RIPARIAN RESTORATION AND MANAGEMENT OF ARID AND SEMIARID WATERSHEDS

by
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DEDICATION

To my dad, Winston Scot Bunting, who passed away in June 2009.

Through you I live my life by the Golden Rule and always aspire to better myself so I can help others.
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ABSTRACT

Riparian ecosystems are valued for ecosystem services which have impacts on the well-being of humans and the environment. Anthropogenic disturbances along rivers in arid and semiarid regions have altered historical flow regimes and compromised their integrity. Many rivers are hydroecologically deteriorated, have diminished native riparian forests, and are pressured for their water supplies.

My first study is founded on the premise that river restoration has increased exponentially with little documentation on effectiveness. We designed a conference to discuss lessons learned from past restoration activities to benefit future efforts. Participants, who included scientists, managers, and practitioners, agreed that creating measurable objectives with subsequent monitoring is essential for quantifying success and employing adaptive management. Attendees stated that current projects are local and have limited funding and time, whereas future efforts must have longer funding cycles, larger timeframes, should contribute to regional goals, and address factors responsible for ecological decline. Bridging gaps among science, management, and policy in the 21st century is a key component to success.

My second study focused on the benefits of long-term monitoring of local riparian restoration. Many efforts include revegetation components to re-establish native cottonwood-willow communities, but do not address how high-density establishment impacts vegetation dynamics and sustainability. Over five years, we documented significantly higher growth rates, lower mortality, and higher cover in cottonwood
compared to non-native tamarisk. Cottonwood height, diameter at breast height, growth rates, and foliar volumes were reduced at higher densities. Herbaceous species decreased every year but native shrubs volunteered after two years resulting in a reduction of overall plant diversity from 2007-2009 with a slight increase from 2009-2011.

My third study focused on improving basin-scale evapotranspiration (ET), a large component of the water budget, to better inform water resource allocation. My research suggests that multiple models are required for basin-scale ET estimates due to vegetation variability across water-limitation gradients. We created two empirical models using remote sensing, a multiplicative riparian ET model ($r^2=0.92$) using MODIS nighttime land surface temperature ($LST_n$) and enhanced vegetation index, and an upland ET model ($r^2=0.77$) using multiple linear regression replacing $LST_n$ with a precipitation input.
1. INTRODUCTION

1.1 Background

Riparian ecosystems provide ecosystem services (e.g., cycling soil and nutrients, replenishing aquifers, treating pollutants, providing water supply, providing recreation opportunities, etc.) worldwide and support human water demands for agriculture, municipal use, and industry (Malanson, 1995). Ecosystem services are also extremely important in arid and semiarid regions where they support (e.g., via refuge, habitat, migratory corridors) the majority of the region’s wildlife species and maintain high biodiversity and ecosystem productivity (Knopf et al. 1988; Malanson, 1995). Riparian ecosystems in Arizona for example, make up less than 0.5% of the total land surface area, yet up to 75% of all resident wildlife depends on them for at least part of their lifespan (Arizona Riparian Council 1995). Often referred to as the “kidneys of the landscape,” wetlands and other riparian ecosystems play an integral role in maintaining ecological function and will continue to be valued by humans and wildlife whose well-being depends on them.

Western riparian ecosystems in the United States (US) have undergone significant changes in the past century to prevent flooding, supply energy, satisfy agricultural demands, and supply water to growing desert metropolitan areas (Rood and Mahoney 1990, 1991, Nilsson and Svedmark, 2002). River impoundment, land-use change, surface and groundwater extraction, and artificial inter- and intra-basin transfers profoundly alter natural flow regimes (Poff et al. 1997, Naiman et al. 2002, Postel and
Richter 2003, Revenga et al. 2005, Pearce 2007). These anthropogenic disturbances have compromised the integrity of many western US river systems (Poff et al. 1997; Ward & Stanford 1995), meanwhile urban population growth in the Southwest continues to increase human consumptive water use as well as agricultural, commercial, and industrial demands (Gollehon and Quinby 2006). Climate change predictions, which include higher temperatures (IPCC 2007), less precipitation in arid and semiarid regions (IPCC 2007, Hall et al. 2008), and decreased snow accumulation in the Rocky Mountains (IPCC 2007, Pederson et al. 2011) exacerbate concerns surrounding future water resources and allocations.

Rising concerns over the deterioration of riparian ecosystems and subsequent impacts on ecosystems services, wildlife habitat, and water resources have stimulated increased efforts to restore rivers across the US. In fact, restoration efforts have increased exponentially since the 1980s and thousands of restoration projects have been implemented since the turn of the 21st century (National River Restoration Science Synthesis 2006). However, concerns regarding project efficacy and foci are arising when considering the magnitude of projects and associated costs with little documentation on overall effectiveness (Follstad Shah et al. 2007, Palmer et al. 2007). Restoration efforts have been typically directed toward site-specific projects within small areas or with a single disciplinary focus narrowly defined and confined to a specific species that may be threatened-endangered or a species with sport or commercial interest (Hauer and Lorang 2002). In the future, efforts must be made in a basin-wide perspective because planning
based on isolated components (fish, vegetation, restoration of specific stream sections) is ecologically incomplete and may lead to unintended consequences (Naiman et al. 1993).

Costs associated with restoration and enhancement of streams and rivers within the continental US has been estimated at $14 to $15 billion since 1990, and is now over $1 billion per year (Bernhardt et al. 2005). Although river restoration projects have increased exponentially in the last 25 years, most have failed to report successes and failures due to the lack of a sound monitoring program (Bernhardt et al. 2005). Smaller projects often fail to include post-treatment evaluations during the planning and design phases (Palmer et al. 2007), thus valuable information regarding success and failures is lost and not disseminated to the broader restoration community. Often overlooked and underfunded, ecological monitoring is a valuable component of restoration that can provide the data needed to alter management in a manner that will improve effectiveness (Bernhardt and Palmer 2011, Briggs 2011). River restoration efforts implemented in the southwestern US follow the national trends elucidated by Bernhardt et al. (2005) and Follstad Shah et al. (2007), including increased project numbers and high costs with little investment toward reporting results. While the Southwest generally has a higher percentage of projects with monitoring components compared to nation-wide statistics, Bernhardt et al. (2005) also found that most projects with monitoring protocols do not have appropriate information for quantifying success nor do they disseminate results.

River restoration needs to be redefined in the 21st century. Current water management practices may no longer be appropriate for the unpredictable flow regimes of a warmer and more densely populated world (Vörösmarty et al. 2004, Alcamo et al. 2005).
Owing to the fact that it is currently impossible to use existing databases to determine whether desired environmental benefits of river restoration are being achieved (Follstad Shah et al., Palmer and Allan 2006), it is essential that the river restoration community makes a concerted effort to change restoration directives. Future restoration will face graver challenges and have different goals and objectives than past efforts (Palmer et al. 2007). Independent of scale, future river restoration planning and implementation could be improved if practitioners would report results from current management practices, regardless of whether they were effective or not. Opening communication between river scientists, managers, practitioners and policy-makers could foster collaboration and partnerships required to meet broad-scale restoration goals in the future. The US needs regulatory and legislative federal policy reforms in order to improve the effectiveness of river restoration and thus the health of the nation’s waterways (Palmer and Allan 2006). Unless political and institutional cooperation can be achieved, the rich array of biologically diverse resources and ecological services associated with riparian corridors will continue to be exploited or wasted (Naiman et al. 1993).

Native riparian forests are perhaps the most affected by the physical degradation of western US rivers. Native riparian forests in the southwestern US consist primarily of cottonwood (*Populus* spp.) and willow (*Salix* spp.) communities that support wildlife, biodiversity, watershed protection, and recreation in an otherwise unproductive arid region (Fenner et al. 1985, Szaro 1989, Patten 1998). They are considered among the most threatened forest types in the US (Swift 1984, Busch and Smith 1995, Shafroth et al.)
2005) due to severe degradation resulting from altered riparian ecosystem hydrology (Rood and Mahoney 1990, 1991, Stromberg 1993, Lite and Stromberg 2005, Hultine et al. 2010). Dam regulation and water diversions alter flow regimes and often result in decreased soil moisture and increased soil salinization (Busch and Smith 1995, Sala and Smith 1996, Glenn et al. 1998, Nagler et al. 2008), channel narrowing and incision (Shafroth et al. 2002), floodplain disconnection, and non-native species invasions that facilitate higher fire frequency (Busch 1995). Collectively, these impacts diminish the ability for native species to recruit and establish large stands (Howe and Knopf 1991, Johnson 1992, Merrit and Poff 2010, Mortensen and Weisberg 2010). Re-establishing cottonwood-willow communities has become a major component of river restoration in the Southwest. Although revegetation alone cannot return full hydroecological function to southwestern rivers, many efforts will continue to include management that focuses on habitat creation for avian and other terrestrial communities that are considered sensitive, threatened, or endangered species under federal regulations.

While small-scale projects provide limited hydroecological benefit and typically cannot address the main stressors causing hydroecological deterioration, they can be completed quickly, are less costly than large-scale projects, provide opportunities for learning how to improve large-scale success, and can be visited by funders and stakeholders to garner support for larger-scale efforts (Briggs 2011). Small-scale or pilot projects, when implemented appropriately, can become important contributions to broad-scale restoration programs and goals. Local scale projects can demonstrate success often necessary to engage public support and stakeholder buy-in, which is important to the
long-term viability of restoration activities (Briggs 2011). Pilot projects also provide baseline research and experience for restoration practitioners to determine the feasibility for implementing restoration strategies. For instance, as part of the Multi-Species Conservation Program (MSCP), the US Bureau of Reclamation is conducting baseline research and implementing demonstration projects on the Cibola National Wildlife Refuge to determine the viability and cost-effectiveness of various management practices (USBR 2011). The revegetation component of the MSCP, in particular, aims to create and maintain 2400 ha of cottonwood-willow habitat. Due to ability to purchase agricultural lands and transfer associated water rights, many reclaimed agricultural areas will be used for revegetation practices. Concerns have been raised regarding whether reclaimed agricultural fields are even suitable for long-term restoration of native riparian habitat (see Aaron 2001). The overall effectiveness of each method applied on agricultural lands will require long-term evaluations on ecological factors (e.g., vegetation dynamics and sustainability, quality of habitat created, ability to support targeted wildlife species) as well as hydrological factors (Nelson and Anderson 1999, Stromberg et al. 2007). Other research suggests that long-term vegetation success can only be quantified with short-term data that evaluates biotic integrity, hydrologic function, and soil and site stability (Harrick et al. 2006).

Regardless of the river restoration tactic employed, restoration activities and management have important implications on water resources in the desert Southwest (Lite and Stromberg 2005, Nagler et al. 2008). Restoration often involves planting native trees that require large amounts of water during the establishment phase and, in some
cases, the life of the project (Hartwell et al. 2010). Revegetation for instance, often involves establishing native hydoriparian phreatophytes, which are capable of returning large amounts of water to the atmosphere via transpiration (Nagler et al. 2008). This high consumptive water use by native vegetation has direct implications on water resources management, especially considering that water lost by riparian vegetation cannot be used for human uses. Basin-wide scales are important from a management perspective because accurate evapotranspiration (ET) estimates are required to estimate how much regional water can be safely allocated to human needs while supporting environmental needs (Commission for Environmental Cooperation 1999, Congalton et al. 1998, Hansen and Gorbach 1997, U.S. Department of Interior 2002, Leenhouts et al. 2006). River restoration and agricultural practices alike can affect regional water budgets because consumptive water use may differ spatially and temporally. Scientists need to develop tools and models to inform and facilitate ecologically sustainable water management, thereby balancing human and ecological demand for freshwater in complex, dynamic and changing social and political environments (Arthington et al. 2010). The ability of researchers to use novel strategies to estimate ET at large scales will benefit managers who aim to sustain water resources for humans and the environment.

1.2 Research Objectives

The objective of this research is to contribute to river restoration and watershed management in arid and semiarid regions. River restoration in the 21st century faces more challenges than ever before and successful recovery of degraded riparian
ecosystems will be determined by realistic strategies that incorporate science and socio-political needs. It is now more important than ever to begin approaching river restoration in a broad context to understand the factors responsible for hydroecological decline. Managers should embrace localized projects for their ability to provide baseline research, but at the same time, define how these projects that provide limited hydroecological benefit contribute to broad-scale restoration. In this context, managers can benefit from highlighting successes and failures of small-scale projects, which provide invaluable experience for understanding the potential and improving the efficacy of future large-scale efforts aimed at meeting broad-scale goals. Regulated rivers may not be able to achieve true restoration (i.e., returning a river back to its wild state, or pre-impact levels) which often aims at restoring hydroecological processes. However, if steps toward ecological recovery can be achieved in a sustainable manner supported by science, then it may be worth exploring novel approaches for restoring regulated rivers to benefit biotic communities. Implementing monitoring protocols in future restoration efforts and disseminating results is perhaps the only way to quantify success and employ adaptive management strategies. Because water resources are already over-allocated and will continue to be pressured by human populations, environmental needs, and climate change, the ability to understand and quantify the amount of water entering and leaving riparian landscapes is essential. Models are necessary to estimate ET and compute water budgets at large scales so resource managers can make sound water allocation decisions, especially considering the impacts that land-use changes (e.g., agriculture, restoration, or development) may have on future water resources.
This research aims to answer three broad questions:

1) What are the most important lessons learned from past restoration activities that can be used effectively to improve restoration in the 21st century considering such inherent challenges as securing water supply, understanding the socio-political environment, and meeting broad-scale goals in the face of climate change?

In order to address this question, I started by helping to organize a conference entitled “Restoring Rivers in the Southwestern U.S. and Northern Mexico: A Bi-national Conference on Learning from the Past for the Benefit of the Future,” which convened in Tucson, Arizona on December 10, 2010. Headed by the World Wildlife Fund, this conference brought together river scientists, managers, practitioners, private citizens and policy-makers from federal and state agencies, academic institutions, non-governmental organizations (NGOs), and tribal nations in the southwestern US, northern Mexico, and southeastern Australia to discuss lessons learned from their river restoration experiences. Specific focus was given to dryland rivers and the biotic communities they support in semi-arid and arid climates. The necessity for such a conference stemmed from the continuing ecological deterioration of rivers; increased understanding of the impacts of climate change; and the need to share information, foster collaboration, and document what has been learned to benefit natural resource practitioners. The inherent challenges faced by all river restoration efforts, coupled with reduced resources for their implementation, makes it increasingly critical to implement efforts that will be
successful, which is contingent upon learning from past restoration efforts, both effective and ineffective, to improve future success. I synthesized and summarized key points highlighted at the conference and submitted the results as a “Set-backs and Surprises” article through the journal of Restoration Ecology entitled “Restoring Rivers in the Semiarid and Arid Southwestern United States and Northern Mexico: Lessons Learned from the Past to Benefit Future Riparian Restoration.” Goals were to disseminate valuable information to the scientific community in a way that emphasized the importance for: 1) improving efficacy of future restoration projects by learning from past experiences; 2) valuing ecological monitoring that quantifies objectives and is shared among practitioners; 3) designing local restoration projects around broad-scale programs and regional goals; 4) highlighting the need for political and institutional will that results in longer funding cycles, longer restoration timeframes, and policy reform to achieve regional goals. This effort also aimed to foster input and feedback for an applied guidebook tentatively titled, “Planning and Implementing River Restoration in the Southwestern United States and Northern Mexico, An Applied Guidebook for Restoration Practitioners” that will be drafted by the World Wildlife Fund.

2) What are the impacts on vegetation diversity and structural dynamics at river restoration sites that aim to re-establish native vegetation at high densities on reclaimed agricultural lands?
I addressed the second question through a long-term study at a local restoration site (i.e., Cibola National Wildlife Refuge) established using native seed collection and distribution techniques. This project is unique in that data has been collected extensively over a five-year period in a field setting where limited maintenance and minimal disturbance has allowed species to naturally compete for resources. This study, implemented at a retired agricultural field on the historical Colorado River floodplain, also contributed baseline research regarding the efficacy and sustainability of seeding riparian species at high densities on reclaimed agricultural lands. Research was conducted to understand the long-term impacts of using agricultural infrastructure for restoration practices, especially considering that these areas are disconnected from the main river channel and require long-term maintenance (e.g., irrigation). Few studies have monitored and disseminated results after the first few years of establishment. This research specifically focused on how density affects vegetation diversity and structural dynamics (e.g., terminal height, diameter at breast height, number of primary and secondary limbs) over time. Results benefit managers aiming to establish vegetation at specific densities that will meet desired measures of diversity and structural characteristics important for habitat creation.

This research was also part of a MSCP demonstration project and supplies baseline research for the USBR, who is tasked with re-establishing 2400 ha of cottonwood-willow habitat. If this method of restoration can be used to establish trees with desired structural attributes and community characteristics, large-scale restoration efforts can be implemented using these same strategies at much lower costs compared to traditional practices using pole-plantings or out-planted nursery stock.
3) How can remote sensing and eddy flux data be used to generate wide-area estimates of ET, an important component of the water budget in arid and semiarid systems?

To address the last question, I evaluated and refined empirical ET models using remote sensing and micrometeorological data. Sound water resource managers must benefit from knowing how much water can be allocated for human needs while addressing environmental needs. Because ET is often a large component of the water budget in arid and semiarid systems, tools are needed for large-scale estimations. Eddy flux tower networks produce the most accurate ET measurements at moderate scales (< km), but cannot be used alone for extrapolating ET to larger scales due to the spatial heterogeneity of desert vegetation. Remote sensing is perhaps the only feasible way to extrapolate ET to larger scales (> km). While a global ET product (i.e., MOD16 available through the ORNL DAAC) exists, distinct and large differences in this product to actual ET measured at flux towers has been observed, justifying the need to further improve upland and riparian ET models. I aimed to understand what drives ET in riparian-influenced vegetation compared to upland, water-limited systems. Identifying the variables that control ET across different vegetation associations is key to modeling ET at large scales. I addressed how ET is differentially affected across water limitation gradients (e.g., shallow groundwater versus no water table) and mosaic desert landscapes. The end products were empirical ET models incorporating the most appropriate inputs for estimating ET across riparian and upland landscapes characteristic of arid and semiarid
systems. This work also contributed to research associated with ET and groundwater projects funded by the Southwest Biological Science Center, US Geological Survey.

Watershed management in arid and semiarid ecosystems broadly encompasses many disciplines. Much of my research is dedicated to improving river restoration practices in the future. Realizing that only 10% of river restoration projects since the 1980s have incorporated monitoring and many have failed to determine whether they were effective at restoring ecological function (Bernhardt et al. 2005), it is essential that we improve current restoration practices. This will take a concerted effort among river managers, practitioners, and funders and requires bridging the gap between science and policy. This dissertation aims to highlight and disseminate key points expressed by river enthusiasts from private restoration firms, trying to contribute to broad-scale restoration while employing good science, to federal and state representatives in charge of setting overarching goals and distributing funding. My research is also dedicated to the science aspect and provides an example of how a local effort can contribute to baseline restoration research. In this project, I share approaches for monitoring restored vegetation over long timeframes but also disclose that monitoring the ecological component is only part of determining the overall effectiveness of river restoration. I emphasize the need to select meaningful methods and demonstrate how to quantify vegetation trends over time that will benefit managers who aim to create specific habitat. Embracing these types of local projects for what they offer in a lessons-learned context is invaluable to future efforts with larger goals and multiple objectives. Lastly, addressing how water resources are managed in arid and semiarid systems is of great importance in
the 21st century. With most dryland rivers over-allocated or threatened by human extractions, we as watershed managers and ecohydrologists will be faced with the challenge to sustain water resources for humans while addressing environmental needs. The health of our riparian ecosystems will depend on how we manage our water in the future, which includes, but is not limited to, improving methods for estimating water budgets, reforming water allocations, reducing human water use when possible (e.g., reduced agriculture, urban water conservation, etc.), improving irrigation schedules, and securing water for environmental needs. An ecohydrologist in the 21st century must be well-rounded, broadly trained, and well-informed in science and policy. Whether managing watersheds, restoring rivers, or managing water resources, the ability to openly communicate, employ novel strategies, document successes and failures, and adapt to changes will improve resilience and the ability to succeed in the future.

1.3 Dissertation Format

Following the introduction is the “Present Study” which provides a summary of the analyses and conclusions of three papers prepared for peer-reviewed publication. The three articles included as appendices are as follows:

Appendix A: “Restoring Rivers in the Semiarid and Arid Southwestern United States and Northern Mexico: Lessons Learned from the Past to Benefit Future Riparian Restoration.” In this paper I synthesized and summarized key points from a ‘lessons learned restoration conference,’ interpreted results, and drafted the manuscript;
Appendix B: “Long-term Vegetation Dynamics after High-density Seedling Establishment: Implications for Riparian Restoration and Management.” In this paper I performed all analyses, interpreted results, and drafted the manuscript;

Appendix C: “Using Remote Sensing and Eddy Covariance Data to Determine Evapotranspiration across a Water-limitation Gradient.” In this paper I performed all analyses, interpreted results and drafted the manuscript.

Altogether these three papers discuss findings that improve the efficacy of restoring rivers in the 21st century, re-evaluate the importance of water as a precious commodity in arid and semiarid systems, and enhance our understanding of how radiation, vegetation, and soil affect water transport processes.

2. PRESENT STUDY

The methods, results, and conclusions of this study are presented in the papers appended to this dissertation. The following is a summary of the most important findings in these documents.
2.1 Appendix A: RESTORING RIVERS IN THE SEMIARID AND ARID SOUTHWESTERN UNITED STATES AND NORTHERN MEXICO: LESSONS LEARNED FROM THE PAST TO BENEFIT FUTURE RIPARIAN RESTORATION

A conference convened in Tucson, Arizona, in December 2010 brought together river practitioners, scientists, private citizens, and conservationists from federal and state agencies, academic institutions, and non-governmental organizations to discuss challenges and lessons learned from past river restoration experiences for the benefit of future river restoration efforts. Open platform discussions focused on a variety of river restoration topics including planning, design strategies, monitoring, adaptive management, data sharing, and community involvement.

Developing realistic restoration objectives is a key component to all river restoration efforts and should be based on both a biophysical evaluation of river conditions and a sound understanding of the sociopolitical environment. If the factors that underlie a river’s ecological deterioration cannot be addressed, it is important to ask if the river can be restored, or if it even should be restored. While it is often not possible to achieve true restoration, returning a river system back to its pre-impact level or wild state, all restoration projects should aim to not only improve current conditions, but to achieve a certain level of sustainability. Small-scale projects provide useful lessons and garner stakeholder and public support, but typically produce limited biophysical benefit so every effort in the future should contribute to broad-scale objectives and goals.

Discussions also focused on the importance of having an interdisciplinary team,
developing a realistic timeline, and—from the outset—planning all phases of the project: design, implementation, monitoring, and evaluation. Of particular interest, was the need to develop measureable objectives that allow practitioners to quantify success, which was often lacking in many past projects. This also paves the way to implement adaptive management, which, in an ecological or restoration context, is the ability to alter management strategies to redirect a project’s trajectory toward success, given an inherent amount of uncertainty. Learning from past mistakes, and disclosing restoration failures is just as important as highlighting successes, so many participants agreed that sharing data and disseminating findings will be important for improving river restoration success in the future.

In addition to these topics, the conference also focused on the emerging river restoration themes of climate change, environmental flows, and restoration of trans-boundary rivers. Planning and design that includes climate change adaptation planning will be key to successful and resilient river restoration practices that take into account how rising temperatures and potential shifts in snowpack and precipitation will affect water resources. Because US and Mexico share many river miles, and policies and water laws differ among nations, it will be increasingly important to create bi-national collaborations. Both countries share similar concerns over water quantity and quality and river restoration in particular may require liaisons and partnerships between the US Department of Homeland Security to ensure that rivers can be protected while employing safe strategies along international borders.
2.2 Appendix B: LONG-TERM VEGETATION DYNAMICS AFTER HIGH-DENSITY SEEDLING ESTABLISHMENT: IMPLICATIONS FOR RIPARIAN RESTORATION AND MANAGEMENT

Human disturbances have contributed to the deterioration of many western US rivers in the past century. Cottonwood-willow communities, present historically along the Colorado River, protect watersheds and provide wildlife habitat, but are now among the most threatened forests. As a result, restoration efforts have increased to re-establish and maintain cottonwood-willow stands. While successful establishment has been observed using multiple strategies with varying investments, few projects are evaluated to quantify efficacy and determine long-term sustainability. We monitored a seeded cottonwood-willow site over a five-year period beginning in 2007, with particular interest in how density affected vegetation diversity and stand structure over time.

Fremont cottonwood (Populus fremontii) and volunteer tamarisk (Tamarix ramosissma) were the only abundant riparian trees at the site after one year. Compared to T. ramosissma, P. fremontii had higher growth rates, lower mortality, and dominated overstory and total cover each year. P. fremontii had an average terminal height of 2.43 m and an average diameter at breast height (DBH) of 0.76 cm, compared to 1.90 m and 0.36 cm, respectively, for T. ramosissma. While T. ramosissma was not significantly affected by density, higher densities suppressed terminal height, DBH, and resulted in lower growth rates of P. fremontii. We also showed that height-to-stem diameter relationships of P. fremontii were explained by a power relationship. If specific and
desired structural attributes were measured in situ (e.g., foliar volumes, number of stems or branches, etc.), allometry (e.g., relating height to DBH at different densities) could be used to determine areas suitable for avian habitat. This could prove to be an important tool for long-term restoration management and studying habitat suitability.

Vegetation diversity decreased from 2007-2009, but slightly increased from 2009-2011. This likely resulted from a large reduction in herbaceous vegetation throughout the study period, but with positive recruitment of native shrub species throughout the study period. As expected, diversity was highest in the lowest density class (1-12 stems/m²), but similar diversity was found among all other classes (13-24, 25-42, 43+). This suggests that plant diversity is not affected after a certain density is reached (e.g., 13 trees/m²). Although examined qualitatively, it was apparent that the trees at our high-density site had smaller stems and foliar volumes than trees restored at lower densities at adjacent fields. Also of interest, was that larger trees at our site, determined by sample root excavations, were phreatophytic after two growing seasons, and our biggest trees, not including in our surveys due to edge effects have reached ~10 m in height with 9-10-cm DBHs. Understanding long-term trends at densely-planted or seeded sites can benefit restoration managers who aim to establish specific community structure and vegetation diversity for wildlife habitat.
2.3 Appendix C: USING REMOTE SENSING AND EDDY COVARIANCE DATA TO DETERMINE EVAPOTRANSPIRATION ACROSS A WATER-LIMITATION GRADIENT

A responsibility for water resource managers is to ensure long-term water supplies for increasing human populations. Evapotranspiration (ET) is an important component of the water balance and accurate estimates are important to quantify how much water can be safely allocated for human use while supporting environmental needs. ET measurements are often representative of small (<1 km) spatial scales, but scaling up to basins has been problematic due to spatial and temporal variability. Satellite data, however, provide spatially distributed remote sensing products that account for seasonal climate and vegetation variability.

We used Moderate Resolution Imaging Spectroradiometer (MODIS) products [i.e., Enhanced Vegetation Index (EVI) and nighttime land surface temperatures (LST$_n$)] to create an empirical ET model calibrated using measured ET from three riparian-influenced and two upland, water-limited flux tower sites. Results showed that combining all sites introduced systematic bias, whereby upland sites tended to over-predict ET while riparian sites tended to under-predict ET. In practice, error would be introduced as a function of how much upland vegetation was present compared to riparian vegetation. We found ET to be differentially affected by groundwater and soil moisture at riparian-influenced sites compared to upland, water-limited sites so we developed separate models to estimate riparian ET and upland ET separately. While EVI
and LST$_n$ were the main drivers for ET in riparian sites, precipitation replaced LST$_n$ as the secondary driver of ET in upland sites. Riparian ET was improved with a multiplicative approach ($r^2 = 0.92$) while upland ET was adequately modeled using a multiple linear regression approach ($r^2 = 0.77$). These models can be used in combination to estimate total annual ET at basin scales provided that each region is classified appropriately and precipitation data is available.

Our approach was limited temporally by MODIS 16-day VI products of 250 m. Finer spatial resolution such as 30-m Landsat products may improve estimates but may not be offered at high enough temporal resolution to account for inter-seasonal fluctuations of ET important for water budgeting. Our upland model also requires a precipitation input, but some areas may not have sufficient precipitation data. Accuracy may be further impacted by interpolating rain from the nearest rain gauges or relying on precipitation models or forecasting to estimate upland ET. However, advances in satellite sensors will soon produce near-surface soil moisture readings which will greatly improve ET modeling in the future.
2.4 Future Research Opportunities

Here, I outline suggestions for future research opportunities that could follow research developed for my dissertation.

2.4.1 Regional River Restoration

Most arid and semiarid regions with stressed and degraded river systems encounter similar challenges for restoring rivers in the face of uncertainty. Some unpredictability stems from climatic stressors such as rising temperatures and other impacts related to climate change; others may stem from changes in social and political values or needs. What we do know is that past river restoration efforts are not working as well as they should, river restoration will remain a big industry in the future, and it is essential to develop strategies to improve the efficacy of future efforts. Below are topics that outline a few future research needs.

- Gaps among science, management, and policy still hinder river restoration progress and efficiency.

In the context of river restoration, scientists may stress the importance of addressing the factors responsible for ecological decline and may advocate the return of hydroecological processes while policy-makers may view practical options that account for overall the investment and societal needs. Social impacts or broader impacts often drive the types of river restoration employed. Social science may be a useful avenue to evaluate and justify which types of policy reforms could effectively
change river restoration practices that are not currently working. Perhaps platforms (e.g., conferences, seminars, blogs, interactive Internet) could be created to openly communicate the needs of scientists, river practitioners, policy-makers, and society as they relate to river restoration. New incentives and novel strategies should be investigated to improve efficacy of future efforts.

- *Most river restoration projects are local in scale and are limited by funding and time.*

  Future efforts should explore methods to promote longer funding cycles and longer time frames. Exploring or defining what an ideal river restoration framework looks like in the context of each project would guide river practitioners and funding agencies when designing restoration templates (e.g., a 5, 10, or 20-yr project investment; a 25%:50%:25% template on planning & design, implementation, and post-project monitoring, respectively; or a 25%:25%:50% template to accommodate adaptive management during post-implementation). Designing river restoration with funding to support all phases from planning to post-project monitoring is essential, but realistic planning may have to consider that funding may not be evenly appropriated over time.

- *Past restoration efforts are not well-documented, and as a result, are not valuable for future efforts.*

  The lessons-learned conference convened in Tucson, AZ was invaluable to participants because the first day was treated as a workshop, invoking open
discussions in smaller, eclectic groups. However, owing to the fact that river restoration has increased exponentially, the experiences from 1000s of projects and practitioners remain an untapped resource of information. With little accessible documentation, strategies to mine these past efforts could rebuild the foundation of river restoration. Perhaps interviews or additional conferences and/or seminars could gather more information that could add to the current restoration knowledge base. Currently, the majority of past projects are considered bad investments with lost information, but finding a way to learn from these projects could still be valuable for future efforts.

2.4.2 *Local River Restoration*

River restoration efforts have increased exponentially and the majority of efforts are small-scale, or local, projects. Many regulated rivers, in particular, have greater challenges for restoring rivers when dams cannot be regulated to produce or mimic floods. Despite the fact that hydroecological processes will not be returned along these rivers, many revegetation projects will continue to be implemented in the future. This document shares a case study along the lower Colorado River located on Field 51 at Cibola National Wildlife Refuge (CNWR) and exemplifies how longer-term monitoring could be used to not only assess tree health over time, but give insights to the ecological value of such projects. Because evaluation of the long-term sustainability of such efforts is in its infancy, future research will be necessary to document overall effectiveness.
Below, potential future project ideas are given and a few opportunities are highlighted to share how monitoring could continue into the future.

- **Local restoration sites require monitoring over longer time frames.**

  Revisiting the same field site (Field 51, CNWR, Unit #1) 10 years after initial seeding (i.e., 2007) to re-assess impacts of high-density establishment on vegetation dynamics (e.g., height, stem diameter, foliar volume, etc.) would add to the value of the current study outlined in this document. Additional surveys and repeat sampling will be valuable to managers interested in the long-term stability of seeding techniques. Revegetation strategies are usually implemented to create habitat for threatened or endangered avian species so long-term monitoring should also incorporate surveys to determine bird presence, abundance, and diversity.

- **High densities negatively impact vegetation diversity and structure.**

  High-density sites, such as Field 51, could be strategically managed to augment negative impacts on plant diversity and structure. Manually thinning woody trees at the field site could result in larger trees with higher foliar volumes, which may benefit avian communities that require specific structural attributes for nesting or stop-over habitat. Management should also consider how herbaceous vegetation and native shrubs could be seeded or planted in the open spaces to increase habitat value. Perhaps thinning trees could coincide with other management objectives or collaborate with other projects (e.g., cutting trees during dormancy could be
beneficial to other river restoration projects that need source trees for revegetation; this site could be a nursery to assist other revegetation projects that require genetically diverse native cottonwood trees). Managers should also consider estimating the overall cost and effort associated with long-term maintenance and management.

- **Re-establishing vegetation on disconnected floodplains requires long-term irrigation and maintenance so concerns arise regarding long-term sustainability.**

  Ceasing irrigation when trees become phreatophytic may save water resources, but may impact biotic and abiotic factors. I recommend a future experiment ceasing irrigation at Field 51 with subsequent monitoring to evaluate riparian tree health and give insights on the sustainability of such revegetation practices. Abiotic factors should be monitored to assess changes in soil and groundwater salinity. For instance, previous irrigation management may have leached salts and replenished groundwater, but ceasing irrigation could increase salinity levels. Assessing ecological value based on how hydrology and soils impact vegetation is also important to evaluate the suitability of using reclaimed agricultural fields for restoration practices.

2.4.3 **Water Resources**

Water is an important and valued commodity across arid and semiarid regions. Because large amounts of water are returned to the atmosphere via evapotranspiration...
(ET), large-scale estimates of ET are necessary to compute annual water budgets. The research in this document outlines an empirical approach that uses remote sensing to extrapolate ground measurements of ET to larger scales. These methods inherently have challenges and error related to the spatial and temporal resolution of data collected for modeling ET. Stemming from current research detailed in this document, a few future research needs are summarized below.

- Applying multiple ET models requires pixels to be classified into vegetation associations.

  Strategies to classify pixels representing “riparian-influenced” vegetation and “upland, water-limited” vegetation are required when applying our two empirical ET models. Spatial analysis software (e.g., ArcGIS, ERDAS Imagine) could be used to manually classify vegetation pixels before applying the appropriate model. Perhaps riparian-influenced sites could be differentiated from water-limited sites by exploring differences in remotely-sensed data (e.g., EVI or NDVI). For instance, pixel characteristic of riparian vegetation may have higher maximum Vegetation Indices (VIs) due to differences in water use and growth trends during the growing season. Exploring contrasting patterns or changes in VIs within the growing season may help isolate which pixels are influenced by groundwater versus those influenced by precipitation.
- *Upland models are limited by the difficulty of acquiring precipitation data at high spatial and temporal resolutions.*

  Data developed from models (e.g., Parameter-elevation Regressions on Independent Slopes Model, PRISM) that provide precipitation at better spatial resolution than point measurements (e.g., meteorological stations, rain gauges) should be explored for their value in upland ET models. If modeled precipitation data cannot be used to improve the upland ET estimates, then a sensitivity study could be performed to explore the number of rain gauges required within a given area to obtain precipitation data that produces valid upland ET estimates.

- *Upland models could be improved with a remotely sensed input related to moisture.*

  Current remote sensing products, such as water indices, should be explored as an improvement over precipitation in the upland empirical model. Future remote sensing efforts (e.g., SMAP) may offer improved soil moisture products that are more representative of the moisture status of uplands, and these should also be explored as an improvement to the upland empirical model.
3. REFERENCES


APPENDIX A: RESTORING RIVERS IN THE SEMIARID AND ARID SOUTHWESTERN UNITED STATES AND NORTHERN MEXICO: LESSONS LEARNED FROM THE PAST TO BENEFIT FUTURE RIPARIAN RESTORATION

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Abstract

A conference convened in Tucson, Arizona, in December 2010 brought together river practitioners, scientists, private citizens, and conservationists from federal and state agencies, academic institutions, and non-governmental organizations to discuss challenges and lessons learned from past river restoration experiences for the benefit of future river restoration efforts. Discussions focused on a variety of river restoration topics including planning, community involvement, data sharing, monitoring, biophysical evaluation strategies, and adaptive management. Developing realistic restoration objectives is a key component of all river restoration efforts and should be based on both a biophysical evaluation of river conditions and a sound understanding of the sociopolitical environment. Discussions also focused on the importance of having an interdisciplinary team, addressing broad-scale objectives, developing a realistic timeline, and—from the outset—planning all phases of the project: design, implementation, monitoring, and evaluation. Small-scale projects typically produce limited biophysical benefit but can provide useful lessons and garner stakeholder and public support for subsequent larger-scale efforts. However, only by identifying and addressing the main drivers of ecological deterioration can significant progress be made toward true restoration. In addition to these topics, the conference also focused on the emerging river restoration themes of climate change, environmental flow, native fish conservation, and restoration of trans-boundary rivers.
Key words: climate change, community involvement, environmental flows, ecosystem services, hydroecology, monitoring, native fish, trans-boundary rivers
1. Introduction

Riparian ecosystems provide a variety of ecosystem services (e.g., cycling soil and nutrients, treating pollutants, replenishing aquifers, supplying water, etc.) that benefit humans and the environment. Ecosystem services support human water demands for agriculture, industry, and municipal use and provide an array of recreational and experiential opportunities (Malanson, 1995). They are also extremely important in arid and semiarid regions, in particular, where the majority of the region’s wildlife species depend on them for survival (e.g., via refuge, habitat, migratory corridors; Arizona Riparian Council 1995). Riparian ecosystems in arid and semiarid regions are characteristic of cottonwood (Populus spp.) and willow (Salix spp.) communities, that maintain high productivity and biodiversity, support wildlife, protect watersheds, and provide recreation in an otherwise unproductive region (Fenner et al. 1985, Knopf et al. 1988, Szaro 1989, Malanson, 1995, Patten 1998).

A variety of human-related perturbations in the southwestern United States (US), northern Mexico, and southeastern Australia have resulted in over-allocated river water and hydroecologically deteriorated rivers. These anthropogenic disturbances have impacted ecosystem functions and services and have compromised the integrity of dryland rivers (Poff et al. 1997; Ward & Stanford 1995). Southwestern riparian ecosystems in the US, for example, have undergone significant changes in the past century to prevent flooding, supply energy, satisfy agricultural demands, and supply water to growing desert metropolitan areas (Rood and Mahoney 1990, 1991, Nilsson and Svedmark, 2002). River impoundment, land-use change, surface and groundwater...
extraction, and artificial inter- and intra-basin transfers have profoundly altered natural flow regimes (Poff et al. 1997, Naiman et al. 2002, Postel and Richter 2003, Revenga et al. 2005, Pearce 2007). Altered riparian ecosystem hydrology is responsible for severely degrading many of these dryland rivers (Rood and Mahoney 1990, 1991, Stromberg 1993, Lite and Stromberg 2005, Hultine et al. 2010). Water diversions following the construction of dams, channels, and levees have often resulted in decreased soil moisture and increased soil salinization (Busch and Smith 1995, Sala and Smith 1996, Glenn et al. 1998, Nagler et al. 2008), channel narrowing and incision (Shafroth et al. 2002), floodplain disconnection, and non-native species invasions that facilitate higher fire frequency (Busch 1995). Regulated flows also often impede the fluvial processes (e.g., vegetation scouring, sediment deposition, and soil/nutrient cycling) necessary for native riparian vegetation recruitment and survival (Howe and Knopf 1991, Ward & Stanford 1995, Poff et al. 1997, Taylor et al. 1999, Stromberg 2007). As a result, riparian forests are now considered among the most threatened forest types in the US (Swift 1984, Busch and Smith 1995, Shafroth et al. 2005).

In recent years, considerable efforts are being invested to recover deteriorated rivers. In fact, a representation of river restoration projects implemented across the US has shown an increase from 10s of river restoration projects in the 1980s, to 100s in the 1990s, to 1000s since the turn of the 21st century (NRRSS 2007). Unfortunately, most projects have lacked any form of assessment, monitoring, or evaluation to quantify effectiveness so, given the proliferation of restoration activities, concerns are rising regarding their overall efficacy, foci, and cost (Follstad Shah et al. 2007). River
restoration efforts implemented in the southwestern US follow the national trends elucidated by Bernhardt et al. (2005) and Follstad Shah et al. (2007), including increased project numbers and high costs with little investment toward post-project monitoring. While the Southwest generally has a higher percentage of projects with monitoring components compared to nation-wide statistics, Bernhardt et al. (2005) also found that most projects with monitoring protocols do not have appropriate information for quantifying success nor do they disseminate results. Smaller projects, specifically, make up the majority of all river restoration activities, yet monitoring is often left out of the planning and design phase (Bernhardt et al. 2005, Palmer et al. 2007), thus valuable information regarding success and failures is not shared with the broader restoration community. Given myriad restoration activities across the southwestern US and northern Mexico with little documentation; the knowledge, experiences, and lessons learned by restoration practitioners remains an untapped resource of valuable information.

The southwestern US and northern Mexico, like many parts of the world, have a diverse population of river practitioners who have been involved in river restoration efforts for years. Together, these practitioners have learned from an array of projects that focused on a variety of different river systems. Tapping into this resource to discuss and document lessons learned could be extremely valuable for future river restoration efforts. Protecting and restoring dryland rivers will face greater challenges with fewer resources for restoration activities, making it increasingly critical to implement efforts that will be successful. Combined, the southwestern US and northern Mexico form a region where concerns over water resources are anomalous to most regions of the world. Water
supplies in these regions are pressured by urban population growth as well as agricultural, commercial, and industrial demands (Gollehon and Quinby 2006). With water resources already over-allocated along most dryland rivers, it is also inevitable that impacts of climate change will likely exacerbate pressures on water availability in the future.

Considering that rivers are dynamic and unpredictable ecosystems, the lack of documentation of many river restoration projects has reduced the ability to implement efficient and robust restoration strategies. Therefore, it is essential to directly communicate with river practitioners (the scientists, river managers, conservationists, etc. who have worked directly in river restoration projects) to synthesize what has been learned from past river restoration experiences. Adding to the current restoration knowledge base will provide a stronger foundation for moving forward in the 21st century. One feasible way to gather information that addresses regional concerns is to present a platform to allow river practitioners to share their experiences. Compiling lessons learned from past restoration efforts is not new. Warner and Hendrix (1984) headed a restoration conference which resulted in 128 technical papers which outlined the ecology of California river systems and highlighted key factors that influenced their decline, what lessons had been learned, and what future management plans should be considered.

Despite the important information that these types of synthesis efforts provide, considerable and important questions remain regarding how the restoration and conservation world can most effectively address the challenges of rivers in arid and semiarid climates. Questions of particular importance include: What restoration
strategies have been effective? What approaches were not successful? How does this body of lessons learned inform future restoration? What are the main steps that practitioners need to contemplate in developing a sound and viable river restoration plan? Can communication between practitioners and funders be improved to meet broad-scale objectives? What gaps are present among science, management, and policy that may hinder future river restoration efforts? Can climate change adaptation be incorporated into the restoration equation to give greater confidence for meeting long-term goals in the context of a changing climate? Taking into account what new and continued challenges river practitioners will face in the 21st century, these questions aim to promote positive feedback and advance our understanding of our current river systems and restoration practices in effort to improve future river conditions with sustainable outcomes.

In December 2010, we designed and convened a conference in Tucson, Arizona entitled “Restoring Rivers in the Southwestern US and Northern Mexico: A Bi-national Conference on Learning from the Past for the Benefit of the Future.” We brought together river practitioners, scientists, private citizens, and conservationists from federal and state agencies, academic institutions, and non-governmental organizations from southwestern US, northern Mexico, and southeastern Australia to discuss lessons learned from their river restoration experiences. The necessity for such a lessons learned conference stemmed from the convergence of several factors, including the continuing and dramatic ecological deterioration of many of our rivers, increased understanding of climate change and its impacts on river systems, and growing appreciation of the need to come together as group, share information, collaborate, and document what we have
learned for the benefit of future natural resource practitioners. Discussions of particular importance at the conference centered on restoration planning and design as well as the emerging themes of climate change, environmental flow, trans-boundary rivers, and native fish conservation. In this article, we briefly discuss the design and intent of the conference and summarize lessons learned and key findings relayed during the five main conference sessions: development of restoration objectives, project design, and implementation; post-implementation considerations (e.g., monitoring, evaluation, site maintenance); quantifying and securing environmental flows; river restoration in the context of a changing climate; and restoration along trans-boundary rivers.

2. Conference Organization

The conference was attended by natural resource practitioners, scientists, and conservationists representing 13 federal agencies; 3 state agencies; 15 universities; 10 private organizations; 15 NGOs; 2 US tribal nations; and 3 county and city agencies from the US, Mexico, and Australia. Conference sessions mirrored the typical sequence of restoration project steps, progressing through goals, objectives, planning, implementation, and post-implementation. Four breakout groups of 40-50 participants each undertook discussions of these topics. Attention then turned to the emerging themes of river restoration in the context of climate change, environmental flows, native fish conservation, and trans-boundary rivers. Facilitators and note-takers documented discussion points, and English/Spanish translators provided information to all
participants. The end products were extensive session notes summarizing key information from the assembled practitioners, organized according to key steps in the restoration process (Figure 1).

3. Conference Results

3.1 Lessons Learned from the Planning and Design Phase

3.1.1 Determining the Overarching Restoration Goal. A project goal is a broad statement that brings the intention of the project into focus. The importance of developing a realistic restoration goal was a strong conference theme. Participants noted the value of asking ‘why’ a river should be restored, and even ‘if’ it could or should be restored. Having available restoration funds does not necessarily mean restoration should be initiated if other key ingredients are not in place. Participants felt that if practitioners and stakeholders could not agree about restoration along a particular river reach or system, or if restoration proved too daunting, funding mechanisms should be flexible enough to provide support in more feasible areas. For this reason, the importance of site prioritization was stressed given inherent political, legal, social, economic, and environmental challenges.

River restoration potential is strongly linked to a river’s hydroecology—the way hydrology impacts the ecosystem (Kundzewicz 2002, Zalewski 2002). Along these lines, participants agreed that successful restoration requires understanding current hydroecological conditions, how conditions have changed, the main stressors driving
change, and the possibilities of addressing stressors given current trends and the socio-
economic environment of the river. A project’s ability to effectively address stressors
determines feasible restoration endpoints. If stressors cannot be eliminated, or their
impacts at least reduced, it may be impossible to return a river to its natural or wild state.
In such a situation, river practitioners will be greatly limited as to what can be
accomplished and should apply a cost/benefit analysis that provides the basis for
identifying sites where restoration resources will produce the most benefit (Figure 2).

True river restoration (e.g., complete structural and functional return to a pre-
disturbance state; Cairns, 1991) aims to restore both hydrological and ecological
processes, which seldom may be possible at regulated rivers (Wohl, 2007). Conference
participants agreed that true restoration is impossible in most situations in the Southwest
given the degree of biophysical deterioration. A roadmap for developing realistic river
restoration goals emerged, understanding that a practical approach would likely aim to
improve current river conditions through rehabilitation strategies that have a sustainable
outcome. Preliminary steps to eliminate or reduce stressors include evaluating current
hydroecological conditions and understanding what can be accomplished given socio-
political will. Gathering such information allows the formation of a realistic vision—
articulated in a goal statement—that depicts the end product after objectives are met.
Goals should be at the watershed scale and have a long-term outlook. Strengthening
information gathering, team building, and planning at the outset of a project can provide a
strong foundation for developing goals that are spatially, temporally, and fiscally
realistic.
3.1.2 Developing Realistic Restoration Objectives. Restoration project objectives are descriptions of specific steps that contribute to achieving the project goal. Measuring the extent to which project objectives are achieved indicates progress toward meeting the goal. Participants observed that developing realistic restoration objectives requires preparing preliminary objectives, evaluating site-specific river conditions, and using the improved knowledge base to refine objectives. The initial objective might grow out of a question, such as whether it is possible to restore native fish populations. If subsequent baseline evaluations appear promising, the question might evolve to a stated objective: *We intend to restore native fish by improving habitat conditions necessary to support them with subsequent reintroduction of native fish stock.* The statement needs to be further refined to be quantifiable, providing the basis for a monitoring program that is capable of quantifying progress toward achieving objectives.

Evaluating baseline river conditions prior to planning contributes to formulating realistic restoration objectives (Kondolf and Downs 1996); provides information for project design (Jungwirth et al. 2002, Hoepfensperger et al. 2007); provides initial monitoring data for comparison to future conditions (Kondolf and Downs 1996); and is essential in planning, gauging restoration progress, evaluating effectiveness of restoration tactics (NRC 1992, Henry and Amoros 1995), and formulating adaptive management responses (Downs and Kondolf 2002). Despite the importance of this step, it is often minimized or overlooked due to time and funding constraints (Stromberg 2007).
The parameters included in a baseline evaluation vary with project focus, scale, funding, and data availability, among other considerations. Conference participants stressed the importance of documenting physical processes, especially water and sediment movement. Assessment of channel morphology, seasonal depths to groundwater, groundwater salinity, and average and variable streamflow statistics are important in this context. To reestablish native hydoriparian trees, site abiotic parameters such as soil texture, salinity, and seasonal soil moisture availability must be evaluated. Based on evaluation results, preliminary restoration objectives can be refined and finalized to describe specific measurable actions that contribute to achieving a realistic goal.

3.1.3 Developing the Restoration Plan. Participants noted that many projects have failed due to lack of planning or the lack of effective planning. Failures to reach restoration goals have been attributed to inadequate understanding of the ecologic system being restored (Westman 1991, Wohl 2005), poor definition of the initial problem (Hopfensperger et al. 2007), and unpredicted natural or anthropogenic disturbances (National Research Council [NRC] 1992). One of the first steps to remedy this is to develop a diverse interdisciplinary team, which might include engineers, scientists, biologists, ecologists, as well as politicians, community leaders, and other caring visionaries, who can help develop realistic restoration objectives and tactics based on the river system of interest. Multiple expertise within a restoration team can provide a holistic approach for understanding the science while accounting for social and political
values when implementing river restoration tactics. Routine meetings, checkpoints, and progress reports strengthen team coordination, foster flexibility in planning, and maintain project momentum.

River restoration takes time. Many river restoration projects run out of funds before completion due to unrealistic planning. Thoughtful fiscal planning at the onset is key to developing a realistic project timeline and a budget that supports all project activities, including baseline assessments at target sites as well as monitoring and evaluation activities after project implementation. Fiscal planning should also be sufficiently flexible to accommodate unforeseen obstacles. Working with government agencies early in the planning process helps participants gain a common understanding of required permits, objectives, personnel and material needs, and other important factors.

Pressure to complete work within fiscal rather than ecological timeframes is common. The challenge sometimes lies in meeting time and budget constraints imposed by grant obligations or project bidding. Conference participants were concerned that projects are sometimes awarded to applicants who knowingly underbid competitors, submitting unrealistic proposals that receive support rather than more thoughtful proposals that address overarching river restoration objectives. Successful river restoration in the future will require funders who embrace the bigger picture and who can build diverse partnerships among agencies, institutes, organizations, and citizens over longer time frames and funding cycles.

3.2 Lessons Learned from the Implementation Phase
3.2.1 Planting. Successful revegetation is fostered by knowledge of historical hydrologic and vegetative conditions and by designing restoration tactics that mimic natural site conditions (Wohl 2005). Conference participants noted that using locally collected and grown plant materials and collecting seed stock from multiple native plants near the restoration site ensures genetic diversity and increases establishment rates, which is consistent with the literature (e.g., Landis et al. 2003, Withrow-Robinson and Johnson 2006). Possibly the most important consideration is understanding and making certain that site soil texture, soil salinity, and above- and below-ground water availability will support the species being planted in the long-term (e.g., Anderson 1995, Aaron 2001, Dreesen 2002).

Many methods (e.g., seed distribution, pole-plantings, out-planting containerized nursery stock) with variable investments have been successful at re-establishing cottonwood-willow communities (e.g., Raulston 2003, USBR 2011a). Success has often been measured by the ability to simply establish trees or to establish trees at certain densities (Taylor et al. 2006); however, success should be measured by a project’s sustainability and ecological value (Palmer et al. 2005, Stromberg 2007), especially if strategies did not aim to restore historical hydroecological processes (e.g., flooding, scouring, cycling nutrients). Although revegetation is often an important element of restoration work, it alone cannot counteract causes of degradation and should be accompanied by strategies that directly address the main drivers of ecological deterioration, promote natural river dynamics, and enhance resilience.
3.2.2 Unanticipated Obstacles. Conference attendees shared stories of logistical challenges that impeded the ability to achieve restoration objectives. Progress can be improved by considering potential obstacles and anticipating speed bumps along the way. Conference attendees noted the importance of reviewing legislation and policy-related issues and securing necessary permits in advance of restoration work. Project planners need to identify and work within physical, legal and/or ecological constraints. Questions to ask include: Are any threatened or endangered species present at the restoration site? Are permits required to access chosen sites? Should bird migration windows be addressed in project design? Is coordination needed with other research groups or outside parties?

3.2.3 Project Management and Timing. Practitioners emphasized that thoughtful planning increases project efficiency and progress. Explaining goals and objectives; providing guidance and support; building relationships; and ensuring cooperation with staff, team members, and contractors helps all parties contribute fully. Fiscal and management timelines should coincide with natural processes and/or biological time windows. In some cases, pole-plantings of hydoriparian species have failed because poles were not collected during dormancy (USBR 2011a), or were not planted at appropriate times or in appropriate locations to accommodate groundwater fluctuations (Dreesen 2002). Efforts to establish vegetation from seed have failed because seeding did not coincide with natural seed dispersal windows or did not account for short-lived viability.
3.2.4 Importance of Small-scale and Pilot Projects. Tackling river issues at the watershed scale is key to addressing stressors and achieving long-lasting results but can be challenging, costly, and time consuming. As compared to restoration at the watershed scale, small-scale projects (e.g., pilot, site-based projects) will not produce dramatic biophysical and social benefits. Yet, such limited efforts can provide numerous short-term benefits and, when planned appropriately, can be part of long-term watershed-scale programs. In this vein, conference participants observed that small-scale projects provide opportunities to organize project teams, better understand current biophysical conditions and trends, and identify main drivers of decline. They also provide demonstrable, tangible results in the near-term, which are often critical for learning lessons and fostering support from community leaders and funders while garnering public support and stakeholder buy-in that pave the way for future collaborative efforts.

3.2.5 Community Involvement. Restoration programs need to benefit society as well as aquatic and riparian biota (Naiman et al. 2002; Richter et al. 2003). Public perception of a river’s condition often drives the decision to undertake the restoration as well as the decision about what type of restoration is needed (Wohl 2005). Highlighting the social benefits of restoration can enhance community support and appreciation, which is key to long-term viability. Obtaining public input can help shape restoration plans to better address such public needs as flood control, water supply, water quality, recreation, and aesthetics that can go hand-in-hand with environmental restoration goals. Community involvement can be promoted by emphasizing the ecosystem services that
rivers provide and how alterations such as damming and diversions pressure water quality and supply. An engaged community can assist in restoring, monitoring, and maintaining sites. Conference participants from several trans-border rivers gave examples of how riverside citizens have directly participated in constructing gabions, eradicating invasive species, planting native vegetation, and monitoring. With such collaboration, the project truly becomes a community experience, providing the foundation for the project’s long-term viability as well as garnering support for future restoration opportunities.

3.3 Lessons Learned from the Post-implementation Phase

3.3.1 Monitoring. The principal aim of monitoring is to measure progress toward meeting project objectives, underscoring the importance of developing quantifiable objectives and success criteria from the outset (England et al. 2008, Buchanan et al. 2012). The cost of monitoring must be built into the restoration budget, a practice not applied frequently enough. Conference participants noted that monitoring methods should be peer-reviewed, useful to scientists, meaningful to managers, and easily understandable to the community. Project team members should stay involved during the monitoring process to ensure effectiveness.

Ideally, monitoring should connect physical, chemical, biological, and even social parameters (Skinner et al. 2007, England et al. 2008). If restoration objectives aim to establish vegetation as habitat for native bird species, the monitoring program should aim to measure the vegetation characteristics (e.g., density, percent cover), key physical-chemical parameters (e.g., soil salinity, depth to saturated soils), bird presence (e.g.,
if/when/how birds use the habitat), and even such parameters as human visitation to quantify socioeconomic impacts of restoration.

Monitoring approaches need to be repeatable, rely on quantification, and provide data that measures progress toward meeting restoration objectives relative to baseline or control conditions (England et al. 2008; e.g., Underwood 1994). Participants reiterated the importance of selecting appropriate indicators that can reliably depict stream health, and cautioned that false indicators can lead to misinterpreting restoration outcomes. Approaches that rely too heavily on indirect measurements of stream health or subjective assessment without sufficient data collection should be avoided.

Participants also noted the importance of standardizing and documenting monitoring protocols to reduce impacts of spatial and temporal variability. The same methods should be employed year-to-year and be conducted at the same time each year. Ideally, monitoring should also include establishing control or reference sites that have similar climatological and pre-project biophysical conditions, to allow comparisons between treated and untreated sites (Jungwirth et al. 2002; e.g., Before-After-Control-Impact designs, reviewed in Underwood 1994). Standardizing monitoring so that local-scale programs can be linked to large-scale programs helps document progress toward meeting broad-scale objectives.

Given the potential diversity of scientists, managers, and policy-makers involved in river restoration, conference participants noted that defining whether objectives are successful may vary depending on the orientations and backgrounds of the restoration team. For this reason, different monitoring approaches, such as compliance-based
evaluations (e.g., verification of task completion or meeting quality standards; e.g., Mirauda, and Ostoich 2011) or functional monitoring (e.g., quantification of whether hydroecological processes have been restored and/or whether target species use the restoration site; e.g., Gonzalez and Garcia 2001) are commonly employed to evaluate results. As reviewed by Palmer et al. (2005), success is perceived differently among various disciplines and can be separated into a few categories: stakeholder (e.g., aesthetics, costs, compliance), learning (e.g., lessons learned and experience gained), and ecological (e.g., returning river processes, sustainability) success.

3.3.2 Adaptive Management. Participants noted the importance of employing adaptive management protocols as a means of improving the success of restoration efforts. While definitions vary with discipline or by the approach used (e.g., Walters 1986, Halbert and Lee 1991, Johnson et al. 2002), the essence of adaptive management is to employ an iterative process that optimizes management strategies over time in the face of uncertainty. The goal is to implement management and measure progress while recognizing that inherent uncertainties exist in dynamic and unpredictable riparian ecosystems (Walters, 1986). Ecological and social complexities also arise when implementing river restoration strategies, so strategic adaptive management can also incorporate how science, management, and policy influence outcomes (Kingsford et al. 2011). Adaptive management uses each step of a management program as an information-gathering exercise in which results are used to modify or design the next stage (Halbert and Lee 1991). Therefore, adaptive management requires the collection
and analysis of monitoring data to evaluate progress toward meeting a restoration objective (Thom 1997). Restoration results must be monitored, documented, and evaluated in a manner that provides the information needed to determine not only how well progress toward objectives is being achieved, but what is driving the level of progress. If progress is deemed unacceptable, knowing how to adjust restoration tactics to improve progress is the essence of adaptive management. Incorporating lessons learned into adaptive management strategies will improve future chances for project success while reducing uncertainties.

3.3.3 Data Sharing. Conference participants highlighted the importance of standardizing, centralizing, and sharing restoration data. Participants concluded that accomplishing these goals and disseminating data to the broader community will improve restoration, but will require coordinated efforts by the restoration community. Benefits of sharing qualitative and quantitative information include improving project coordination, learning from others’ experiences, avoiding “reinventing-the-wheel” scenarios, disseminating site background information, and identifying potential restoration team members.

Although Internet resources and technology can be used to build and archive restoration data, sharing data between different entities in an open, transparent manner can be a challenge. Proprietary rules regarding data sharing often need to be understood and addressed. Data sharing requires identifying which member or group of a diverse restoration team will take on the on-going responsibility of data entry, quality assurance,
and management. While standard data protocols exist at the federal level, an integrated, web-based database with standardized protocols could improve collaboration and data sharing between non-federal entities. Conference participants involved in trans-boundary river projects noted the additional challenge of establishing shared databases that are internationally recognized.

3.4 Emerging River Restoration Themes

3.4.1 Environmental Flows. As defined at the 2007 International River Symposium in Brisbane, Australia, environmental flows refer to “the quantity, timing, and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend upon these ecosystems” (Brisbane 2007). Securing environmental flows is a priority in semiarid regions in the US, Mexico, Australia, and other countries where water resources are often over-allocated with no water dedicated to supporting native river ecosystems. Inherent complexities of water law, regulations, compacts, and policies make securing water, as well as transporting secured water to its destination, a significant challenge. Despite these challenges, a variety of innovative strategies have been recently developed, which tend to fall into one of three categories: supply, demand management, or reallocation (Garrick et al. 2011). Identifying the most appropriate strategy for a stream or river requires understanding the relative costs and benefits of allocating water between multiple demands, including environmental needs.
In the Colorado River Basin, strategies for securing water for restoration activities focus mainly on purchasing water rights from owners of retired agricultural lands (John Swett pers. comm.). Water to meet these rights is stored in reservoirs and prescribed for future needs. Proposals are in the works in effort to secure in-stream flows whereby Mexico could store water rights in US reservoirs during surplus years to be used during dry years or for restoration that requires water prescriptions (e.g., quantified peak flow or baseflow discharges; Osvel Hinojosa pers. comm.). In contrast, along the unimpounded San Pedro River in southern Arizona, management has focused on retiring riverside agricultural lands in order to reduce groundwater extraction, thus protecting base flow that is needed to support native flora and fauna.

Linking water saved via water conservation actions to direct benefits for native ecosystems is a key legal and political challenge. Harvesting stormwater runoff can assist in recharging shallow aquifers, which may elevate water tables and potentially increase streamflows. Some regions have established water conservancy districts, developed management plans, and/or established grant programs that address water quantity and quality concerns. In Sierra Vista, Arizona, for example, the Partnership Water Conservation Grant Program gives grants to businesses to improve water conservation and efficiency, the Desert Hospitality Water Conservation Program offers incentives to lodging and dining establishments to save water, and the County Toilet Rebate Program offers local residences the option to replace old commodes with new efficient water-saving commodes (CCWCO 2012). Tucson, Arizona, is conducting a Conserve to Enhance pilot project that allows money saved on water bills to be pooled
and used to purchase environmental water rights (Schwarz and Megdal 2008).

Conference participants noted that environmental flow initiatives can take significant time to negotiate and implement, and it is critical to show near-term tangible progress (e.g., on-the-ground pilot projects) while long-term initiatives are prepared.

3.4.2 Climate Change. The issue of how to incorporate climate change information into the river restoration ‘equation’ was a key theme of the conference. It is likely that riparian ecosystems will be among the most sensitive to climate change given their dependence on precipitation patterns (Ormerod 2009). Future warming and drying predicted throughout the southwestern US into Mexico (Intergovernmental Panel on Climate Change [IPCC] 2007) will shift the timing and intensity of streamflows and consequently alter transport of water and sediment to and within channels (Pfister et al. 2004), producing a new level of uncertainty for setting restoration objectives and for managing restoration projects (Ingram 2009).

Despite the importance of designing and implementing climate change adaptation into restoration projects, only a few publications have addressed hydro-climatology in support of long-term water resources management and planning (Serrat-Capdevila et al. 2007). As part of a forthcoming river restoration guidebook based on conference results (see Discussion below), conference organizers are developing a climate-adaptive strategy specific to river restoration that centers on assessing the vulnerability of restoration objectives to climate change impacts.
A variety of programs and information sources are available to provide restoration practitioners climate change data and assistance, including CONAGUA in Mexico, National Oceanic and Atmospheric Administration (NOAA) in the US, many academic institutions in both countries, as well as the Department of Interior’s Climate Science Centers and Landscape Conservation Cooperatives, the Interagency Climate Change Adaptation Task Force created in 2009 by the White House, various state climatologist offices, and the National Phenology Network, among numerous others.

3.4.3 Native Fish Conservation. Of all fauna, native fish species may be the most affected by the biophysical deterioration of our rivers. Most regulated rivers can no longer sustain historical native fish populations because streamflow cannot be managed to mimic pre-dam conditions (e.g., turbidity, temperature, sediment transport) necessary for migration, spawning, and survival (Stanford and Ward 1986, Minckley and Meffe 1987, Marchetti and Moyle 2001, Brouder 2001). Conversely, non-native fish species often thrive in altered rivers and compete with natives (Stanford and Ward 1986, Richter 1997). Future restoration of native fish populations could be further impacted by climate change exacerbating stresses on native fish and potentially shifting geographic ranges (USBR 2011b). A key priority identified by conference participants was the need to develop criteria to identify high priority river reaches where long-term restoration would be most likely to succeed. Understanding which fish species may be imperiled and which species can benefit from even partial restoration of the natural flow regime can help
prioritize sites and benefit future efforts for native fish restoration (Poff et al. 1997, Propst and Gido 2011).

The urgency to protect and restore native fish habitat highlights the importance for strengthening communication and collaboration between ichthyologists, or other aquatic scientists, and river practitioners working on other types of restoration projects. Partnerships can promote opportunities to address complicated challenges, such as when river management is dictated by other priorities (e.g., recreational fishing). Fortunately, approaches for improving native fish habitat often parallel restoration tactics intended to restore hydroecological processes, thus the ability of a river to maintain native fish populations is a good indicator of overall river health.

3.4.4 Trans-boundary River Conservation. The US and Mexico share many river miles and both nations are interested in the conservation and protection of dryland rivers. Water quality and quantity are concerns in international talks and agreements. Trans-boundary cooperation and collaboration are of utmost importance when trying to ensure healthy riparian ecosystems. Challenges in trans-boundary work include political, social, cultural, language, training, and technological issues, along with differences in national and regional policies and goals.

Recently, security issues related to drug smuggling have made collaborative trans-boundary efforts even more challenging. Building partnerships with the US Department of Homeland Security could benefit restoration. Acquiring permits or work visas and establishing liaisons can increase travel efficiency and safety across borders. Improving
safety for restoration workers has helped efforts on the Rio Colorado by giving a former “no-man’s land” back to the community. Along the Rio Grande/Rio Bravo, work permits are obtained through the International Boundary Water Commission (IBWC) and Comisión Internacional Límite y Agua (CILA) that allow bi-national work crews to conduct restoration-related tasks on both sides of the river.

By building confidence and trust, on-the-ground pilot projects can provide the necessary foundation for larger, longer-term bi-national efforts. A key ingredient noted by several trans-boundary restoration experts was the importance of involving riverside citizens by both addressing socioeconomic community needs in restoration objectives, and by hiring citizens to conduct project activities. Involving river communities and showing short-term project progress was critical in the Colorado River Delta to obtain long-term agency buy-in and restoration plan support—another key ingredient of successful bi-national projects.

Project recognition and involvement with agencies from the highest government levels is also important. Encouraging government buy-in, working closely with key water users, and explaining broader goals are important when developing trans-boundary environmental flow programs. Discussing the lack of threats that conservation efforts have on water supply and quality and emphasizing overall benefits is also necessary for trans-boundary collaboration. The IBWC, CILA, and the Commission for Environmental Cooperation provide a variety of services supporting trans-boundary work, including fostering transparency between US and Mexican teams, enhancing technical and
scientific dialog, developing long-term objectives, conducting cross-training, and providing a mechanism for effective data exchange.

4. Discussion

All major rivers in the southwestern US and northern Mexico have experienced some degree of ecological deterioration. Efforts to restore rivers have increased exponentially over the last 25 years. However, little documentation exists describing or validating the best management practices for achieving restoration objectives. The 2010 Tucson conference provided river practitioners a platform at which to share experiences and lessons to benefit future efforts. The overarching goal is to promote regional river restoration such that local scale projects, which make up the majority of all river restoration projects, can begin to contribute to broad-scale programs and goals. Broad-scale river restoration that attempts to meet multiple objectives will require coordination at multiple scales, with funding that supports projects that target more than a single outcome (Gilvear et al. 2012). Perhaps the only way to accomplish such a feat, is to promote larger funding cycles that support longer project timeframes. Building partnerships and collaborations will allow varying expertise to attack multiple objectives that all are working toward one goal, protecting and saving our dryland rivers for generations to come.

An applied river restoration guidebook is being developed that incorporates results from the conference and draws on contributing restoration experts, with the
preliminary title: “Planning and Implementing River Restoration in the Southwestern United States and Northern Mexico.” The guidebook will provide detailed information on all aspects of stream restoration processes, from planning and implementation to monitoring and evaluation. Additional chapters will address climate change, quantifying and securing environmental flows, native fish conservation, invasive plants, and the development of restoration plans for rivers that cross political boundaries. The focus will be on dryland rivers in arid and semiarid regions, but it will also serve as a useful reference for planning river restoration in more mesic climates.
Implications for Practice
The ‘top ten” lessons learned were key points cited frequently during the conference or stressed in post-conference communications. All lessons are equally important.

- Stop the bleeding please! Effective river restoration cannot occur unless we understand and address the stressors that are driving hydroecological decline.
- As river experts, we are responsible for gathering the information needed to develop viable restoration objectives that describe what we want and where we want it in a manner that galvanizes support from public leaders and citizens.
- As ecosystems change through time, so do restoration targets. Therefore, the formulation of restoration objectives and tactics must be flexible and be based on up-to-date science and monitoring.
- Rivers are bodies of water and sediment. Understanding changes in both water and sediment budgets and the factors behind the changes is essential to sound restoration design.
- Maintaining heterogeneity of species and habitat types is critical. Protecting or reestablishing flood pulses is essential to meet this objective.
- Protecting streamflow for natural river ecosystems is key to restoration efforts. Securing novel water sources (e.g., effluent, agricultural return flows) through innovative strategies should be emphasized.
- River restoration programs need to be strongly supported by targeted scientific inquiry that illuminates key unknowns regarding river conditions, providing the foundation for moving forward despite continued uncertainty.
- Climate change is happening now. It is essential to incorporate an understanding of the latest climate forecasts and their potential hydroecological consequences into river restoration design and implementation.
- Though challenging, trans-boundary river restoration is essential and provides a variety of benefits beyond environmental that include enhanced communication and collaboration between adjoining governments and communities.
- Highlight success! When projects are successful—even if successes are small in scale—publically highlight your effort. Good news is always welcome, fosters good will, and can increase long-term support.
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Figure Captions

Figure 1. Flow chart for river restoration steps including phases for planning and design, implementation, and post-implementation.

Figure 2. A cost/benefit analysis conducted for potential restoration sites may assist in site prioritization and determining restoration feasibility. Taking environmental and socio-political constraints into account, restoration projects can fall into three categories: A) full recovery potential, resources are available for rapid improvement with long-term stability; B) limited recovery potential, resources may be available, but rehabilitation may take time to achieve a certain level of stability; or resources may be limited, but current conditions can be rehabilitated to attain an improved level of long-term stability; or C) no recovery potential, there are insufficient resources to succeed, or the site has irreversible anthropogenic damage (e.g., regulated dams, levees, channel diversion, disconnected floodplains, etc.) that cannot be mitigated or managed to improve hydroecological processes.
Figures

Figure 1.
Figure 2.

Cost/Benefit Analysis
APPENDIX B: LONG-TERM VEGETATION DYNAMICS AFTER HIGH-DENSITY SEEDLING ESTABLISHMENT: IMPLICATIONS FOR RIPARIAN RESTORATION AND MANAGEMENT

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Abstract

Human disturbances have contributed to the deterioration of many western US rivers in the past century. Cottonwood-willow communities, present historically along the Colorado River, protect watersheds and provide wildlife habitat, but are now among the most threatened forests. As a result, restoration efforts have increased to re-establish and maintain cottonwood-willow stands. While successful establishment has been observed using multiple strategies with varying investments, few projects are evaluated to quantify efficacy and determine long-term sustainability. We monitored a seeded cottonwood-willow site over a five-year period beginning in 2007, with particular interest in how density affected vegetation diversity and stand structure over time. Fremont cottonwood (Populus fremontii) and volunteer tamarisk (Tamarix ramosissma) were the only abundant riparian trees at the site after one year. P. fremontii, compared to T. ramosissma, had higher growth rates, lower mortality, and dominated overstory and total cover each year. Vegetation diversity decreased from 2007-2009, but was similar from 2009-2011 as a result of decreased herbaceous and increased shrub species richness. Diversity was highest in the lowest density class (1-12 stems/m²), but similar among all other classes (13-24, 25-42, 43+). High initial woody species densities resulted in single-stemmed trees with depressed terminal and radial growths. Allometry, relating height to DBH at different densities, could prove to be an important tool for long-term restoration management and studying habitat suitability. Understanding long-term trends at densely-
planted or seeded sites can benefit restoration managers who aim to establish specific community structure and vegetation diversity for wildlife habitat.

Key Words: Colorado River, cottonwood, diameter at breast height, Populus fremontii, revegetation, saltcedar, Tamarix ramosissima
1. Introduction

Western rivers in the United States (US) have been severely degraded throughout the past century. Land-use change, river impoundment, surface and groundwater extraction, and artificial inter- and intra-basin transfers have profoundly altered natural flow regimes (Poff et al., 1997; Naiman et al., 2002; Postel and Richter, 2003; Revenga et al., 2005; Pearce, 2007). Perhaps most affected by the physical degradation of western US rivers are native riparian forests. Southwestern riparian forests in the US consist primarily of cottonwood (*Populus* spp.) and willow (*Salix* spp.) communities that support wildlife, biodiversity, watershed protection, and recreation in an otherwise unproductive arid region (Fenner et al., 1985; Szaro, 1989; Patten, 1998). They are considered among the most threatened forest types in the US (Swift, 1984; Busch and Smith, 1995; Shafroth et al., 2005) due to severe degradation resulting from altered riparian ecosystem hydrology (Rood and Mahoney, 1990; 1991; Stromberg, 1993; Lite and Stromberg, 2005; Hultine et al., 2010). River regulation, in particular, has resulted in decreased soil moisture and increased soil salinization (Busch and Smith, 1995; Sala and Smith, 1996; Glenn et al., 1998; Nagler et al., 2008), channel narrowing and incision (Shafroth et al., 2002), floodplain disconnection, and non-native species invasions that facilitate higher fire frequency (Busch, 1995), which collectively have diminished the ability for native species to recruit and establish large stands (Howe and Knopf, 1991; Johnson, 1992; Merrit and Poff, 2010; Mortensen and Weisberg, 2010).
Because of the negative impacts from human disturbances and poor management practices imposed in the past, much attention has been given to restore western US rivers. Riparian ecosystems provide ecosystem services (e.g., cycling soil and nutrients, replenishing aquifers, treating pollutants, providing water supply, providing recreation opportunities, etc.) valued by humans (Carothers, 1977; Patten, 1998), but are also extremely important in arid and semiarid regions where they support the majority of the region’s wildlife species and maintain high biodiversity and ecosystem productivity (Knopf et al., 1998; Malanson, 1995). Riparian ecosystems in Arizona for example, make up less than 0.5% of the total land surface area, yet 60-75% of all resident wildlife depends on them for at least part of their lifespan (Arizona Riparian Council, 1995). For this reason, re-establishing cottonwood-willow communities has become a major component of river restoration in the Southwest. Although revegetation alone cannot return full hydroecological function to southwestern rivers, many efforts will continue to include management that focuses on habitat creation for avian and other wildlife communities that are considered sensitive, threatened, or endangered species under federal regulations.

Restoration projects have increased exponentially since the 1980s; however, monitoring data are lacking or inaccessible, raising concerns on project efficacy, numbers, foci, and costs (Follstad Shah et al., 2007). Many riparian revegetation methods have proven effective, but success has often been measured by the ability to establish native plants at high densities over short time windows (< 3 years; Taylor et al., 2006). The reasoning for establishing trees at high densities is derived from the
competitive exclusion theory, which, in an ecological context, states that two functionally similar species competing for the same resources cannot coexist (Grime, 1973). Implementing such a strategy for riparian restoration allows the competitive superiority of *Populus* spp. to limit success of volunteer species such as *Tamarix* spp. when water is not limiting (Sher et al., 2000; 2002).

Many projects have been successful at re-establishing native riparian vegetation at high densities using native seeds (Friedman et al., 1995; Taylor et al., 1999; Sprenger et al., 2002; Raulston, 2003; USBR, 2005; Bhattacharjee et al., 2006; Grabau et al., 2011). However, long-term monitoring data are required to quantify effectiveness and sustainability of this strategy over time (Taylor et al., 2006). To date, few studies have focused on longer-term persistence and competitive relationships of cottonwood and tamarisk stands, or on the relationship of initial seedling densities to long-term restoration of native riparian habitat in the southwestern US (Taylor et al., 2006). Many studies have reported the restoration of a certain number of species and age classes along regulated rivers, yet an understanding of long-term vegetation dynamics such as landscape heterogeneity and native species diversity is lacking (Stromberg et al., 2007).

The Colorado River, considered the most regulated river in North America (Adler, 2007), experiences negative impacts from river impoundment and agricultural practices, including native vegetation clearing, high soil salinity (Busch and Smith, 1995; Glenn et al., 1998), disconnected floodplains, and non-native species invasions that facilitate higher fire frequency (Busch, 1995). However, the Lower Colorado River (LCR) still serves as a greenbelt, an important migratory corridor for a host of birds, and
home to many aquatic and terrestrial wildlife species (USFWS, 2012). To ensure that these habitats continue to support wildlife, myriad conservation and restoration practices have been put into place along the LCR (USFWS, 2012; USBR, 2011a), particularly for re-establishing native cottonwood-willow habitat. Variable results with different levels of investment have been observed for many revegetation methods in the past (Raulston, 2003; USBR, 2011b). Selecting methods for future restoration practices should not only aim for specific densities, but should understand how different densities will affect structure and diversity as they relate to desired habitat characteristics.

The main objective of this paper is to understand longer-term (~5 years) vegetation dynamics after seeding riparian vegetation at high densities. To our knowledge, few studies have reported monitoring results spanning over a few years since seedling establishment. However, quantitative and qualitative information regarding vegetation stand structure, diversity, and insights to the overall trajectory of such practices are important to managers when choosing restoration strategies in the future. In 2006, as part of the LCR Multi-Species Conservation Program (LCR-MSCP), the US Bureau of Reclamation (USBR) implemented a study to test the feasibility of using native seeds for riparian restoration. The project aimed to investigate potential methods for seeding native riparian vegetation on a retired agricultural field with supplemental irrigation (Grabau et al., 2011). Through the first three growing seasons, P. fremontii showed competitive superiority over T. ramosissma (Bunting et al., 2011). Additional surveys in 2010 and 2011 and a secondary analysis of data spanning from 2007-2011 allowed us to examine density effects on growth, mortality, and diversity over a five-year
period. While previous research demonstrates the ability of *P. fremontii* to be successfully seeded and established in a field setting, this research explores the long-term sustainability of such practices with particular emphasis on how woody tree density affects growth and survival, overall vegetation diversity, and tree stand structure over time.

True river restoration (e.g., complete structural and functional return to a predisturbance state; Cairns, 1991) aims to restore both hydrological and ecological processes, which seldom may be possible at regulated rivers (Wohl, 2007; Briggs, 2011). Regardless, management to re-establish cottonwood-willow habitat will continue to occur on both reclaimed agricultural lands as well as more natural settings because it satisfies the Habitat Conservation Plan for covered (i.e., endangered species) and evaluation (i.e., could be listed in the future) species on the LCR (USBR, 2011a) and provides an ecological improvement from current conditions, which are characteristic of patchy, discontinuous vegetation and tamarisk monocultures that do not provide quality habitat. The success of converting agricultural land for restoration practices will be determined, in part, by long-term evaluations of abiotic and biotic factors that underlie the overall quality, function, and sustainability of the habitat created. Determining levels of passive or active maintenance required to sustain such practices in the long term may influence when and where restoration is executed. If true restoration is not feasible, efforts should be invested at suitable sites or should develop quantifiable objectives that will improve current conditions to a certain level of sustainability, understanding the physical limitations that are imposed at the project site.
2. Methods

2.1 Study site

This study was conducted on a former agricultural field on the Cibola National Wildlife Refuge (CNWR) and is managed by the USBR. The field lies 1.5 km east of the Colorado River (33°22′03″ N, 114°40′50″ W; 71 m elevation) in La Paz County, AZ and is located on the historical floodplain terrace (Figure 1). Annual temperature ranges from 4°C to 49°C and rainfall averages under 100 mm yr$^{-1}$.

Soil at the site is predominately silt and silt loam down to a meter below ground surface. Sand and sandy loam strata are prevalent below, especially toward the southern portion of the field (GSA, 2007). Prior to irrigation, average surface soil salinity was less than 2 dS m$^{-1}$ and tended to be higher near the northern portion of the study site and lower at depth in coarser, sandy soils (GSA, 2008). Salinity measured from 1-m soil cores in winter 2007 was highly variable among plots, ranging from 1.8 dS m$^{-1}$ to 26.9 dS m$^{-1}$ (soil salinity at 15 cm below ground surface (EC$_{15}$) averaged 4.4 dS m$^{-1}$ and ranged from 1.8 to 22.3 dS m$^{-1}$; EC$_{46}$ was 8.4 dS m$^{-1}$ and ranged from 2.4 to 22.9 dS m$^{-1}$; and EC$_{91}$ was 6.3 dS m$^{-1}$ and ranged from 0.5 to 26.9 dS m$^{-1}$). Groundwater salinity measured in March, April, and June 2009 ranged from 1.6 to 2.7 dS m$^{-1}$. Depth to groundwater, measured with on-site well-point piezometers prior to irrigation ranged from 2.2 to 2.6 m below ground surface and ranged from 1.6 to 2.7 m throughout the study period (GSA, 2008; Bunting et al., 2011).
2.2 Irrigation management

The field site, originally seeded in May 2007, was surface or sprinkler-irrigated daily for the first 10 days. Between day 10 and day 25, the field was watered every other day, and was watered occasionally thereafter with a maximum gap of eight days between irrigation events during the first growing season. Non-growing season irrigation was infrequent. Irrigation in the 2008 growing season was scheduled to approximate 80% of reference crop evapotranspiration (ET\textsubscript{0}) computed at the Cibola Weather Station (WRRC, 2012). Soil moisture and tree health condition monitoring in 2008 showed no water stress, so irrigation in 2009 and 2010 was prescribed to approximate 60% of ET\textsubscript{0}. In 2011, irrigation infrastructure was modified such that calculation of applied water was not possible; however, plots were flood-irrigated occasionally to prevent stress and provide water during the growing season. Average irrigation across the field ranged from 5-15 cm every one to four weeks during each growing season with occasional watering in the winter.

2.3 Field design

The field site was divided into 36 individual plots (6 m x 12 m) to test germination success across various treatments (Grabau et al., 2011). In May 2008, three randomized transects (6 m) were laid across each plot to measure overstory and total cover percentages for all vegetation at the field site over time. Permanent quadrats (1.0 m x 0.5 m) were randomized along transects to assess health condition and to measure
target tree (i.e., Fremont cottonwood, *P. fremontii*; tamarisk, *T. ramosissma*) growth rates, survival, and density over time. Target trees within each quadrat were assigned a permanent tag with a unique number to allow measurement of the same tree in subsequent surveys. To reduce edge effects and avoid disturbed areas following 2008 root excavations, this study analyzes two of the three original transects and quadrats from each plot (i.e., those on the interior of each plot).

Initial tree densities ([stems/m²]; i.e., number of trees within each 0.5 m x 1.0-m quadrat multiplied by two) were computed by counting how many target trees (not stems) were present in each quadrat. Densities, ranging from 0 to 56 stems per 0.5 square meter, were extrapolated to stems per square meter and assigned using the histogram distribution. Density classes were chosen as follows: 0 (n = 7); 1-12 (n = 16); 13-24 (n = 16); 25-42 (n=17); and 43+ (n=16). We analyze and compare vegetation metrics among density classes to assess how density impacts native versus non-native trees over time.

2.4 Vegetation monitoring

2.4.1 Growth and survival. Overall growth rate [cm/year] was determined by subtracting height measurements for every target tree within each quadrat from each fall survey, which were conducted every year at the end of the growing season. Tree height in this study is defined by the terminal height of the primary stem. Growth rates were analyzed among density classes and between *P. fremontii* and *T. ramosissima*. Mortality percentages were assessed by documenting all dead trees in each quadrat upon subsequent surveys (i.e., Mortality % = (d/n)*100, where d is the total number of deaths
within each quadrat during the year and \( n \) is the total number of trees documented in each quadrat). Mortality was also analyzed among density classes and between species.

### 2.4.2 Cover and diversity.

Cover percentages for both overstory and total cover were computed using the point-intercept method (i.e., Cover \( \% = \frac{x}{n} \times 100 \), where \( x \) is the number of “hits” of a given cover type and \( n \) is the number of sample points in each transect for each plot). Four categories were selected to document species richness: 1) Riparian (all target species including \( P. \) fremontii, \( S. \) gooddingii, \( S. \) exigua, and \( T. \) ramosissma); 2) Shrubs (all volunteer shrubs including \( P. \) spp., \( B. \) spp., and \( P. \) sericea); 3) Grasses/Sedges; and 4) Herbaceous (all non-grass annuals).

Vegetation diversity at the species level was computed using the Shannon-Wiener Index (i.e., \( H' = -\sum p_i \ln p_i \), where \( H' \) is species diversity and \( p_i \) is the proportion constituted by species \( i \) of all species present; Shannon, 1948), which was chosen because it accounts for the relative abundance of each species and is the preferred index for communities with few species (Tiegs, et al., 2005). Real communities often have rare species present at very low numbers and the Shannon-Wiener Index, unlike other methods that treat rare and common species the same, addresses diversity by weighting species by their proportions (Huston, 1994; Jost, 2012). Vegetation diversity was analyzed among density categories as well as total diversity after four years as observed using the 2010 fall survey (2011 was not analyzed because only a sub-sample of plots was surveyed due to limited resources).
2.4.3 *Stand structure.* Qualitative and quantitative information was gathered to document various structural attributes (e.g., tree form, number of stems per tree, terminal height, and diameter at breast height). A total stem count (i.e., vegetation stem counts include primary stems and secondary stems, when an additional stem branches out between the base of the tree up to 10 cm above ground surface) was tallied and compared to the total tree count. Relationships between terminal height and diameter at breast height (DBH) were compared between target trees and between years 2008 and 2009. Radial growth [cm/yr], calculated by subtracting DBHs between two field surveys (2009 and 2008), was compared between target species and among density classes.

3. Results

3.1 Growth and survival

Overall *P. fremontii* growth rate was significantly higher in 2009 (31.2 cm/yr) as compared to 2010 (15.2 cm/yr) and 2011 (15.0 cm/yr) (Figure 2a). *T. ramosissima* growth rate was higher in 2009 and 2010 (4.75 cm/yr and 5.52 cm/yr) compared to 2011 (0.88 cm/yr), although not significant (Figure 2b). *P. fremontii* growth rates were significantly higher than *T. ramosissma* in all years (Figure 2).

*P. fremontii* growth rate generally decreased with increased density in each year except 2011 where the highest growth rates were observed in the highest density class (43+ stems/m²; Figure 2a). *T. ramosissma* growth rates did not follow a trend when
Comparing among density classes; the highest growth rates were observed in lowest density class (1-12 stems/m²) and the second highest density class (13-42 stems/m²).

*P. fremontii* mortality was similar to *T. ramosissima* in 2009, but was significantly higher in 2010 and 2011 (Figure 3). The highest *P. fremontii* mortality (11.5%) was observed in 2009, whereas the highest *T. ramosissima* mortality (49.4%) was observed in 2011, which was significantly higher than all other mortality rates during the study.

Mortality rates were not affected by density for either species. No *P. fremontii* mortality occurred in the lowest or second lowest density class in 2010 or the lowest density class in 2011, however, *T. ramosissima* mortality was observed in every density class every year (Figure 3).

### 3.2 Cover and diversity

In 2007, a few months after seeding, grasses and sedges dominated both overstory and total cover percentages (Figure 4). After two growing seasons, *P. fremontii* dominated the overstory cover and after three growing seasons *P. fremontii* dominated total cover. *P. fremontii* was the only vegetation class that increased in both overstory and total cover each year with both percentages equaling 89% in 2011. As a result, no other vegetation class had high overstory cover percentages throughout the study period. *T. ramosissima* and grasses and sedges decreased in total cover percentage throughout the study period, while shrubs and forbs experienced a slight increase from 2009-2011.

Species richness within the herbaceous and riparian classes decreased over time, while species richness in the shrub class increased over time (Figure 5). Species richness
for grasses and sedges varied throughout the study period. Vegetation diversity decreased significantly from 2007-2009, but was similar from 2009 and 2011 (Figure 5). Diversity was lowest in the zero density class (which lacked any woody species), was highest in the lowest density class (1-12), and did not differ in the highest three density classes (Figure 6).

3.3 Stand structure

Only three *P. fremontii* trees in the stem-count survey had secondary stems, resulting in a stem count of 548 from a total of 545 trees tagged. Total *T. ramosissma* stem count matched the tree count of 500. The average *P. fremontii* terminal height was 2.43 m while the average *T. ramosissma* terminal height was 1.90 m. *P. fremontii* height and DBH were significantly reduced at higher densities (Figure 7a), whereas *T. ramosissma* height and DBH were not affected (Figure 7b). The average *P. fremontii* DBH was 0.76 cm with a maximum of 3.35 cm while the average *T. ramosissma* DBH was 0.36 cm with a maximum of 1.82 cm. Radial growth for *P. fremontii* more than doubled that of *T. ramosissma* from 2008 to 2009 (Figure 8). Excluding the highest density class, radial growth of *P. fremontii* decreased significantly as density class increased, but no significant differences were found when relating radial growth of *T. ramosissma* among density classes (Figure 8).

As expected, terminal height and DBH were strongly related (Figure 9, Table 1). We looked at year-to-year differences between relationships to depict changes in growth patterns over time. Regressions of *P. fremontii* terminal height and DBH showed slightly stronger linear relationships in 2009 ($r^2 = 0.86$) compared to 2008 ($r^2 = 0.84$), but *T.*
**ramosissima** was opposite, with a stronger linear relationship observed in 2008 (Table 1). Overall, terminal height-to-DBH ratios were higher in *P. fremontii* compared to *T. ramosissma* as depicted by slightly higher-sloped trendlines (Figure 9, Table 1). Because we expected this relationship could change as trees grew older, we analyzed non-linear relationships such as logarithmic and power functions that may account for reduced growth over time. Only in the case of *P. fremontii* did a power function improve the relationship (Figure 9). A linear trend better represented height-to-DBH relationships for *T. ramosissma* in both years (Table 1).

4. Discussion

4.1 Long-term stand community dynamics

Based on all evaluation measures (e.g., growth rate, mortality rate, overstory and total cover percentages, radial growth, etc.) through five years, *P. fremontii* has clearly demonstrated competitive superiority over *T. ramosissma* at this field site. This complements many studies that have highlighted the competitive advantages that *Populus* spp. have over *Tamarix* spp. when water is not limiting (Stromberg, 1997; Sher et al., 2000; 2002; Bunting et al., 2011). Rapid growth rates and minimal mortality have resulted in a dominant *P. fremontii* overstory, but have consequently reduced grass and herbaceous cover in the understory. While our tallest tree heights in our dataset reflect *P. fremontii* and *T. ramosissma* terminal heights approaching six and four meters (Figure
respectively, some trees at this site have reached over 10 meters after only five years of growth.

Allometric relationships are often used in forest ecology to estimate biomass production, carbon stocks, fuel load, or simply to characterize trees at different life stages (e.g., Gibbs et al., 2007; Wolf et al., 2011). Generally, height is roughly proportional to diameter among understory saplings, but increases more gradually among large canopy trees (Kira, 1978; Sterck and Bongers, 1998). This may partially explain why after two years of growth, *P. fremontii* trees, which have a larger range of terminal heights, displayed a stronger power relationship than *T. ramosissma* which, on average, consisted of much smaller trees. Interestingly, adding two large trees (10.0 m, 1000-cm DBH and 9.2 m, 920-cm DBH) from ancillary data as tie points produced a *P. fremontii* power relationship with the same $r^2$ with an almost identical equation as observed in our dataset, supporting the use of a power function for a wider range of tree sizes. Tree allometry also varies in relation to the environment (King et al., 2009). For example, trees grown in wide open spaces have trunks that can be several times as thick as trees of the same height grown in dense patches (Ek, 1974; King, 1981). Our *P. fremontii* trees, having lower DBHs at higher densities, support this relationship. Restoration that targets a specific density can relate established tree density to metrics such as tree height, DBH, and stem count. These relationships could be valuable to resource managers that are interested in how density will affect tree structure and growth over time.

Similar to approaches used in forest management (Leftsky, 2001; Mette, 2004), allometric relationships built around height, which can be quickly estimated in large
spatial extents using modern technologies such as LiDAR (Farid et al., 2006; Popescu, 2007), could be used to identify areas with appropriate structure for nesting habitat. For instance, selected vegetation characteristics collected in situ can be related to height and then, using LiDAR, larger areas could be analyzed to find suitable habitat. As a first step, we built linear and power relationships between height and DBH for young target trees. Additional allometric relationships valuable to habitat could include biomass, crown or canopy width, and foliage density or vegetation volume. This research suggests that allometry could be a valuable tool for long-term restoration management if large databases were created to supply pertinent information on how different species heights are influenced by density and how density, in turn, affects overall structural dynamics. These relationships could then be used in restoration design and could be evaluated and validated after establishment to assess overall effectiveness or suitability.

4.2 Vegetation diversity

Restoration endpoints often aim to maintain a desirable level of vegetation diversity; however, as observed in this study, restoring native woody vegetation at high densities negatively impacts recruitment of new species. Overall species richness decreased every year with the most prevalent reduction in herbaceous cover. Herbaceous growth established and maintained in open spaces, however, is one of the best wildlife habitat improvements within large tracts of woodland (VDGIF, 2012). The Shannon-Wiener Index has been commonly used to measure vegetation diversity because it can track changes in relative species abundance over time (e.g., Huang et al., 2007; Ishida et
al., 2010; Shaheen et al., 2011). For instance, studies using the Shannon-Wiener Index have shown decreases in species diversity and richness over time since flood disturbance (Tiegs et al., 2005). Diversity at our field site reduced rapidly in the first couple years, but leveled out three years after initial seeding. A slight increase in diversity in 2011 was likely attributed to the recruitment of native shrub species, concurrent with reduced proportions of riparian species. Native shrubs volunteering at the field site included *Pluchea sericea*, *Prosopis glandulosa*, *Baccharis salicifolia*, and *Atriplex lentiformis*, all of which are native species used in conservation plantings (USBR, 2011a). Vegetation diversity could be augmented by adding seeds from desired native species to the seed mix. Given their ability to volunteer in low-light, dense conditions, manually thinning trees at high-density restoration sites could allow more desired shrubs to establish and perhaps allow herbaceous species to proliferate in open spaces between woody vegetation. Supplemental management could also include subsequent planting or seeding of desired understory species, but long-term site maintenance may not be cost efficient.

4.3 Implications for wildlife habitat

Community composition and stand dynamics are important topics to discuss when establishing trees for avian habitat, yet few studies focus on how revegetation strategies affect these attributes. Some land managers consider high density establishment to be more successful than low-density establishment, without taking into account that higher densities may result in lower DBHs and hence lower stand volume (Bhattacharjee et al., 2008; Taylor et al., 2006). The ability of native riparian communities with rich canopy
structures and abundant decadent trees to support more birds than saltcedar monocultures, and to also support nest cavities and harbor greater species diversity has been well documented (Anderson and Ohmart, 1982; 1984; Busch et al., 1992; Ellis, 1995; Ohmart et al., 1988; Sedgwick and Knopf, 1986). Furthermore, certain structural attributes may favor a particular bird species over another (e.g., Southwestern Willow Flycatcher versus Brown-headed Cowbird), thus management efforts should take into account specific nesting and refuge requirements for targeted bird species. For instance, high-density native communities with shorter canopies may be desired to support habitat requirements for the Southwestern Willow Flycatcher, while low initial densities with larger Populus spp. forests may favor cavity-nesting bird species (Taylor et al., 2006).

Although foliar density was not quantified, high-density establishment of woody species at our field site resulted in noticeably lower foliar volumes and DBHs compared to recently restored riparian vegetation at adjacent fields, which is consistent with other studies analyzing density effects on tree structure (Taylor et al., 2006). Furthermore, few trees, out of over a thousand tagged, had greater than one basal stem, which is likely a product of the high-density establishment of both P. fremontii and T. ramosissma. Single-stemmed riparian vegetation may or may not be a desired structural attribute when considering habitat requirements (e.g., bird nesting) for certain bird species. However, larger trees observed at our study site have secondary branches large enough to support nests and multiple nests have been observed in both P. fremontii and T. ramosissma trees, although no official bird surveys have been conducted to quantify distribution, abundance, or diversity.
4.4 Reclaimed agricultural fields and native seed distribution

The suitability of using reclaimed agricultural fields for revegetation, particularly on the LCR, has been challenged (Aaron, 2011; Stromberg, 2007). For restoring cottonwood-willow communities, in particular, it has been suggested to rely on the following edaphic conditions: a) groundwater level (0.1 m - 3m); b) soil moisture; c) salinity (< 2 dS); d) pH (6 - 8.5); and e) soil texture (coarse) (Aaron, 2011). In one study using these guidelines, only one of nine chosen sites along the Colorado River was considered suitable for restoration, and the majority of these sites were limited by insufficient groundwater levels (Aaron, 2011). However, canal infrastructure at agricultural sites can be used advantageously to supplement water until roots of riparian vegetation reach the water table (Bunting et al., 2011). The ability of larger *P. fremontii* and *T. ramosissma* trees at our site to reach shallow groundwater (< 3 m) after two growing seasons suggests that using agricultural irrigation infrastructure would likely succeed at sites having deeper groundwater with similar soil textures.

Despite not meeting all of the proposed edaphic conditions (e.g., our site had salinity levels reaching well over 3 dS m⁻¹ and soils characteristic of silt and silt loam), *P. fremontii* and *T. ramosissma* trees have established and persisted for over five years. However, irrigation has been applied consistently each growing season throughout the five-year study period so tree response if irrigation were to cease is still unknown. Irrigation not only supplements soil moisture, but flushes potentially harmful salts that may evapoconcentrate on surface soils (ASCE, 2011). Results to date show site and soil suitability have been sufficient for sustaining riparian vegetation, given extensive
irrigation. Typically, soil and groundwater salinity increase as a function of distance from the main channel (Glenn and Nagler, 2005). Our site is further from the main channel but has similar depth to groundwater to the majority of proposed MSCP sites on the CNWR.

Importantly, the intent of this USBR project was not to restore hydroecological processes, but to determine the feasibility of using and to demonstrate how to use native seed distribution techniques for native riparian revegetation. Irrigation regimes were implemented to grow riparian vegetation according to reference crop calculations and to provide moist soils to promote conditions for invertebrate communities, but did not necessarily mimic natural flood conditions nor aim to restore hydrological conditions. Nonetheless, large *P. fremontii* trees are now phreatophytic and have the potential to be sustained by shallow groundwater even after irrigation is ceased in the future. While restoring riparian vegetation on a disconnected floodplain is not ideal, sustaining a mixture of native and non-native species may contribute to continuity and refuge necessary for migratory birds. Addressing all the factors responsible for degradation along regulated rivers may not be possible, but restoration efforts may be able to mitigate some negative impacts with sustainable outcomes. Recognizing the limitations inherent in the present-day system, a compromise between purity and practicality has to occur to progress toward establishing a significant amount of restored habitat on the LCR (Raulston, 2003). For this reason, monitoring and quantified assessments of local restoration projects, such as this, are important contributions to the restoration knowledge
base as future efforts aim to improve current and past management practices while addressing broad, regional goals.

4.5 Conclusions

After five years, high-density establishment of woody species has resulted in lower vegetation diversity and reduced overall growth potential of woody species. Reduced terminal and radial growth rates are more pronounced in *P. fremontii* compared to *T. ramosissma*; however, this is precluded by the competitive superiority of *P. fremontii* observed at this field site. Relatively low mortality throughout the study period suggests that high recruitment of other woody, shrub, or herbaceous species will not likely occur without additional management to thin trees and create open spaces. The decision to revegetate native species at high densities could benefit from a better understanding of what ideal measures of biodiversity are, why they are important, and how targeted wildlife species will be affected. Overall effectiveness will depend on defining quantifiable objectives and addressing other underlying factors (e.g., moist soils; invertebrate presence, abundance, and diversity; distance to surface water) that may not only be impacted by the type of management imposed, but could be just as important for attracting targeted wildlife species as the trees themselves. Monitoring efforts such as this offer insight to long-term restoration trajectories and can be used to improve future efforts for sustaining native riparian vegetation with desired densities and structure.
Acknowledgements

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Table Captions

Table 1. Height to diameter at breast height relationships through two and three growing seasons since seedling establishment.
Figure Captions

Figure 1. Site map of Cibola National Wildlife Refuge. Our study site is located on Field 51, a retired agricultural field located on Unit#1 on the north side of the refuge approximately 1.5 km east of the Colorado River.

Figure 2. Growth rates over time for: a) Fremont cottonwood; and b) tamarisk. Student’s t tests were used to compare means where lower case letters denote significance among density classes within each year while capital letters denote significance among years surveyed. Error bars depict standard error.

Figure 3. Mortality rates over time for: a) Fremont cottonwood; and b) tamarisk. Error bars denote significance using 1.96 standard deviations (α = 0.05).

Figure 4. Vegetation cover percentages by selected vegetation classes surveyed over five years for: a) overstory cover; and b) total cover. Error bars are not shown to avoid clutter, although all error bars were small due to high sampling numbers each survey (n=1512).
Figure 5. Species richness by vegetation class (bars) and Shannon-Wiener Index for diversity (bold numbers) over time. All vegetation at the field was divided into four categories: Riparian (all target species including cottonwood, tamarisk, and willow trees); Shrubs (all non-target woody species including mesquite, baccharis, arrowweed, etc.); Grasses/Sedges; and Herbaceous (all annual and forb species).

Figure 6. Shannon-Wiener Index for diversity measured during the Fall 2010 survey. Target stems include all woody target species (i.e., Fremont cottonwood and tamarisk) per square meter.

Figure 7. Relationships between 2009 density and: a) Average diameter at breast height (DHB); and b) Average height of Fremont cottonwood and tamarisk trees. P-values denote significant differences (slope) across density classes using an Analysis of Variance.

Figure 8. Radial growth among density classes for Fremont cottonwood and tamarisk from year 2008-2009. Student’s t tests were conducted to test significance where lower case letters denote significance among density classes within species and capital letters denote significance for average radial growth between species.
Figure 9. Height and diameter at breast height relationships after two (2008, closed circles) and three (2009, open circles) years of growth for a) Fremont cottonwood using a linear trend; b) Fremont cottonwood using a power trend; c) tamarisk using a linear trend; and d) tamarisk using a power trend.
### Tables

**Table 1.**

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Figures

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APPENDIX C: USING REMOTE SENSING AND EDDY COVARIANCE DATA TO DETERMINE EVAPOTRANSPIRATION ACROSS A WATER-LIMITATION GRADIENT

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Abstract

A responsibility for water resource managers is to ensure long-term water supplies for increasing human populations. Evapotranspiration (ET) is an important component of the water balance and accurate estimates are important to quantify how much water can be safely allocated for human use while supporting environmental needs. ET measurements are often representative of small (<1 km) spatial scales, but scaling up to basins has been problematic due to spatial and temporal variability. Satellite data, however, provide spatially distributed remote sensing products that account for seasonal climate and vegetation variability. We used Moderate Resolution Imaging Spectroradiometer (MODIS) products [i.e., Enhanced Vegetation Index (EVI) and nighttime land surface temperatures (LST_n)] to create an empirical ET model calibrated using measured ET from three riparian-influenced and two upland, water-limited flux tower sites. Results showed that combining all sites introduced systematic bias, so we developed separate models to estimate riparian and upland ET. While EVI and LST_n were the main drivers for ET in riparian sites, precipitation replaced LST_n as the secondary driver of ET in upland sites. Riparian ET was successfully modeled using an inverse exponential approach ($r^2 = 0.92$) while upland ET was adequately modeled using a multiple linear regression approach ($r^2 = 0.77$). These models can be used in combination to estimate total annual ET at basin scales provided that each region is classified appropriately and precipitation data is available. This approach improves
accuracy of ET estimates at large scales while accounting for daily to seasonal fluctuations in ET.

Key Words: enhanced vegetation index, flux tower, land surface temperature, MODIS, riparian, semiarid
1. Introduction

A common goal for water resource managers in the southwestern United States, and arid and semiarid regions worldwide, is to ensure long-term water supplies for increasing human populations. In these regions, evapotranspiration (ET) is an important component of the water budget [Nichols, 1994; Dahm et al., 2002; Glenn et al., 2010] and has large implications for water resources management [Jackson et al., 2001; Newman et al., 2006]. As a result, accurate ET estimates are important to quantify how much water can be allocated safely for human use while supporting environmental needs [Hansen and Gorbach, 1997; Congalton et al., 1998; Commission for Environmental Cooperation, 1999; U.S. Department of Interior, 2002]. Over the last 15 years, much attention has been given to estimating ET along natural riparian corridors, across agricultural fields, within disturbed floodplains, or a mixture of each. Although riparian ET has been well-studied, most ET measurements are made at local scales. Sound water resource management, however, requires accurate ET estimates at watershed and basin scales, which have been difficult to obtain due to heterogeneous patterns of land cover as well as mixed riparian and upland vegetation [McDonnell et al., 2007; Scott et al., 2008]. Spatial variability in such environmental factors as precipitation and below ground water availability (e.g., groundwater, soil moisture) further complicates the ability to scale ET.

An understanding of water resources and the hydrologic cycling of these resources within any given system is necessary to effectively scale ET. Determining actual rates of water consumption by riparian vegetation requires knowledge of
interspecies differences in water use, as well as broad-scale estimates over large river sections [Nagler et al., 2005a]. Bowen ratio and eddy covariance flux towers are now often used to monitor ET for representative riparian plant associations on southwestern rivers [e.g., Devitt et al., 1998; Goodrich et al., 2000; Coonrod and McDonnell, 2001; Cleverly et al., 2002; Dahm et al., 2002; DeMeo et al., 2003; Scott et al., 2004]. At scales of a hundreds to thousands of square meters, the eddy covariance technique has been regarded as the best and most reliable method to estimate ET [Rana and Katerji, 2000; Drexel et al., 2004], and many studies have confirmed reasonable estimates of evaporation using this approach [Barr et al., 2000; Wilson et al., 2001; Schume et al., 2005; Kosugi and Katsuyama, 2007].

However, the eddy covariance method can only provide measurements representative of a particular type of vegetation cover when there is an extensive, uniform area of that vegetation immediately upwind of the tower [Unland et al., 1998; Baldocchi, 2003]. Therefore, directly extrapolating patch-scale ET to larger landscapes may lead to biased regional estimates because towers may not adequately represent larger areas [Wylie et al., 2003]. Wylie et al. [2003] recommended that site-specific tower data could be used in combination with remote sensing and other data sources to develop statistical algorithms for extrapolating tower data to regional scales. Due to the difficulties encountered with varying spatial and temporal scales, the use of satellite-based remote sensing is perhaps the only feasible means of estimating ET over wide areas of mixed landscape types [Glenn et al., 2010], owing to the fact that it is the only technology that can provide representative parameters such as radiometric surface temperature, albedo,
and vegetation index in a globally consistent and economically feasible manner [Choudhury, 1989; Kustas and Norman, 1996].

Physically-based remote sensing models using thermal near infrared (NIR) bands are available for estimating ET at large scales [reviewed in Kalma et al., 2008; (e.g., SEBAL, Bastiaanssen et al., 1998; METRIC, Allen et al., 2007; and ReSET, Elhaddad and Garcia, 2008)]. These methods compute latent heat as a residual of the surface energy balance and are often preferred because they measure physical properties of the surface which allow ET estimations across different landscapes on a pixel-by-pixel basis [Overgaard et al., 2006]. This approach does not require calibration using ground measurements of ET, but does require micrometeorological data and high spatial resolution at the cost of low temporal resolution. These methods are often more appropriate for detecting stress at the level of individual agricultural fields [Nagler et al., 2009]. Furthermore, these methods may produce erroneous ET estimates in natural ecosystems because they can falsely assume a constant evaporative fraction when scaling up instantaneous measurements of ET from a single satellite over pass [Nagler et al., 2009; Glenn et al., 2010].

Another promising approach, however, has been to find empirical relationships between flux tower ET and remotely sensed data such as vegetation indices (VIs). Despite apparent obstacles, a number of successful VI methods have been developed to estimate ET in a variety of semiarid landscape types across the globe including agricultural districts [Kim and Hogue, 2008; Gonzalez-Dugo et al., 2009], riparian zones and desert phreatophyte communities [Guerschman et al., 2009; Nagler et al., 2005a, b,
grasslands and savannas [Cleugh et al., 2007; Alfieri et al., 2009; Guerschman et al., 2009; Nagler et al., 2009], and water-limited shrublands [Nagler et al., 2009]. Empirical VI approaches may be more appropriate when estimating highly variable daily and seasonal ET because flux tower networks record time-series data at high temporal resolution (i.e., every half-hour) while satellite data provide high temporal resolution and spatially distributed products that account for seasonal climate and vegetation variability. A statistical algorithm can then be developed to scale ground measurements of ET to wide areas (> 1 km) using remote sensing.

As an example, Nagler et al. [2005a] developed a single, empirical equation for estimating ET across multiple river systems. River-reach ET was estimated by correlating flux tower ET to the enhanced vegetation index (EVI) acquired from the Moderate Resolution Imaging Spectroradiometer (MODIS) and maximum daily air temperature measured from flux tower sites. In a similar study, Scott et al. [2008] developed an empirical ET model using EVI and nighttime land surface temperature ($LST_n$) acquired solely from MODIS products. Such ET models can be used in practical applications such as quantifying water budgets or determining groundwater use in a riparian zone or catchment [Scott et al., 2008].

Typical approaches for applying empirical riparian ET models require the desired river reach to be extracted and digitized. Then, an algorithm can be applied to this area of interest assuming that the inputs to the ET model can be attributed to each pixel as necessary. As a potential drawback, the digital images used for these types of analyses
inherently include pixels representing upland vegetation or bare soil along with pixels of common riparian vegetation such as large cottonwood/willow associations or mixed mesquite woodlands. Models such as the one developed by Scott et al. [2008] are calibrated using eddy covariance towers located in close proximity to the San Pedro River and influenced by relatively shallow groundwater (≤ 10-m depth). In practice, the model generally over-predicts ET when the region of interest consists of mosaic vegetation (e.g., mixed riparian and upland vegetation) [Scott, pers. comm. 2011]. This error is likely caused by the presence of pixels characteristic of upland vegetation and/or bare soil. The algorithm applied to these upland regions is the same as the algorithm calibrated with and used for adjacent riparian pixels. As a result, overall estimates of water lost at the catchment or river-reach scale are over-estimated as a function of upland area or bare soil present in the scene. For this reason, calibrating empirical ET models such as these with additional data from adjacent, upland landscape types may improve ET estimates.

Our objective is to develop an empirical ET model to improve basin-scale estimates of ET in semiarid regions characterized by mosaic vegetation. By re-calibrating an existing ET model with upland sites and riparian sites spanning a water-limitation gradient, we aim to produce a model useful for management decisions. To do this we will incorporate data from two water-limited flux tower sites along with three riparian sites influenced by shallow groundwater. We will examine whether these upland sites capture the difference in EVI signal and thus the variation in daily and seasonal ET observed between upland and riparian sites. Finally, we will evaluate which variables
explain the greatest variation in actual ET at each site and investigate how the inclusion of inputs that address the dependency of upland vegetation on soil moisture, such as precipitation, affect model predictions.

2. Methods

2.1 Study Sites

Five flux tower sites were used for this research including three riparian-influenced (i.e., all sites have access to shallow groundwater during at least part of any given year) sites (CM, LM, LS) and two water-limited (i.e., neither site has access to groundwater reserves) upland sites (KG, SR) in southeastern Arizona, US (Table 1, Figure 1). These regions extend from the Santa Cruz River Watershed in the west to the San Pedro River Watershed in the east. All sites are considered to be semiarid with cool winters and warm summers where rainfall varies spatially and temporally with mean annual precipitation (MAP) ranging from 250 mm to 390 mm [McClaran et al., 2002; Scott et al., 2008; Nagler et al., 2007]. Precipitation is characterized by bimodal patterns in which 50-60% of the total annual rainfall arrives during the summer (July-Sept) as part of the North American Monsoon [Adams and Comrie, 1997]. MAP was documented at each of the five sites during the study period (2003-2007; Table 1).

The three riparian sites are located in the San Pedro River Natural Conservation Area (SPRNCA) within the San Pedro Basin and are influenced by relatively shallow groundwater supported by intermittent San Pedro River stream flow. Soils consist of
sandy loam alluvium with interspersed layers of gravel and clayey materials (Scott et al., 2008). The Charleston Mesquite Woodland riparian site (CM, N 31.6636 W -110.1778) is adjacent to the main river channel and had a mean annual temperature of 16.8 °C during the study period. CM is a dense mesquite woodland dominated by velvet mesquite (Prosopis velutina) with an understory consisting of perennial sacaton bunchgrass (Sporobolus airoides), greythorn (Zizyphus obtusifolia), and various summer annuals. The Lewis Springs Mesquite riparian site (LM, N 31.5658 W -110.1344) is located within one kilometer of the main channel and had a mean annual temperature of 16.2 °C during the study period. LM is a mesquite shrubland with sparsely distributed P. velutina, scattered patches of S. airoides, and other various shrubs within the tree interspaces. The Lewis Springs Sacaton riparian site (LS, N 31.5615 W -110.1403) is adjacent to the main river channel and had a mean annual temperature of 15.4 °C during the study period. Dominant vegetation includes dense S. airoides with occasional summer annuals.

The two upland sites include a grassland and a mesquite savannah. The Kendall Grassland upland site (KG, N 31.7365 W -109.9419) lies in the northwest portion of the Walnut Gulch Experimental Watershed and is located over 20 km from the San Pedro River. The mean annual temperature during the study period was 17.4 °C. Surface soils are generally very gravelly with sandy to fine sandy loams with some silty clay to clay loams at depth. Historically, vegetation consisted of diverse patches of native bunchgrasses including black grama (Bouteloua eriopoda), hairy grama (Bouteloua hirsuta), curly mesquite grass (Hilaria belangeri), and hook three-awn (Aristida
hamulosa). Following the peak of the drought (2003-2006), the dominant vegetation has been replaced by non-native Lehmann love grass (*Eragrostis lehmanniana*). The Santa Rita Mesquite Savannah upland site (SR, N 31.8214 W -110.8661) is located in the Santa Rita Experimental Range approximately 65 km west of the San Pedro River and 45 km south of Tucson, AZ. The mean annual temperature during the study period was 19.2 °C and dominant vegetation includes *P. velutina* and *S. airoides*.

2.2 Eddy Covariance Flux Data

The eddy covariance technique was used to estimate daily ET [Moncrieff et al., 2000]. At each of our sites, typical instruments included three-dimensional sonic anemometers to measure wind speed and infrared gas analyzers to measure CO₂ and H₂O fluxes. Additional instrumentation included net radiometers, ground heat flux plates, and tipping bucket rain gauges. Methods specific to the San Pedro sites can be found in Scott et al. [2004, 2008] while the upland sites are described in Scott et al. [2009] and Scott [2010b].

Daily ET values were estimated by averaging the original half-hourly flux tower ET measurements. A 16-day average was then computed to match 16-day composites offered by remote sensing products. Likewise, precipitation was summed for each 16-day window to match remote sensing products. Due to the lag time associated with processing and quality-assuring flux tower data, eddy flux data used for this research included five years of riparian site data (2003-2007) and four years of upland site data (2004-2007).
2.3 Remotely Sensed Data

Pre-processed MODIS products are available through the Oak Ridge National Laboratory Distributed Active Archive Center (ORNL-DAAC; http://daac.ornl.gov/MODIS/). Vegetation indices (MOD13Q1 products) were obtained as 16-day composites with 250-m resolution and land surface temperatures (LSTs, MOD11A2 products) were obtained as 8-day composites with 1000-m resolution. Single pixels containing the coordinates for each tower were extracted to match the time period available from all flux tower sites.

ET models using pre-processed MODIS data acquired from the ORNL-DAAC have a temporal resolution limited by the 16-day VI products. As such, two 8-day LST products were averaged to match each 16-day window offered by VI products. The MOD13Q1 products include calculations of both NDVI and EVI while the MOD11A2 products include measurements of both daytime (LST\textsubscript{d}) and LST\textsubscript{n}. Coefficients of determination (r\textsuperscript{2}) were used to determine which MODIS variables explained the most variation in measured ET, and thus which variables were chosen as inputs for the empirical ET model.

2.4 ET Models

While many empirical ET models have been created [see Nagler et al., 2005a, 2005b, 2007; Murray et al., 2009], the Scott et al. [2008] model was selected for this study because inputs can be acquired solely from remote sensing products. This makes it
a valuable approach for researchers and managers limited by data acquisition. The empirical ET model introduced in Scott et al. [2008] is:

\[
ET = a(1-e^{-bEVI}) + c(e^{dTs}) + e
\]

where \(a\), \(b\), \(c\), \(d\), and \(e\) are calibration coefficients, and \(Ts\) is LSTn.

This ET model was calibrated using all five study sites combined spanning from 2003-2006. EVI and LSTn were the sole inputs and the model was calibrated using measured ET from the five flux tower sites. The surface fitting tool in MATLAB (The Mathworks, Inc., Natick, MA) was used to identify parameters. The new equation with calibration coefficients took the form:

\[
ET = -14.45(1-e^{-0.622EVI}) + 0.197(e^{0.089Ts}) - 1.028
\]

To validate the model, Equation (2) \(r^2 = 0.78\), \(RMSE = 0.57\) was used to compute ET predictions for 2007 at each of the five sites. Predicted ET was then compared to measured ET in 2007. Root mean square error (RMSE) was used to compare model performance for predicting ET at each study site individually. The location of predicted ET values with respect to the 1:1 line gave insight to whether a particular site tended to under- or over-predict ET.

3. Results
3.1 Estimates of catchment ET using combined approach

Regression analyses were conducted to compare relationships of ET among selected variables across each site as well as each year of the study period (Table 2). Excluding the KG site, EVI explained more variation in ET than NDVI at each site. Similarly, MODIS LST\textsubscript{n} was a stronger predictor of ET than average daily temperature (T\textsubscript{avg}) or MODIS LST\textsubscript{d} at all sites. These findings were consistent with Scott et al. [2008] and justified the use of Equation (1) to develop the parameters given in Equation (2). Equation (2) (r\textsuperscript{2} = 0.78, RMSE = 0.57) successfully predicted 2007 ET (RMSE = 0.62); however, values among each site were not evenly distributed along the 1:1 line, suggesting a bias (Figure 2). Upland sites with the majority of values observed above the 1:1 line tend to over-predict ET, while riparian sites with the majority of values below the 1:1 line tend to under-predict ET when all sites are combined (Figure 2). Plotting residuals shows an even distribution when all sites are combined; however, when examining residuals by vegetation association, it is evident that riparian sites have a positive skew while upland sites have a negative skew. If this model was used in practice, estimated ET along a river reach would be systematically under- or over-estimated depending on the dominant vegetation association and plant water status in the scene.

3.2 Influence of upland vegetation on estimates of catchment ET
Actual ET, examined qualitatively with precipitation, showed a much different trend at riparian sites versus upland sites (Figure 3). Riparian site ET often more than doubled ET values observed at upland sites and riparian site ET was less affected by individual rainfall events whereas upland site ET had noticeable spikes in response to rainfall events. In fact, coefficients of determination ($r^2$) showed that EVI and $LST_n$ were the strongest predictors of ET for all three riparian sites, but EVI and precipitation explained most of the variation in ET at the two upland sites (Table 2). These findings suggest that Equation (1) is better suited for predicting riparian ET. Furthermore, calibrating Equation (1) with both riparian and upland sites negatively impacts the model’s ability to accurately estimate basin-scale ET. Because ET between riparian and upland sites was differentially affected by precipitation, riparian and upland sites were separated in effort to improve ET predictions within each association.

### 3.3 Improving ET models for estimating riparian ET

Equation (1) takes the form of an additive and inverse exponential function to account for lower and upper ET thresholds [see Nagler et al., 2005; Scott et al., 2008]. This equation assumes that riparian ET approaches zero when winter $LST_n$ and EVI values reach a minimum, while summer ET reaches full potential, stabilizing when $LST_n$ and EVI reach optimal levels. Theoretically, ET should be dependent on temperature (e.g., LST) multiplied by a scaling factor to account for the amount of light intercepted by the canopy (e.g., EVI) [Nagler et al., 2005]. A multiplicative approach is more realistic for estimating ET at riparian sites because it accounts for an interaction between $LST_n$
and EVI rather than having additive terms. Furthermore, this approach is analogous to traditional crop coefficient approaches that multiply potential ET of a reference crop by a crop coefficient. Replacing crop coefficients with VIs has been explored on a theoretical basis because VIs give a measure of the actual state of a crop canopy, instead of a crop coefficient produced by expert opinion [Choudhary et al., 1994; Glenn et al., 2010]. As a result, we altered Equation (1) to take a multiplicative form in effort to further improve riparian ET predictions specifically:

\[
ET = a(1-e^{-bEVI^*})e^{cT_s} + d
\]  

(3)

Equation (3) also simplifies the model from five to four parameters and scales EVI between averaged values acquired from bare soil sites (i.e., 0.059) to represent \(EVI_{\text{min}}\) and fully vegetated sites (i.e., 0.704) to represent \(EVI_{\text{max}}\). The three bare soil sites included an area under construction and two mine tailings and the three fully vegetated sites included a golf course, a fully vegetated park, and a grass lawn, all located within the study region. The following equation was used to scale EVI:

\[
EVI^* = 1 - (EVI_{\text{max}} - EVI)/(EVI_{\text{max}} - EVI_{\text{min}})
\]  

(4)

Scaled EVI \((EVI^*)\) allows ET to reach a minimum (i.e., the intercept) when \(EVI^*\) approaches 0.0 and allows ET to reach full potential when \(EVI^*\) approaches 1.0.
Equation (3) was calibrated with all sites combined, riparian sites only, and upland sites only to obtain and compare three new empirical ET models, respectively:

\[
ET = 2.587(1-e^{-1.218EVI^*}) \ast (e^{0.064LST}) + 0.016
\] (5)

\[
ET = 1.109(1-e^{-3.464EVI^*}) \ast (e^{0.075LST}) + 0.062
\] (6)

\[
ET = 294.5(1-e^{-0.001EVI^*}) \ast (e^{0.164LST}) + 0.318
\] (7)

Although Equation (5) \(r^2 = 0.85, \text{RMSE} = 0.47\) produced favorable 2007 ET predictions (RMSE 0.63) and was an improvement from Equation (2) \(r^2 = 0.78, \text{RMSE} = 0.57\), the model still produced bias and skewed residuals (Figure 4). Equation (6) \(r^2 = 0.92, \text{RMSE} = 0.37\), as expected, produced the best 2007 riparian ET predictions (RMSE = 0.40) and values among each riparian site were evenly distributed along the 1:1 line (Figure 5a), suggesting that the model is unbiased. Furthermore, plotted residuals were randomly dispersed, validating this approach for estimating riparian ET. Not surprisingly, Equation (7) \(r^2 = 0.67, \text{RMSE} = 0.37\), calibrated with upland sites only, did not predict 2007 upland ET as well (RMSE = 0.46), suggesting that this empirical model may not be the most appropriate for estimating ET at water-limited sites (Figure 5b).

3.4 Accounting for rainfall when estimating upland ET
Equation (2) produced over-predictions of ET in the upland sites and prompted an analysis of ET relationships with other variables. Because changes in ET in arid and semiarid ecosystems are often coupled to growing season moisture inputs that wet upper soil layers [Scott et al., 2006b; Williams et al., 2006], it was not surprising that precipitation explained more variation of ET than LST$_n$ at the upland sites. Models in the form of Equation (1) and Equation (3) are sensitive to temperature and greenness but do not sufficiently account for near surface soil moisture. However, if ET is strongly coupled to pulses of moisture in upland ecosystems, then ET models need to have a variable that explains how soil moisture affects ET. Therefore, a new model was created in effort to improve upland site ET specifically by using a multiple linear regression approach with precipitation replacing LST$_n$ as a new model input:

$$ET = aEVI + bPPT + c$$  \hspace{1cm} (8)$$

where PPT [mm $^{-16\text{day}}$] is precipitation summed over each 16-day window at each flux tower. Equation (8) was calibrated with all sites combined, riparian sites only, and upland sites only to obtain three new empirical ET models, respectively:

$$ET = 12.00EVI + 0.013PPT – 0.956$$  \hspace{1cm} (9)$$

$$ET = 12.43EVI + 0.011PPT – 0.900$$  \hspace{1cm} (10)$$
ET = 6.93EVI + 0.017PPT – 0.507  \hfill (11)

These multiple linear regression models produced favorable $r^2$ values for each case (Table 3). Equation (9) predicted overall 2007 ET well (RMSE = 0.58, Figure 6), but at similar performance to Equation (2) (RMSE = 0.62) and Equation (5) (RMSE = 0.63, Table 4). This model also shows similar bias in which upland sites tend to over-predict and riparian sites tend to under-predict 2007 ET. Furthermore, unevenly dispersed residuals among each site suggest that combining all sites together does not create a robust multiple regression model. Equation (10) did not improve 2007 riparian ET predictions (RMSE = 0.60, Figure 7a) compared to Equation (6) (RMSE = 0.40), suggesting that a multiple linear regression model does not improve riparian ET at these sites. Equation (11), as expected, greatly improved 2007 upland site ET predictions (RMSE = 0.23, Figure 7b) compared to Equation (7) (RMSE = 0.46). Equation (11) also produced evenly distributed values along the 1:1 line and residuals were not skewed. Furthermore, this approach reduced RMSE at both upland sites suggesting that using multiple linear regression provides the best estimates of ET at water-limited sites.

Statistics were tabulated for all empirical ET models after parameterization (Table 3). The highest $r^2$ was 0.92 which shows that models in the form of Equation (1) and Equation (3) calibrated by riparian sites only are the best for estimating riparian ET. Upland site ET, on the other hand, is best estimated by using multiple linear regression with precipitation as an input calibrated with upland sites only ($r^2 = 0.77$). RMSE for all validations (i.e., 2007 ET model predictions vs. 2007 measured ET) were broken down
among sites (CM, LM, LS, KG, SR) and vegetation associations (riparian, upland) to observe site-specific trends (Table 4). Smaller RMSE values show the best riparian site (CM, LM, LS) ET predictions were made using a model in the form of Equation (3) while upland site (KG, SR) ET predictions were consistently better using a model in the form of Equation (8).

4. Discussion

4.1 Implications of including upland vegetation in current empirical models to estimate catchment-scale ET

Our modified empirical models that combine riparian and upland vegetation provide reasonable estimates of overall ET ($r^2 = 0.78, 0.85, 0.74$) and perform similar to other empirical models in which VIs were used in combination with temperature inputs to estimate ET in riparian zones [Nagler et al., 2005a ($r^2 = 0.82$); Nagler et al., 2005b ($r^2 = 0.76$); Nagler et al., 2009 ($r^2 = 0.80$); and Murray et al., 2009 ($r^2 = 0.80$)]. Our model represented by Equation (6) is a good predictor of riparian ET, specifically, because meteorological (e.g., temperature) and vegetation conditions (e.g., leaf area) control ET when soil moisture is not limiting [Shuttleworth, 1991]. Our riparian sites are composed of phreatophytic vegetation with extensive root systems that access groundwater reserves. Phreatophytic vegetation in this region is known to derive 80-100% of their transpiration water from groundwater and is minimally impacted by monsoon rain events [Snyder and Williams, 2000; Horton et al., 2001; Yepez et al., 2003]. This is supported by other
studies in the region where pre-monsoon physiological responses of phreatophytic vegetation such as transpiration did not change significantly when near surface soil moisture increased due to monsoon rainfall events during the growing season [Scott et al., 2003; Potts et al., 2006]. Because vegetation is not limited by soil moisture at these sites, the amount of available energy becomes the limiting factor driving ET during the growing season [Gazal, 2006; Williams et al., 2006; Moore and Heilman, 2011].

In contrast, upland vegetation often depends on pulses of water for productivity [Noy-Meir, 1973; Sala and Lauenroth, 1982; Scott et al., 2000; Huxman et al., 2004; Potts et al., 2006]. ET in arid and semiarid regions dominated by upland vegetation is not limited by available energy, but by near-surface soil moisture supplied by seasonal rainfall [Kurc and Small, 2004; Serrat-Capdevila et al., 2011; Moore and Heilman, 2011]. In fact, ET in upland ecosystems has been shown to rapidly increase following rainfall events [Sala and Lauenroth, 1982; Kurc and Small, 2004; Scott et al., 2006b], which in part explains how >90% of precipitation in semiarid regions can be returned to the atmosphere via ET [Wilcox et al., 2003]. This is not accounted for, however, in our modified empirical models represented by Equation (1) and (3) because ET is driven by increased temperature and greenness, independent of moisture. Consequently, these models over-predict overall ET along river catchments where upland vegetation encompasses the majority of pixels in the scene. While pixels characterized by upland vegetation may have much smaller EVI values, the temperatures are much higher than adjacent riparian-dominated pixels which may in part be responsible for biased predictions. Therefore, while models incorporating VIs and temperature may be
appropriate for riparian-dominated regions, they do not properly account for the main
drivers of ET in regions dominated by upland vegetation.

We conclude that our modified empirical models combining riparian and upland
sites will not be able to consistently or accurately estimate ET along river reaches or
basins, especially considering that the amount of upland vegetation varies greatly from
catchment to catchment and will continue to change due to desertification in semiarid
systems [Kerley and Whitford, 2000, Archer 1994]. Accounting for shifts in upland
vegetation are exceptionally important for watershed management because these shifts
can alter hydrology at both local and regional scales [Wilcox, 2002; Huxman et al.; 2005].
ET models that cannot effectively incorporate both riparian-influenced and water-limited
upland vegetation are not adequate for making appropriate basin-scale ET estimates.

4.2 Incorporating the sensitivity of upland sites to pulses of moisture in empirical models

ET is limited by soil moisture in arid and semiarid environments when
precipitation is much less than potential ET [Noy-Meir, 1973; Rodriguez-Iturbe, 2000].
Our research shows that ET in our upland sites is highly responsive to rainfall events and
this is supported by relatively high coefficients of determination, especially when
compared to riparian sites (Table 2). Other studies have documented rapid ET response
to rainfall events in upland vegetation and have indicated that near-surface soil moisture
(0-5 cm) from these events is the primary source of water for ET during the summer
monsoon [Sala and Lauenroth, 1982; Kurc and Small, 2004; Scott et al., 2006b].
Empirical ET models for upland ecosystems without access to groundwater must account for these rainfall events that provide plant available moisture at the surface.

As a first iteration, we developed an empirical model that incorporates precipitation as an input for estimating upland ET specifically. While EVI, as an indicator of greenness, captures the variation and contribution of transpiration to total ET, we make the assumption that precipitation accounts for rapid evaporation following rainfall events. To do this, we acquired precipitation data from tipping-bucket rainfall gauges installed at each upland flux tower site. A multiple linear regression approach was then used to create an empirical ET model assuming that EVI can represent total ET when precipitation is not present and that precipitation is additive when present. We eliminated the energy input because LST, did not improve the predictive power of the model. This multiple regression approach represented by Equation (11) predicted upland ET adequately ($r^2 = 0.77$) with similar performance to other studies [Nagler et al., 2007 ($r^2 = 0.74$)].

This approach can estimate water lost from upland vegetation along river reaches where weather stations or rain gauges are available. When precipitation data is not available, however, the ability to accurately estimate ET is not likely unless rainfall can be interpolated from nearby rainfall gauges or forecasted using a suitable method or model for predicting rainfall at the site of interest. Future advancements and accessibility of satellite-based remote sensing will improve models for estimating basin scale ET. The National Polar-orbiting Operational Environmental Satellite System (NPOESS) and Soil Moisture Active-Passive (SMAP) missions aim to improve weather forecasting and will
likely improve model predictability. While no remote sensing products are currently available to measure soil moisture, SMAP, scheduled to launch in 2014, will measure near-surface soil moisture at high spatial and temporal resolution. The capabilities of SMAP or other remotely-sensed soil moisture products will improve ET models by understanding how pulses of moisture from rainfall affect ET across different regions.

4.3 Integrating upland and riparian empirical models to estimate catchment-scale ET

While a single predictive equation is desired to estimate ET at basin scales, this research suggests that differences in plant function (e.g., the ability or inability to access and use groundwater) and ecohydrology introduce complications that need to be addressed to improve ET estimates. However, an approach using multiple models could estimate ET along riparian reaches if suitable vegetation maps were available or created. Independent ET models appropriate for each vegetation association could then be applied to each land surface classification. Unland et al. [1998], for example, estimated ET from a riparian system in a semiarid environment by classifying five different surfaces and attributing multiple approaches to best estimate ET for each surface. A similar approach using multiple empirical ET models to represent the different riparian and upland vegetation classified along a river reach could be used to improve overall ET estimates along the entire reach or catchment.

This approach requires adequate flux tower data, both spatially and temporally, to calibrate models for each vegetation association and assumes that classifying vegetation along a mosaic corridor into riparian-influenced versus upland, water-limited vegetation
is feasible and sufficient for documenting all vegetation within the river reach. By combining functional groups, this method follows the assumption observed in many studies that riparian vegetation such as cottonwood, willow, tamarisk, and mesquite have similar transpiration rates on a leaf area basis and therefore could be combined into one riparian class [Anderson, 1982; Busch and Smith, 1995; Sala et al., 1996, Glenn et al., 1998; Nagler et al., 2008]. Accuracy would be improved if riparian classes were broken into species-specific communities (i.e. cottonwood/willow, tamarisk, mesquite), but would require additional flux towers to provide patch-specific or monoculture ET data for calibration as well as remote sensing products with high enough spatial resolution to discriminate between species. If necessary, a combination using empirical approaches and other methods (e.g., sap flux, crop coefficient approaches, etc.) for each riparian class could result in the most appropriate estimates of river reach ET.

4.4 Water resource applications of empirical ET models

Seasonal riparian ET is difficult to estimate over large areas [Nagler et al., 2005; Devitt et al., 2010], but is often a significant factor in the basin water balance in semiarid regions [Goodrich et al., 2000; Maddock et al., 2000; Dahm et al., 2002]. Furthermore, river managers need accurate estimates of consumptive water use (ET) by crops and riparian species spanning large areas to manage water supplies for human use, primarily agriculture, and environmental restoration [USBOR, 1997; LCR-MSCP, 2004; Scott et al., 2006c; Trezza, 2006; Murray et al., 2009]. Improved estimates of riparian ET and its seasonal distribution are also necessary to improve regional groundwater models for
near-term management tools [Goodrich et al., 2000]. Empirical ET models are valuable for their ability to use VIs over a wide range of temporal and spatial resolutions for scaling local or patch-scale ET measurements to river-reach and catchment scales [Glenn et al., 2010]. Using a combination of multiple flux towers and remote sensing products can produce models that account for temporal and spatial heterogeneity in vegetation along riparian zones. Growing availability and accessibility of data from eddy flux networks and improvements in remote sensing products will further increase spatial and temporal resolutions needed to produce more accurate and near, real-time estimates of ET in the future.

A degree of uncertainty, however, is inherent when modeling river-reach to basin-scale ET due to errors propagating through subsequent steps [Devitt et al.; 2010]. Due to the empirical nature of the approach represented here, we caution that this method requires model re-calibration using accurate ground measurements of ET representing the vegetation captured in the satellite products. This study used free MODIS products to avoid pre-processing raw satellite data. While MODIS data is available almost daily, the available ORNL-DAAC composites used in this research reduce the temporal resolution to 8 or 16-day products. ET models can be improved by using remotely sensed products with better temporal resolution. Likewise, remote sensing products with finer spatial resolution (e.g., Landsat, 15 to 30 m) would improve empirical methods of modeling ET, but may sacrifice the temporal resolution necessary to account for seasonal fluctuations in ET. With the advent of longer-term in situ ET measurements and high-frequency repeat-
overpass remote-sensing data, there is great potential for improving empirical methods for estimating ET in the future.

Acknowledgements

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References


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Table Captions

Table 1. Locations and Descriptions for Five Study Sites

Table 2. Relationships of Evapotranspiration to Selected Variablesa

Table 3. Evapotranspiration Model Statistics Reported after Parameterizationa

Table 4. Root Mean Square Error (RMSE) Reported for each Modela
Figure Captions

Figure 1. Locations of the five study sites, including one in the Santa Cruz River Watershed (Santa Rita Mesquite Savannah, SR) and four in the San Pedro River Watershed (Charleston Mesquite Woodland, CM; Lewis Springs Mesquite, LM; and Lewis Springs Sacaton, LS); and Kendall Grassland, KG) in Arizona, USA.

Figure 2. Predicted 2007 ET (\( ET_{\text{predicted}} \)) versus measured ET (\( ET_{\text{actual}} \)) plotted for all sites along the 1:1 line by calibrating Equation (1) using all sites combined to yield Equation (2). Locations include three riparian influenced sites (filled symbols): Charleston Mesquite Woodland (CM), Lewis Springs Mesquite (LM), and Lewis Springs Sacaton (LS); and two upland, water-limited sites (hollow symbols): Kendall Grassland (KG) and Santa Rita Mesquite Savannah (SR).

Figure 3. Precipitation and evapotranspiration (ET) plotted over time for all study sites [Charleston Mesquite Woodland (CM), Lewis Springs Mesquite (LM), and Lewis Springs Sacaton (LS), Kendall Grassland (KG), and Santa Rita Mesquite Savannah (SR)] during a 3-yr period. Precipitation was summed over 16-day windows measured using tipping-bucket rain gauges at each flux tower site and ET was calculated using the eddy covariance technique and averaged over each 16-day window.
Figure 4. Predicted 2007 ET (ET\textsubscript{predicted}) versus measured ET (ET\textsubscript{actual}) plotted for all sites along the 1:1 line by calibrating Equation (3) using all sites combined to yield Equation (5). Locations include three riparian influenced sites (filled symbols): Charleston Mesquite Woodland (CM), Lewis Springs Mesquite (LM), and Lewis Springs Sacaton (LS); and two upland, water-limited sites (hollow symbols): Kendall Grassland (KG) and Santa Rita Mesquite Savannah (SR).

Figure 5. Predicted 2007 ET (ET\textsubscript{predicted}) versus measured ET (ET\textsubscript{actual}) plotted along the 1:1 line by calibrating Equation (3) using: 5a) riparian sites only to yield Equation (6), and 5b) upland sites only to yield Equation (7). Locations include three riparian influenced sites (filled symbols above): Charleston Mesquite Woodland (CM), Lewis Springs Mesquite (LM), and Lewis Springs Sacaton (LS); and two upland, water-limited sites (hollow symbols below): Kendall Grassland (KG) and Santa Rita Mesquite Savannah (SR).

Figure 6. Predicted 2007 ET (ET\textsubscript{predicted}) versus measured ET (ET\textsubscript{actual}) plotted for all sites along the 1:1 line by calibrating Equation (8) using all sites combined to yield Equation (9). Locations include three riparian influenced sites (filled symbols): Charleston Mesquite Woodland (CM), Lewis Springs Mesquite (LM), and Lewis Springs Sacaton (LS); and two upland, water-limited sites (hollow symbols): Kendall Grassland (KG) and Santa Rita Mesquite Savannah (SR).
Figure 7. Predicted 2007 ET ($ET_{predicted}$) versus measured ET ($ET_{actual}$) plotted along the 1:1 line by calibrating Equation (8) using: 7a) riparian sites only to yield Equation (10), and 7b) upland sites only to yield Equation (11). Locations include three riparian influenced sites (filled symbols above): Charleston Mesquite Woodland (CM), Lewis Springs Mesquite (LM), and Lewis Springs Sacaton (LS); and two upland, water-limited sites (hollow symbols below): Kendall Grassland (KG) and Santa Rita Mesquite Savannah (SR).
Tables

Table 1.

<table>
<thead>
<tr>
<th>Site</th>
<th>Long. (W)</th>
<th>Lat. (N)</th>
<th>Elev. (m)</th>
<th>MAP&lt;sup&gt;a&lt;/sup&gt; (mm)</th>
<th>Dist. to San Pedro River (km)</th>
<th>Dominant Veg. Type and Species</th>
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<sup>a</sup>Mean annual precipitation (MAP)
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2003 0.85 0.79 0.90 0.67 0.86 0.51 0.54 -0.22 0.58 0.21 1.00 -0.38 0.43 0.45 0.56
2004 0.84 0.79 0.64 0.63 0.65 0.01 0.42 -0.34 0.73 0.47 1.00 -0.03 0.46 0.30 0.54
2005 0.85 0.82 0.66 0.45 0.62 0.52 0.29 -0.41 0.64 0.20 1.00 -0.30 0.30 0.22 0.51
2006 0.86 0.81 0.71 0.43 0.65 0.59 0.16 -0.42 0.66 0.12 1.00 -0.37 0.03 0.15 0.47
2007 0.89 0.83 0.68 0.50 0.65 0.46 0.31 -0.42 0.70 0.32 1.00 -0.27 0.26 0.24 0.53

*a* A multivariate analysis reported coefficients of determination (r^2) for ET versus selected independent variables: enhanced vegetation index (EVI), normalized difference vegetation index (NDVI), nighttime land surface temperature (LST_n), daytime land surface temperature (LST_d), average daily temperature (T_avg), precipitation (PPT), vapor pressure deficit (VPD), wind speed (W_spd), net radiation (R_n), ground heat flux (G), latent heat flux (LE), sensible heat flux (H), incoming shortwave radiation (SW_inc), potential ET computed by The Arizona Meteorological Network (ET_oAZMET), and potential ET computed using the Shuttleworth approach (ET_oSW). Regressions were analyzed to determine which variables explained most of the variation in ET among each site while regressions across each year determined overall variations among years during the study period.
Table 3.

<table>
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<tr>
<th>Training Set</th>
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</table>

*a* Each model was trained by all five sites combined (ALL), riparian sites only (CM, LM, LS), and upland sites only (KG, SR).

*b* Original *Scott et al.* [2008] model

*c* Modified multiplicative model

*d* Multiple linear regression model
Table 4.

<table>
<thead>
<tr>
<th>Training Set</th>
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<th>Eq. (5) Model&lt;sup&gt;c&lt;/sup&gt;</th>
<th>Eq. (9) Model&lt;sup&gt;d&lt;/sup&gt;</th>
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</table>

<sup>a</sup>RMSE was used to determine overall closeness of fit where lower values represent empirical ET models that provide the closest 2007 ET predictions compared to measured ET.

<sup>b</sup>Original Scott et al. [2008] model

<sup>c</sup>Modified multiplicative model

<sup>d</sup>Multiple linear regression model
Figures

Figure 1.
Figure 2.
Figure 3.
Figure 5.
Figure 6.
Figure 7.