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Estimating phreatophyte evapotranspiration from Diel groundwater fluctuations in the middle Rio Grande Bosque

Christian Gunning

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Estimating Phreatophyte Evapotranspiration from Diel Groundwater Fluctuations in the Middle Rio Grande Bosque

by

Christian Gunning

B.S, University of Georgia, 2001

THESIS

Submitted in Partial Fulfillment of the Requirements for the Degree of Master of Science Water Resources

The University of New Mexico

Albuquerque, New Mexico

April, 2010
Dedication

To a long line of teachers, all of whom have been more patient than any reasonable person has the right to expect. I hope it was worth the effort.
Acknowledgments

First and foremost, I want to extend my deepest gratitude to Dr. Roy Jemison, Regional Hydrologist, USDA Forest Service, Southwestern Division, formerly of USDA Rocky Mountain Research Station, for giving me the opportunity to work on this project. He’s provided invaluable guidance throughout. I’ve tried his patience and found it ample, if not limitless. Roy has been a mentor in the truest sense. He has prodded, asked incisive questions, and provided much-needed perspective, all in appropriate doses.

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In my years at UNM, the Water Resources Program has fostered an warm, collegial environment for hands-on learning and experimentation. Dr. Bruce Thomson, director of the program and someone who seemingly aspires to be pretty darn good at most everything, is largely responsible for this collegial environment, and for the completion of this project.

Dr. James Cleverly’s insights into riparian hydroecology have been both focused and congenial. I’ve found working with him a great and educational pleasure.

To my wonderful colleagues in the field, thank you. Lynda Price, Jordan Wollak, and Nichole Carnevale, I’d have lost my mind long ago if it weren’t for your extensive help and more extensive conversations.

This work involved multiple agencies, along with change-overs in investigators and administration. Many people patiently answered my many questions. Without their assistance, I would not have been able to navigate the requisite maze of grant paperwork and technical reports. Dr. Julie Coonrod, Dr. John Stormont, Dr. Cliff Crawford, Dr. Bill Flemming, Dr. Cliff Dahm, Dr. D. Max Smith, Dr. Kim Eichhorst, Annamarie Cordova, James Thibault, and Jen Schuetz, thanks for all the answers.
Estimating Phreatophyte Evapotranspiration from Diel Groundwater Fluctuations in the Middle Rio Grande Bosque

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Christian Gunning

B.S, University of Georgia, 2001
M.S., Water Resources, University of New Mexico, 2010

Abstract

Throughout the Southwest, non-native phreatophytes such as saltcedar have rapidly replaced native cottonwoods along river corridors. The USDA Forest Service (USFS) Fuels Reduction Study (FRS) is an effort to quantify the effects of the removal of non-native riparian riparian vegetation on multiple parameters of riparian ecology, including groundwater variation over time. Here, 8 years of measurements from 12 wells at 5 research sites in the Middle Rio Grande riparian corridor (bosque) of Albuquerque area are considered. Of these research sites, 4 were cleared of non-native vegetation, 2 experienced wildfires, and 1 was a control.

This professional project quantified the connection between river flow and groundwater level at each well using ordinary least squares (OLS) regression of groundwater level onto river flow. Groundwater at each well was strongly influenced by river flow. Pronounced hysteresis and inter-well variability were observed.

Phreatophyte consumption of groundwater via evapotranspiration (ET) constitutes a significant use of available water resources in arid riparian ecosystems.
Phreatophyte-induced diel groundwater fluctuations in unconfined, shallow aquifers have been used to estimate well-specific ET since the introduction of the White method in 1932 (White, 1932). This professional project estimated well-specific ET from diel groundwater fluctuations using the White method. Modest reductions in ET were observed following treatment, and cessation in ET was observed immediately following wildfire. White method ET from 4 wells at one site was compared with on-site eddy covariance ET estimates. Despite the high variance of the White method, estimates from the two methods match closely when averaged over long time periods.
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Chapter 1

Introduction

The USDA Forest Service (USFS) Fuels Reduction Study (FRS) is an effort to quantify the effects of the removal of non-native riparian vegetation on multiple parameters of riparian ecology, including groundwater variation over time. Numerous sites have burned since the study began, allowing for a consideration of the effects of fire on the measured parameters (Finch et al., 2003; Smith et al., 2009).

In December of 2005, the University of New Mexico (UNM) Water Resources Program (WRP) entered into a joint venture with the USDA Forest Service Rocky Mountain Research Station (RMRS) to provide continuing maintenance, data collection, and analysis for the groundwater portion of the FRS, as seen in Price (2008). Groundwater levels were monitored at research plots at numerous phreatophyte-dominated sites in the riparian corridor of the Middle Rio Grande (MRG), also known as the bosque (MRGB). Through this joint venture, FRS groundwater data have been distributed to regional stakeholders by request. These data have assisted in the selection of appropriate sites for restoration activities that are sensitive to groundwater levels, such as pole planting of cottonwoods.

This professional project examines 8 years of measurements from 8 wells at 5 research sites in the Albuquerque area. Analysis was conducted in 3 stages. First, the connection between river flow and groundwater level at each site was quantified. Second, site-specific evapotranspiration (ET) was estimated from diel groundwater
fluctuations. Third, ET estimates were compared with eddy covariance estimates at one site.

Phreatophyte consumption of groundwater via ET constitutes a significant use of water resources in the MRG (Dahm et al., 2002). Accurately accounting for this use can aid in constructing accurate regional water budgets, and in the equitable allocation of scarce water resources. Early attempts at quantifying riparian ET on the Pecos River, for example, led to a costly and historic legal battle between New Mexico and Texas after saltcedar eradication failed to yield any of the expected “salvaged water” as measured by an increase in base flow (Frederick, 1993). Likewise, a more recent large-scale saltcedar eradication program in Western Texas yielded no observable increases in river flow (Hart et al., 2005). A review of 2 eradication efforts indicated that “water salvage was lower than expected based on previous ET estimates of saltcedar and other riparian phreatophytes”, and that revegetation strongly influences long-term water salvage (Shafroth et al., 2005).

Phreatophyte-induced diel fluctuations in unconfined shallow aquifers have been used to estimate ET since the introduction of the White method (White, 1932). Recent work by Butler et al. (2007), Gribovszki et al. (2008), and Loheide et al. (2005) have refined estimates by accounting for meteorological, plant, and soil variations. Much work on diel groundwater variations to date has been conducted in hydrological settings with low spatial and temporal variation. Butler et al. (2007), for example, studied a site adjacent to the Arkansas River in Kansas that experiences some effects from upstream pumping and precipitation, yet experienced more than two weeks of undisturbed hydrology, increasing confidence in the quality of diel variation measurements. Observing groundwater fluctuations is inexpensive compared to other, more direct ET sensing methods, such as eddy covariance. Yet considerable uncertainty remains in estimating ET through diel fluctuations of groundwater, especially in highly variable hydrological regimes.

Declining groundwater levels in riparian corridors can have profound ecological consequences (Rood et al., 2000; Williams and Cooper, 2005; Stromberg et al., 1993). Cottonwood canopies require a minimum depth to groundwater for survival,
and rapid groundwater declines cause high cottonwood mortality (Scott, 1999). Decreasing groundwater below this minimum level could radically change the canopy structure of the MRGB and the fauna that depend upon this canopy. Restoration efforts in the MRGB to date have focused on maintaining native cottonwood forests. The long-term success of these restoration efforts and the MRGB at large depends upon a clear understanding of interactions between stream flow, groundwater levels, phreatophyte ET, overbank flooding, and fire frequency in the MRGB.
Chapter 2

Background

2.1 Surface water and groundwater interactions in the Middle Rio Grande

The location of human and ecological communities in the Southwest have historically been limited by surface water availability. Diverse human and ecological communities cluster around riparian habitats, where available water greatly exceeds yearly precipitation (Crawford et al., 1993). Historically, legal fights over groundwater allocation in the Southwest have lagged behind fights over surface water (Clark, 1987). This is due in part to the large amount of energy and expertise required to extract economically significant quantities of groundwater, and to delayed scientific understanding of groundwater flows compared to groundwater (Kanazawa, 2003). Throughout the 20th century, groundwater has come under increased scrutiny, and linkages between surface water and groundwater are now carefully considered by stakeholders and policy makers.

In the MRG, five primary groups of surface water stakeholders and their associated uses are generally recognized for administrative and legal purposes: ecological, agricultural, indigenous, urban, and downstream. A complex legal framework allocates water between these stakeholders. Ecological communities, particularly riparian
communities, include a number of endangered species that are protected under the Endangered Species Act, including the southwestern willow flycatcher (*Empidonax traillii extimus*) (Finch and Stoleson, 2000) and the Rio Grande silvery minnow (*Hybognathus amarus*) (Bestgen and Platania, 1991). The indigenous and agricultural communities generally hold the most senior rights under the doctrine of prior appropriation (also known as “first in time, first in right”) (Milliman, 1959). Urban communities have experienced the largest growth in water demand in modern times and, despite their less senior water rights, can often pay higher prices for water in open markets. A summary of New Mexico water market transfers found that 96% of purchases were made by municipalities (Brookshire et al., 2004). This market-based allocation of water resources to urban areas has coined the euphemism that “water flows uphill towards money”. Finally, an interstate compact guarantees water delivery to Texas (Hill, 1974), and a treaty guarantees water delivery to Mexico (Mumme, 1993).

Throughout the world, human demand for and use of groundwater resources have grown rapidly, along with associated complications, including water shortages and water quality degradation (Shah et al., 2003; Villholth, 2006). Groundwater resources are frequently and increasingly extracted from aquifers by humans at rates that greatly exceed recharge. Throughout the arid Southwest, including the MRG, deep aquifer water resources have been increasingly exploited for human uses (Bexfield and Anderholm, 2002). Albuquerque’s waste water treatment plant has been a major tributary to the Rio Grande (Cleverly et al., 2002). Overall, deep aquifer water has supplemented in-stream river flows. These supplemental river flows have aided endangered species and consequently limited Endangered Species Act (ESA) lawsuits (Robert, 2005). In addition, these supplemental flows have aided New Mexico’s fulfillment of compact-required Rio Grande water delivery to Texas and Mexico (Booker et al., 2005; Hill, 1974).

In response to groundwater depletions, Albuquerque has begun diverting water from the Rio Grande (Bexfield and McAda, 2003; Authority, 2009a), increasing contention for already scarce surface water. The city’s switch from groundwater to

surface water will potentially reduce net water flux from the deep aquifer into the river in the short-term. Increased use of reclaimed wastewater on city landscaping such as parks and golf courses (Authority, 2009b) will further decrease return flows into the Rio Grande.

In the MRG, surface water and groundwater are coupled via numerous pathways. The human-utilized deep aquifer is loosely coupled to surface water and shallow groundwater resources due to vertical anisotropy (Bexfield and McAda, 2003), such that changes in deep aquifer levels propagate slowly into shallow aquifer dynamics, and vise versa. On the other hand, surface water and the shallow riparian aquifer are tightly coupled. The MRG contains a network of irrigation canals and ditches where surface water can rapidly infiltrate into the shallow aquifer, providing groundwater recharge. Water also leaks out of the shallow aquifer into riverside drains, whence it flows into the Rio Grande. Illustrating these connections, Bexfield and McAda (2003) estimate that in the summer of 1999, 33% of total groundwater recharge (slightly more than 200 cfs) came from “river leakage” and 31% of total groundwater outflow (slightly less than 200 cfs) flowed into irrigation drains, showing an approximate balance of flows between surface and shallow groundwater at that time.

2.2 Methods of estimating riparian ET

In the MRG, riparian vegetation relies primarily on shallow groundwater. Recent estimates place total riparian ET at 20–33% of total regional river depletions (Dahm et al., 2002). Accurately quantifying riparian ET is now a major research focus, and a number of methods have been used to measure riparian ET at a variety of scales. Most recently, remote sensing techniques have been employed to estimate riparian ET over large areas via Landsat imagery, Enhanced Vegetation Index (EVI), and Normalized Difference Vegetation Index (NDVI) (McDonnell et al., 2002; Nagler et al., 2005). Eddy covariance has been used to measure site-level ET under a variety of plant regimes (Cleverly et al., 2006). Sap flow measurements have been employed to measure plant-specific ET estimates, and is often used as a control against other
measurements (Nagler et al., 2003; Moore et al., 2008).

Estimating phreatophyte water use from diel groundwater fluctuations was first employed by White (1932) using the assumption in equation 2.1,

$$\frac{Sdh}{dt} = Q_{net} - ET$$

(2.1)

where $S$ [unitless] is storativity, $dh/dt$ [$LT^{-1}$] is the rate of change of groundwater elevation, $Q_{net}$ [$LT^{-1}$] is the groundwater recharge rate, and $ET$ [$LT^{-1}$] is the rate of water loss to phreatophyte evapotranspiration. Equation 2.2 shows the relationship of storativity to specific storage ($S_s$ [unitless]) to specific yield ($S_y$ [unitless]), where $h$ [$L$] is the groundwater elevation.

$$S = S_y + S_s \times h$$

(2.2)

In an unconfined aquifer, $S_y \gg S_s$, and thus $S \simeq S_y$. The White method assumes that $Q_{net}$ is constant, yielding equation 2.3,

$$ET = \hat{S}_y \times (24r - s)$$

(2.3)

where $\hat{S}_y$ [unitless] is the readily available specific yield of the sediment (i.e. $S_y$ over short time periods), $r$ [$LT^{-1}$] is the tangent to the rising limb of one day’s diel groundwater fluctuation, and $s$ [$L$] is the inter-day change in groundwater level, or storage. Butler et al. (2007) investigated this method at length, comparing White method ET estimates with estimates based on sap flow, noting that White method estimates provide spatial integration yet are unreliable in fine-grained soils. $\hat{S}_y$ [unitless] is a major source of uncertainty, and is difficult to measure in situ (Loheide et al., 2005). Recently, Martinet et al. (2009) compared White method ET estimates to eddy covariance ET estimates and found that estimate agreement was higher at sites with greater depths to groundwater, reduced groundwater-river connectivity, and coarser-grained soils. Gribovszki et al. (2008) proposed several modifications to the White method that include a variable $Q_{net}$ estimated from the maximum and minimum $dh/dt$, or from a background monitoring well.

The White method can only be employed to measure daytime groundwater consumption from unconfined aquifers. This presents a number of potential problems.
Chapter 2. Background

Nighttime ET is unaccounted for, as well as water loss from the soil surface and the vadose zone. Arid riparian corridors commonly contain primarily phreatophyte vegetation in unconfined aquifers, though nighttime ET remains a significant question.

2.3 Ecological disturbance: native vs. non-native vegetation

Throughout the Southwest, riparian hydrology and ecology have been transformed by anthropic impacts. Dams, irrigation canals and other structures have altered the frequency, duration, and intensity of floods, and changed base flow patterns (Crawford et al., 1993). Cottonwoods depend upon overbank flooding for reproduction. Growth of cottonwoods in alluvial floodplains is strongly correlated with streamflow (Stromberg and Patten, 1996), and floodplain water table declines have been shown to cause high mortality in native cottonwood communities (Scott, 1999; Williams and Cooper, 2005; Lite and Stromberg, 2005). Consequently, non-native phreatophytes including Russian olive (Elaeagnus angustifolia) and saltcedar (Tamarix pentandra) now dominate areas formerly inhabited by flood-adapted native cottonwoods and willows (Busch and Smith, 1995), largely as a result of human-altered flow regimes (Zouhar, 2003).

The comparative benefits, risks, and costs of removing non-native vegetation have been debated in numerous forums (Dudley and DeLoach, 2004; Merritt et al., 2009). Non-native vegetation, particularly saltcedar, is associated with high frequency and intensity of uncontrolled fires in the riparian forest. Such high-intensity fires cause high mortality among flood-adapted native vegetation while invasive vegetation regrows quickly, leading rapidly to a fire-dominated ecosystem (Busch, 1995; Smith et al., 2006). Prevention of fire is a dominant concern driving non-native vegetation removal, as evidenced by the FRS. Nonetheless, non-native vegetation has been shown to provide suitable habitat for many endemic species (Ellis, 1995), including the endangered Willow Flycatcher (Finch and Stoleson, 2000).
Chapter 2. Background

Numerous stakeholders have asserted the intrinsic value of a bosque dominated by native vegetation. The instream flows that support riparian ecosystems in the MRG were found to be valued by the region’s residents in a phone surveys (Berrens et al., 1996). Numerous conservation and restoration projects have worked at restoring native riparian ecosystem dynamics (Molles Jr. et al., 1998; Crawford et al., 1993), while several lawsuits have been filed, for example to preserve in-stream flows for endangered species health (Robert, 2005).

This study examines groundwater trends across relatively small spatial and temporal scales. Groundwater is a key driver of MRGB ecology, and changes in groundwater hydrology have been very rapid in recent years. As such, this study provides useful insight into the future of MRGB vegetation, and to the ecosystem at large.
Chapter 3

Methods

3.1 Study sites and treatment descriptions

Twelve wells at five different study sites were employed. All of the study sites are located in Albuquerque’s South Valley between Middle Rio Grande Conservancy District (MRGCD) riverside drains and the river channel (Figures 3.1 and 3.2). The predominant hydraulic gradient is from the river channel into the MRGCD riverside drains. River and riverside drain flows are highly variable across space and time (see Figure 4.3). River flow typically peaks in during spring following up-gradient snowmelt (see Figures 3.3 and 4.2). A second river flow peak often occurs in late summer during the North American monsoon. The large drainage area of the MRG means that regional hydrological events including precipitation, snow melt, and dam releases effect local hydrology.
Chapter 3. Methods

Figure 3.1: Overview map of study area

Figure 3.2: Overview map of groundwater monitoring wells and USGS river gage
Chapter 3. Methods

Figure 3.3: Rio Grande USGS gage #08330000 at Albuquerque, NM. Daily mean flow (cfs) from 1942 to present

At the four FRS study sites, two wells were installed per site, one each at the northern and southern ends of the site. Each well-pair was separated by between 350 and 700 meters. In addition, the UNM Biology Department operates a tower that measures evapotranspiration via eddy covariance in the close proximity of the FRS sites. This tower also collects micrometeorological measurements. The UNM site contains 5 groundwater wells – a center well at the tower and 4 additional wells 40 meters north, south, east and west of the tower. This site is located on the east side of the river. Usable data was available for all but the east well.

Each well was given a label according to the following scheme:

- One letter indicating direction from river: East (E) or West (W);

- Three letters indicating the treatment category: Control (CON), chipping followed by planting of native shrubs (PLT), chipping alone (CHP), stacking (STK), and tower site (TOW);

- One letter indicating the relative direction of the well compared to the tower: North (N), South (S), East (E), West (W), and Center (C).
Chapter 3. Methods

Treatment descriptions, dates, and areas are enumerated in Table 3.1.

At all FRS treatment sites, all non-native shrubs and trees were removed, and herbicide was applied to saltcedar stems. Each treatment site received one of three additional treatments: cut wood was chipped and spread across the site, followed by planting of native shrubs (W.PLT); cut wood was chipped and spread across the site (W.CHP); or cut wood was stacked (E.STK). At these sites, a small amount of pre-treatment groundwater data were gathered. Pre-treatment control data are especially limited. No treatment was applied to the FRS control site.

The tower sites (E.TOW) experienced the same treatment as the CHP sites, with the exception that treatment was spread across 2 years. An eddy covariance tower is sited above E.TOW.C. The remaining wells are 40 meters in each cardinal direction from the tower. The close proximity of the TOW wells to the eddy covariance tower allowed comparison of White method with eddy covariance ET estimates at this site.

The TOW sites and W.CHP.S experienced a wildfire during the project, in June of 2006. W.CHP.S appears to have experienced much more intense temperatures, and vegetation loss was complete. The TOW sites, on the other hand, burned patchily and suffered only limited cottonwood mortality.

3.2 Site vegetation and hydrology

Substantial variation exists between well sites in vegetation, soil texture, location with respect to the river channel, levees, and irrigation ditches. Well labels and post-treatment vegetation descriptions are shown in Table 3.2.

Study site vegetation is predominantly cottonwoods (Populus spp.) of various sizes and ages, along with Russian olive (Elaeagnus angustifolia), saltcedar (Tamarix spp.), willow (Salix spp.), and tree of heaven (Ailanthus glandulosa). The W.CON well sites have well-developed cottonwood canopies, significant saltcedar understories, and high woody fuel loads (e.g. Figure 3.5). Treatment sites with intact canopies have
### Table 3.1: List of study site treatments

<table>
<thead>
<tr>
<th>Treatment Abbreviation</th>
<th>Treatment Description</th>
<th>Mechanical Removal Dates</th>
<th>Area Treated (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>W.CON</td>
<td>Control – No treatment</td>
<td>None</td>
<td>16.07</td>
</tr>
<tr>
<td>W.PLT</td>
<td>Mechanical removal and chipping, herbicide, and planting of native shrubs</td>
<td>Nov 2002 - Nov 2003</td>
<td>18.33</td>
</tr>
<tr>
<td>W.CHP</td>
<td>Mechanical removal and chipping, herbicide</td>
<td>Apr 2003 - Apr 2004</td>
<td>17.03</td>
</tr>
<tr>
<td>E.STK</td>
<td>Mechanical removal and stacking with later removal, herbicide</td>
<td>Nov 2002 - Apr 2003</td>
<td>23.43</td>
</tr>
<tr>
<td>E.TOW</td>
<td>Mechanical removal and chipping, herbicide</td>
<td>Aug 2002 - Oct 2002 (North), - May 2004 (South)</td>
<td>No Data</td>
</tr>
</tbody>
</table>

an open, park-like under-canopies (e.g. Figure 3.10). Vegetation at W.CHP.S post-wildfire consists primarily of fast-growing annuals that depend upon precipitation-derived soil moisture (e.g. Figure 3.14). The same fire burned the E.TOW sites to a lesser extent.

Canopy cover surveys were conducted at one circular 375 m$^2$ plot for each FRS study site. Each vegetation plot was between the study site’s two wells. Since vegetation on each study site is heterogeneous, the surveys are not necessarily representative of the vegetation at each well. In particular, the post-treatment canopy cover at E.TOW.W is approximately 100%, while E.TOW.S has 0% canopy cover. The results are shown in Table 3.3.

All wells are between 35 and 235 meters from the Rio Grande river bank as indicated by USGS National Hydrography Dataset Area layer (U.S. Geological Survey, 2008a). The true location of the river channel is highly variable from year to year as the river channel shifts. In addition, higher river flows increase the channel width and lower the distance between each well and the river channel. Figures 3.2, 3.4, 3.7, 3.9, 3.11, 3.13, 3.15, and 3.17 show the close proximity of the river channel and the
Table 3.2: List of all well names and well site vegetation

<table>
<thead>
<tr>
<th>Well ID</th>
<th>Upper Canopy Vegetation</th>
<th>Understory Vegetation</th>
</tr>
</thead>
<tbody>
<tr>
<td>W.CON.N</td>
<td>Old Cottonwood</td>
<td>Saltcedar, Russian olive</td>
</tr>
<tr>
<td>W.CON.S</td>
<td>Old Cottonwood</td>
<td>Saltcedar, Russian olive</td>
</tr>
<tr>
<td>W.PLT.N</td>
<td>Mature Cottonwood</td>
<td>Elm, kochia, Russian thistle</td>
</tr>
<tr>
<td>W.PLT.S</td>
<td>Mature cottonwood</td>
<td>Willow, tree of heaven, grasses and annuals</td>
</tr>
<tr>
<td>W.CHP.N</td>
<td>Mature Cottonwood</td>
<td>Primarily open, some elm</td>
</tr>
<tr>
<td>W.CHP.S,</td>
<td>Mature Cottonwood</td>
<td>No data</td>
</tr>
<tr>
<td>pre-fire</td>
<td></td>
<td></td>
</tr>
<tr>
<td>W.CHP.S,</td>
<td>None</td>
<td>Kochia, Russian thistle</td>
</tr>
<tr>
<td>post-fire</td>
<td></td>
<td></td>
</tr>
<tr>
<td>E.STK.N</td>
<td>None</td>
<td>Kochia, Russian thistle</td>
</tr>
<tr>
<td>E.STK.S</td>
<td>Old Cottonwood</td>
<td>Primarily open, some tree of heaven</td>
</tr>
<tr>
<td>E.TOW,</td>
<td>Mature Cottonwood</td>
<td>Saltcedar, Russian olive</td>
</tr>
<tr>
<td>all wells</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Table 3.3: Vegetation survey of FRS sites. Note that the E.STK survey was conducted after clearing. All other surveys were conducted before clearing.

<table>
<thead>
<tr>
<th>Site</th>
<th>Survey Date</th>
<th>Canopy Cover (%)</th>
<th>Cottonwood Canopy Cover (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>W.CON</td>
<td>2002-08-26</td>
<td>108</td>
<td>74</td>
</tr>
<tr>
<td>W.PLT</td>
<td>2002-08-25</td>
<td>77</td>
<td>63</td>
</tr>
<tr>
<td>W.CHP</td>
<td>2002-08-24</td>
<td>125</td>
<td>90</td>
</tr>
<tr>
<td>E.STK</td>
<td>2004-05-11</td>
<td>75</td>
<td>75</td>
</tr>
</tbody>
</table>

MRGCD irrigation drain to the monitoring wells (for detailed maps of the MRGCD system see District (2004)). Connection between groundwater and irrigation structures was not investigated due to limited availability of data.

Aerial photographs were obtained for each study site via Google Earth (Google, 2009). Each photograph is viewed from an elevation of approximately 1,500 meters with north facing up and with comparable scales. All aerial photographs are from 22 June, 2005.
Figure 3.4: Aerial photograph – control wells – W.CON.N and W.CON.S

Figure 3.5: Site photograph – control well – W.CON.N
Figure 3.6: Site photograph – control well – W.CON.S
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Figure 3.7: Aerial photograph – mechanical removal with chipping and revegetation – W.PLT.N

Figure 3.8: Site photograph – mechanical removal with chipping and revegetation – W.PLT.N
Figure 3.9: Aerial photograph – mechanical removal with chipping and revegetation – W.PLT.S

Figure 3.10: Site photograph – mechanical removal with chipping and revegetation – W.PLT.S
Chapter 3. Methods

Figure 3.11: Aerial photograph – mechanical removal with chipping – W.CHP.N and TOW wells

Figure 3.12: Site photograph – mechanical removal with chipping – W.CHP.N
Figure 3.13: Aerial photograph – mechanical removal with chipping – W.CHP.S

Figure 3.14: Site photograph – mechanical removal with chipping – W.CHP.S
Figure 3.15: Aerial photograph – mechanical removal with stacking – E.STK.N

Figure 3.16: Site photograph – mechanical removal with stacking – E.STK.N
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Figure 3.17: Aerial photograph – mechanical removal with stacking – E.STK.S

Figure 3.18: Site photograph – mechanical removal with stacking – E.STK.S
Figure 3.19: Site photograph – TOW wells – pre-treatment

Figure 3.20: Site photograph – TOW wells – post-treatment
3.3 Well construction and groundwater sampling

At each FRS well, a hole was manually bored and a steel well-casing was pounded in so that 0.8 meters protruded above the soil surface. Each well casing has an interior diameter of 5.08 cm (2 inches) and an external diameter of 6 cm (2.38 inches). Each well extends approximately 4 meters below the soil surface and is screened across the bottom 1.52 meters of the casing with 0.5 mm slotted stainless steel screen. Each well is equipped with a gaged (vented-cable) data logger (miniTROLL, In-Situ, Inc.). A picture of a well with the logger cable connected for data download is shown in Figure 3.21. Depth to groundwater (DTW) was sampled every 15 or 30 minutes. All times were recorded in Mountain Standard Time (UTC-7). DTW was averaged by hour to ensure temporal alignment and reduce the total number of data points for subsequent analysis.

At each TOW well, a bore hole of 7.62 cm was drilled. A filter pack of 10-20 mesh silica sand was added, and a surface seal of 0.95 cm of bentonite chips was applied. Schedule 40 PVC was used for the well casing with identical diameters as above. Each well extends approximately 3 meters below the soil surface, and is screened across the bottom 0.9 meters of the casing with a 0.25 mm slotted screen. Each well was originally equipped with an Electronic Engineering Innovations Model 5.0 vented logger. On the 27th of April, 2005, the loggers were replaced with Solinst 2001 M10 Mini LT Levelloggers. Correction for barometric pressure was achieved via an on-site Solinst 3001 M1.5 Mini LT Barologger.

3.4 Data acquisition and processing

Manual groundwater levels were recorded at each well visit using a tape measure and conductive-activated flashlight. A GPS waypoint that included time, date, and approximate elevation was recorded at each well visit. In addition, 6 pictures of each well were recorded with a digital camera – one in each cardinal direction, one of the canopy, and one of the ground. Distance from well casing to soil surface and from
well casing to logger pressure sensor was manually measured once for each well to calculate the distance from water table to soil surface (DTW).

Hourly river flow data were obtained from the USGS for gage numbers #08330000 (Rio Grande at Albuquerque, NM), #08330875 (Rio Grande at Isleta Lakes near Isleta, NM), and #08354900 (Rio Grande Floodway at San Acacia, NM) via personal correspondence and via the NWISweb Automated Retrieval system (U.S. Geological Survey, 2008b). Linear regression of #08330875 onto #08330000 yielded slope and adjusted $R^2$ values within 2% of unity, showing very close agreement between the two gages. As the record of #08330000 was more complete for the period in question, it was used for subsequent analysis.

All data files were pre-processed via a perl script (Wall et al., 2000). The perl script checks for errors and documents the exact format of data files. All data were stored in a postgresql relational database (PostgreSQL Global Development Group, 2008). The R statistical computing environment (R Development Core Team, 2008)
was used for data analysis. The *dynlm* R package (Zeileis, 2008) was used for time series linear regression. Figures were prepared with the *Lattice* R package (Sarkar, 2008). The *Sweave* R package (Leisch, 2002) was used for manuscript preparation.

### 3.5 Calculating River-Groundwater Connection

Groundwater measurements were matched to river flow measurements by hour. Time periods were discarded from analysis if they contained fewer than 90 concurrent measurements in groundwater and river flow. Each year was split into a growing season and a non-growing season. The growing season was defined as April through October, inclusive. OLS linear regression of groundwater onto river flow was conducted for each season, year, and well. Regression of lagged groundwater onto river flow was conducted for lags up to 60 days.

### 3.6 Calculating ET

A readily available specific yield ($\hat{S}_y$) of 0.1 was employed as per Martinet et al. (2009). The choice of $\hat{S}_y$ has more effect on White method estimates than any other single parameter, and deserves further attention. Soil samples were not available for the study wells, and thus well-specific $\hat{S}_y$ determination was not possible.

Daily standard deviation (s.d.) of river flow was calculated for the period of record. Any day when flow s.d. exceeded 100 cfs was excluded from further consideration.

For the full period of record, hourly $dDTW/dt$ was calculated. To estimate recharge ($r$), the 5 $dDTW/dt$ estimates corresponding to DTW measurements from 11 p.m. to 5 a.m were averaged for each day. Recharge ($s$) was calculated as the difference between consecutive midnight measurements of DTW for each day. Daily estimates of ET were calculated using the White method (see Equation 2.3). All
Figure 3.22: One day hydrograph of DTW for two wells, August 2005. Daily storage change was calculated as the difference between consecutive midnights, shown as black X’s. The red shading indicates the period over which recharge was calculated. Values over 0.015 mm/day, and negative values, were discarded as outliers following a visual inspection of outliers.

The employed method is illustrated for a short time period in Figures 3.22, 3.23, 3.24, 3.25, and 3.26. A single day of DTW and $d\text{DTW}/dt$ for two wells is shown in Figures 3.22 and 3.23. An exemplary week of DTW measurements highlighting diel variations are shown in Figure 3.24. The resulting $d\text{DTW}/dt$ is shown in Figure 3.25 (note that different instrumentation and well designs were employed at the TOW wells).
Figure 3.23: One day hydrograph of the time derivative of DTW (dDTW/dt) for two wells, August 2005, as in Figure 3.22. Note the derivative is backwards-looking, so that the first and second time-points from Figure 3.22 are used in the first time-point here. Recharge is calculated as the mean of the 5 slopes in the recharge calculation period, shown as red X’s. The dashed red and solid black horizontal lines shows calculated hourly recharge (r) and $d\text{DTW}/dt = 0$, respectively.
Figure 3.24: One week hydrograph of DTW for all wells, 2005-08-05 to 2005-08-10.
Figure 3.25: One week hydrograph of the time derivative of DTW for all wells, 2005-08-05 to 2005-08-10. The black horizontal line shows $d\text{DTW}/dt = 0$. 
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3.7 Comparison of White Method with Eddy Covariance

A linear regression of eddy covariance estimates onto White method estimates was performed to assess the degree to which White method estimates can predict the more widely used and generally reliable eddy covariance estimates. In this regression, an intercept of 0 was enforced to ensure that both estimates converged on zero. Daily ET and weekly and monthly mean daily ET were tested for goodness-of-fit. The effects of varying $S_y$ and the recharge time window on goodness-of-fit were also tested.
Chapter 4

Results

4.1 Data quality and completeness

Significant gaps in the groundwater record were caused by equipment malfunction and operator error. Certain loggers and wells showed much higher rates of failure and rapid battery death (e.g. W.CON.N). Clock synchronization was especially troublesome. Many observation files contain inaccurate timestamps due to failure to synchronize the logger clock with the handheld computer. When the correct location of these files could be inferred from record gap size and groundwater level, they were manually corrected.

Agreement between DTW as measured by logger and by manual well flashing was poor. Over 43 samplings, the mean difference was -0.0099 meters, while the standard deviation of the difference was 0.1344 meters. In W.CON.N, a sudden drop in DTW in April of 2008 (Figure 4.3) was likely caused by replacement of the groundwater logger, and was not observed in manual DTW measurements. These results indicate that the absolute error of logger DTW measurements is high. However, the White method depends on relative DTW measurements, and static, absolute errors are absent from the time derivative. Overall, relative DTW measurements appear accurate, as evidenced by the observed high correlation over time between wells and
4.2 River-groundwater interactions

Precipitation and river flow over the study period are shown in Figures 4.1 and 4.2. A strong and significant connection (p < 0.001) between river flow and DTW is evident from a comparison of Figures 4.2 and 4.3, as well as from Figure 4.5. In Figure 4.4, a short, high peak in river flow is seen to elicit a broad, low peak in depth to groundwater, indicating delay and hysteresis of the river flow’s effect on DTW. The reactivity of groundwater to river flow is also seen in Figure 4.4 to vary from well to well. Linear regression of DTW onto river flow helps quantify the river-groundwater connection, and provides further evidence that the connection is not linear. Hysteresis is also evident in Figure 4.5, with groundwater rising and falling more slowly than river flow. Here, the path through time is counter-clockwise from spring to autumn, indicating that high river flows cause increased soil saturation that persists after the floodwaters recede. All lags up to 60 days were found to be significant but exhibit very small effects.

At low river flows, during the autumn and winter months (especially in 2005), the fit of the linear regression is poor, as seen in Figure 4.6. Figure 4.7 shows the relation between each regression’s fitted slope and its $R^2$. Figure 4.7 shows that regressions with greater slopes have better fits, suggesting that the linear regression is most appropriate for identifying strong, rapid response of groundwater to river flow. Likewise, regression $R^2$'s are higher during the growing season, when river levels are typically much higher. Negative slopes were only seen in linear regressions with poor fits (low $R^2$), primarily during the non-growing season.
4.3 Interwell comparison of depth to groundwater

Significant variation in DTW exists both within and between wells. The CON and STK wells experience the largest DTW, reaching 2.5 to 3.25 meters, while TOW and CHP wells have the least DTW, ranging from 1.5 to 0.5 meters. DTW exhibits a non-normal distribution, as seen in Figures 4.8 and 4.9. Figure 4.8 shows the density of DTW for all years taken together for each well, while Figure 4.9 shows DTW for each well in 2005. In these figures, a long tail of shallow DTW is observed, corresponding with transient periods of high river flow. The flood pulse is especially apparent in Figure 4.9, during 2005. A high peak of deeper DTW is also seen here, corresponding with long periods of low groundwater levels during autumn and winter.

Figures 4.10 and 4.11 show comparisons of years for each well and wells for each
year, respectively. Within each well over multiple years (Figure 4.10), the variance changes from year to year, while the median is relatively conserved. Within each year across multiple wells (Figure 4.11), years of low and high variance are especially evident (2003 and 2005, respectively). Variance of DTW in the growing season is much higher than in the non-growing season.

Figure 4.3: Mean daily depth to groundwater (m) for each study well for study period.
Figure 4.4: Response of groundwater to transient river flow spikes in select wells, 2005. The distance from each well to the river channel is as follows: W.CON.N, 140 meters; W.PLT.N, 146 meters; W.CHP.N, 46 meters; and E.STK.N, 218 meters.
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Figure 4.5: Linear regression of DTW onto log10(river flow) during the growing season for all wells and selected years.
Figure 4.6: Linear regression of DTW onto log10(river flow) during the non-growing season for all wells and selected years.
Figure 4.7: Goodness of fit versus slope for linear regression of DTW onto river flow by season for all study wells. Only regressions with $p<0.05$ are shown.
Figure 4.8: Mean daily DTW frequency (density kernel) by well for all years for both seasons.

Figure 4.9: Mean daily DTW frequency (density kernel) by well for 2005 for both seasons.
Figure 4.10: Box-and-whisker plot of mean daily DTW by season by year by well.
Figure 4.11: Box-and-whisker plot of mean daily DTW by season by well by year.
Chapter 4. Results

4.4 ET estimates

Overall, ET estimates appear realistic (Figure 4.12), peaking in mid-summer and falling to zero in the non-growing season (Figure 4.15). Most estimates are within previously published ranges for riparian phreatophytes, W.CHP.S is an exception, where estimates are unrealistically high. This indicates that the \( \hat{S}_y \) used for this well is too high. Clay lenses are known to occur in the MRGB (Thorn et al., 1993; McAda and Barroll, 2002), and can lead to very low well-specific \( \hat{S}_y \) (as low as 0.03, see Johnson (1967)). Finally, the employed method is prone to making isolated outlier estimates, and these sole estimates should not be regarded as significant.

Significant interwell differences are evident, and little similarity within treatment is observable, except within the E.TOW wells (Figures 4.17 and 4.18). The E.TOW wells are noteworthy for their close proximity to each other. Inter-year variation shows an overall decrease in ET over time (Figure 4.19). Within some wells, ET is seen to vary considerable from year to year (e.g. W.PLT.S and W.CHP.S), with an overall decline in ET over time. Wells with low ET seem to exhibit much less year-to-year variation (e.g. W.CHP.N and E.STK.N, Figure 4.19).

4.4.1 Treatment effects, fire effects, and yearly trends in ET

These results indicate that the effect of treatment on ET may modestly decrease ET. All sites for which reliable data are available show a long-term trend of decreasing ET rates through 2006, with a slight uptick of ET in 2007 and 2008 (Figures 4.12 and 4.19). This could signify a decrease in ET following treatment, with a concomitant post-treatment rebound in ET as vegetation re-established. Sparse pretreatment data and high intra-site and inter-year variability, especially with respect to control sites, renders these conclusions uncertain. Drought conditions in 2002 and 2003 have been identified as alternate causes of increased cottonwood ET due to increased insolation from reduced cloudcover (Cleverly, personal communication). For reference, net daily radiation and mean daily canopy temperature are shown in Figures 4.13 and 4.14.
Chapter 4. Results

The effect of fire and consequent cottonwood mortality on ET was dramatically observed in June of 2006 at W.CHP.S, as seen in Figure 4.16. Post-fire visual assessment of the site indicated complete cottonwood mortality. During the monsoon season following the fire, the site was densely colonized by the non-native annuals kochia (*Kochia scoparia*) and Russian thistle (*Salsola kali*). Figures 4.3 and 4.5 show that DTW did not exceed 2 meters at W.CHP.S in the summer of 2006. Kochia commonly grows taproots to depths of 2.4 meters (Esser, 1995). Thus, kochia likely had direct access to groundwater and may have contributed to the observed post-fire ET. Note that, in 2007, ET was reduced but detectable. This indicates either that consumption of groundwater by non-native annuals was appreciable, or that the site experienced background diel fluctuations in DTW from off-site vegetation.

### 4.4.2 Comparison of ET between yearly well-pairs

ET exhibits a non-normal distribution, as seen in Figures 4.21 and 4.22. A similar density distribution is seen when each year is considered individually. As such, a t-test is misleading when comparing ET estimates between wells. In addition, interyear hydrological and climactic differences preclude comparisons between years. To assess differences between yearly well-pairs, the Mann-Whitney test was applied within year to each well-pair (Hollander and Wolfe, 1973). A summary of results of the test are shown in Figure 4.23. The full results are shown in Figure 4.24.

Significant differences in yearly ET estimates were found between well sites, as seen in Figure 4.23. The E.TOW well sites exhibited significantly higher ET than the remaining wells. Within-treatment variation was high, indicating that treatment alone is not sufficient to explain variation in ET estimates. The well site with the lowest estimated ET (E.STK.N) appears to have burned prior to the study and lacks a phreatophyte canopy, suggesting that absence of phreatophyte ET can be accurately estimated by the White method. Figure 4.24 shows yearly comparisons for each well-pair, with one point per year per well-pair.
4.4.3 Comparison of depth to groundwater versus ET

No systemic pattern between depth to groundwater and estimated ET was observed, as seen in Figure 4.25. This indicates that cottonwood mortality due to lowered groundwater levels is not yet a significant factor at the study sites, nor do shallow groundwater levels correlate with significantly greater ET. If groundwater levels drop below 3 meters at any well, increased cottonwood mortality and a concomitant decrease in ET is expected at that well.
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Figure 4.12: Estimated ET for each well over study period.
Figure 4.13: Net radiation per day, measured at E.TOW, 10 day rolling average.

Figure 4.14: Mean daily canopy temperature, measured at E.TOW, 10 day rolling average.
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Figure 4.15: Estimated ET for select wells in 2005.

Figure 4.16: Estimated ET for W.CHP.S, 2006 and 2007. This well experienced a catastrophic wildfire in June of 2006 (red vertical line), resulting in complete cottonwood mortality. A sharp reduction in estimated ET following the fire is evident.
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Figure 4.17: Daily ET by well, April–October (inclusive), for all years with > 30 days of data per year per well.

Figure 4.18: Daily ET by year, April–October (inclusive), for all years with > 30 days of data per year per well.
Figure 4.19: Daily ET by year between wells, April–October (inclusive), for all years with > 30 days of data per year per well.
Figure 4.20: Daily ET by well between years, April–October (inclusive), for all years with > 30 days of data per year per well.
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Figure 4.21: Daily ET frequency (density kernel) by well, April–October (inclusive), for all years with > 30 days of data per year per well.

Figure 4.22: Daily ET frequency (density kernel) by well, April–October (inclusive), for 2005.
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Figure 4.23: Estimated within-year difference in daily ET between well-pairs aggregated by well. Only differences with $p < 0.05$ as per Mann-Whitney test are shown. April–October (inclusive), for all years with $> 30$ days of data per year per well.
Figure 4.24: Estimated within-year difference in daily ET between well pairs. Only differences with $p < 0.05$ as per Mann-Whitney test are shown. April–October (inclusive), for all years with > 30 days of data per year per well.
Figure 4.25: Daily DTW versus estimated ET by well, April–October (inclusive), for all years with > 30 days of data per year per well.
4.5 Comparison of White Method with Eddy Covariance

Overall, the correlation between White method and eddy covariance estimates of ET is significant and high. The goodness-of-fit of the model increased when monthly means of daily ET were employed, compared to daily estimates or weekly means of daily estimates. As the time window used to estimate recharge was extended forward in time past 6 a.m., the estimated slope rose (i.e. White method estimates decreased). As the time window was extended backward from 12 a.m., the estimated slope fell (i.e. White method estimates increased).

Using $\hat{S}_y = 0.1$ and a recharge time window of 12 a.m. to 6 a.m., the estimated slope is within 10% of unity for all wells, and within 1% of unity for 3 out of 4 wells; the adjusted $R^2$ values are within 6% of unity for all wells. Results of the regressions are shown in Table 4.1.

A graphical comparison of White method estimates with eddy covariance estimates is shown in Figure 4.26, with best-fit lines for each well shown. In addition, a mean of all wells at each point in time is computed and shown as E.TOW.Mean. Figure 4.27 shows the same comparison as Figure 4.26 by year with weekly means.

It is important to note that eddy covariance and the White method measure ET over significantly different spatial extents. The fetch over which eddy covariance measures ET varies with wind direction and speed, and may extend more than 100 meters upwind of the tower. The soil volume over which an observation well measures ET is a function of the soil’s hydraulic conductivity and, to a lesser extent, the DTW. In addition, eddy covariance estimates measure total ET, while White method estimates measure only phreatophyte extraction of groundwater. Thus, the two methods measure different quantities and areas, and the long-term agreement between the two methods is somewhat surprising.
<table>
<thead>
<tr>
<th>Predictor Well</th>
<th>Fitted Slope</th>
<th>Degrees of Freedom</th>
<th>Adjusted $R^2$</th>
<th>P Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>E.TOW.C</td>
<td>0.909</td>
<td>44</td>
<td>0.954</td>
<td>0.000</td>
</tr>
<tr>
<td>E.TOW.N</td>
<td>0.996</td>
<td>32</td>
<td>0.952</td>
<td>0.000</td>
</tr>
<tr>
<td>E.TOW.S</td>
<td>1.006</td>
<td>39</td>
<td>0.953</td>
<td>0.000</td>
</tr>
<tr>
<td>E.TOW.W</td>
<td>1.002</td>
<td>39</td>
<td>0.949</td>
<td>0.000</td>
</tr>
<tr>
<td>E.TOW.Mean</td>
<td>0.941</td>
<td>44</td>
<td>0.947</td>
<td>0.000</td>
</tr>
</tbody>
</table>

Table 4.1: Linear regression of ET estimated by eddy covariance versus White method, monthly means.
Figure 4.26: Monthly means of daily ET estimates at UNM control site – Eddy Covariance versus White method, full period of record. The dashed black line shows a slope of unity. Colored lines show regression slopes for corresponding wells, as shown
Figure 4.27: Weekly means of daily ET estimates at UNM control site – Eddy Covariance versus White method by year. The dashed black line shows a slope of unity. Colored lines show regression slopes for corresponding wells, as shown in legend.
Chapter 5

Discussion

5.1 Efficacy of the White method for Estimating ET

The White method has been the subject of extensive debate over the years. A thorough historical overview is given by Gribovszki et al. (2010), detailing various innovations such as novel methods of estimating $S_y$ and accounting for time-variable $Q$. The results presented here indicate that the White method is a useful tool for estimating riparian evapotranspiration provided the following conditions are met. First, traditional point-in-time estimates of recharge rates inflate estimate variance. By estimating recharge from average nighttime $dH/dt$, the effects of time- and well-specific hydrology, such as transient groundwater changes and complete predawn recharge, are minimized. Second, measured sites should contain predominantly phreatophyte vegetation with well-developed root systems that extend below the water table.

Similarly, soil hydrological characteristics can be highly variable over short spatial scales. By using a sufficiently large number of wells, such intrinsic well-specific variance can be estimated. The low cost of groundwater monitoring well installation and maintenance compared to eddy covariance towers, the comparative ease of groundwater data collection and analysis, and the favorable performance of the
White method when compared with eddy covariance, suggest that further use of the White method is warranted.

As shown in Figures 4.26 and 4.27 and Table 4.1, the White method provides a reliable estimate of ET when long-term records are considered. The variance of White method ET measurements is much higher than eddy covariance measurements, yet the two methods are highly and significantly correlated over the long term. $S_y$ controls the slope of the regression line (colored lines in Figures 4.26 and 4.27 and shown in Table 4.1). A slope of 1 shows that the White method accurately predicts ET as estimated by eddy covariance. Thus, $S_y$ can be estimated by comparing the White method with eddy covariance estimates by selecting an $S_y$ that gives a slope of 1.

The White method’s assumptions of zero nocturnal phreatophyte ET is appropriate for cottonwood dominated sites, but is not always appropriate. Using eddy covariance, sap flux, and leaf gas exchange measurements, Moore et al. (2008) have shown that nighttime ET constitutes a significant portion of total ET at saltcedar-dominated sites. Saltcedar comprised no more than a small fraction of total phreatophyte vegetation at any of the sites considered in this study, and nighttime ET is not expected to be significant here. At saltcedar-dominated sites, the White method would be expected to underestimate ET, both by failing to count nighttime ET and by underestimating recharge.

5.2 Vegetation Removal, Wildfire, and ET

Reduction in phreatophyte ET as a result of removing non-native vegetation was difficult to observe, and a small proportion of total ET. All wells, including the control sites, experienced decreased ET from 2003 to 2004, and all wells except for burn sites experienced increased ET from 2006 and 2007 to 2008 (Figure 4.19). This indicates that year-to-year variability significantly influenced ET. The reduction in ET found in other studies at similar sites and at the TOW site is estimated to
be approximately 10-20% over pre-treatment values (Cleverly et al., 2006; Martinet et al., 2009). A 10-20% decrease appears to be just at the threshold of detection of the employed method and the available data. A more complete pretreatment and control record would improve confidence in ET reduction estimates.

Wildfire, on the other hand, caused an immediate, readily detectable, and almost complete cessation of phreatophyte ET. These results indicate that the White method is capable of detecting large changes in phreatophyte ET on very short timescales. Despite the low absolute accuracy and precision of the White method as employed here, these results indicate that it is able to reliably detect relative, within-site changes in phreatophyte ET over time.

5.3 Flooding, Fire, and the Middle Rio Grande Bosque

Historically, the MRGB was a flood-dominated ecosystem (Crawford et al., 1993). Sporadic, large-scale flooding created a patchwork mosaic of habitats as the braided river carved new channels and old channels were colonized by vegetation. Flooding increased nutrient cycling and decreased fire risk through increased forest litter decomposition (Ellis et al., 1998; Molles Jr. et al., 1998; Shah and Dahm, 2008). Cottonwoods have historically been the dominant vegetation of the MRGB, and depend upon flooding for successful germination and establishment.

Human activity in the 20th century has dramatically reduced flooding in the MRGB, with significant ecological consequences. Hydrological modification of the MRGB began in the 1920s with the construction of levees and drainage canals, and culminated with the completion and closure of Cochiti dam in 1973 (Crawford et al., 1993). The last major flood experienced by the region was in 1942. The most extensive cottonwood galleries date from this flood, and Howe and Knopf (1991) assert that little regeneration of cottonwoods has occurred in the MRGB since 1955-1960. Decreasing litterfalls have been observed in the last 20 years, and indicate
ongoing senescence of existing cottonwood canopies (Molles Jr et al., 1998).

Increasingly, fire has emerged as a novel ecological driver in the MRGB. While historical and pre-historical records of wildfires in North American riparian habitats are scarce (Zouhar, 2003), available evidence suggests that fires were uncommon in the MRGB before modern times. In particular, cottonwoods’ intolerance to fire indicates that they evolved in an environment where fire was not a dominant factor. Modern wildfires have caused large-scale cottonwood mortality, while facilitating the establishment of non-native saltcedar and Russian olive. Saltcedar, in particular, can rapidly re-establish from underground parts after wildfires (Smith et al., 2009; Zouhar, 2003), and mature saltcedar stands burn readily and with high intensity. Saltcedar is a particularly interesting invasive species in that it was intentionally introduced into the U.S. in general throughout the 19th and 20th centuries as an ornamental plant and a windbreak, and specifically into the MRGB starting in 1926 for erosion and silt control (Crawford et al., 1993).

As the catastrophic burning of W.CHP.S in 2006 shows, removal of non-native vegetation alone is not always sufficient to prevent cottonwood mortality via wildfire. Likewise, lack of overbank flooding means that establishment of new cottonwoods is greatly diminished. As such, the MRGB is rapidly shifting towards a fire-dominated ecosystem. Flood prevention measures were originally instituted, in part, to prevent loss of human life and property. The new fire-dominated ecosystem already poses many challenges to stake-holders, including threats to human life and property. Further work is needed to assess fire risks and possible mechanisms of fire prevention and fire damage reduction.
5.4 Mediating and monitoring human-mediated ecosystem changes in the Middle Rio Grande bosque

Human-mediated hydological modification of the Rio Grande, and thus the MRGB, is not new. However, with rising populations comes increased urban water demand, while incipient climate change is predicted to lead to decreases in regional precipitation (Seager et al., 2007). Altogether, human effects will likely lead to lower instream river flows and reduced groundwater levels, leading to increased cottonwood mortality.

This study shows that cottonwood mortality leads to an immediate decrease in phreatophyte ET. Other studies have shown that the non-native vegetation that colonizes vacated land has comparable ET (Cleverly et al., 2002, 2006). Frequent controlled burning of saltcedar stands might realize long-term reductions in ET while improving nutrient cycling. Such controlled burns of standing saltcedar intermixed with cottonwoods could cause high mortality of native riparian vegetation, and may only be appropriate where prior burns have already occurred or where no native riparian vegetation remains.

This study clearly demonstrates that the White method can reliably detect cessation of cottonwood ET. Continued monitoring of riparian groundwater under declining groundwater regimes will allow for the future estimation of critical DTW values that determine cottonwood mortality. Previous studies have cited 3 meters as the critical DTW past which cottonwoods experience rapid crown dieback, though the speed of DTW change and the maturity of cottonwood stands are both cited as influential factors in cottonwood mortality (Horton et al., 2008). At least 2 wells in this study (the W.CON wells), are currently at this 3 meter threshold. Future decreases in DTW at these sites as a result of human activities are likely, and ongoing observation of these sites could provide a unique opportunity to directly observe DTW-mediated cottonwood mortality via ET estimates.
References


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