildlife Refuge, AZ	2.7–3.4 m DGW	1,100 mm yr^{-1} (3.0 mm day^{-1})	MODIS vegetation indices	Nagler et al. 2008
T. ramosissima	Chibola National Wildlife Refuge, AZ	<100 mm yr^{-1} 2.5–3.3 m DGW	3.7–9.5 mm day^{-1}	Granier sap flow	Nagler et al. 2009

Obligate phreatophytes, such as many species from the *Populus* genus, generally extend their roots downward into the water table to support high transpiration rates and depend on access to groundwater for growth and survival. These species are generally pioneers that maximize growth and transpiration rates at the expense of drought and shade tolerance. For example, Fremont cottonwood (*Populus fremontii*) transpiration rates were higher in individuals growing along perennial streams where roots had access to the permanent water table than in those growing along intermittent streams (1.0–5.7 vs. 1.3–3.1 mm day^{-1}, respectively; Schaeffer et al. 2000; Gazal et al. 2006). Stand age also affects the amount of water consumed (Willms et al. 2006), which usually peaks after a stand has gone through self-thinning and competition has removed weaker trees from the stands.

Ewers et al. (2005) examined water use of upland trembling aspen (*Populus tremuloides*) along a chronosequence after fire disturbance. Aspen dominated the overstory at 12 to 37 years after fire. Over the chronosequence, growing-season canopy transpiration in this species rose from 8 mm observed in 12- and 20-year-old stands, to 37 mm in 37-year-old stands. This trend was explained by the increase in the ratio of sapwood area supporting a unit of leaf area with height, suggesting that taller, more dominant trees can support higher transpiration rates than shorter, generally suppressed trees. As the pioneer *Populus* trees matured, individual trees declined in health, growth, and transpiration rates as more shade-tolerant species like black spruce (*Picea mariana*) assumed complete dominance of the stands at age 70.

The ability of the riparian vegetation to resist environmental stresses like drought, the most common limiting factor in these systems, depends in large part on the species and ecotype. Species such as eastern cottonwood (*P. deltoides*) of the *Aigeiros* section of the *Populus* genus are faster growing and less drought-tolerant than species such as black cottonwood (*P. trichocarpa*) of the *Tacamahaca* section of *Populus* (Dickmann 1979). Anatomically, *Aigeiros* cottonwood species, including *P. fremontii*, *P. angustifolia*, *P. acuminate*, and *P. deltoides,* tend to have larger earlywood vessels than the *Tacamahaca* aspen species (e.g., *P. tremuloides*; Tennessen et al. 2002). While these larger vessels can support faster transpiration rates, they are more vulnerable to lose water-conducting capacity under drought due to xylem cavitation that occurs when air breaks the vertical columns of water, thus reducing the number of xylem elements that carry water up the stem (Hacke et al. 2006). Resisting xylem cavitation and maintaining hydraulic conductivity, therefore, become

crucial for species survival under drought stress in riparian areas. Several of the *Aigerus* section *Populus* species respond to water stress first by xylem cavitation in twigs and branches, next by early leaf senescence, and then by branch dieback (Rood et al. 2000). Xylem air-entry point, the start of cavitation, in black cottonwood was reported to be at water potentials of -0.71 to -1.32 MPa (a measure of water status in leaves) in trees from wet sites and at -1.55 to -1.67 in trees from dry sites (Sparks and Black 1999). Cochard et al. (2007), in an evaluation of five interspecific poplar clones, confirmed both soil water requirements and cavitation thresholds (-1.5 MPa) and low drought resistance in these species. Recovery from drought depends on the severity and length of water stress (Amlin and Rood 2003). If water stress leads to death of twigs and branches, then cottonwood photosynthesis and evapotranspiration on a tree- and stand-level basis will decline, ultimately leading to cottonwood mortality. Coupled with the fact that cottonwood cannot regenerate under shade (Farmer and Bonner 1967), increased cottonwood mortality could speed up successional replacement of cottonwood by more drought- and shade-tolerant species, such as eastern redcedar, Russian olive, and saltcedar, unless flooding occurs at sufficient frequency (Johnson 1992; Lytle and Merritt 2004).

Eastern redcedar (*Juniperus virginiana*), an evergreen species, has the capability to grow under high (Holthuizjen and Sharik 1985) as well as low levels of irradiance (maintain positive carbon gain at around 5% of full light; Lassoie et al. 1983), photosynthesize under a wide range of temperatures (0°–40°C), and tolerate drought conditions (Eggemeyer et al. 2006, 2009; Bihmidine et al. 2010). Although xylem cavitation in eastern redcedar is reported to begin at water potentials below -4 MPa, the species is able to maintain 50% xylem conductivity at -5.8 MPa, and 10% conductivity at water potentials as low as -9 MPa (Sperry and Tyree 1990; Wilson et al. 2008). These characteristics, together with its long growing season and ability to extract water from both the saturated and unsaturated soil profiles, make eastern redcedar a significant competitor and component in many riparian and upland communities in the Great Plains (Ormsbee et al. 1976; Eggemeyer et al. 2006, 2009; Bihmidine et al. 2010). Data on tree-level water consumption in eastern redcedar are few. At a site located by the Platte River near Odessa, NE, one mature 22.9-cm-diameter eastern redcedar tree was reported to use an average of 62.3 L day^{-1} in April and 48.4 L day^{-1} during a five-month period between May and August (Landon et al. 2009). Our preliminary data indicate that water use of eastern redcedar trees ranging

in size from 4 cm to 28 cm in diameter averaged 5 to 50 L day^{-1} at two sites located along the Republican River, Nebraska, depending on tree size in late summer.

Saltcedar (*Tamarix ramosissima*) is the most commonly occurring *Tamarix* species out of the 8 to 10 species that have been introduced in North America (Smeins 2003). Like eastern redcedar, drought-tolerant saltcedar (facultative phreatophyte; Busch et al. 1992; Glenn and Nagler 2005) and Russian olive can significantly alter the water balance in riparian areas by consuming water from both the unsaturated and saturated soil profiles (Truman 1996; Zhao et al. 2007). Researchers have observed Russian olive displacing native woody species in droughty and shady environments in the seedling stage (Shafroth et al. 1995). Saltcedar trees during the record 2006 drought in Kansas adapted by shifting from using water from the unsaturated soil profile when water was available to using groundwater during seasonal drought (Nippert et al. 2010). How well saltcedars avoid cavitation depends on site conditions and ecotype. At the Cienga Creek Natural Preserve in Arizona, where annual precipitation averages 310 mm, cavitation of xylem water conducting elements of *Tamarix ramosissima* did not occur until water potential dropped below -7 to -8 MPa (Pockman and Sperry 2000). In contrast, the same species growing at sites located adjacent to rivers or in a seasonal swamp in five states throughout the United States (Idaho, Washington, Florida, Georgia, and South Carolina) lost 75% of their xylem or water-conducting capacity at -0.86 MPa (Pratt and Black 2006). Saltcedar transpiration rates were also found to reflect site conditions, ranging from 13.4 mm day^{-1} on relatively dry sites (depth to groundwater [DGW] 2–3 m), to 23.5 mm day^{-1} in wet areas (DGW 0.5–1 m), along the lower Virgin River, Nevada (Sala et al. 1996). Higher transpiration rates in wet areas corresponded with the ability of saltcedar to (1) extract water from the various zones in the soil profile; and (2) regulate water use during the growing season by adjusting leaf production (Sala et al. 1996; Cleverly et al. 2002; Dahm et al. 2002). Therefore, control of saltcedar in areas that are flooded or have a shallow water table has greater potential of increasing water yields than control efforts in drier areas. Saltcedar's tolerance to a number of disturbances such as salinity, fire, flooding, and herbivory, and its ability to resprout after topkill, have facilitated its spread throughout the western United States and have made it difficult to control (Smeins 2003).

Although few studies have examined the ecophysiology of Russian olive (*Elaeagnus angustifolia*), work conducted in the Canyon de Chelly National Monument by Reynolds and Cooper (2010) indicates that its ability to survive low light and droughty conditions as a seedling may enable it to increase its range. In a survey of stands in the region, Russian olive seedlings were found at light levels of 1,233 ± 46.3 μmol m^{-2} s^{-1} under a mean canopy cover of 55.6% ± 18.4% up to 8 m above the stream channel. In experiments monitoring the growth and survival of seedlings, about 75% of the Russian olive seedlings were able to survive under 99% shade when they had access to a shallow water table and some (10%–35%) were able to survive when subjected to water stress. In contrast, only 10% of the cottonwood (*Populus fremontii*) seedlings survived under 99% shade when they had access to shallow water, and none of the tamarisk (*Tamarix ramosissima*) seedlings survived under 99% shade. Analysis of oxygen isotopes indicated that Russian olives less than 15 years old utilized water from a shallow soil (10–30 cm depth) compared with trees 15 years or older, which utilized both shallow and deeper (40–70 cm depth) water. Thus, Russian olive can spread in densely shaded stands, replacing cottonwood and later successional native species in the absence of disturbances.

STAND–LEVEL WATER USE

Evapotranspiration accounts for much of the water lost from riparian zones in semiarid ecosystems, and precise estimates of riparian evapotranspiration are essential to accurately allocate river water for environmental and human needs (Nagler et al. 2005a, 2005b). Estimates of evapotranspirational losses suggest that 20%–50% of water depletion can be attributed to riparian vegetation in semiarid systems (Dahm et al. 2002). In Nebraska, if a phreatophyte cover of 1,289 km^2 is assumed (0.64% of the total state land area), Szilagyi et al. (2005) calculated, using water balance equations and an automated base flow separation, a mean groundwater phreatophyte evapotranspiration of 887 mm yr^{-1}. Estimates increase significantly as woody species density increases and invasive species spread. For example, evapotranspiration rates were 20% higher in cottonwood stands with saltcedar and Russian olive understory (1,230 mm yr^{-1}), and in saltcedar stands (1,110 to 1,220 mm yr^{-1}), compared to closed canopy cottonwood stands with an understory of coyote willow (*Salix exigua*), seep willow (*Baccharis glutinosa*), and false indigo bush (*Amorpha fruticosa*) (980 mm yr^{-1}) in New Mexico (Dahm et al. 2002). Likewise, a study in the Pecos River of New Mexico found that removing saltcedar and Russian olive from the understory of *Populus* stands reduced groundwater fluctuations by 6.7% and 18.1%, respectively (Martinet et al. 2009).

Within the *Populus* genus, riparian cottonwood trees of the warm arid southwestern region of the United States were found to transpire more water than upland poplar trees of the cooler northern regions and in Canada. The reported riparian stand-level transpiration rates are 0.7–3.4 mm day^{-1} in *P. deltoides* (New Mexico and Texas, *Aegeros* section) and 1.0–9.3 mm day^{-1} in *P. fremontii* (Arizona and California, *Aegeros* section) (Table 1). In comparison, in Canadian upland forest sites *P. tremuloides* stands transpired 0.3–1.2 mm day^{-1}. When contrasting stand-level water use of poplar with those reported for *Juniperus* sp., Table 1 shows that, generally, rates reported in the literature were lower for *Juniperus* than those for *Populus* species, ranging from 0.23–1.21 mm day^{-1} in Utah juniper (*J. osteosperma*, Arizona and New Mexico) and alligator juniper (*J. deppeana*, Arizona), to 1.90 mm day^{-1} in ash juniper (*J. ashei*, Texas; Table 1). Although, these comparatively low transpiration rates reflect the *Juniperus* conservative water use, which allow for survival in dry areas, the evergreen nature and longer growing season of *Juniperus* may offset the relatively low water daily uptake.

ECOHYDROLOGICAL RESPONSES TO WOODY PLANT REMOVAL

Reducing woody plant cover, particularly within an entire watershed, has been hypothesized to increase water yields in riparian ecosystems by reducing transpirational demands. Studies examining this hypothesis have been conducted by removing trees over an entire watershed or catchment area rather than within selected reaches. Gains in water yields, reported in the literature, have been found to vary greatly. In a review of 39 catchments, Hibbert (1967) found that, although the reduction of forest cover generally increased water yield and the establishment of forest cover on sparsely vegetated land decreased water yield, responses to removing woody vegetation were highly variable and unpredictable. In another study that examined the average streamflow water yield rather than the maximum increase in yield, Sahin and Hall (1996) found that a small (10%) reduction in conifers and deciduous hardwoods significantly increased annual water yield by an average of 22.5 and 18 mm, respectively, while reduction in scrub vegetation increased yield by only 5 mm. Variations in water yield response to thinning and cutting forest vegetation can be largely explained from differences in the interannual variability in water input, in vegetation regrowth, between riparian and stream types, and among geographic and climatic regions (Zhang et al. 2001; Sun et al. 2005; Adams and Fowler 2006).

As precipitation increases, stream yield responses to tree harvesting increases until a maximum response is reached (Zhang et al. 2001; Wilcox et al. 2005). Analysis of published studies indicate that annual stream-water yield following tree removal in areas receiving 450 to 600 mm of precipitation showed either no response (i.e., 0 increase), or increased up to 90 mm after watershed-wide clear-cutting (mostly in areas with higher precipitations), a variation attributed to regional climatic conditions and topographic features (Table 2; Sun et al. 2005; Adams and Fowler 2006). In the southeastern United States, Sun et al. (2005) modeled potential water yields based on regional variability in climate and topography and found that in general, mountainous areas with cool summer temperatures and high rainfall showed the most increase in water yields in response to vegetation removal, while coastal wetland areas with moderate rainfall and high potential evapotranspiration exhibited the least response. In studies where clear-cutting was conducted on a portion of the watershed, the increased water yield reflected the portion of land where forest cover had been removed. Bosch and Hewlett (1982) in a review of 94 catchments found that water yield response following plant removal depended on reduction levels and type of vegetation. Water yield increases were not detectable following forest cover reductions of less than 20% of the catchment (similar observations were reported by Stednick 1996), while maximum streamflow water yield increased by 400 mm yr^{-1} when coniferous trees were completely removed from the watershed, compared to an increase of 250 and 10 mm yr^{-1} following the complete removal of deciduous hardwood trees and grasses, respectively, from the watershed.

In general, forest treatments conducted in riparian forests where trees can use groundwater affect stream yields more than those conducted in upland forests where trees cannot access groundwater. Nagler et al. (2010) reported that of the 637,000 km^2 Colorado River basin, the 180 km^2 area covered by saltcedar monocultures (0.03% of the total basin area) growing along the Lower Colorado River consumes between 0.75 and 1.45 m yr^{-1} of water (1 m yr^{-1} average) and accounts for a loss of 1% of all river flow. Wilcox (2002) reviewed experiments examining water yields response to removing honey mesquite (*Prosopis glandulosa*) and juniper (*J. ashei* and *J. pinchotii*) from nonriparian rangelands in Texas. He concluded that mesquite control in upland regions is unlikely to significantly increase water yields because (1) the typical herbaceous regrowth and tree sprouting on sites utilized most of

TABLE 2

PREDICTED CHANGES IN ANNUAL WATER YIELD AFTER REMOVAL OF CONIFER OR HARDWOOD TREES IN A WATERSHED AS A FUNCTION OF AVERAGE ANNUAL PRECIPITATION OBSERVED IN THE GREAT PLAINS BASED ON WATERSHED-LEVEL MODELS

| | --------------- Change in water yield (mm) --------------- | | | |
| | Adams and Fowler (2006) regressions on conifer removal | | Brown et al. (2005) | |
Average annual pre-cipitation (mm)	Bosch and Hewlett (1982) data	New Zealand	Conifers	Hardwoods
300	128	88	100	100
400	151	115	140	130
500	174	143	175	160
600	197	170	200	190
700	220	197	240	220
800	243	225	280	250
900	266	252	340	275
1,000	289	280	360	300
1,200	335	334	400	350

Note: Adapted from Brown et al. (2005) and Adams and Fowler (2006).

the available soil moisture; (2) deep soils isolated the groundwater from the surface; (3) much of the increased soil water flows horizontally over the land when soils become saturated; and (4) flood-producing rainfall events produced massive runoff. Conversely, Wilcox (2002) suggested juniper control can be used to effectively increase streamflow.

The majority of paired catchment studies reviewed above and others listed in the literature (Table 2) are based on water yield changes within one to five years of vegetation removal (Brown et al. 2005). Ffolliott et al. (2003) studied dryland oak transpiration in the southwestern United States and found that mature trees in unharvested upland stands transpired about 45% of the annual regional precipitation, leaving 55% of the rain available for groundwater recharge, streamflow input, and evaporation. However, mature trees and numerous stump sprouts in harvested stands transpired 80% of the annual precipitation, leaving only 20% available for site water use. These results highlight the impact of harvest events on the water budget in semiarid regions. Consideration thus should be given to postharvest vegetation response when managing riparian forests for water use (Ffolliott et al. 2003). Several studies have demonstrated that sustained increases in water yield depend not only on vegetation response but also on site management following initial vegetation removal. Hornbeck et al. (1993) in a study of the long-term impacts of vegetation change

found that increases in water yield immediately after tree removal could only be sustained if regrowth was controlled. Similarly, studies on deforestation conducted by Ruprecht and Schofield (1989) and Silberstein et al. (2003) demonstrated increased water yields in cleared catchments in the first year after treatment, with steady yield decreases observed thereafter until a new vegetation equilibrium was established. Brown et al. (2005) generalized that transpiration rates, vegetation age, and soil storage changes to treated systems required at least five years for establishing an equilibrium following catchment alteration.

Inconsistencies in models predicting water yields as a function of vegetation type and cover caused by precipitation variations could be significantly reduced by including climatic variables, such as annual precipitation in the year of maximum change in annual yield, as explanatory variables (Brown et al. 2005; Adams and Fowler 2006). If these results are mapped out according to precipitation isocline across the Great Plains, potential water savings from clearing areas entirely covered with deciduous woody vegetation from the Platte River would range between 130 mm yr^{-1} in the west to 250 mm yr^{-1} in the east, and potential water savings from clearing areas with a 100% cover of eastern redcedar from the Platte River would range from 140 mm yr^{-1} in the west to 280 mm yr^{-1} in the east (Fig. 1). Wilcox et al. (2006) urge caution when projecting results from small catchments to larger scales.
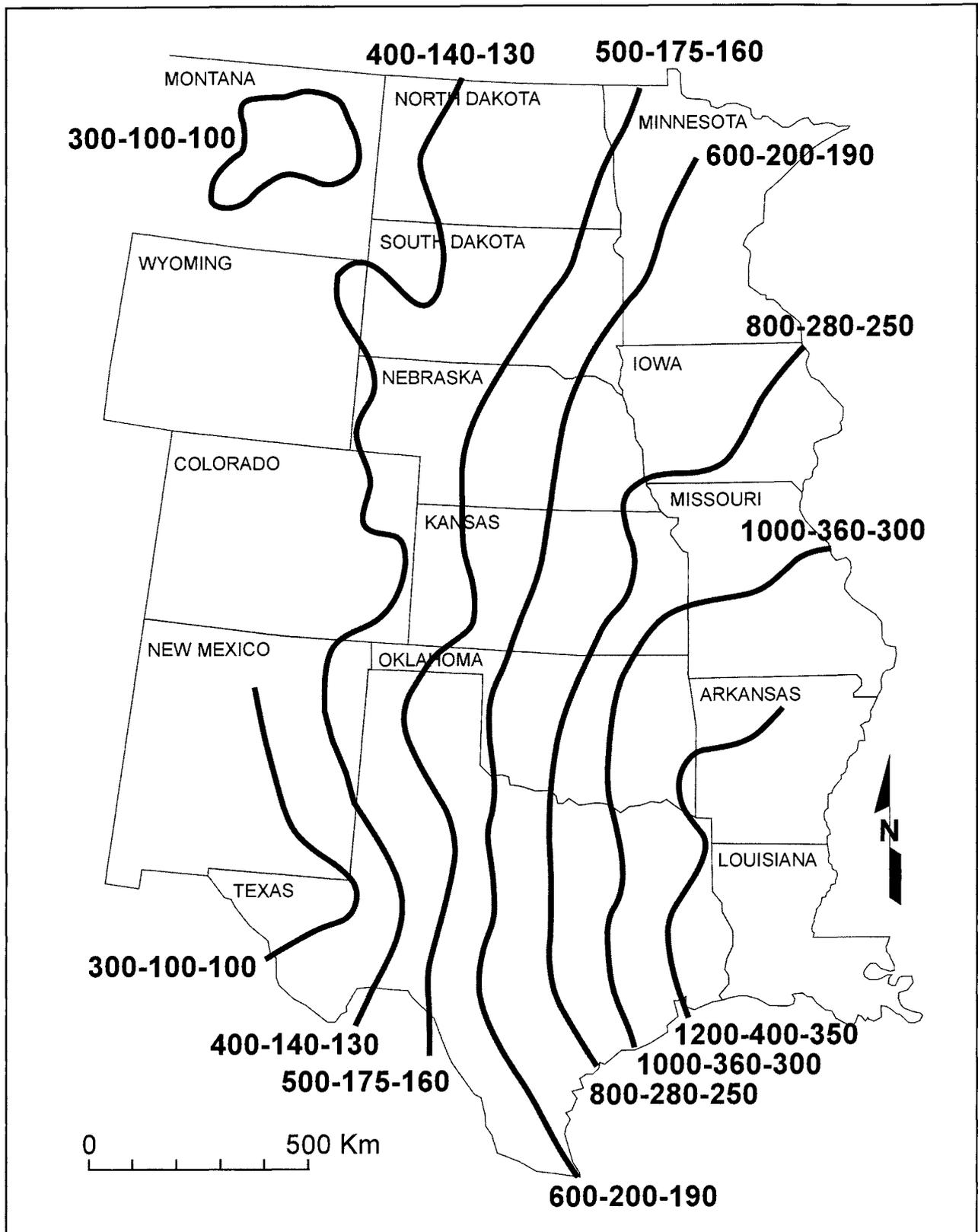
Figure 1. Precipitation isoclines in the Great Plains and adjacent areas. Numbers from left to right: the isoclines of the Great Plains according to average annual precipitation (mm, modified from Schimel et al. 1990), the predicted increase in water yields after the complete (100%) removal of conifer trees (mm yr^{-1}), and the predicted increase in water yield after the complete (100%) removal of deciduous trees (mm yr^{-1}, after Brown et al. 2005).

Models that estimate the evapotranspiration rates based on plant functional type (e.g., obligate wetland, shallow-rooted riparian, deep-rooted riparian, transitional riparian, and upland) and water table depth can potentially integrate physiological measurements across larger scales (Baird and Maddock 2005).

On a landscape scale, occurrence of trees within grasslands or agricultural systems can reduce evapotranspiration. Shading and mulch from trees reduce evaporative losses from soil (Stormont et al. 2009). On farm systems windbreaks, riparian forest buffers and alley-cropping systems (growing crops between rows of trees) can be used to reduce evapotranspiration by agricultural systems by reducing evaporative losses caused by wind (Brandle et al. 2003). An optimum tree density and distribution likely exists where the reductions in understory water losses by shade and wind reduction are greater than water losses from trees. Promotion by land managers of more efficient water use through improved management coupled with sound water-trading principles will help producers achieve economic goals and help society attain ecological interests such as greater water yields and streamflows (Clayton 2009).

ECONOMIC COSTS AND BENEFITS OF ECOHYDROLOGICAL MANAGEMENT

Removing woody vegetation from rangelands and forested areas has been proposed as a way to increase water yields from land in the western United States (Hibbert 1983; Wilcox and Thurow 2006). Managing riparian forests to balance social and economic needs and ecosystem capacity poses important challenges to managers and policy makers. Attaching an economic value to various services and goods provided by ecosystems can provide a benchmark principle for managing resources (Pearce and Smith 2001). For example, the direct role of forests in regulating water quantity and quality around the world provides an important argument for sustainable forest management and protection (Dudley and Stolton 2003). Whether or not forested riparian areas should be harvested must be assessed using both ecological and economical criteria.

In heavily timbered regions where harvests are conducted for income, streamside management zones (SMZs) are commonly written into management plans to protect riparian ecosystem functions. Trees are harvested only outside of these SMZs to provide economic benefits. In a literature review on the effects of timber harvest on U.S. eastern hardwood streams, LeDoux and Wilkerson

(2006) found that 45 m wide was enough for streamside management zones to provide all of the five ecological functions they assessed (woody debris, shade, sediment filtering, aquatic communities, riparian bird communities). Streamside management zones that were only 30 m wide provided 87% of these benefits as coarse woody debris and sediment filtering benefits from riparian forests declined. They calculated that streamside management zones cost landowners with yellow-poplar and mixed hardwood forests in the eastern United States between $30.54 and $67.02 ha^{-1} yr^{-1} ($12.36 to $27.12 per acre) depending on the stand type and logging technology used. The economic return of harvesting stands having about 300 m^3 ha^{-1} standing timber averaged $9,200 ± $940 ha^{-1} ($3,725 ± $380 per acre) in yellow-poplar stands and $11,500 ± $1,270 ha^{-1} ($4,650 ± $510 per acre) in mixed hardwood stands. In contrast to large expanses of eastern hardwood forests, Great Plains forests largely occur as linear gallery forests or as planted windbreaks (Barker and Whitman 1988). Biomass production of linear windbreak forests planted in Great Plains agro-ecosystems was calculated to be higher in narrow 10 m strips than in wider 30 m strips (Guo et al. 2004).

Riparian forests in the Great Plains are not generally managed for forest production. Here it is largely the ecological costs and benefits that must be weighed when considering whether or not to harvest or even plant trees within SMZs. Costs associated with Great Plains riparian forests include water lost through evapotranspiration, elevated sedimentation rates of dams and reservoirs, reduced forage for grazing, and losses in biodiversity (Zavaleta 2000). On the other hand, woody riparian buffers help stabilize eroding banks, filter dissolved pollutants, improve stream habitat for fish, provide forest habitat, provide flood protection, shelter livestock and yield economic products (Dosskey 1998).

Zavaleta (2000) estimated that increases in water losses, sedimentation rates, and in subsequent flooding damage caused by tamarisk costs $280–$450 ha^{-1} yr^{-1} in the western United States. Costs of removing this invader and restoring native vegetation could be recovered as soon as 17 years after removal efforts. Ecosystem adaptation to invading species like tamarisk complicates removal and restoration planning. Although many bird species have declined as tamarisk has spread, some species, such as the endangered southwestern willow flycatcher (*Empidonax tralii extimus*), now nest in tamarisk (Sogge et al. 2008). A threshold response of bird species to tamarisk at 40%–60% canopy cover appears to exist (Van Riper et al. 2008). Zavaleta et al. (2001) recommend

that removal of tamarisk and restoration of native habitat be done in a stepwise fashion to provide adequate habitat for woodland animal species like the southwestern willow flycatcher. In areas where woody species are invading grasslands, such as the Niobrara River valley, where eastern redcedar is invading, woodland bird species are replacing grassland bird species (Frost and Powell 2010). Restoring native grasslands following juniper removal would thus be essential to restoring the native fauna in these systems.

The cost of increased water yields depends on a number of factors including removal cost, increases in water yields, and economic direct and indirect returns from clearing. Differences in size and in increased water yields following clearing largely explained why water from juniper and mesquite brush removal from Texas rangelands cost $320 ha^{-1} in the Edwards Plateau and $820 ha^{-1} in the Twin Buttes watershed (Olenick et al. 2004). Although removing a low cover of juniper was less expensive than removing a low cover of mesquite, the reverse held true when woody cover was high (Table 3). Mechanical treatments were more expensive than chemical treatments.

The cost of tree removal is another crucial management consideration. In general, aerial removal of saltcedar is cheaper than ground methods (Table 3). A study comparing four control methods conducted between 1989 and 2001 at 20 sites located in seven states in the southwestern United States estimated the efficacy of saltcedar removal to be 89% ± 13% for helicopter herbicide application; 93% ± 10% for fixed-wing herbicide application; 78% ± 25% for cut-stump and herbicide application; and 88% ± 14% for foliar herbicide application (Sisneros 1994; U.S. Bureau of Reclamation 2009). Thus, aerial control methods are cheaper and more effective over large areas than labor-intensive ground methods. Including herbicide in control treatments helps ensure the long-term control of saltcedar (O'Meara et al. 2010). Follow-up control must be done to ensure restoration of native species in controlled areas. In the 23 states of the western United States, where tamarisk has invaded 470,000–650,000 ha of riparian zones, the total cost for eradicating tamarisk from riparian zones is estimated as high as $7,400 ha^{-1} (Zavaleta 2000). Cost of controlling juniper ranges from $5 ha^{-1} to $508 ha^{-1} depending on method and cover (Table 3).

In addition to the potentially improved water yield, wood removed during forest harvesting and thinning could be marketed and sold in emerging biomass energy and biofuels markets, as well as traditional forest markets. However, riparian forests support fundamental ecosystem services such as internal nutrient cycling, soil protection, biodiversity conservation, carbon sequestration, climatic regulation, and quality water supply. Tree removal, when not done in an environmentally sound manner, can harm these ecosystem services. Tabacchi et al. (2000) warn that large-scale logging and fragmentation of floodplain forests can lower the ecosystem's buffering capacity to reduce water input and delay backwater drainage, thus altering the exchange of surface and ground water. Early successional species such as cottonwood depend on periodic flooding and scouring events to provide the moist bare mineral soil needed for regeneration (Smith and Linnartz 1980). Treatments such as thinning can be used to remove competing vegetation and raise light levels, which allow cottonwood to establish. Cuttings are often used to quickly regenerate cottonwood in plantations and on open sites (Taylor 2001).

However, caution must be used in ecosystems where tree removal might stimulate the spread of nonindigenous species. Changes in ecosystem structure have been shown to increase encroachment by nonindigenous species in riparian zones (Planty-Tabacchi et al. 1996). These native and non-native invaders exhibit greater adaptability to physical disturbance in the form of tree removal, physical disturbance, and hydrological alterations than do endemic riparian species (Katz and Shafroth 2003). McIntyre and Lavorel (1994) report that riparian invaders demonstrate enhanced colonization and reproduction capabilities in disturbed habitats when compared with native pioneer species.

Clear-cutting riparian forests can degrade downstream habitat for fish and wildlife species. Channel narrowing has been documented along the braided Platte River due to upstream flow modifications following damming (Johnson 1994), but studies by Johnson (1997) report that over time these channel areas have stabilized. On the other hand, vegetation removed along a portion of the river to increase open channel area for migrating whooping (*Grus americana*) and Sandhill cranes (*Grus canadensis*) resulted in downstream channel area disequilibria. The vegetation clearing may have liberated excess sediment, which raised the channel and stimulated tree and shrub recruitment, causing a 10% channel loss (Johnson 1997). Channel area losses in unmanaged reaches may offset gains in managed areas, suggesting that management procedures should be reevaluated before further reaches are cleared (Johnson 1997). Jones et al. (1999) investigated the impact of riparian forest removal on downstream fish assemblages in southern Appalachian streams. The studies concluded that clearing

TABLE 3
WOODY VEGETATION TYPE AND LOCATION, TREATMENT, AND COST

Woody vegetation type and location	Treatment	Cost (US$/ha)		Reference
Juniperus, moderate cover (10%–30%)	Various clearing methods	365	± 87	Olenick et al. (2004)
Juniperus, heavy cover (>30%)	Various clearing methods	508	± 111	Olenick et al. (2004)
Juniperus virginiana open (<30-year-old stand) and dense stands (30-year-old, 5 m tall), Custer County, NE	Burned	5		Ortmann et al. (1998)
	Burned and cut	40		
	Individual trees burned	15		
	Picloram	90		
	Cutting	65		
Prosopis, moderate cover (10%–30%)	Various clearing methods	256	± 73	Olenick et al. (2004)
Prosopis, heavy cover (>30%)	Various clearing methods	438	± 228	Olenick et al. (2004)
Tamarix	Helicopter herbicide application	168		U.S. Bureau of Reclamation (2009); Sisneros (1994)
Tamarix	Fixed-wing herbicide application	138		
Tamarix	Cut-stump and herbicide application	2,617		
Tamarix	Foliar herbicide application	849		
Tamarix	Cut and sprayed with imazapyr	1,250	± 370	Taylor and McDaniel (1998)
Tamarix	Aerial spray of imazapyr with and without glyphosphate; burning	430	± 140	Taylor and McDaniel (1998)
Tamarix	Individual cut and spray imazapyr	3,952	± 6,175	Taylor and McDaniel (2004)
Tamarix	Individual herbicide application *or* mechanical grubbing	99	± 741	Taylor and McDaniel (2004)
Tamarix	Large-scale control methods	1,010	± 460	Taylor and McDaniel (2004)

vegetation over 1–3 km in width or reach was associated with the decreased abundance of benthic-dependent fish species, causing sediment-tolerant and often invasive species to supplant them. However, the study focused on deforested but still vegetated riparian zones in otherwise wooded landscapes, so the results may not apply to riparian buffers in other forested areas (Jones et al. 1999).

CONCLUSION

Despite their apparent small percentage of cover on a regional scale, riparian ecosystems play a major functional role in the hydrologic cycle in the Great Plains. Water crises and shortages in arid and semiarid areas of the United States and elsewhere due to climate change, drought, population growth, agriculture, and new demand for biofuels and other energy systems are driving

researchers and managers to find ways to reallocate water and increase water yields in streams and rivers. Because riparian forests use more water than upland forests on an area basis, approaches to increase water yields could include the removal of invasive and aggressive woody vegetation from riparian areas. Realistic projections of increased water yields following tree removal must consider many variables, including land use within the entire watershed, relative water use with each land-use category, climate, and pre- and post-treatment vegetation. As a first cut, relative areas and water use rates of different vegetation and land-use types can be used by managers and policy makers to identify regions where changes in management can have the most impact on water yields and streamflows. In the face of escalating political and socioeconomic pressures, it is critical to manage riparian habitats in an ecologically sustainable

manner in order to preserve their integrity, function, structure, productivity, and species composition. Otherwise, short-term gains in water yields after removing woody vegetation could be offset by long-term losses in ecosystem services, including water yields, as aggressive, more water-consumptive species spread in untreated and cleared areas. Targeted efforts to control woody plants in water-rich riparian zones can help minimize water lost by invasive facultative phreatophytes, such as saltcedar, Russian olive, and eastern redcedar, potentially increasing water yields. However, our review has shown that such increases are unpredictable in semiarid regions; they vary within geographic regions and stream types, and are most often temporary, with benefits leveling off after a few years unless continuous control of woody and weedy vegetation is adopted on these sites. More research on the ecologic, hydrologic, and economic consequences of the spread and management of invasive species in riparian areas is needed in the northern Great Plains. Quantifying these responses to invasive species is needed to plan effective large-scale invasive species control programs where efficient and sustainable use of water resources is a management priority.

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