Instream Flow Requirements for Maintenance of Wildlife Habitat and Riparian Vegetation:

Cherry Creek, Tonto National Forest, Arizona

Submitted to Arizona Division of Water Resources, June, 2010
by

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Table of Contents

Introduction

Cherry Creek is a free-flowing, perennial tributary of the Salt River in central Arizona. Cherry Creek originates in spring fed tributaries on the Mogollon Rim at an elevation of 2,360 m above mean sea level (amsl) and flows 95 km to its confluence with the Salt River at 754 m. Cherry Creek and its tributaries flow through confined canyons along much of their length and through wide alluvial valley bottoms as the main stem nears its confluence with the Salt River. The catchment area of Cherry Creek is 721 km^2, most of which is managed by the Tonto National Forest. In 1999, the Tonto National Forest applied for an instream flow water right of 8,837 acre feet for the lower reaches of Cherry Creek (Appendix A). The objective of the current study is to determine streamflow characteristics necessary to maintain wildlife habitat along an alluvial reach downstream from US Geological Survey (USGS) streamflow gage 09497980. In support of our objective, we initiated a study to determine the relationships between wildlife habitat, riparian vegetation and streamflow in Cherry Creek. We accomplished this by measuring the principle landscape feature that supports terrestrial-avian wildlife habitat along Cherry Creek – riparian vegetation. We investigated the physical linkages between surface water (streamflow), the alluvial groundwater supporting riparian vegetation/wildlife habitat, and developed relationships between streamflow and groundwater elevation over a range of flow conditions. We used these data to develop a predictive model of riparian forest cover as it relates to wildlife habitat.

Wildlife Habitat Characterization

It is well-accepted that structurally and compositionally diverse habitats provide more and more varied resources for a larger number of wildlife species than do more compositionally and structurally homogeneous habitats (MacArthur and MacArthur 1961, MacArthrur 1964, Roth 1976, Powell and Steidl 2000, 2002, McElhinny *et al.* 2005, Kissling *et al.* 2008). We gathered data to characterize the range of riparian wildlife habitats that exist along intermittent and perennial segments of Cherry Creek, including a flow-diverted reach. We compared quantitative characteristics of wildlife habitat (in the form of structural complexity, species composition, and productivity) along intermittent, perennial, and flow-diverted reaches. By comparing habitat characteristics from different reaches, we can describe habitat conditions impacted by varying river flows and thus relate reductions in flow to responses of wildlife as expressed through changes in habitat.

The following report is organized into three sections. In the first section, we present information describing the wildlife habitat and its relationship to riparian vegetation composition and structure. Next, we compare and contrast wildlife habitat diversity and river discharge along a reach with intermittent and perennial flows. In the final section, we integrate information on riparian vegetation composition and fitness and groundwater-surface water interactions. We conclude by describing the drivers of habitat diversity, and describe implications of variations in river flow to wildlife habitat along Cherry Creek.

Methods

Study Design

The main study segment of Cherry Creek is located at 516753E, 3729660N (UTM coordinates) in central Arizona (Figures 1 and 2). This segment has a free-flowing reach with riparian and xeroriparian areas (drier riparian sites further from the channel *sensu* Stromberg *et al*. 2008), as well as a flow-diverted reach. We examined the structural and compositional characteristics of these habitats and differences in flow between reaches.

Because wildlife habitat includes the physical and biological components of a landscape, we examined physical conditions (e.g., hydrology, soil texture, and geomorphology) as well as riparian vegetation structure and composition. We measured vegetation in 12 m diameter (113 m^2) circular plots oriented along transects established perpendicular to the stream. Transects were systematically placed at 50 m intervals along the length of the stream. Along each transect, streamside plots were placed at a distance of 6 m from the active channel on either side of the stream. Subsequent plots were placed at 20 m intervals along each transect for a total of three 12 m diameter plots on either side of the stream (Figure 2). The reach had 14 transects along the undiverted reach, with four to six plots per transect (14 transects, n = 78 plots).

Four km upstream from the main study reach is a wide alluvial valley with a section of no or intermittent flow which then narrows and becomes more confined by bedrock in a downstream direction (517795E, 3733110N; Figures 1 and 2). This change in valley dimensions results in an intermittent upper segment along the wider valley reach (groundwater depth is below the channel bed along most of the reach) and a perennial lower segment (shallower water table and surface flow) along the narrower valley reach. Perennial flow occurs along the widest alluvial reaches only in areas where the thalweg is deepened (i.e., scour holes).

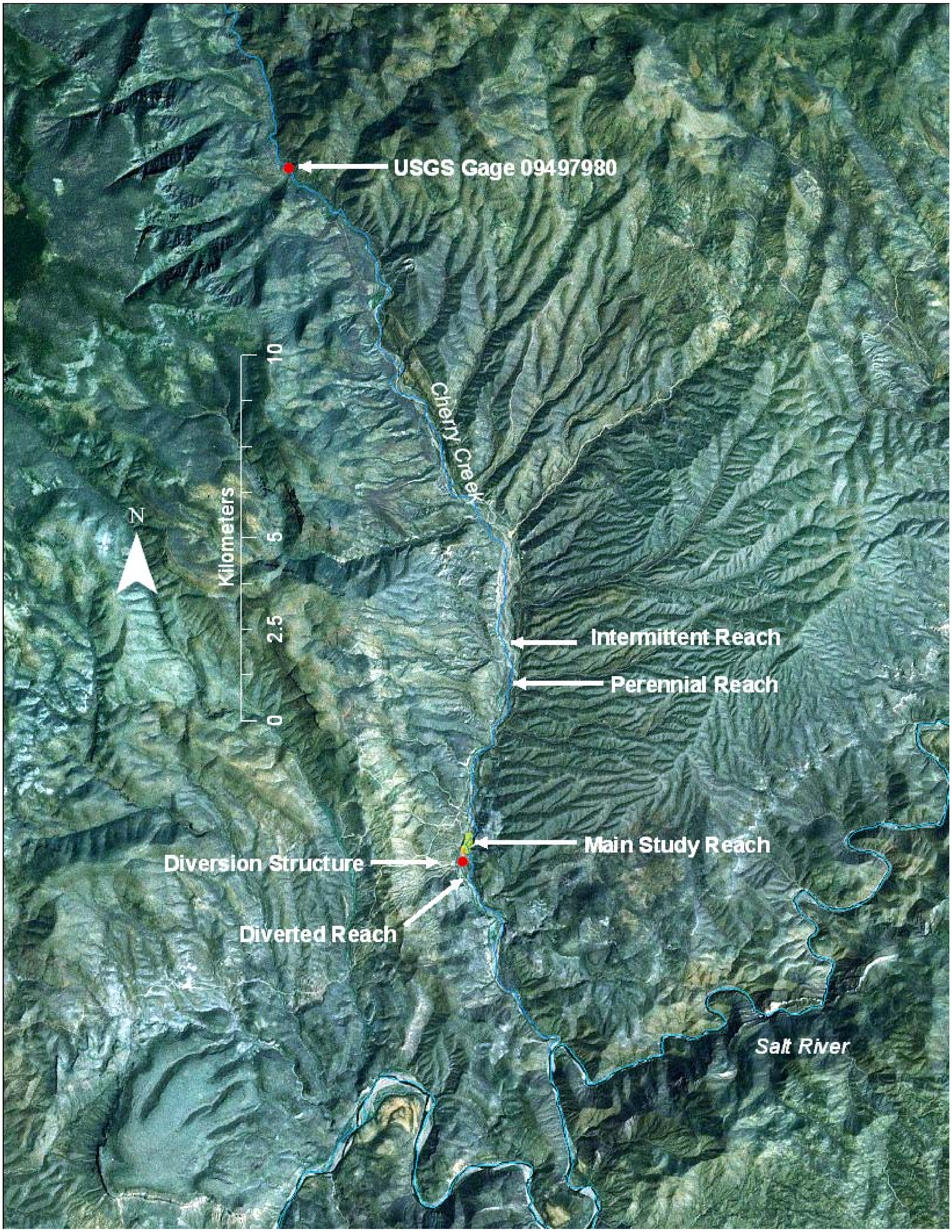

Figure 1. Cherry Creek showing the location of the USGS streamflow gage, the intermittent-perennial study reach, the main study reach, and the diverted reach.

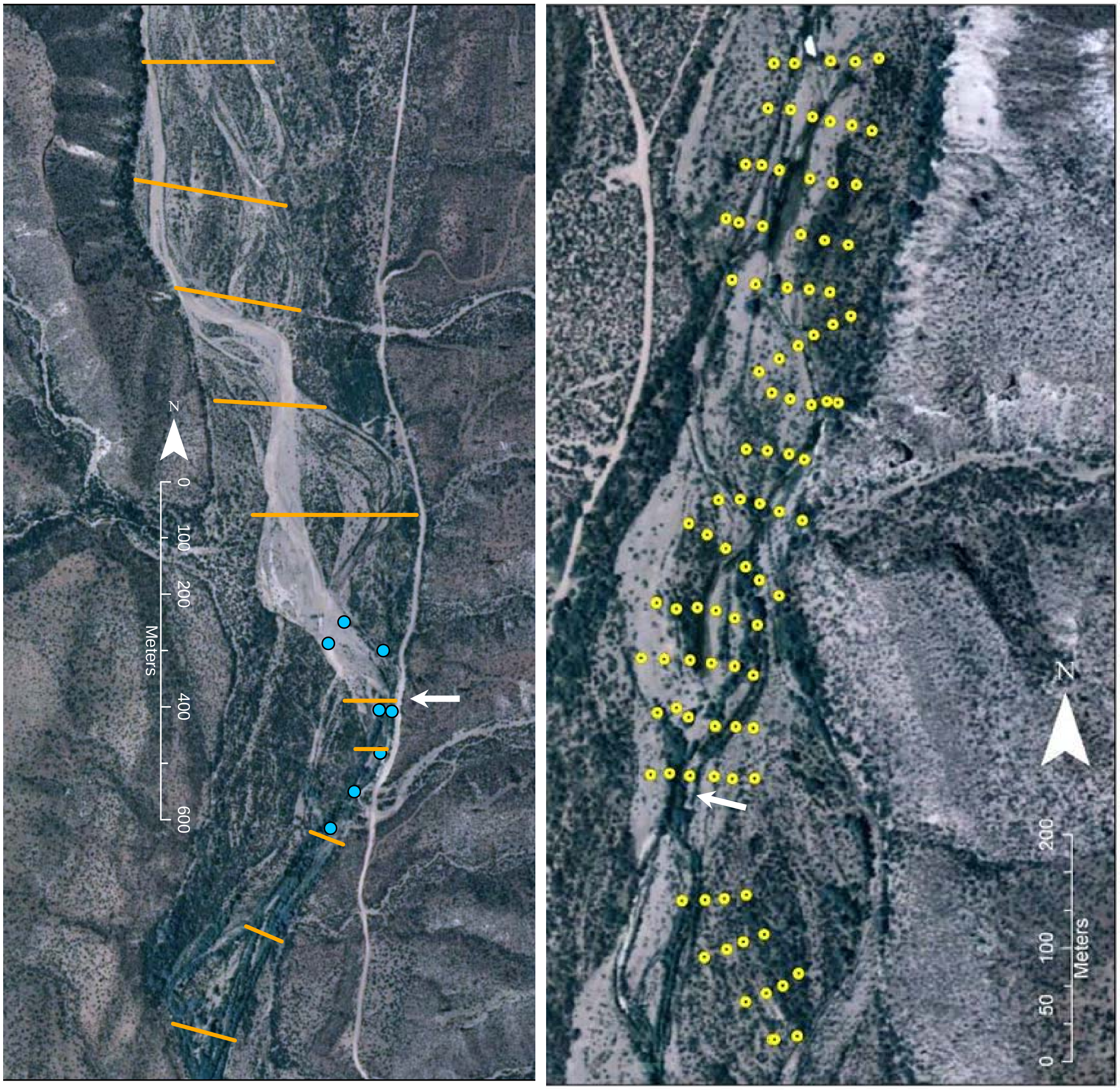

Figure 2. Study sites. Left panel. Upper study segment with intermittent (upstream) and perennial (downstream) reaches; transition from intermittent to perennial flow is indicated by white arrow. Vegetation sampling transects (orange lines) and perennial pools and flowing water (blue symbols) are shown. Right panel. Main study reach (upstream) and diverted reach (downstream from white arrow). Habitat sampling sites are indicated by yellow symbols. Streamflow direction is south (top to bottom) in both frames.

Wildlife Habitat Characterization, Vegetation Sampling, and River Flow Measurements

Structural complexity of wildlife habitat is a measure of the multiple structural attributes (e.g., plant size variation, cover, and vertical distribution of foliage) and the relative abundance of each of these attributes (McElhinny *et al*. 2005). At each circular plot, we measured characteristics of avian, small mammal, and reptile habitat. The characteristics we measured represent wildlife requirements in terms of breeding, cover, and foraging resources. For example, vertical substrate provides a wider variety of habitat niches, increasing potential bird species diversity (MacArthur and MacArthur 1961). Similarly, reptiles respond to structural changes in habitat (Pianka 1967) and mammals forage amongst diverse vegetation cover types (Oakley *et al*. 1985).

Variables measured to quantify avian habitat include: basal area and size of woody species for cavity-building species. We measured foliage height diversity and percent vegetation cover using the pole method (Carothers 1974) at five points within each circular plot. Measurements were taken from the center point of each plot and at 4 additional points, sampled 3 meters away from the center point in each of the cardinal directions. Foliage height diversity was estimated by tallying the number of times living vegetation intersects a 10 cm disc surrounding the pole at each height class (e.g., 0-0.013 m, 0.013-0.6 m, 0.6-1.5 m, 1.5-3.0 m, 3.0-4.5 m, 4.5-6.0 m, 6.0-9.0 m, and > 9.0 m; Figure 3). Diversity was calculated with the Shannon diversity index (H). Percent vegetation cover was determined by estimating cover in each of three height classes (0-1.5, 1.5-4.0, and >4 m), using modified Braun-Blanquet cover classes (> 1, 1-5, 6-15, 26-50, 51-75, 76-100%; Braun-Blanquet 1965). The height classes correspond to the distinct vegetation community types found at the site (i.e., herbs and forbs, shrubs, and trees). Total woody cover was determined by the percent canopy cover of all woody vascular plant species (83 species of shrubs and trees in total) 1 m above ground, as well canopy cover for all woody species. Stem diameter of every stem present was measured using calipers for several woody species of particular interest: Arizona sycamore (*Platanus wrightii*), Arizona alder (*Alnus oblogifolia*), Frémont cottonwood (*Populus frémontii*), Goodding willow (*Salix gooddingii*), saltcedar (*Tamarix ramosissima*), sandbar willow (*S. exigua*), velvet ash (*Fraxinus velutina*), and Arizona walnut (*Juglans major*).

Figure 3. Measuring foliage height diversity and percent vegetation cover using the pole method.

The variables measured to quantify compositional and structural complexity of habitat for small mammals and reptiles included: woody structure and diversity from 0 to 1.5 m above the ground, seedlings (food source), diameter of trees (to represent log and woody debris recruitment), amount of sandy substrate (for lizard locomotion), and total canopy cover to represent areas of cover and areas for reptiles to thermoregulate.

To relate habitat structure and diversity to the plant communities found along Cherry Creek, we measured plant species abundance (percent cover and basal area), tree size and age, and association with depth to groundwater. Stem size and stem size diversity was determined by calculating the geometric mean and geometric standard deviation of selected woody species occurring within each circular plot. Geometric mean diameter corresponds to basal area, stand volume, and other ecologically important stand characteristics, so is preferred to arithmetic mean for characterizing size-classes of woody vegetation (Curtis and Marshall 2000).

To determine the age of woody species, we removed stem cores or harvested tree slabs for range of sizes of each species of interest from the entire study reach. These samples were aged by counting ring annuli and incremental growth was measured using a linear encoder and stereo microscope. Analyses were conducted at the USGS dendrochronology laboratory in Fort Collins, CO. Methods for preparation and analysis followed Cook and Kairiukstis (1990). Individual cores and slabs were progressively sanded with an orbital sander (finest sand paper ANSI 440-grit). Ring width and year were digitized on a Velmex TA Unislide measuring system with an ACU-Rite linear encoder and QC1100 digital readout device (Velmex, Inc., Bloomfield, New York). The ring-reading software used was Measure J2X, Version 3.1 (Project J2X, Voortech Consulting, Holderness, NH).

Size- and age-class distributions were developed for each species of interest. Age distributions and establishment dates were calculated for each species and compared to long-term streamflow records. Annual incremental growth was also compared to a range of flow-related variables to determine the relationships between growth and streamflow attributes. These variables included average annual monthly flow for each year of growth (e.g., average of each daily average flow measured in the month of February calculated for each year).

Soil particle size composition of each vegetation stand was determined by estimating the percent cover of the surface of Phi diameter classes using the Wentworth scale (Appendix B). Average particle size was calculated using a weighted average by the percent cover of each Phi class in the plot.

Along the intermittent-perennial segment we sampled riparian vegetation along systematically spaced transects oriented perpendicular to valley walls. A transect was placed at the midpoint of the segment of interest (where stream transitions from intermittent to perennial) and transects were placed at 200 m intervals upstream and downstream from this midpoint (Figure 2). A total of ten transects were established, 5 upstream, 4 downstream and one at the midpoint. Vegetation was sampled along each transect. Samplers walked each transect until vegetation was encountered, then located plots to represent stands. Vegetation was sampled in nested plots (1 x 2 m for herbaceous vegetation, 2 x 5 m for trees and shrubs) in each of the vegetation patches. All vascular plant species were identified and percent cover estimated using the modified Braun-Blanquet cover-class scale. Within each 1 x 2 m plot, cover of each vascular species and ground cover feature (e.g., water, bare ground, litter, large wood pieces) was recorded.

We systematically surveyed the water surface elevation of pools and stream surface along the intermittent to perennial segment. Along the perennially flowing reach, discharge was measured at three different sites to calculate the rate of gain in discharge as a function of downstream distance.

Hydrologic Conditions

To determine the relationships between surface and groundwater, a grid of 8 groundwater wells was established along the main study reach. Four wells were installed along two transects spanning the width of the riparian zone and separated by 100 m along the length of the valley. One of these wells was placed immediately adjacent to the stream to serve as a staff gage to measure water surface level of the stream. All wells and staff gages were instrumented with Hobo U20 or Solinist Levelogger Gold (model 3001) water level loggers and depth to water (±0.5 cm) and temperature ($\pm 0.1^{o}C$) measured at 15 minute intervals (Onset Computer Corporation, Pocasset, MA., USA and Solinist Canada Ltd., Georgetown, ON). One well was instrumented with a pressure transducer to measure atmospheric pressure for corrections in groundwater measurements for variations in water temperature and atmospheric pressure. Seven additional wells were installed in September, 2009 to extend existing well transects and to add an additional transect. In addition, physical measurements of groundwater depth were taken each time the data loggers were downloaded. Streamflow was measured over a range of flows (0.057 to 1.42 cms (2 to 50 cfs)) when the study site was visited between July, 2008 through October, 2009.

Rating curves were constructed using stage measured by a pressure transducer at the staff gage and discharge was measured in the field and used to relate discharge to surface stage and groundwater levels. Separate rating curves were constructed for the time period before and after a flood which occurred in December, 2008. The streamflow record from Cherry Creek near Globe (USGS streamflow gage 09497980) was obtained and exceedance probabilities calculated and plotted for the more than 16,000 daily average flows measured at the gage during the period between May, 1965 and October, 2009. This exceedance probability curve provides the percentage of the time that each daily flow value is equaled or exceeded. Flood frequency/recurrence interval of flooding was calculated from a time series of instantaneous peak flow for the period 1965 to 2008 following USGS guidelines (USGS 1988).

Each vegetation plot and groundwater well was surveyed by a USGS hydraulic modeling team using Trimble survey grade global positioning system (GPS; Waddle and Bovee 2009). The survey equipment consisted of a Trimble® 5800 and R8 receivers using real-time kinematic positioning (RTK) and multipath reduction (Trimble Navigation, Ltd., http://www.trimble.com). Such survey-grade systems use carrier phase processing that enables centimeter accuracy. We used a Leica TC800 total station to survey areas where the GPS equipment could not be used. All data were recorded in Universal Transverse Mercator (SI) coordinates, zone 12 N, using the WGS84 horizontal datum, and the NAVD88 vertical datum.

To estimate depth to water table in the vegetation sampling plots, we used a spline fit to interpolate a groundwater surface (0.3 m grid size) within our groundwater well grid. We used measurements from a period of stable, low flow (October 5, 2009 which was 0.127 cms (4.5 cfs)) to avoid confounding effects of groundwater recharge or discharge related to floods or pulses in streamflow. Surveyed elevations of each vegetation plot enabled us to calculate the depth to groundwater in the vegetation plots at this stable, low flow.

Age and annual growth rates (corrected for tree age) were related to hydrologic data (annual average monthly discharge) from the USGS gage located upstream from the sites. This enabled us to isolate discharges and sequences of flows associated with plant recruitment events, fitness of adult trees, and to provide insight into possible effects of changes in flow.

Tree growth corresponding to median flow years and years of exceptionally low flow were examined. Interactions between age classes, elevation above channel, substrate composition, and incremental growth were also evaluated. For example, saplings and juveniles may be more sensitive to minor changes (e.g., reductions) in flow compared to adults.

Several cottonwood, Goodding willow, Arizona sycamore, and saltcedar individuals were excavated to determine maximum rooting depth relative to groundwater depths. Roots were exposed in holes excavated by backhoe and roots were revealed and followed to their terminus by hand with a digging bar and shovel. Depth to root terminus was measured and base of the trunk of the individual was surveyed as outlined above. Depth to exposed groundwater surface was also measured. These data and the groundwater data from the wells were used to determine the groundwater elevations below which the roots of cottonwood, Goodding willow, Arizona sycamore and saltcedar would be isolated from the water table.

Together, these relationships between surface and groundwater and riparian plant species of interest enabled us to draw conclusions about the probable effects of modified (e.g., reduced) flow regimes on recruitment, fitness, and stand characteristics along the study reaches.

Water Status of Riparian Species

To develop an understanding of the effects of flow diversion on the water status of riparian species of interest, we measured pre-dawn and mid-day water potential of cottonwood, Goodding willow, Arizona sycamore, and saltcedar. The tension between the atmosphere and roots is expressed as negative pressure (water potential) in the xylem vessels of the tree. Water potential provides a measure of water stress level in plants; higher water potential (less negative) indicates that plants have sufficient water, low water potential (more negative) indicates increasing water stress. Sampling stands were selected within 200 m upstream and downstream from an earthen diversion structure, which at the time of surveys was diverting 95 percent of the flow of Cherry Creek into a diversion ditch (Figure 4). Branch cuttings were taken from individuals and placed into PMS Model 670 Scholander-type pressure chambers (PMS Instruments, Corvallis, OR) and internal xylem pressure potential (Ψ) of the individual measured. Measurements were taken simultaneously by two investigators upstream and downstream from the diversion. In addition, at each stand, temperature and relative humidity were measured at 15 min. intervals with sheltered Onset Hobo RH/TEMP H8 data loggers (±5% RH, ±0.7°C at 20°C; Onset Computer Corporation, Bourne, MA). Vapor pressure deficit was calculated for the period of mid-day Ψ measurement (2:45-5:00 pm September 28, 2008).

Figure 4. Diverted reach of Cherry Creek. Looking upstream from the diversion (left panel), at the earthen dam (center), and downstream from the diversion (right panel). Ninety five percent of the flow in Cherry Creek was being diverted at the time of photographs.

Statistical Analyses

We related wildlife habitat to instream flows by first developing a habitat index for each vegetation plot, comparing indices between intermittent and perennial reaches, and then relating instream flows to differences observed in habitat. We developed habitat indices for birds and for small mammals and reptiles by using a subset of variables described in a full model of important habitat characteristics. We included variables based upon their weightings on factors in principal components analysis (PCA). Variables were identified based upon the number of significant factors determined by scree plots, factors with eigenvalues ≥ 1, and significance $P \leq 0.05$ (Legendre and Legendre 1998). This index included variables which contributed to significant factors in the PCA, reduced redundancy and correlation between variables (collinearity), and did not significantly differ from the index produced using the full model. We standardized (mean = 0 and standard deviation = 1) each of the selected variables and calculated their sum which gave us habitat index for each vegetation plot.

To relate wildlife habitat to instream flows, we compared the habitat index between two reaches (perennial and diverted) with varying stream flows. In addition, we compared riparian areas near the stream (hydro- and mesoriparian, *sensu* Stromberg *et al*. 2008) to drier riparian sites further from the channel (xeroriparian). Hydric plants are associated with wetland conditions and permanent water sources; mesic species require supplemental water exceeding that supplied by local precipitation, and xeric species are able to survive using moisture solely from local precipitation.

Woody vegetation was classified into distinct cover types using two-way indicator species analysis (TWINSPAN) – a hierarchical, divisive cluster analysis technique (Hill and Šmilauer 2005). We compared wildlife habitat indices among vegetation cover types using analysis of variance (ANOVA), and in cases of significant differences we conducted pairwise-comparisons (Tukey's adjustment) to determine which plant communities significantly differed in habitat quality. If the diversity of habitat significantly differed among vegetation cover types, we related habitat indices to specific woody species abundances that were dominant components of those cover types.

Presence of vegetation cover types was modeled as a function of depth to groundwater using logistic regression. Model fit was evaluated using -2 log likelihood statistics and Hosmer-Lemeshow goodness of fit tests. In addition, we determined relationships between several tree and shrub species (dominant species in vegetation cover types) and the habitat index.

We used discriminant function analysis[1] (DFA) to develop a discriminant function for classifying vegetation cover types as a function of water table elevation at stable low groundwater conditions (groundwater levels corresponding to a stable period of streamflow of 0.127 cms (4.5 cfs), this flow is exceeded 88 percent of the time). We combined cottonwood-dominated cover types into a single 'riparian forest' cover type, shrub-dominated cover types into a single 'shrubland' cover type, and left Goodding willow-dominated cover types as a third category. Goodding willow-dominated cover type was not used in the final discriminant analysis due to the small sample size (*n* = 3) of this vegetation cover type. We used this discriminant function to examine how frequency of cover types would likely change under a range of declining groundwater levels (0 to 1 m declines in groundwater at decimeter intervals). We used the discriminant function from groundwater levels at low flow and then forced groundwater levels down in decimeter (3.9 inch) intervals and reclassified cover types. These models are realized niche models and emphasize only the potential for cottonwood to exist at a particular site without regard for seed availability, germination, and recruitment requirements.

In addition to constructing statistical models and simulating effects of groundwater decline on riparian forest cover types, we determined the area of potential cottonwood habitat along the portion of the main study reach using depth to groundwater at a stable low flow (indicated above) and average rooting depth of cottonwood (1.74 m) from six excavated individuals along Cherry Creek. Using measured groundwater depths at this stable flow condition and the detailed digital elevation model generated by the USGS (Waddle and Bovee

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[1] Discriminant function analysis (DFA) is a multivariate statistical technique for developing a mathematical rule for classifying observations into predetermined categories based upon quantitative variables associated with each observation. Here we used depth to groundwater to classify measured vegetation plots into statistically classified vegetation cover types.

2009) we constructed maps of potential cottonwood habitat. We then mapped changes in cottonwood forest habitat as a function of lowering the water table.

The 172 tree cores for which we were able to determine age were used to develop diameter-age relationships for cottonwood, Goodding willow, sycamore, and saltcedar along the main study reach. These relationships were used to predict ages for all individuals measured in the vegetation sampling plots. Of the 6,972 stem diameters measured, the maximum stem diameter for each of the 3,641 individual plants was used for age estimates. These values were plotted for each species to develop age-class distributions and to evaluate years of establishment.

Along the upper segment, ordination was used to quantify plant species turnover and compositional change along the gradient from intermittent to perennial. Analysis of similarity (ANOSIM) was used to statistically compare species composition between intermittent and perennial reaches (Legendre and Legendre 1998). These analyses enabled us to examine species compositional change along the xeric to mesic gradient as an analogue to species change caused by decreasing water availability.

Hydrographs of surface and groundwater were constructed at various stream stages to determine temporal patterns of groundwater and surface water interactions (e.g., gaining versus losing stream reaches) and lag times between changes in stream stage and groundwater levels. Surfaces were interpolated using spline fit using a grid size of 0.3 m. Statistical relationships between each well and staff gage were determined using Spearman rank order correlation.

Differences in mean water potential of each of the species of interest were compared between reaches upstream and downstream from the diversion using two-way *t*-tests. Temperature, relative humidity, and vapor pressure deficit were also evaluated between upstream and downstream from the diverted reach using *t*-tests.

When data satisfied assumptions of statistical tests we used parametric statistical tests (e.g., *t*-test and ANOVA), when data did not comply with assumptions we used non-parametric tests (e.g., Mann-Whitney *U* test or Kruskal-Wallis test; Zar 1996).

Results

Vegetation Cover Types

Thirty seven species of woody plants and 82 herbaceous species were identified in 175 vegetation plots sampled along all studied reaches of Cherry Creek. Species richness ranged from 0 (bare plots) to 21 species and cover ranged from 0 to 250 percent (multilayered canopies). The first three axes of the detrended correspondence analysis (DCA) explained 49 percent of the variation in composition of woody species. DCA axis 1 was most strongly associated with cottonwood, Goodding willow, Arizona alder (*Alnus oblongifolia*), seepwillow (*Baccharis salicifolia*), saltcedar, and sandbar willow (high scores on axis 1) and desert broom (*Baccharis sarothroides*), and burrobrush (*Hymenoclea monogyra*; low scores on axis 1). Axis 2 was driven by mesquite (*Prosopis velutina*) and snakeweed (*Gutierrezia sarothrae*; high values on axis 2) and saltcedar (low value on axis 2). Classification of woody vegetation resulted in nine distinct vegetation cover types along the intensive study reach of Cherry Creek and one additional unique cover type along the intermittent reach (Table 1).

Table 1. Riparian vegetation cover types in 170 vegetation plots sampled along four reaches of Cherry Creek determined using cluster analysis. Reaches included two reaches along an intermittent to perennial gradient, a main perennial study reach, and a diverted reach.

Riparian vegetation cover types	Frequency of occurrence (%)				
	Main study reach *n* = 79	Diverted reach *n* = 14	Intermittent reach *n* = 23	Perennial reach *n* = 54	All reaches *n* = 170
1. Mesquite/desert broom/ burrobrush/catclaw acacia	3	7	13	6	9
2. Desert broom/ burrobrush/ mesquite	8	29	4	9	16
3. Burrobrush	6	0	18	13	16
4. Burrobrush/saltcedar	11	7	4	0	6
5. Burrobrush/desert broom	18	21	52	11	21
6. Saltcedar/cottonwood/ Goodding willow	13	0	0	0	6
7. Cottonwood/willow/ seepwillow/burrobrush	16	29	0	4	11
8. Cottonwood/Goodding willow/ sycamore/saltcedar	22	7	0	0	11
9. Goodding willow/seepwillow	4	0	9	30	12
10. Cottonwood/Goodding willow/ seepwillow/Arizona alder	0	0	0	28	9

The most frequently occurring cover types in the 80 plots sampled along the main study reach were Cottonwood/Goodding willow/sycamore/saltcedar (20 percent of plots) followed by Cottonwood/willow/seepwillow/burrobrush and Burrobrush/desert broom which both occurred in 18 percent of plots. The least frequent cover types were Goodding willow/seepwillow (3 percent) and Mesquite/desert broom/burrobrush/catclaw acacia (4 percent).

Bird Habitat

The bird habitat index (Figure 5) was best explained by seven variables quantifying diversity of vegetation height and size and foliage cover (PCA analysis; Factors 1 & 2, *P* = 0.05;

Table 2). This model avoided collinearity and did not significantly differ from the full model of 15 variables ($r^2 = 0.98$, $P \leq 0.0001$). Overall, index of bird habitat differed among reaches in Cherry Creek ($F_{2,91} = 4.1$, $P = 0.02$; Figure 6). The main reach had a significantly greater average habitat complexity (0.23 ± 1.07) than the diverted reach (-0.52 ± 0.76) and the xeroriparian area (-0.21 ± 0.81). The later was intermediate between main and diverted reaches (Pairwise comparisons with Tukey's adjustment; Table 2). We found that species composition of the vegetation along the main reach and the diverted reach differed significantly (ANOSIM $R = 0.12$. $P = 0.03$)[2]. Hydro- and mesoriparian vegetation near the channel differed from xeroriparian further from the channel (ANOSIM $R = 0.11$. $P = 0.001$) along the free-flowing reach. Xeroriparian along the perennial reach and the vegetation adjacent to the channel along the diverted reach did not significantly differ (ANOSIM $R = 0.09$. $P = 0.08$).

Species that were more abundant along the free-flowing reach compared to both the diverted and xeroriparian sites included cottonwood (average cover 11 percent), Goodding willow (cover 8 percent), and saltcedar (cover 5 percent). Species that were more abundant along both the xeroriparian and the diverted reach included burrobrush (cover 10-24 percent) and desert broom (cover 16 to 17 percent). In addition, mesquite was more abundant along the diverted reach (cover 9 percent) compared to the free-flowing (cover 2.4 percent) and the xeroriparian reach (cover 2 percent).

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[2] *R* provides an index of how different plant community composition is ranging from very similar, *R* near 0, to completely different, *R* approaching 1.0.

Table 2. Variables included in habitat models to describe bird and mammal and reptile habitat complexity along Cherry Creek. Values are the mean ± 1 standard error.

Wildlife Habitat Variables	Main reach (*n*= 54)	Diverted reach (*n* = 14)	Xero-Riparian (*n* = 24)	$F_{2,91}$	P
Canopy cover (%)	54.5 ± 35.4	46.0 ± 27.1	54.5 ± 31.7	0.4	0.07
Basal area (cm^2/plot) [A,B]	449.3 ± 737	98.2 ± 313.4	141.6 ± 398.5	3.2	0.05
Quadratic mean woody tree diameter	4.12 ± 4.8	2.9 ± 6.2	3.0 ± 4.5	0.6	0.54
Standard deviation woody tree diameter [A,B]	4.1 ± 5.4	2.9 ± 7.8	1.9 ± 3.7	1.4	0.25
Riparian tree overstory cover (%) [B]	23.0 ± 32.3^{a}	4.5 ± 10.7^{b}	7.0 ± 17.2^{a}	4.6	0.01
Abundance of seedlings [B]	72.4 ± 199.9	7.5 ± 20.9	0.68 ± 1.4	2.3	0.10
Foliage height diversity (H') [A]	1.1 ± 0.5^{a}	0.7 ± 0.5^{b}	1.0 ± 0.5^{ab}	4.8	0.01
Foliage hits [A]	7.3 ± 4.7^{a}	3.9 ± 3.1^{b}	5.5 ± 3.0^{ab}	4.5	0.01
Foliage height diversity (H') 0-1.5 m	0.74 ± 0.33^{a}	0.42 ± 0.45^{b}	0.75 ± 0.33^{a}	5.1	0.01
Mean vegetation cover height class 1 (%) [A]	2.2 ± 1.0	1.6 ± 0.9	2.1 ± 0.9	2.0	0.14
Mean vegetation cover height class 2 (%)	1.2 ± 0.2	0.7 ± 0.8	3.9 ± 0.10	1.9	0.16
Mean vegetation cover height class 3 (%)	0.6 ± 1.3	0.2 ± 0.4	0.26 ± 0.75	1.2	0.29
Mean vegetation cover height class 4 (%) [A]	0.3 ± 1.0	0.0 ± 0.0	0.1 ± 0.24	1.3	0.28
Standard deviation vegetation cover ht. class 1 (%)	1.2 ± 0.5	1.0 ± 0.5	1.1 ± 0.5	1.7	0.18
Standard deviation vegetation cover ht. class 2 (%) [A]	1.1 ± 0.7	0.9 ± 0.8	0.9 ± 0.8	1.0	0.38
Standard deviation vegetation cover ht. class 3 (%)	0.5 ± 0.9	0.4 ± 0.9	0.3 ± 0.5	0.8	0.44
Standard deviation vegetation cover ht. class 4 (%)	0.3 ± 0.8	0.0 ± 0.0	0.1 ± 0.5	1.3	0.29
Mean vegetation cover for mammals/retiles (%) [B]	163 ± 193.4	69.0 ± 57.8	97.2 ± 98.4	2.80	0.07
Stand. Dev. Veg. cover for mammals/retiles (%) [B]	31.6 ± 29.8	20.3 ± 22.7	19.8 ± 21.2	2.20	0.12
Proportion of sandy substrate (%) [B]	0.57 ± 0.31	0.59 ± 0.39	0.64 ± 0.30	0.43	0.65

[A] Included in bird habitat index

[B] Included in mammal/reptile habitat index

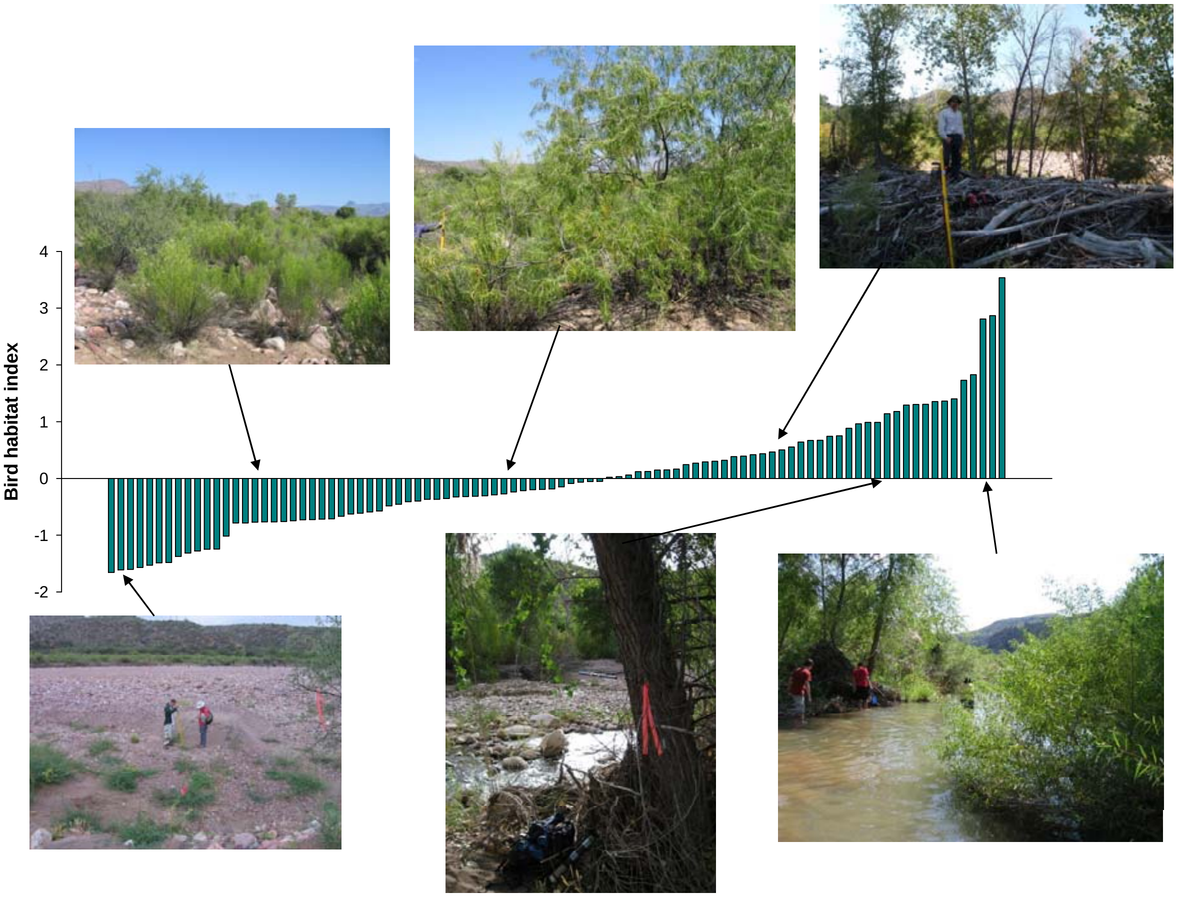

Figure 5. Bird habitat index for 67 vegetation plots along the main study reach of Cherry Creek. Scale is relative having been adjusted so the mean and standard deviation of all observations equals 0 and 1, respectively. Low values indicate relatively simple habitat that is not likely to support species-rich bird communities; high numbers represent complex habitats capable of harboring relatively species-rich communities.

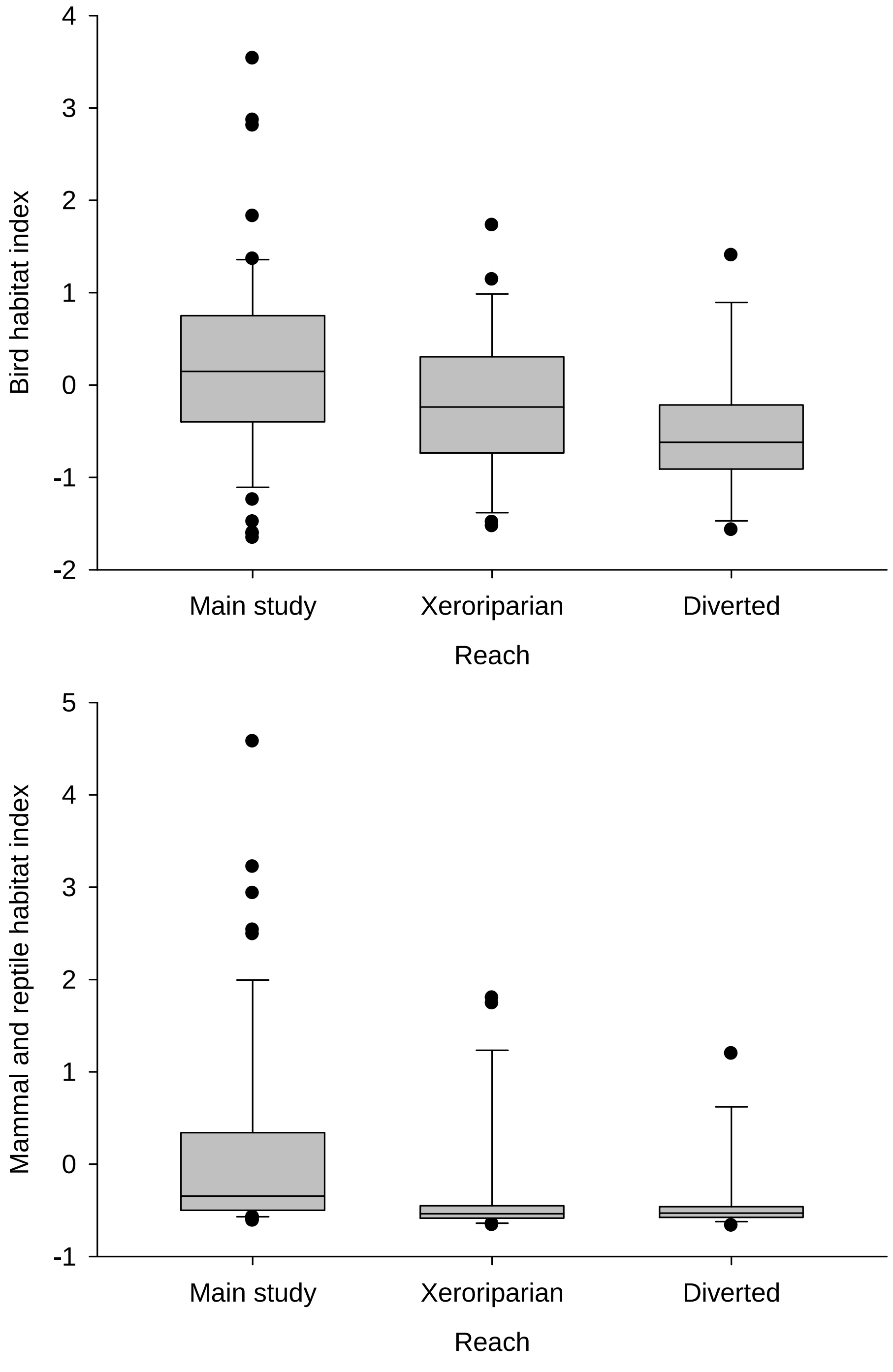

Figure 6. Bird and mammal and reptile habitat indices for three segments of Cherry Creek. Values are relative; higher values indicate higher habitat values.

Riparian vegetation attributes were statistically associated with bird habitat. The index of bird habitat was significantly greater in cottonwood/Goodding willow/sycamore (cover type 8) communities compared to burrobrush (cover type 3) and burrobrush/saltcedar (cover type 4) communities ($F_{8,85} = 3.5$, $P = 0.0017$; Figure 7; Table 3). Relating bird habitat to specific woody species from the vegetation types, showed that the index of bird habitat was greater in areas with more cover of riparian tree species (i.e., cottonwood, $F = 48.1$, $P \leq 0.0001$, $r^2 = 0.35$; Goodding willow, $F = 18.1$, $P \leq 0.0001$, $r^2 = 0.17$; and sycamore $F = 15.3$, $P = 0.0002$, $r^2 = 0.15$; Figures 8 through 10) and unrelated to areas where species of upland/xeric plant species dominated (i.e., burrobrush, $F = 1.0$, $P = 0.31$, $r^2 = 0.01$; mesquite $F = 0.10$, $P = 0.75$, $r^2 = 0.001$; and saltcedar $F = 2.8$, $P = 0.11$, $r^2 = 0.07$; Figures 11 through 13). Saltcedar only contributed to the habitat index where it co-occurred with cottonwoods and willows ($F = 5.4$, $P = 0.02$, $r^2 = 0.06$; Figure 13). The best descriptor of Bird habitat index was cover of native riparian tree species combined ($F = 106.0$, $P < 0.0001$, $r^2 = 0.54$; Figure 14).

Table 3. Comparison of vegetation cover types in relation to bird habitat index. Bolded cover types significantly differed in bird habitat. Cover types corresponding to numbers are described in Table 1 and Figure 7.

Comparisons of vegetation cover types	*P-value*
2,8	> 0.05
3,6	> 0.05
3,8	**< 0.05**
3,9	> 0.05
4,8	> 0.05
5,6	> 0.05
5,8	**< 0.05**
5,9	> 0.05
7,8	> 0.05

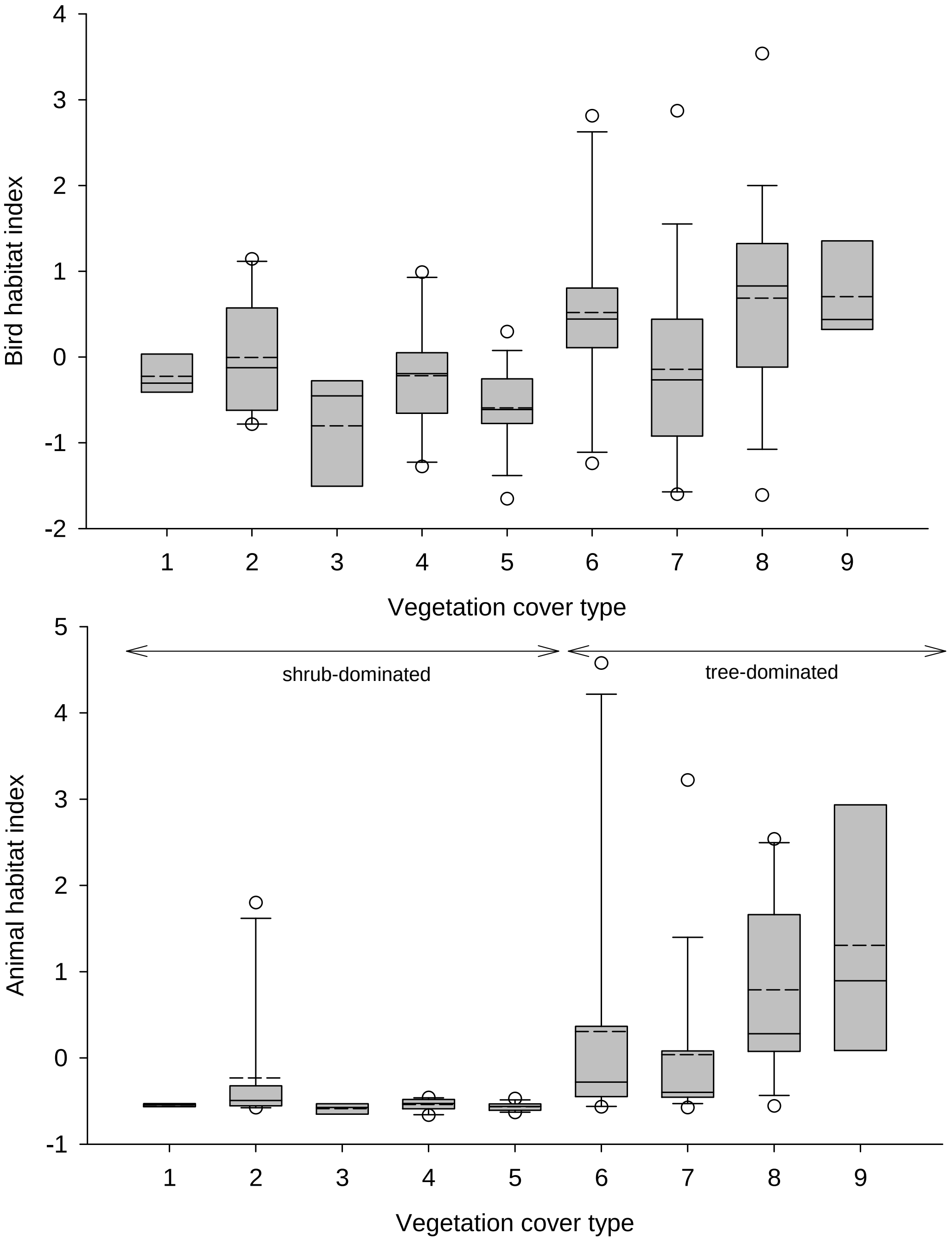

Figure 7. Bird and animal habitat index within nine vegetation cover types along the main and diverted study segment of Cherry Creek. Vegetation cover types: 1. Mesquite/desert broom/burrobrush/catclaw acacia (9 vegetation plots), 2. Desert broom/burrobrush/mesquite, 3. Burrobrush, 4. Burrobrush/saltcedar, 5. Burrobrush/desert broom, 6. Salt cedar/cottonwood/Goodding willow, 7. Cottonwood/willow/seepwillow/burrobrush, 8. Cottonwood/Goodding willow/sycamore/saltcedar, 9. Goodding willow/seepwillow.

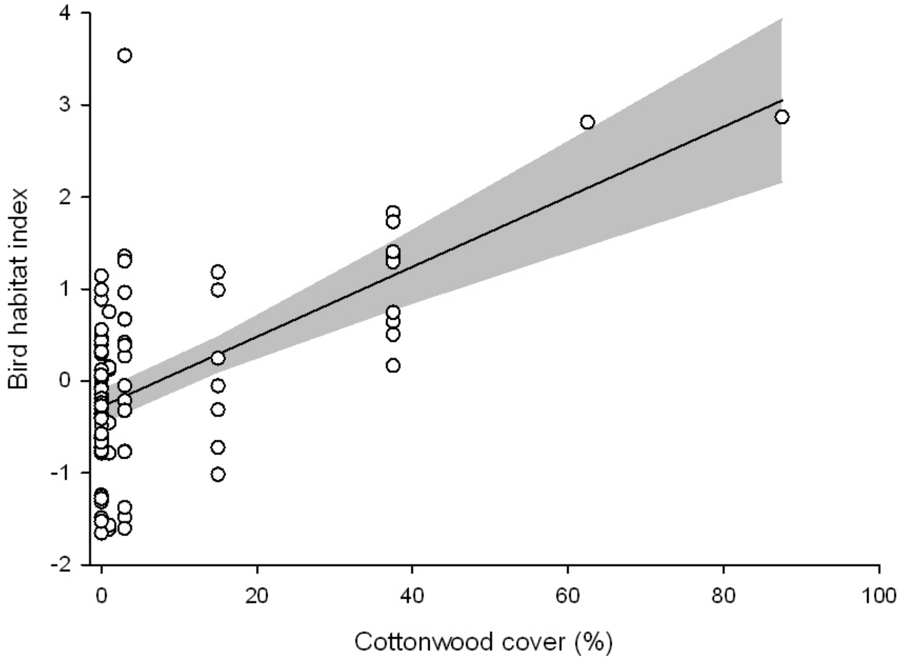

Figure 8. Bird habitat index related to percent cover of cottonwood trees (in the absence of non-native saltcedar; $r^2 = 0.35$; $P < 0.0001$).

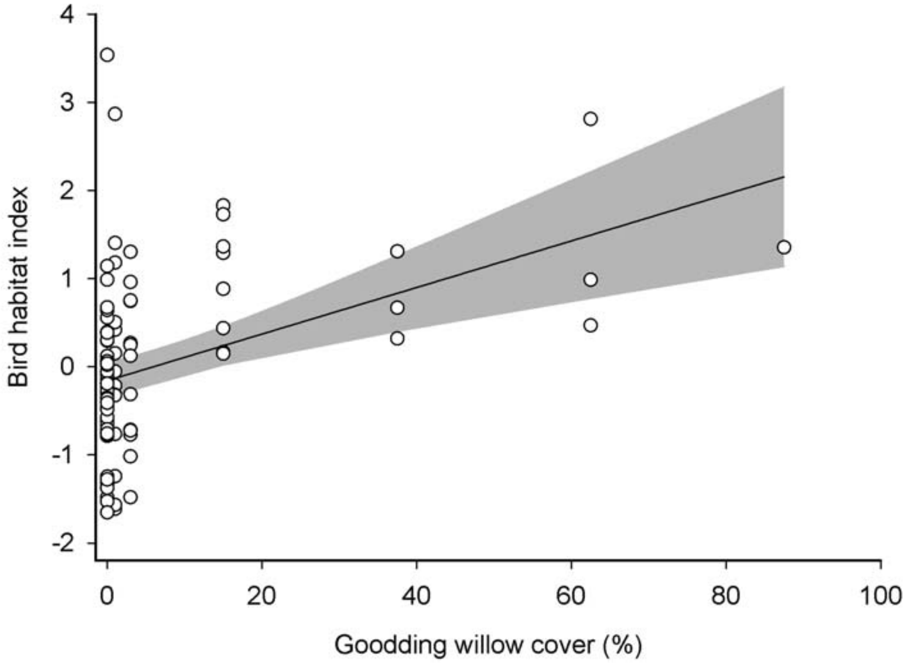

Figure 9. Bird habitat index related to percent cover of Goodding willow trees ($r^2 = 0.17$; $P < 0.0001$).

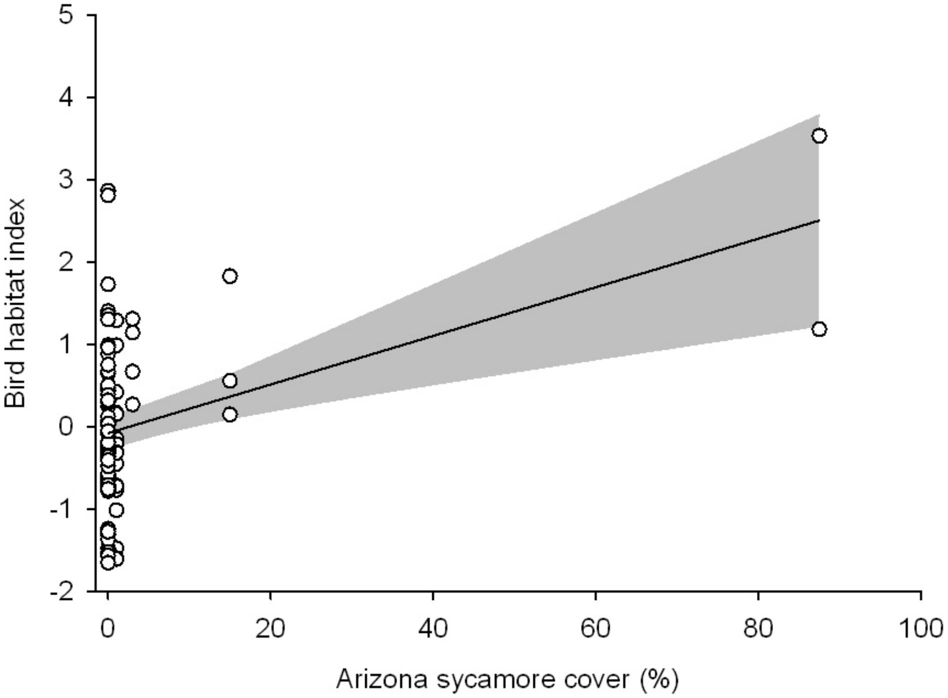

Figure 10. Bird habitat index related to percent cover of Arizona sycamore trees (r^2 = 0.17; *P* = 0.0002).

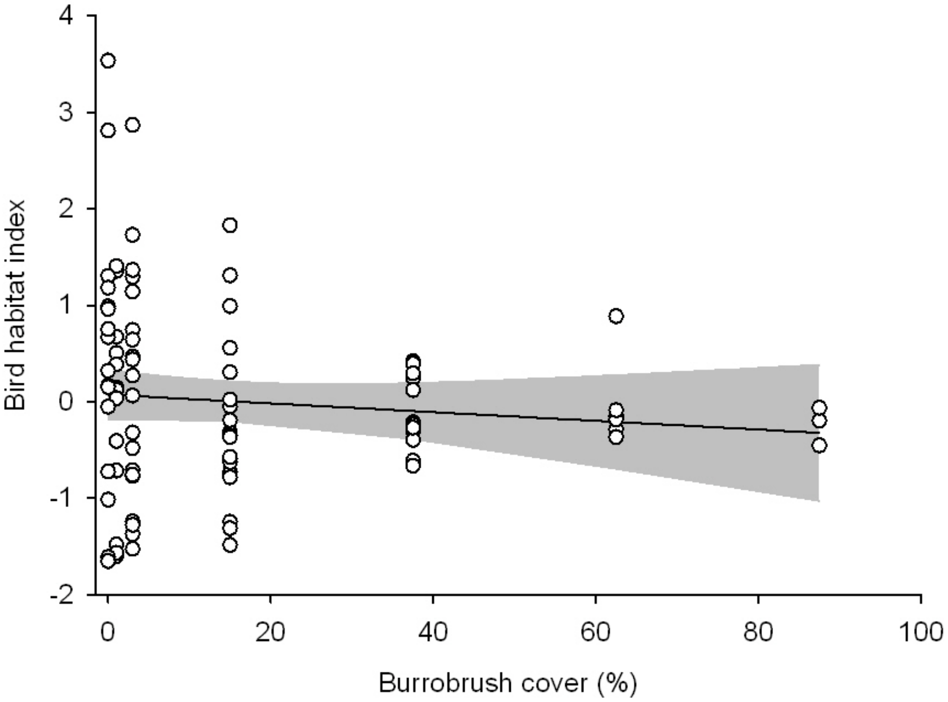

Figure 11. Bird habitat index related to percent cover of burrobrush (r^2 = 0.01; *P* = 0.3).

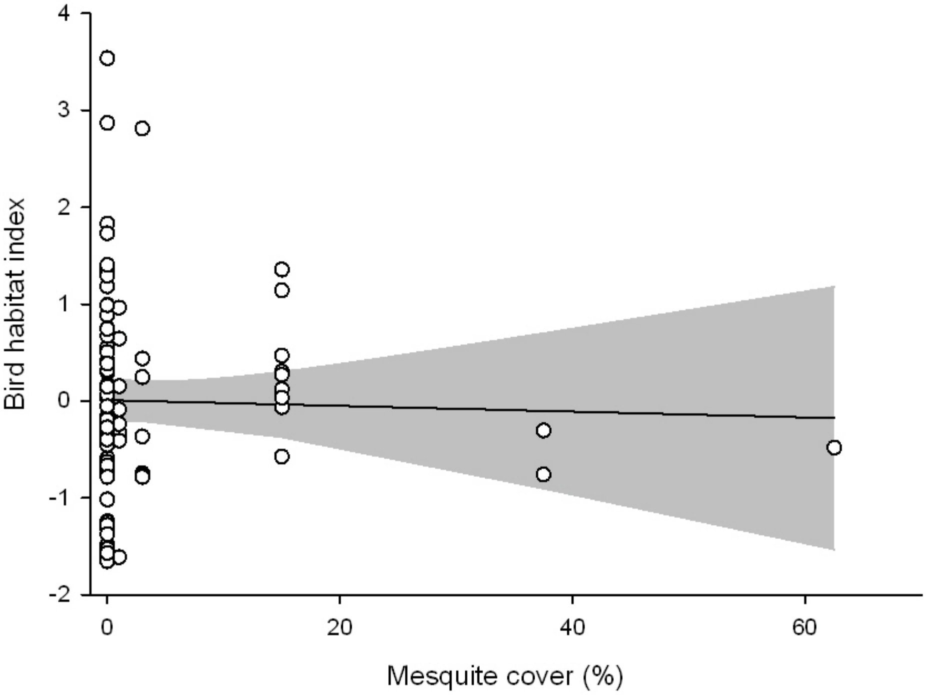

Figure 12. Bird habitat index related to percent cover of mesquite (r^2 = 0.001; *P* = 0.8).

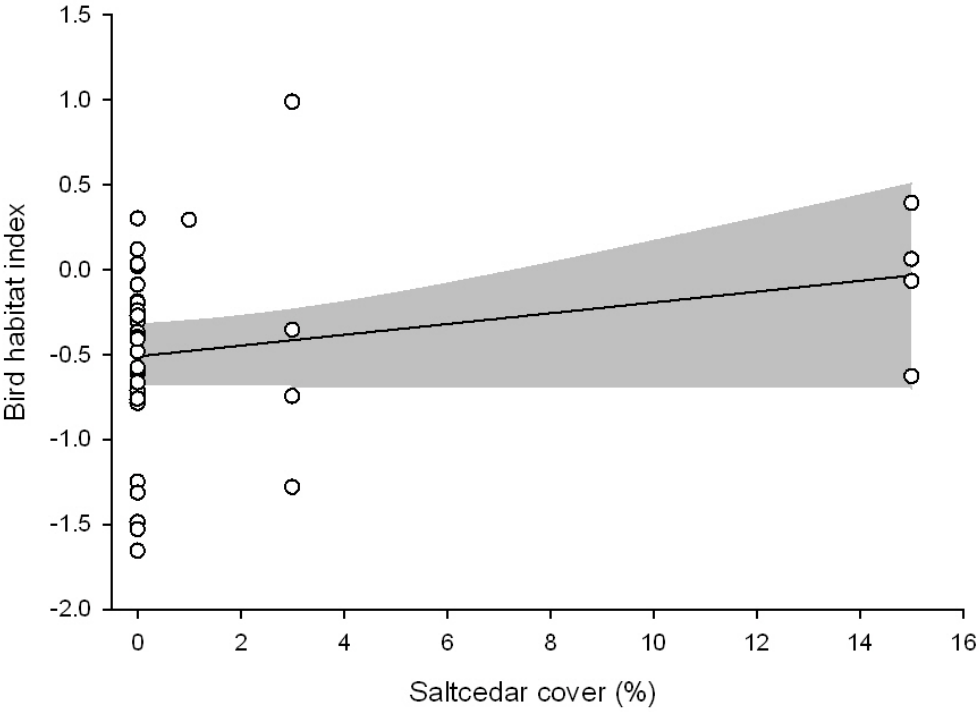

Figure 13. Bird habitat index related to cover of saltcedar in the absence of cottonwood, willow, and sycamore trees (r^2 = 0.07; *P* = 0.1).

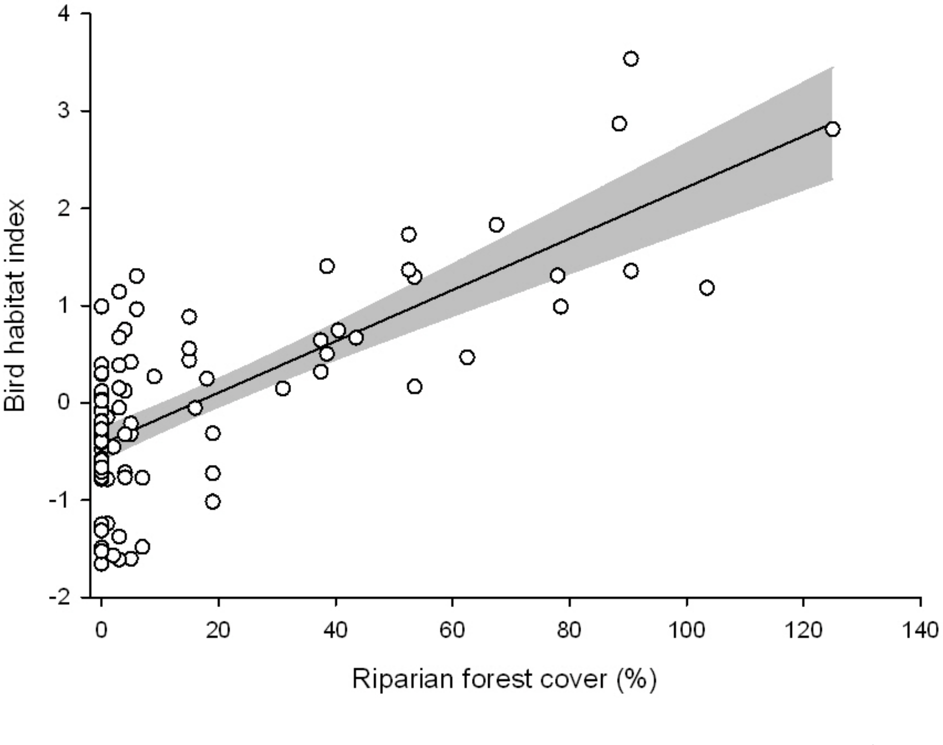

Figure 14. Bird habitat index related to cover of native riparian forest species (cottonwood, Goodding willow, and Arizona sycamore; r^2 = 0.54; *P* < 0.0001).

Small Mammal and Reptile Habitat

The index of small mammal and reptile habitat was best explained by seven variables describing foliage cover and diversity, abundance of cottonwoods, and substrate (PCA analysis; Factors 1 and 2, $P = 0.05$). This model reduced collinearity and did not significantly differ from the full model ($r^2 = 0.97$, $P \leq 0.0001$). Because a Levine's test indicated that the equal variance assumption was violated, we conducted ANOVA on ranks. Overall, index of habitat differed among reaches in Cherry Creek (Kruskal Wallis $x^2 = 15.2$, $P = 0.0005$, Bonferroni adjustment significance level $P = 0.017$; Figure 6). The main reach had a significantly greater habitat diversity than the diverted reach (Mann-Whitney U $x^2 = 8.4$, df = 2, $P = 0.004$) and the xeroriparian area ($x^2 = 10.6$, df = 2, $P = 0.001$). The diverted reach and outer riparian area did not differ ($x^2 = 0$, df = 2, $P = 0.99$).

Riparian vegetation attributes were also important drivers of small mammal and reptile habitat. The index of habitat was significantly greater in riparian forest communities dominated by cottonwood (types 6 through 9) compared to non-cottonwood, shrub-dominated communities (types 1-5; Kruskal Wallis $x^2 = 32.6$, df = 1, $P < 0.0001$; Figure 7). Structural complexity increased from an average low of -0.79 to a high of 1.2 along the measured species composition gradient. Since small mammal and reptile habitat indices were constructed using abundances of species of woody plants, no further analyses were conducted to relate habitat to percent cover of these species.

Vegetation and Hydrological Factors

We determined that depth to groundwater was an important driver of riparian tree species presence, abundance (basal area and cover), and health (indicated by water stress) along Cherry Creek. Generalized linear modeling of plant-habitat community type as a function of depth to water table resulted in a family of models for predicting probability of cover type across the range of observed depths to groundwater across the floodplain. Riparian tree species were more common in areas with shallow groundwater. For example, cottonwood, Goodding willow, and saltcedar (Figures 15 through 17) were more likely to occur in areas with shallow groundwater (less than 1.5. m deep); whereas, desert broom was more likely to be present where groundwater was deeper (greater than 1.5 m deep; Figure 18).

Cottonwood/Goodding willow/sycamore/saltcedar, Cottonwood/willow/ seepwillow/burrobrush, and Saltcedar/cottonwood/Goodding willow cover types occurred where the groundwater averaged 1.0, 1.1, and 1.3 m in depth, respectively. These cover types were not found in areas where groundwater was deeper than 2.03 m below the ground surface. In contrast, Desert broom/burrobrush/mesquite (mean and maximum groundwater depths were 1.5 m and 2.0), Burrobrush (mean and maximum groundwater depths were 1.8 and 2.4 m), Burrobrush/saltcedar and Burrobrush/desert broom (mean and maximum depths to groundwater were 1.9 and 2.5 m) were found on sites with deeper water tables. The two plots classified as Mesquite/desert broom/burrobrush/catclaw acacia in the main study reach were in sites with groundwater depths of 1.4 and 1.5 m.

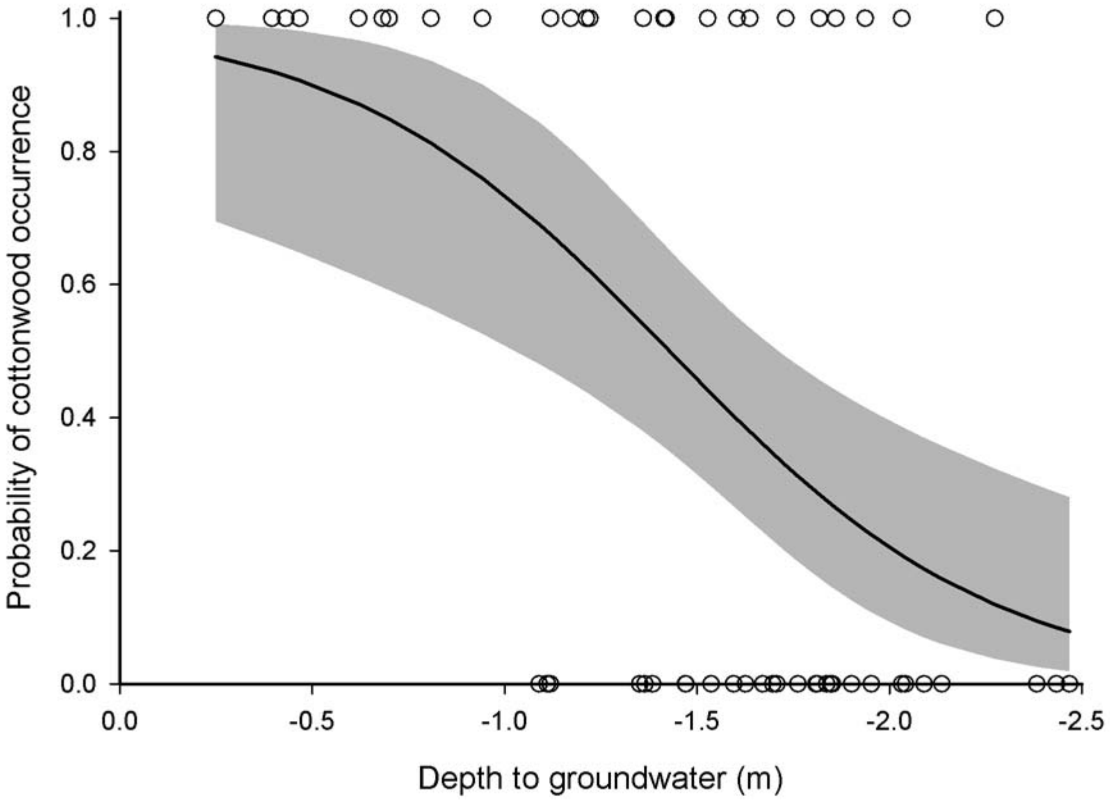

Figure 15. Probability of cottonwood occurrence in relation to groundwater depth. Points represent plots with absences (0) and presences (1) of cottonwood. The model was significant and correctly predicting cottonwood presence or absence for 75 percent of the observations (Likelihood ratio *P* < 0.0001; Hosmer-Lemeshow goodness of fit *P* = 0.8).

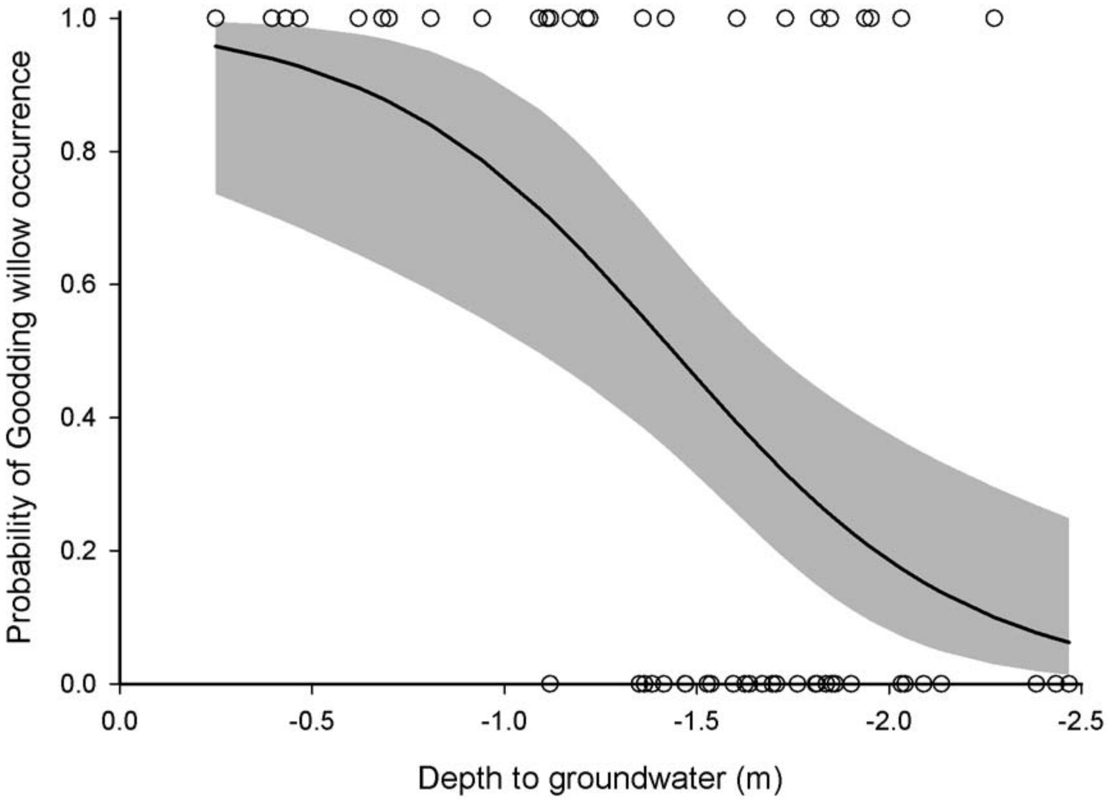

Figure 16. Probability of Goodding willow occurrence in relation to groundwater depth. Points represent plots with absences (0) and presences (1) of Goodding willow. The model was significant and correctly predicting Goodding willow presence or absence for 77 percent of the observations (Likelihood ratio $P < 0.0001$; Hosmer-Lemeshow goodness of fit $P = 0.2$).

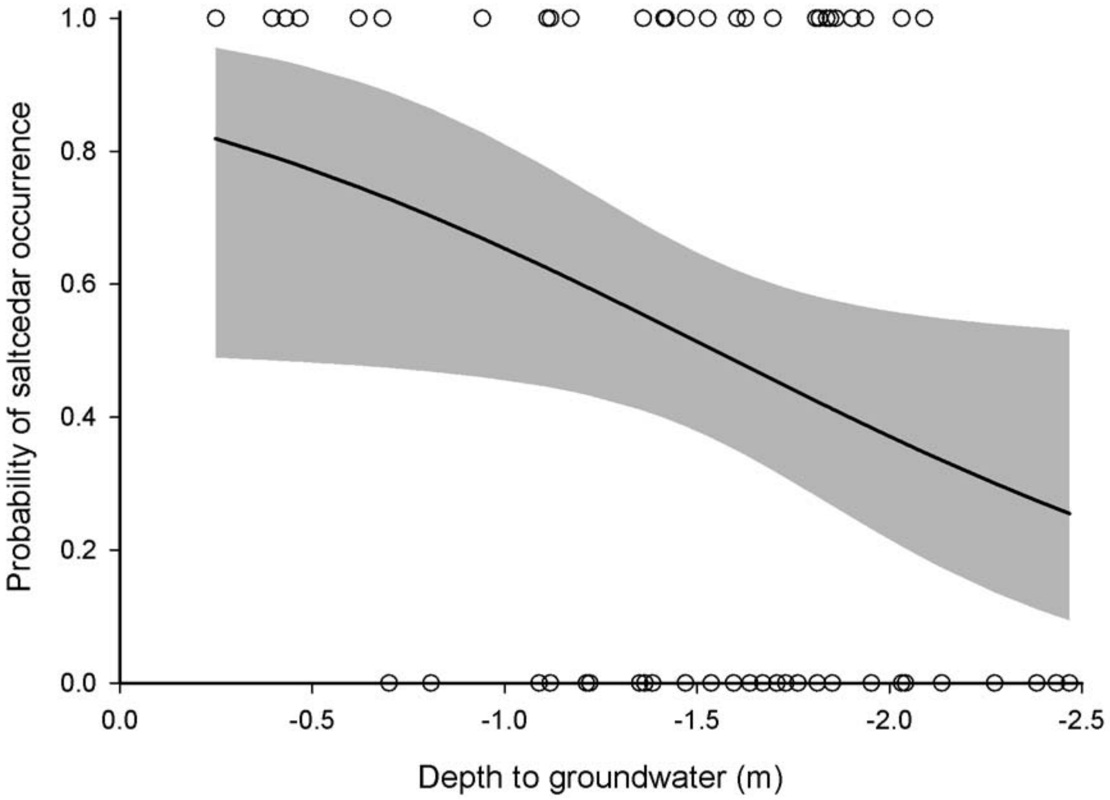

Figure 17. Probability of saltcedar occurrence in relation to groundwater depth. Points represent plots with absences (0) and presences (1) of saltcedar. The model was significant and correctly predicting tamarisk presence or absence for 63 percent of the observations (Likelihood ratio *P* = 0.03; Hosmer-Lemeshow goodness of fit *P* = 0.6).

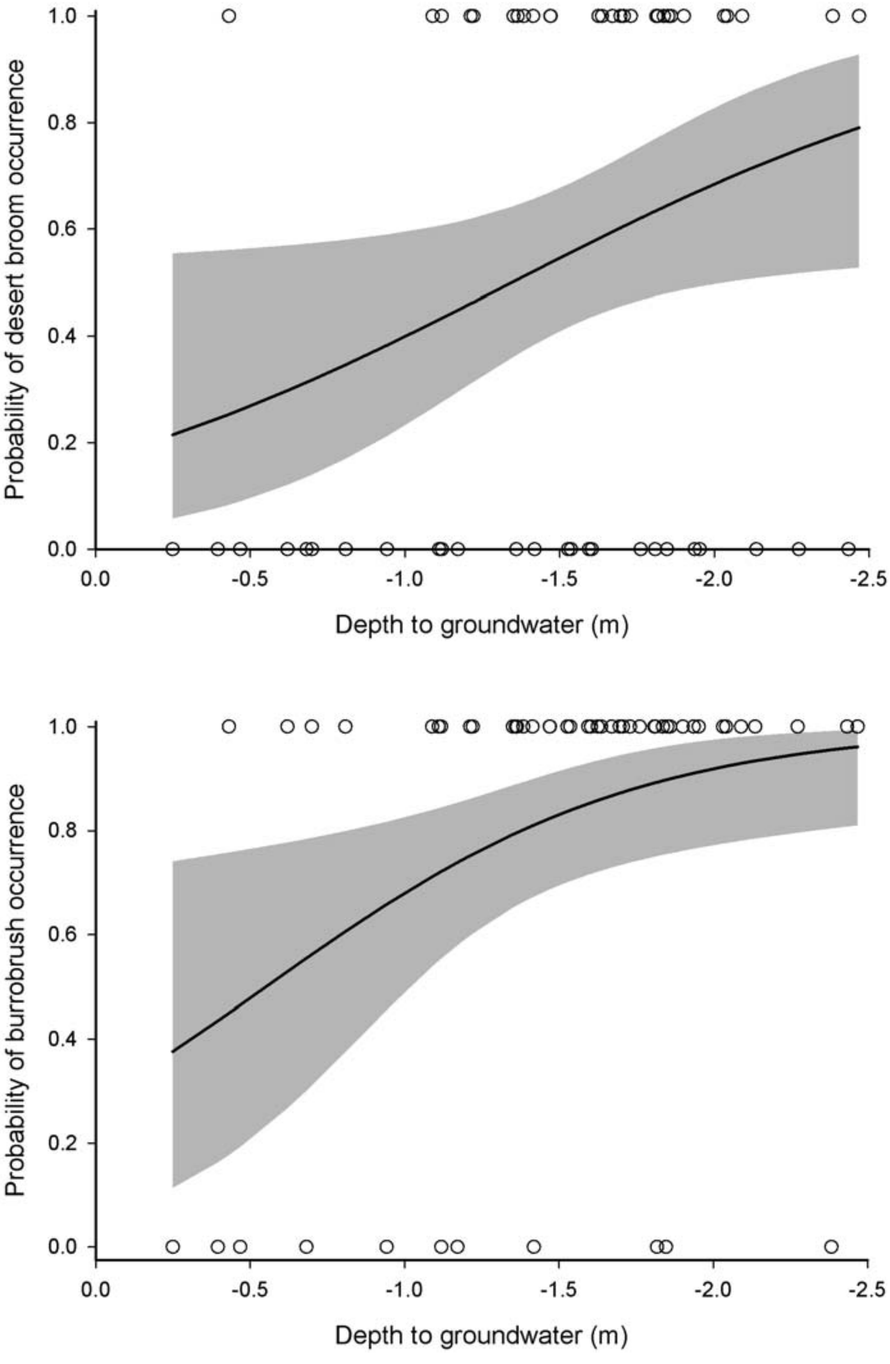

Figure 18 a and b. Probability of desert broom and burrobrush occurrence in relation to groundwater depth. Points represent plots with absences (0) and presences (1) of each species. The models were both significant and correctly predicted presence or absence for 65 and 72 percent of the observations (Likelihood ratio *P* = 0.03; Hosmer-Lemeshow goodness of fit *P* = 0. 5 and 0.3 for desert broom and burrobrush, respectively).

The discriminant function did an excellent job of correctly classifying vegetation cover types into riparian forest or shrubland cover types based upon depth to groundwater (Wilks' Lambda 0.74, $F_{1,51} = 18.0$; $P < 0.0001$). Seventy-four percent of plots were correctly classified as *riparian forest* or *shrubland* based upon depth to water table in the plot at a stable, base-level. Riparian forest was correctly classified in 67 percent of cases and Shrubland was correctly classified at a rate of 81 percent. At the stable modeled flow, depth to groundwater in the cover types averaged 1.1 ± 0.63 m (± standard deviation) in riparian forest and 1.7 ± 0.37 m in shrublands. Shrubland frequency remained relatively stable over the entire range of increasing depth to groundwater, ranging between 59 and 93 percent (Figure 19). This suggests that shrublands are not sensitive to water table declines of less than 1 m below baseline. However, the riparian forest cover type declined 83 percent from 41 percent frequency at the baseline condition to 7 percent at a groundwater depth of 1 m below modeled baseline condition. A simulated groundwater decline of 2 m below base level resulted in a nearly complete loss of riparian forest (1.8 percent frequency) and dominance of plots by shrubland (98.2 percent frequency).

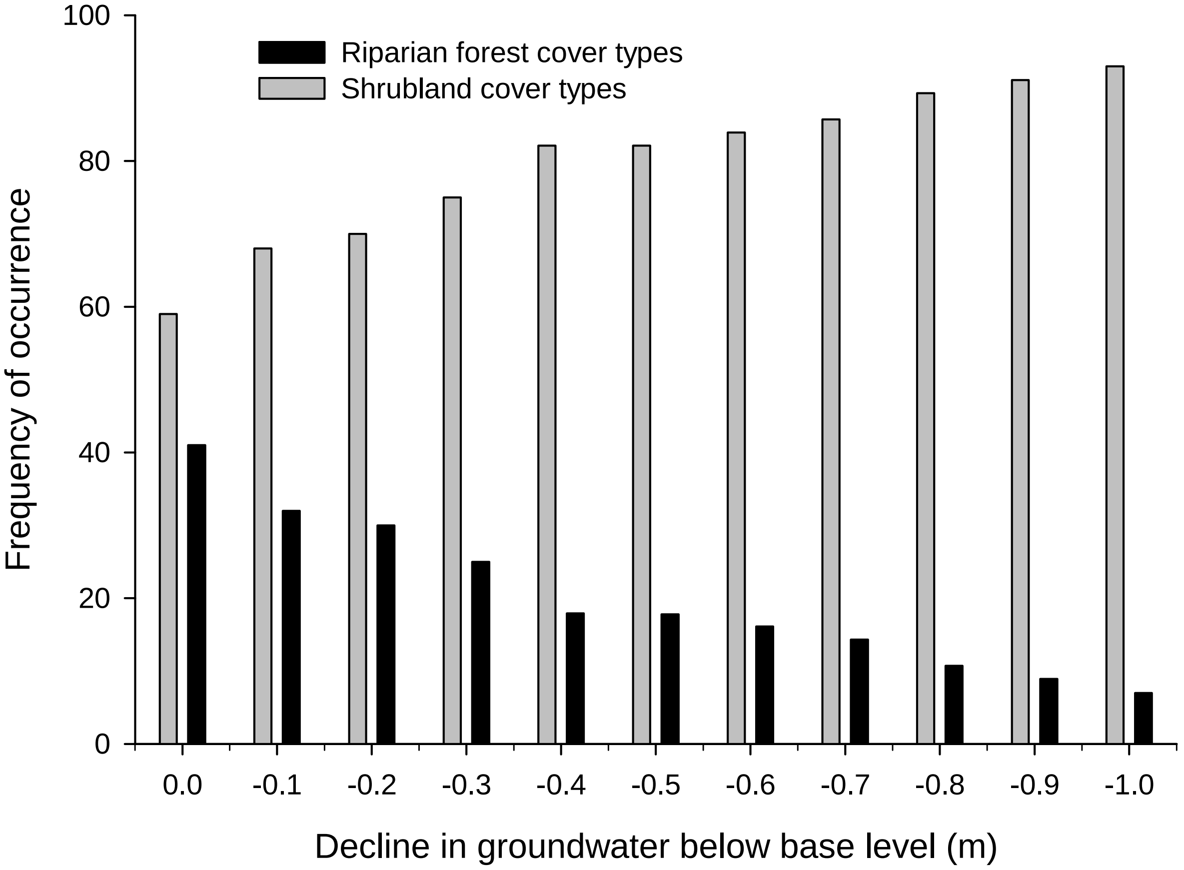

Figure 19. Frequency of plots (*n* = 67) classified as Riparian forest cover type and those classified as Shrubland cover types as a function of lowering groundwater levels (and river stage) in the main study reach of Cherry Creek. The discriminant function was constructed using the groundwater levels measured at streamflow of 0.127cms (4.5 cfs).

Fourteen trees were excavated and root systems measured. Average root depth of Frémont cottonwood was 1.74 ± 0.22 m (± standard error; range 1.25 to 2.51 m), Goodding willow was 1.50 ± 0.005 m (range 1.52 to 1.55 m), Arizona sycamore was 1.41 ± 0.14 m (range 1.20 to 1.68 m), and saltcedar was 1.68 ± 0.21 m (range 1.27 to 1.91 m). A ten year old cottonwood had the shallowest cottonwood roots excavated and a 16 year old cottonwood located at the extreme western edge of the valley had the deepest roots excavated (Figure 20).

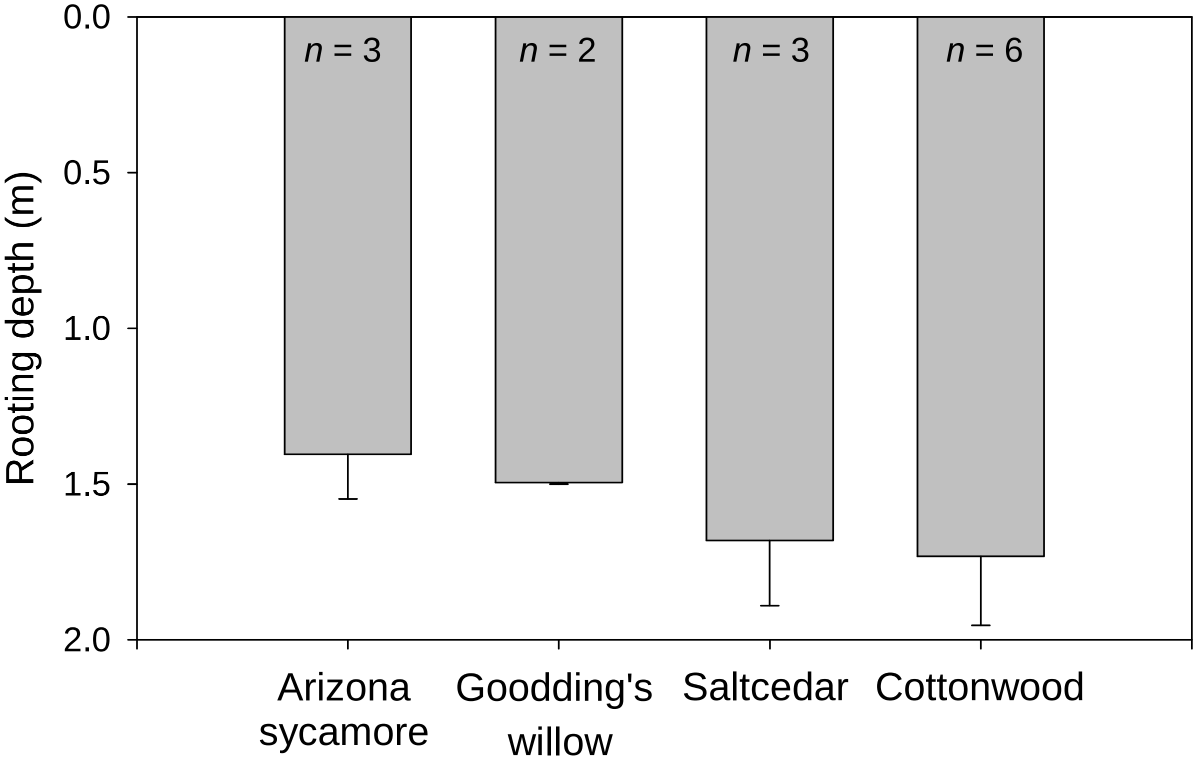

Figure 20. Measured depth (mean ±1 S.E.) to water table for four species of interest that were excavated from the study reach of Cherry Creek.

Models of cottonwood habitat suitability based upon mean cottonwood rooting depth indicated that about 65 percent of the floodplain is potential cottonwood habitat at modeled low flow conditions (0.127cms (4.5 cfs) depth to groundwater within 1.74 m of ground surface). Simulating groundwater declines of 10 to 100 cm resulted in incremental reductions of potential riparian forest habitat to a low of 16 percent of the floodplain (Figure 21 and 22). Declines of 1.5 to 2.0 m (4.9 to 6.6 ft) below base groundwater level resulted in forest cover of 7 and 3 percent and dominance of the valley bottom and riparian areas by shrubs (93 to 97 percent). It should be noted that these models only indicate where mature cottonwood could potentially exist, they do not account for the fluvial and hydrologic processes necessary for recruitment to occur. Though herbaceous riparian species may be more vulnerable than woody species to groundwater declines (Stromberg *et al.* 1996, J.C. Stromberg in Haney *et al.* 2008), we focus only upon woody species here due to their role in conditioning habitat structure.

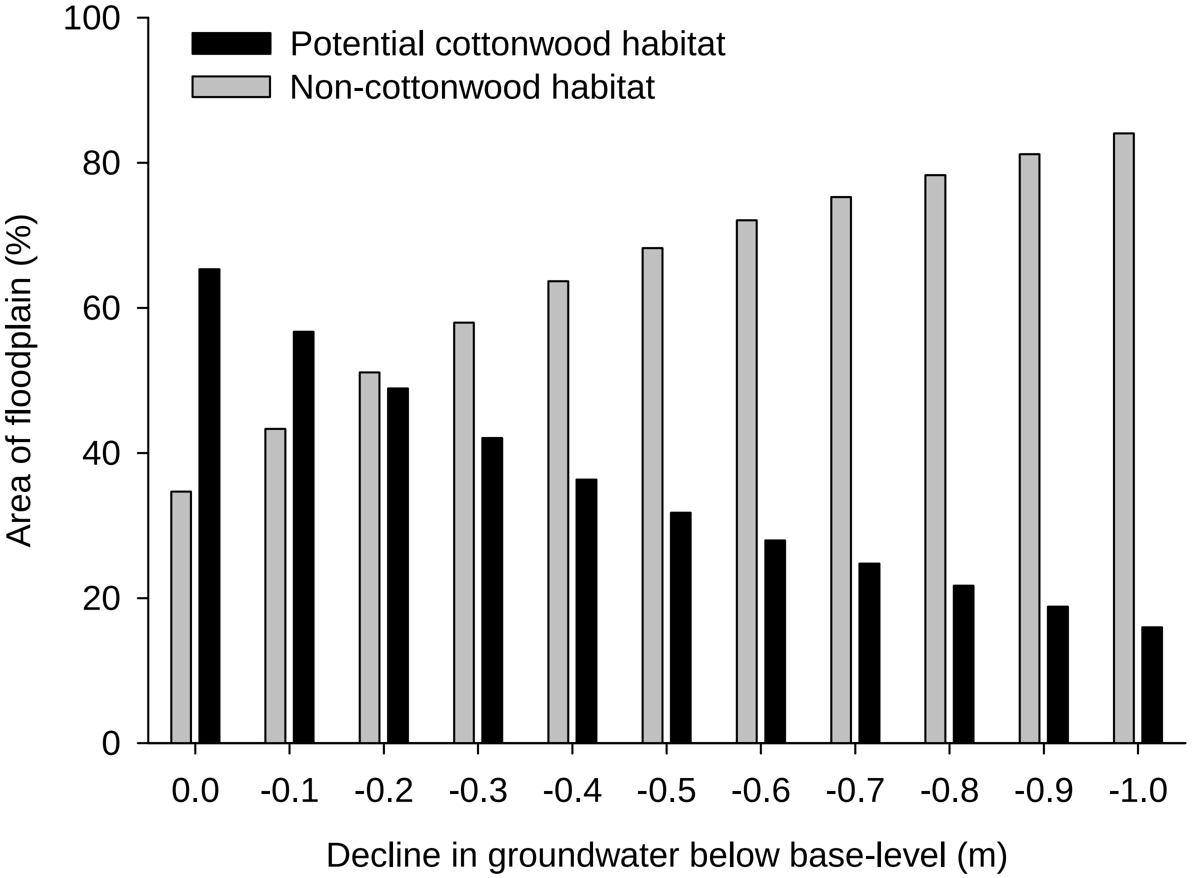

Figure 21. Percent of floodplain area that provides potential cottonwood habitat at a base groundwater condition (corresponding to a streamflow of 0.127cms (4.5 cfs)) and at a range of levels below this groundwater level. Potential cottonwood habitat falls to 7 and 3 percent at 1.5 and 2 m (4.9 and 6.6 ft) below base level, respectively.

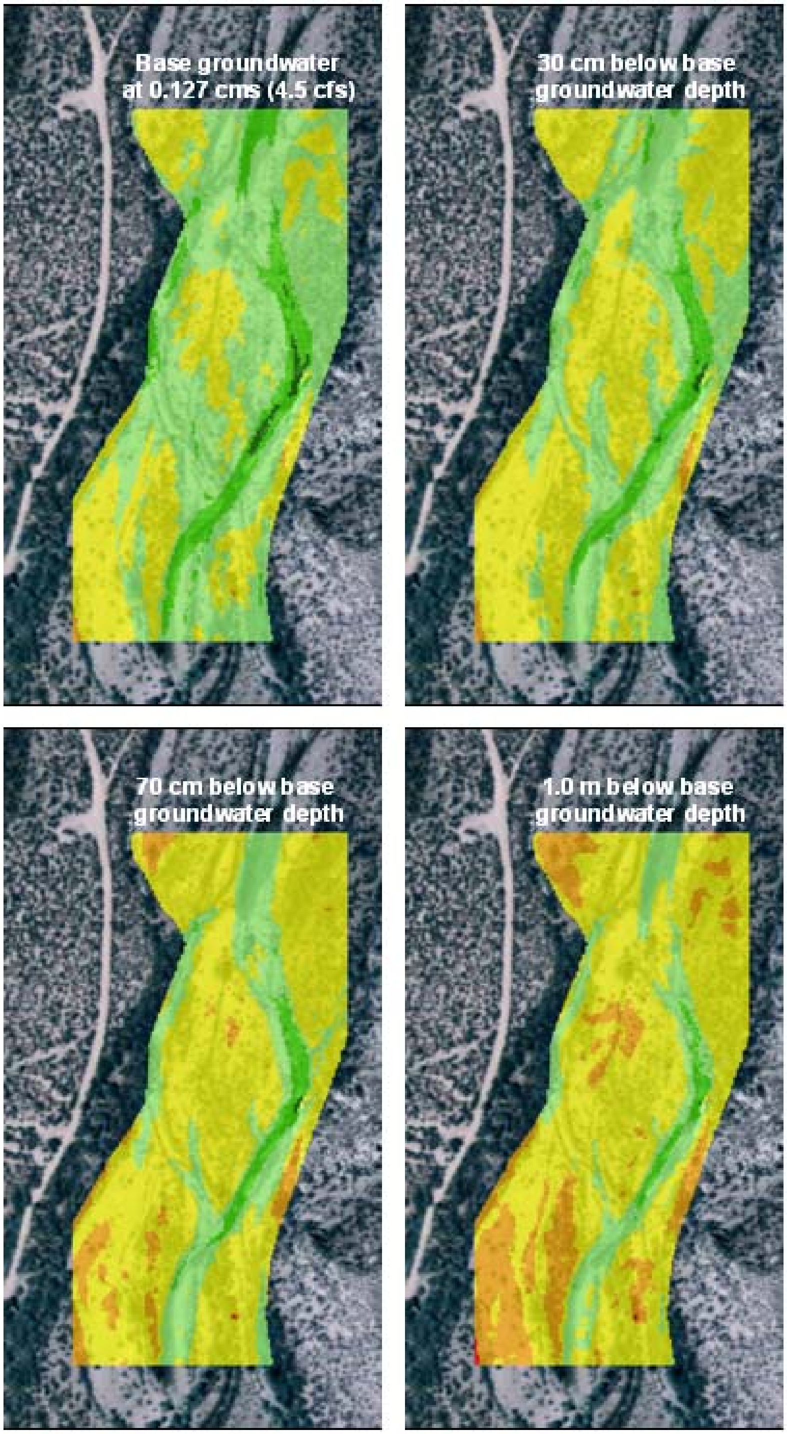

Figure 22. Potential cottonwood habitat (green) and unsuitable cottonwood habitat (yellow, orange and red) at a base groundwater level (at discharge in Cherry Creek of 0.127 cms (4.5 cfs)). Areas with depths to groundwater less than the average rooting depth of cottonwood (1.74 m) are considered potential cottonwood habitat. Rooting depth of cottonwood was calculated from six excavated trees along Cherry Creek. Area of potential habitat declines from 65 to 16 percent of the modeled floodplain area from base groundwater level to 1 m below base level.

Diverted reach

Streamflow in the main channel of Cherry Creek was reduced by 95 percent by the earthen diversion structure that was in place at the time of this study (streamflow was 0.122 cms (4.3 cfs) above the diversion; 0.006 cms (0.2 cfs) downstream from the diversion; Figure 4). The stream was discontinuous (intermittent) downstream from the diversion structure, but the channel was clearly visible. Average temperature during the plant water potential measurements was higher below the diversion (35.3^{o}C; 95.5^{o}F) compared to above (30.5^{o}C; 86.9^{o}F; t = 15.6; P < 0.0001) during the mid-day water potential measurements (Figure 23). Median relative humidity was higher downstream (26%) compared to upstream (24%) from the diversion (Mann-Whitney T = 15; P = 0.008). Mid-day xylem water potential was significantly lower downstream from the diversion for cottonwood (t = 4.7; P < 0.001), Goodding willow (t = 4.5; P = 0.002), and Arizona sycamore (t = 4.0; P = 0.017), but was not significantly different for saltcedar (t = 0.8; P = 0.46; Figure 23). Nighttime recovery occurred at a higher rate upstream from the diversion for cottonwood, Goodding willow, and Arizona sycamore but was no different relative to the diversion structure for saltcedar. Vapor pressure deficit was higher downstream from the diversion (2.17 kPa) compared to above (1.59 kPa; t = -19.9, P < 0.0001) during the mid-day water potential measurements. The threshold for stomatal closure and reduced photosynthetic rate of cottonwood is between 1.2. and 1.6 kPa (Horton *et al*. 2001a,b).

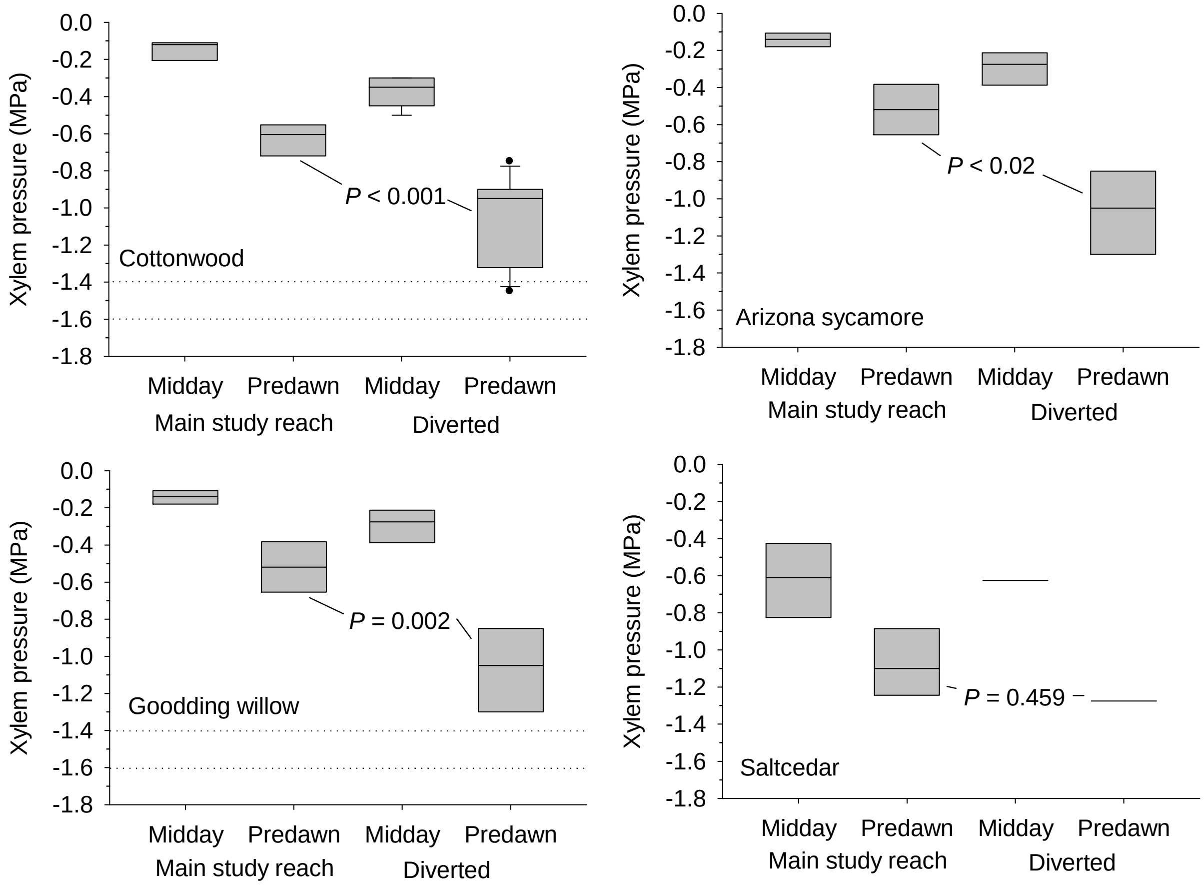

Figure 23. Pre-dawn (reference) and mid-day plant xylem water potential (Ψ) for cottonwood, Goodding willow, Arizona sycamore and saltcedar downstream from a water diversion on Cherry Creek. Stomatal closure and reduced photosynthetic rates of cottonwood and willow occur at -1.2 MPa and critical (cavitation) values for cottonwood and Goodding willow are between -1.4 and -1.6 MPa as indicated by dotted lines (Pockman *et al.* 1995).

Age-class Structure and Growth of Riparian Species along the Main Study Reach

Diameter-age relationships were strong and significant for cottonwood (Age= $3.44D_{cm}^{0.46}$; $r^2 = 0.69$; $P < 0.0001$), Goodding willow (Age= $3.12D_{cm}^{0.52}$; $r^2 = 0.77$; $P < 0.0001$), and saltcedar (Age= $0.016D_{cm}^{3.12}$; $r^2 = 0.71$; $P = 0.0006$). Cottonwood, Goodding willow, and saltcedar had skewed age-class distributions (skewed toward younger age-classes) along the main study reach (Figure 24). The oldest cottonwood of the 241 individuals measured was 32 years, with most individuals being two to three years of age (77 percent; skewness of distribution -2.3). The year 2005 was an exceptional recruitment year for cottonwood (peak flow in 2005 was 152.91 cms (5,400 cfs); 5 to 10 year recurrence interval flow). The median age of cottonwood was six years old (excluding seedlings from current year). The oldest Goodding willow (*n* = 92) measured was 27 years old and the median age (excluding seedlings from current year) was 10 years (skewness of the distribution -2.0). The majority (82 percent) of individuals were one year old. The oldest saltcedar (*n* = 40) measured was 47 years (skewness of the distribution -21.5). Most individuals were current and one-year old seedlings (58 percent). The median age of saltcedar was three years old (excluding current year seedlings).

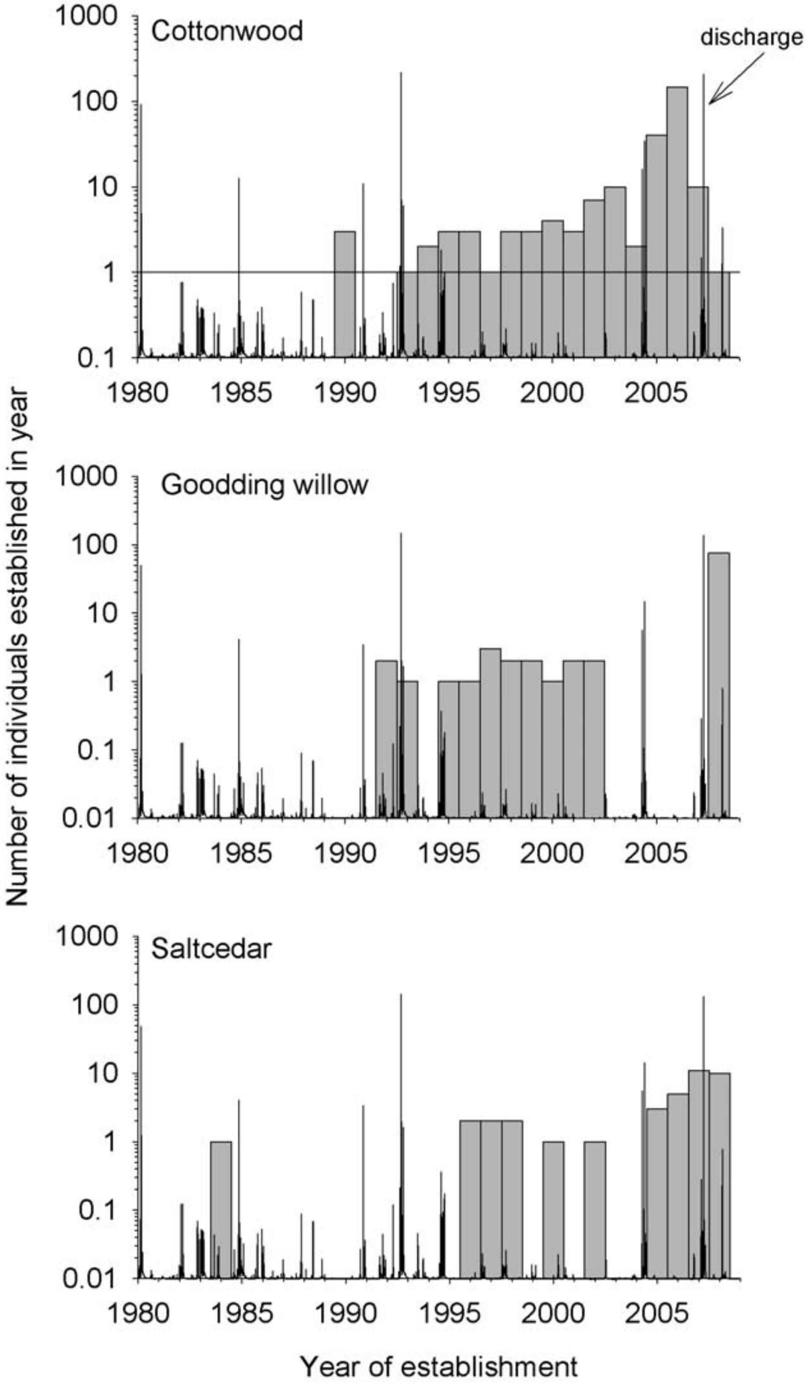

Figure 24. Distributions of establishment dates of cottonwood, Goodding willow, and saltcedar along the main study reach of Cherry Creek (grey bars; note log scale of count of number of individuals). A hydrograph of daily average flows is superimposed for reference (black line; no scale for discharge shown but the range is from 0.057 to 118.08 cms (2 to 4,170 cfs).

Cottonwood growth was best explained by May, June, and July average streamflow. Average monthly flow explained from 25 to 43 percent of the variability in cottonwood incremental growth (Figure 25) and between 25 and 44 percent of the incremental growth of Goodding willow. Incremental growth of cottonwood decreased an average of 1.5 mm per 0.142 cms (5 cfs) reduction in May flow, 3 mm reduction in growth per 0.142 cms (5 cfs) reduction in average June flow, and 3.5 mm per 0.142 cms (5 cfs) for average July flow. Goodding willow growth was also sensitive to May, June, and July flows, decreasing by 2, 3.5, and 4 mm per 0.142 cms (5 cfs) reduction in flow, respectively (Figure 26). Interannual May and June mean flows were highly correlated ($r = 0.96$; $P < 0.0001$) but July flow was not correlated with May or June ($P > 0.05$). Forty-eight percent of the variability in cottonwood growth was explained by two variables: mean June and July flows (Growth (cm) = $0.045+0.03Q_{Jun}+0.054Q_{Jul}$; $P = 0.02$). Similarly, 48 percent of the variability in Goodding willow growth was explained by mean June and July flows (Growth (cm) = $-0.059+0.03Q_{Jun}+0.066Q_{Jul}$; $P = 0.02$).

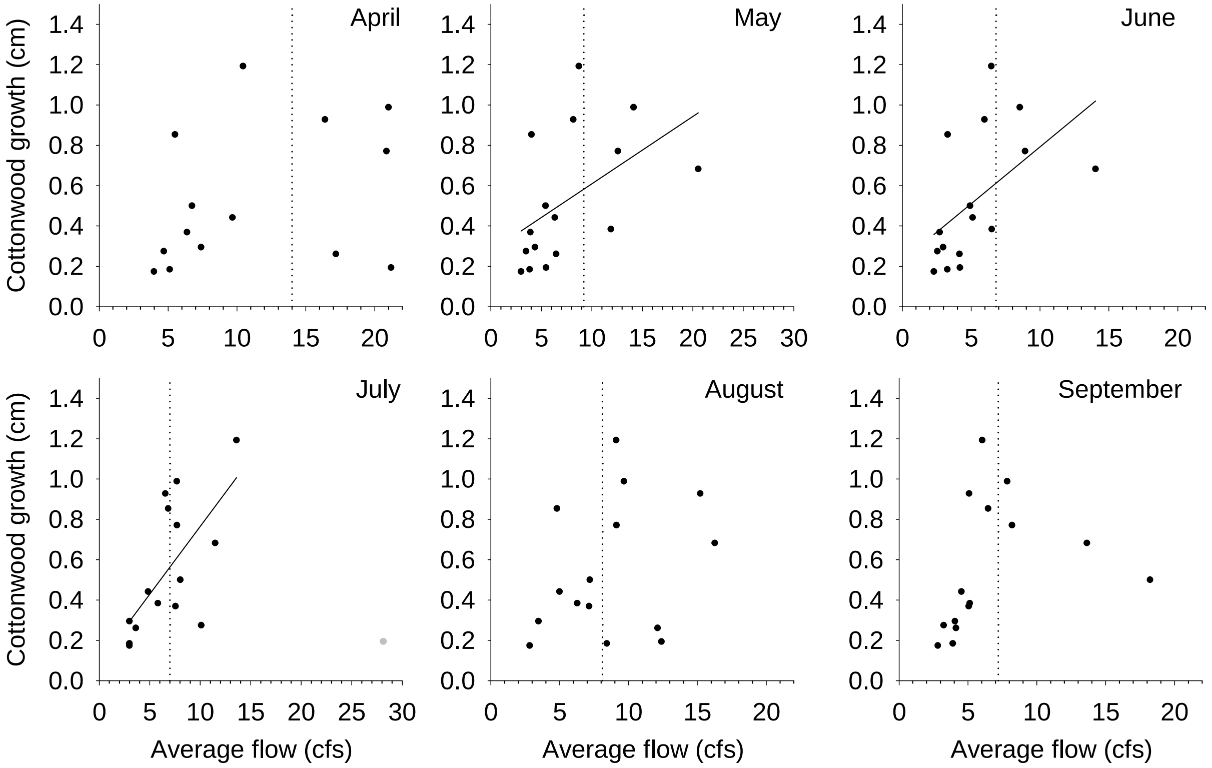

Figure 25. Relationships between average flow calculated for each month during the growing season and related to cottonwood growth for each year. Only significant relationships are shown. May: Growth (mm) = 0.28+0.033*Flow (cfs); r^2 = 0.25; *P* = 0.05. June: Growth (mm) = 0.23+0.056*Flow (cfs); r^2 = 0. 28; *P* = 0.03. July: Growth (mm) = 0.09+0.067*Flow (cfs); r^2 = 0.43; *P* = 0.008. Relationships between April, August, and September flow and incremental growth were not significant (*P* > 0.05). Outlier for July removed (shown in gray). Vertical dotted lines indicate the median instream flow for each month (period of record).

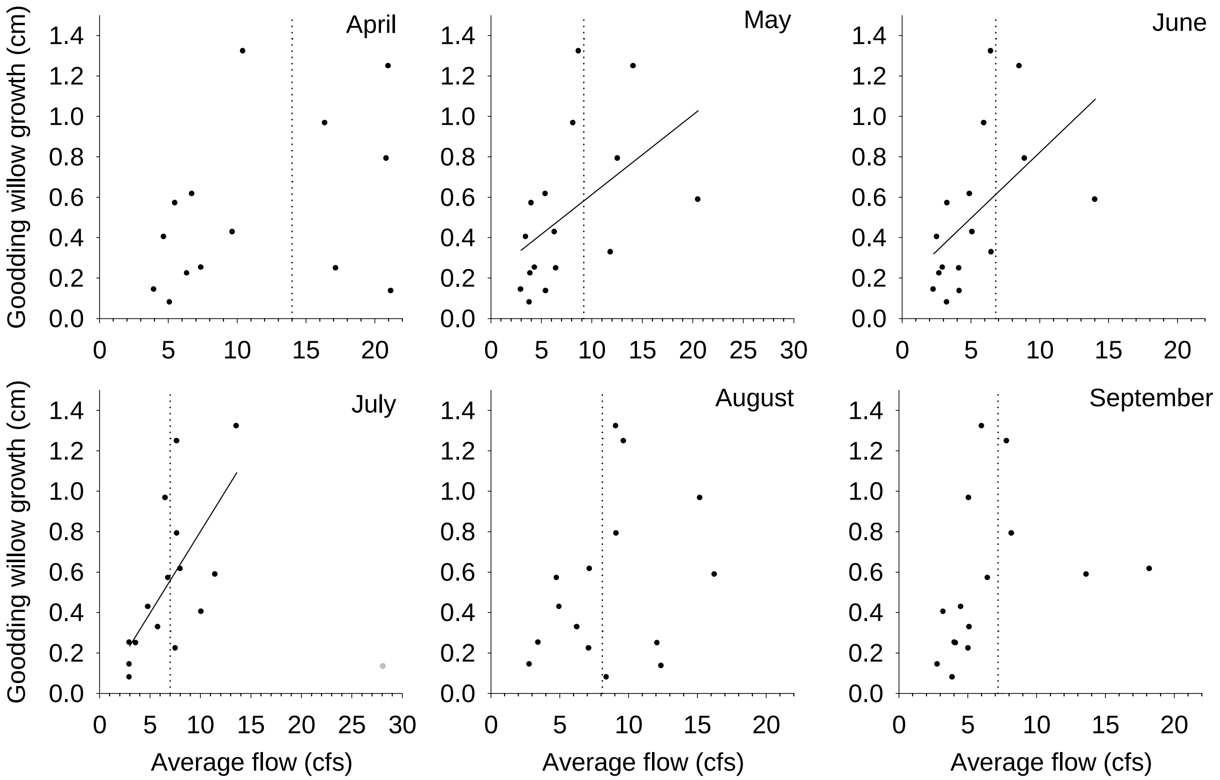

Figure 26. Relationships between average flow calculated for each month during the growing season and related to Goodding willow growth for each year. Only significant relationships are shown. May: Growth (mm) = 0.22+0.039*Flow (cfs); r^2 = 0.25; P = 0.05. June: Growth (mm) = 0.17+0.065*Flow (cfs); r^2 = 0. 27; P = 0.04. July: Growth (mm) = -0.008+0.08*Flow (cfs); r^2 = 0.44; P = 0.007. September: Growth (mm) = 0.09+0.067*Flow (cfs); r^2 = 0.43; P = 0.008. Relationships between April, August, and September flow and incremental growth were not significant ($P > 0.05$). Outlier for July removed (shown in gray). Vertical dotted lines indicate the median instream flow for each month (period of record).

Perennial to Intermittent Flow Reach

Along a perennial to intermittent flow gradient, species composition shifted from that dominated by native riparian forest species to xeroriparian and desert shrub communities (Figure 2). Turnover in species composition was high, species at either end of the intermittent to perennial gradient being completely different (Detrended Correspondence Analysis axis 1 spans 3.98 standard deviation units; for reference 4 sd units is considered complete species turnover – no shared species between sites (Jongman *et al*. 1995)). Along the perennial reach seven vegetation cover types were represented: Mesquite/desert broom/burrobrush/catclaw acacia (6 percent), Desert broom/burrobrush/mesquite (9 percent), Burrobrush (13 percent), Burrobrush/desert broom (11 percent), Cottonwood/willow/ seepwillow/burrobrush (4 percent), Goodding willow/seepwillow (30 percent), and Cottonwood/Goodding willow/seepwillow/Arizona alder (28 percent). Along the intermittent reach six community types were represented: Mesquite/desert broom/burrobrush/catclaw acacia (13 percent), Desert broom/burrobrush/mesquite (4 percent), Burrobrush (18 percent), Burrobrush/saltcedar (4 percent), Burrobrush/desert broom (52 percent), and Goodding willow/seepwillow (9 percent). Riparian forest-dominated plots were the most frequently occurring along the perennial reach, (62 percent of plots), whereas desert and non-native shrub-dominated cover types dominated the intermittent reach (90 percent). Total cover of native riparian tree species (cottonwood, Goodding willow, and Arizona sycamore) was significantly higher in the perennial (40 percent) compared to the intermittent reach (1.9 percent; Mann Whitney *U* test; z = -5.15; P < 0.0001).

Woody and tree species richness, woody and tree cover, and tree richness were all higher in the perennial compared to the intermittent reach (*t*-test or Mann-Whitney *U* test; P < 0.05; Figure 27). Herbaceous richness and cover were both higher in the perennial reach but not significantly so (Mann-Whitney *U* test; P > 0.05).

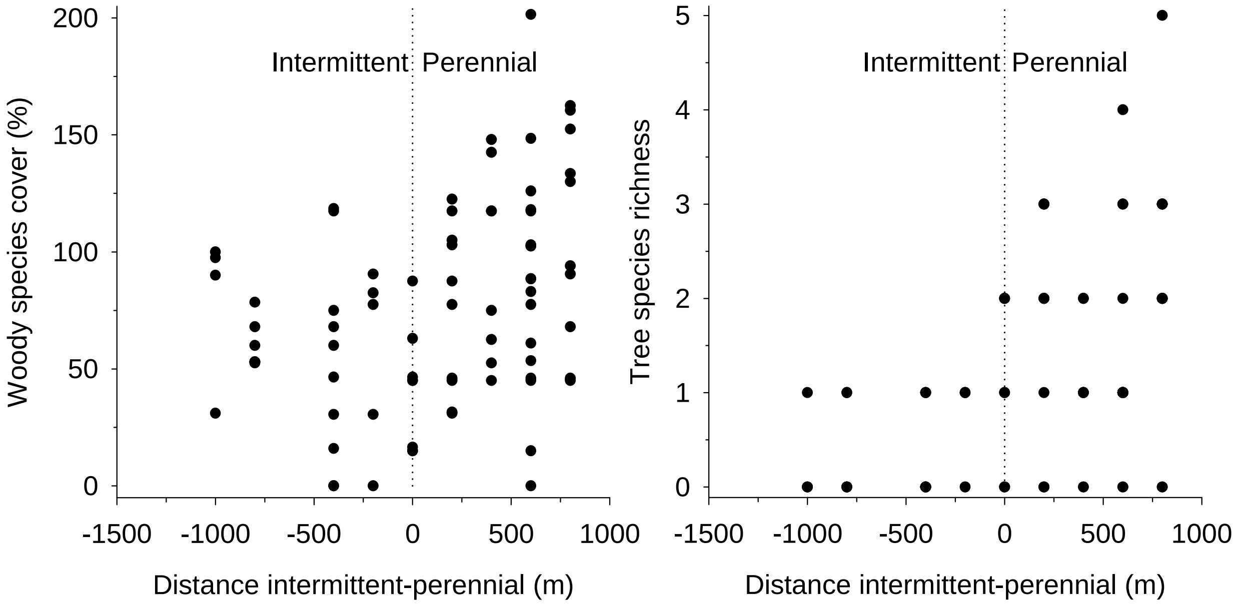

Figure 27. Woody species cover and tree species richness along a gradient from intermittent to perennial streamflow along Cherry Creek. Vertical dotted line indicates the transition from intermittent to perennial flow; distance is stream distance upstream and downstream from this transition.

Groundwater – Surface water Interactions

Discharge measurements taken along the study reach were associated with those measured at the USGS streamflow gage 15 km upstream ($Q_{USGS} = 1.47Q_{study}^{0.84}$; r^2 =0.91, $P <$ 0.0001). The relationship was linear for flows from 0.057 to 0.425 cms (2 to 15 cfs). Flow measurements tended to be 12 to 19 percent lower through the study reach compared to those measured at the USGS gage at flows exceeding 0.425 cms (15 cfs). Discrepancies are certainly due to the presence of the intermittent reach which lies between the USGS gage and the main study reach. Because the channel through this reach is dry except during higher discharges, it serves as a hydrologic buffer, delaying and reducing the peak volume of flow pulses reaching the study reach. Rating curves were constructed relating discharge to stage measured at the staff gage.

Groundwater level was measured at 15 minute intervals from November, 2008 through October, 2009. During this period, the site experienced a frontal rain storm resulting in three major peaks in streamflow. Over the course of the storm, discharge measured at the USGS streamflow gage ranged from a low of 0.17 cms (6 cfs) to a high of 106 cms (3,750 cfs), encompassing the range of instream flow values 0.20 to 0.88 cms (7 to 31 cfs) that were filed for by the Forest Service (Appendix A). The three flood peaks occurred in succession over a nine day period, beginning with 13.62cms (481 cfs) on December 18, 2009, 12.18cms (430 cfs) on December 23, and 106 cms (3,750 cfs) on December 26. These flood pulses provided an opportunity to examine groundwater recharge, stage, and discharge associated with high flow events, and to use the data collected to develop a better understanding of the linkage between streamflow and groundwater levels.

Stream stage and groundwater levels were highly correlated across the floodplain. The weakest correlations were between the staff gage and the upstream well on river left (GW 1) and between the staff gage and the wells on far river right (GW 4 and 8). The weaker correlations are due to the distances between the staff gage and the wells and lags in recharge and discharge in response to changes in streamflow. Distances from the staff gage to well 1, 4, and 8 were 72 m, 97 m, and 70 m, respectively. Although wells near the channel were tightly coupled with streamflow-stage fluctuations, wells further from the channel exhibited suppressed and lagged fluctuations. For example, there was a 22 hr. lag between peak flow in the stream (measured at the staff gage February 24, 2009) and peak groundwater levels in the groundwater well 8 which was 70 m from the stream.

These correlations provide a quantification of the connections between streamflow/ stage and groundwater levels. Further, groundwater profiles show that the groundwater movement through the valley along the main study reach is from the northeast toward the southwest. Groundwater elevations were higher than the staff gage along the left side of the valley, and lower than the channel along the right side of the valley. This gaining or groundwater inflow appears to be related not to some extra-floodplain groundwater source, but rather to a side channel located on the left side of the valley beyond the well transect (Figure 1). Average daily depth to groundwater along the left (east) side of the valley were significantly related to daily average discharge measured at the USGS gage several km upstream ($r^2 = 0.67$; $P < 0.0001$). This indicates that variability in flow at the USGS gage explains 67 percent of the variability of the flow in the groundwater well on river left in the study reach. Because the groundwater level in the well to the far left fluctuates as a function of streamflow but remains at a higher elevation than the water surface in the channel, it is clear that the side channel is likely hydrologically connected upstream from the transect, fluctuates with changes in streamflow, and supplies the main channel with water through subsurface flow. Groundwater profiles on river right (west of main channel) indicate that the main channel is losing to floodplain alluvium across a range of flows (0.28 to 13.6 cms (10 to 481 cfs)). Surface and groundwater measurements indicate that flows in Cherry Creek are the principle source of water supporting groundwater levels across the floodplain. Incremental decreases in streamflow (and stream stage) result in similar but lagged reductions in groundwater levels.

Discussion

Structurally and compositionally heterogeneous vegetation is an important component of wildlife habitat by providing a variety of food resources and shelter (Arnold 1988, Mac Nally 1990, McElhinny *et al*. 2005). Through their provision of resources, structurally and compositionally heterogeneous stands of vegetation support diverse species assemblages (Woinarski *et al*. 1997, McElhinny *et al.* 2005). Riparian habitats, with their exceptional structural and compositional diversity, provide a variety of resources not found in adjacent upland habitats (Powell and Steidl 2000, 2002, Palmer and Bennett 2006, Kirkpatrick *et al.* 2009). This difference in structure is one important reason why bird and small mammal species composition differs between upland and riparian habitats (Finch 1989, Doyle 1990, Ellis *et al*. 1997). Kirkpatrick *et al.* (2009) found that bird numbers and richness of bird communities increased 75 and 68 percent, respectively, in riparian areas compared to areas 200 m from stream channels along southwestern Arizona streams.

Maintenance of riparian forest requires perennial streamflow. Reduction in streamflow and streamflow intermittency may result in declines in forest structure, loss of mature cottonwood stands, and conversion of forest to grassland-shrubland system (Boggs and Weaver 1994, Stromberg *et al*. 2005, Stromberg *et al.* 2007a, Merritt and Poff 2010). Further, in riparian areas, the presence of perennial standing water is associated with higher bird species richness and total bird numbers (Brand *et al.* 2008, Kirkpatrick *et al.* 2009). Managers interested in conserving biodiversity in riparian areas must consider efforts to maintain and/or restore riparian plant communities (i.e., those with a cottonwood, Goodding willow, and Arizona sycamore component) and promote a dense and diverse vegetation understory (Powell and Steidl 2002) and maintain surface water.

Our findings suggest that riparian forests along Cherry Creek are dependent upon alluvial groundwater systems that are supported most directly by streamflow. Maintenance of sufficient streamflow to support shallow water tables reduces the likelihood of conversion of cottonwood and Arizona sycamore-dominated forests to shrub communities and those more typical of upland and xeroriparian habitats. Since groundwater levels are a function of the balance between surface and groundwater inflows and outflows into and from the valley, and since streamflow is the principle source of water entering the valley, maintaining the structure and quality of wildlife habitat requires the maintenance of adequate streamflow timing and volume.

Bird Habitat

The relationship between birds and habitat has been well-evaluated. For example, habitat structural complexity has long been known to positively influence bird species richness and abundance in general (MacArthur and MacArthur 1961, Willson 1974, Roth 1976, Cody 1981) and in riparian areas in particular (Emmerich and Vohs 1982, Finch 1989, Knopf and Samson 1994, Sanders and Edge 1998, Powell and Steidl 2002). Riparian habitats have more horizontal structure and vertical layering of habitat and are comprised of plant species not generally found in adjacent upland areas (Palmer and Bennett 2006). Within the American Southwest, more than 50% of breeding birds are dependent upon riparian habitats at some point in their annual cycle (Knopf *et al.* 1988, Kirkpatrick *et al.* 2009).

Bird habitat diversity along Cherry Creek was greatest where deciduous riparian tree species occurred. Bird habitat index showed a significant positive relationship with habitat containing native plant cover, specifically frequencies of native plants as cottonwood, Goodding willow, and Arizona sycamore. Our bird habitat results were similar to those from Powell and Steidl (2002) who quantified nesting habitat of songbirds in Arizona. Seven species of songbirds nested in habitat containing specific riparian plant species (e.g., Arizona sycamore and hackberry) as well as choosing sites with a dense volume of vegetation. Cottonwood and other large-diameter trees are important resources for woodpecker and sapsucker species (Piciformes) to excavate cavities (Brenowitz 1978, Sedgwick 1997). Woodpeckers are primary cavity-nesting birds that play important roles in forest ecosystems by excavating cavities that, once vacated, provide nesting and roosting sites for other birds and mammals and are therefore considered keystone species (Jones *et al*. 1994, Martin and Eadie 1999). According to Powell and Steidl (2002), maintaining habitats containing native riparian tree and shrub species is important to meeting the habitat requirements of songbirds inhabiting arid regions of the Southwest.

Habitat indices along Cherry Creek were lowest in the diverted and in xeroriparian areas adjacent to the main study reach. These habitats are comprised of upland shrubs (e.g., burrobrush) and non-native shrubs (e.g., saltcedar). These xeroriparian habitats were more abundant further from the stream channel and cover of xeroriparian dominants was not associated with high quality bird habitat. Habitat indices were highest in the main reach and associated with native riparian vegetation communities. Among the vegetation communities, cottonwood and Arizona sycamore are especially important for maintaining keystone species like woodpeckers. Reducing the amount of large diameter nesting trees could reduce primary-cavity excavating species from Cherry Creek and, in turn, impact several species of secondary-cavity nesting birds (e.g., species of flycatchers, swallows, and titmouse, and Lucy's warbler) and mammals (e.g., bats).

Small Mammal and Reptile Habitat

Compared to our knowledge of avifauna, the importance of riparian habitat structure is less understood for mammal and reptile communities (Naiman *et al*. 2005), but likely provides resources for these wildlife during part of their life cycle (Oakely *et al*. 1985). Herpetofauna respond to structural changes to their habitat (Pianka 1967); therefore, their presence and abundance can be good indicators of habitat changes in riparian areas (Bateman *et al*. 2008a). Small mammal species in the arid and semi-arid Southwest are often more numerous in riparian habitats than in adjacent uplands (Stamp and Ohmart 1979, Doyle 1990, Falck *et al*. 2003). However, it is likely but poorly understood whether populations in riparian areas serve as source populations for uplands (Ellison and van Riper 1998). The diverse ground cover in riparian habitats provides rich foraging areas for seed-feeding and insect-feeding small mammals (Oakley *et al*. 1985). Food items found in riparian habitats are also important for beaver (*Castor canadensis*) which prefer to feed on cottonwood and willow (Kimball and Perry 2008).

Riparian areas can provide shelter and corridors for movement of mammal and reptile species. For example, woody debris can provide denning sites for muskrat (*Ondatra zibethicus*), raccoon (*Procyon lotor*), and snakes (Colubrids and Vipers, Oakley *et al*. 1985). Medium size mammalian predators use the high edge-to-area ratio of riparian areas as corridors for commuting among habitat patches (Oakely *et al*. 1985). Woodrats (*Neotoma albigula*), which are common in a variety of habitats including riparian areas, construct large nests in areas with greater tree cover and tree density (Ellison and van Riper 1998). These nests can provide microhabitats used by other small mammal and reptile species (e.g., Colubrid snakes and diurnal active lizards).

Small mammal and reptile habitat index was greatest along Cherry Creek in areas with high foliage cover and variation of cover, abundance of cottonwoods, and where variable sandy substrate occurred. Mammal and reptile habitat index was greatest in the main reach and lowest in the xeroriparian where the vegetation was dominated by non-cottonwood communities. Other studies within Arizona indicate that small mammal distributions are strongly correlated with riparian areas having grass and herbaceous ground cover (Anderson and Nelson 1999) and a high frequency of trees (desert willow *Chilopsis linearis*, ash *Fraxinus pennsylvanica*, and Arizona sycamore) and tree cover (Ellison and van Riper 1998). Riparian areas having a mosaic of canopy cover can provide thermoregulatory benefits for mammals and reptiles (Oakely *et al*. 1985, Bateman *et al*. 2008a). This variation in foliage cover creates areas shaded from sunlight which is beneficial to small mammals and reptiles.

Maintaining cottonwood-dominated habitat will be important to provide food and shelter resources for mammals and reptiles along Cherry Creek. One study in New Mexico captured more species of small mammals in monotypic stands of saltcedar compared to native cottonwood; however, this increase in species richness was likely caused by the proximity of saltcedar stands to source populations because five of the species were typical of upland grassland habitats instead of riparian habitats (Ellis *et al*. 1997). Therefore, if the vegetation along Cherry Creek were to shift from a cottonwood-willow dominated riparian forest habitat to a saltcedar-burrobrush shrubland habitat it would likely support mostly upland species found in adjacent xeroriparian areas instead of both riparian and upland species.

Groundwater, Streamflow, and Wildlife Habitat Maintenance

As mentioned, the most structurally and compositionally complex bird and mammal habitats contain a riparian forest component. Many of the riparian species governing habitat complexity are considered obligate riparian due to their reliance upon shallow water tables (*sensu* J.C. Stomberg in Haney *et al.* 2008). Shrub-dominated vegetation cover types (which were most abundant in intermittent, diverted and xeroriparian sites) had lower habitat indices and were similar to upland habitats. Maintenance of riparian forest species that support this complex habitat requires maintenance of perennial flows of sufficient magnitude to maintain elevated groundwater tables during the growing season (Figure 27). Further, regeneration and maintenance of riparian forest species are dependent upon high flow events, which create sites for recruitment by disturbance-adapted species (e.g., cottonwood, Goodding willow, sandbar willow), hydrologic conditions that exclude upland species from riparian areas, serve to recharge and maintain alluvial groundwater levels, and support biochemical processes on the floodplain (Figure 24). Low flows of sufficient magnitude enhance the survival likelihood of disturbance-adapted regenerants and support root development, growth, and survival to reproductive age (Mahoney and Rood 1998, Horton and Clark. 2001).

Riparian tree species were most abundant in areas along Cherry Creek where groundwater was within 2 m of the ground surface. Incremental declines in surface and groundwater levels were predicted to cause frequency of cottonwood-dominated riparian forest habitat to decline from its current 41 percent frequency at an average rate of 3 percent per decimeter decline in groundwater level from the modeled base flow (Figure 19). Incremental reductions in surface water and groundwater levels increase the likelihood of conversion of riparian forest to shrubland more typical of xeroriparian areas, the migration of the riparian forest-xeroriparian edge nearer the stream channel, and the reduction in the extent or complete loss of riparian forest, depending on the severity and persistence of groundwater decline.

Multiple lines of evidence suggest that, because the source of alluvial groundwater is from surface flow in Cherry Creek along the main study reach, incremental reductions in streamflow will have negative consequences for riparian forest species and associated wildlife habitats. Our data and the literature suggest that these consequences include: 1.) elevated water stress and decreased growth rates (fitness) of riparian tree species, 2.) reduction in spatial extent of potential habitat for dominant forest species (e.g., cottonwood), and 3.) likely conversion of complex riparian forest cover types to structurally and compositionally simple shrubland and bare cover types (Busch *et al*. 1992, Pockman and Sperry 2000; Figures 19 & 21). The risk of such a conversion of riparian wildlife habitat to that similar to terrestrial habitat increases as a function of streamflow reduction and associated groundwater decline (Figures 19 and 21).

Water Stress and Growth of Riparian Forest Species

Because the dominant species in the riparian forest cover types along Cherry Creek (cottonwood and Goodding willow) are obligate phreatophytes, maintenance of suitable soil moisture through maintenance of water tables within reach of the rooting zone is the governing factor supporting their health and persistence (Busch *et al.* 1992, J.C. Stromberg in Haney *et al.* 2008). Over short periods of time, water stress can cause wilting of leaves, reduced photosynthesis, leaf death, reduced leaf area, cavitation, and branch sacrifice (Tyree *et al.* 1994, Pockman and Sperry 2000). Chronic water stress may make riparian trees more susceptible to disease and may cause permanent loss of canopy volume, reduced growth rates, and may result in tree death if severe and prolonged (Kozlowski and Pallardy 2002). Reductions in streamflow from low (0.122 cms (4.3 cfs); 90 percent of flows equal or exceed) to extremely low (0.006 cms (0.2 cfs); >99.9 percent exceedance probability) due to the flow diversion along the main study reach was related to reduced cover of riparian forest species along Cherry Creek and resulted in elevated water stress for extant individuals (Appendix C; Figure 23). Though mid-day cottonwood water stress never fell below -0.83 MPa upstream from the diversion (mean ψ = -0.64 MPa), several individuals downstream from the diversion approached the critical ψ threshold for xylem cavitation and branch loss (e.g., three individuals had ψ of -1.39, 1.40 and 1.45 MPa indicating severe water stress). Shoot water potentials beyond which xylem cavitation will occur is -1.50 to -1.60 MPa for cottonwood and -1.40 MPa for Goodding willow (Pockman *et al*. 1995). Along Cherry Creek, Goodding willow mid-day ψ averaged -0.52 MPa upstream and -1.07 MPa downstream from the diversion. Arizona sycamore mid-day ψ

averaged -0.80 MPa upstream from the diversion and -1.18 MPa downstream from the diversion, indicating internal water stress caused by response to depleted streamflow.

It is likely that some cottonwood trees in areas downstream from the diversion have already compensated for water shortages through branch sacrifice and reduction in canopy volume, thereby reducing water stress to remaining branches and preventing tree death (Tyree *et al.* 1994, Rood *et al.* 2000). Average cottonwood tree height downstream from the diversion was 8.4 m compared to 9.9 m above, though this difference was not statistically significant (t = 0.9; P = 0.4). In addition the lower cover, lower stem density, and smaller basal area of stands downstream from the diversion may relieve water stress for remaining individuals in stands. Branch sacrifice is an individual response to cope with water stress; thinning through mortality of individuals is a stand-level response to inadequate water supplies.

In contrast to native riparian tree species, xeric (facultative) phreatophytes and upland shrubs can tolerate water potentials ranging from –4.0 to –9.0 MPa, suggesting variable degrees of water stress tolerance in these species (Pockman and Sperry 2000, Pataki *et al.* 2008). For example, xylem cavitation threshold for saltcedar is -7.0 MPa (Pockman and Sperry 2000). Saltcedar never approached this threshold along either upstream (average ψ = -1.07 MPa; range -0.84 to -1.39 MPa) or downstream (average ψ = 1.28 MPa; range -0.90 to -1.65 MPa) from the diversion along Cherry Creek. Saltcedar is 3 to 4 times more resistant to cavitation than native riparian trees, but is also tolerant of flooding unlike many desert species (Pockman and Sperry 2000). Its greater resistance to cavitation and ability to tolerate flooding give saltcedar a competitive advantage, allowing it to better tolerate periods of drought and water depletion associated with water extraction.

Whereas many native desert shrubs are not abundant in riparian areas because of trade-offs between resistance to cavitation and xylem vessel hydraulic conductivity, velvet mesquite, acacia and desert broom are exceptions and can grow well and compete with native riparian species and persist in drought conditions (Pockman and Sperry 2000). Desert upland shrubs have three to six times lower critical ψ than native riparian tree species (cottonwood and Goodding willow; Pockman and Sperry 2000). Some desert shrubs have been shown to tolerate low soil water conditions and survive leaf ψ to -12 MPa because they have cavitation-resistant xylem (Pockman and Sperry 2000). Desert broom is particularly adept at both surviving during low water availability and surviving floods that exclude many other upland species, making it a likely species to become more abundant in riparian areas in the event of water depletion. Indeed, along the two perennial flow reaches sampled along Cherry Creek, desert broom cover averaged 5 percent or less, whereas along the intermittent, upland, and diverted reaches had significantly higher cover ranging from 12 to 17 percent. Therefore, elevated water stress caused by lower streamflows over time would likely result in conversion of riparian woodland dominated by cottonwoods, Goodding willow, and Arizona sycamore to more drought tolerant species such as saltcedar and desert broom which offer less habitat structure and complexity.

The higher temperature and vapor pressure deficit downstream from the diversion along Cherry Creek was likely due to sparser vegetation cover associated with the diverted reach and lower water surface area of Cherry Creek due to depleted flow. Downstream from the diversion, riparian tree cover in the vegetation plots averaged 4.5 percent compared to 23 percent upstream (*t*-test; $t = -21$; $P = 0.04$). Vapor pressure deficit was significantly higher along the reach downstream from the diversion (2.2 kPa) compared to above (1.6 kPa). Atmospheric water stress caused by high leaf-to-air vapor pressure deficit have been shown to cause stomatal closure and reduced photosynthesis rate at high VPD (threshold 1.2 to 1.6 kPa) for cottonwood and Goodding willow (Horton *et al.* 2001a), leading to decreases in growth rate. The effects of the diversion on the vegetation immediately downstream from it would likely be more severe if not for leakage from the bed of the ditch and contribution to local groundwater levels. This leakage and subsurface flow from upstream reaches are supported by water delivered from upstream to the study reach.

Depth to Groundwater

Previous studies addressing ecophysiological responses of cottonwood to hydrologic modifications documented reduced stomatal conductance and xylem pressure with increased depth to groundwater (Busch and Smith 1995; Horton *et al*. 2001a,c). As a consequence of high vulnerability to xylem cavitation, riparian trees have higher incidence of branch or crown mortality when roots lose contact with groundwater (Tyree *et al*. 1994, Scott *et al.* 2000, Shafroth *et al.* 2000, Horton *et al.* 2001a).

In Arizona, cottonwood depend on shallow floodplain groundwater recharged by the adjacent stream, which makes them particularly vulnerable to modifications of river flow (Smith *et al.* 1991; Busch *et al.* 1992; Kolb *et al.* 1997). Goodding willow is generally known as an obligate phreatophyte that is relatively shallow rooted (0.1 to 3.2 m; Stromberg 1993, Stromberg *et al.* 1996; mean of 1.5 in current study). Saltcedar is a deep rooted, facultative phreatophyte that obtains water from unsaturated soil and physiologically adapted to a higher degree of water stress than cottonwood and Goodding willow (Busch *et al.* 1992, Busch and Smith 1995, Cleverly *et al.* 1997, Horton *et al.* 2001a,b). Abrupt, sustained water table declines greater than 1.5 m in alluvial sands and coarse, porous substrate like that along Cherry Creek have produced cottonwood mortality in excess of 80 percent (Scott *et al.* 1999, Shafroth *et al.* 2000), however trees may tolerate short-term groundwater declines. Mature cottonwood trees have been killed by abrupt permanent drops in water table of 1 m with lesser declines of 0.5 m reducing stem growth (Scott *et al.* 1999, 2000). Water stress is most pronounced in juveniles of a species (Smith *et al.* 1991). Seasonal declines of 1 m may cause mortality of saplings of cottonwood and willow (Shafroth *et al.* 2000). Reduced streamflow and subsequent groundwater decline has more of an impact on native species than on saltcedar, favoring saltcedar (Cleverly *et al.* 1997, Horton *et al.* 2001c, Pockman and Sperry 2000).

Streamflow Intermittency

Cottonwood that grow along dry river beds may persist, yet forests in such situations tend to be sparse and of low age-class diversity and thus may not provide desired wildlife benefits (Stromberg *et al.* 1996). The sparse older cottonwood on the valley margins along the study reach of Cherry Creek provide examples of such deeper-rooted, isolated cottonwood that have persisted far from the channel. Dense cottonwood/Goodding willow forests appear to be restricted to perennial rivers, or along intermittent rivers in areas with shallow water tables (Lite and Stromberg 2005; Stromberg *et al.* 2007a).

Saltcedar is known to be more drought tolerant than native riparian species in part due to its deeper potential rooting depth (Stromberg *et al*. 1996, Busch and Smith 1995, Horton *et* *al.* 2001a,c) and ability to function and maintain dominance under both water stressed and flooded conditions (Cleverly *et al.* 1997, Horton *et al.* 2001a,b,c). Lite and Stromberg (2005) found that cottonwood and Goodding willow were dominant over saltcedar at sites where flows were perennial at least 76 percent of the time, interannual groundwater fluctuation was less than 0.5 m and average depth to groundwater was less than 2.6 m. Sites with deeper water tables along dewatered rivers had greater aerial coverage of shrublands and less of woodlands. When flow permanence fell below 42 percent, saltcedar became the dominant plant. Our work associates increases in shrub cover with lower wildlife habitat quality. Cottonwood and Goodding willow had higher cover and basal area and were dominant along perennial compared to paired intermittent reaches in a study of ten rivers in Arizona (Stromberg *et al.* 2007a) and the opposite was true of saltcedar. Shifts in dominance from native forest to exotic-dominated shrubland were associated with flow regulation and streamflow intermittency. Even in the presence of saltcedar, rivers with flow sufficient to support cottonwood and Goodding willow maintained native forest-dominated vegetation. This suggests that reductions in flow that approach and result in intermittent conditions along Cherry Creek increase the likelihood of a shift in riparian cover over time from cottonwood and Goodding willow-dominated to saltcedar and desert shrub-dominated habitat.

Findings from the studies cited above and our work along Cherry Creek suggest that maintenance of dominance by native forest species and prevention of the establishment of shrub-dominated cover types requires maintenance of perennial flows. Hydraulic models of the main study reach indicated that the transition between perennial and intermittent occurs between 0.113 to 0.227 cms (4 and 8 cfs; Waddle and Bovee 2010). Saltcedar is not particularly competitive against cottonwood and Goodding willow under well-watered conditions (Sher *et al.* 2000). Furthermore, saltcedar does poorly under cottonwood canopy (Lesica and Miles 2001) and other native trees (e.g., *Acer negundo*; Dewine and Cooper 2008), so maintenance of flows that support native forest species is key to preventing non-native shrub dominance in riparian areas. Because saltcedar does well in areas with deeper water tables and drier soils as well as in areas near the stream that flood, the more water extracted or otherwise reduced in the stream, the higher likelihood of shifts in riparian vegetation structure from native riparian forest-dominated stands to saltcedar and desert shrub-dominated shrubland cover types (Horton *et al.* 2001 a,c) and as a consequence, poorer wildlife habitat.

Seasonal and Interannual variability in flow

Seasonal and interannual variability in flow is important for maintaining heterogeneous riparian habitat. Fluctuations between high and low flows, variation in the timing and rate of change in flows, in conjunction with some seasonal predictability in the timing of biologically important flow events, foster heterogeneity and enhances the quality, productivity, fitness, age-and size-class structure of vegetation and structure of habitat. Variation in flow regime through time (timing, magnitude, frequency and sequencing of high and low flow) provides opportunities for different species to become established at different periods of time and for multiple age classes of individual species to be present at any point in time.

Maintaining static flows from year to year would lead to simplification in both habitat structure and quality through benefiting species better suited to those conditions and excluding those that might require different combinations of conditions. The historic flows in Cherry

Creek exhibit high flow variability (Figure 24, Appendix C), and the mosaic of forest, shrubland, bare ground, and patches of herbaceous vegetation present along Cherry Creek are a biological expression of this variability.

Low flows

Integration of hydraulic models and rating curves from field measurements of streamflow indicate that the main study reach of Cherry Creek becomes intermittent at flows between 0.113 and 0.227 cms (4 and 8 cfs; Waddel and Bovee 2009). Maintenance of flows at this level during the growing season would provide only for maintenance of riparian tree species adjacent to the channel and would likely result in reduction in the extent of riparian forest across the floodplain (in the absence of occasional flooding). Maintenance of low flows approaching the perennial-intermittent threshold for Cherry Creek would increase the likelihood of non native and upland shrubs becoming dominant in riparian areas, resulting in reduced wildlife habitat scores (Figures 19, 21, and 22).

Historic low flows that have supported extant riparian forests have never been recorded to fall below 0.057 cms (2 cfs) in the nearly 60 years of flow record along Cherry Creek. Extreme low flows have occurred during periods of prolonged drought (e.g., during drought years 1978, 2002, 2003, and 2004), the most recent of which are likely responsible for standing dead individuals (cottonwood and Goodding willow) observed during site visits in 2008. Channel avulsion and abandonment of former channels such as occurred during floods in 2005 may also be responsible for standing dead riparian trees along portions of Cherry Creek.

It is likely that riparian forest extent along Cherry Creek has expanded and contracted in response to wetter periods of time with increased streamflow and prolonged periods of drought. Tree growth rates were particularly sensitive to reductions in May, June and July flows, declining by as much as 50 percent between the highest flow years and those approaching median monthly flow (Figures 25 and 26). Extended periods of low flow could also inhibit survival of seedlings and saplings in areas normally favorable for survival, confining survivors to areas near the channel where they are likely to be killed by subsequent flooding (Mahoney and Rood 1998). Chronic low flows would be detrimental and have lasting effects on riparian forest cover as other species would become established, potentially excluding native riparian trees even when favorable conditions return (Stromberg *et al.* 2007a, Merritt and Poff 2010).

The maintenance of groundwater levels sufficient to support the current extent of native-dominated riparian forest requires support of the inputs of water to the alluvial aquifer along Cherry Creek. Lowering of alluvial water tables may be caused by reduced flow from upstream (caused by climate change or water extraction), channel incision (caused by headcutting, downcutting, or sand and gravel mining), groundwater pumping, and localized streamflow diversion (Stromberg *et al.* 1996, Kondolf 1997). Though the channel through the study reach is fluvially active, with signs of localized filling, scour, natural levee deposits, and evidence of past channel avulsion, there does not appear to be any widespread downcutting or channel incision along Cherry Creek. Our data suggest that groundwater levels and fluctuations across the valley are strongly related to the volume of surface water flowing in Cherry Creek. Streamflow measured at the USGS gage (approximately 15 km upstream), explained nearly 70 percent of the variability in measured water table fluctuations in a groundwater well located on river left (east side) along the study reach.

Along Cherry Creek, groundwater levels are a function of the balance between inflows and outflows of water from the valley. Inputs of water to the valley are from precipitation, inflow from tributaries and side slopes, and streamflow in Cherry Creek. In this arid valley, streamflow is the principal source of water to the alluvial water table in the main study reach. Outflows of water from the valley include loss from streamflow exiting the valley downstream, evaporation and transpiration by vegetation, and flow diversion. The short term effects of lowered volume of water flowing into the valley (through reduced flow from upstream in Cherry Creek) would be a corresponding reduction in groundwater levels. Comparison of surface and groundwater hydrographs indicates that water table levels are highly responsive to even small pulses in streamflow. Groundwater levels increase and decrease over short periods of time due to high porosity and hydraulic conductivity of the substrate along Cherry Creek. The chronic or long-term effects of a reduction in flow entering the valley (due to water extraction or drier climate) would be a shift in the water balance. Continued outflow of water from the valley thorough stream and groundwater movement and evapotranspiration without sufficient inflow of water to replace it will likely lead to depletion and lowering of the water table. With outflow exceeding inflow, the volume of the alluvial aquifer and the water table levels across the valley would continue to decline through time. Therefore, an examination of water balance (inputs versus outputs) would suggest that the effects of chronically reducing streamflow in Cherry Creek would be a progressive decrease in groundwater volume and level through time. Maintenance of near natural levels of low flow during the growing season (April through October; median (0.193 to 0.396 cms; 6.8 to 14 cfs)) is advisable (Appendix A and C). Avoidance of flows below 4 to 8 cfs (0.113 to 0.227 cms) is advisable as the stream shifts from connected perennial flows to disconnected and intermittent flows within this range.

High flows

Occasional high flows are necessary for maintenance of growth and vigor as well as the long-term sustainability of the riparian forests along Cherry Creek. Flow exceeding 8.212 cms (290 cfs) at least once a year is typical for Cherry Creek (1 year recurrence interval; Appendix C). Five and ten year recurrence interval flows have been 126.46 to 220.50 cms (4,466 cfs and 7,787 cfs), respectively. Extreme high flows are the principle mechanism forming the channel along Cherry Creek, as has been noted along other rivers in the region such as the Verde (Haney *et al.* 2008). High flows in riparian areas play an important role in creating and maintaining the channel as well as physical habitat for vegetation and wildlife species including amphibians (Bateman et al. 2008b). High flows mediate regeneration, biotic interactions, and growth and vigor of riparian and aquatic organisms (Stromberg *et al*. 2007b).

Through moistening the soil column and elevating groundwater levels, high flows serve to exclude upland species from riparian areas, reducing competition with native riparian species. High flows create and maintain new sites for seedlings of native riparian species to become established, recharge groundwater, moisten the soil column, maintain fine roots, and stimulate decomposition and enhance nutrient cycling and nutrient availability to plants (Molles *et al.* 1998). Vegetation near the contemporary channel along Cherry Creek is likely in a perpetual state of recovery from the last large flood.

Appropriately timed high flow pulses (e.g., in the spring when native riparian forest species are dispersing seed) are as important (or more important) biologically than the extreme high flows associated with winter monsoons along Cherry Creek. Spring and summer flow pulses translate as brief rises in groundwater levels, recharge and moisten the soil column and plant roots within it, and enhance transpiration and growth of riparian vegetation.

Prolonged periods without flooding along Cherry Creek would lead to accumulation of organic material on the floodplain (litter and wood), increased fuel and wildfire intensity, nutrient limitations for some species, drier soils, lowering of water tables and amplified water stress for phreatophytic vegetation during periods of low streamflow, and higher probability of shifts towards shrub-dominated vegetation in areas nearer the channel.

Conclusions

Frequency and duration of high and low flow, timing of flows, and interannual variability in flow characteristics are the principle factors that determine the structure and quality of wildlife habitat in riparian areas along Cherry Creek. The most certain means of maintaining the present qualities and characteristics of the wildlife habitat along Cherry Creek is continuation of historical long-term patterns of low flows, high flows, timing and sequencing of flows and the maintenance of the natural range of variability in seasonal and interannual flows. Any modification of flows from this natural range of variability would result in vegetation change. The degree and direction vegetation change in response to changes in flow regime depends upon the degree of deviation from historic patterns.

Adherence to a strict set of flows from year to year would lead to simplification of habitat along Cherry Creek (Appendix A), but would likely serve to maintain riparian forest. Strategies for managing wildlife habitat and recreational needs along Cherry Creek requires maintenance of perennial and seasonally variable flows from upstream. Reductions in low flows below historic ranges and durations would be detrimental to wildlife habitat along Cherry Creek. Maintenance of occasional high flows, such as the 1, 2 and 5 year recurrence interval flows of record (8.212, 43.210, and 126.459 cms (290, 1,526, and 4,466 cfs), resp.) are necessary for continued recruitment of riparian tree species. Such flows are in excess of the flows filed for by the Forest Service (Appendix A).

Occasional high flows equaling or exceeding the 0.878 cms (31 cfs) filed flow for the month of February is important for groundwater recharge and wetting of the soil column (Appendix A). On some years, flow during the low flow period could be reduced, and the volume of water delivered during a concentrated (short duration) period of high flows to enhance the geomorphic, hydrologic, and biogeochemical functions performed by high flows. It is well understood that the timing of these flows and the shape and timing of drawdown could be strategically designed to enhance regeneration of native riparian forest species such as cottonwood and Goodding willow (Mahoney and Rood 1998, Stromberg *et al*. 2007b). Maintenance of near historic levels of low flow during the growing season (April through October; median = 0.193 to 0.396 cms (6.8 to 14 cfs)) is advisable to prevent chronic lowering of the alluvial water table (Appendix A and C).

Our models provide a risk assessment of riparian forest loss as a function of groundwater decline. However, isolating precise flows associated with specific levels of groundwater is complicated by the fact that these relationships vary at any point along the study reach of Cherry Creek as a function of variability in channel form which varies spatially and through time due to flooding and fluvial processes. Furthermore, short and long term effects of variations in streamflow on water table levels differ; groundwater levels track short term increases and decreases in streamflow, but chronic low flows would result in groundwater declines exceeding those observed in groundwater measurements or those simulated in our models.

Scores of riparian studies in arid parts of North America cited in this report as well as the multiple lines of evidence in the current study from data collected along the main study reach of Cherry Creek, strongly suggest that incremental reductions in streamflow would result in increased likelihood of shifts from high quality riparian habitat (Frémont cottonwood, Goodding willow and Arizona sycamore-dominated riparian forest) to lower quality habitat dominated by native desert and xeroriparian shrubs such as burrobrush, desert broom, mesquite, and non-native saltcedar along Cherry Creek. Incremental reductions in streamflow and corresponding reductions in groundwater levels resulted in such shifts both in our models of groundwater decline and in patterns of vegetation measured along diverted flow-reaches and intermittent flow reaches.

Relationships between surface flow, groundwater, vegetation cover types and indices of avian, small mammal, and reptile habitat structural and compositional complexity suggest that incremental flow reductions in Cherry Creek would result in vegetation change and an increased risk of degradation in riparian habitat quality. Deviations from the range of natural flows (currently expressed in the patterns of present riparian vegetation and habitat along

Cherry Creek) will result in vegetation change along Cherry Creek. Our work and the extensive literature from this region suggest that reductions in streamflow would increase the likelihood (risk) of transition from forest- to shrub-dominated riparian vegetation and consequent degradation of avian, mammal, and reptile wildlife habitat. Though many of the desired functions of the extant riparian ecosystems of Cherry Creek could be supported by the filed flows, the risk of the loss of such functions is amplified as a function of degree of deviation from the historic range of flows.

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Appendices

Appendix A. Tonto National Forest instream flow water right filing with the state of Arizona (filing #33-96609) in cubic feet per second (cfs) and cubic meters per second (cms). Exceedance probability (percentage of the time that daily flow is equaled or exceeded) was calculated for the entire period of record from USGS streamflow gage no. 09497980.

Month	Filed flows (cfs)	Filed flows (cms)	Exceedance probability
January	16.0	0.453	22.6
February	23.0	0.651	17.8
March	31.0	0.878	14.5
April	14.0	0.396	24.7
May	9.2	0.261	39.6
June	6.8	0.193	58.5
July	7.0	0.198	55.8
August	8.1	0.229	45.1
September	7.2	0.204	53.3
October	6.9	0.195	57.3
November	7.9	0.224	46.9
December	10.0	0.283	34.2

Appendix B. Modified Wentworth particle size (grain size) scale.

Φ scale	Size range (SI units)	Size range (English units)	Aggregate class (Wentworth)
< -8	> 256 mm	> 10.1 in	Boulder
-6 to -8	64–256 mm	2.5–10.1 in	Cobble
-1 to -6	2–64 mm	1.26–2.5 in	Gravel
4 to -1	62.5 µm–2 mm	0.039–0.079 in	Sand
> 4	<62.5 µm	0.00015–0.0025 in	Fines

Appendix C. Exceedance probability and flood frequency for daily flows from the Cherry Creek near Globe, AZ USGS streamflow gage (no. 09497980). Daily average and flood instantaneous peak flow data are from the period May 1965 and through October 2009. Values are rounded to the nearest whole number for cfs.

Exceedance probability (percentage of the time flow equaled or exceeded)	Discharge (cfs)	Discharge (cms)	Flood recurrence interval (years)	Discharge (cfs)	Discharge (cms)
99.99	2	0.057	1	290	8.212
90	4	0.113	2	1526	43.210
80	5	0.142	5	4466	126.459
70	6	0.170	10	7787	220.497
60	7	0.198	25	14030	397.274
50	8	0.227			
40	9	0.255			
30	12	0.340			
20	20	0.566			
10	52	1.472			