RESEARCH ARTICLE

Plant, Avian, and Butterfly Response to a Native-Grassland Restoration in Southern Texas

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ABSTACT

Non-native, invasive grasses can pose a threat to biodiversity in the southern U.S. *Pennisetum ciliare* (buffelgrass) is an example of an introduced invasive grass that has established in southwestern rangelands and negatively influenced biodiversity. Since its introduction, millions of hectares in the southwestern U.S. have been planted with, or invaded by, buffelgrass. Buffelgrass can form monocultures that not only reduce biodiversity but can also change ecosystem processes. Native-grassland restorations may be able to mitigate such negative impacts of non-native grasses. We conducted a study to document the response of herbaceous plants (grasses and forbs) and wildlife (grassland breeding birds, grassland wintering birds, and butterflies) to a 118-ha grassland restoration (involving prescribed fire, multiple discing and herbicide applications, and native-plant seeding) in La Salle County, Texas during 2013–2019. In general, we documented a numerical increase for all three taxa (native plants, birds, and butterflies) in species richness and relative abundance on the restoration site compared to a control. Our results suggest that native-grassland restoration is possible in a landscape dominated by buffelgrass. These restoration efforts can increase plant and wildlife diversity, although the time and expense required to achieve such responses are great.

Keywords: biodiversity, buffelgrass, butterfly diversity, grassland bird diversity, non-native grasses

Restoration Recap

- Large-scale sites may be able to implement treatments to reduce the abundance of non-native, invasive grass species such as buffelgrass while successfully increasing native plants.
- Native-grassland restoration may provide a means to mitigate the negative impacts that invasive, non-native plants have on biodiversity.
- However, given that vast seed sources of non-native plants exist on landscapes dominated by these plants, land managers must be committed to routine maintenance on restoration sites to sustain long-term benefits.

Invasive, non-native species have become a global conservation concern (Powell et al. 2011). Introductions of non-native species can pose serious threats to biodiversity and represent the second leading cause of species endangerment in the U.S. (Wilcove et al.1998, Simberloff 2005). In the southwestern U.S., several species of non-native grasses have invaded millions of hectares of native rangeland and have transformed previously diverse plant communities into monotypic stands of non-native grasses (Cox et al. 1988, Ibarra et al. 1995, Bock et al. 2007). One notable example of a non-native grass that has resulted in such a profound ecological impact is *Pennisetum ciliare* (hereafter buffelgrass) (Stevens and Falk 2009).

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Buffelgrass is a perennial, warm-season species that is native to Africa and Asia. Buffelgrass was introduced into the U.S. during 1917 for cattle forage and was successfully introduced into southern Texas soon after (Marshall et al. 2012). Since then, buffelgrass has spread and become dominant in many portions of the Rio Grande Plains ecoregion (Marshall et al. 2012). Like other invasive non-native grasses, buffelgrass has the ability to create monocultures and negatively impact the biodiversity of grasslands (Flanders et al. 2006, Wright 2011).

The establishment of a diverse plant community of forbs and grasses may be the basis on which successful grassland restorations depend. A general tenant of ecology is that "diversity begets diversity" (Maynard et al. 2017). If plant diversity increases following grassland restoration, then it is reasonable to expect faunal diversity also to increase. Meta-analysis of past studies indicates that diversity of arthropods, herps, birds, and mammals significantly increase with increasing plant diversity (Castagneyrol and Jactel 2011). Native-grassland restoration therefore may represent an important tool for restoring biodiversity on invaded rangelands. Past research aimed at controlling buffelgrass and increasing native plants has been conducted using small plots (< 100 m²), areas too small to document a wildlife response (Tjelmeland et al. 2008, Ruffner and Barnes 2012). Given that buffelgrass is an aggressive invader, particularly following disturbance, it is unknown whether invaded areas can be successfully restored to native-plant communities in landscapes it dominates (Tjelmeland et al. 2008, Marshall et al. 2012). The goal of our research was to evaluate whether native-grassland restoration could successfully establish a diverse herbaceous plant community within a buffelgrass-dominated area and consequently increase wildlife diversity. Specifically, our objectives were to document changes of three general taxa (herbaceous plants, birds, and butterflies) in response to a large-scale (118-ha) grassland-restoration project. We hypothesized that 1) the native-plant community on the restoration site would become more diverse and increase in cover while the native-plant community on a control site would remain constant or decrease; and 2) concomitantly, the diversity and relative abundance of fauna (birds and butterflies) would increase on the restoration site as native plants became established.

Study Sites

This study was conducted on the Hixon Ranch (La Salle County, Texas, USA) located within the western Rio Grande Plains ecoregion of Texas (Figure 1). The study included a restoration site and a control site. The restoration site was a 118-ha plot that prior to the late 1970s was used to grow vegetables and possibly was sown with buffelgrass afterwards. Prior to restoration, buffelgrass was the dominant herbaceous species along with sporadic

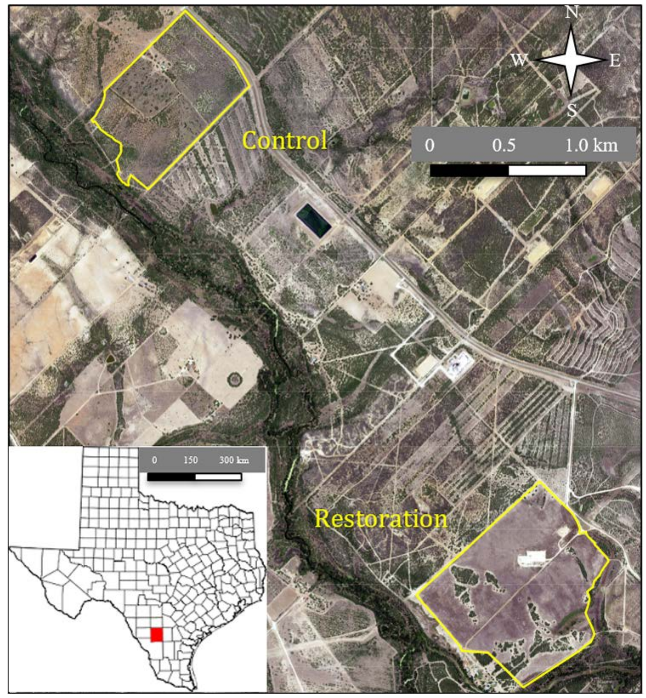

Figure 1. Restoration and control sites in relation to one another, La Salle County, Texas, 2018. The two sites are located 4.8 km apart and are in the western Rio Grande Plains of Texas. Bottom Left: La Salle County within the state of Texas, US.

areas of *Dichanthium annulatum* (Old World bluestem), with these non-native grasses comprising > 85% cover of the herbaceous community. Mottes of *Prosopis glandulosa* (honey mesquite) and *Vachellia farnesiana* (huisache) were present pre-restoration. The restoration area was dominated by Caid very fine sandy loam and Brystal very fine sandy loam soils.

The control site was a 109-ha plot that was of similar size, land history, and vegetation composition as the restoration site prior to restoration activities. The control site was an old agriculture field now dominated by buffelgrass, mixed with Old World bluestem, and contained mottes of honey mesquite and huisache. Soils in the control site consisted of Caid very fine sandy loam and Duval very fine sandy loams. This site was located 4.8 km west of the restoration site (Figure 1)

Methods

Experimental Design

This study consisted of two experimental units: a restoration site and a control site. There was only one experimental unit for each treatment resulting in lack of replication. However, replication was not practical in this study due to the relatively large-size (118-ha) of restoration and associated economic costs. In addition, the ability to have true statistical replication (i.e., areas of land with similar soil series, land history, plant composition, etc.) is virtually impossible for field studies of this size. There were no areas within 15 km with native grassland and so it was not possible to have a native reference site. Despite these limitations, we attempted to strengthen the study design by using a before-after control impact design (BACI) (Stewart-Oaten et al. 1986). Therefore, our study was a temporal comparison of data on vegetation and wildlife response pre-restoration (2013–2014) and post-restoration (2018–2019). The post-restoration period of our study consisted of data collected two years post-seeding (termed Post 1) and three years post-seeding (termed Post 2).

The restoration treatment was a labor-intensive process that involved an initial prescribed fire (February 2014) to remove herbaceous standing crop followed by repeated discing (April 2014, January 2015, May 2015, June 2015, and July 2015) and repeated herbicide applications of glyphosate (2.3 kg/ha, Bayer, Luling, Louisiana) in March 2016, June 2016, August 2016, September 2016, and October 2016 to exhaust the non-native grass seed bank. In autumn (October–November) 2016, a diverse mix of ecotypic native herbaceous plant seeds was seeded by drill seeding. We used ecological site descriptions from the USDA Natural Resource Conservation Service as an initial guide for the development of the seed mixture. We then modified this seed mixture based on various information sources: 1) a pilot study that was conducted on this ranch evaluating different types of restoration methods and seed mixtures; 2) discussions arising from a non-native grass technical group that we formed and was comprised of state and federal agency personnel, academics, landowners, and managers; 3) our own experience and knowledge of the plant communities occurring on soil series similar to those of the restoration site; and 4) availability of commercial and hand-collected seed. We used all these information sources to develop the final seed mixture by soil series (Brystal very fine sandy loam, Caid very fine sandy loam, and Cochina clay/Bookout clay loam). Seed sources primarily came from seeds harvested in Texas. Seed mixtures were sown at an average rate of 215 pure live seeds/m². The final native-seed mixture consisted of ≈ 26–27 grasses and ≈ 37–43 forbs/subshrubs depending on soil series (Table S1). In October 2017, we reseeded a few small sections of the restoration site where native plants had failed to establish using the criteria to reseed if < 5 native plants/m² occurred in areas. For a complete description of the restoration process, the reader is referred to Vreugdenhil (2019) and Gowdy (2020).

Vegetation Surveys

We conducted vegetation sampling during March, June, and October to capture both cool-season and warm-season plants (Vreugdenhil 2019). Data for the plant community consisted of pre-restoration (June and October 2013), Post 1 (March, June, and October 2018) and Post 2 data (March, June, and October 2019). Vegetation sampling consisted of 48 transects (n = 24 in the restoration site; n = 24 in the control site) that were 30-m long and permanently placed. We established transects using a stratified, random approach that allocated transects by soil series with the criteria of ≥ 1 transect per soil series. We used a coin flip to determine orientation of transects (north-south or east-west) with no more than half of transects in either direction. We sampled vegetation using a 20 × 50-cm frame placed at 0–m, 15-m, and 30-m along the right side of a transect for a total area of sampling of 7 m² (n = total 72 frames/site) (Daubenmire 1959). We identified all herbaceous and subshrub plants to species and visually estimated the percent canopy cover of each species. Canopy cover was estimated in 5% increments unless < 5% canopy cover was present, in which case we estimated canopy cover in 1% increments. A single species' percent canopy cover within the frame could not exceed 100%; however, total percent canopy cover of all species present could exceed 100% because of overlapping canopies.

Avian Surveys

We conducted avian point-transect surveys during the winter (December–January) and summer (June) to document wintering birds and breeding birds, respectively (Flanders et al. 2006). We did not conduct surveys during inclement weather (e.g., wind speeds ≥16 km/hr, cloud cover ≥ 75%, or during light rain or storms). Data for wintering birds consisted of Pre-restoration (December 2013–January 2014), Post 1 (December 2018–January 2019), and Post 2 (December 2019), whereas data for breeding birds consisted of Pre-restoration (June 2014), Post 1 (June 2018), and Post 2 (June 2019). We established survey points uniformly across the interior area of each study site using a 200 × 200 m grid that was randomly placed onto maps of the study areas using ArcGIS where each grid point represented a survey point. Because the restoration and control sites differed in size and shape, the number of survey points varied by site (n = 23 survey points in the restoration site; n = 15 survey points for the control).

We conducted avian surveys beginning at sunrise and continued for three hours (Somershoe et al. 2006). We recorded all birds heard or seen within 100-m of a point and identified individuals to species for five minutes. In addition, we categorized species as either grassland bird or shrubland bird based on vegetation-community affinity per Henehan (2016). We surveyed each point once per season (i.e., once per summer and once per winter), with sampling at each site (restoration and control) taking about 2–3 weeks to complete.

Butterfly Surveys

We conducted butterfly surveys during autumn (October) of 2014, 2018, and 2019 to encompass the general timing when the greatest number of species would be active as adults (Tracy et al. 2019). Data collection for butterflies corresponded to Pre-restoration (October 2014), Post 1 (October 2018), and Post 2 (October 2019). We used the coordinates of the avian point counts to establish transects for butterfly surveys by radiating a transect 50 m in a random, cardinal direction from each avian survey point. We conducted surveys between 0900 and 1300 hours during sunny, non-inclement weather days. While walking each transect at two km/hr, we identified to species butterflies occurring within 2.5 m of each side of the transect (Pollard and Yates 1994, Koh et al. 2002). We photographed any unknown species for subsequent identification.

Data Analysis

Data collection for two of the three taxonomic groups (avian and butterfly) occurred only once a year—breeding birds (June), butterflies (October), and wintering birds (December)—and therefore required no aggregation or averaging within a year. However, plants were sampled three times per year (March, June, and October) to capture both cool-season and warm season plants. Because we were interested in the entire plant community, we aggregated plant data across these three samplings within a given year for analysis. Using these datasets, we calculated species richness for native plants, birds, and butterflies at both the site level (total number of species for a given taxonomic group for a given site) and sampling-unit level (no. species/transect or no. species/point) by site. We also calculated mean cover for plants (% native cover/transect) and mean relative abundance of birds (no. individuals/point) and butterflies (no. individuals/transect) by site.

We used a linear mixed model with site (i.e., treatment), period, and their interactions as fixed effects, and points or transects nested within site as a random effect and time (Post 1 and Post 2) as a repeated measures effect. A Shapiro-Wilk test (1965) indicated the assumption of normality of residuals was reasonably satisfied. Butterfly and avian response variables were analyzed with a generalized linear mixed model with the same fixed and random effects specified above. We used the Pearson Chi-Square / df statistic to select between Poisson or negative binomial distributions associated with these response variables. Hypotheses about equality of means and interaction effects were tested with an F test. Because the model included two main effects (treatment and time) as well as their interaction, there were three F tests: one each for the main effects and one for the interaction. Consequently, we evaluated whether response variables differed between treatments (i.e., restoration and control) or across time (Post 1 and Post 2). All analyses used the initial measure (Pre) as a co-variable. Results were reported as mean ± standard error. We used an alpha of 0.05 to determine statistical significance. We conducted all data analyses using SAS (SAS 9.4, SAS Institute, Cary, NC, USA).

Results

Vegetation Surveys

Total species richness of native plants increased from Pre (39 species) to Post 2 (62 species) on the restoration site, whereas it only slightly increased (28 to 39 species, respectively) on the control site (Figure 2A). Mean species richness was greater in the restoration site (4.69 ± 0.27 species/transect) than the control site (1.27 ± 0.27 species/transect) ($F_{1,45} = 80.01$, $p = 0.0001$) (Figure 2B). Regarding native plant cover, we documented a treatment × time interaction ($F_{1,46} = 14.06$, $p = 0.0005$) and therefore analyzed data by period. During Post 1, the restoration site had greater mean native plant cover (45.50 ± 0.20 % cover/transect) than the control site (1.81 ± 0.20 % cover/transect) ($F_{1,45.2} = 98.8$, $p < 0.0001$). During Post 2, mean cover of native plants also was greater on the restoration site (34.72 ± 0.20 % cover/transect) than the control site (3.79 ± 0.20 % cover/transect) ($F_{1,45.3} = 50.23$, $p < 0.0001$) (Figure 2C).

Avian Surveys

Total species richness of grassland breeding birds increased from Pre (13 species) to Post 2 (18 species) on the restoration site, whereas it remained relatively constant (11–13 species) on the control site (Figure 3A). In addition, mean species richness was greater on the restoration site (3.42 ± 0.08 species/point) than the control site (1.87 ± 0.14 species/point) ($F_{1,72} = 14.08$, $p = 0.0004$) (Figure 3B). Regarding relative abundance, we documented a treatment × time interaction ($F_{1,45.61} = 8.37$, $p = 0.0058$) and consequently reported by period. During Post 1, there was no difference in mean relative abundance of grassland breeding birds between the restoration site and control site ($F_{1,67} = 0.40$, $p = 0.5316$). During Post 2, however, mean relative abundance of grassland breeding birds was greater on the restoration site (9.36 ± 0.13 individuals/point) than the control site (3.08 ± 0.12 individuals/point) ($F_{1,67} = 18.77$, $p = 0.0001$) (Figure 3C).

We documented similar responses by grassland wintering birds. Total species richness for grassland wintering birds remained relatively stable on the restoration site from Pre (12 species) to Post 2 (15 species) but decreased considerably on the control site from Pre (9 species) to Post 2 (3 species) (Figure 4A). We documented a treatment × time interaction ($F_{1,45.43} = 8.84$, $p = 0.0047$) for mean species richness of grassland wintering birds and reported by period. During Post 1, mean species richness of grassland wintering birds was greater on the restoration site (2.83 ± 0.10 species/point) than the control site (1.13 ± 0.19 species/point) ($F_{1,79.09} = 18.89$, $p < 0.0001$). During Post 2, mean species richness of grassland wintering birds also was greater on the restoration site (2.39 ± 0.10 species/point) than the control site (0.27 ± 0.38 species/point)

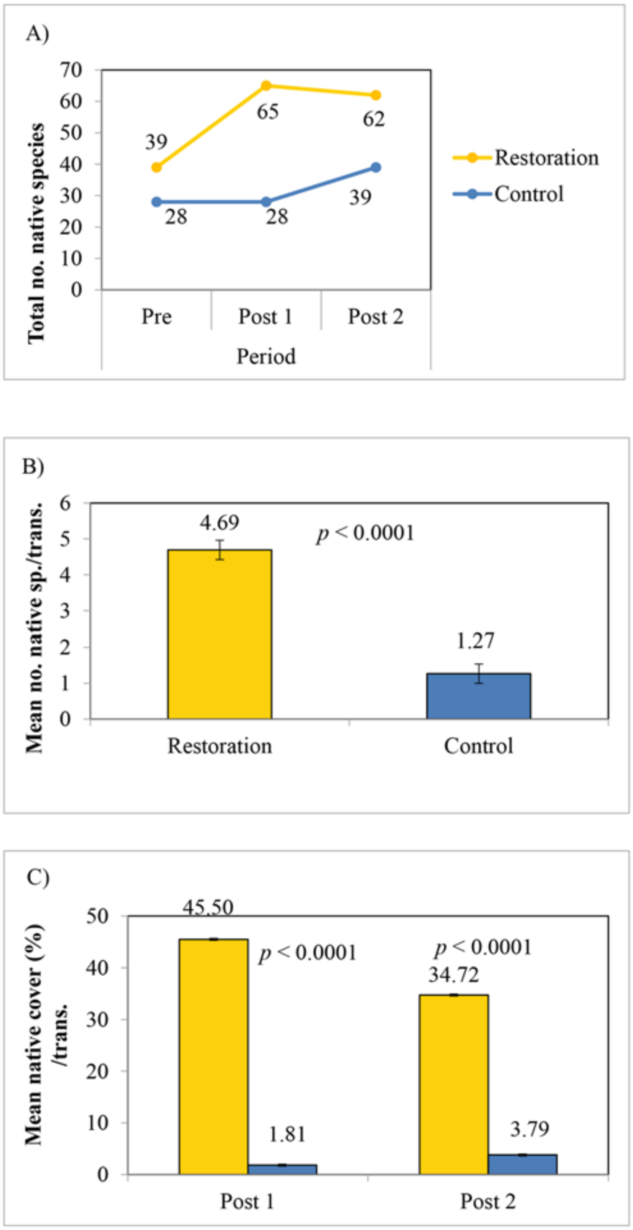

Figure 2. A) Total species richness of native plants on a restoration (yellow) and control (blue) site during Pre (2013), Post 1 (2018), and Post 2 (2019) survey periods in La Salle County, Texas, 2013–2019. B) Mean species richness of native plants (mean no. species/transect ± SE) pooled over time period (Post 1 and Post 2) on a restoration and control site. C) Mean native-plant cover (mean % cover /transect ± SE) by time period on a restoration and control site.

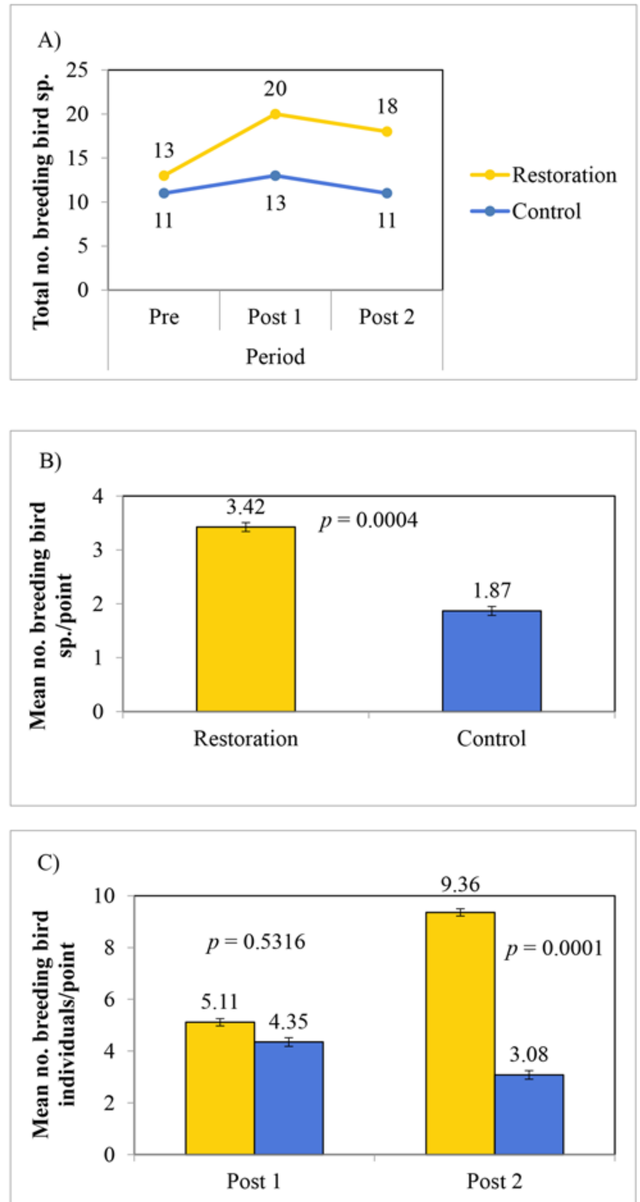

Figure 3. A) Total species richness of grassland breeding-birds on a restoration (yellow and control (blue) site during the Pre (2014), Post 1 (2018), and Post 2 (2019) survey periods in La Salle County, Texas, June 2014–2019. B) Mean species richness of grassland breeding birds (mean no. species/point ± SE) pooled over time (Post 1 and Post 2) on a restoration and control site. C) Mean relative abundance of grassland breeding birds (mean no. individuals/point ± SE) by time period on a restoration and control site.

($F_{1,71.09} = 30.13$, $p < 0.0001$) (Figure 4B). Like breeding birds, we documented greater mean relative abundance of grassland wintering birds in the restoration site (8.46 ± 0.16 individuals/point) than the control (3.03 ± 0.31 individuals/point) ($F_{1,57} = 8.72$, $p = 0.0046$) (Figure 4C).

Butterfly Surveys

Total species richness of butterflies increased from Pre (9 species) to Post 2 (14 species) on the restoration site but remained stable from Pre (7 species) to Post 2 (9 species) on the control site (Figure 5A). Mean species richness of butterflies also was greater on the restoration site (3.32 ± 0.08

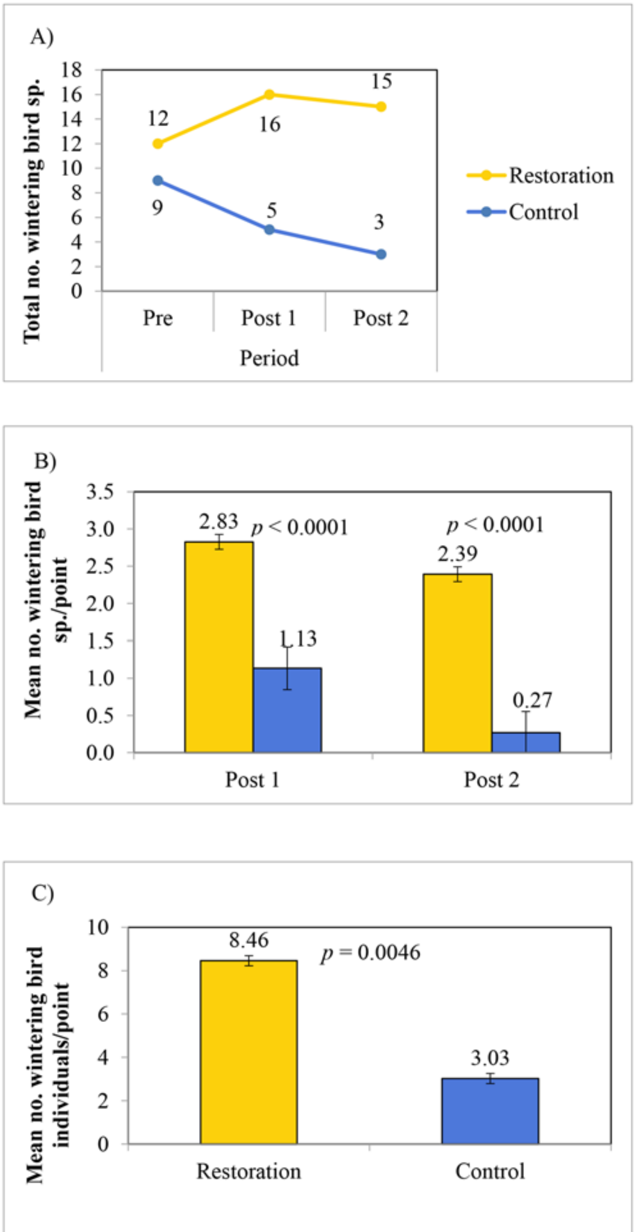

Figure 4. A) Total species richness of grassland wintering birds on a restoration (yellow) and control (blue) site during the Pre (2013), Post 1 (2018), and Post 2 (2019) survey periods in La Salle County, Texas, Dec–Jan 2013–2019. B) Mean species richness of grassland wintering birds (no. species/point ± SE) by time period on a restoration and control site. C) Mean relative abundance of grassland wintering birds (no. individuals/point ± SE) pooled over time period (Post 1 and Post 2) on a restoration and control site.

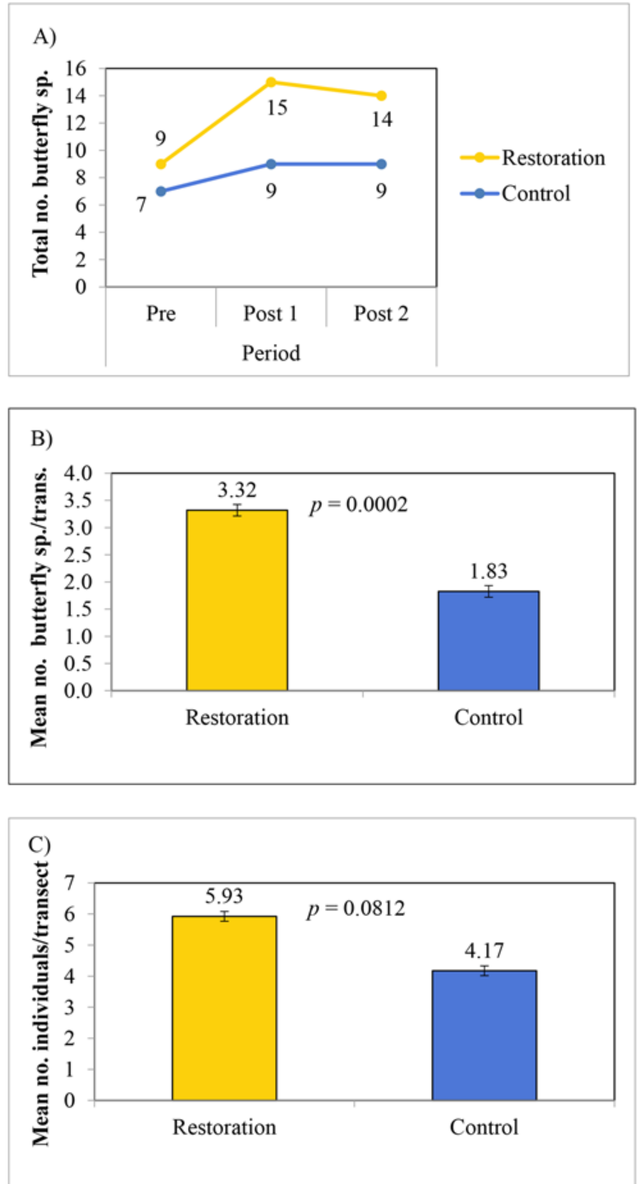

Figure 5. A) Total species richness of butterflies on a restoration (yellow) and control (blue) site during the Pre (2014), Post 1 (2018), and Post 2 (2019) survey periods in La Salle County, Texas, October 2014–2019. B) Mean species richness of butterflies (mean no. species/transect ± SE) pooled over time period (Post 1 and Post 2) on a restoration and control site. C) Mean relative abundance of butterflies (mean no. individuals/transect ± SE) pooled over time period (Post 1 and Post 2) on a restoration and control site.

species/transect) than the control site (1.83 ± 0.13 species/transect) ($F_{1,72}$ = 14.86, p = 0.0002) (Figure 5B). Mean relative abundance of butterflies was numerically greater on the restoration site (5.93 ± 0.13 individuals/transect) than the control site (4.17 ± 0.18 individuals/transect), but only approached statistical significance ($F_{1,72}$ = 3.13, p = 0.0812) (Figure 5C).

Discussion

Our hypothesis that increasing native-plant diversity would result in a positive faunal diversity response was supported. In general, by the end of the study, all taxa on the restoration site had greater species richness and relative abundance than the control site, while taxa on the control site tended to remain constant or decline. Below we elaborate on the positive response of each taxon specifically.

Vegetation

Species richness of native plants increased dramatically on the restoration site. This increase in species richness was the result of the restoration efforts because natural recolonization by native grasses and forbs would have been highly unlikely given the dominance of buffelgrass on the surrounding landscape and the restoration treatments used to deplete the seedbank. Despite the overall increase in native plants, coverage and composition of the native-plant community fluctuated over time. However, such temporal fluctuations are common in natural communities and are driven by a suite of biotic and abiotic factors such as rainfall and herbivory (McDougall and Morgan 2005). In our study, rainfall may have been a contributing factor to the observed fluctuations given that annual rainfall fluctuated considerably during the study (41–74 cm) and herbivory by domestic livestock was absent.

Past attempts of grassland restoration of buffelgrass- or Old World bluestem-dominated rangelands have yielded mixed results or only short-term (< 1 year) success. Tjelmeland et al. (2008) evaluated vegetation response in southern Texas in small plots receiving an herbicide application of glyphosate and subsequently planted with a mixture of three native grasses—*Leptochloa dubia* (green sprangletop), *Setaria leucopila* (plains bristlegrass), and *Chloris pluriflora* (four-flower trichloris). They documented that buffelgrass cover returned to pre-treatment conditions in less than one year and canopy cover of native grasses never exceeded 8% on any treatment. Ruffner and Barnes (2012) evaluated Old World bluestem response to herbicide and herbicide plus discing treatments in plots in a coastal prairie in southern Texas. They also documented that, although repeated applications of herbicide and discing produced the best results, cover of Old World bluestems recovered back to, or was greater than, pre-treatment levels after two years post treatment. Our study supports some of the findings of past research: 1) depletion of the seed bank is critical, 2) multiple applications of treatments are necessary to accomplish this and control non-native plants, and 3) follow-up maintenance treatments are required to minimize non-native grass establishment after restoration. We were able to successfully establish native-grasses up to three years post-treatment using multiple treatments of discing and herbicide to suppress buffelgrass on the restoration site and permit seeded native-plants an opportunity for establishment. In our study, we targeted the emergence of non-native seedlings after rain events using hand-sprayed herbicides or hand-pulling to minimize establishment of non-native grasses and replenishment of seed bank.

Avian Communities

We observed an increase in species richness of grassland birds (both breeding and wintering) on the restoration site. These results are similar to that of da Silva and Fontana (2020), who documented that restoration of a site previously degraded by agriculture was able to support grassland bird species. Silva and Fontana found that grassland bird species richness increased over three years on the restoration site compared to the reference area. This trend closely mirrors our results given that species richness was higher on the restored site compared to the control three years post-seeding.

Regarding species relative abundance, we also recorded an increase in both grassland breeding and wintering birds. Prior studies have reported similar findings. For example, Keyser et al. (2020) and Saalfeld et al. (2016) were able to increase abundance of breeding birds and wintering birds, respectively, in response to native-grassland restorations. The increase in native vegetation, which occurred in both studies, had an additional effect of increasing bird species richness on their respective sites. Native-plant restoration in our study increased not only native vegetation cover but also species richness of both plants and birds.

It is important to note that our avian surveys involved single visits to points and did not account for detection probability (Anderson 2001). Various factors such as weather, observer, and vegetation can affect the probability of detection and influence survey results (Buckland et al. 2015). We attempted to account for these influences by conducting surveys only during defined weather conditions and distributed over a two-week period, and the same observers conducted surveys throughout the study. In addition, vegetation was structurally similar (i.e., grassland savannah) between control and restoration sites. Although detection probability may have influenced given species, our emphasis is on general trends of increasing avian diversity and relative abundance on the restoration site relative to the control.

Butterflies

We observed an increase in butterfly diversity and abundance over time on the restoration site. This is a key finding given that native-grassland restorations recently have become an important management tool for addressing this declining taxon. Butterflies provide important ecological services such as pollination and habitat loss due to urban development and non-native species establishment is a contributing factor to the butterfly decline (Melo et al. 2018). However, an "if you build it, they will come" approach appears to work in many restorations conducted for pollinators (Rotchés-Ribolta et al. 2018). This general result may be partially due to pollinators being highly mobile and being able to locate habitat that best provides for species-specific needs (Rotchés-Ribolta et al. 2018). A positive response by pollinators to restoration has been documented in numerous studies where plant species richness increased following restoration (Winsa 2016, Breland et al. 2018, Lettow et al. 2018, Rotchés-Ribolta et al. 2018, Luong et al. 2019).

One noteworthy consideration is that, in addition to butterfly diversity increasing on restoration sites, restorations also may also serve as migration corridors for butterflies by acting as an "island of plant diversity" in a sea of non-native grasses. Restoration areas therefore could act as important stopovers for butterflies as they migrate or move long distances to more suitable areas. Shuey et al. (2016) found that a tall-grass prairie restoration in the Midwest was able to provide a habitat corridor for the endangered *Speyeria idalia* (regal fritillary butterfly) and facilitate their dispersal to other sites. The grassland restoration site in our study could be important for migratory butterfly species such as *Danaus plexippus* (monarch) due to the geographical location of the site being within major migratory routes of monarchs.

We note that, like avian surveys, butterfly surveys did not account for detection probability. However, we employed the same corrective measures (i.e., same observers, defined weather conditions for surveys, etc.) to minimize the influence of factors potentially affecting probability of detection of butterflies.

Management Implications

We documented that native-grassland restoration of a buffelgrass-dominated area was able to increase the diversity in native plants, grassland breeding birds, grassland wintering birds, and butterflies on a relatively large scale (118 ha). Our findings suggest that native-grassland restoration is possible in monocultures of buffelgrass on cultivatable soils in southern Texas and, if successful, can result in positive population responses from several wildlife taxa. Land managers must have a firm understanding of plant identification and plant-community phenology, as well as be committed to routine maintenance on any restoration project to sustain the benefits. One important aspect for land managers is the time and cost of restoration. This study involved approximately a 2-year preparation phase to remove standing crop of buffelgrass and exhaust its seed bank, 1-year phase involving seedbed preparation and awaiting appropriate environmental conditions for seeding, and a 2-year maintenance phase. Restoration costs over this 5-year period were about $1,509/ha (Vreugdenhil 2019). This high cost of restoration resulted from the multiple treatments used (prescribed fire, discing, herbicide, and native-plant seeding) as well as multiple applications of treatments (discing, herbicide, reseeding, hand-pulling) employed to decrease non-native grass cover, exhaust the non-native grass seed bank, and prevent outside influx of non-native grasses. Although scaling up restoration appears to result in economies of scale with costs/ ha decreasing with increasing size of restoration area (Powell et al. 2017), restoration costs remain high and greatly vary depending on restoration methods (Kimball et al. 2015). Consequently, although restoration efforts can increase plant and wildlife diversity in buffelgrass-dominated rangelands, the time and expense required to achieve such responses are great. When economically feasible, we recommend land managers implement native-grassland restoration as a manner of mitigating the negative effects of non-native grasses and increasing biodiversity across the landscape.

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