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In cooperation with the U.S. Department of Agriculture Natural Resources Conservation Service, the Edwards Region Grazing Lands Conservation Initiative, the Texas State Soil and Water Conservation Board, the San Antonio River Authority, the Edwards Aquifer Authority, Texas Parks and Wildlife, the Guadalupe Blanco River Authority, and the San Antonio Water System

Effects of Brush Management on the Hydrologic Budget and Water Quality In and Adjacent to Honey Creek State Natural Area, Comal County, Texas, 2001–10

Scientific Investigations Report 2011–5226

U.S. Department of the Interior
U.S. Geological Survey

Cover: Evapotranspiration station in the treatment watershed, Honey Creek State Natural Area, Comal County, Texas, July 23, 2010.

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By J. Ryan Banta and Richard N. Slattery

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U.S. Department of the Interior
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Conversion Factors

Inch/Pound to SI

Multiply	By	To obtain
	Length	
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
acre	0.004047	square kilometer (km^2)
square foot (ft^2)	0.09290	square meter (m^2)
	Volume	
cubic foot (ft^3)	0.02832	cubic meter (m^3)
	Flow rate	
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)
inch per year (in/yr)	25.4	millimeter per year (mm/yr)
	Mass	
ton per day (ton/d)	0.9072	metric ton per day
	Pressure	
atmosphere, standard (atm)	101.3	kilopascal (kPa)
	Density	
pound per cubic foot (lb/ft^3)	16.02	kilogram per cubic meter (kg/m^3)
	Energy	
kilowatthour (kWh)	3,600,000	joule (J)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$°F=(1.8\times°C)+32$$

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:

$$°C=(°F-32)/1.8$$

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius (µS/cm at 25°C).

Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L) or micrograms per liter (µg/L).

Effects of Brush Management on the Hydrologic Budget and Water Quality In and Adjacent to Honey Creek State Natural Area, Comal County, Texas, 2001–10

By J. Ryan Banta and Richard N. Slattery

Abstract

The U.S. Geological Survey, in cooperation with the U.S. Department of Agriculture Natural Resources Conservation Service, the Edwards Region Grazing Lands Conservation Initiative, the Texas State Soil and Water Conservation Board, the San Antonio River Authority, the Edwards Aquifer Authority, Texas Parks and Wildlife, the Guadalupe Blanco River Authority, and the San Antonio Water System, evaluated the hydrologic effects of ashe juniper (*Juniperus ashei*) removal as a brush management conservation practice in and adjacent to the Honey Creek State Natural Area in Comal County, Tex. By removing the ashe juniper and allowing native grasses to reestablish in the area as a brush management conservation practice, the hydrology in the watershed might change. Using a simplified mass balance approach of the hydrologic cycle, the incoming rainfall was distributed to surface water runoff, evapotranspiration, or groundwater recharge. After hydrologic data were collected in adjacent watersheds for 3 years, brush management occurred on the treatment watershed while the reference watershed was left in its original condition. Hydrologic data were collected for another 6 years. Hydrologic data include rainfall, streamflow, evapotranspiration, and water quality. Groundwater recharge was not directly measured but potential groundwater recharge was calculated using a simplified mass balance approach. The resulting hydrologic datasets were examined for differences between the watersheds and between pre- and post-treatment periods to assess the effects of brush management. The streamflow to rainfall relation (expressed as event unit runoff to event rainfall relation) did not change between the watersheds during pre- and post-treatment periods. The daily evapotranspiration rates at the reference watershed and treatment watershed sites exhibited a seasonal cycle during the pre- and post-treatment periods, with intra- and interannual variability. Statistical analyses indicate the mean difference in daily evapotranspiration rates between the two watershed sites is greater during the post-treatment than the pre-treatment period. Average annual rainfall, streamflow, evapotranspiration, and potential groundwater-recharge conditions were incorporated into a single hydrologic budget (expressed as a percentage of the average annual rainfall) applied to each watershed before and after treatment to evaluate the effects of brush management. During the post-treatment period, the percent average annual unit runoff in the reference watershed was similar to that in the treatment watershed, however, the difference in percentages of average annual evapotranspiration and potential groundwater recharge were more appreciable between the reference and treatment watersheds than during the pre-treatment period. Using graphical comparisons, no notable differences in major ion or nutrient concentrations were found between samples collected at the reference watershed (site 1C) and treatment watershed (site 2C) during pre- and post-treatment periods. Suspended-sediment loads were calculated from samples collected at sites 1C and 2T. The relation between suspended-sediment loads and streamflow calculated from samples collected from sites 1C and 2T did not exhibit a statistically significant difference during the pre-treatment period, whereas during the post-treatment period, relation between suspended-sediment loads and streamflow did exhibit a statistically significant difference. The suspended-sediment load to streamflow relations indicate that for the same streamflow, the suspended-sediment loads calculated from site 2T were generally less than suspended-sediment loads calculated from site 1C during the post-treatment period.

Introduction

The U.S. Geological Survey (USGS), in cooperation with the U.S. Department of Agriculture Natural Resources Conservation Service, the Edwards Region Grazing Lands Conservation Initiative, the Texas State Soil and Water Conservation Board, the San Antonio River Authority, the Edwards Aquifer Authority, Texas Parks and Wildlife, the Guadalupe Blanco River Authority, and the San Antonio Water System, evaluated the hydrologic effects of ashe juniper (*Juniperus ashei*) removal as a brush management conservation practice in and adjacent to the Honey Creek State Natural Area in Comal County, Tex.

The Honey Creek State Natural Area overlies the Trinity aquifer outcrop area, which is also the catchment area for the Edwards aquifer (Maclay, 1995) (fig. 1). Barker and Ardis (1996) noted that the Trinity aquifer is an important water supply to much of northern Bexar County. Additionally, streams originating in the catchment area of the Edwards aquifer that cross the Edwards aquifer outcrop provide recharge to the Edwards aquifer (Maclay, 1995). The Edwards aquifer is the primary source of water for more than 1.7 million people in the greater San Antonio, Tex. area.

Anecdotal reports from local ranchers, as well as scientific studies, have indicated that land use and vegetative land cover might have an effect on streamflow, spring discharge, or groundwater recharge (Baxter, 2009; Tennesen, 2008; Thurow and Hester, 1997). Woody vegetation, including ashe juniper, have encroached upon areas that historically were oak grassland savannahs across much of the Edwards aquifer catchment and outcrop area near the study area. This is generally attributed to overgrazing and fire suppression (Van Auken, 2000; Bray, 1904).

Ashe juniper is commonly thought to be a larger consumer of water (by plant transpiration), in comparison to native grasses (Baxter, 2009). Ashe junipers have taproots as well as an extensive lateral root system in the surface foot of soil allowing the ashe juniper to take advantage of shallow and deeper soil moisture (Sullivan, 1993; Jackson and others, 1999). McCole and Stern (2007) examined seasonal changes in stable isotope concentrations in ashe juniper at Honey Creek State Natural Area as an indicator of the source of water taken up by this species. They found ashe juniper are opportunistic, utilizing water from a more shallow soil water source in the winter, and a deeper water source in the summer presumably because of the drying of the surface soils, or some combination of the two. By removing the ashe juniper and allowing native grasses to reestablish in the area as a brush management conservation practice (hereinafter referred to as brush management), the hydrology in the watershed might change. Hydrologic changes might include changes to surface water runoff, evapotranspiration, or groundwater recharge.

Huang and others (2006) monitored a perennial stream in a 47-acre watershed within the Honey Creek State Natural Area for 2 years before and 2 years after selective brush-control management. Using an event-based regression analysis of summer and non-summer rainfall for pre- and post-treatment, they observed an increase in the streamflow of approximately 5 percent of the average annual precipitation as a result of ashe juniper removal. The watershed Huang and others (2006) examined was not among those studied in this report, and the stream measured by Huang and others was a perennial stream as compared to the ephemeral streams monitored in this report.

Differences in evapotranspiration (the combined processes of evaporation and transpiration) have been observed at the individual plant to watershed scale. Dugas and others (1998) measured evapotranspiration at a 37-acre site in central Texas, approximately 50 miles (mi) west of the Honey Creek State Natural Area. The study measured evapotranspiration from two plots where the vegetation was predominately ashe juniper. The first (reference) plot was left in its original condition, and brush management was performed at the second (treatment) plot, allowing native grasses to reestablish in place of the brush. Evapotranspiration at the reference and treatment plots were measured 2 years before and 3 years after brush removal. The results of the study showed an average of 26 millimeters per year (mm/yr) difference in evapotranspiration between the reference and treatment plots. Most of the differences in evapotranspiration were observed in the first 2 years following treatment, but no appreciable differences were observed in the third year, possibly because of an increase in herbaceous vegetation.

Saleh and others (2009) examined the effects of brush-control management on two 200-acre watersheds in the North Concho River watershed, approximately 200 miles northwest of San Antonio. Similar to the study design used by Dugas and others (1998), Saleh and others (2009) selected one watershed for brush management (treatment watershed), and the other watershed was left with the mesquite-dominated brush intact (reference watershed). Comparing the post-treatment evapotranspiration rates, Saleh and others (2009) found that the reference watershed had a higher evapotranspiration rate than the treatment watershed (46 mm cumulatively during the 4-year study).

The physical linkages between evapotranspiration and other effects, such as streamflow, are not well understood (Wilcox and others, 2008, 2010; Saleh and others, 2010). Wilcox and others (2005) found that brush-control practices did not result in a notable difference in the amount, timing, or magnitude of ephemeral streamflow on watersheds of 7 to 15 acres in size. However, they noted changes in vegetation might affect streamflow when base-flow conditions are present (for example, presence of springs). Additionally, Wilcox and Huang (2010) found a general upward trend in streamflow in three of four large watersheds in the Edwards Plateau region of Texas using available data from about 1925 to about 2007; a period when woody encroachment presumably has been occurring. The authors attribute this streamflow trend to changes in land use, specifically, reduced grazing intensity of livestock on rangelands. Wilcox (2002) and Wilcox and others (2006) report that at sites where annual rainfall is greater than 500 mm and soil conditions are such that deep drainage can occur (for example, soils overlying a highly permeable karst layer), conversion of woody encroachment to healthy native grasslands might increase groundwater recharge.

Although previous studies have shown that changes in evapotranspiration might occur as a result of changes in vegetation, the effects on the hydrologic budget (for example, streamflow, spring discharge, and groundwater recharge), especially at the watershed scale, are still not well understood. Observed changes in the hydrologic budget are likely linked to both land use and vegetation cover and to some degree are site and scale specific.

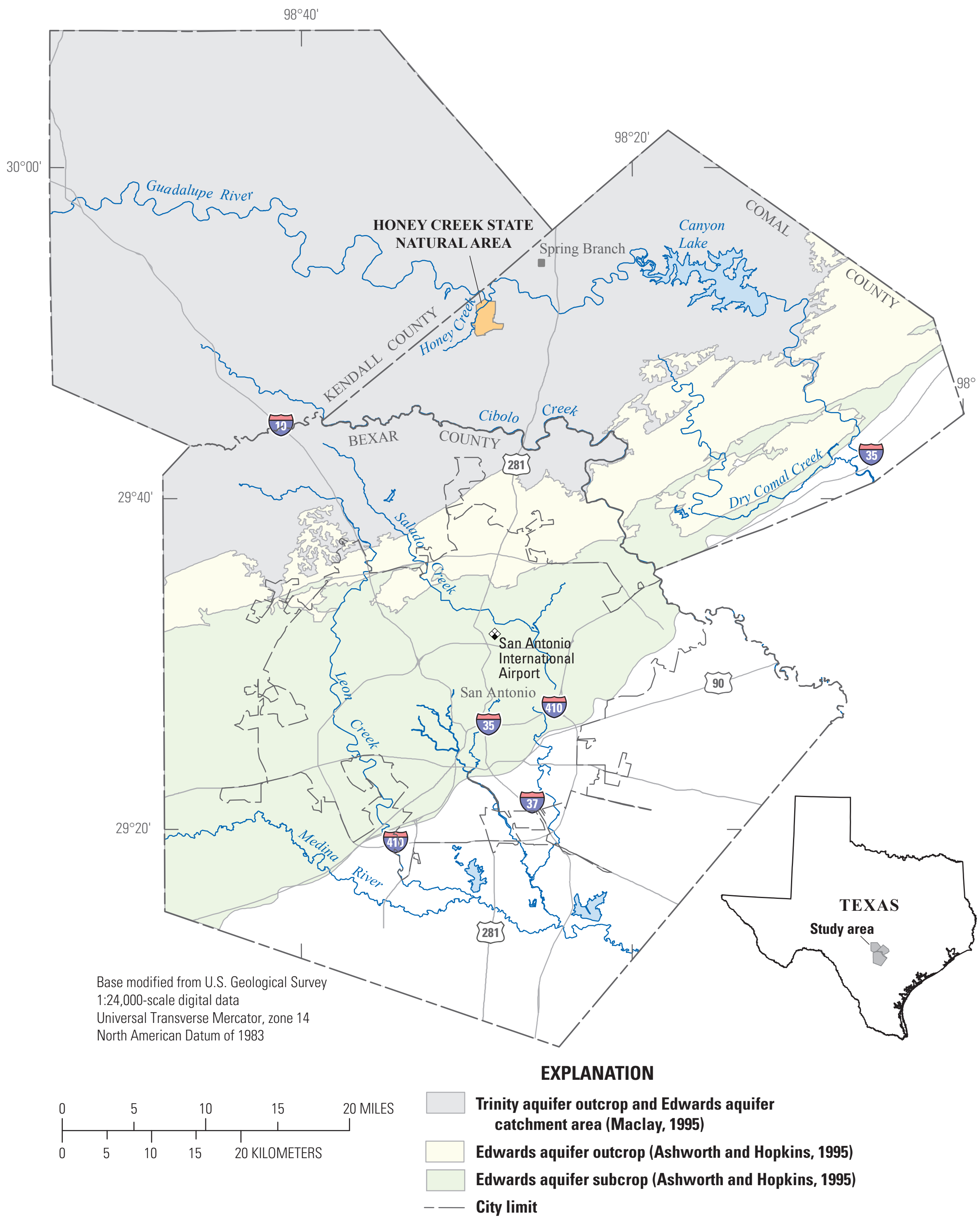


Figure 1. Location of the Honey Creek State Natural Area, Comal County, Texas.

Purpose and Scope

This report evaluates selected hydrologic effects of a brush-management conservation practice (ashe juniper removal) using a paired watershed approach in and adjacent to the Honey Creek State Natural Area, near San Antonio, Tex. Hydrologic budget and water-quality data were collected in both watersheds during 2001–10. The hydrologic budget data collected include rainfall, streamflow, and evapotranspiration rates. The groundwater-recharge values were not directly measured but potential groundwater-recharge values were calculated using a simplified mass balance approach to the hydrologic budget. The water-quality constituents analyzed include selected major ions, nutrients, and suspended sediments. The resulting hydrologic datasets were examined for differences between the watersheds and between pre-treatment (2001–04) and post-treatment (2005–10) periods to assess the effects of brush management.

Description of Study Area

The study area consists of two watersheds in and adjacent to the Honey Creek State Natural Area, about 10 miles north of San Antonio, Tex., which is in the Trinity aquifer outcrop and Edwards aquifer catchment area (figs. 1 and 2). The reference watershed, defined as the drainage area upstream from the streamflow-gaging station 1C, is 223 acres (fig. 2, table 1). The treatment watershed, defined as the drainage area upstream from the streamflow-gaging station 2T, is 340 acres. The upper part of the treatment watershed is on private land outside the Honey Creek State Natural Area boundary (fig. 2). An informal agreement was made with the landowner to maintain consistent vegetation cover and land-use practices during the study period (Phillip Wright, Natural Resources Conservation Service, written commun., 2011).

The long-term average annual rainfall, as calculated from the 1957–2009 annual rainfall record at the National Weather Service station at Spring Branch, Tex. (fig. 1) was 34 inches per year (in/yr), with interannual variability of 10 in. (estimated as one standard deviation from the mean for the period of record) (National Climatic Data Center, 2011). The rainfall events generally were evenly distributed throughout the calendar year, as determined from the rainfall data collected within the Honey Creek State Natural Area during 2001–10 (appendix 1). The watersheds are drained by first-order (headwater) ephemeral streams that are tributaries to Honey Creek, which in turn is a tributary to the Guadalupe River (fig. 2).

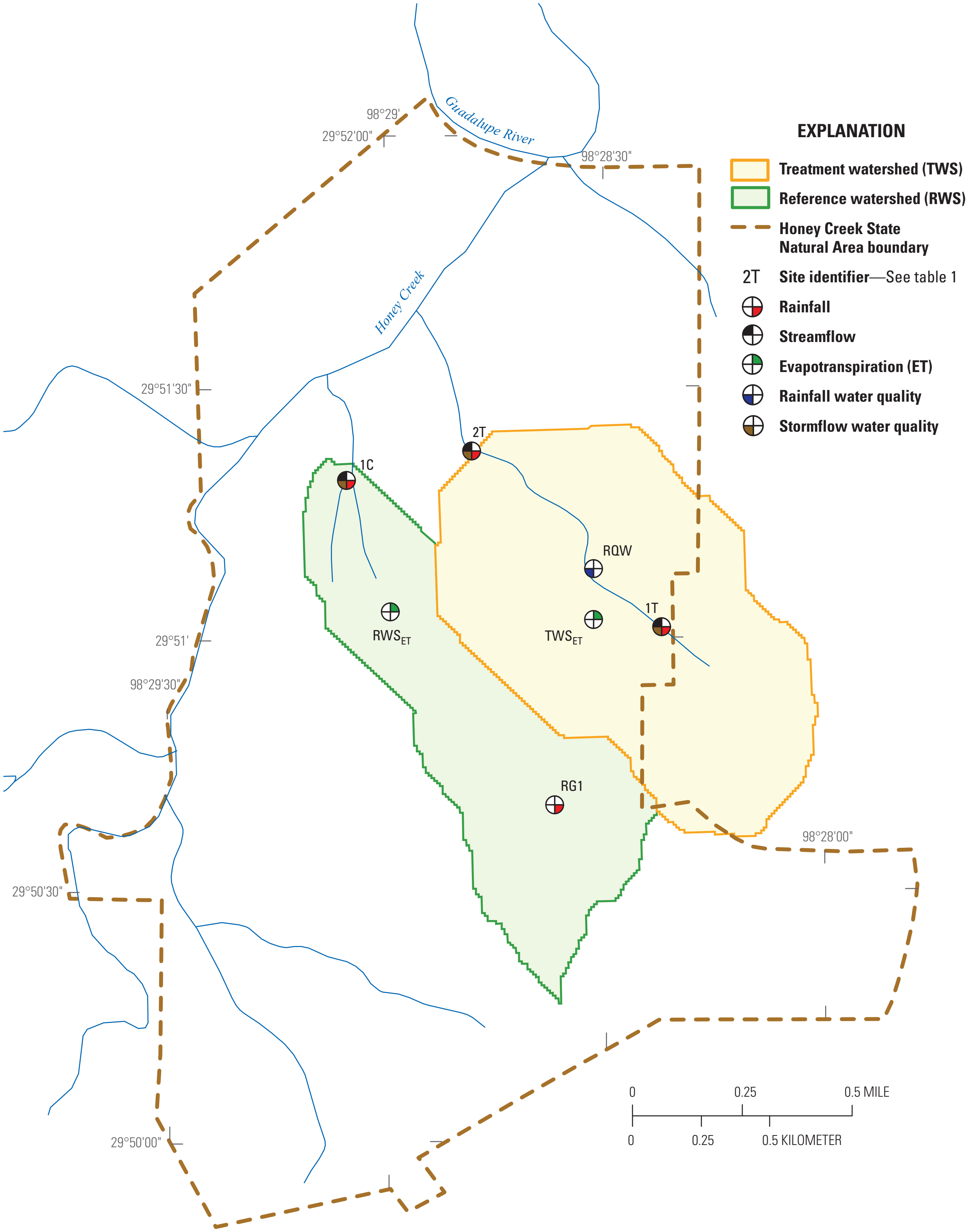
Most areas in the reference and treatment watersheds are gently sloped (less than 5 percent), with steeper slopes in the ravines along the stream channels and at the outlet of the watersheds (northern part of the watersheds). Soils are generally calcareous stony clay and clay loam with rock outcrop (U.S. Department of Agriculture, Soil Conservation Service, 1984), overlying the Trinity group. The Trinity group is comprised of lower Cretaceous rocks, including the Cow Creek Limestone, Hensel Sand, and lower Glen Rose Limestone, which in turn compose the Trinity aquifer (table 2) (Stricklin and others, 1971; Perkins, 1974; Inden, 1974; Stricklin and Smith, 1973; Amsbury, 1974; and Ashworth, 1983). The Cow Creek Limestone and lower member of the Glen Rose Limestone have karst characteristics favorable to subsurface groundwater flow (Hunt and others, 2011).

Woody vegetation covers most of the landscape in and near the Honey Creek State Natural Area, and consists primarily of dense ashe juniper woods, with stands of live oak (*Quercus fusiformis and virginiana*). Interspersed are other woody species including cedar elm (*Ulmus crassifolia*), Spanish oak (*Quercus buckleyi*), and Texas persimmon (*Diospyros texana*). Grasses include buffalograss (*Buchloe dactyloides*), cedar sedge (*Carex planostachys*), curly mesquite (*Hilaria belangeri*), indiangrass (*Sorghastrum nutans*), King Ranch bluestem (*Bothriochloa ischaemum*), little bluestem (*Schizachyrium scoparium*), perennial threeawn (*Aristida oligantha*), meadow dropseed (*Sporobolus asper*), sideoats grama (*Bouteloua curtipendula*), and Texas wintergrass (*Stipa leucotricha*) (Wright and others, 2009).

Brush Management Conservation Practices

At the beginning of the study, the percent coverage of ashe juniper in the reference and treatment watersheds were similar (Wright and others, 2009). From November 2003 through July 2005, selective cutting removed about 70 percent of the ashe juniper from the treatment watershed in the Honey Creek State Natural Area (Phillip Wright, Natural Resources Conservation Service, written commun., 2011). Removal was done by cutting the ashe juniper near ground level with hydraulic tree shears attached to a skid-steer loader. This method kills the tree with minimal soil disturbance compared to tree dozing. Cut trees were left in place. Selective ashe juniper removal was done to ensure that the habitat and nesting season of the Golden-cheeked Warbler were not adversely affected (Texas Parks and Wildlife Department, 2009). Ashe juniper removal was also avoided in the ravines along the stream channels and on steeper slopes—the landscape niche ashe juniper might have historically occupied (Bray, 1904).

Initial brush removal occurred at a slower rate because of mechanical limitations. Brush removal in the treatment watershed began in the winter of 2004–5 at the evapotranspiration station (USGS station 295102098283200), proceeded outwards to the surrounding area, and was completed by May 2005. Consequently, January 1, 2005, was set as the representative date delineating “pre-treatment” and “post-treatment” time periods. The pre-treatment period is defined as January 2001 through December 2004. The post-treatment period is defined as January 2005 through December 2010.



Base modified from U.S. Geological Survey
1:24,000-scale digital data
Universal Transverse Mercator, zone 14
North American Datum of 1983

Figure 2. Location of data-collection sites in the Honey Creek State Natural Area, Comal County, Texas.

Table 1. Hydrologic and water-quality data-collection sites in the Honey Creek State Natural Area, Comal County, Texas.

[NA, not applicable; RF, continuous rainfall; SF, ephemeral streamflow; QWS, periodic stormflow water-quality; QWRF, periodic rainfall water-quality; ET, evapotranspiration]

Site identifier (fig. 2)	U.S. Geological Survey station number	U.S. Geological Survey station name	Latitude	Longitude	Drainage area (acres)	Type of data	Period of record
RG1	295040098283701	Honey Creek rain gage number 1 near Spring Branch, Tex.	29°50'40"	98°28'37"	NA	RF	Jan 2001–Dec 2010
1C	08167347	Unnamed tributary of Honey Creek site 1C near Spring Branch, Tex.	29°51'19"	98°29'05"	223	RF	Jan 2001–Dec 2010
						SF	Apr 2001–Dec 2010
						QWS	Aug 2001–Oct 2009
1T	08167350	Unnamed tributary of Honey Creek site 1T near Spring Branch, Tex.	29°51'01"	98°28'22"	105	RF	Jan 2001–Dec 2010
						SF	Jan 2001–Dec 2010
						QWS	Aug 2001–Oct 2009
2T	08167353	Unnamed tributary of Honey Creek site 2T near Spring Branch, Tex.	29°51'22"	98°28'48"	340	RF	Jan 2001–Dec 2010
						SF	May 2001–Dec 2010
						QWS	Aug 2001–Oct 2009
RQW	295108098283201	Honey Creek rainfall water quality near Spring Branch, Tex.	29°51'08"	98°28'32"	NA	QWRF	Oct 2001–Oct 2009
RWS_{ET}	295104098285900	Honey Creek reference evapotranspiration near Spring Branch, Tex.	29°51'04"	98°28'59"	NA	ET	Mar 2002–Dec 2010
TWS_{ET}	295102098283200	Honey Creek treatment evapotranspiration near Spring Branch, Tex.	29°51'02"	98°28'32"	NA	ET	Mar 2002–Dec 2010

Data Collection Methods

Hydrologic data were collected in the study area during 2001–2010. Rainfall data were collected from USGS stations 295040098283701 Honey Creek rain gage number 1 near Spring Branch, Tex. (hereinafter site RG1), 08167347 Unnamed tributary of Honey Creek site 1C near Spring Branch, Tex. (hereinafter site 1C), 08167350 Unnamed tributary of Honey Creek site 1T near Spring Branch, Tex. (hereinafter site 1T), and 08167353 Unnamed tributary of Honey Creek site 2T near Spring Branch, Tex. (hereinafter site 2T) (fig. 2, table 1).

Streamflow data were collected at sites 1C, 1T, and 2T. Meteorological data used for the calculation of evapotranspiration (ET) were collected in the reference watershed at USGS station 295104098285900 Honey Creek reference evapotranspiration near Spring Branch, Tex. (hereinafter site RWS_{ET}) and in the treatment watershed at USGS station 295102098283200 Honey Creek treatment evapotranspiration near Spring Branch, Tex. (hereinafter site TWS_{ET}) (fig. 2, table 1). Rainfall and stormflow water-quality samples were collected at four sites: USGS station 295108098283201 Honey Creek rainfall water quality near Spring Branch, Tex. (hereinafter site RQW), and at sites 1C, 1T, and 2T (fig. 2, table 1).

Rainfall

Rainfall data were collected at four sites (RG1, 1C, 1T, and 2T) using a 12-in. diameter tipping-bucket rain gage (fig. 2, table 1) (NovaLynx Corporation, 2009). Measurements were recorded at 5-minute intervals and transmitted every 4 hours by the Geostationary Operational Environmental Satellite (GOES) to the USGS National Water Information System (NWIS) database (U.S. Geological Survey, 2010). To maintain

Table 2. Summary of the geologic and hydrogeologic framework of the Trinity aquifer outcropping in the study area at the Honey Creek State Natural Area, Comal County, Texas.

[Period, group, formation, members, and lithology modified from Stricklin and others (1971), Perkins (1974), Inden (1974), Stricklin and Smith (1973), Amsbury (1974); approximate thicknesses from Wierman and others (2010); hydrogeologic framework from Ashworth (1983)]

Geologic framework						Hydrogeologic framework
Period	Group	Formation	Member	Approximate thickness (feet)	Lithology	
Lower Cretaceous	Trinity	Glen Rose Limestone	Lower	160	Medium to thick beds of limestone, dolostone, and dolomitic limestone	Trinity aquifer
		Hensel Sand	*	55	Weakly cemented mixture of ferruginous clay, quartz and calcareous sand	
		Cow Creek Limestone	*	80	Upper part: crossbedded beach coquina formed by oyster-shell detritus, quartz grains and chert pebbles. Middle part: silty calcarenite with carbonate concretions and quartz sand grains. Lower part: fine to coarse grained calcarenitic limestone containing oyster fragments	
		Hammett Shale	*	50	Burrowed mixture of clay, terrigenous silt, carbonate mud, particles of carbonate and dolomite.	Confining unit

*Not subdivided into members.

the accuracy of the rain gages, the instruments were periodically inspected and cleaned, and calibration checks were performed as described by the manufacturer and USGS protocols (NovaLynx Corporation, 2009; U.S. Geological Survey, 2005). Instruments not meeting calibration standards (calibration values different from expected values by more than 8 percent) were replaced. Raingages found with debris in the tipping bucket were cleaned, and the affected data were removed from the NWIS database. Measurements made by the rain gage also can be affected by environmental conditions, which can cause recorded rainfall values to differ from the actual rainfall amounts. These conditions can include high winds and result in the undercatch of rainfall (Duchon and Essenberg, 2001). During low-intensity rainfall, the measurement accuracy might be affected by losses to evaporation; during high-intensity rainfall, the accuracy might be affected by the ability of the instrument to register rainfall at the rate of input (Legates and Deliberty, 1993; Duchon and Essenberg, 2001). Other than removing anomalous values caused by instrumentation noise (anomalous values not corroborated by preceding and subsequent values), no further corrections were made to the rainfall data. On the basis of field calibration checks, the rainfall data are considered accurate to within 8 percent of actual rainfall.

The daily rainfall totals at the four sites were highly correlated during 2001–10, with an average site-to-site coefficient of determination (R^2, Helsel and Hirsh, 2002) of 0.98. Given the close proximity of the sites and similar physical site conditions (for example, similar slopes and aspects), a representative daily rainfall amount of the study area was developed by averaging the available daily rainfall totals from the four rainfall sites (hereinafter daily rainfall). Annual rainfall for the study area was calculated as the summation of the daily rainfall for the respective year (hereinafter annual rainfall). Average annual rainfall was calculated from the annual rainfall values during pre- and post-treatment periods. The daily and annual rainfall totals for the period 2001–10 are listed in appendix 1.

Streamflow

Streamflow-gaging stations were installed in the lower parts of the reference and treatment watersheds, upstream from the confluences with Honey Creek (sites 1C and 2T, respectively) (fig. 3). A third streamflow-gaging station was installed in the upper part of the treatment watershed, immediately downstream from the boundary of the Honey Creek State Natural Area and private land (site 1T). Concrete weirs constructed at the sites were used as streamflow measuring devices.

Streamflow at sites 1C, 1T, and 2T was computed at 5-minute intervals by using a theoretical stage-discharge

Figure 3. Example of streamflow channel and weir in the treatment watershed at site 2T (U.S. Geological Survey station 08167353 Unnamed tributary of Honey Creek site 2T near Spring Branch, Texas), Honey Creek State Natural Area, Comal County, Tex., June 28, 2004.

relation for each site (Kennedy 1983, 1984) and transmitted every 4 hours by the GOES to the USGS NWIS database (U.S. Geological Survey, 2010). To compute discharge, water-surface elevation (stage) was measured and recorded every 5 minutes using a nonsubmersible pressure transducer and data recorder. The recorded stage was verified during site visits by comparing recorded stage values to a reference gage mounted in the stream channel. A crest-stage gage mounted in the stream channel was used to mark the peak stage occurring during runoff events and to verify the recorded peaks. The recorded stage was corrected to the reference gage and crest-stage gage when the difference was greater than 0.01 ft (Rantz and others, 1982a; Sauer and Turnipseed, 2010).

A theoretical stage-discharge relation (rating) was developed for each of the weirs (Kennedy, 1983, 1984). From the stage-discharge relation, the corrected stage data were used to compute discharge. The computed discharges were estimated to be within 8 percent accuracy (Turnipseed and Sauer, 2010). During periods of instrument failure, discharge was estimated by using hydrographic comparisons, interpolation between recorded values, or crest-stage measurements (Rantz and others, 1982b). The rating from each site was applied to calculate discharge for the entire period of record (Bos, 1989; Hulsing, 1967; Rantz and others, 1982b). The daily mean streamflow for sites 1C, 1T, and 2T for the period 2001–10 are listed in appendixes 2A-C.

Evapotranspiration

Evapotranspiration (ET) refers to the combined processes of evaporation and transpiration. Through these coupled processes, water is converted from a liquid to a vapor and is transferred from Earth's surface to the atmosphere. Sources of water available for evaporation include open bodies of water, soil moisture, precipitation, and water condensate on surfaces. In the process of transpiration, water is transpired by plants, changing from a liquid to a vapor and passing through the stomata (Brutsaert, 1982).

The process of evapotranspiration utilizes energy from the environment and measuring this transfer of energy is one basis for determining evapotranspiration (Bowen, 1926). The energy at Earth's surface can be described by the surface-energy budget (hereinafter the energy budget). The energy budget balances the incoming and outgoing energy fluxes at Earth's surface and is in equilibrium when all sources of energy in their different states of transformation are taken into account. Assuming that energy fluxes from other sources and sinks are negligible, the simplified form of the energy budget can be expressed as follows (Brutsaert, 1982; Wilson and others, 2002):

$$R_n - G = H + \lambda E \quad , \tag{1}$$

where

R_n is net radiation, the difference between incoming and outgoing radiation, in watts per square meter;

G is soil-heat flux, in watts per square meter;

H is the sensible-heat flux, in watts per square meter; and

λE is the latent-heat flux, energy utilized in the process of evapotranspiration, in watts per square meter.

Evapotranspiration is related to the latent-heat flux (λE) and can be calculated as follows:

$$ET = 1000 * \frac{\lambda E}{\lambda_v \rho_w} \quad , \tag{2}$$

where

ET is the rate of evapotranspiration, in millimeters per second;

λE is latent-heat flux, in watts per square meter;

λ_v is latent heat of vaporization for water, in joules per kilogram; and

ρ_w is density of water, in kilograms per cubic meter.

To calculate ET, the latent-heat flux was calculated using the energy budget Bowen ratio method whereby the latent- and sensible-heat are estimated from the Bowen ratio (Bowen, 1926). The Bowen ratio (β) is defined as the ratio between H and λE, which, assuming equal turbulent bulk-transfer coefficients, also is expressible in terms of vertical gradients of air temperature and air vapor pressure. Thus, the equation for β is:

$$\beta = \frac{H}{\lambda E} = \gamma \frac{\Delta T}{\Delta e} \quad , \tag{3}$$

where

β is the Bowen ratio, dimensionless;

H is the sensible-heat flux, in watts per square meter;

λE is the latent-heat flux, in watts per square meter;

γ is the psychrometric constant, in kilopascals per degree Celsius (Radiation and Energy Balance Systems, Inc., 1996);

ΔT is the difference in air temperature at two different heights, correcting for the adiabatic lapse rate, in degrees Celsius (Radiation and Energy Balance Systems, Inc., 1996); and

Δe is the difference in vapor pressure at two different heights, correcting for the pseudoadiabatic lapse rate, in kilopascals (Radiation and Energy Balance Systems, Inc., 1996).

The Bowen ratio (β) is substituted into equation 1 and the latent-heat flux calculated by algebraic rearrangement:

$$\lambda E = \frac{R_n - G}{1 + \beta} \quad , \tag{4}$$

where all terms are as previously defined.

To obtain the meteorological and surface-energy flux data needed for the calculation of ET by the Bowen method, the following manufacturer equipment (and model number where applicable) was used at sites RWS_{ET} and TWS_{ET}: a Campbell Scientific data logger (CR23X); a Radiation Energy Balance Systems net radiometer (Q7.1), two Radiation Energy Balance Systems temperature and humidity sensors with aspirated radiation shields (THP-1), a Radiation Energy Balance Systems automatic exchange mechanism (AEM), a Met One Instruments wind speed and direction sensor (034B wind sensor), three Radiation Energy Balance Systems soil-heat flux plates (HFT-3.1), Campbell Scientific soil temperature sensors (TCAV), and two Campbell Scientific soil moisture sensors (CS615) (figs. 4 and 5).

The net radiation, R_n, is the algebraic sum of all incoming and outgoing longwave and shortwave radiation (Arya, 2001). Measurements of R_n are obtained from the net radiometer installed on a tower approximately 36 ft above the land surface.

The wind sensor is mounted 20 ft above the land surface. Wind speed was used to calculate a correction coefficient applied to the measurement of R_n (Campbell Scientific Inc., 1996). For periods when the wind sensor had instrumentation issues, the correction coefficient was calculated using the wind speed at the other site because the wind speed at sites RWS_{ET} and TWS_{ET} were correlated during pre-treatment periods (R^2 = 0.81) and post-treatment periods (R^2 = 0.72).

Soil heat flux, G, is calculated using measurements made by the soil heat flux plates, soil temperature probes, and soil moisture probes buried in the area near the tower. Three soil heat flux plates were buried at a depth of 6 in. at various exposures of shade and sun. The soil temperature probes are spaced vertically at about equal intervals in the soil layer above the soil heat flux plates. Soil moisture probes measure the volumetric water content of the soil. The default calibration of the soil moisture probes was refined using soil moisture data from the study area. Soil cores with a diameter of approximately 0.75 in. and length of about 3 to 4 in. were collected using a straight barrel sampler, placed in 4-ounce stainless steel cups, and weighed in the field. The soil cores were then transported to the Texas Water Science Center laboratory, where they were oven dried for 24 hours, re-weighed, and soil moisture was calculated (Campbell Scientific Inc., 2004). The change in

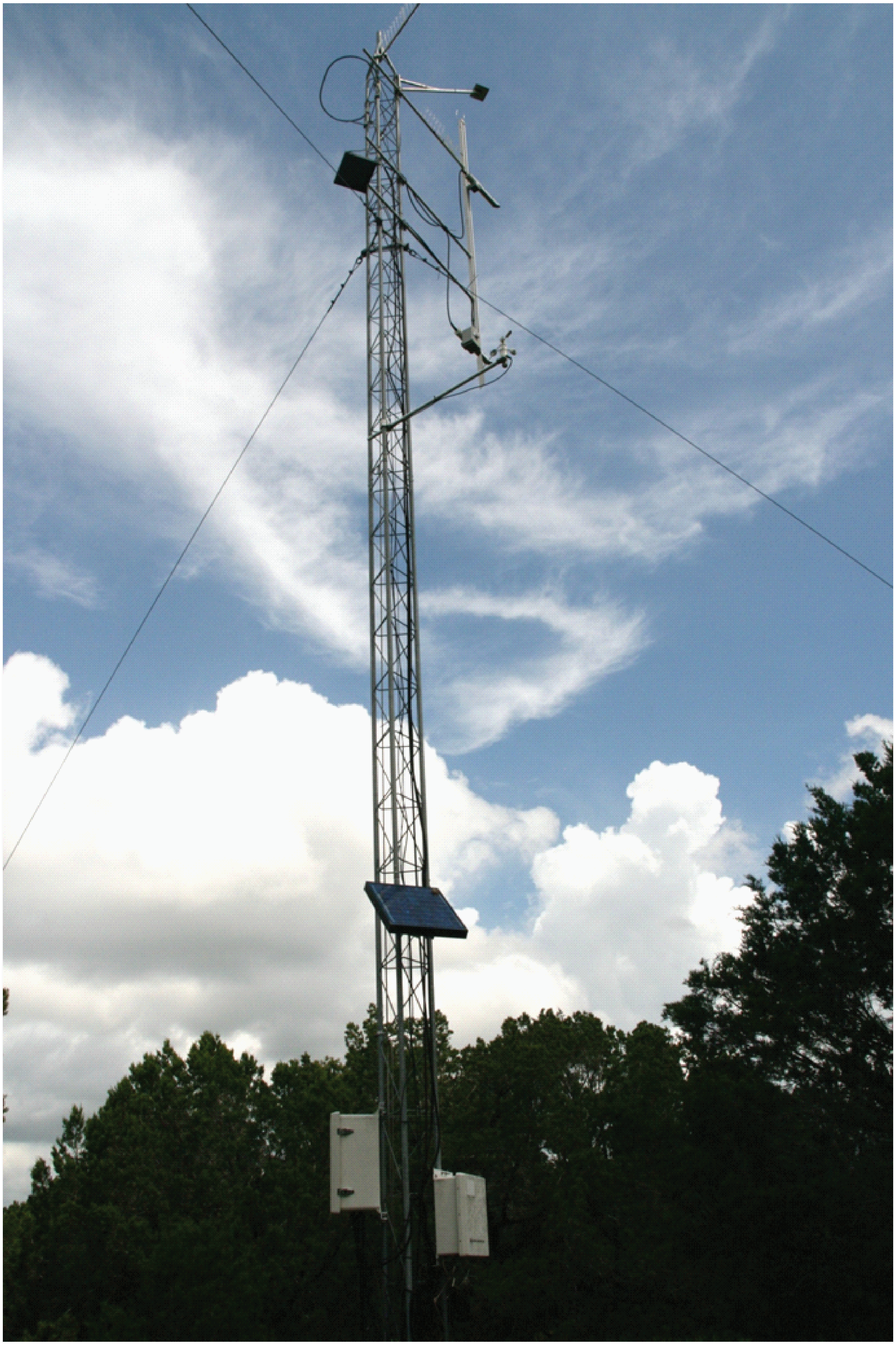


Figure 4. Evapotranspiration station in the reference watershed at site RWS_{ET} (U.S. Geological Survey station 295104098285900 Honey Creek reference evapotranspiration near Spring Branch, Texas), Honey Creek State Natural Area, Comal County, Tex., July 23, 2010.

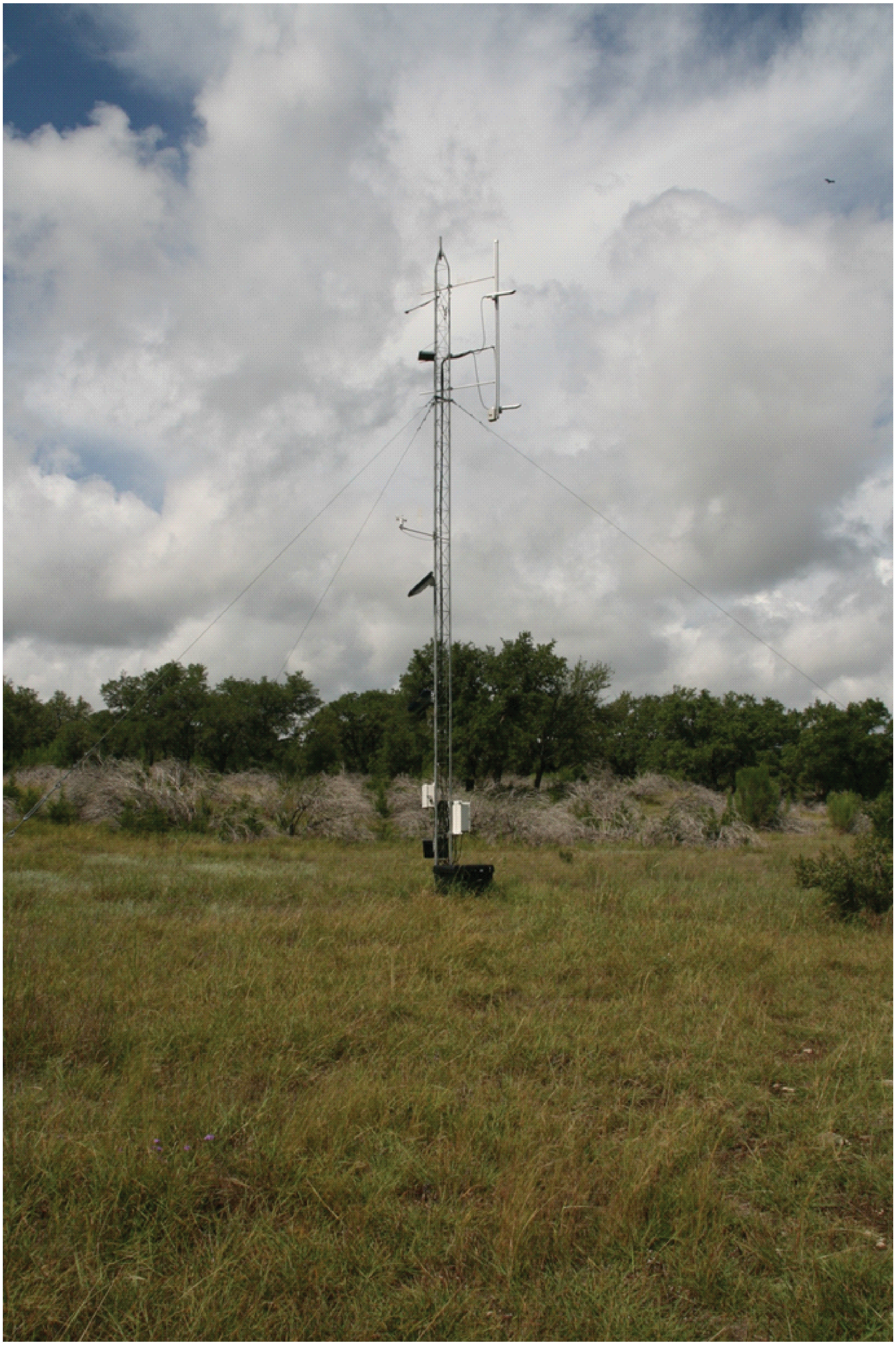


Figure 5. Evapotranspiration station in the treatment watershed at site TWS_{ET} (U.S. Geological Survey station 295102098283200 Honey Creek treatment evapotranspiration near Spring Branch, Texas), Honey Creek State Natural Area, Comal County, Tex., July 23, 2010.

heat stored in the soil solids and soil water is added to the heat flux measured by the soil heat flux plates to obtain G (Campbell Scientific Inc., 2003).

Temperature and vapor-pressure gradients, ΔT and Δe, were calculated from the two temperature and humidity sensors mounted on the AEM. The bottom temperature and vapor-pressure sensors are positioned 28 ft above ground surface and the top sensors are 34.6 ft above ground surface (6.6 ft separation between the sensors). The ΔT and Δe values were corrected for the adiabatic lapse rate and pseudoadiabatic lapse rate, respectively, taking into account the 6.6-ft separation in sensor heights (Radiation and Energy Balance Systems, Inc., 1996). The AEM exchanges the position of the sensors every 15-minutes. Averaging the temperature and vapor-pressure gradients over a 30-minute period (where the sensors switch positions at 15-minute intervals) removes possible bias between sensors (Radiation and Energy Balance Systems, Inc., 1996).

To maintain the instruments, site visits were made at 4- to 6-week intervals. During the visits, data were retrieved from the data logger and the instruments were inspected for damage. The net radiometer windshields were inspected and flushed with deionized water to remove dust. The polyethylene windshields of the net radiometers were replaced about every 3 months, or when the windshields were damaged (for example, by hailstorms or bird pecks). Evapotranspiration was not calculated for periods when the R_n sensor was damaged. The THPs were inspected during visits and washed every 6 months to remove accumulated dust.

Evapotranspiration-related parameters were measured every 20 seconds, and 15-minute averages were calculated and recorded by the data-logger. The calculated ET rates utilize the two previous, 15-minute averaged ET-related parameters (30-minute moving average), and are reported on a 15-minute interval (Radiation and Energy Balance Systems, Inc., 1996). The calculated ET rates were determined using data from several different instruments and were considered to be accurate to within about 5 percent (Radiation and Energy Balance Systems, Inc., 1996). ET rates were calculated for daytime periods only. The Bowen ratio calculation of ET can be problematic during nighttime because of the required high precision in measurements needed (Jensen and others, 1990). Furthermore, the focus of this study was to evaluate differences in ET rates as a result of changes in vegetation type, a difference which is expected to be greatest during daytime hours. Therefore, ET rates were calculated for daylight hours (ET rates were set to zero for non-daytime periods), where daytime is defined as the time between one hour after sunrise and one hour before sunset.

The calculated ET rates were examined to identify values that were physically implausible or suspected to be erroneous. The ET calculation becomes mathematically unstable as the Bowen ratio (β) approaches -1 because it is undefined at -1. This results in extreme calculated values of λE (equation 4). Consequently, the calculated evapotranspiration rate was rejected when the β was within the interval of $-1.3 < \beta < -0.7$. Additionally, following Ohmura (1982), if fluxes of Δe and ΔT are not physically consistent with the calculated sign of available energy, the evapotranspiration rate was rejected. There were periods where the measured parameters passed the above mentioned tests, however, the calculated ET rates were unrealistic and were rejected (for example, negative ET values). Negative ET rates were not considered physically plausible and were rejected. Additionally, because of instrumentation issues, there are occasional data gaps and the calculated ET rates for the period of missing data were rejected. Rejected ET rates were set to zero. If the calculated ET rates were rejected during a time period of less than one hour, and the data on either side of the questionable period were considered good, a linear interpolation was used to fill the data gap.

Daily ET rates were calculated as the sum of the calculated ET rates during daytime periods. Daily ET rates for days where more than 25 percent of the calculated ET rates were rejected are not reported. Daily ET rates at sites RWS_{ET} and TWS_{ET} for the period 2002–10 are listed in appendixes 3A and 3B.

The fetch area is defined as the area extent that might affect evapotranspiration measurements (for example, the type of vegetation or presence of an open water body in the fetch area may influence the evapotranspiration measurements). The fetch area upwind from the sensor can potentially influence the measurement, although as the distance from the sensor increases, its percent contribution to the measurements decreases. That is to say, the farther away from the sensor, the less that area affects the measurement. The fetch area traditionally is approximated as 100 times the vertical height of the temperature-humidity sensor (100:1 fetch-to-height ratio) (Heilman and others, 1989; Burba and Anderson, 2010). However, Heilman and others (1989) demonstrated the fetch-to-height ratio can be as low as 20:1 for Bowen-ratio systems. Stannard (1997) developed a theoretically based determination of fetch requirements for site-specific conditions. The percent equilibrium of the Bowen ratio is calculated as (Stannard, 1997):

$$\%Eq = 100e^{-z\left[\ln\left(\frac{z}{z_0}\right)-1+\left(\frac{z_0}{z}\right)\right]/x_i k^2\left(1-\frac{z_0}{z}\right)} \tag{5}$$

where

- $\% Eq$ is the percent equilibration of the Bowen ratio, dimensionless,
- e is the base of the natural logarithm;
- $\ln$ is the natural logarithm;
- z is the geometric mean of the sensor heights above the zero-plane displacement height, in feet;
- z_0 is the roughness length, in feet;
- k is von Karman's constant, dimensionless; and
- x_i is the upwind distance, in feet.

The geometric mean of the sensor heights was approximated as the square root of the product of the heights above the zero-plane displacement height, where the zero-plane displacement height was calculated as 0.65 times the canopy height (Campbell, 1972). The roughness length was calculated as 0.13 times the canopy height for dense canopies (Campbell, 1972). A 20-ft canopy height of mature ashe juniper was used for the reference watershed and treatment watershed during the pre-treatment period (Phillip Wright, Natural Resources Conservation Service, written commun., 2011) (fig. 6). A roughness length of 1.31 ft, representative of savannah scrub (Brutsaert, 1982), was used for the treatment watershed during the post-treatment period (fig. 7). A value of 0.4 was used for von Karman's constant (Arya, 2001).

To account for the area affecting the evapotranspiration measurements, the percent equilibration (Stannard, 1997) was calculated for each watershed. This was calculated by dividing the 360 degree radius into 16 equal sectors (for example, 0 degrees for north, 22.5 degrees for north-northeast, 45 degrees for northeast). The distance from the tower to the edge of the watershed was measured for each of the 16 sectors. These distances were input into equation 5 as x_i, and the percent equilibration was calculated. The percent of time that the winds were coming from a given sector during daytime hours (WRPLOT View, 2010; table 3) were multiplied by the percent equilibration. These weighted percent equilibriums for each sector were summed to determine the total percent equilibrium. Evapotranspiration rates during periods of light variable winds (that is, less than 1.6 ft/sec (0.5 m/sec)) were assumed to be representative of their respective watershed. This results in the percent equilibration that is contained within the respective watershed.

Because the vegetation types are mostly similar beyond the watershed boundary (for example, field observations and aerial photographs (U.S. Department of Agriculture, Farm Service Agency, 2010) indicate dense ashe juniper vegetation is present to the west and south of the reference watershed boundary), the x_i distances were adjusted to extend to an edge of similar vegetation representative of the watershed in order to calculate a more representative percent equilibrium of the watershed. For example, the reference watershed x_i distance to the west was extended to 2,000 ft where ashe juniper are present, approximately coincident with the Honey Creek State Natural Area boundary. Distances for the north through the south sectors (counter clockwise) also were extended to 2,000 feet, and the distances for the southeast and south-southeast sectors were reduced to the edge of the ashe juniper cover. For the treatment watershed during the post-treatment period,

Figure 6. Landscape of reference watershed viewed from the top of the tower at the reference watershed evapotranspiration site RWS_{ET} (U.S. Geological Survey station 295104098285900 Honey Creek reference evapotranspiration near Spring Branch, Texas), Honey Creek State Natural Area, Comal County, Tex., January 19, 2011.

Figure 7. Landscape of treatment watershed viewed from the top of the tower at the treatment watershed evapotranspiration site TWS_{ET} (U.S. Geological Survey station 295102098283200 Honey Creek treatment evapotranspiration near Spring Branch, Texas), Honey Creek State Natural Area, Comal County, Tex., January 19, 2011.

distances to the south/southeast and southeast were extended to the southern Honey Creek State Natural Area boundary because ashe juniper were removed in that area in 2001 by the Natural Resources Conservation Service as part of a separate project (Phillip Wright, Natural Resources Conservation Service, written commun., 2011). Distances for the south and south/southwest sectors were extended to 3,094 ft and 3,166 ft, respectively, as that area was also largely free of ashe juniper. Using the updated distances, the percent equilibration was 91 and 94 percent for the reference and treatment watersheds, respectively, during the pre-treatment period, and 91 percent and 88 percent for reference and treatment watershed, respectively, during the post-treatment period. This indicates that during pre-treatment periods, the fetch is largely representative of the respective watershed. During post-treatment periods, the fetch at the reference watershed is still largely representative of the watershed, but the slightly lower percent equilibrium at the treatment watershed indicates some "contaminating" area outside of the treatment watershed may be influencing the ET measurements at site TWS_{ET}. The effects of a contaminating area are discussed later in this report in the section, "Determination of Change in Water from Evapotranspiration."

Water Quality

Collection and Processing of Rainfall and Stormflow Samples

Rainfall water-quality samples were collected at site RQW, and stormflow water-quality and suspended-sediment samples were collected at sites 1C, 1T, and 2T. Rainfall samples collected at site RQW were analyzed for the physical properties (pH and specific conductance), major ions, nutrients, and the isotope ratios for hydrogen (deuterium/protium [δD]) and for oxygen (oxygen-18/oxygen-16[ŭ^{18}O]). Stormflow samples collected at sites 1C, 1T, and 2T were analyzed for the field properties (pH and specific conductance), major ions, nutrients, total organic carbon, and the isotope ratios for hydrogen (deuterium/protium [δD]) and for oxygen (oxygen-18/oxygen-16[ŭ^{18}O]) and suspended sediment (suspended sediment concentration and percent of sand and fine-sized material). Sample preparation, collection, and processing techniques followed standard USGS protocols (U.S. Geological Survey, variously dated).

Table 3. Percentage of wind direction by sector during pre- and post-treatment periods at the reference watershed evapotranspiration site RWS_{ET} (U.S. Geological Survey station 295104098285900 Honey Creek reference evapotranspiration near Spring Branch) and treatment watershed evapotranspiration site TWS_{ET} (U.S. Geological Survey station 295102098283200 Honey Creek treatment evapotranspiration near Spring Branch, Tex.).

Sector, in degrees from north	Pre-treatment (2002–04)[1]		Post-treatment (2005–10)	
	RWS_{ET} (percent)	TWS_{ET} (percent)	RWS_{ET} (percent)	TWS_{ET} (percent)
0.0	6.0	6.0	5.4	5.3
22.5	6.7	6.4	5.0	4.3
45.0	6.5	5.4	4.6	3.7
67.5	5.1	5.0	4.9	3.9
90.0	4.8	4.4	5.9	4.1
112.5	6.0	4.8	6.8	5.1
135.0	8.0	10.0	9.3	12.8
157.5	12.1	15.6	9.7	17.7
180.0	12.9	12.7	8.7	12.0
202.5	7.7	5.5	5.7	5.2
225.0	4.7	3.4	4.6	3.1
247.5	2.5	2.5	3.9	1.8
270.0	2.1	2.5	4.4	2.2
292.5	2.7	2.6	4.4	2.6
315.0	3.1	3.4	5.2	3.6
337.5	4.2	4.9	5.4	4.7
Frequency of calm winds	4.9	4.9	6.1	9.0

[1]Time period during pre-treatment limited to time periods of evapotranspiration data–evapotranspiration data were not available during 2001.

Rainfall water-quality samples were collected by using an automated Aerochem Metrics 301 atmospheric deposition sampler (Graham and Robertson, 1990). When rainfall occurs, a moisture sensor activates a mechanism to uncover a clean polyethylene collection bucket. When rainfall ends, the sampler mechanism recovers the collection bucket to prevent evaporation and contamination of the sample. Each rainfall sample is a single-composite sample representing average quality conditions during the runoff event. The samples were retrieved as soon as practical after a runoff event, chilled, and transported to the USGS South Texas Program Office in San Antonio for processing. Subsamples of rainwater were drawn from the collection bucket and shipped for analysis to the USGS National Water Quality Laboratory (NWQL) in Denver, Colo., and the USGS Reston Stable Isotope Laboratory (RSIL) in Reston, Va., depending on the constituent.

Stormflow water-quality and suspended-sediment samples were collected during stormflow using a point-integrated sampling method. The sample was drawn through a fixed intake mounted at a mid-point in the stream channel using a suction-lift type automatic sampler. The automatic sampler was programmed to begin sampling when flow occurred (stage rose above the crest of the weir), then filled as many as 24 1-liter bottles at variable time intervals during the runoff event. The samples were collected at shorter intervals at the beginning of the runoff event (coinciding with increasing discharge and the peak of discharge) and at longer time intervals at the end of the runoff event (coinciding with decreasing discharges). The samples were retrieved at the end of the runoff event or soon after all 24 samples were obtained, the samples were then chilled, and transported to the South Texas Program Office for processing.

To process the samples, as many as 5 of the 1-liter samples were selected from the set of 24 for analysis of suspended sediments. These samples were chosen to represent different phases of the runoff hydrograph (rising stage, rising stage midway to peak, peak stage, falling stage midway of recession after peak, and falling stage at the tail of the recession). The suspended-sediment samples were shipped to the USGS sediment laboratory in Iowa City, Iowa to determine suspended-sediment concentration and for the separation of sand and fine-sized material. The remaining 1-liter samples were flow-weighted and composited into a single water-quality sample representing the event mean concentration. To composite the samples, an aliquot, proportional to the amount of flow at the time the sample was taken, was measured from each of the remaining bottles. The measured volumes were then poured into a Teflon-lined stainless-steel churn. As the composited sample was mixed in the churn, subsamples were drawn off for analysis by the USGS NWQL or the USGS RSIL, depending on the constituent.

Sample Analysis

Composited water-quality samples were analyzed for the field properties, pH, and specific conductance by the USGS South Texas Program Office using methods described in the USGS National Field Manual for the Collection of Water Quality Data (U.S. Geological Survey, variously dated). Major ions, nutrients, and total organic carbon were analyzed and reported by the USGS NWQL in Denver, Colo. (appendix 4A-D). Major inorganic ions were analyzed using methods described by Fishman and Friedman (1989) and Fishman (1993). Nutrients were analyzed using methods documented by Fishman (1993), O’Dell (1993), and Patton and Truitt (2000). Total organic carbon was analyzed using methods described by Clescari and others (1998). Samples for analysis of the environmental isotopes were submitted to the USGS Reston Stable Isotope Laboratory in Reston, Va. δD was analyzed using a gaseous hydrogen equilibration technique at 30°C (Coplen and others, 1991). ŭ^{18}O was analyzed using a carbon dioxide-water equilibration technique (Epstein and

Mayeda, 1953). The hydrogen and oxygen isotope results are reported as δD and $\delta^{18}O$ relative to the Vienna Standard Mean Ocean Water (Coplen, 1994).

Quantified values of constituents represent measured concentrations greater than or equal to the detection level, as determined by the laboratory at the time of analysis, and were reported as specific numerical values. Censored values represent measured concentrations less than the reporting level, as determined by the laboratory at the time of analysis, were and reported as "<RL", where RL is the numerical reporting level. For each constituent, the numerical values of the detection and reporting levels can vary over time. In some cases, the reporting level used by the laboratory was greater than the detection level, so some quantified values can be reported which are less than censored values for the same constituent. In these cases the quantified value is reported with an "E" remark code.

Suspended-sediments samples were analyzed using methods described by Guy (1969) and Matthes and others (1991) to determine suspended sediment concentration in milligrams per liter and separate sand from fine-sized material. Sand-size material is defined as particles sieved to a size between 0.0625 mm and 0.125 mm and fine-size material (silt and clay) is defined as particles sieved to less than 0.0625 mm. The percentages by weight of the suspended-sediment concentration composed of sand and fine-size material were determined (Guy, 1969) (appendix 5A-C).

Quality Assurance

USGS quality-assurance methods were followed in the collection and processing of the water-quality and suspended-sediment samples to minimize potential sample contamination, document possible biases, and preserve the sample integrity (U.S. Geological Survey, variously dated). To minimize the potential contamination of the environmental samples, the autosamplers and collection bottles were cleaned between times when the samples were collected. To clean the autosamplers, the sample tubing was flushed with a soap solution, rinsed with deionized water, flushed with a 5-percent solution of hydrochloric acid, and followed by another rinse with deionized water. Sample-collection bottles were washed in the laboratory following the same procedures. To document possible contamination, six field-equipment blanks were collected during the study. These samples were collected using inorganic blank water (certified ASTM Type I deionized water) provided by the NWQL. The blank water was pumped through the autosampler tubing and into 1-liter collection bottles. The equipment-blank samples were processed following the same methods as for the environmental samples and analyzed by the NWQL for major ions, nutrients, and total organic carbon. The majority of reported concentrations of these constituents were less than the reporting levels. The few exceptions where the equipment-blank samples were greater than the reporting levels, the concentrations were an order of magnitude less than those of the environmental sample concentrations and, consequently, its influence on the environmental sample concentration was considered negligible (appendix 4E). There were four environmental sample data points that were anomalously high (more than one order of magnitude greater in concentration than any of the other samples). The data points were filtered chloride and filtered sulfate concentrations collected at site 1C on June 27, 2004; and filtered orthophosphate as phosphorus and filtered phosphorus collected at site RQW on Aug. 16, 2007. These data points were not considered plausible, thus they are not presented in subsequent data discussions, but are included in the appendixes for completeness.

Hydrologic Budget

By removing the ashe juniper and allowing native grasses to reestablish in the area as a brush management conservation practice, the hydrology in the watershed might change. This idea is based on a simplified mass balance approach of the hydrologic cycle (Zhang and others, 2002):

$$RF = SW + ET + GW + \Delta S \tag{6}$$

where

- RF is the rainfall into the system;
- SW is the surface-water runoff out of the system;
- ET is the evapotranspiration out of the system;
- GW is the groundwater recharge out of the system; and
- ΔS is the change in storage in the system

Assuming that long-term average annual change in storage in the system is negligible, equation 6 simplifies to:

$$RF = SW + ET + GW \tag{7}$$

In this simplified approach where rainfall accounts for the water coming into the system, rainfall is distributed to surface-water runoff (streamflow), evapotranspiration (combination of evaporation and transpiration), or groundwater recharge (subsurface flow that contributes to the groundwater table or contributes to spring discharge downstream from the study area). If the rainfall remains constant, but the evapotranspiration rates change because of a change in vegetation cover, then the surface-water or groundwater components of the hydrologic budget will change.

The components of the hydrologic budget measured in this study include rainfall, streamflow, and evapotranspiration. Groundwater recharge was not directly measured, but was calculated using a simplified mass balance approach to the hydrologic cycle (eq. 7). The effects of brush management on the watersheds were evaluated by comparing the hydrologic budgets (and their respective components) of the two watersheds during pre- and post-treatment periods.

Rainfall

The annual rainfall ranged from 19.27 in. (2008) to 54.23 in. (2002). The average annual rainfall was approximately 43 in. during pre-treatment years and 30 in. during post-treatment years. The most extreme event during the study period occurred in 2002, when more than 22 in. of rainfall fell during a 2-week period, which accounted for more than 40 percent of the annual rainfall for that year. Extreme events can exert heavy leverage on calculated annual averages, which should be interpreted with caution.

Streamflow

Ephemeral streams in the study area result from rainfall events—perennial streams are not present in the study area. Because the streams in the study area are ephemeral and only flow during periods of stormwater runoff, base flow is considered negligible. The streamflow-gaging stations at sites 1C and 2T measure the discharge from the respective watersheds. To evaluate the effects of brush management on streamflow, the rainfall and computed unit runoff were examined for possible relations. Computed unit runoff normalizes the streamflow to allow comparisons between watersheds. Daily unit runoff (in inches) was computed by multiplying the daily mean streamflow in cubic feet per second (ft^3/s) by the number of seconds in a day and dividing by the drainage area of the respective streamflow-gaging station. The rainfall and unit runoff for each event were calculated as the summation of the respective daily values for the storm event (hereinafter event rainfall and event unit runoff, respectively). Only events resulting in daily mean streamflow amounts greater than 0.01 ft^3/s at both streamflow-gaging stations (sites 1C and 2T) were included in the following analyses. Because event rainfall and event unit runoff were non-normally distributed, a natural logarithm transformation, ln(x), was applied to both datasets.

The pre-treatment event unit runoff (dependent variable) exhibited a linear relation with event rainfall (independent variable) at both watersheds, with a R^2 of 0.57 and 0.70 at sites 1C and 2T, respectively (p value less than 0.01 [$p < 0.01$] for both regressions) (fig. 8*A*). The residuals are approximately uniformly distributed, indicating a natural logarithmic

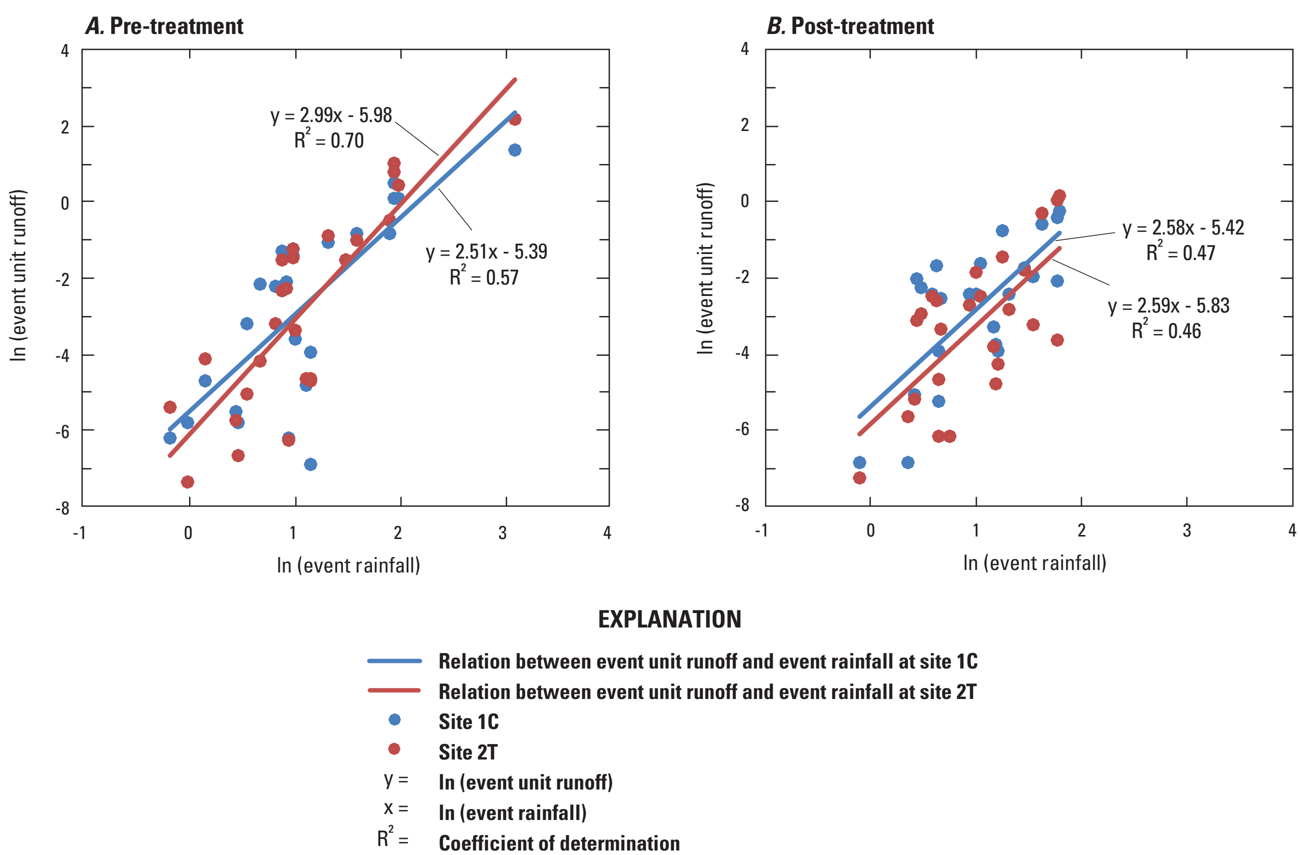


Figure 8. Event unit runoff compared to event rainfall during *A*, pre- and *B*, post-treatment periods at the reference watershed streamflow-gaging station site 1C (U.S. Geological Survey station 08167347 Unnamed tributary of Honey Creek site 1C near Spring Branch, Texas) and the treatment watershed streamflow-gaging station site 2T (U.S. Geological Survey station 08167353 Unnamed tributary of Honey Creek site 2T near Spring Branch, Tex.). Both datasets have been transformed through natural logarithm (ln), and the axes are ln scale.

transformation is appropriate for these data. The two linear regression lines are not statistically different from each other at the 99-percent confidence level (that is, the null hypothesis that both lines are the same cannot be rejected) (Zar, 1984).

The post-treatment event unit runoff at sites 1C and 2T also exhibited a statistically significant linear relation with event rainfall (fig. 8*B*). The R^2 was 0.47 and 0.46 for 1C and 2T, respectively ($p < 0.01$ for both regressions). The residuals are approximately uniformly distributed, indicating a natural logarithmic transformation is appropriate for these data. The two linear regression lines are not statistically different from each other at the 99-percent confidence level (that is, the null hypothesis that both lines are the same cannot be rejected) (Zar, 1984).

These results are consistent with previous studies that did not find a difference in the streamflow to rainfall relation as a result of brush removal at a small plot or small watershed scale (that is, less than 37 acres) (Dugas and others, 1998; Wilcox and others 2005). The relation between unit runoff and rainfall is influenced by more than rainfall amount, as the rainfall intensity and duration of intensity also will likely play a role. Further, the antecedent site conditions will directly affect the amount of water that may reach the stream channel. Antecedent conditions include (but are not limited to) the soil moisture content immediately prior to the rainfall event and the type and amount of vegetation cover that can intercept rainfall, which in turn is partially dependent on the seasonal growing cycle (for example, when plants are budding compared to full leaf extent).

Evapotranspiration

The daily evapotranspiration rates at sites RWS_{ET} and TWS_{ET} exhibited a strong seasonal cycle during the pre- and post-treatment periods, with substantial intra- and interannual variability (figs. 9*A,B*). Daily evapotranspiration rates generally were lowest in the winter months when solar insolation was low (Bendta and others, 1981). Daily evapotranspiration rates began increasing around April 1, coinciding with the beginning of the growing season, and reaching a maximum around July 7. The daily ET rates decreased around October 1, coinciding with the end of the growing season. While these patterns in daily ET rates are observed each year, daily ET rates can vary substantially from day to day and year to year depending on weather patterns (for example, available energy, available moisture, drought periods). An example of this is in 2006 when drought conditions were present in the peak of the growing season (summer). The daily ET rates at both sites reached about 4 mm/d by mid-June, and then as water availability decreased, the daily ET rates began to decrease and reached near winter levels of less than1.5 mm/d by mid-August. After rainfall events in late-August and early-September, the daily ET rates returned to normal seasonal levels. Hence, vegetation at sites RWS_{ET} and TWS_{ET} exhibit the ability to influence the daily ET rates in response to environmental conditions.

Data gaps are present in the daily ET rate datasets from sites RWS_{ET} and TWS_{ET}. These data gaps are generally the result of instrument failure. The data gaps in the daily ET datasets from sites RWS_{ET} and TWS_{ET} may or may not be coincident in time. Consequently, annual total ET rates were not calculated, as each year had at least one day of missing data. Instead, an average seasonal cycle was developed using a first order Fourier transformation function (hereinafter Fourier transformation) that was fit to the data (TIBCO Software Inc., 2011):

$$ET(t) = B_1 sin(2\pi t) + B_2 cos(2\pi t) + B_0, \tag{8}$$

where

- $ET(t)$ is the average evapotranspiration at time t, in millimeters per day;
- t is the time, in decimal year;
- B_1, B_2, B_0 are parameters derived empirically to fit the data, in millimeters per day;
- *sin* is the sine trigonometric function;
- *cos* is the cosine trigonometric function; and
- π is the mathematical constant approximately equal to 3.14.

Four Fourier transformations were developed (RWS_{ET} and TWS_{ET} for pre- and post-treatment periods) (fig. 10, table 4). These Fourier transformations represent the average ET seasonal cycle (based on Julian day; January 1 is day 1 and December 31 is day 365) over the time period that incorporates wet years, dry years, and intermediate years. Using the Fourier transformations, the average annual ET rate was computed by multiplying B_0 by 365 days for the pre- and post-treatment periods. The average annual ET rate at site RWS_{ET} during pre-treatment was 721 mm/yr (28.39 in/yr), and 645 mm/yr (25.39 in/yr) during the post-treatment period. The average annual ET rate at site TWS_{ET} during pre-treatment was 693 mm/yr (27.28 in/yr), and 564 mm/yr (22.21 in/yr) during the post-treatment period.

Lastly, to ensure that the Fourier transformations did not overly smooth unique month-to-month or year-to-year variability, monthly average ET rates at each site were calculated for all possible months where data were present. The monthly average ET rates were averaged together to develop a monthly composite–a generalization of the ET annual cycle (by month) (fig. 10). These monthly composites were used to calculate composite average annual ET rates. The composite average annual ET rates calculated at sites RWS_{ET} and TWS_{ET} during pre- and post-treatment periods were within 2 percent of the respective Fourier transformations annual averages. Hence, though the Fourier transformations smooth some unique month-to-month variability for a given site for given year, they are considered representative of the general patterns in ET during the study period.

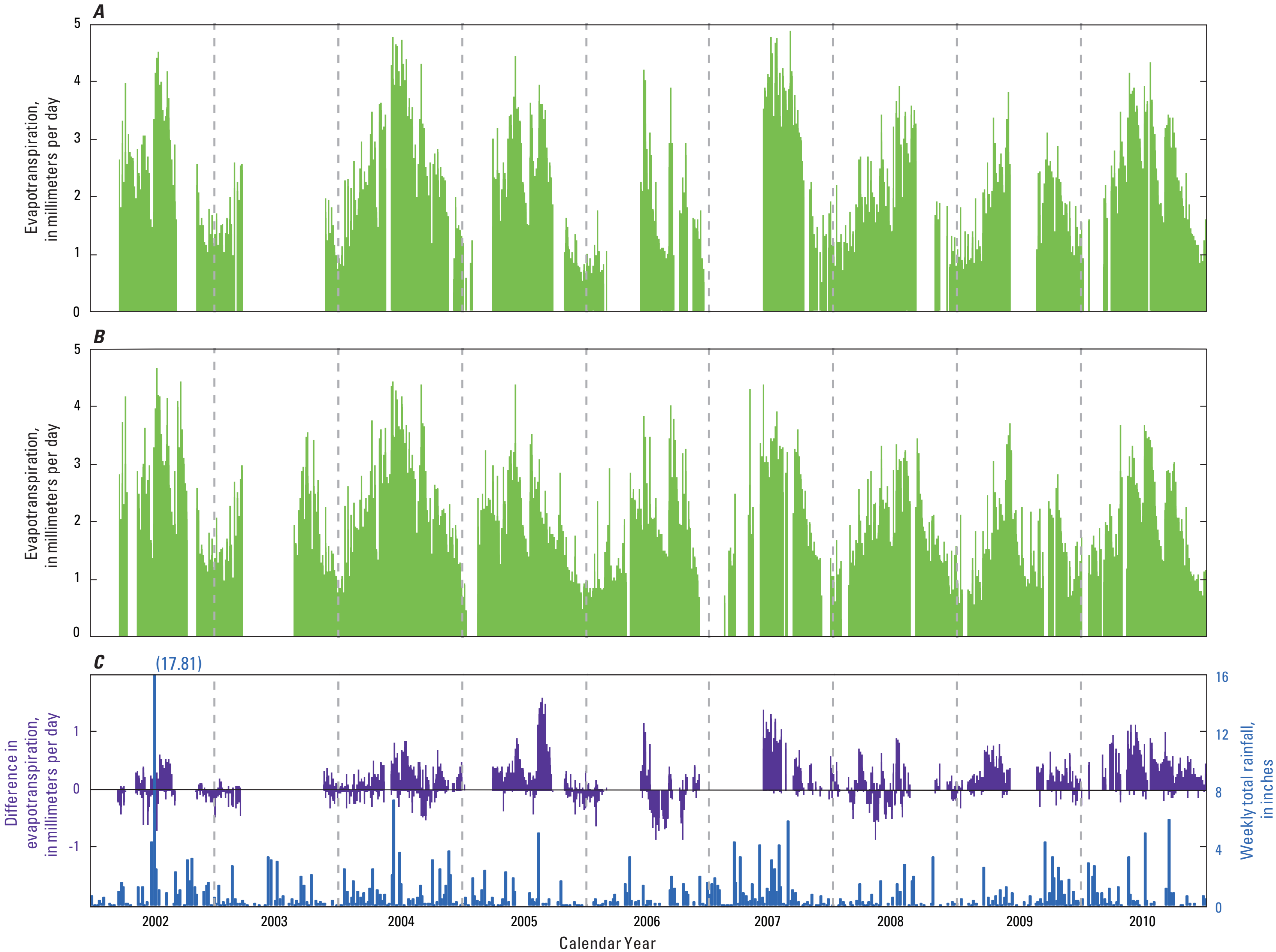


Figure 9. Daily evapotranspiration data at A, the reference watershed evapotranspiration site RWS_{ET}, (U.S. Geological Survey station 295104098285900 Honey Creek reference evapotranspiration near Spring Branch, Texas), and B, the treatment watershed evapotranspiration site TWS_{ET}, (U.S. Geological Survey station 295102098283200 Honey Creek treatment evapotranspiration near Spring Branch, Texas). C, the difference in evapotranspiration between the sites (RWS_{ET} minus TWS_{ET}).

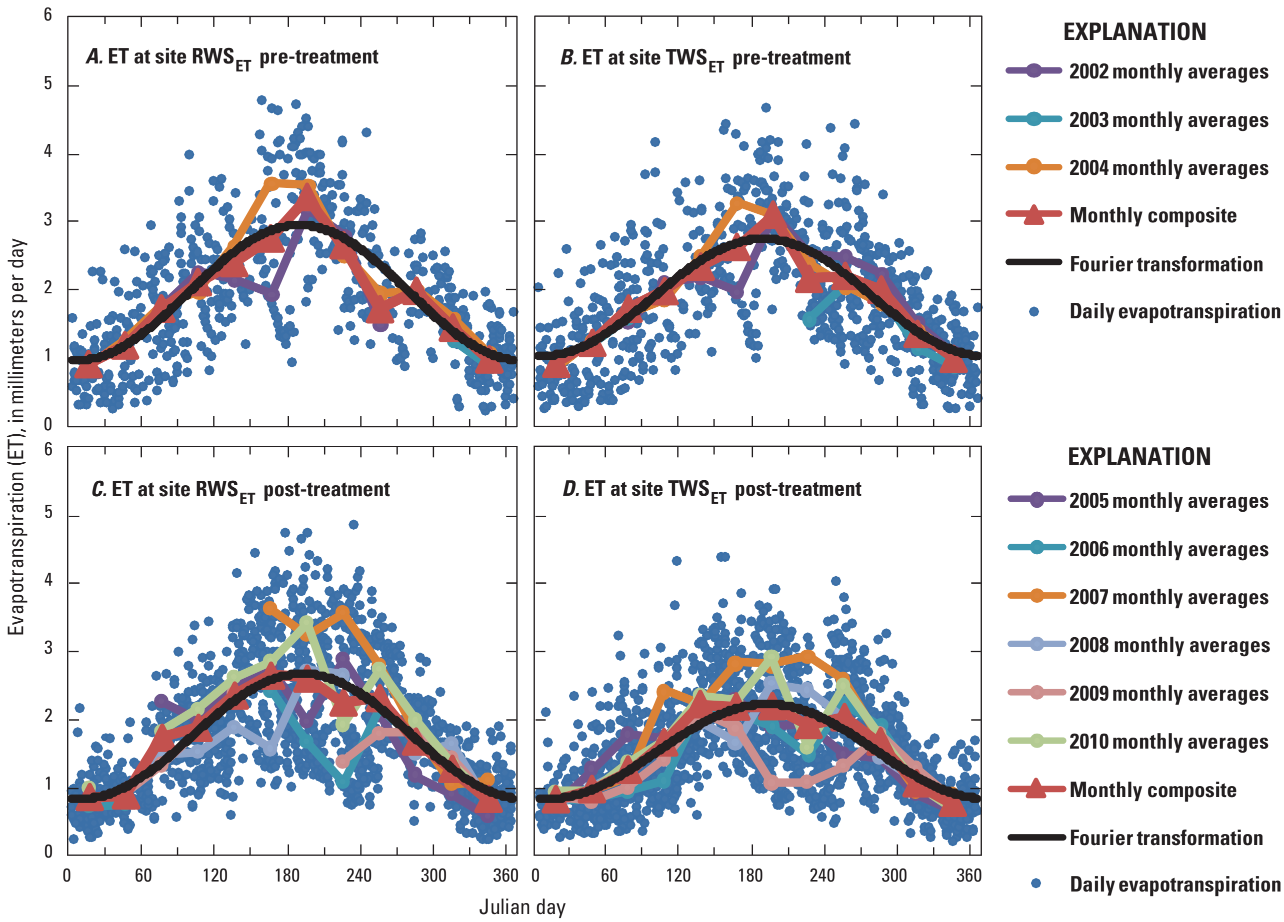


Figure 10. Annual cycle of evapotranspiration data by Julian day during pre- and post-treatment periods at the reference watershed evapotranspiration site RWS$_{ET}$ (U.S. Geological Survey station number 295104098285900 Honey Creek reference evapotranspiration near Spring Branch, Texas) during pre-treatment period, and the treatment watershed evapotranspiration site TWS$_{ET}$ (U.S. Geological Survey station 295102098283200 Honey Creek treatment evapotranspiration near Spring Branch, Tex.).

Table 4. Fourier transformation parameters of the evapotranspiration data during pre- and post-treatment periods at the reference watershed evapotranspiration site RWS$_{ET}$ (U.S. Geological Survey station 295104098285900 Honey Creek reference evapotranspiration near Spring Branch, Texas) and treatment watershed evapotranspiration site TWS$_{ET}$ (U.S. Geological Survey station 295102098283200 Honey Creek treatment evapotranspiration near Spring Branch, Tex.).

Site identifier (fig. 2)	U.S. Geological Survey station number	Time period[1]	Parameter B_1 (millimeters per day)	Parameter B_2 (millimeters per day)	Parameter B_0 (millimeters per day)
RWS$_{ET}$	295104098285900	Pre-treatment (2002–04)	-0.0798	-0.9946	1.9759
TWS$_{ET}$	295102098283200	Pre-treatment (2002–04)	-.0672	-.8799	1.8983
RWS$_{ET}$	295104098285900	Post-treatment (2005–10)	-.1269	-.9167	1.7666
TWS$_{ET}$	295102098283200	Post-treatment (2005–10)	-.0846	-.6975	1.5457

[1]Time period during pre-treatment limited to time periods of evapotranspiration data–evapotranspiration data were not available during 2001.

Statistical Analysis of Evapotranspiration

To assess if the observed daily ET rates had an effect on the hydrologic budget, the daily ET rate time series were examined for differences between watersheds as well as pre- and post-treatment. Parametric and non-parametric tests used to evaluate datasets were tested for statistical significance based on the resulting p-value (for example, $p < 0.01$ indicates the null hypothesis can be rejected at the 99-percent confidence level).

The non-parametric Wilcoxon signed-rank test for match pairs (hereinafter, Wilcoxon matched pairs; Wilks, 2006) was utilized to assess if a statistically significant difference is observed between the post-treatment daily ET rates at the two evapotranspiration sites. The test includes all available paired daily ET data—if daily ET data from both sites are not available, then that day was not included in the test statistic. Though non-uniformly distributed data gaps have a potential to affect the results, the data gaps of paired daily ET data are assumed to be spread relatively evenly throughout the study period (fig. 11).The null hypothesis is both sites exhibit the same daily ET rates during the post-treatment period. The Wilcoxon matched pairs test results indicate a statistically significant difference between the daily ET rates post-treatment (two-sided, $p < 0.01$; that is, reject null hypothesis), with the daily ET rates at site RWS_{ET} being greater than the daily ET rates at site TWS_{ET}.

Pre-treatment daily ET rates also were evaluated using the Wilcoxon matched pairs test. The pre-treatment test results also indicated that the daily ET rates at site RWS_{ET} were statistically different (greater) than the daily ET rates at site TWS_{ET} (two sided, $p < 0.01$). This indicates that though a paired watershed approach was used, the watersheds were not identical. Hence, differences in the watershed-specific environmental characteristics might be affecting the differences in the measured ET rates such that the observed differences are not solely a result of brush management (hereinafter referred to as a potential site bias).

To evaluate if the daily ET rates in one watershed have changed in time as a result of brush management, the non-parametric Wilcoxon-Mann-Whitney rank-sum test (hereinafter Wilcoxon-Mann-Whitney; Wilks, 2006) was applied to each evapotranspiration dataset separately. The null hypothesis is the daily ET rate during the pre-treatment period at the site is the same or less than during the post-treatment time period. The test results indicate a statistically significant difference (one-sided, $p < 0.01$) between the daily ET rates (that is, reject null hypothesis) at site TWS_{ET}, with the post-treatment ET rates being less than the pre-treatment ET rates.

As with the previous test, this finding may be the result of more than just a change in vegetation type (ashe juniper to grasses). Specifically, daily ET rates are linked to climate conditions. This test assumes that all other factors have remained constant in time. However, measured average annual rainfall was greater during the pre-treatment period than during the post-treatment period. Hence, climate variability might be affecting the differences in the daily ET rates observed over time (hereinafter referred to as a potential climate variability).

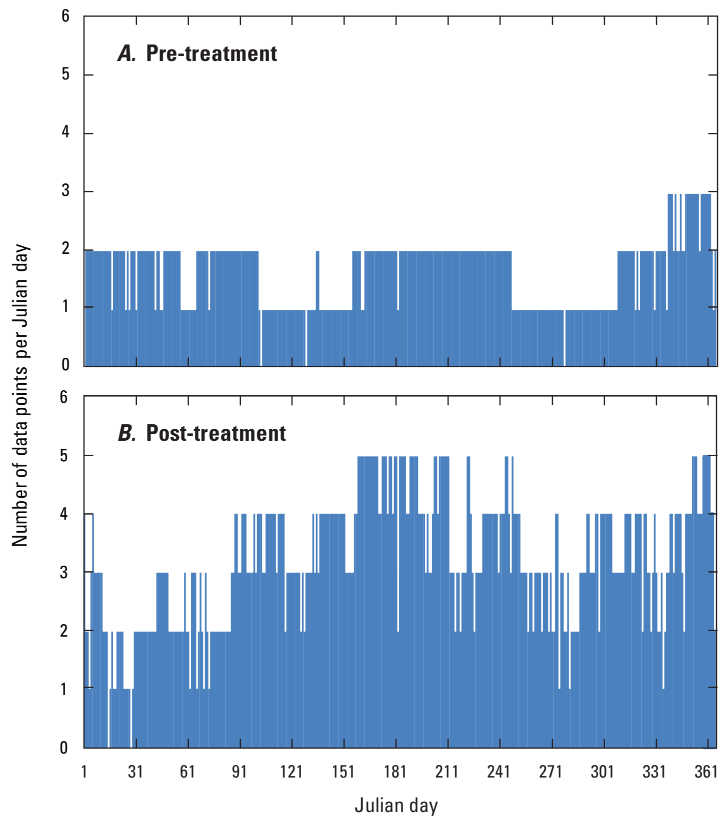


Figure 11. Number of available paired daily evapotranspiration data points by Julian day during A, pre- and B, post-treatment periods at the reference watershed evapotranspiration site RWS_{ET} (U.S. Geological Survey station 295104098285900 Honey Creek reference evapotranspiration near Spring Branch, Texas), and the treatment watershed evapotranspiration site TWS_{ET} (U.S. Geological Survey station 295102098283200 Honey Creek treatment evapotranspiration near Spring Branch, Tex.).

To evaluate if potential climate variability might be influencing the Wilcoxon-Mann-Whitney test results, the same analysis was performed on the daily ET rates at site RWS_{ET}. The null hypothesis is the daily ET rates at site RWS_{ET} during the pre-treatment time period is equal to or less than the daily ET rates during the post-treatment period. The test results indicate no rejection of the null hypothesis (one-sided, $p < 0.01$) (that is, there is not a statistical difference between pre- and post-treatment daily ET rates); however, the null hypothesis can be rejected at the 95-percent confidence level (one-sided, $p < 0.05$). These results indicate potential climate variability between the two time periods might have been a contributing factor to the observed changes in daily ET rates.

To reduce the potential effects of potential site bias or climate variability, the daily ET rates at site TWS_{ET} were subtracted from the daily ET rates at site RWS_{ET} (fig. 9C). A positive difference indicates the daily ET rates were greater at site RWS_{ET}, and a negative difference indicates the daily ET rates were greater at site TWS_{ET}. This assumes that the potential site biases were constant in time and that both sites were influenced similarly by potential climatic variability. The difference in daily ET rates were evaluated pre-treatment compared to post-treatment using a Wilcoxon-Mann-Whitney test. The null hypothesis was the difference in daily ET rates was the same or greater during pre-treatment period than during the post-treatment time period. The test results indicated rejection of the null hypothesis (one-sided, p less than 0.01); that is, the post-treatment differences in evapotranspiration rates were greater than those of the pre-treatment period.

Each of above analyses assumes that the daily ET rates composing the datasets are serially independent. The daily ET data do exhibit a degree of autocorrelation and, hence, the findings should be understood in that context. This is partly because the daily ET rates are determined from atmospheric and climatic conditions, which might exhibit autocorrelation (Wilks, 2006). To account for autocorrelation, a modified z-test statistic can be used to test if the mean differences in the daily ET rates between the two sites (RWS_{ET} - TWS_{ET}) during the pre- and post-treatment periods are the same (Wilks, 2006). The pre- and post-treatment datasets of difference in daily ET rates generally are symmetric and without extreme outliers. The null hypothesis was the mean difference in daily ET rates is the same or greater during pre-treatment time period than the post-treatment time period. The modified z-statistic indicates that the null hypothesis should be rejected (one-sided, $p < 0.01$). That is, the mean difference in daily ET rates between the two sites is greater during the post-treatment period than during the pre-treatment period.

Ashe juniper and grasses enter into a period of lower daily ET rates during the winter months. One objective was to examine differences in daily ET rates between the watersheds during the growing season when daily ET rates were high. Hence, a modified z-test statistic was performed focusing only on the mean differences in daily ET rates during the growing season (April 1 to October 1). To calculate the mean growing season difference in daily ET rates, all available daily differences in ET rates from April 1 to October 1 were used to determine the mean difference, which were approximately normally distributed as determined by a Kolmogorov-Smirov test (StatSoft Inc., 2011). The null hypothesis is the mean difference in daily ET rates during the growing season is the same or greater during the pre-treatment period compared to the mean difference in daily ET rates during the post-treatment time period. The modified z-statistic indicated that the null hypothesis should be rejected (one-sided, $p < 0.05$). That is, the mean differences in daily ET rates were higher during the post-treatment period than during the pre-treatment period.

The statistical tests described in this section indicated the mean difference in daily ET rates between the two sites (RWS_{ET} - TWS_{ET}) was greater during the post-treatment than during the pre-treatment period, even when accounting for possible site bias and climate variability. The above analyses tested population groups, where the daily ET differences were grouped into two populations: pre- and post-treatment. However, as with the RWS_{ET} and TWS_{ET} ET records, there was intra- and interannual variability in the record of difference in daily ET rates. One notable example period was during the growing season of 2006. The daily ET rates were greater at site RWS_{ET} than at site TWS_{ET} in mid-June, resulting in a positive difference (fig. 9*C*). A prolonged dry period began during the mid-growing season of 2006, and daily ET rates at sites RWS_{ET} and TWS_{ET} began to decrease. However, the daily ET rates at site TWS_{ET} were slightly higher than the daily ET rates at site RWS_{ET} during this dry period, likely because vegetation present at site TWS_{ET} (more grasses) were potentially better adapted to hot and dry environmental conditions than vegetation at site RWS_{ET} (more ashe juniper). This pattern of greater daily ET rates at site TWS_{ET} compared to site RWS_{ET} continued until appreciable rainfall (about 1 inch per week) occurred more frequently, beginning in late August. The RWS_{ET} and TWS_{ET} sites then had sufficient water to support plant growth, and the daily ET rates returned to seasonally normal levels–where the daily ET rates at site RWS_{ET} were again greater than at site TWS_{ET}. This dynamic response to rainfall might be related to the plant physiologies within each watershed—in particular whether the plants use a C4 or C3 photosynthesis pathway. Plants that employ a C4 photosynthesis pathway (for example, certain grasses) are better adapted to hot and arid climates, compared to plants that employ a C3 photosynthesis pathway (for example, ashe juniper) which are better adapted to comparatively cooler and moister climates. This is because during hot, arid conditions, C4 and C3 plants begin to close their stomata to reduce water loss, however, during partial stomatal closure, photosynthesis is reduced less in C4 plants compared to C3 plants (Weier and others, 1982).

In contrast to 2006, 2007 was a wet year; the annual rainfall was approximately 49 in. without any extended dry periods. The daily ET rates at site RWS_{ET} were consistently higher in 2007 than at site TWS_{ET}. Hence, when grouping multiple years of data together (for example, 6 years during the post-treatment), the average difference between pre- and post-treatment were statistically significant, but daily ET differences between sites RWS_{ET} and TWS_{ET} varied on a daily to annual time step.

Determination of Change in Water from Evapotranspiration

The effects of differences in daily ET rates between sites RWS_{ET} and TWS_{ET} on the hydrologic budget can be evaluated. A dimensionless ratio of the daily ET rates can be used to estimate the difference in the amount of water returned to the atmosphere from evapotranspiration between sites (hereinafter referred to as the change in water or ΔW), calculated as:

$$\Delta W = 1 - \left(\frac{\Sigma ET\ RWS_{ET}\, pre-treatment}{\Sigma ET\ TWS_{ET}\, pre-treatment}\right)\left(\frac{\Sigma ET\ TWS_{ET}\, post-treatment}{\Sigma ET\ RWS_{ET}\, post-treatment}\right) \quad (9)$$

where

$\Sigma ET\ RWS_{ET}$ *pre-treatment* is the sum of daily ET rates at site RWS_{ET} during the pre-treatment period;

$\Sigma ET\ TWS_{ET}$ *pre-treatment* is the sum of daily ET rates at site TWS_{ET} during the pre-treatment periods;

$\Sigma ET\ TWS_{ET}$ *post-treatment* is the sum of daily ET rates at site TWS_{ET} during the post-treatment periods; and

$\Sigma ET\ RWS_{ET}$ *post-treatment* is the sum of daily ET rates at site RWS_{ET} during the post-treatment periods.

By incorporating daily ET rates at both sites during pre- and post-treatment periods, the calculated change in water reduces the effects of potential site bias and climate variability. The calculated ΔW using all available paired daily ET data was 0.08, indicating an approximate 8 percent reduction in water as a result of evapotranspiration at site TWS_{ET} as compared to site RWS_{ET} during the post-treatment period is likely. This reduction is a percentage of the amount of water attributed to ET, not the annual rainfall. As discussed previously, the daily ET rates vary seasonally and interannually. Given the long study period, the 8 percent difference of change in water represents a general annual average (which includes dry, wet, and intermediate conditions) and can vary in magnitude from year to year depending on environmental conditions. To reduce potential influences of data gaps on the change in water calculation, the average annual ET rates (as determined by the Fourier transformation methodology) were substituted in equation 9. The resulting average ΔW is 0.09 or 9 percent. This indicates that the 8 percent difference, calculated using all the paired daily ET data, is likely not biased high as a result of data gaps. As a final check, the ΔW was calculated using the composite average annual ET rates (as determined by the average monthly composite methodology). This resulted in a ΔW of 0.09 or 9 percent. The results of the three methods of calculating the average change in water ratio are generally consistent with each other. Dugas and others (1998) performed a similar plot-scale study where ET was measured for 2 years before and 3 years after ashe juniper was removed from study watersheds. Using those results in equation 9, the ΔW was 5 percent for the 3-year post-treatment period. Much of the difference in ET was observed in the first 2 years with negligible difference in the third year. Dugas and other (1998) attributed the differences to changes and regrowth of vegetation.

As discussed previously, the fetch areas are largely representative of the study watersheds, but some amount of "contaminating areas" may be influencing the daily ET rates. To address potential "contamination areas", a paired set of equations can be solved iteratively to determine a theoretical daily ET rate, as compared with the measured daily ET rate. This assumes that the contaminating area for the reference watershed is from the treatment watershed and conversely the contaminating area for the treatment watershed is from the reference watershed (or area of similar vegetation cover).

$$RWS_{ETT} = \frac{100 * RWS_{ETM} - (100 - \%EQ_{RWS}) * TWS_{ETT}}{\%EQ_{RWS}}, \quad (10)$$

$$TWS_{ETT} = \frac{100 * TWS_{ETM} - (100 - \%EQ_{TWS}) * RWS_{ETT}}{\%EQ_{TWS}}, \quad (11)$$

where

RWS_{ETT} is the theoretical sum of daily ET rates at site RWS_{ET}, in millimeters;

RWS_{ETM} is the measured sum of daily ET rates at site RWS_{ET}, in millimeters;

TWS_{ETT} is the theoretical sum of daily ET rates at site TWS_{ET}, in millimeters;

TWS_{ETM} is the measured sum of daily ET rates at site TWS_{ET}, in millimeters;

$\%EQ_{RWS}$ is the percent equilibration at site RWS_{ET}, dimensionless; and

$\%EQ_{TWS}$ is the percent equilibration at site TWS_{ET}, dimensionless.

Equations 10 and 11 can be iterated for the pre- and post-treatment periods. Using the RWS_{ETT} and TWS_{ETT} values in equation 9, the ΔW increased from 0.08 to 0.10. Hence, the reported 8 percent change in water from evapotranspiration is considered to be a conservative value.

Potential Groundwater Recharge

Groundwater recharge was not directly measured, but was calculated as the amount of residual water from the measured components of hydrologic budget; that is, by algebraically rearranging equation 7 to solve for groundwater recharge:

$$GW = RF - SW - ET \quad (12)$$

where all terms are previously defined.

Groundwater-monitoring wells were not installed in the study area, and groundwater discharges were not measured as part of this study. Thus, it is not possible to determine where the residual water went. Wilcox and others (2006) indicate groundwater recharge might occur in shrub land areas with shallow soils overlying permeable karst geology, characteristic of conditions at the Honey Creek State Natural Area. Because measuring the residual water was beyond the scope of this study, the calculated groundwater recharge is hereinafter broadly categorized as potential groundwater recharge.

Hydrologic Budget Partitioning

Average annual rainfall, streamflow (expressed as average annual unit runoff), evapotranspiration, and potential groundwater-recharge can be incorporated into a single hydrologic budget (eq. 7) applied to each watershed before and after treatment to evaluate the effects of brush management. The average annual values during the pre-treatment period were calculated for the time period common to each of the data types (2002–04; evapotranspiration data were not collected during 2001). The average annual rainfall was calculated from the annual rainfall values for the pre- and post-treatment periods. Average annual unit runoff was calculated from the annual unit runoff values (summation of the daily unit runoff values for the respective year). The average annual evapotranspiration was calculated by the Fourier methodology. The average annual potential groundwater recharge was calculated using average annual rainfall, unit runoff, and evapotranspiration values in equation 12.

The effect of each component on the total water budget can be assessed as a percentage of the average annual rainfall (fig. 12, table 5). These percentages represent the average annual values for the study period for the respective hydrologic component. Caution is warranted when attempting to quantify these types of comparisons, as differences might be affected by potential site bias or climate variability. To be more representative of typical conditions observed in the study area, the extreme rainfall event in 2002 (22 in. of rainfal during a 2-week period) was removed from the rainfall record, unit runoff, and consequently, potential groundwater totals in the calculated percentages. This is because the rainfall amount during that event was more than three times greater than the next largest rainfall event over the 10-year study period and accounted for more than 40 percent of the annual rainfall for that year. The ET data during this period was not removed because identifying ET rates resulting from the storm event is not possible with this dataset, and average annual ET rates were calculated using the Fourier transformation methodology.

A comparison of the overall percent contributions during the study period indicate a change in percent contribution of water budget components (fig. 12). Specifically, during the pre-treatment period, the percent average annual unit runoff, evapotranspiration, and potential groundwater recharge (each expressed as a percentage of total rainfall) in the reference watershed were similar to those in the treatment watershed: unit runoff (5 percent compared to 5 percent), evapotranspiration (81 percent compared to 78 percent), and potential groundwater recharge (14 percent compared to 17 percent), respectively. During the post-treatment period, the percent average annual unit runoff in the reference watershed was similar to the percent average annual unit runoff in the treatment watershed, however, the difference in percentages of average annual evapotranspiration and potential groundwater recharge were more appreciable between the reference and treatment watersheds than during the pre-treatment period: unit runoff (2 percent compared to 2 percent), evapotranspiration (85 percent compared to 74 percent), and potential groundwater recharge (13 percent compared to 24 percent), respectively.

Water Quality

To evaluate effects of brush management on water quality, water samples were collected at the rainfall water-quality site (RQW) as well as at the three streamflow-gaging stations during selected storms. Water-quality samples were collected during both pre- and post-treatment periods. The samples were analyzed for water chemistry (appendix 4A-D) and suspended-sediment concentrations (appendix 5A-C). Samples were separated into pre- and post-treatment periods. Statistical comparisons between pre- and post-treatment or between sites is not possible because of the small sample size; comparisons presented here are based on visual inspection.

Physical Properties and Chemical Constituents

Specific conductance is a physical measurement of the amount of electrical current that water can transmit and is a direct reflection of the ionic strength, or total amount of dissolved solids in the water (Hem, 1992). Rain has a very low specific conductance (Herczeg and Edmunds, 2000). Surface water has higher specific conductance than rainfall because of chemical reactions with the land surface, soils, and streambed; groundwater typically has a higher specific conductance than surface water because of the dissolution of the rock matrix of the aquifer. The specific conductance measured in the laboratory for the surface-water samples (collected at sites 1C, 1T, 2T) ranged from 86 to 223 microsiemens per centimeter (µS/cm), with a median value of 129 (µS/cm). These specific-conductance values are less than the specific-conductance of 586 µS/cm measured in a groundwater sample collected from a nearby well in the Honey Creek State Natural Area, which was completed in the Edwards-Trinity aquifer (U.S. Geological Survey station 295013098285601) (U.S. Geological Survey, 2010). These results indicate the surface-water flows are consistent with a stormwater pulse compared to groundwater discharge.

Lower concentrations of major ions were generally measured in the rainfall samples than in surface-water samples collected from the streamflow-gaging stations during the same sampling event. This is the case of both pre- and post-treatment periods. Graphical comparison of major-ion

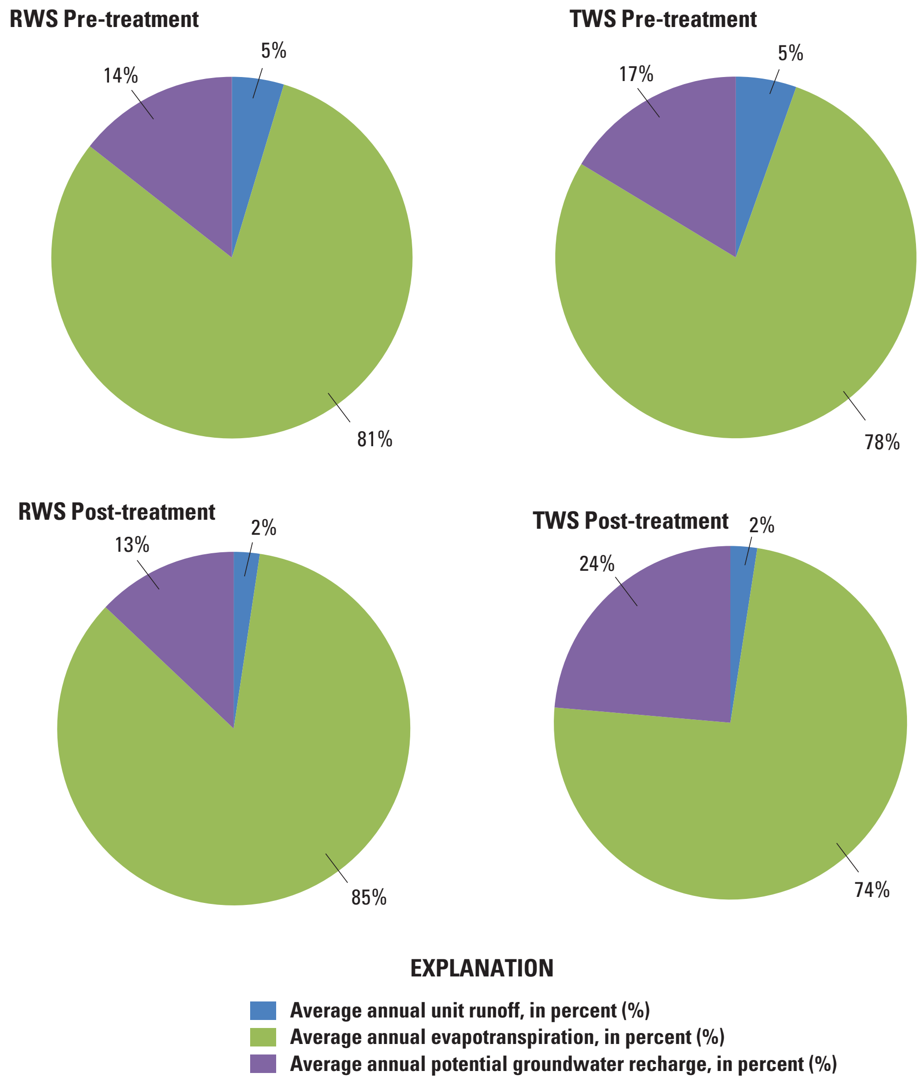


Figure 12. Percent contribution of the hydrologic components to the total water budget during pre- and post-treatment periods at the reference watershed (RWS), and the treatment watershed (TWS) (table 5).

constituents in filtered samples collected at sample sites did not exhibit notable changes between pre- and post-treatment periods (fig. 13).

Total nitrogen and the associated species of nitrate, nitrite, ammonia, and organic nitrogen in filtered samples collected from sample sites did not exhibit a notable change between pre- and post-treatment periods (fig. 13). The possible exception is organic nitrogen. From graphical comparisons, concentrations of organic nitrogen measured in samples collected from site 2T appear relatively similar compared to concentrations measured in samples collected from site 1C during pre-treatment periods. From graphical comparisons during the post-treatment period, concentrations of organic nitrogen measured in samples collected from site 2T appear somewhat higher compared to concentrations measured in samples collected from site 1C. Organic nitrogen

Table 5. Average annual rainfall, unit runoff, evapotranspiration, and potential groundwater recharge during pre- and post-treatment periods in the reference watershed (RWS) and treatment watersheds (TWS).

Watershed	Time period[1]	Average annual rainfall (inches/year)	Average annual unit runoff (inches/year)	Average annual evapotranspiration (inches/year)	Average annual potential groundwater recharge (inches/year)
RWS	Pre-treatment (2002–04)	35.09	1.63	28.39	5.07
TWS	Pre-treatment (2002–04)	35.09	1.90	27.28	5.91
RWS	Post-treatment (2005–10)	30.04	.71	25.39	3.94
TWS	Post-treatment (2005–10)	30.04	.73	22.21	7.09

[1]Time period during pre-treatment limited to time periods common to all data–evapotranspiration data were not available during 2001.

concentrations measured in samples collected from sites 1T and 2T were similar during the post-treatment period. The similarity between concentrations measured in samples collected from sites 1T and 2T during the post-treatment period indicate that land use upstream from site 1T might be affecting water quality at both sites. However, these graphical comparisons are based on all available data points, some of which were censored values. Orthophosphate and phosphorus in filtered samples, and organic carbon in unfiltered samples collected from sample sites do not exhibit a notable change between pre- and post-treatment periods.

Stable isotopes from rainfall (RQW samples) and surface-water samples (1C, 1T, 2T samples) ranged from -61.24 to -6.6 per mil for hydrogen (δD), and -9.25 to -1.84 per mil for oxygen ($\delta^{18}O$). Comparing these values with the global meteoric water line (Craig, 1961), the sample data indicate that the surface-water flow is meteoric in origin, corroborating the finding based on the specific conductance data that the surface-water flows are consistent with a stormwater-runoff pulse.

Suspended Sediment

Suspended-sediment samples were collected from the reference and treatment watersheds at sites 1C and 2T, respectively. Multiple samples were collected during selected storm events on the rising and falling limbs of the hydrograph. Samples collected from both watersheds during the same storm events were used in the following comparisons so that antecedent conditions would be consistent in the comparison (for example, similar dry and wet periods prior to the sampled storm event). The suspended-sediment loads are primarily composed of fine-sized particles. Typically, more than 90 percent of suspended sediment has grain size of less than 0.0625 mm, representative of silts and clays. Suspended-sediment loads (in units of tons per day) were calculated as (Porterfield, 1972):

$$L = 0.0027 * CQ \tag{13}$$

where

- 0.0027 is a unit-conversion factor;
- L is the suspended-sediment load, in units of tons per day;
- C is the suspended-sediment concentration, in units of milligrams per liter; and
- Q is the streamflow, in units of cubic feet per second.

The suspended-sediment-load data were examined for a relation to streamflow. Both suspended-sediment load and streamflow data were non-normally distributed, and consequently, a natural logarithm transformation, ln(x) was applied to both datasets. Samples where the streamflow was less than 0.1 ft^3/s were not included in the following analyses because at such low flows, pooling behind the weir may have potentially influenced the sample composition. All values are reported in appendix 5A-C for completeness. Only samples collected from the same storm events were included in the following analyses.

The pre-treatment suspended-sediment load (dependent variable) exhibited a log-linear relation with streamflow (independent variable) at both watersheds, with R^2 values of 0.82 and 0.92 calculated at sites 1C and 2T, respectively, and $p < 0.01$ for both regression equations (fig. 14*A*). The residuals are approximately uniformly distributed, indicating a natural logarithmic transformation is appropriate for these data. Using a one-sided test (Zar, 1984), the regression lines are not statistically different from each other at the 95-percent confidence level, indicating suspended-sediment load to streamflow relation in both watersheds are similar during the pre-treatment period.

The post-treatment suspended-sediment loads at sites 1C and 2T also exhibited a statistically significant log-linear relation to streamflow (fig. 14*B*). The R^2 was 0.74 and 0.86 for sites 1C and 2T, respectively, and $p < 0.01$ for both regressions. The residuals are approximately uniformly distributed,

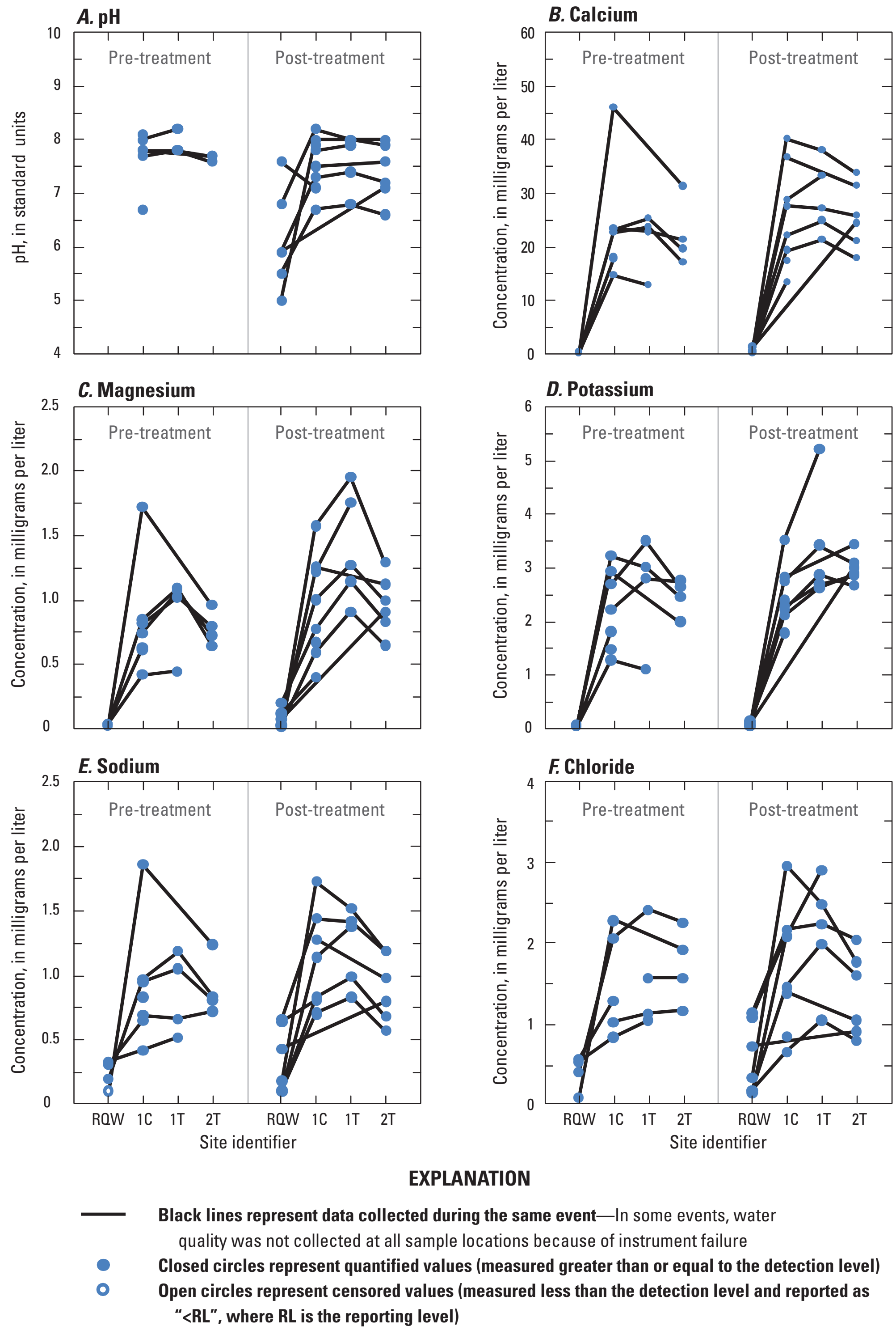
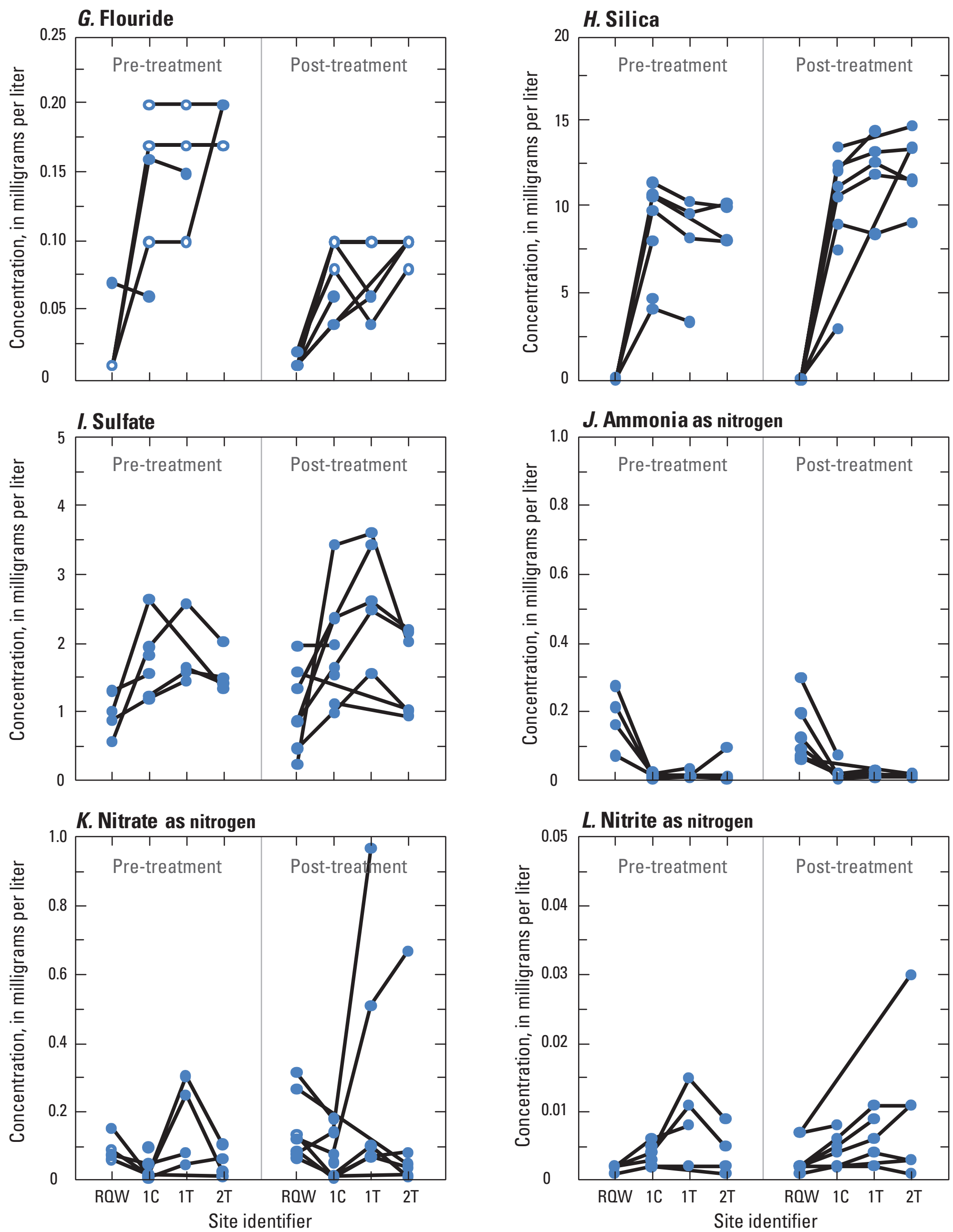


Figure 13. Water-quality data for selected constituents during pre- and post-treatment periods from the rainfall water-quality site RQW (U.S. Geological Survey station 295108098283201 Honey Creek rainfall water-quality near Spring Branch, Texas), reference watershed streamflow-gaging station site 1C (U.S. Geological Survey station 08167347 Unnamed tributary of Honey Creek site 1C near Spring Branch, Tex.), upstream treatment watershed streamflow-gaging station site 1T (U.S. Geological Survey station 08167350 Unnamed tributary of Honey Creek site 1T near Spring Branch, Tex.), and downstream treatment watershed streamflow-gaging station site 2T (U.S. Geological Survey station 08167353 Unnamed tributary of Honey Creek site 2T near Spring Branch, Tex.). *A*, pH. *B*, Calcium. *C*, Magnesium. *D*, Potassium. *E*, Sodium. *F*, Chloride. *G*, Fluoride. *H*, Silica. *I*, Sulfate. *J*, Ammonia as nitrogen. *K*, Nitrate as nitrogen. *L*, Nitrite as nitrogen. *M*, Organic nitrogen. *N*, Orthophosphate as phosphorus. *O*, Phosphorus. *P*, Total nitrogen. *Q*, Organic carbon.



EXPLANATION

Black lines represent data collected during the same event—In some events, water quality was not collected at all sample locations because of instrument failure

Closed circles represent quantified values (measured greater than or equal to the detection level)

Open circles represent censored values (measured less than the detection level and reported as "<RL", where RL is the reporting level)

Figure 13. Water-quality data for selected constituents during pre- and post-treatment periods from the rainfall water-quality site RQW (U.S. Geological Survey station 295108098283201 Honey Creek rainfall water-quality near Spring Branch, Texas), reference watershed streamflow-gaging station site 1C (U.S. Geological Survey station 08167347 Unnamed tributary of Honey Creek site 1C near Spring Branch, Tex.), upstream treatment watershed streamflow-gaging station site 1T (U.S. Geological Survey station 08167350 Unnamed tributary of Honey Creek site 1T near Spring Branch, Tex.), and downstream treatment watershed streamflow-gaging station site 2T (U.S. Geological Survey station 08167353 Unnamed tributary of Honey Creek site 2T near Spring Branch, Tex.). *A*, pH. *B*, Calcium. *C*, Magnesium. *D*, Potassium. *E*, Sodium. *F*, Chloride. *G*, Fluoride. *H*, Silica. *I*, Sulfate. *J*, Ammonia as nitrogen. *K*, Nitrate as nitrogen. *L*, Nitrite as nitrogen. *M*, Organic nitrogen. *N*, Orthophosphate as phosphorus. *O*, Phosphorus. *P*, Total nitrogen. *Q*, Organic carbon. —Continued

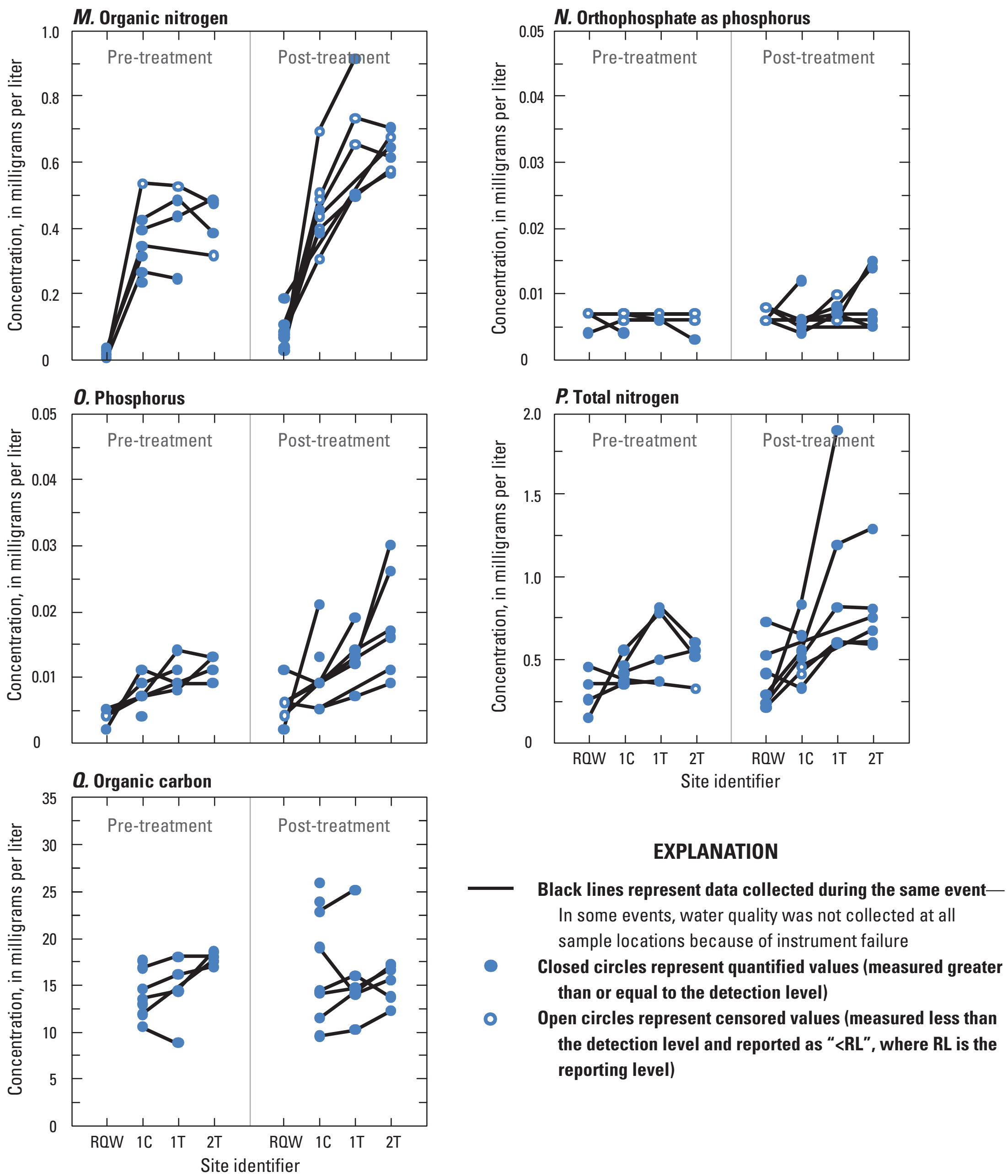


Figure 13. Water-quality data for selected constituents during pre- and post-treatment periods from the rainfall water-quality site RQW (U.S. Geological Survey station 295108098283201 Honey Creek rainfall water-quality near Spring Branch, Texas), reference watershed streamflow-gaging station site 1C (U.S. Geological Survey station 08167347 Unnamed tributary of Honey Creek site 1C near Spring Branch, Tex.), upstream treatment watershed streamflow-gaging station site 1T (U.S. Geological Survey station 08167350 Unnamed tributary of Honey Creek site 1T near Spring Branch, Tex.), and downstream treatment watershed streamflow-gaging station site 2T (U.S. Geological Survey station 08167353 Unnamed tributary of Honey Creek site 2T near Spring Branch, Tex.) *A*, pH. *B*, Calcium. *C*, Magnesium. *D*, Potassium. *E*, Sodium. *F*, Chloride. *G*, Fluoride. *H*, Silica. *I*, Sulfate. *J*, Ammonia as nitrogen. *K*, Nitrate as nitrogen. *L*, Nitrite as nitrogen. *M*, Organic nitrogen. *N*, Orthophosphate as phosphorus. *O*, Phosphorus. *P*, Total nitrogen. *Q*, Organic carbon. —Continued

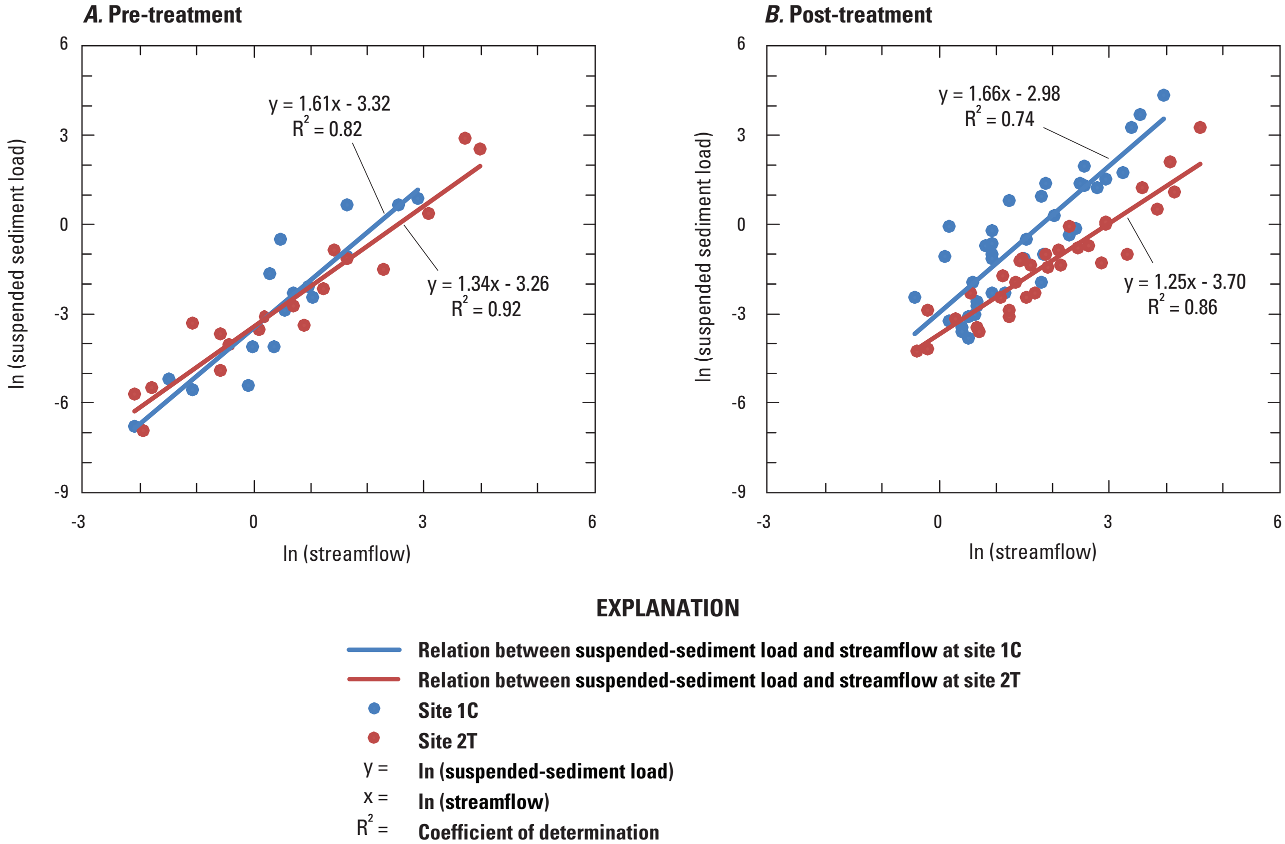


Figure 14. Suspended-sediment loads compared to streamflow during *A*, pre- and *B*, post-treatment periods at the reference watershed streamflow-gaging station site 1C (U.S. Geological Survey station 08167347 Unnamed tributary of Honey Creek site 1C near Spring Branch, Texas) and the treatment watershed streamflow-gaging station site 2T (U.S. Geological Survey station 08167353 Unnamed tributary of Honey Creek site 2T near Spring Branch, Tex.). Both datasets have been transformed through natural logarithm (ln), and the axes are ln scale.

indicating a natural logarithmic transformation is appropriate for these data. Using a one-sided test (Zar, 1984), the two linear regression lines were statistically different at the 95-percent confidence level. The suspended-sediment load to streamflow relations indicate that for the same streamflow, the suspended-sediment loads calculated from site 2T were generally less than suspended-sediment loads calculated from site 1C during the post-treatment period. This reduction in loads may be a result of the grasses acting as an obstruction to overland flow, causing overland flow to move in a slower, more tortuous path, thereby resulting in deposition of some of the suspended sediment before the overland flow reaches the stream channel (Thurow and others, 1986).

Changes in the auto-sampler program specifying when to collect samples (based on stage), as well as the size of storm events occurring during the sampling efforts, resulted in collecting samples during different streamflow conditions for the study period. The streamflow at site 1C during the pre-treatment sampling period, when stormflow samples were collected (for streamflows greater than 0.1 ft^3/s) and suspended-sediment load was calculated, ranged from 0.12 to 51 ft^3/s, and ranged from 0.66 to 52 ft^3/s during the post-treatment period. Compared to the streamflow at site 1C, the streamflow during sample collection events at site 2T was similar, ranging from 0.12 to 55 ft^3/s during the pre-treatment period and 0.7 to 101 ft^3/s during the post-treatment period. However, there are more datapoints associated with smaller streamflows in the pre-treatment period than the post-treatment period (fig. 14). To assess if differences in sample populations unduly influenced the results, the data were culled to more uniformly represent similar streamflows between the datasets. Specifically, samples that were collected when the streamflow was less than 0.66 ft^3/s or greater than 55 ft^3/s were removed from the datasets, and the statistical analyses were rerun. The results confirmed the original findings that the regression lines for sites 1C and 2T were not statistically different from each other during the pre-treatment period, but were different in the post-treatment period (one-sided tests, $p < 0.05$). Suspended-sediment samples also were collected in the upper part of the treatment watershed at site 1T. However, because of sample size limitations at site 1T during the pre-treatment period, similar statistical analyses were not possible.

Summary

The U.S. Geological Survey (USGS), in cooperation with the U.S. Department of Agriculture Natural Resources Conservation Service, the Edwards Region Grazing Lands Conservation Initiative, the Texas State Soil and Water Conservation Board, the San Antonio River Authority, the Edwards Aquifer Authority, Texas Parks and Wildlife, the Guadalupe Blanco River Authority, and the San Antonio Water System, evaluated the hydrologic effects of ashe juniper (*Juniperus ashei*) removal as a brush management conservation practice in and adjacent to the Honey Creek State Natural Area in Comal County, Tex.

Woody vegetation, including ashe juniper, have encroached historically oak grassland savannah areas across much of the Edwards aquifer catchment and outcrop area. This is generally attributed to overgrazing and fire suppression. By removing the ashe juniper and allowing native grasses to reestablish in the area as a brush management conservation practice (referred to as brush management), the hydrology in the watershed might change. This idea generally is based on a simplified mass balance approach of the hydrologic cycle. In this simplified approach, where rainfall accounts for the water coming into the system, rainfall is distributed to surface-water runoff (streamflow), evapotranspiration (combination of evaporation and transpiration), or groundwater recharge (subsurface flow that flows into the groundwater table or contributes to spring discharge downstream from the study area). If the rainfall remains constant, but the evapotranspiration rates decrease because of a change in vegetation cover, then the surface-water or groundwater components of the hydrologic budget will change. Because the streams in the study area are ephemeral and only flow during periods of stormwater runoff, base flow is considered negligible. If the rainfall remains constant, but the evapotranspiration rates change because of a change in vegetation cover, then the surface-water or groundwater components of the hydrologic budget will change.

After hydrologic data were collected in adjacent watersheds for 3 years, brush management occurred on the treatment watershed while the reference watershed was left in its original condition. Hydrologic data were collected for another 6 years. Hydrologic data include rainfall, streamflow, evapotranspiration, and water quality. Groundwater recharge was not directly measured but potential groundwater recharge was calculated as the residual of a simplified mass balance approach of the hydrologic budget. The resulting hydrologic datasets were examined for differences between the watersheds and between pre- and post-treatment periods to assess the effects of brush management. Statistical comparisons of the relation of streamflow and rainfall (expressed as event unit runoff to event rainfall relation) did not change between the watersheds during pre- and post-treatment periods.

Evapotranspiration was measured at the reference watershed site RWS_{ET} (U.S. Geological Survey station 295104098285900 Honey Creek reference evapotranspiration near Spring Branch, Tex.) and in the treatment watershed site TWS_{ET} (U.S. Geological Survey station 295102098283200 Honey Creek treatment evapotranspiration near Spring Branch, Tex.) using the Bowen ratio methodology. The daily ET rates at sites RWS_{ET} and TWS_{ET} exhibited a seasonal cycle during the pre- and post-treatment periods, with intra- and interannual variability. Daily ET rates generally were lower in the winter months, began increasing around April, coinciding with the beginning of the growing season, and reached a maximum rate around July. The daily ET rates decreased around October, coinciding with the end of the growing season. An average seasonal cycle of ET was developed using a first order Fourier transformation function that was fit to the data.

The effects of differences in daily ET rates between sites RWS_{ET} and TWS_{ET} on the hydrologic budget were evaluated. Statistical analyses indicate the mean difference in daily ET rates between the two sites (RWS_{ET} - TWS_{ET}) is greater during the post-treatment period than during the pre-treatment period. A dimensionless ratio of the daily ET rates at sites RWS_{ET} and TWS_{ET} can be used to estimate the difference in the amount of water returned to the atmosphere from evapotranspiration between sites (ΔW). Using all available paired daily ET data between sites RWS_{ET} and TWS_{ET}, the ΔW was calculated as 0.08, indicating an approximate 8 percent reduction in water as a result of evapotranspiration at site TWS_{ET} is likely. This reduction is a percentage of the amount of water attributed to evapotranspiration, not the annual rainfall.

Average annual rainfall, streamflow, evapotranspiration, and potential groundwater-recharge conditions were incorporated into a single hydrologic budget applied to each watershed before and after treatment to evaluate the effects of brush management. The influence of each component of the hydrologic budget can be assessed as a percentage of the total rainfall amount. A comparison of the overall percent contributions during the study period indicate a change in percent contribution of water budget components. Specifically, during the pre-treatment period, the percent average annual unit runoff, evapotranspiration, and potential groundwater recharge (expressed as a percentage of total rainfall) in the reference watershed were similar to those in the treatment watershed. During the post-treatment period, the percent average annual unit runoff in the reference watershed was similar to the percent average annual unit runoff in the treatment watershed, however, the difference in percentages of average annual evapotranspiration and potential groundwater recharge were more appreciable between the reference and treatment watersheds than during the pre-treatment period.

Using graphical comparisons, no notable differences in major ion or nutrient concentrations were found between samples collected at the reference watershed (site 1C) and treatments watershed (site 2C) during pre- and post-treatment periods. Suspended-sediment loads were calculated from samples collected at sites 1C and 2T. The relation between suspended-sediment loads and streamflow calculated from samples collected from sites 1C and 2T did not exhibit a statistically significant difference during the pre-treatment period,

whereas during the post-treatment period, relation between suspended-sediment loads and streamflow did exhibit a statistically significant difference. The suspended-sediment load to streamflow relations indicate that for the same streamflow, the suspended-sediment loads calculated from site 2T were generally less than suspended-sediment loads calculated from site 1C during the post-treatment period.

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