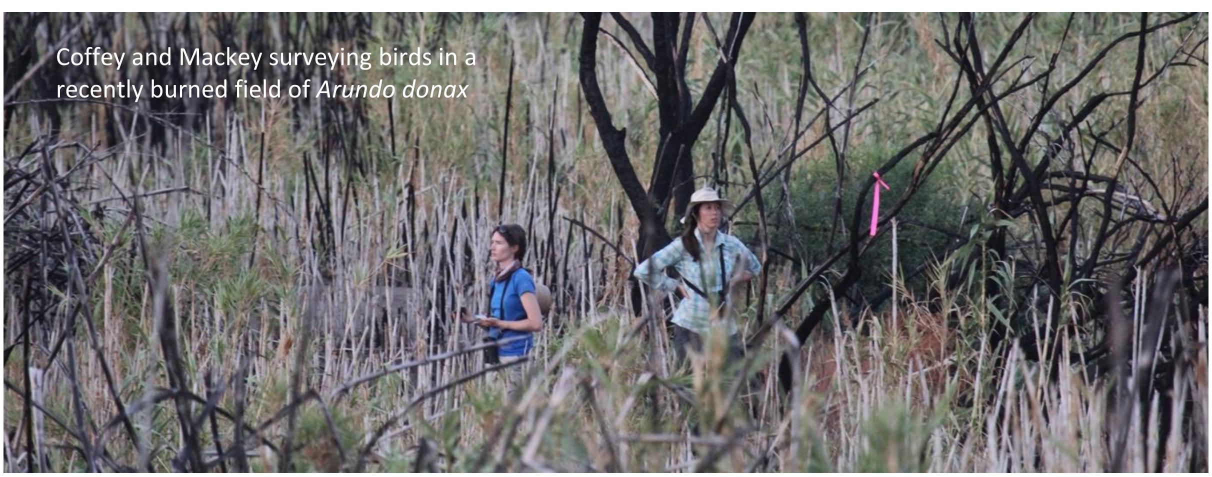
Annual Report 2017

Bird and butterfly community response to large-scale invasive plant removal and native plant restoration in desert riparian habitat along the Rio Grande/Bravo, Big Bend National Park, Texas


The Rio Grande/Bravo near Rooney's Place, Big Bend National Park, with *Arundo donax* visible along its banks


Coffey and Mackey surveying birds in a recently burned field of *Arundo donax*

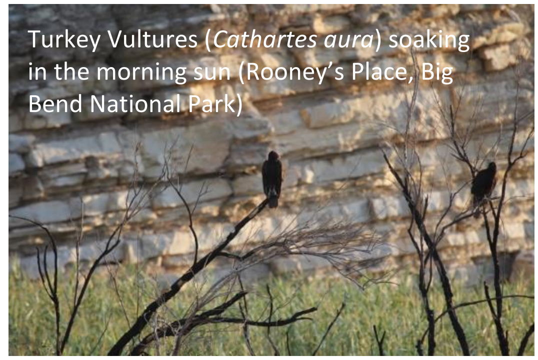
Authors

Julie Coffey
Graduate Student
California State University of Los Angeles
5151 State University Dr.
Los Angeles, CA 90032-8201

Heather Mackey
Graduate Student
California State University of Los Angeles
5151 State University Dr.
Los Angeles, CA 90032-8201

Lars Pomara
Research Ecologist
Eastern Forest Environmental Threat Assessment Center
US Forest Service Southern Research Station
200 W.T. Weaver Blvd
Asheville, NC 28804
Phone: (512)-501-0260

Eric M. Wood
Assistant Professor
Department of Biological Sciences
California State University of Los Angeles
5151 State University Dr.
Los Angeles, CA 90032-8201



Turkey Vultures (*Cathartes aura*) soaking in the morning sun (Rooney's Place, Big Bend National Park)

Acknowledgements

Park staff Raymond Skiles, David Larson, Jeanette Juraldo, and Marie Landis were instrumental in assisting with planning logistics, sampling design input, acquisition of remote-sensing data, and assistance with plant and butterfly identification and understanding historical and present riparian conditions throughout the duration of this study. This survey was carried out under permit # BIBE-2016-SCI-0037.

Project Summary

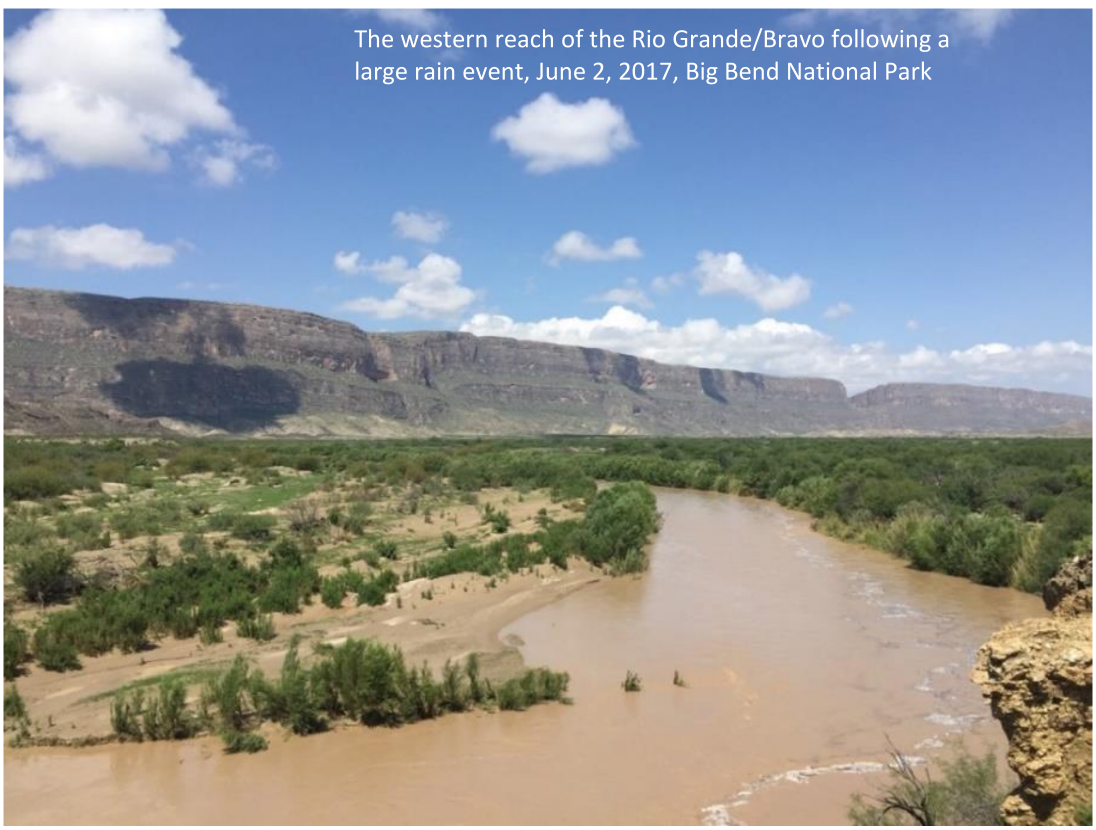
This report summarizes the 2017 annual findings from a two-year study, established in 2016, aimed at assessing how National Park Service efforts to remove giant cane (*Arundo donax,* hereafter, *Arundo*) from the riparian corridor of the Rio Grande/Bravo in Big Bend National Park influences terrestrial vertebrate and invertebrate wildlife. For additional information on the project background, justification, and survey approach, please see the 2016 annual report from this project (Mackey et al., 2016). Here, we present findings from our 2017 summer field season (i.e., our second year of fieldwork) to build on our initial findings concerning how terrestrial wildlife are responding to habitat manipulations. Altogether, we hope our findings will serve as a baseline for future monitoring of the restoration of the Rio Grande floodplain within the park.

The two objectives of our study were: (1) to quantify the response of bird and butterfly communities to restoration activities; and (2) to model habitat associations and determine nesting sites and nesting phenology throughout the floodplain by Yellow-billed Cuckoos (*Coccyzus americanus*). The western sub-population of the Yellow-billed Cuckoo is listed as threatened under the US Endangered Species Act, warranting a special focus on this species. We show that habitat structure and vegetation composition, data that were collected for field sites via both field sampling and remote sensing, continue to regenerate following prescribed burning and herbicide application. We documented positive responses in riparian-affiliated bird and butterfly species richness and abundance in *Arundo* removal locations that were burned > 4 years ago, compared with more recently burned locations. The Yellow-billed Cuckoo appeared unaffected by the cane removal, likely because they were less abundant in the narrow strips of floodplain habitat where most burning has occurred. We continue to detect cuckoos in virtually all wooded survey areas in the floodplain, highlighting the importance of the Big Bend National Park reach of the Rio Grande/Bravo for this species.

Patterns in the response of birds and butterflies relative to restoration activities remained mostly stable between 2016 and 2017 (see Mackey et al 2016 for comparative results). However, conditions for much of the floodplain – especially west of Mariscal Canyon - were dramatically different between years. On May 22, 2017, the Big Bend region experienced heavy rainfall, resulting in a measured flow in the Rio Grande/Bravo of 30,000 cubic feet per second, according to the USGS gauging station near Castalon (USGS, 2017). The effects of this pulse of water on the habitat were noticeable through increased herbaceous cover and wildflowers across most of the park's western floodplain – including more upland sites - as well as inundation of low-lying sites. These phenomena may help to explain the increase in butterfly abundance at survey locations that were unoccupied in 2016, as butterflies likely tracked the shifting phenology of flower and nectar resources found in what had previously been bare ground. Another impact of the flood, the closure of the river road, delayed our access to survey locations downriver from Castolon until the second week of June. However, we were still able to complete breeding bird surveys within the recommended window.

For our second season of surveys, we added three additional components to our fieldwork related to our two objectives. To bolster objective one, we (1) monitored the influence of plant phenology (i.e. flowering) on habitat use by butterflies, and we (2) added additional survey locations embedded within *Arundo*-dominated habitat. To quantify butterfly habitat use, we monitored butterfly foraging behavior (i.e. 'use' of flowering plants) and estimated abundance of flowering plants (i.e. 'availability' of foraging substrates) at each visit to survey locations. Finally, to better understand how bird and butterfly communities utilize habitat dominated by *Arundo*, we added five additional bird-monitoring surveys in locations of the floodplain where *Arundo* was present. To expand on objective two, we (3) assessed the breeding phenology of Yellow-billed Cuckoos by adding an additional round of passive/playback surveys, and several weeks of nest searching throughout our study area. While we did not record any active nests, our observations hinted at the possibility of asynchronous timing of nesting between birds found on the eastern and western sections of the park. Whether this was due to environmental causes (e.g., the flooding events), or biological differences (i.e., a distinction between eastern and western populations of Yellow-billed Cuckoo) remains unclear and warrants further study.

Taken together, the results from our multi-year effort continue to suggest that the removal of *Arundo* positively affects early-successional components of riparian vegetation, which likely positively influenced butterfly species, and hints at the recovery of later-successional woody vegetation, which is a strong component of habitat for breeding birds.


The western reach of the Rio Grande/Bravo following a large rain event, June 2, 2017, Big Bend National Park

Background

The Rio Grande/Rio Bravo constitutes one of the most important desert riparian systems in the US southwest, in terms of economic, cultural, and biological value. To maintain the biological diversity of the 118-mile stretch of the Rio Grande within Big Bend National Park (BIBE), the National Park Service, in collaboration with the World Wildlife Fund and the Mexican government, is undertaking a major restoration effort to remove *Arundo* and *Tamarix* along the BIBE corridor. The removal efforts are accomplished using a combination of burn and herbicide applications (for *Arundo*) and a biocontrol (saltcedar leaf beetle, *Diorhabda sublineata*), for *Tamarix.* This is part of a binational effort to promote "desired future conditions" along this historically, culturally, and recreationally significant international waterway (CEC 2014).

Broad expectations are that restoration efforts in BIBE will result in regenerating native plant communities in restored areas, which, in turn will result in a more diverse wildlife community. It is essential to document the ecological impacts of restoration efforts in order to evaluate project effectiveness and help inform future resource management strategies. One way to measure the success of the restoration work in BIBE is to monitor the response of vegetation, and bird and butterfly communities, all of which are useful indicators of riparian health (Nelson and Andersen 1994, Gardali et al. 2006).

We had two objectives for our 2017 field season: (1) quantify the response of bird and butterfly communities to restoration activities; and (2) model habitat associations and determine nesting sites and nesting phenology throughout the floodplain by Yellow-billed Cuckoos. We expected that bird and butterfly diversity and composition would be lower prior to burning in *Arundo* dominated habitat in part, because *Arundo* may displace other plant species (e.g. biotic homogenization) and thus reduce available niches for wildlife. Immediately after the removal of *Arundo* via prescribed burning, we expected that bird and butterfly diversity and composition would further drop due to loss of habitat. However, we expected a general rebound in bird and butterfly communities, and possibly enhanced diversity and abundances, following burning as habitat succession ensued (Fig. 1).

Methods

Sampling Design

We established a systematic sampling design to monitor breeding bird and butterfly communities throughout the Rio Grande/Bravo in BIBE (Mackey et al. 2016, Fig. 2). To place sampling locations, we used a high-resolution (1-m^2) National Agricultural Imagery Program image, taken during the summer of 2015. In a GIS (ArcMap) environment, we digitized all riparian habitat, including bosques and vegetation clearly within the floodplain, along the 118-mile stretch of the Rio Grande/Rio Bravo, outside of canyon reaches. We also relied upon a digitized spatial layer, provided by the National Park Service, depicting cane stands. We then drew a rectangular polygon covering the entirety of the riparian habitat polygons, and created a 300-m^2 square grid across this rectangle. All line intersections of the grid that fell within the areas of riparian habitat on the US side of the river were identified as candidate locations for sampling points for birds and butterflies (hereafter, sample points or sample transects, Mackey et al. 2016, Fig. 2). We used the minimum 300-m spacing, as this is a standard distance to minimize the potential for double counting individual birds during breeding bird point count surveys (Ralph et al. 1995). Within contiguous sections of riparian habitat, we established walking routes, consisting of seven to ten sample points, moving some candidate points to ensure sampling of restored areas (but adhering to the minimum separation distance). Our sampling design resulted in 168 points, arranged among 21 walking routes; 11 along the western reach of the BIBE section of the river, and 10 along the eastern reach; Mariscal Canyon being the boundary between western and eastern reaches (Mackey et al. 2016). Most significant riparian vegetation blocks on the US side of the river were saturated with sample points using this approach, with the important exception of a large section immediately west of Mariscal Canyon that was excluded due to remoteness (Mackey et al. 2016). We geolocated all survey points with a GPS in the field, and the resulting locations were intersected in a GIS with the spatial layer indicating *Arundo* removal areas and years treated.

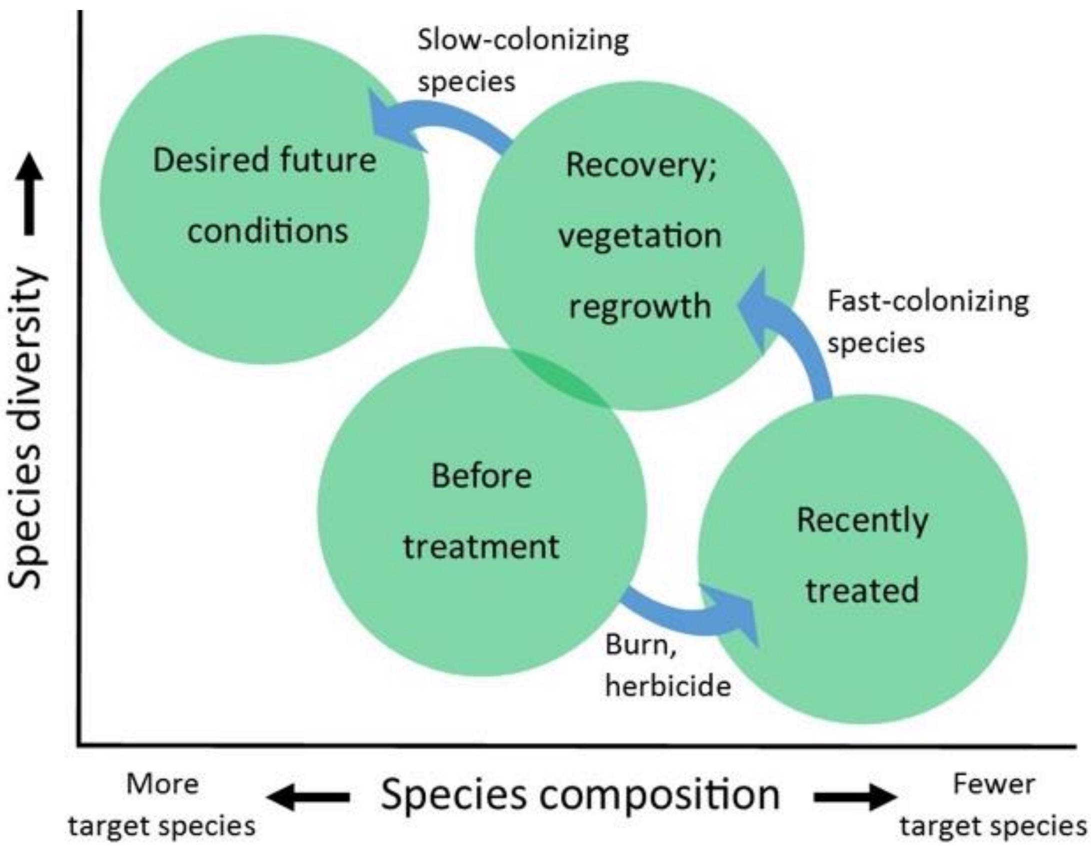


Figure 1. Conceptual diagram depicting the expected response of bird and butterfly diversity and composition before and after removal of Arundo donax from the Big Bend National Park Rio Grande/Bravo floodplain.

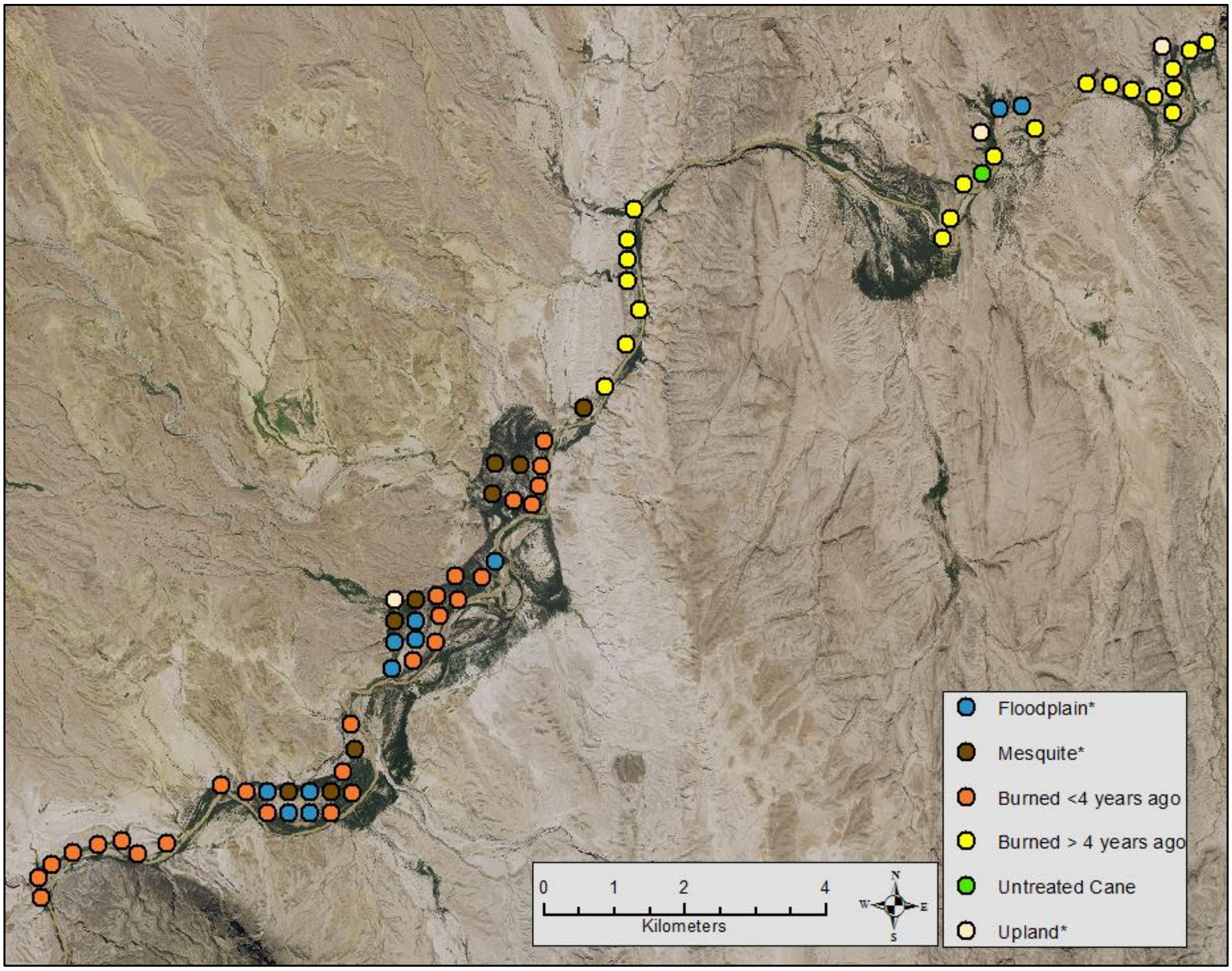


Figure 2. An example of survey locations East of Mariscal canyon, colored by habitat categories indicated in legend on bottom right. For full sampling design details, see Mackey et al. 2016. Floodplain sites are representative of untreated floodplain vegetation without strong Arundo (i.e. cane) or mesquite components. Mesquite sites were typically situated within large bosques.

Bird Surveys

We used standardized, five-minute point count surveys to characterize the breeding bird community at each sample point (Ralph et al. 1995, Mackey et al. 2016, Fig. 2). We visited each sample point six times; three visits during each field season, with each visit occurring one to two weeks apart, between May 19th-June 24th, 2016 and May 25th-July 1st, 2017. We began point counts typically within ten minutes after sunrise and ended within four hours. We varied the starting point of our walking routes with each visit to control for temporal variation in bird detectability and rotated observers among route visits to minimize observer bias. Graduate students, Julie Coffey and Heather Mackey, conducted all counts. They were trained in auditory and visual identification of birds found along the Rio Grande/Bravo by Pomara and Wood. For each bird detection during a point count, we estimated the approximate distance, and detection method (visual, call or song). We used the following distance categories for each detection: 0-10 m, 10-20 m, 20-30 m, 30-40 m, 40-50 m, with categories for >50-m and birds that were detected as flying over the habitat. We did not conduct surveys during high winds or rain.

Butterfly surveys

We established 100-m butterfly transects at each sample point, with the midpoint of the transect either centered on the bird point count location (*n* = 47), or the sample point as a start- or end-point of a transect (*n* = 121) (Mackey et al. 2016). We established straight-line transects through walkable habitat following protocols designed to monitor the federally endangered Karner Blue Butterfly (*Lycaeides melissa samuelis*, Brown and Boyce 1998). Our starting point was dependent on our ability to walk through dense vegetation and survey butterflies. On seven occasions, the bird point count was located in nearly impenetrable vegetation (i.e. mesquite, *Prosopis* spp.) making it difficult to maneuver. In these seven cases, we repositioned transects along the edge of the dense vegetation. Since many of our sample points were adjacent to the Rio Grande/Rio Bravo, we generally positioned transects parallel to the river and flagged the start, center and ends. We used compasses to ensure our walking routes were consistently followed, as any deviation from transects would influence distance estimation for butterflies.

We visited butterfly transects seven times over the duration of the study: three visits during May 19th - June 27th, 2016 and four during May 19th-July 23rd, 2017. All surveys began between 8:00 am and 1:45 pm, with an average start time of 10:30 am. To survey butterflies, we walked transects slowly in a straight line, identifying all butterflies detected within a search area that was approximately 10 x 100 m (Mackey et al. 2016). Butterfly detections typically occurred in two ways: a butterfly sitting on a plant surface, or a butterfly flying within our survey area. If we observed a butterfly feeding on a flower surface, we noted the species of butterfly and the plant that it fed on. This feeding behavioral observation documented 'use' by butterflies for a particular plant species. The purpose of this analysis was to quantify habitat-use by butterflies (Objective 1; see *Habitat Quantification – field based* below for detailed methods on 'availability' of flowering plants to foraging butterflies). When we detected a butterfly on a plant surface, we noted the distance, perpendicularly, from the butterfly, or plant if the butterfly was repeatedly changing locations, to a transect, and we identified the plant that was being 'used'. When we detected a butterfly flying within our survey area, we marked the distance that the butterfly was first detected, perpendicularly, to a transect. We used the following distance categories for each detection: 0-0.5 m, 0.5-1 m, 1-1.5 m, 1.5-2.25 m, 2.25-3 m, 3-4 m, and 4-5 m, and >5-m (Brown and Boyce 1998). We surveyed each transect for a minimum of five minutes. If we detected a butterfly but were not able to identify it from the location of a transect, we would maneuver from the transect to a better viewing position to identify the species before resuming the transect survey. We would not record any new individuals during this time.

Yellow-billed Cuckoo Passive and Playback Surveys

A target species of our bird surveys was the Yellow-billed Cuckoo. The Yellow-billed Cuckoo is separated into two populations or subspecies, the 'western' (*Coccyzus americanus occidentalis* and 'eastern' *Coccyzus americanus americanus*) (US Fish and Wildlife Service 2014). The western population was listed as threatened in 2014 due to losses in its preferred breeding habitat, riparian gallery forest (US Fish and Wildlife Service 2014). The USFWS-recognized geographic boundary between the western and eastern populations bisects BIBE, occurring at Mariscal Canyon along the Rio Bravo/Rio Grande (US Fish and Wildlife Service 2014). Given uncertainty surrounding the precise location of this boundary (or possibly hybrid zone), management of Yellow-billed Cuckoo habitat throughout BIBE is warranted, as is further study of the taxonomic identity of Yellow-billed Cuckoos breeding in BIBE.

To increase detection rates for Yellow-billed Cuckoos, we used a call playback protocol at each sample point (Halterman et al. 2015). Following a passive point count, we broadcasted recordings of Yellow-billed Cuckoo calls for a one-minute period. We then listened for a minute, before repeating the process for an additional two minutes. Evidence suggests the western sub-population utilizes sonically unique vocalizations compared with the eastern subpopulation (US Fish and Wildlife Service 2014). Therefore, we designed our playback surveys to account for potential differences in responses from individuals belonging to western or eastern populations. At sample points east of Mariscal Canyon, we played eastern calls for the first minute of our survey, followed by western calls for the second minute. We did the reverse for sample points west of Mariscal Canyon. We obtained western and eastern Yellow-billed Cuckoos recordings from the bird-vocalizations sharing website, xeno-canto (http://www.xeno-canto.org/). We broadcast recordings consisting of contact and territorial calls ("kowlp call" and "coo coo" call, respectively) using a smartphone and handheld speaker. If we detected a Yellow-billed Cuckoo during a passive point count, we did not perform call playback and recorded its distance following our standard protocol for bird point counts. If we detected a Yellow-billed Cuckoo during the call playback, we stopped the survey and recorded the distance to, and the location of, the individual. We conducted Yellow-billed Cuckoo surveys at each point three times during the 2016 field season and four times during 2017.

Yellow-billed Cuckoo Nest Searching

To gather information on nesting locations and timing by breeding Yellow-billed Cuckoos, we revisited survey locations to conduct nest searching from July 2-July 25th, 2017. We conducted nest searching every other day during this period, alternating with a fourth round of playback surveys and an additional round of butterfly surveys. We selected survey locations for nest searching based on density of Yellow-billed Cuckoo detected during previous playback surveys, as well as feasibility of walking through the mesquite. We initially conducted nest searching west of Mariscal Canyon within the Cottonwood/Castalon, Boat Launch, Santa Elena, and Triangulation Station bosques, and east of Mariscal Canyon in San Vicente, Casa de Piedra, and La Clocha bosques. Our approach was similar to that of Flippo and Flippo (2015). We began nest searching at sunrise and passively observed and listened for Yellow-billed Cuckoo activity. If we detected a Yellow-billed Cuckoo, we attempted to track the individual, or pair, and determine what locations within the bosque they were nesting, and what behaviors they were engaged in (e.g. nest material carry). If we observed nest building, fledglings, or other sensitive breeding behavior, we noted the behavior and left the area to minimize disturbance.

Habitat quantification - field based

We quantified vegetation cover, at both the bird and butterfly survey extents from July 3-23rd, 2017. For bird-count vegetation surveys, we used a relevé method to estimate percent cover at the ground, shrub (<4 m), and canopy (>4 m) level within the 100-m radius circle at each sample point (Wood et al. 2011). Broad vegetation categories included tree (>4 m in height), shrub (<4 m in height), dead shrub, grass, herbaceous, water, duff, and bare ground. Estimates of tree (>4 m) or shrub (<4 m) percent cover included mesquite, *Tamarix*, and willow (*Chilopsis* spp., and *Salix* spp.). We identified any species that covered 5% or more of the plot, and we estimated the percent cover of the individual species.

For butterfly-transect vegetation surveys, we estimated percent cover of the same variables as the bird-count vegetation surveys, but within a 10 x 100-m rectangle centered on the butterfly transect (Mackey et al. 2016). In addition to percent cover, we noted the presence of flowering plant species. From our vegetation surveys, we identified six habitat groupings, which were the basis for components of our analyses described below (Table 1).

Further, we collected data on plant flowering on each visit to a transect. We noted the plant species that was flowering and estimated the number of stems for any flowering plants encountered within 5-m of the transect. The purpose of this data collection method was to estimate the ‘availability’ of flowers as a nectar source for foraging butterflies.

Habitat quantification - remote sensing

To calculate metrics of landcover and habitat structure remotely, we used aerial imagery collected by the National Agriculture Imagery Program (NAIP) and the Texas Natural Resources Information Systems (TNRIS). We calculated the extent of landcover habitat types as the number of pixels of the respective landcover as a function of the total number of landcover pixels within a 100-m radius circular buffer of a sample point (Wood et al. 2016). *Arundo* percent cover and total riparian vegetation cover were quantified by using a combination of image segmentation results and manual classification in ArcGIS 5.1 (ESRI 2017). We quantified the Normalized Difference Vegetation Index (NDVI) on the NAIP imagery. We calculated the 1st-order standard deviation (e.g. standard deviation of the pixel values within a moving window), which is useful in characterizing the variability of horizontal habitat structure throughout an image (Wood et al. 2012). By using both the 1-m aerial imagery and the 10-m satellite imagery, we captured both fine- and broad-scale habitat variability.

Table 1. Habitat categories, sample sizes, and descriptions for bird and butterfly sampling locations. Bird and butterfly surveys differ because we categorized habitat at two spatial extents: (a) 100-m radius circles surrounding bird sample points and (b) 10 m x 100 m rectangle surrounding butterfly transects.

Category	*n* = bird	*n* = butterfly	Description
Burned < 4	34	28	Locations burned in any year from 2014 to 2017. This category captures 'recent' burning efforts.
Burned > 4	21	16	Locations burned in 2014 or prior. This category captures 'older' burning efforts.
Unburned cane	9	14	Unburned locations with >10 percent cover of *Arundo*
Floodplain	64	58	Locations in open floodplain (non-forested)
Mesquite*	31	27	Honey mesquite dominated gallery forest
Upland*	9	25	Outside of the floodplain - upland desert vegetation

**Sites excluded from analyses, as not directly relevant to Arundo management hypothesis*

Analytical methods

For our first objective, we performed the following two analyses:

1. We quantified differences in vegetation cover, and bird and butterfly richness and abundance among four habitat categories detailed in Table 1. To refine our analysis, we removed passage migrants, and focused on birds that breed within the floodplain (Appendix Table 1A). To evaluate the degree of difference in vegetation cover and bird and butterfly richness and abundance among habitat categories, we used a one-way analysis of variance (ANOVA). Prior to analysis, we checked assumptions for ANOVAs, which revealed all assumptions were satisfied. Following significant ANOVAs, we performed a multiple-comparisons routine, using a Tukey-Kramer's test. Because we made three comparisons during the multiple comparisons routine, we used a Bonferroni adjusted *alpha level* of 0.08 to assess significance (=0.05/6 = 0.08).
2. As a preliminary evaluation of univariate habitat associations, we assessed the correlation, using Spearman's *rho*, between bird and butterfly richness and abundance and habitat characteristics (i.e., habitat and remotely sensed variables). We used the graphical and statistical program *R* for all analyses described above (R Core Team 2013).

For our second objective, we determined whether restoration efforts affected habitat use of the Yellow-billed Cuckoo by evaluating Yellow-billed Cuckoo sample-point occupancy in a formal habitat occupancy analysis. We fitted multi-season, single-species occupancy models by relating Yellow-billed Cuckoo habitat occupancy to habitat characteristics and remotely sensed variables using the PRESENCE statistical software (MacKenzie et al. 2006). We held detection probability as constant within a year, and fitted single-variable models to estimate the proportion of site occupancy. We used a multi-season approach to capture occupancy patterns over the two years of our surveys.

Results

OBJECTIVE 1: EVALUATING THE IMPACT OF RESTORATION ON THE BIRD AND BUTTERFLY COMMUNITY

Analysis 1 – Response by bird and butterfly communities and vegetation to restoration

Some patterns of vegetation cover were distinct among bird and butterfly habitat groupings (Tables 2 and 3). *Arundo* was significantly lower in burned sites compared with unburned, and similar to floodplain sites, while grass cover showed a similar pattern (though far lower in floodplain) (Table 1). Remotely sensed variables, such as the normalized vegetation difference index (NDVI), a measure of vegetation greenness, characterized trends expected from post-disturbance successional dynamics. NDVI of burned sample points was significantly lower than that of unburned floodplain sample points (Table 2).

In comparing bird and butterfly diversity and abundance among habitat groupings, we found that bird richness and abundance were lower in recently burned sites, but not significantly different between older burned sites and the floodplain (Table 2). We found that, although there were strong differences in the means of butterfly richness and abundance among habitats, there were no statistically significant differences among habitat categories (Table 3).

Analysis 2 – Relationships among bird and butterfly communities and vegetation

When examining relationships among bird and butterfly richness communities with habitat characteristics, we found that bird and abundance was positively correlated with total vegetation cover, mesquite shrub cover, and average habitat greenness (mean NDVI) (Table 4). Bird abundance was positively associated with grass cover, while bird richness was not. For butterflies, richness and abundance were strongly positively correlated with herbaceous cover (Table 4).

Butterfly richness was negatively correlated with mean distance to river, indicating that survey transects further from the river recorded fewer species of butterflies. This observation supports that butterflies likely move to locations of the floodplain where food resources (and possibly host plants as well) are abundant. Butterfly abundance was negatively correlated with bare ground, but positively associated with grass cover (Table 4).

Table 2. Average values, plus 95% confidence intervals in parentheses, for bird richness and abundance, and habitat characteristics at 128 sample points, grouped among four habitat categories described in Table 1. Categories followed by an asterisk are habitat characteristics within the 100-m radius around a point count location that were estimated using remote sensing. Letters that differ, following average values within a row, indicate a significant difference among habitat categories for a particular bird or vegetation metric based on a one-way analysis of variance (ANOVA), followed by a Tukey-Kramer multiple comparisons test for significant ANOVAs. We assessed significance at the Bonferroni adjusted alpha-value of 0.008 (=0.05/6 comparisons).

	Burned < 4		**Burned ≥ 4**		**Unburned Cane**		**Floodplain**	
Bird richness	8.59^{A}	(7.97,9.20)	9.43^{AB}	(8.63,10.22)	10.22^{AB}	(8.84,11.60)	10.44^{B}	(9.80,11.07)
Bird abundance	15.70^{A}	(14.10,17.27)	22.90^{B}	(20.70,25.10)	21.56^{B}	(18.5,24.60)	23.66^{B}	(21.80,25.49)
Mesquite > 4m	12.71^{A}	(7.69,17.71)	1.43^{B}	(0.23,2.63)	7.78^{AB}	(1.43,14.12)	12.91^{A}	(9.50,16.30)
Mesquite < 4m	18.97	(15.25,22.7)	14.40	(9.83,18.93)	12.20	(6.78,17.67)	14.10	(10.90,17.22)
Tamarix	5.94	(2.63,9.25)	2.52	(0.83,4.22)	3.33	(0.50,6.20)	5.14	(3.40,6.88)
Herbaceous	7.35	(4.36,10.34)	2.90	(1.21,4.59)	7.56	(4.12,10.98)	5.14	(3.50,6.78)
*Percent cane**	4.71^{A}	(2.42,7)	2.52^{A}	(0.89,4.15)	15.16^{B}	(12.85,17.46)	2.44^{A}	(1.72,3.17)
Bare ground	35.89^{AB}	(30.6,41.14)	50.48^{A}	(41.8,59.16)	22.22^{B}	(8.94,35.5)	36.75^{AB}	(31.29,42.20)
Grass	4.85^{A}	(1.83,7.89)	8.81^{AB}	(3.46,14.16)	8^{AB}	(3.50,12.50)	19.34^{B}	(15.10,23.58)
*Mean distance to river (m)**	123^{AB}	(88,157)	73^{A}	(57,89)	93^{AB}	(68,117)	182^{B}	(147,217)
*Percent vegetation cover**	33.34	(24.92,41.77)	34.70	(26.74,42.67)	71.82	(63.17,80.47)	52.78	(47.47,58.08)
*Mean NDVI**	0.09^{A}	(0.06,0.12)	0.09^{A}	(0.07,0.11)	0.26^{B}	(0.19,0.33)	0.19^{B}	(0.17,0.21)

*****Remotely-sensed variables estimated from 2016 aerial imagery (1-m, National Agricultural Imagery Program)

Table 3. Average values, plus 95% confidence intervals in parentheses, for butterfly richness and abundance, and habitat characteristics at 116 sample transects, grouped among four habitat categories that are described in Table 1. Letters that differ, following average values within a row, indicate a significant difference among habitat categories for a particular butterfly or vegetation metric based on a one-way analysis of variance (ANOVA), followed by a Tukey-Kramer multiple comparisons test for significant ANOVAs. We assessed significance at the Bonferroni adjusted alpha-value of 0.008 (=0.05/6 comparisons).

	Burned < 4		**Burned ≥ 4**		**Unburned Cane**		**Floodplain**	
Butterfly Richness	3.71	(2.71,4.72)	5	(4.31,5.69)	5.36	(4.17,6.55)	3.67	(3.13,4.22)
Butterfly Abundance	9.50	(5.67,13.33)	20.63	(13.37,27.88)	19.36	(11.19,27.52)	12.71	(9.84,15.58)
Bare ground	56.50	(45.25,67.75)	42.68	(24.92,60.46)	29.57	(14.69,44.52)	49.40	(40.72,58.08)
Mesquite > 4m	1	(0.12,1.88)	2.25	(0.06,4.45)	2.86	(0,8.46)	3.40	(0.70,6.13)
Mesquite < 4m	4.61	(0.67,8.54)	2.69	(0.80,4.58)	3.21	(0.57,5.85)	3.89	(1.37,6.42)
Herbaceous	7.29	(2.63,11.94)	14.50	(4.93,24.07)	10.43	(4.89,15.97)	9.53	(5.77,13.30)
*Percent Cane**	5.36^{AB}	(2.64,8.09)	2.35^{B}	(0.43,4.27)	9.09^{A}	(5.90,12.27)	2.62^{B}	(1.55,3.70)
*Percent vegetation**	34.33	(24.75,43.92)	37.10	(27.56,46.58)	62.94	(52.65,73.22)	51.93	(46.16,57.69)
*Mean NDVI**	0.01^{A}	(0.06,0.14)	0.10^{A}	(0.07,0.12)	0.24^{B}	(0.19,0.29)	0.19^{B}	(0.16,0.21)
*Mean distance to river (m)**	129	(88,170)	63	(48,77)	81	(47,114)	217	(174,260)
Tamarix	5.54	(1.28,9.79)	2.81	(0,6.03)	5.36	(0.82,9.89)	3.67	(1.84,5.5)
Total Grass	4.04^{A}	(2.19,5.88)	9.94^{AB}	(5.17,14.71)	14.43^{B}	(9.33,19.53)	12.71^{B}	(10.45,15.21)

Remotely-sensed variables estimated from 2016 aerial imagery (1-m, National Agricultural Imagery Program)

Table 4. Spearman's correlation coefficient (rho) for richness and abundance of birds and butterflies with ten habitat characteristics. Values in bold indicate a significant correlation ($p < 0.05$).

	Bird richness (*n*=128)		**Bird abundance (*n*=128)**		**Butterfly richness (*n*=116)**		**Butterfly abundance (*n*=116)**	
	Rho (ρ)	*p-value*	*Rho (ρ)*	*p-value*	*Rho (ρ)*	*p-value*	*Rho (ρ)*	*p-value*
Mesquite > 4m	0.07	0.41	-0.01	0.88	**-0.19**	**0.05**	0.04	0.70
Mesquite < 4m	**0.16**	**0.07**	0.12	0.18	-0.05	0.58	0.02	0.86
Herbaceous	0.07	0.45	-0.10	0.26	**0.35**	**<0.01**	**0.42**	**<0.01**
Bare ground	-0.03	0.73	-0.04	0.68	-0.15	0.11	**-0.19**	**0.04**
Grass	0.11	0.20	**0.20**	**0.03**	0.18	0.06	**0.20**	**0.03**
*Percent Arundo**	0.07	0.40	0.06	0.47	-0.07	0.46	-0.05	0.62
*Percent vegetation cover**	**0.24**	**0.01**	**0.32**	**<0.01**	0.10	0.26	0.16	0.08
*Mean distance to river (m)**	0.07	0.42	-0.11	0.21	**-0.20**	**0.03**	-0.18	0.06
*Mean NDVI**	**0.28**	**0.01**	**0.33**	**<0.01**	0.10	0.28	0.18	0.05

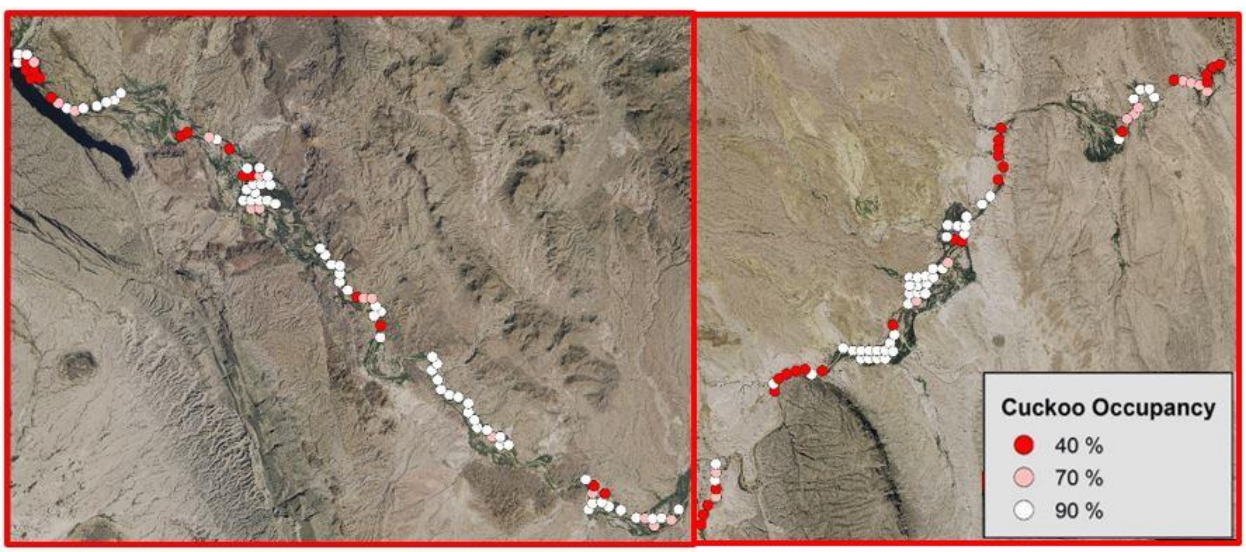
*****Remotely-sensed variables estimated from 2016 aerial imagery (1-m, National Agricultural Imagery Program)

OBJECTIVE 2: HABITAT RELATIONSHIPS OF THE YELLOW-BILLED CUCKOO

Yellow-billed Cuckoo naïve occupancy was 51.5% among all sample points. Sample points that were occupied by Yellow-billed Cuckoos had 53% higher mesquite cover than unoccupied sample points. We found there was no significant difference between *Arundo* or *Tamarix* percent cover at sample points that were occupied by Yellow-billed Cuckoos compared with unoccupied sample points.

Based on the multi-season, single-species occupancy analysis (patterns of occupancy in 2016 and 2017), we found that our detection probability was 0.25 (95% CI, 0.20 - 0.30), and our estimate of sample-point occupancy was 0.51 (0.37 - 0.66). Site colonization between 2016 and 2017 was 0.62 (SE, 0.12) and site extinction was 0.10 (SE, 0.12). Our top (lowest-AIC) model strongly supported the expectation that Yellow-billed Cuckoos occupy sites with high tree cover, with 59% of model deviance accounted for by proportion of mesquite bosque at the 100-m scale, identified by NAIP image classification. The model with the second highest support had 33% of model deviance accounted for by percent cover of mesquite bosque estimated by relevé. All other habitat variables had low support (ΔAIC > 2.0) for predicting Yellow-billed Cuckoo occupancy.

Our occupancy analysis revealed that the following locations had the highest probability of hosting Yellow-billed Cuckoos: San Vicente and Casa de Piedra, (east side of Mariscal Canyon), and Johnson's Ranch, Black Dike, Smokey Creek, Buenos Aires, and Castolon (west side of Mariscal Canyon) (Fig. 4).



***F**igure 4. Map of Yellow-billed Cuckoo predicted site occupancy calculated from a multi-season, single-species intercept-only, occupancy model [ψ(.), p(.)]. Colored circles indicate sites predicted occupancy (ie. red circles have predicted occupancy of 40%).*

Discussion

Our results from the 2017 field season suggest that breeding bird communities in *Arundo* removal sites continue to rebound to levels comparable with the surrounding floodplain. Our results broadly support our expectations regarding the response of vegetation and wildlife following removal of *Arundo*. In general, we found support for an increase in bird and butterfly richness and abundance from newly burned sites with relatively low abundance of native flora and fauna, to older burned sites, to floodplain and *Arundo*-dominated sites. We note that the higher richness and abundance of bird and butterfly species in *Arundo*-dominated sites could be due to the many other habitat conditions found in those locations. On the other hand, the strong rebound in bird and butterfly richness and abundance following removal of *Arundo* suggests the management efforts are having a positive effect on the flora and fauna throughout the floodplain. Lastly, our remotely sensed variables highlight the return of vegetation and structure (increased greenness), which is likely a contributing driver to the increasing diversity and abundance of the bird and butterfly communities.

Butterflies exhibited significant differences in richness and abundance among habitat types in our first season, with older burned sites harboring the greatest abundance of butterflies. In our second season, this pattern held, however differences between groups were no longer significant, perhaps due to increases in herbaceous plants and butterfly abundance across the floodplain dampening this effect somewhat. We did find that herbaceous cover continued to be significantly correlated with butterfly abundance, and that both butterfly abundance and herbaceous cover increased in the unburned floodplain from 2016 to 2017. While the NAIP 1-m imagery we used for habitat greenness calculations for 2016 is not available for 2017, we hope to use additional imagery to be able to quantify fine-scale habitat greenness for 2017, to better understand the changes in butterfly distribution between years.

Consistent with the first-season data, Yellow-billed Cuckoos appeared to be unaffected by the restoration activities and were found in nearly every floodplain route we surveyed on the river, including sites with no detections in 2016. Our nesting observations, while limited, hint at potential breeding asynchrony between Yellow-billed Cuckoos populations East and West of Mariscal canyon. We observed one possible pair in the early stages of building a nest in the Castalon/Cottonwood campground area (West side) in the same week as a parent was observed feeding a fledgling in Casa de Piedra (East side). While it is difficult to attribute our observation to any biological differences between Yellow-billed Cuckoos found on the western and eastern portions of the park (e.g. migration and nesting phenology), we believe a more detailed analysis is warranted to determine whether the park hosts both the western- and eastern subspecies of Yellow-billed Cuckoo.

Continuing Project Directions

Future directions for this project include more detailed occupancy modelling of individual bird and butterfly species' responses to vegetation changes and *Arundo* removal, as well as community species composition analyses of the bird and butterfly data sets to characterize drivers of species turnover among survey locations, including annual variability and cane removal in particular. We will also continue with quantitative analysis and a model selection framework of habitat variables at different spatial extents, in order to capture whether local or regional phenomena are driving patterns in bird and butterfly communities within the park. For Yellow-billed Cuckoo, we are working toward a spatially explicit occupancy likelihood layer, based on further development of occupancy models using remotely sensed data, which can be extended throughout the Big Bend NP and adjacent Mexican portions of the international floodplain.

Literature Cited

Bateman, H. L., and S. M. Ostoja. 2012. Invasive woody plants affect the composition of native lizard and small mammal communities in riparian woodlands. Animal Conservation 15:294–304.

Bateman, H. L., and E. H. Paxton. 2010. Saltcedar and Russian olive interactions with wildlife. In Saltcedar and Russian olive control demonstration act science assessment: 49–63. Shafroth, P.B., Brown, C.A. & Merritt, D.M. (Eds). Reston, Virginia: U.S. Geological Survey.

Brand, L. A., G. C. White, and B. R. Noon. 2008. Factors influencing species richness and community composition of breeding birds in a desert riparian corridor. The Condor 110:199–210.

Brown, J. A., and M. S. Boyce. 1998. Line transect sampling of Karner blue butterflies (*Lycaeides melissa samuelis*). Environmental and Ecological Statistics 5:81–91.

Durst, S. L., T. C. Theimer, E. H. Paxton, and M. K. Sogge. 2008. Temporal variation in the arthropod community of desert riparian habitats with varying amounts of saltcedar (*Tamarix ramosissima*). Journal of Arid Environments 72:1644–1653.

Flippo, M., and K. Flippo. 2015. Status of western yellow-billed cuckoo (*Coccyzus americanus occidentalis*) in Big Bend National Park, 2015. Report to Division of Science and Resource Management Big Bend National Park, Texas Professional Services Procurement BIBE-15-106, RM-BL-12.

Flory, S. L., and K. Clay. 2009. Invasive plant removal method determines native plant community responses. Journal of Applied Ecology 46:434–442.

Gardali, T., A. L. Holmes, S. L. Small, N. Nur, G. R. Geupel, and G. H. Golet. 2006. Abundance patterns of landbirds in restored and remnant riparian forest on the Sacramento River, California, U.S.A. Restoration Ecology 14:391–403.

Greco, S. . 2013. Patch change the shifting mosaic of an endangered bird's habitat on a large meandering river. River Research and Applications 29:707–717.

Gurevitch, J., and D. K. Padilla. 2004. Are invasive species a major cause of extinctions? Trends in Ecology and Evolution 19:470–474.

Halterman, M., M. J. Johnson, J. A. Holmes, and S. A. Laymon. 2015. A natural history summary and survey protocol for the western distinct population segment of the Yellow-billed Cuckoo: U.S. Fish and Wildlife Techniques and Methods, 45 p.

Lambert, A. M., T. L. Dudley, and K. Saltonstall. 2010. Ecology and impacts of the large-statured invasive grasses Arundo donax and Phragmites australis in North America. Invasive Plant Science and Management 3:489–494.

MacKenzie, D. I., J. D. Nichols, K. H. Pollock, J. A. Royle, L. L. Bailey, and J. E. Hines. 2006. Occupancy estimation and modeling: inferring patterns and dynamics of species occurrence. BOOK, Academic Press, San Diego, California, USA.

Nelson, S. M., and D. C. Andersen. 1994. An assessment of riparian environmental quality by using butterflies and disturbance susceptibility scores. Southwestern Naturalist 39:137–142.

Pimentel, D., R. Zuniga, and D. Morrison. 2005. Update on the environmental and economic

costs associated with alien-invasive species in the United States. Ecological Economics 52:273–288.
R Core Team. 2013. A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL http://www.R-project.org/. ICOMM.
Ralph, C. J., S. Droege, and J. R. Sauer. 1995. Managing and monitoring birds using point counts: standards and applications. Page Monitoring Bird Populations by Point Counts. USDA Forest Service, Pacific Southwest Research Station, General Technical Report PSW-GTR-149. JOUR, Forest Service, Pacific Southwest Research Station. Albany, California, USA.
van Riper III, C., K. L. Paxton, C. O'Brien, P. B. Shafroth, and L. J. McGrath. 2008. Rethinking avian response to Tamarix on the lower Colorado River: A threshold hypothesis. Restoration Ecology 16:155–167.
Sogge, M. K., S. J. Sferra, and E. H. Paxton. 2008. Tamarix as habitat for birds: Implications for riparian restoration in the Southwestern United States. Restoration Ecology 16:146–154.
Soykan, C. U., et al. 2012. Multitaxonomic diversity patterns along a desert riparian–upland gradient. PLoS ONE 7:e28235.
Swengel, A. B. 2001. A literature review of insect responses to fire, compared to other conservation managements of open habitat. Biodiversity and Conservation 10:1141–1169.
U.S. Geological Survey, 2018, National Water Information System data available on the World Wide Web (USGS Water Data for the Nation), accessed January, 08, 2018, at URL https://waterdata.usgs.gov/tx/nwis/uv/?site_no=08374550&PARAmeter_cd=00065,00060
US Fish and Wildlife Service. 2014. Endangered and threatened wildlife and plants; determination of threatened status for the western distinct population segment of the Yellow-billed Cuckoo (*Coccyzus americanus*); final rule. Federal Register 79:48548–48652.
Wiggins, D. A. 2005. Yellow-billed Cuckoo (*Coccyzus americanus*): a technical conservation assessment. [Online}. USDA Forest SErvice, Rocky Mountain Region, https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5182002.pdf
Wood, E. M., M. D. Johnson, and B. A. Garrison. 2011. Quantifying bird-habitat using a relevé method. Transactions of the Western Section of the Wildlife Society 46:25–41.
Zavaleta, E. S., R. J. Hobbs, and H. A. Mooney. 2001. Viewing invasive species removal in a whole-ecosystem context. Trends in Ecology and Evolution 16:454–459.