Community structure and seasonality of bees and flowering plants in a riparian corridor of the lower Rio Grande River in Webb County, Texas (USA)

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Abstract:

The Lower Rio Grande River (LRGR) in Texas is the physical boundary between the United States and Mexico and is considered one of the world's most at-risk rivers due to intensified management of the riparian corridor and human use. Exotic plant invasions have significantly altered the native floral communities because of invasive giant reed, with potential impacts to the native wildlife using resources in the riparian corridor. This study was conducted along a 3.22 km stretch of the LRGR in southwestern Webb County, TX to assess bee (Anthophila) communities and their flowering-plant resources among proximal and distal terrestrial upland and river-adjacent sub-corridors. Patterns related to the bee community across the two habitats consisted of low variation and dominance by common taxa, suggesting the riparian corridor could be used as a resource for bee foraging and soil-nesting. Although a lack of community structure similarities among habitats were found, indicator species analysis produced two bee genera that were more common and abundant in the upland habitat. Total number of individual bees and genera collected across 26 dates and 2 years show a bimodal trend, with peaks in March-April and September – October, with bees increasing following floral blooms primarily during the spring growing season. Findings provide a preliminary assessment of bees and flowering plants in this managed riparian corridor, but further research is needed. Conservation

efforts should include an assessment of soil and vegetation structure and their influence on native bees, considering the geographical expanse of these riparian habitats.

Introduction:

Pollinators and flowering plants are intricately linked by the mutualistic relationships that have evolved over time (Potts et al. 2010, Fiedler et al. 2012). Pollination is a vital ecosystem service provided by bees, which sustains important ecosystem services in natural ecosystems and in the production of agricultural crops (Kremen et al. 2002). It is estimated that bees pollinate over half of the world's crop varieties and are responsible for an estimated 15 billion dollars in annual revenue (Kremen et al. 2002, Losey and Vaughan 2006, Kimoto et al. 2012). In addition to managed systems and important ecosystem services highlighted by crop pollination, wild flowering plant communities are especially dependent on bees. Plant interdependence on bees is primarily to maintain seed production and species genetic variation to sustain wild plant communities (Kimoto et al. 2012).

Currently, global threats to pollinators are expected to continue if environmental stressors go unmitigated (Potts et al. 2010), with impacts to vegetation and further potential effects to ecosystem services provides by insects (Losey and Vaughan 2006). The European Honeybee, *Apis mellifera*, has been a bee pollinator receiving much attention, with managed honey bee colonies in the United States declining by over 50% in the last two decades (Ragsdale et al. 2007). Concurrent with honey bee loses reported, some native bees have become threatened because of reduced range or rarity relative to historical studies and accounts (Cameron et al. 2011). Moreover, managed bees can affect wild native bees through vector disease causing agents during foraging in flowering plants (Fürst et al. 2014). Concomitantly, anthropogenic

inputs to managed systems can affect pollinators, such as *Aspergillus flavus* in corn crops that can cause stonebrood in honey bee colonies (Klich 2007). Habitat fragmentation or invasion by non-native species is another major environmental stressor to wild systems (Potts et al. 2010). Consequently, resources such as native flowering plant communities and undisturbed areas of bare ground that support foraging and ground nesting bees can become depauperate or depleted. Overall, pollinator losses could dramatically affect ecosystem services, and therefore understanding how habitats support wild bee populations (e.g. of focal taxa) and communities remains an important area of research. This is especially critical where wild lands are affected by anthropogenic disturbances and biological invasions simultaneously. Furthermore, managing natural areas for wildlife, such as extensive riparian corridors, could be an effective strategy for conservation of local and migratory animal species, such as monarch butterflies requiring nectar during southerly annual migrations to Mexico.

The Rio Grande begins in the San Juan Mountains of Colorado and travels approximately 3,200 kilometers to drain into the Gulf of Mexico and in Texas serves as a geographical boundary between the United States and Mexico (Karatayev et al. 2012). The river and its associated riparian corridors are one of the most anthropogenically affected and yet understudied systems in the world (Karatayev et al. 2012). The river is also a primary source of drinking water and supports much of the municipal, industrial, and agricultural water needs for both nations on the U.S.-Mexico border. Regionally, the endemic flora and fauna depend on the Rio Grande's life sustaining properties; maintenance of food webs, providing refugia and habitat for animals, and a steady source of available water (Ellis et al. 2001). The Rio Grande has been critically affected by over-extraction of freshwater, pollution, invasive plant species and the compounding threat of climate change (Karatayev et al. 2012). Watershed disturbances, especially those

occurring proximal to river channels (i.e. riparian corridors) impact resources for wildlife (Fowler et al. 2018). For example, invasive plant species pose a significant threat to plant and pollinator communities along the Rio Grande in Texas (Rubio et al. 2014), but this has not been investigated. Invasive grasses such as buffelgrass (*Cenchrus ciliaris*), guinea grass (*Urochloa maxima*), and giant reed (*Arundo donax*) dominate the riparian corridor of the LRGR, which was once a native mixed grass prairie (Sands et al. 2012). Studies suggest that the rapid growth and spread of invasive grasses can have a severe negative impact on floral resources for pollinators (Fierke and Kauffman 2006, M.M.T Beater 2008, Kristine J. Brooks 2010), including those along the LRGR in Texas.

Although impacts from anthropogenic activities and invasive grasses are widespread, the LRGR riparian corridor in Southwestern Webb County, TX remains understudied. A need exists to better understand how wildlife uses the riparian corridor to better align conservation goals for target species. Little is known about the current state of flowering plant and bee communities provided by the riparian corridor, with potentially significant areal coverage of resources for nesting (i.e. sandy soil) and foraging (i.e. flowering plants). The objectives of this study were to survey the riparian and upland habitats in the LRGR and document bee and flowering plant generic and species richness, respectively, and to determine if differences in communities existed across riparian habitats (i.e. upland and riparian habitats within the riparian corridor). Information on pollinator habitat preferences, diversity and seasonality support further conservation actions and strategies for ecological restoration in this intensively managed system.

Methods:

Study Area

This project was conducted within a 3.22 km stretch along the banks of the Lower Rio Grande River (LRGR) in Southwestern Webb County, TX, (27.5013°N; 099.52697°W). The area is a steppe climate and located within a subtropical zone (NRCS 2006) with short periods of humidity (less than 5 humid months) and dry winters. The average annual temperature is 30.2°C and the average precipitation is 54.7 cm (NRCS 2006) . Typically, May, June and September are the wettest months averaging 7.26 cm of precipitation combined (NRCS 2006). The LRGR (RG) soil series primarily dominates the study area; the soil is deep, well drained, very fine sandy loam, and moderately alkaline (Sanders and Gabriel 1985). The LRGR's soil is able to sustain riparian vegetation through periods of prolonged drought due to its flood water holding capacity (Moore et al. 2016). Disturbances along the Rio Grande may create an ideal substrate to be exploited by invasive plants such as giant reed and other exotics. The unique LRGR plant communities can provide suitable habitat for many vertebrate and invertebrate species, including polyphagous beetles (Osbrink et al. 2018) and native bee communities (Henne et al. 2012).

Sampling Design and Habitats

In February 2017, twenty (10 in each habitat) 50 m-long x 1m-wide belt transects were established parallel to the Rio Grande in riparian and upland habitats for identifying extant floral diversity and sampling bee communities. In March 2017 – May 2017, 24 triplet bowl bee traps (12 in riparian zone and 12 in upland terrace zone) were placed 50 m apart within the 3.22 km sampling area (Fig. 1). The upland habitats were between 180 and 530 m from the main stem of the river, while riparian habitats were located from 50 to 130 m from the river. Sampling plots in both habitats were separated on average by 172 m. The total area representative of the sampled habitats was approximately .003 km^2, and the selected habitats were characteristic of the riparian corridor exposed to invasion by exotic invasive plants (Rubio et al. 2014) and human

management (Fowler et al. 2018). Each transect and pan trap was sampled monthly from February 2017 to May 2019.

Vegetation Sampling

To supplement transect flowering-vegetation sampling, flowering plants were also identified *in situ* around a 5 m radius of each bee pan trap cluster. Blooming plants were censused monthly within riparian and upland habitats on 25 visits. During each visit, plants with visible blooms were recorded as present, representing one individual or count of that plant species. This facilitated an analysis of the seasonality of bloom occurrences and the relationship with bee activity in both habitats and using pooled data. All unknown plant species were photographed and/or harvested for identification in the lab. Voucher specimens and digital images of flowering plants are held in the Texas A&M International Teaching Herbarium.

Bee Sampling

The goal of bee community sampling was to census the seasonal abundances and diversity of bees using the riparian corridor for foraging or other behaviors such as nesting in sandy substrate (Fellendorf et al. 2004). Bee data was biased because of our selected sampling method, towards some bee families such as the sweat bee family Halictidae being more collected using bee bowls (Hall 2016). However, this family is commonly abundant and represents a large portion of native biodiversity in the region (Wilson and Carril 2015) that could benefit from warm, sandy soils and diverse flowering vegetation (Michener 2007). Bees communities were sampled using hand collecting with aerial nets along the 50 m belt transects and using pan traps (i.e. bee bowls) (LeBuhn et al. 2016). Belt transect sampling involved visually locating floral resources and collecting bees directly from plants by sweeping nets (i.e. hunt sampling). Hunt

sampling the length of each transect was conducted in pairs (one person netting and the other recording), lasting approximately 25 minutes. Bee bowl collection structures were an adaptation of Droege et al. (2010). Bowls traps consisted of 3.5 oz cups painted three different fluorescent colors (Blue, White, and yellow) (New Horizons Entomology Services, Upper Marlboro, MD USA). Four-foot metal T-posts with metal wire utilized to secure the bowls in place for sampling. Soapy water solution (water + a few drops of ivory dish soap) was added to each container to capture pollinators. Bee bowls were set on two dates each month between 09:00 am and 011:00 am CST and bees were collected from bee bowls after 24 hours. Bees collected from hand netting and bee bowl pan traps were placed into 4oz Whirl Pak® (Nasco Fort Atkinson, WI) bags or vials containing 70% ethanol. In the laboratory, bees were identified to the level of genus using available taxonomic keys available online (Discover Life http://www.discoverlife.org/mp/20q?guide=Bee_genera and, Bugguide http://bugguide.net/node/view/8267/bgpage) and in published keys (Michener et al. 1994; Michener 2007; Wilson and Carril 2015). Voucher specimens were deposited in the insect collection in the Department of Biology and Chemistry at Texas A&M International University.

Temperature data was collected monthly using a Kestrel 5000 Environmental Meter with LINK (Kestrel Meters, Boothwyn, PA) at each bee bowl cluster (n = 24) and averaged across samples to yield one value per sampling date.

Data Analysis

Data matrices consisting of bee genera and blooming plant counts for each month across two years was created to preliminarily assess community structure regarding generic richness and abundance among the riparian and upland habitats. The row and column summary command

in PC-ORD was used to produce values for generic richness and three community diversity indices (Shannon's H, Simpson's D and Evenness). Generic richness and total number of individuals within a genus (i.e. abundance) was used to statistically compare bee and plant communities among habitats and months (see below).

Non-metric multidimensional scaling (NMDS) was used to initially compare bee communities among upland and riparian habitats. Bee genera counts were used in the main matrix to ordinate the total sample plots (n =10 from each habitat). Ordinations were conducted using raw counts of bee genera. Using the same data matrix as for NMDS, indicator species analysis (ISA) was performed to determine if any bee genera were collected in greater disproportionately from one of the two habitats. NMDS and ISA were conducted using PCORD 7.0 (Wild Blueberry Media LLC, Corvallis, OR, USA).

A mixed model with a repeated measures analysis was utilized (JMP 14, SAS Institute Inc, Cary, NC) to compare bee genera richness and abundances among the riparian and upland habitats. The model was constructed using the fixed main effects of bee generic richness and abundance, a full factorial between month and habitat (upland and riparian) and a random effect of year with nested month.

Correlation analysis (i.e. non-parametric correlation Spearman's ρ) was used to determine bivariate relationships among genera richness, total number of individuals, number of plants in bloom, and air temperature (JMP 14, SAS Institute Inc, Cary, NC). Pooled data was used (across habitats) to determine relationships of bee generic and abundance seasonality and total bloom characteristics within the study area.

Results

A total 1489 bee individuals representing 29 genera were collected across the riparian and upland habitats (Table 1). The 10 most dominant bee genera comprised 90 percent of the total number of individuals collected, with the remaining 19 bee genera comprising the remainder of the community (10 percent). *Lasioglossum* (*Dialictus*) was the genera collected most frequently (618 individuals) and was collected over three times more than the next most abundant bee (*Apis*, 212 individuals).

Across the total sampled area (i.e. upland and riparian habitats), a total of 57 flowering plants species with blooms were counted representing 24 families (Table 2). Fifty flowering plant species were counted in the riparian and 24 in the upland habitat. Sunflower (*Helianthus annuus*), Narrowleaf globemallow (*Sphaeralcea angustifolia*), and silverleaf nightshade (*Solanum elaegnifolium*) were the dominant plants comprising 20 percent of the grand total number of blooming plants observed in the study. Bee abundance and vegetation data were summarized graphically (Fig. 2), showing a higher diversity of flowering plants in riparian corridor compared to upland habitat yet with similar bee communities.

Riparian Habitat Associations

Pooled riparian and upland community data showed that bee genera richness between both habitats were not significantly different (26 and 29 bee genera, respectively). NMDS ordinations showed that bee communities at the taxonomic level of genera were similar among the upland and riparian locations and dominated by sweat bees. Two bee genera in relatively lower abundances (*Halictus* and *Ashmeadiella*) were found to be significant indicator taxa, both

showing affinity for the upland habitats. (*Ashmeadiella* IV = 55.1, P = 0.007; *Halictus* IV = 41.7, P = 0.0374). A total of four individuals were collected as singletons: *Pseudopanrugis*, *Sphecodogastra*, *Florilegus* and *Epeolus*. Three of these four genera occurred in the riparian habitat, while one (*Pseudopanrugis*) occurred in the upland habitat but because of rarity in our sample they are not indicator taxa of one of the two habitats. The top two dominant bee genera regarding abundances (*Dialictus* and *Apis*) accounted for 56 percent of the total number of individual bees collected. Bee community evenness was very similar in riparian (0.65) and upland communities (0.66) (Table 1). Shannon diversity was not significantly different among riparian (2.13) and upland (2.10) communities, and the Simpson diversity index showed slight differences between riparian (.80) and upland (.76) communities but without significant differences detected among these habitats (Table 1).

Among 57 flowering plant species recorded, 7 were found only in upland and 35 were found only in riparian habitats (yet bee communities were generally similar). A total of 17 plant species occurred in both riparian and upland habitats, with most of these plants producing blooms attractive to specialist or generalist pollinators. A total of six plant species were most frequently encountered occurring in over 50 percent of sample plots and blooms of these species persisted an average of four months across all years in the current study: silverleaf nightshade (*Solanum elaeagnifolium*) (April-September), common sunflower (*Helianthus annuus*) (April-September), narrowleaf globemallow (*Sphaeralcea angustifolia*) (March-July), annual sowthistle (*Sonchus oleraceus*) (February-April), Texas vervain (*Verbena officinalis*)(February-April), and cowpen daisy (*Verbesina encelioides*) (March-May). Blooming plant community evenness differed significantly between riparian (0.94) and upland communities (0.88) (Table 2). Shannon diversity was higher in riparian (3.66) than upland (2.79) plant communities, and Simpson

diversity index showed slight differences between riparian (0.96) and upland (0.91) communities (Table 2).

Seasonal Bee Communities

Analysis of pooled bee abundance data showed no significant difference in abundance between years of data collection. However, effect tests in our statistical model showed a significant difference for monthly abundance, across all sites and years (DF = 11; F Ratio = 4.9134; P = 0.0048) ($\alpha = 0.05$). Least-squares means plots showed three peaks of higher bee abundance in the months of March (0.0170), April (P = 0.0001) and September (P = 0.0139) ($\alpha = 0.05$) (Fig. 3 a). The two most abundant bees showed clear peaks, with *Apis* had the highest peak abundances in March and April and *Lasioglossum* (*Dialictus)* in September.

Analysis of pooled community data showed no significant difference in genera richness between years of data collection. Effect tests in our statistical model showed a significant difference for monthly genera richness, across all sites and years (DF = 11; F Ratio = 2.8308; P = 0.0473) ($\alpha = 0.05$). Least-squares means plots shows a bimodal trend of increasing genera richness in the months of April (P = 0.0115) (averaged 17 genera) and September (averaged 20 genera) (P = 0.0162) ($\alpha = 0.05$) (Fig. 3 b).

Bee Community, Bloom and Temperature Relationships

Due to non-significant differences between the years of data collection, plant, bee and environmental data was pooled across years prior to analysis. There was a strong positive correlation between bee genera and blooming plant richness, which was highly statistically significant, ($r_s = 0.7964$; $P < 0.0001$) ($\alpha = 0.05$) (Fig. 4 a). Similarly, bee abundance was positively correlated with blooming plant richness and was statistically significant, ($r_s = 0.6260$; P

= 0.0006) (α = 0.05) (Fig. 4 b). Average monthly temperature did not have a statistically significant effect on bee genera richness or bee abundance; however, both abundance and richness generally decreased at temperatures below 15°C and above 30°C which was highly evident in the months of April and September across years (Fig. 4 c-d). Bee abundance was strongly associated with genera richness and was statistically significant (r_s = 0.5821; P = 0.0018) (α = 0.05). A plot of bee abundance by genera was best represented by a log-linear model, which showed bee abundance increasing logarithmically with increasing genera (Fig. 5).

Discussion

Riparian areas and associated watersheds are extremely vulnerable to natural and human caused disturbances. The intensity of such disturbances can often be associated with irreversible changes in these landscapes, usually resulting in loss of biodiversity and overall ecological identity and function. Along the Lower Rio Grande River (LRGR) near Laredo, TX (including our study area) urbanization, over extraction of freshwater, and invasive plant species continue to threaten ecological communities. Recently, the Rio Grande river has been at the forefront of a growing socio-political issue. The United States government has secured funding to build a border wall along the Rio Grande between the U.S. and Mexico (Fowler et al. 2018). The LRGR in Texas will be the most impacted by the construction since there are approximately 2000 km of border between countries (Fowler et al. 2018). There is little or no data that can serve as a baseline to determine the effects of ongoing ecological change in this important riparian corridor. The sampling area is unique in that there are overlapping community structures reminiscent of both riparian forest, upland scrub habitats, and urban landscapes. Our study shows that diverse native bees are utilizing riparian habitat resources and if current trend of disturbances continue, this could have a significant impact on extant bee communities. To date, the current study

provides the only account of bee diversity and flowering plant diversity for this important habitat and freshwater resource in the region.

Among the dominant genera of bees collected in the study, *Lasioglossum (Dialictus)* was most abundant. They are ground nesting bees and can have an array of social behaviors that range from strictly solitary to parasitic. The high number of collected *Lasioglossum (Dialictus)* may be attributed to our primary sampling method (bee bowls) that has been show to bias samples towards higher relative abundances for bees in the family Halictidae (Roulston et al. 2007). Bees in the genus *Diadasia* (Tribe Emphorini) are small to medium sized hairy bees that range in size from 5-20 mm (Michener 2007). The bees encountered from this genus were observed mainly foraging on narrowleaf globemallow (*Sphaeralcea angustifolia*) which was a common plant present in both habitat types. Many of the bees in this genus are foraging specialists and make shallow nests often with tubular entrances around the opening (Michener 2007). The genus *Melissodes* (Tribe Eucerini) are medium to large bodied bees 7.5-16 mm (Michener 2007). Many of the *Mellisodes* collected were in early - mid fall (September-October) which is characteristic of this genus (Wilson and Carril 2015). The bees in this genus are specialists that primarily forage on flowers of the family Asteraceae, but few may be generalists (Michener 2007). All *Melissodes* are ground nesting solitary bees (Michener 2007). Unexpectedly, September showed are large spike bee genera and abundance although blooming plants remained low. Upon further investigation, blooming invasive and weedy plant san miguelito vine (*Antigonon leptopus*) was found growing within the riparian habitat along with other dominant flowering plant species. The combined effects of late blooming common sunflower, silverleaf nightshade and presence of san miguelito vine provide resources for late season bees such as some *Melissodes*.

There was a clear lack of heterogeneity between upland and riparian zones sampled in the study. The NMDS procedure failed to find a solution that was associated with habitat differences, and this was probably because of the overlap of the dominant taxa in the two habitats. Only genera *Halictus* and *Ashmeadiella* showed a strong affinity for upland habitats. *Halictus* are common bees, medium bodied and foraging generalists (Michener 2007). *Ashmeadiella* are small bees that have an affinity for drier environments and can be both generalist and specialist foragers (Wilson and Carril 2015). Some *Ashmeadiella* are known to forage on mesquite flowers which in our study was only encountered on the upland sites (Wilson and Carril 2015). The upland site was patchier than the riparian habitat and had a higher invasive buffelgrass cover, which is a weedy grass that grows in large aggregate groups. Invasion by grasses like buffelgrass and giant reed, have severely fragmented the landscapes and decreased floral resources available to foraging bees (Everaars et al. 2018). Grasses are mostly wind pollinated or propagate vegetatively and don't provide resources for bees. In addition to patchy vegetation, upland sites also had more bare ground that bees could have used as nesting habitat. Riparian habitats recorded two times more flowering plant species than upland habitats, which likely stimulated upland bees to forage in the riparian zone. This is further supported by distances between the two zones which averaged only 172 m, which might not have been enough spatial distance to present differences. Consequently, the proximity of both habitats created overlap of similar plant communities in which would be within bee foraging range. In a study conducted by Gathmann and Tscharntke (2002) showed that bees averaged 150 – 600 m of foraging distance between nesting sites and floral resources, which comparatively is well within our distance measure between habitats. Other covariables that drive distances between habitats,

elevation and distance to river, likely in part drive soil and plant differences in riparian and upland sites but did not significantly affect bee genera richness and abundance.

Overall, results show high similarities among habitats and dominant, soil-nesting bees in both habitats. From a conservation standpoint, singletons (4 bee genera) and those occurring as rare (e.g. fewer than 5 individuals) could warrant further study, while the dominant taxa are apparently using the river corridor as a habitat and foraging resource. The succession of invasion, primarily as a result of giant reed grass and riparian disturbance, is a dynamic process. As giant reed grass continues to spread and create large monotypic stands, floral diversity and potential pollinator/bee resources may decline (Herrera and Dudley 2003). Consequently, this may cause extirpation of rare species from the riparian corridor. Furthermore, investigating how disturbances affect soil nesting native bees would advance our understanding of bee biology in a unique riparian community. Further, ecological restoration involving native plants could assist in management of invasive reed grass, coupled with other benefits from intensified riparian management involving giant reed (Patiño et al. 2018). Ecological restoration towards native vegetation could support initiative for management of the corridor by reestablishing native vegetation to replace dense stands of giant reed.

Seasonally, bee communities can vary significantly over time, largely depending on the availability of floral resources, seasonal phenology and environmental factors (Kimoto et al. 2012). Bee genera and abundance showed a high seasonal/monthly variation but conversely not significant inter-annual differences. Lack of significant inter-annual differences in bee diversity may be in part due to the regions relatively consistent subtropical climate, which in turn may develop patterns in bee behavior (Boucek et al. 2016). Genera across months and years, were significantly different with April and September having the greatest richness. Similarly,

abundance across months and years was significantly different with March, April and September having the highest bee abundance which may be largely attributed to floral availability and temperature (Classen et al. 2015). Kimoto et al. (2012) showed similar trends in their study where during the spring growing season had the highest bee activity which was also strongly associated with available floral resources and average monthly temperature. In our study temperature extremes negatively affected bee behavior since the data showed both abundance and richness decreased at temperatures below 15°C and above 30°C. To support this blooming plants, bee genera and abundance are strongly associated in the months of April and September (across years) which show a temperature range of 25°C -30°C (Fig. 4 c-d). Temperature extremes could have limited bee access to floral resources although they were abundant. Seasonal rainfall that may have provided significant information of associations with study variables like genera richness, bee abundance, and blooming plant counts was not measured.

Conclusion

Along a narrow two mile stretch of the Lower Rio Grande River (LRGR) we recorded previously undocumented bee and flowering plant communities, which supports further studies and conservation actions involving this important river and its riparian corridor. How wild and native bees use this habitat remains an important area of investigation, especially considering intensified management in the riparian corridor. The community approach and findings of the current study show diverse bees using resources provided in this variable habitat, while the diversity and areal coverage of flowering plant communities in the riparian are likely affected by competition from highly invasive plants such as giant reed grass. These environmental flow-mediated habitats are facing additional severe threats from anthropogenic activity and invasive plant species. The flowering plant communities, soil structure (i.e. affecting bee nesting) and bee

communities could serve as biological targets for ecological restoration conducted in this intensively managed riparian corridor.

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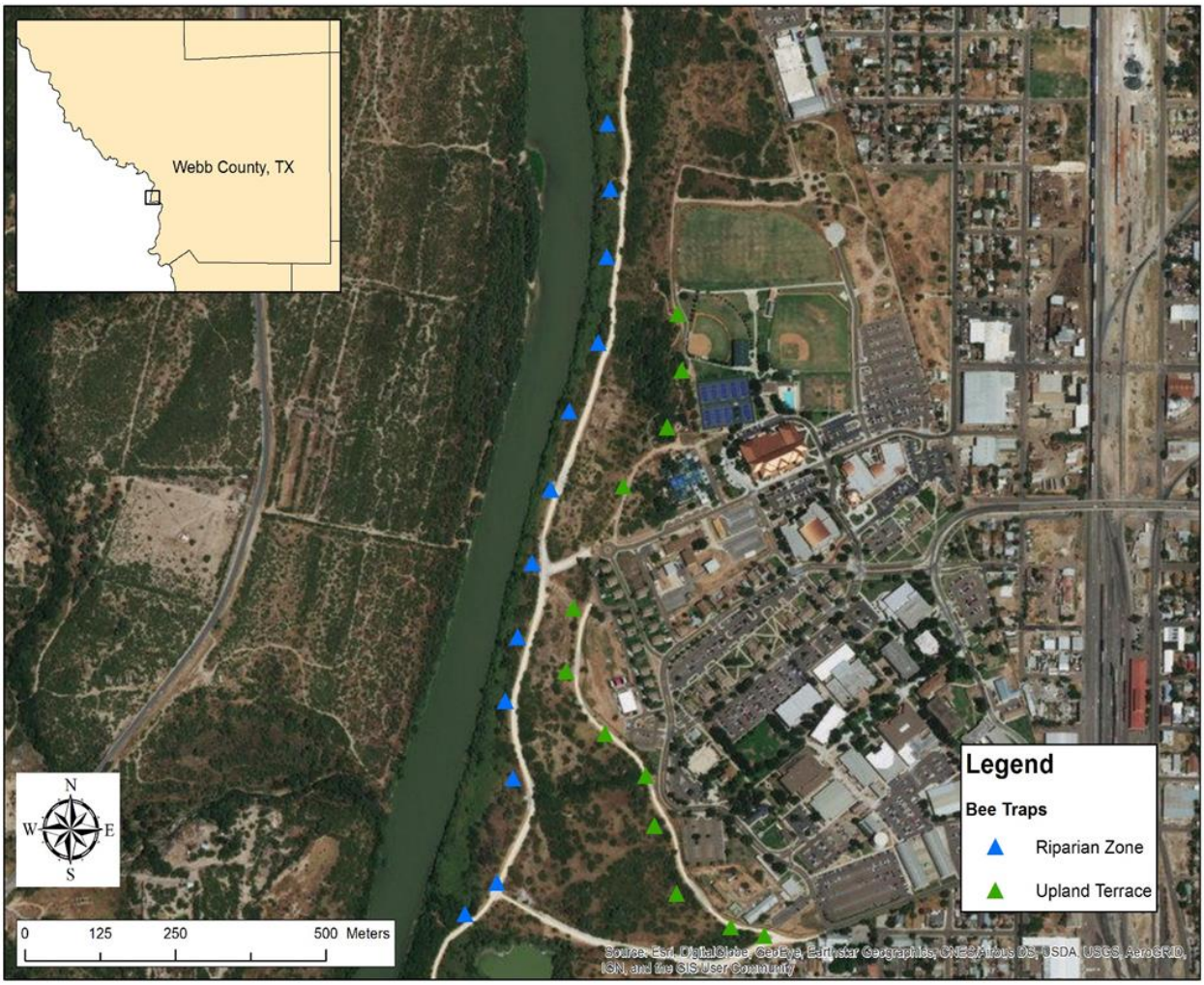
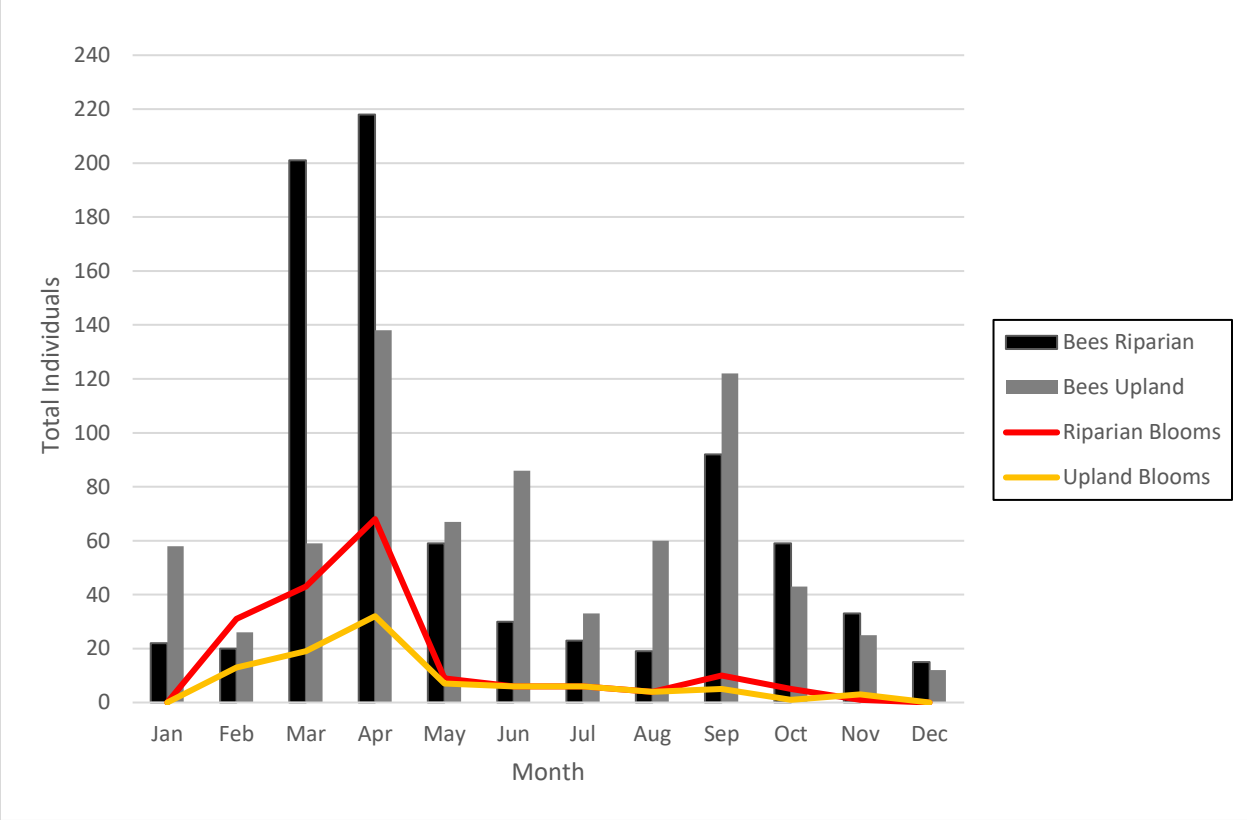


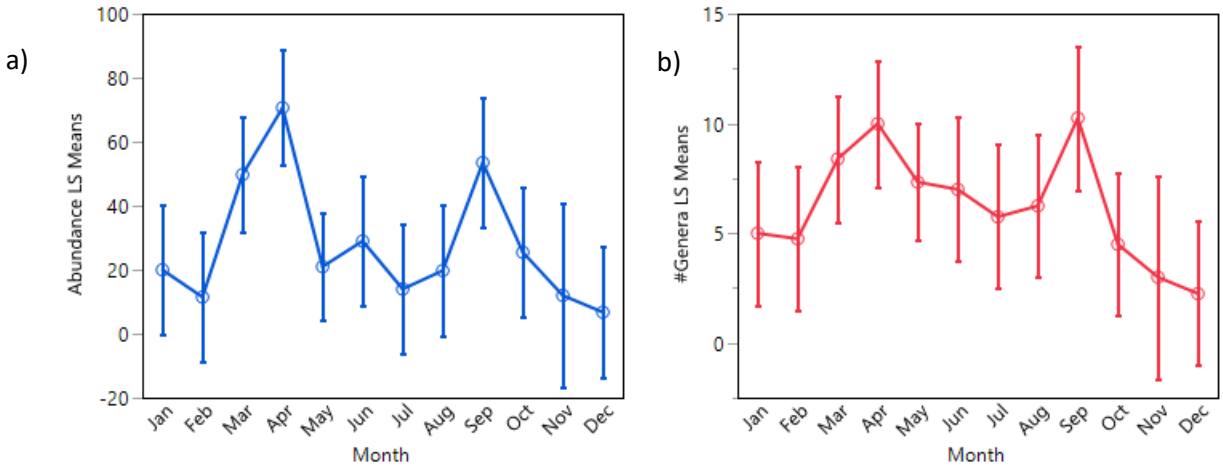
Figure 1. Map of the study area with bee bowl trap locations in riparian and upland habitats (triangles) Locations of vegetation transects for flowering plant surveys (not shown) are within the extent of this sampling area and included habitats within 5 m surrounding bee bowl locations.

Figure 2



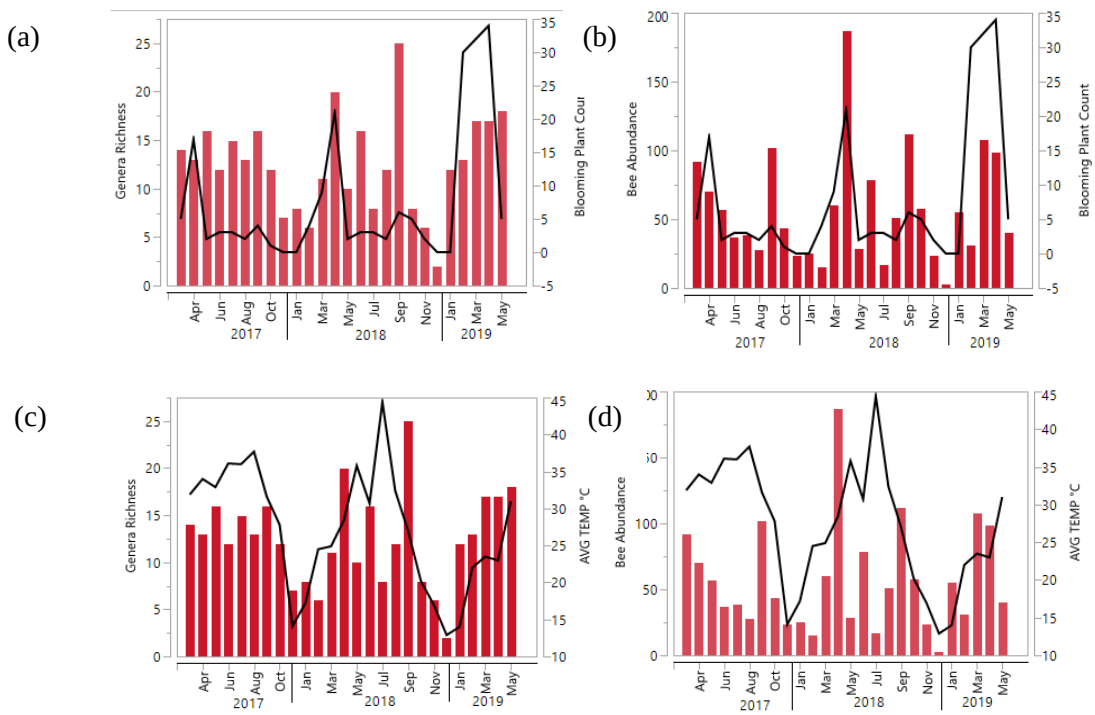
Comparison of total generic bee abundances and total bloom counts in upland and riparian habitats during the study period. Data has been pooled by years and shown as totals by month. Generic bee abundances were significantly correlated to total bloom counts (r_s= 0.6260; P = 0.0006) (α = 0.05).

Figure 3



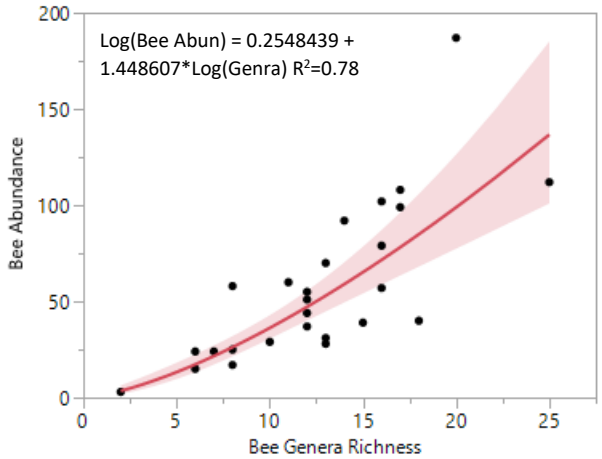
(a) Interaction plot of pooled bee abundance by month (years combined) (a), and interaction plot of pooled genera richness by month (years combined) (b). Error bars represent 95% confidence intervals.

Figure 4



Interaction plot of blooming plant richness (line) and bee genera richness (bars) (a) and bee abundance (bars) (b). Interaction plot of monthly average temperature (line) and genera richness (bars) (c) and bee abundance (bars) (d).

Figure 5



Relationship of pooled bee abundance and genera richness showing a log linear interaction. Shading around fit line shows the 95% confidence intervals.

Table 1. Bee genera collected during the study period and summary of diversity indices in riparian and upland sites.

Genus	Riparian	Upland	Total
Agapostemon	45	40	85
Ancyloscelis	8	16	24
Andrena	2	12	14
Anthophora	40	25	65
Anthophorula	3		3
Apis	140	72	212
Ashmeadiella	1	17	18
Augochlorella	3		3
Augochloropsis	15	9	24
Calliopsis	6	2	8
Centris		2	2
Ceratina	27	20	47
Diadasia	65	38	103
Dialictus	289	329	618
Epeolus	1		1
Eucera	4	3	7
Florilegus	1		1
Halictus		7	7
Lasioglossum	28	31	59
Lithurgus	5	2	7
Megachile	4	2	6
Melissodes	53	55	108
Osmia	5	8	13
Perdita	1	1	2
Pseudopanurgus		1	1
Sphecodogastra	1		1
Svastra	1	2	3
Vespidae	18	20	38
Xylocopa	4	5	9
Total Richness	26	24	Grand Total 1489
Shannon	2.14	2.10	
Simpson	0.805	0.763	
Shannon Evenness	0.656	0.661	

Table 2. Blooming flowering plants and summary of diversity indices in riparian and upland sites.

Species	Riparian	Upland	Total
Acacia rigidula		1	1
Aloysia gratissima	4		4
Antigonon leptopus	1		1
Aphanostephus ramosissimus	3	3	6
Argemone sanguinea	5		5
Astragalus brazoensis		1	1
Brassica juncea	3		3
Chromolaena odorata	5	4	9
Ciclospermum leptophyllum	2		2
Cirsum texanum	3		3
Conyza canadensis	3		3
Croton ciliatoglanduliferus		1	1
Descurainia pinnata	4		4
Ehretia anacua		3	3
Funastrum clausum	1	1	2
Gaillardia pulchella	2		2
Galium aparine	2		2
Gamochaeta pensilvanica	6		6
Gaura parviflora	5		5
Glandularia quadrangulata	3	3	6
Heart Leaf Hibiscus	1		1
Helenium microcephalum	3		3
Helianthus annuus	14	14	28
Lactuca serriola	3	3	6
Lamium amplexicaule	1	1	2
Lantana camara	3		3
Lepidum viginicum	6	5	11
Leucosyris spinosa	2		2
Malva parviflora	2	2	4
Maurandella antirrhiniflora	3		3
Melilotus indicus	2		2
Monarda punctata	5		5
Morus rubra	1		1
Nama hispidum		4	4
Neptunia spp.	2		2
Oenothera speciosa	2	2	4
Oxalis stricta	2		2
Parietaria pennsylvanica	2		2

Table 2. Blooming flowering plants and summary of diversity indices in riparian and upland sites.

Species	Riparian	Upland	Total
Parkinsonia aculeata	3	3	6
Plantago rhodosperma	2	2	4
Prosopis glandulosa		1	1
Ratibida columnifera	3		3
Rubus riograndis	3		3
Ruellia simplex	2		2
Sibara virginica	3		3
Sisymbrium irio	4		4
Solanum americanum	1		1
Solanum elaegnifolium	15	15	30
Solanum triquetrum	3		3
Sonchus oleraceus	7	6	13
Sphaeralcea angustifolia	11	11	22
Teucrium cubense	2		2
Vachelia farnesiana		2	2
Verbena officianalis	8	8	16
Verbena plicata	1	1	2
Verbesina encelioides	6		6
Vicia ludoviciana	5		5
Total Richness	50	24	Grand Total 282
Shannon	3.66	2.79	
Simpson	0.967	0.918	
Shannon Evenness	0.94	0.88	