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ABSTRACT: Restoring and enhancing riparian vegetation on private and communal lands in Mexico is important for biodiversity conservation given the ecological significance of these areas and the scarcity of public protected areas. To enhance riparian vegetation and wildlife habitats and train local people in restoration techniques, we implemented restoration and outreach efforts on private and communal lands in the Sky Islands region of northwest Mexico. We fenced 475 ha of riparian zones from livestock, erected erosion-control structures, planted trees, and developed management agreements for cool-season grazing with landowners on 10 ranches across 3 sites in 2012-2013, then repaired fences and renegotiated agreements in 2017-2019. To foster evaluation, we used a before-after/control-impact design to measure attributes of vegetation structure and bird communities and compared baselines from 2012 with post-treatment estimates from 2019. As predicted, understory vegetation volume generally increased in treatments relative to controls ($P = .09$), especially when one treatment area with the lowest pre-treatment grazing impacts was censored ($P = .01$). Although canopy cover also increased, there was little differential change in treatments relative to controls ($P \geqslant .23$) due likely to longer time periods needed to realize responses. Densities of most focal bird populations varied across time periods in directions that typically matched observed changes in vegetation structure, but fewer species showed signs of differential positive change linked to treatments relative to controls. Densities of Yellow-breasted Chat, a key understory obligate and important focal species, increased in treatments relative to controls across sites, as did densities of Sinaloa Wren, which also use dense underbrush ($P \leqslant .05$). Positive changes by other understory obligates (eg, Common Yellowthroat, Song Sparrow) were more local but sometimes of high magnitude (>8-fold) also suggesting positive impacts of treatments. Despite mixed results over a limited time period, these patterns suggest restoration efforts drove localized recovery of understory vegetation and associated bird populations, but benefits varied widely with environmental and social factors linked to management. Greater ecological benefits to riparian areas on private and communal lands in this region can be fostered by further incentivizing construction, maintenance, and proper use of restoration infrastructure, through education, and by building relationships based on trust and credibility with landowners.

KEYWORDS: Before-after/control-impact, cattle exclosures, conservation incentives, cool-season grazing, density, distance sampling

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Introduction

Habitat loss and degradation are major threats to biodiversity and ecosystems worldwide.[1,2] In western North America, these threats and the ecological significance of riparian areas exemplify this crisis.[3] Riparian areas dominated by galleries of broadleaf deciduous trees are vital to both wildlife and human populations, especially in arid regions.[3] Nonetheless, such areas occupy only tiny portions of landscapes (eg, 1%), are highly threatened by a broad array of stressors, and have been lost or degraded across much of their historical range.[4-7] Habitat restoration is one of the few remaining options to conserve, enhance, and augment riparian areas and the habitats and ecosystem services they provide. Hence, broadleaf riparian forests are important foci for restoration efforts, which can have marked positive impacts on vegetation and wildlife.[8-11] Such efforts are especially significant in Mexico and elsewhere in Latin America where these and many other areas that support high biodiversity are privately or communally owned and have weaker conservation mechanisms than on public protected lands.[12] In Mexico, public lands cover less than 5% of the national territory and in northwestern Mexico, where grazing is a primary land use, grazing impacts are thought to be greater on communal than on private lands, but patterns can vary in complex ways at broader scales.[13] Hence, strategies that foster protection, augmentation, and enhancement of riparian areas on private and communal lands and that increase the capacity of local people to realize these goals are vital.

With the assistance of numerous partners including Sky Island Alliance and Borderlands Restoration L3C, we initiated a multifaceted approach to restore and enhance riparian areas in the Sky Islands region of northwestern Mexico. Our efforts focused on improving degraded riparian areas dominated by cottonwood (*Populus fremontii*) and willows (*Salix* spp.) on private and communal (ejido) lands, educating local people on restoration and management techniques, and monitoring ecological responses to treatments. Areas we considered had long histories of livestock grazing at the start of efforts, but impacts varied with ranch size, and social and ecological factors. During initial efforts in 2012-2013, we fenced 569 ha of riparian areas, erected erosion-control and other structures and planted trees within exclosures, and established 5-year management agreements that protected areas from warm-season livestock grazing across 4 project sites and 11 ranches.[14] Low to moderate levels of cool-season grazing is potentially a viable management strategy on private and communal lands in this region to improve vegetation cover and wildlife habitats, as has been shown in other western North American riparian areas.[15-18] This is because riparian areas are likely more resilient to grazing when soils are drier and vegetation is dormant,[15,19] and because complete livestock exclusion is rarely economically viable. Nonetheless, studies are needed to assess the efficacy of this practice, and resources such as pasture fencing are limited for implementing it in Mexico. To assess the efficacy of this strategy and facilitate long-term monitoring, we gathered data on 2 important ecological indicators in treatments and nearby controls before applications in 2012. In 2017-2019, we repaired fences where needed, renegotiated management agreements at 3 sites across 10 ranches, and reassessed focal indicators in 2019.

Assessing the benefits of restoration efforts for vegetation and wildlife can be complex because treatments can have a broad range of impacts on resources that are challenging to measure.[20] An efficient and direct way to evaluate restoration is to use a before-after/control-impact (BACI) design to compare changes in systems at treated sites to those within controls using data gathered before and after application.[21,22] Before-after/control-impact designs provide a robust way to identify changes associated with treatments because they consider 2 or more time periods and explicit controls, which reduces the potential for confounding due to environmental variation or natural changes in resources over time.[23] Unfortunately, logistical and administrative issues often linked to funding can make rigorous evaluations of restoration impacts difficult and limit our understanding of the benefits and ways to improve future efforts.[21] Here, we addressed some of these challenges through explicit use of a BACI design, data on multiple ecological indicators, and spatial replication across numerous treatment and control areas at 3 sites and 10 ranches.

As a framework for evaluation, we developed a priori hypotheses to guide assessments and used 2 attributes of riparian vegetation structure (understory volume and canopy cover) and both population and community parameters of breeding birds as ecological indicators. Birds are useful ecological indicators because most species depend on specific resources and conditions tied to foraging, nesting, and other requirements, which vary widely among species, and because communities are closely linked to overall ecological integrity.[24-26] Moreover, despite high acuity needed to survey birds, communities are efficient to sample by skilled observers and have high ecological, economic, and flagship values.[27,28] Because short-term livestock impacts are often focused within 1.5 m of the ground, we expected over the relatively short (~7 year) duration of the study that restoration treatments would have the greatest positive impact on understory vegetation near ground level versus in the canopy. Thus, we predicted reduced grazing linked to treatments would have positive impacts on understory vegetation and associated bird populations in treatments relative to controls.[15,29] For canopy cover, we expected little differential change in the short term and predicted either increases in treatments and controls as trees matured naturally over time, or no changes where mature trees were dominant. For birds, we predicted species dependent on understory vegetation such as dense shrubs, low trees, and herbaceous vegetation, and that nest on or near the ground would exhibit the greatest differential changes in response to treatments. In this region, these species include Common Yellowthroat (Latin names in Supplemental Appendix A), Yellow-breasted Chat, and Song Sparrow, which we considered individually and as an understory guild, and that have been used as important focal species for similar monitoring.[15,29] In contrast, we predicted species dependent on mid- and high-canopy resources, tree boles, and other resources would show little or no response to treatments relative to controls over the period considered. Here, we assess the impacts of treatments, evaluate ecological and social factors that may have driven responses, and implications for future efforts.

Materials and Methods

Study area

We considered 3 sites in northern Sonora, Mexico, located along major watercourses that drain several Sky Island mountain ranges in the United States-Mexico borderlands, which dominate the physiography and biogeography of this region (Figure 1).[30,31] Two sites, Río Cocóspera and Río Santa Cruz near San Lazaro, were along major river valleys dominated by lowland riparian associations of cottonwood and willows. A third site, Milpillas, was along a smaller drainage and supported both lowland and montane tree species including sycamore (*Platanus wrightii*) and walnut (*Juglans major*). At Río Cocóspera, many riparian trees along the main river channel were relatively young having germinated following a major flood event in December 1992 (C. Robles Elías, personal communication, Nov. 2019). This event fostered widespread germination and subsequent recruitment of riparian trees in the region. Such was likely the case near the main channel along the Río Santa Cruz near San Lazaro, but here a much broader floodplain promoted more complex age structure with cohorts of very large old cottonwood trees at greater distances from the main channel. Along the relatively narrow canyon along Arroyo Milpillas, in contrast, most riparian trees were large and old, and a more shaded forest floor limited potential for understory growth.

Site conditions varied widely before treatments due to differences in ranch size, ownership, past management, and grazing levels. In general, private lands along the Río Cocóspera

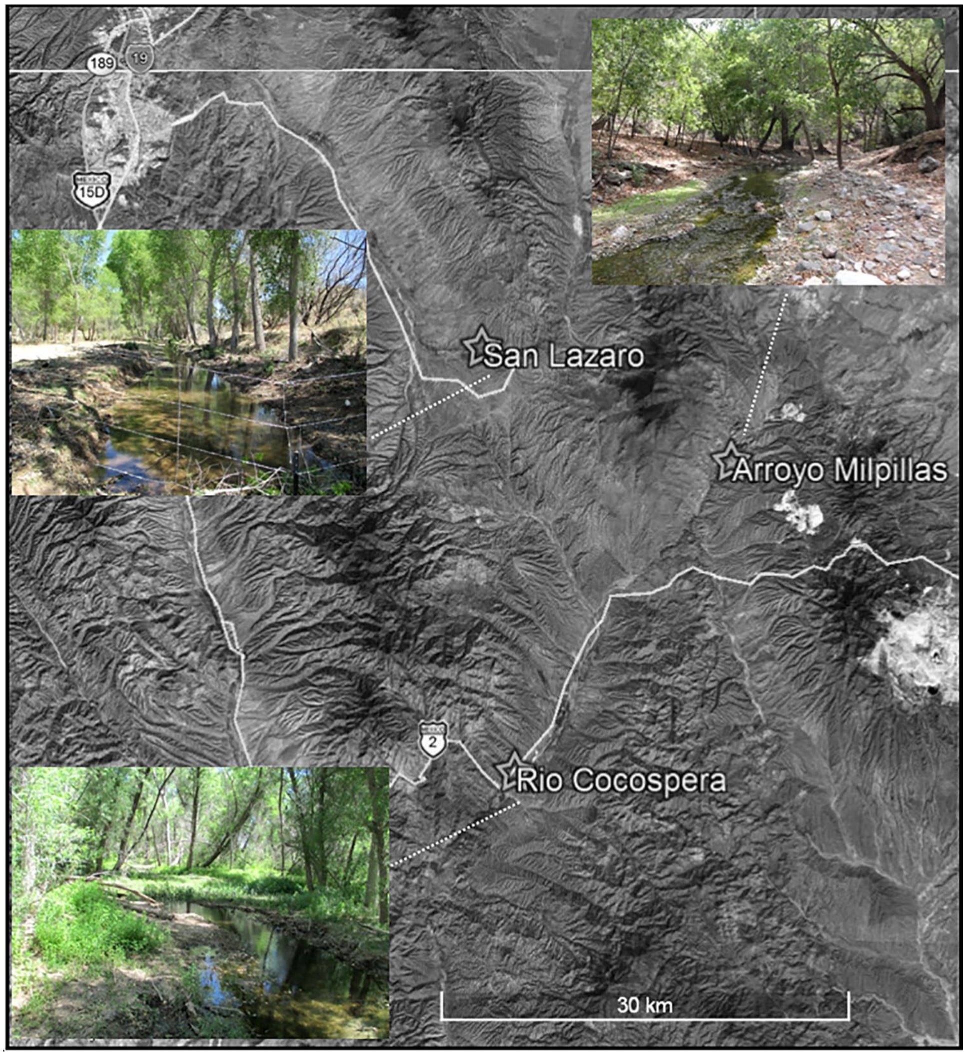

Figure 1. Location of 3 study sites in northern Sonora, Mexico, where we monitored the impacts of riparian restoration treatments on birds and vegetation, 2012-2019. The horizontal line to the north is the United States-Mexico border, with major roads labeled for reference.

were the least intensively managed, included larger parcels, and owned by families with more diverse income sources. In contrast, communal lands along the Río Santa Cruz near San Lazaro were more intensively managed, had smaller pastures, and managers more dependent on income from livestock. Nonetheless, all landowners were committed to enhancing the ecological and economic value of their lands, saw inherent value in conservation, and some were involved in prior conservation, education, and outreach efforts that helped us recruit them into the program.

Design and treatments

Restoration treatments entailed extensive fencing to exclude livestock from riparian zones during the growing season. Fencing was often combined with local construction of plug-and-pond features and efforts to reconnect spring flows with historical river channels, bank stabilization structures, and pole plantings inside exclosures installed in collaboration with landowners, volunteers, and other staff based on local site needs (see Supplemental Appendix B). With each landowner, we negotiated management agreements for a period of 5 years that focused on cool-season grazing and basic maintenance of structures. Grazing regimes sometimes varied somewhat, however, due in part to localized failure of fences, noncompliance, or sometimes more restrictive grazing practices. Due to specific landowner needs and other site considerations, fenced exclosures were typically not randomly assigned across space. Some fences were constructed to allow livestock access water along small 5- to 15-m sections of drainage channels.

For monitoring, we placed 12 transects, 0.29 to 2.38 km in length (total = 15.69 km; Supplemental Appendix C) on terraces oriented to follow drainage channels across the 3 project sites. Transects were established in spring 2012 in areas where restoration was proposed but specific treatment locations not always known a priori. We subdivided transects into 50-m sections to allow data to be linked with eventual treatment and control locations and sampled all treatment and as many nearby control areas as possible. To estimate baseline conditions, we sampled bird communities continuously across the

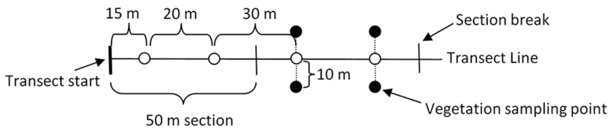

Figure 2. Schematic illustration of transect layout and vegetation sampling points used to monitoring the influence of riparian restoration treatments on canopy cover and understory vegetation volume in Sonora, Mexico, 2012-2019. Only 4 sampling points are shown but points were placed systematically at 20- or 30-m intervals, 10 m on either side of transects.

full length of transects at 2- to 4-week intervals 6 times between early April and mid July 2012, and again in the same manner following treatments 2 times between late April and early June 2019. We measured vegetation once in each year in late spring or early summer after trees had leafed out and spring ground cover was established. Vegetation measurements began 15 m from the start of each transect and were made at alternating distances of 20 and 30 m across the full length of transects (Figure 2). At each of these distances, we measured canopy cover and vegetation volume at 2 points that were each 10 m from transect lines in perpendicular directions (or closer if cliffs or large rocks were present) so that each 50-m section included 4 sampling points (Figure 2). Because some sections were slighter longer than 50 m, this sometimes resulted in >2 pairs of points per section. In 2019, we reduced these measurements to 2 pairs of points per section to streamline efforts.

Field measurements

To estimate canopy cover, we used a spherical densiometer and made 4 readings at each point by turning 90° between readings and averaging estimates for each point. To estimate understory vegetation volume, we used a 1 m × 1 m white cover board. We placed the board at each point and estimated the proportion of the board covered by vegetation (including woody plants and debris, branches, forbs, grasses) from 8 m away at 2 locations that we placed parallel to and 10 m away from transect lines. To assess bird densities independent of variation in detection probability, we used distance sampling methods along line transects.[32] For each bird detection, we recorded the species, number of individuals, sex (if known), detection type (aural or visual), behavior (singing, calling, drumming, flying, silent), and the initial perpendicular distance from transect lines to the actual or estimated location of each individual or center of each flock. We noted the 50-m section in which each bird was observed so data could be linked to specific treatment and control locations. We used a laser rangefinder to measure distances to birds and trained field technicians in distance estimation to assure accuracy. All bird surveys were completed by single observers from 20 minutes before until as late as 3.5 hours after local sunrise during periods of low winds and no precipitation.

Statistical analyses

We used distance sampling methods to derive detectability-corrected estimates of densities within treatments and controls. This approach involves fitting detection functions to frequency histograms of distance data to model the decline in detection probability with increasing distance from observers and the influence of spatial, temporal, individual, survey, and other covariates on the observation process.[32] When vegetation changes markedly between time periods, spatiotemporal variation in detection probability can confound comparisons of bird abundance. We omitted observations of flyovers that were not directly using resources along transects. Because the monitoring period was longer in 2012, we censored data from early April and July 2012 and considered a total of 5 or 6 surveys per transect across time to foster comparisons.

To estimate bird densities and allow inferences at both local and larger scales, we pooled bird data from selected consecutive 50-m transect sections into a set of transect reaches that we sized to match the smallest treatment areas (eg, 150 m in length). This stratification process fostered more precise density estimates at local reach-specific scales because sample sizes within individual 50-m sections were limited even for common species. On average, reaches were 270 m in length (range = 150-360 m) and included 4.7, 50 m sections (range = 2-7; Supplemental Appendix C). Across sites, treatment reaches (n = 21) totaled 5.87 km in length and controls (n = 37) totaled 9.82 km.

We focused on 23 bird species and 1 species group (obligate understory species; Common Yellowthroat, Yellow-breasted Chat, Song Sparrow) for density (no./ha.) estimation. Focal species were selected because they depend on a broad suite of representative resources that enabled us to evaluate changes in species linked to specific environmental attributes of interest, and because they were encountered a sufficient number of times to foster precise estimates (⩾50 total encounters). We computed density estimates at 2 spatial scales: (1) pooled within all treatment or control reaches across sites both before and after treatments (eg, before-treatment, before-control, after-treatment, after-control), and (2) within each transect reach classified as a control or treatment in each time period. We fit both simple detection functions with no covariates and more complex functions with covariates and used program Distance version 7.3.[33,34] We considered spatial (site) and temporal (year, Julian day) covariates and assumed variation in detectability due to vegetation was linked to them. We also considered covariates linked to bird observations (detection type, group size, sex) and survey conditions (temperature, noise level). We compared models with single covariates first and then assessed various additive combinations of covariates for supported models, and used *Akaike information criterion* adjusted for small sample sizes (AIC_c) to rank models. To select final models, we assessed shapes of detection functions, precision of estimates, and goodness-of-fit of highly ranked models and selected the best overall models.[33] We fit uniform, half-normal (HN), and hazard-rate (HR) functions for models without covariates, and HN and HR functions for models with covariates, and considered up to 2 cosine, simple polynomial, or hermite adjustment terms. We considered data grouped into various bin sizes to best smooth histograms and right truncated 1% to 5% of encounter data to foster model fitting.

Species richness or the number of species in a community can be a useful metric for assessing ecological change.[35] Because species are not detected perfectly during surveys, species present but undetected could bias richness estimates. Thus, we used observed species abundance distributions from survey data to estimate richness $(\widehat{N})$ with a bias-corrected version of the Chao 1 estimator[35,36] defined as

$$\widehat{N} = N_{obs} + \frac{f_1(f_1 - 1)}{2(f_2 + 1)}$$

where N_{obs} is the number of species observed, f_1 is the number of species observed once, and f_2 is the number of species observed twice in the sample.

To estimate treatment effects for vegetation parameters and bird species richness, we used linear mixed-effects models with site classification (treatment or control), time period (before or after), and site classification by time period interactions as fixed effects, and site and year as random effects. BACI designs test for differential change in responses within treatments relative to controls across time, which are estimated by treatment by time period interactions (eg, test for nonparallel responses). Thus, we report effect sizes and *P* values for interaction terms and least square means for all combinations of site classification and time period. Because transect 2 at Cocóspera (Robles canyon) had very limited pre-treatment grazing impacts and high baseline values given it was the focus of past conservation efforts, we fit models with and without data from this transect when assessing vegetation change. For bird densities, we used 95% confidence intervals estimated by program Distance to assess the significance of changes within treatment reaches relative to controls before and after treatments. Nonoverlapping 95% confidence intervals were considered evidence ($P < .05$) of statistical significance. All models were fit with JMP version 9.0.[37] To assess more local changes in bird densities, we compared reach-specific estimates across various treatment and control reaches output by program Distance.

Results

Vegetation

We obtained 1446 estimates of vegetation parameters in 2012 and 1082 in 2019. Understory volume and canopy cover both increased across time but not always relative to changes in controls. Changes in the understory were of greater relative magnitude than those in the canopy, especially at Cocóspera, but varied widely among sites (Figure 3). As predicted, understory volume increased in treatments relative to that in controls across all sites and transects combined ($P = .092$ for treatment by time interaction), especially when one transect with the lowest pre-treatment grazing impacts and highest baseline values was censored ($P = .014$; Tables 1 and 2). At San Lazaro, and especially at Milpillas, post-treatment estimates of understory volume were higher in treatments than in controls, with interaction plots indicating nonparallel responses (Figure 3). Nonetheless, understory volume remained fairly low at both sites following treatments, suggesting only modest responses, and increases in treatments at San Lazaro where largely offset by declines in controls (Figure 3). At Cocóspera, post-treatment increases in volume were greatest averaging a 168% increase across 4 of 5 treated transects vs a 120% increase in controls (Table 2), but overall changes were similar (Figure 3). In contrast, variation in canopy cover showed no evidence of differential change in treatments relative to controls ($P \geqslant .23$; Table 1), with similar changes across sites despite markedly different baselines (Table 3). On average, canopy cover was greatest in the mature riparian forest at Milpillas, moderate at Cocóspera, and much lower on the broad floodplain of the Río Santa Cruz.

At a transect scale, variation in understory volume showed both encouraging and discouraging patterns (Table 2). At Cocóspera, magnitudes of change were greatest along treated sections of transect 3.1 (Supplemental Appendix C) where volume increased nearly 4-fold, but only to moderate levels (eg, 22%; Table 2). Downstream in the cienega along transect 1, volume increased by an estimated 117% following treatments to high levels (eg, 51%; see Supplemental Appendix D for before-after photos). Further downstream in the canyon where pre-treatment baselines were high due to low grazing impacts, understory volume increased by just 12%, but reached peak levels that may be near maximum potential vegetation. At Milpillas, post-treatment increases in understory volume were also large (eg, 141%) but reached only moderate levels, whereas estimates in controls remained within 1% of baselines. At San Lazaro, post-treatment increases in volume were more local and increased by 2% to 82%, with volume decreasing by 23% to 28% in controls.

Birds

Spatiotemporal changes in bird densities varied widely among focal species, but patterns of change were often similar for species dependent on similar vegetation resources. Among all focal species and groups, densities of 16 species and 1 group (71%) showed some evidence of increasing across time in treatments, controls, or both (Figure 4, Supplemental Appendix E). Importantly, densities of White-winged Dove, Sinaloa Wren, Yellow-breasted Chat, and Summer Tanager all increased in treatments relative to changes in controls. In contrast, densities of Cassin's Kingbird, which use openings in forest and woodland, increased more in controls than in treatments. Hence, for the focal understory species Yellow-breasted Chat, predictions linked to hypothesized increases in understory vegetation, upon which this species depends, were realized. For other focal

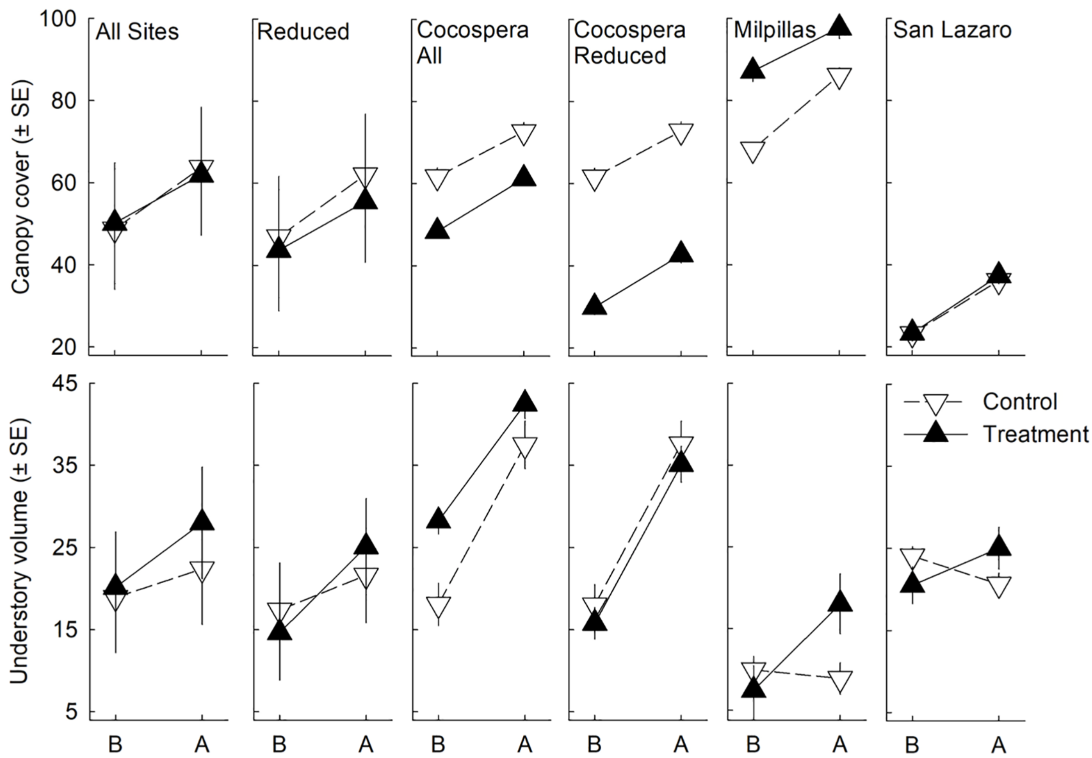

Figure 3. Effects of riparian restoration treatments on canopy cover (percent; top row) and understory vegetation volume (percent; bottom row) across 3 sites in northern Sonora, Mexico, in 2012 and 2019. Points are least square means (±SEs) from multifactor, mixed-effects ANOVA comparing differences between treatments and controls before (B) and after (A) treatments. Reduced data include all transects except for Cocóspera transect 2, which had low pre-treatment grazing impacts. ANOVA indicates analysis of variance.

understory species Common Yellowthroat and Song Sparrow, however, this prediction was not realized, but marked local increases in densities in some treated sections suggested positive effects (Supplemental Appendix E; see below).

For the 2 most abundant species, point estimates of densities of canopy-dependent Yellow Warbler increased by 76% in controls and 61% in treatments across time with nonoverlapping 95% confidence intervals between time periods indicating a highly significant increase. Similarly, densities of woodland-dependent Bewick's Wren increased in both treatments and controls but at somewhat lower magnitudes. Densities of trunk-dependent Gila Woodpecker and woodland-opening-dependent Western Wood-Pewee also increased significantly by somewhat greater magnitudes, but again changes within treatments were similar to those in controls (Supplemental Appendix E).

For focal understory species, variation in reach-specific estimates suggested some auspicious local changes (Supplemental Appendix E). In the cienega at Cocóspera (transect 1), for example, post-treatment increases in densities of Common Yellowthroat, Yellow-breasted Chat, and Song Sparrow were all high. This was especially the case in the upstream reach most heavily impacted before treatments where cover of forbs, graminoids, and shrubs increased markedly (Supplemental Appendix D). Here, point estimates of densities of Common Yellowthroat increased by a notable 747% (from 0.8 to 6.9 individuals/ha) between time periods, with ⩾167% increases of the 2 other focal species. In contrast, densities of these species increased 71% to 100% in the adjacent but much less impacted reach downstream on the Robles property. At San Lazaro, increases in these 3 focal species were largely limited to transect 3. Results were mixed at Milpillas where densities of Yellow-breasted Chat decreased in treatments relative to controls but distribution of Common Yellowthroat expanded into treatment reaches where they were not observed initially.

Over time, we observed 162 bird species at project sites during standardized line transect sampling (Supplemental Appendix A). There was little spatiotemporal change in observed and estimated species richness among transect reaches (Table 4, Figure 5). On average, observed richness was lower along control reaches, but once corrected for detection probability differences were not statistically significant. Across all reaches and visits, estimated richness averaged 16.4 ± 0.9 (±SE) greater than that observed, and 19.4 ± 0.3 overall. On average, Cocóspera (mean ± *SE* = 20.8 ± 0.4) and San Lazaro (20.6 ± 0.5) had higher observed richness than at Milpillas (15.5 ± 0.6). Patterns of estimated richness were similar with somewhat higher estimates at San Lazaro (39.3 ± 1.8) than at Cocóspera (34.3 ± 1.2).

Discussion

Restoration impacts

We assessed the impacts of restoration treatments that included extensive livestock exclosures and cool-season grazing regimes

Table 1. Influence of riparian restoration treatments on canopy cover and understory vegetation volume across 3 riparian sites in northern Sonora, Mexico, where we monitored the impacts of restoration treatments in 2012 and 2019.

PARAMETER; DATA SET (N) / FACTOR	ESTIMATE	SE	*T*	*P* VALUE
Canopy cover; all data (2528)				
Intercept	56.12	10.36	5.42	.0056
Site Classification [Control]	0.09	0.69	0.13	.90
Time Period [After]	6.69	10.36	0.65	.55
Time Period [After]*Site Classification [Control]	0.81	0.69	1.18	.24
Canopy cover; reduced (2318)				
Intercept	52.05	10.42	5.00	.0075
Site Classification [Control]	2.43	0.68	3.55	.0004
Time Period [After]	6.75	10.42	0.65	.55
Time Period [After]*Site Classification [Control]	0.82	0.68	1.20	.23
Understory vegetation volume; all data (2528)				
Intercept	22.35	4.72	4.73	.0093
Site Classification [Control]	–1.72	0.65	2.65	.0081
Time Period [After]	2.83	4.72	0.60	.58
Time Period [After]*Site Classification [Control]	–1.09	0.65	1.69	.092
Understory vegetation volume; reduced (2318)				
Intercept	19.69	4.03	4.88	.0084
Site Classification [Control]	–0.19	0.64	0.30	.77
Time Period [After]	3.67	4.03	0.91	.41
Time Period [After]*Site Classification [Control]	–1.56	0.64	2.46	.014

Estimates are based on a linear mixed-effects models with site classification (treatment or control), time period (before or after), and site classification by time period interactions fit as fixed effects, and site and year fit as random effects, based on a before-after/control-impact design to assess restoration response. Models for each parameter are presented based on all data combined and a reduced data set that censored measurements from one transect along the Río Cocóspera that was classified in the treatment group, but had limited impacts by livestock before the start of project and thus had less potential to illustrate treatment effects.

on riparian vegetation and breeding birds on private and communal lands across 3 sites in northwest Mexico. Understory vegetation volume and densities of some bird populations linked to this key limiting resource generally increased in treatments relative to controls, thus adding to a growing body of evidence for the value of such restoration strategies.[16-18,38] The magnitude of overall change and patterns of differential change within treatments relative to controls, however, indicated varying levels of positive response, increases in relatively few bird populations linked directly to treatments, and hence only limited success. Complete protection from grazing would have almost certainly fostered greater increases in vegetation structure and bird abundances, and results were undoubtedly dampened by some localized failure of fences between project phases.[18,39-41] On private and communal lands in this and other regions of Mexico, however, complete exclusion of livestock is rarely an economically viable option for land managers, thus necessitating evaluations of this and other potential restoration strategies. Whereas recovery times are likely longer when grazing is not completely excluded due to continued but more limited impacts to soil quality, nutrient cycling, plant recruitment, and vegetation volume,[17,18] the approach used here seems viable provided landowners adhere to management agreements and set grazing levels carefully based on site and weather conditions. Evaluating the impacts of these practices on reproductive output, survival, and other demographic attributes of wildlife populations[42,43] is an important future question for assessing the ultimate feasibility of this and other strategies.

As predicted, a key focal indicator of riparian restoration efforts, understory vegetation volume, increased in treatments relative to controls at 2 sites. At a third site (Cocóspera), understory volume increased by a greater overall magnitude but changes in treatments were similar to those in controls, suggesting factors other than treatments alone drove these

Table 2. Understory vegetation volume between 0 and 1 m above ground (%) within restoration treatments and nearby control areas along 12 transects across 2 time periods in northern Sonora, Mexico, where we applied riparian restoration treatments.

SITE NAME	SITE CLASS	TRANSECT NO.	BEFORE (2012)			AFTER (2019)			DIFFERENCE, %
			NO. POINTS	MEAN	SE	NO. POINTS	MEAN	SE	
Cocospera	Treatment	1	40	23.4	4.73	42	50.6	3.63	116.6
Cocospera	Treatment	2[a]	114	47.9	3.13	94	53.7	2.79	12.2
Cocospera	Control	3.1	16	6.6	2.11	12	14.8	3.57	125.4
Cocospera	Treatment	3.1	32	4.6	2.68	20	22.0	3.43	376.7
Cocospera	Treatment	3.2	90	14.8	2.59	60	28.4	2.86	91.7
Cocospera	Treatment	4	24	20.1	4.86	20	37.9	6.01	88.4
Cocospera	Control	5	94	19.5	2.95	74	41.8	2.98	113.8
Milpillas	Control	1	62	8.3	2.19	44	7.2	1.56	–12.5
Milpillas	Treatment	1	82	7.4	1.82	54	17.8	2.71	141.0
Milpillas	Control	2	202	10.3	1.24	156	9.3	1.09	–10.3
San Lazaro	Control	1	154	18.0	2.14	112	17.3	2.22	–4.0
San Lazaro	Control	2	170	14.0	1.92	120	13.4	1.76	–4.6
San Lazaro	Control	3	46	16.7	3.87	34	12.0	3.23	–28.2
San Lazaro	Treatment	3	106	22.1	3.21	82	22.5	2.57	1.9
San Lazaro	Control	4	170	41.7	3.00	128	32.3	2.85	–22.5
San Lazaro	Treatment	4	38	17.0	4.63	30	30.8	4.69	81.5

Before period denotes pre-treatment measurements, whereas after period was ~6 to 7 years post-treatment. Treatments included fencing riparian vegetation to reduce warm-season grazing, erosion control structures, and occasional vegetation planting. Differences note percent change from 2012 to 2019. Understory vegetation volume was measured with a 1 m × 1 m cover broad.
[a]Robles Canyon transect had low pre-treatment grazing impacts and was censored from one set of analyses for comparison.

patterns. Concomitantly, and also as predicted, densities of some bird species that depend on dense understory vegetation (eg, Yellow-breasted Chat), or that frequently use such conditions (eg, Sinaloa Wren), increased systematically across time in treatments relative to controls. Moreover, despite limited evidence of overall changes across sites, there were similar but more local biologically significant increases in densities of other understory-dependent populations of birds (eg, Common Yellowthroat, Song Sparrow) in areas where habitats improved due to reduced grazing pressure. Over relatively short time spans such as we considered here (~7 years), understory vegetation and associated bird populations can respond strikingly and positively to reduced grazing pressure.[38,44,45] For avifauna, however, responses to changes in the understory may not be fully realized unless sufficient overstory vegetation is present,[40] which was likely not an issue here except at one site (San Lazaro) where canopy cover was limited and overall responses were lowest. Assessing whether changes in focal indicators that we targeted translate to more general positive changes in riparian ecosystems is more complex,[46,47] but seems likely given the nature of these systems and general importance of understory vegetation in securing and shading soils, and promoting cooler, moister, and more protected microclimates and habitats. Regardless, patterns of ecological change we observed and the underlying social context suggest additional effort and modifications to our approach could foster greater benefits.

Densities of many bird populations that we considered increased across time as did canopy cover, but changes were often similar in treatments and controls suggesting factors other than restoration work drove these patterns. Many bird populations that increased across time depend on mid- or high-canopy resources (eg, Yellow Warbler), tree trunks (eg, Gila Woodpecker), or other resources that we did not expect to respond markedly to treatments over the relatively short time period considered. In these cases, treatments we applied can take more time to influence canopy vegetation and dependent wildlife, and patterns were likely driven by natural maturation and growth of trees mostly present at the time of baseline measurements. Many trees along the main channel at 2 sites were only ~20 years old at the start of the study having germinated following extensive flooding in late winter 1992. In these contexts, natural tree growth can be rapid when water and nutrients are available, which seemed to be the case at most sites and likely explain these patterns. Regardless, canopy cover

Table 3. Canopy cover of vegetation (%) within restoration treatments and nearby controls along 12 transects across 2 time periods in northern Sonora, Mexico, where we applied riparian restoration treatments.

SITE NAME	SITE CLASS	TRANSECT NO.	BEFORE (2012)			AFTER (2019)			DIFFERENCE, %
			NO. POINTS	MEAN	SE	NO. POINTS	MEAN	SE	
Cocospera	Treatment	1	40	40.2	5.13	42	57.5	5.26	43.3
Cocospera	Treatment	2[a]	114	78.0	2.36	94	89.3	1.38	14.6
Cocospera	Control	3.1	16	10.7	6.09	12	15.2	4.23	42.0
Cocospera	Treatment	3.1	32	4.9	2.23	20	10.4	3.30	112.9
Cocospera	Treatment	3.2	90	25.9	3.19	60	37.6	3.95	45.2
Cocospera	Treatment	4	24	52.5	6.42	20	69.5	6.87	32.5
Cocospera	Control	5	94	68.0	2.31	74	84.9	1.63	24.9
Milpillas	Control	1	62	81.1	2.56	44	89.8	2.45	10.6
Milpillas	Treatment	1	82	89.3	1.29	54	94.4	0.83	5.7
Milpillas	Control	2	202	64.7	1.70	156	84.0	1.04	29.8
San Lazaro	Control	1	154	13.6	1.71	112	20.8	2.00	52.7
San Lazaro	Control	2	170	30.5	2.08	120	42.8	2.30	40.2
San Lazaro	Control	3	46	16.1	3.49	34	38.3	4.70	137.7
San Lazaro	Treatment	3	106	22.4	2.70	82	36.8	3.13	64.5
San Lazaro	Control	4	170	26.8	2.44	128	42.5	2.88	58.4
San Lazaro	Treatment	4	38	27.4	5.06	30	37.7	4.70	37.4

Before period denotes pre-treatment measurements, whereas after period was ~6 to 7 years post-treatment. Treatments included fencing riparian vegetation to reduce warm-season grazing, erosion control structures, and occasional vegetation planting. Differences note percent change from 2012 to 2019. Canopy cover was measured with a spherical densitometer.
[a]Robles Canyon transect had low pre-treatment grazing impacts and was censored from one set of analyses for comparison.

is a useful parameter to monitor even over short periods because it allows comparisons with changes in more dynamic factors, provides insights into the drivers of change in bird communities, and is essential for long-term monitoring given its importance to bird communities. These results match some prior studies that have found greater relative increases in understory vegetation and bird species compared with changes in canopy resources and associated bird populations, at least across similar time scales.[39,41] With more time, positive changes we observed in the understory are important precursors to eventual recruitment of young seedlings and saplings to heights beyond the browsing range of livestock. Ensuring geomorphic processes required for tree germination and establishment are present, however, remains an important management challenge in these and many other riparian systems in North America.[48]

Marked increases in densities of one bird species in treatments relative to controls was likely driven by a combination of both natural range expansion and vegetation change linked to treatments. The Sinaloa Wren was rare or not present in the project area in 2012, and even rarer along Río Cocóspera in 2007 downstream of the project area where it occurred at the northern edge of its breeding range.[14,49,50] Between 2012 and 2019, this species expanded markedly at Cocóspera, was observed for the first time along the Río Santa Cruz in 2019, is expanding its range in northwest Mexico, and was recently observed for the first time in Arizona.[31]

Design and sampling issues

Various design and sampling issues likely also influenced our results and dampened overall effect sizes and responses to treatments. Although BACI designs can provide strong inference, not all treatments were randomly assigned across sites with some locations depending on landowner needs and other considerations. Thus, attributing causation to the treatments themselves is not possible. Some sites, transects, and reaches therein also varied widely in baseline conditions. Whereas BACI designs consider such variation, this likely reduced the magnitude of responses in some areas especially at Cocóspera (El Aribabi) where pre-treatment impacts were lower given years of careful management and prior conservation efforts with landowners. As such, vegetation and habitat conditions in some areas of Cocóspera (eg, in the canyon) were already at favorable levels at the start of the study, which reduced the overall

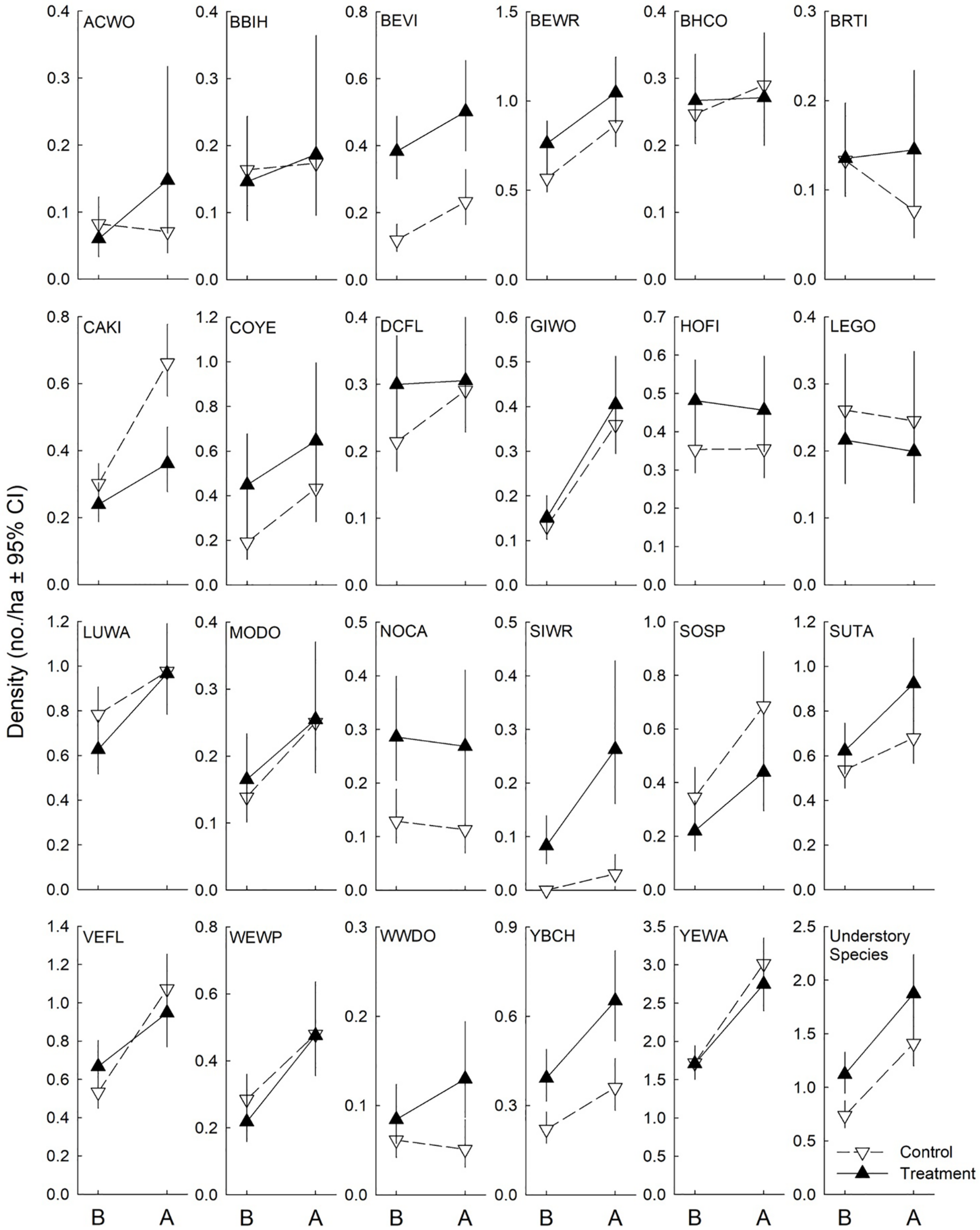

Figure 4. Effects of riparian restoration treatments on density of breeding birds across 3 sites in northern Sonora, Mexico, where we monitored treatments and controls before (B, 2012) and after (A, 2019) treatments. Points are estimates and 95% confidence intervals from distance sampling.

magnitude of responses. Some treatment and control reaches were also in fairly close proximity along the same drainages. Hence, carryover effects from nearby restoration works at broader scales, and landscape-level responses linked to increases in water and other factors linked to treatments, which can impact conditions kilometers away,[51] may have impacted controls. Although we considered nearly 6 linear kilometers of treated riparian areas, sample sizes for bird density estimation were still fairly low for many species, which reduced the precision of estimates and our ability to detect some ecologically relevant differences. Finally, baseline and post-treatment sampling were limited to only single years, and thus broad-scale environmental factors such as those that drive regional population trends of birds could explain some results. Additional time-series data from future years and more sampling will help clarify the patterns and drivers of treatment effects.

Environmental, social, and management contexts

Variation in restoration responses across space were undoubtedly influenced by a variety environmental, management, and social factors. Such factors are important considerations when

Table 4. Model results estimating the influence of riparian restoration treatments on bird species richness across 3 riparian sites in northern Sonora, Mexico, in 2012 and 2019.

PARAMETER (N)	ESTIMATE	SE	*T*	*P* VALUE
FACTOR				
Richness—Observed (317)				
Intercept	19.85	1.13	17.52	<.0001
Site Classification [Control]	−1.15	0.35	−3.29	.0011
Time Period [After]	1.29	1.13	1.14	.32
Time Period [After]*Site Classification [Control]	−0.16	0.35	−0.45	.65
Richness—Chao 1 bias corrected (317)				
Intercept	35.20	2.05	17.13	<.0001
Site Classification [Control]	−1.49	1.17	−1.27	.21
Time Period [After]	−1.66	2.05	−0.81	.46
Time Period [After]*Site Classification [Control]	0.05	1.17	0.05	.96

Estimates are least square means from linear mixed-effects models with site classification (treatment or control), time period (before or after), and site classification by time period interactions fit as fixed effects, and site and year fit as random effects, based on a before-after/control-impact design.

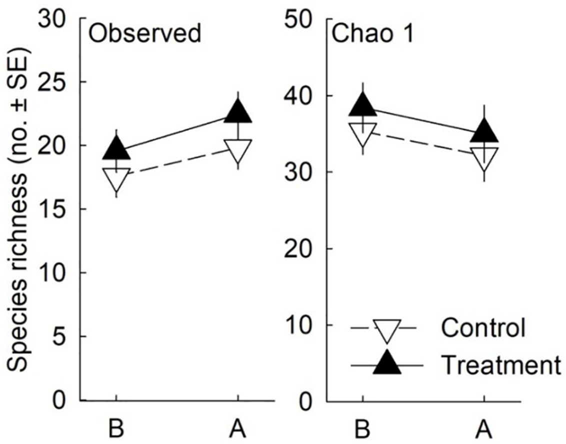

Figure 5. Effects of riparian restoration treatments on bird species richness across 3 sites in northern Sonora, Mexico, where we monitored treatments and controls before (B, 2012) and after (A, 2019) treatments. Points are least square means and SE from linear mixed-effects models.

planning and evaluating restoration efforts on private lands, but were not explicit design elements despite their relevance for improving future efforts. More marked changes at Cocóspera, which often included positive changes in controls, may have been linked to a strong conservation ethic by the Robles Elías family. This family recently established a voluntary reserve on their ranch designated by the Mexican federal government despite lack of financial support.[52] In recent years, under the direction of a younger progressive generation of ranch managers, and resources from this and other projects, most livestock have been excluded from mesic riparian areas on Rancho El Aribabi including during the cool season. Here, landowners have encouraged neighbors to be better stewards of the land, and these influences, combined with the restoration workshops and trainings we provided and nearby examples of successful restoration, may explain similar increases in vegetation structure and associated bird populations in controls. Despite similar commitments to conservation but likely less financial resources of the Rivera family at Milpillas, responses of lower magnitude were likely due mainly to environmental factors. Here, high levels of canopy cover and a gallery of old riparian trees in a deeper, more shaded, north-south facing canyon, likely discourages dense understory vegetation. Regardless, treatment reaches at Milpillas were only lightly grazed by livestock, and numerous young tree saplings had emerged following treatments. At San Lazaro, in contrast, localized failures of some fences combined with lack of strict adherence by one or more landowners to cool-season grazing undoubtedly dampened responses. Despite some positive local changes, site-specific changes in understory volume suggested post-treatment gains were offset by declines of similar magnitude in controls. These patterns may have resulted from redistribution of livestock from treatments to controls, but not overall reductions in herd sizes. Greater income demands from livestock production, less diverse income sources of residents, smaller pasture sizes, and communal ownership likely helped drive these patterns.

Conclusions

We documented some important improvements in understory vegetation and associated bird populations following restoration treatments on private and communal lands in northwest Mexico, which suggest cool-season grazing and efforts to fence and improve habitat conditions can be useful. Despite some limited positive changes in treatments relative to controls, local responses to treatments were often much more significant. Combined with underlying variation in the sites and landowners we engaged, local variation in responses suggests a number of avenues for improving future efforts. Future work here and elsewhere should consider linking restorative fencing and management agreements with small payments or other nonfinancial incentives for ecosystem services that are tied to the needs and values of landowners.[53,54] Such incentives would have likely improved results in some areas we worked while also fostering continued future habitat improvements. Although incentives can be useful, their sustainability over the long term has been questioned and hence long-term costs should be carefully weighed with costs of purchasing the most critical conservation lands. Coupling incentives with increased outreach and education tailored to the perspectives of landowners can also foster important changes in attitudes and values that reduce the needs for payments over the long term. Aside from social factors, incidental observations at sites over time suggest more focused efforts to plant and protect young trees, and water them during the early summer drought, may be critical for fostering recruitment of riparian trees especially in areas where geomorphic processes required for tree germination have been disrupted. These efforts may be especially critical for maintaining tree cover at sites such as San Lazaro where canopy cover is limited, land uses are high, and appropriately timed flooding events required for cottonwood and willow germination are rare. Broader institutional commitments for funding to ensure timely fence repairs and frequent communication with landowners outside of typical grant cycles are also essential for maintaining the continuity of efforts, but in our experience are hard to secure. Together, these and other techniques have great potential to improve the quality and quantity of riparian habitats on private and communal lands in this region of Mexico.

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Author Contributions

ADF designed the study, completed the analyses, and wrote portions of the paper linked to ecology. AE implemented outreach to landowners, restoration efforts, and wrote portions of the paper linked to social science.

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Supplemental Material

Supplemental material for this article is available online.

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