Photographic Monitoring of Defoliation by the Tamarisk Beetle

Middle Rio Grande from Belen to Elephant Butte Reservoir, New Mexico

July 2018, LFCC-02: Direct impacts of defoliation

July 2019: Carry-over effects of 2018 defoliation on vegetation health

Mission Statements

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The mission of the Bureau of Reclamation is to manage, develop, and protect water and related resources in an environmentally and economically sound manner in the interest of the American public.

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Executive Summary

Bureau of Reclamation personnel conducted photographic monitoring of riparian vegetation within occupied Southwestern Willow Flycatcher (*Empidonax traillii extimus;* SWFL) habitat along the Middle Rio Grande, New Mexico from 2015 through 2019. Photographic monitoring was conducted in an effort to quantify the effects of defoliation by the introduced tamarisk beetle (*Diorhabda* spp.). Monitoring consisted of photographs taken at fixed locations within the riparian corridor 1) facing upward into the vegetation overstory (hemispherical photography) and 2) facing inward at the vegetation community (landscape photography). Monitoring was conducted at 52 hemispherical photo stations and 27 landscape photo stations across 6 study sites occupied by SWFLs between Belen, NM and Elephant Butte Dam. Photographs were taken between May and August in all years of the study, in order to correspond with tamarisk beetle activity and the Southwestern Willow Flycatcher breeding season. Additionally, hemispherical photographs were taken during the 2017 through 2019 winter dormant seasons in order to distinguish woody growth from foliage.

The tamarisk beetle was not detected at any of the photo stations in 2015, and only at one study site in 2016. Thus, 2015 and most of 2016 photography provided baseline data on vegetation structure and canopy closure prior to beetle-induced tamarisk (*Tamarix* spp.) defoliation. Severe defoliation was recorded in two study sites in 2017 and in 4 study sites in 2018. Only a small amount of defoliation was observed at one study site in 2019. Despite the lack of direct impact in 2019, landscape and hemispherical photography found an overall decline in vegetation health and canopy cover in some sites as a persistent result of multiple previous years of defoliation. Although canopy cover remained within the apparent range of natural variation observed in the study, the decline was sufficient to result in increased temperature and aridity of breeding habitat. Continued photographic monitoring is recommended as the severity and extent of defoliation by the tamarisk beetle is expected to increase in coming years.

Introduction

Tamarisk (*Tamarix* spp.; aka saltcedar) is a large woody shrub that was introduced to the United States (U.S.) in the early 1800s for erosion control and horticulture. The species spread rapidly in the 1900s, expanding from 4,000 hectares (ha) in 1920 to more than 500,000 ha by 1970, and is now a dominant plant throughout riparian areas in the southwestern U.S. (Neill 1985, Gay and Fritschen 1979). Tamarisk is highly drought- and salt-tolerant, and the species' rapid expansion in the early 20th century is attributed primarily to the alteration of natural, dynamic river flows following construction of large dams and water diversion projects in the western U.S. (Di Tomaso 1998, Everitt 1998). The subsequent drying and salinization of riparian ecosystems, in addition to other disturbances such as grazing and reduced recruitment of native vegetation, has created conditions favorable for rapid colonization by tamarisk (Di Tomaso 1998). Tamarisk has a deep tap root, reaches four to eight meters in height, and frequently expands into dense monotypic stands to the exclusion of native vegetation.

The decline of native riparian vegetation and the corresponding loss of biodiversity resulting from tamarisk invasion are two of the factors that prompted the initiation of efforts to eradicate the species (Di Tomaso 1998, Shafroth et al. 2005). Additionally, tamarisk was reported to cause streamflow depletion, increased soil salinization, increased fire severity, and contribute to the degradation of wildlife habitat (Johnson 1987, Di Tomasso 1998, Shafroth et al. 2005). However, traditional methods to remove tamarisk, such as herbicides, fire, or mechanical treatment, were only marginally successful and often negatively impacted the riparian ecosystem (Shafroth et al. 2005, Harms and Hiebert 2006).

The search for an economical and effective method of controlling and eradicating tamarisk eventually led to the initiation of a biological control program. The tamarisk beetle (*Diorhabda* spp.) defoliates tamarisk repeatedly over multiple growing seasons, eliminating the plant's ability to photosynthesize. This repeated defoliation can eventually lead to plant mortality. The U.S. Department of Agriculture (USDA) approved the release of the exotic tamarisk beetle in 2001, after laboratory and field testing concluded that the beetle would only defoliate tamarisk and that dispersal would be limited to only 1-2 kilometers (km) per year (yr) (DeLoach and Tracy 1997, Tracy and Robins 2009). After an initial caged release, the tamarisk beetle was introduced in multiple watersheds throughout the southwestern United States.

Adult tamarisk beetles emerge from diapause in the spring and immediately begin feeding and mating. Females lay 10 to 20 eggs per day on tamarisk foliage, with total production ranging from 300 to 500 eggs per female. Eggs hatch in approximately seven days, and then go through three instar larval stages of approximately four to seven days each. Upon completion of the third instar stage, the larvae drop to the ground where they enter a 7 to 10 day pupal stage and then emerge as adult beetles. Adults live for two to four weeks and typically produce two to five generations per year. All larval and adult stages feed on tamarisk foliage. Adults drop to the ground and burrow into the soil or leaf litter in autumn, enter diapause, and overwinter there (DeLoach et al. 2003; Lewis et al. 2003).

The tamarisk beetle was indeed highly successful at defoliating tamarisk and had limited impact on non-target plant species (Moran et al. 2009). However, the tamarisk beetle spread through riparian systems far more rapidly than predicted, resulting in defoliation of large expanses of tamarisk (Dudley and Bean 2012). While this was effective for tamarisk control, there were unforeseen ecological consequences. The pre-release projections of tamarisk beetle dispersal rates led to the belief that native vegetation would replace tamarisk as the beetle slowly defoliated one area and moved along the riparian corridor. Instead,

in many instances altered hydrology prevents the immediate re-establishment of native vegetation, or the tamarisk beetle simply defoliates tamarisk much more quickly than native vegetation can regenerate (Paxton et al. 2011). This rapid defoliation of tamarisk without replacement by native vegetation has negatively impacted riparian breeding birds, which are left with drastically reduced or absent nesting habitat, particularly in areas characterized by monotypic tamarisk stands.

The Southwestern Willow Flycatcher (*Empidonax traillii extimus;* SWFL) is one of the most notable avian species to be negatively impacted by tamarisk beetle defoliation, and concern for the species' welfare has led to the cessation of tamarisk beetle releases in many areas (Hultine et al. 2010; Dudley and Bean 2012). Moreover, in August 2017 the U. S. District Court of Nevada ruled that the USDA tamarisk beetle release program is in violation of the Endangered Species Act for its failure to create a concurrent conservation program for the SWFL. The Southwestern Willow Flycatcher is a State- and Federally-listed endangered subspecies of the Willow Flycatcher (*Empidonax traillii*). It is an insectivorous, Neotropical migrant that nests in dense riparian vegetation in the southwestern United States. SWFLs commonly nest in tamarisk, and some SWFL breeding habitat is composed predominately or exclusively of tamarisk. SWFLs typically arrive on their breeding grounds between early May and early June; between late July and mid-August they depart for wintering areas in Mexico, Central America, and northern South America (Sogge et al. 1997, USFWS 2002). The SWFL's brief breeding season coincides with the peak of tamarisk defoliation by the tamarisk beetle.

In October 2005, the U.S. Fish and Wildlife Service (USFWS) designated Critical Habitat for the SWFL along the Middle Rio Grande between the Isleta Pueblo and Elephant Butte Reservoir (USFWS 2005). The designation was updated in January of 2013 to include the Sevilleta and Bosque del Apache National Wildlife Refuges and a portion of the Elephant Butte Reservoir conservation pool. No critical habitat was designated downstream of Elephant Butte Dam (USFWS 2013), although the SWFL does breed there. The Rio Grande currently supports one of the largest breeding populations of SWFLs in the United States.

Although the tamarisk beetle was not released on the Rio Grande in New Mexico, it was released on the Rio Grande in Texas, the Pecos River in New Mexico, as well as on the San Juan River in Colorado. It is likely tamarisk beetles dispersing from these locations are now being detected along the Rio Grande in New Mexico. Beginning in 2016, the beetle was detected throughout the river corridor from north of Albuquerque, NM to Texas, following several years of intermittent tamarisk beetle detections along the Rio Grande in New Mexico. Currently, all four tamarisk beetle sub-species have colonized New Mexico, although only one sub-species was intentionally released in the state (Figure 1). Although the beetle was detected at sample locations throughout the Rio Grande in 2016 through 2018, the population still consisted of multiple, disconnected patches and was not found in all tamarisk continuously throughout the Rio Grande.
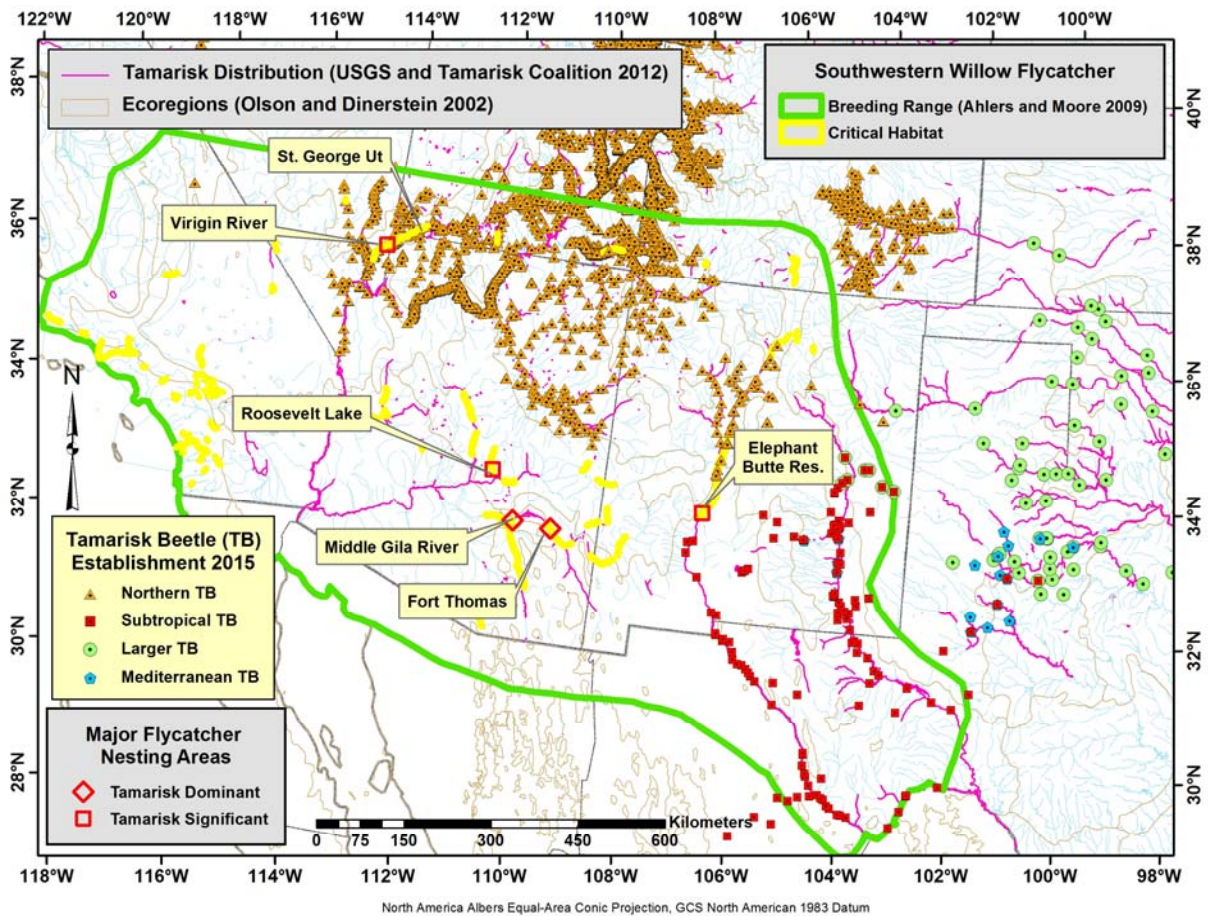
Figure 1. Distribution of the tamarisk beetle by species in 2015 (Tracy 2017).

Reclamation began a photographic monitoring study of the impacts of the tamarisk beetle on SWFL habitat in the Middle Rio Grande in 2015. The objectives of the study were to monitor and quantify defoliation by the tamarisk beetle over time, and the effects of that defoliation on canopy closure and vegetation composition in SWFL breeding habitat.

Methods

Study Area

Eighteen hemispherical photography stations and 10 landscape photography stations were established at two occupied SWFL study sites in the Middle Rio Grande in 2015, one site in the Belen Reach (BL-10) and one site in the San Marcial Reach (DL-12N; Figures 2 and 3). The study was expanded in 2016 with the addition of eight hemispherical and three landscape photography stations at a third study site (LFCC-5B; Figure 4), in 2017 with the addition of nine hemispherical and four landscape photography stations at a fourth study site (EB-15; Figure 5), and in 2018 with the addition of 17 hemispherical and 10 landscape photography stations across two study sites (DL-12S and LFCC-02; Figures 6 and 7). However, LFCC-5B was severely burned in late June 2017 in the Tiffany fire, prior to the third sampling period, leaving minimal suitable SWFL habitat, and was subsequently excluded from most analyses.

The photo stations in BL-10 and EB-15 were located within the active floodplain of the Rio Grande, and the stations in DL-12N, DL-12S, LFCC-2, and LFCC-5B were located west of the Low Flow Conveyance Channel (LFCC) and LFCC Outfall. All sites included in this study have consistently supported breeding SWFL territories throughout many years of surveying (Moore *In Press*).

Figure 2. Hemispherical and landscape photography stations established in 2015 in site BL-10.

Figure 3. Hemispherical and landscape photography stations established in 2015 in site DL-12N.

Figure 4. Hemispherical and landscape photography stations established in 2016 in site LFCC-5B.

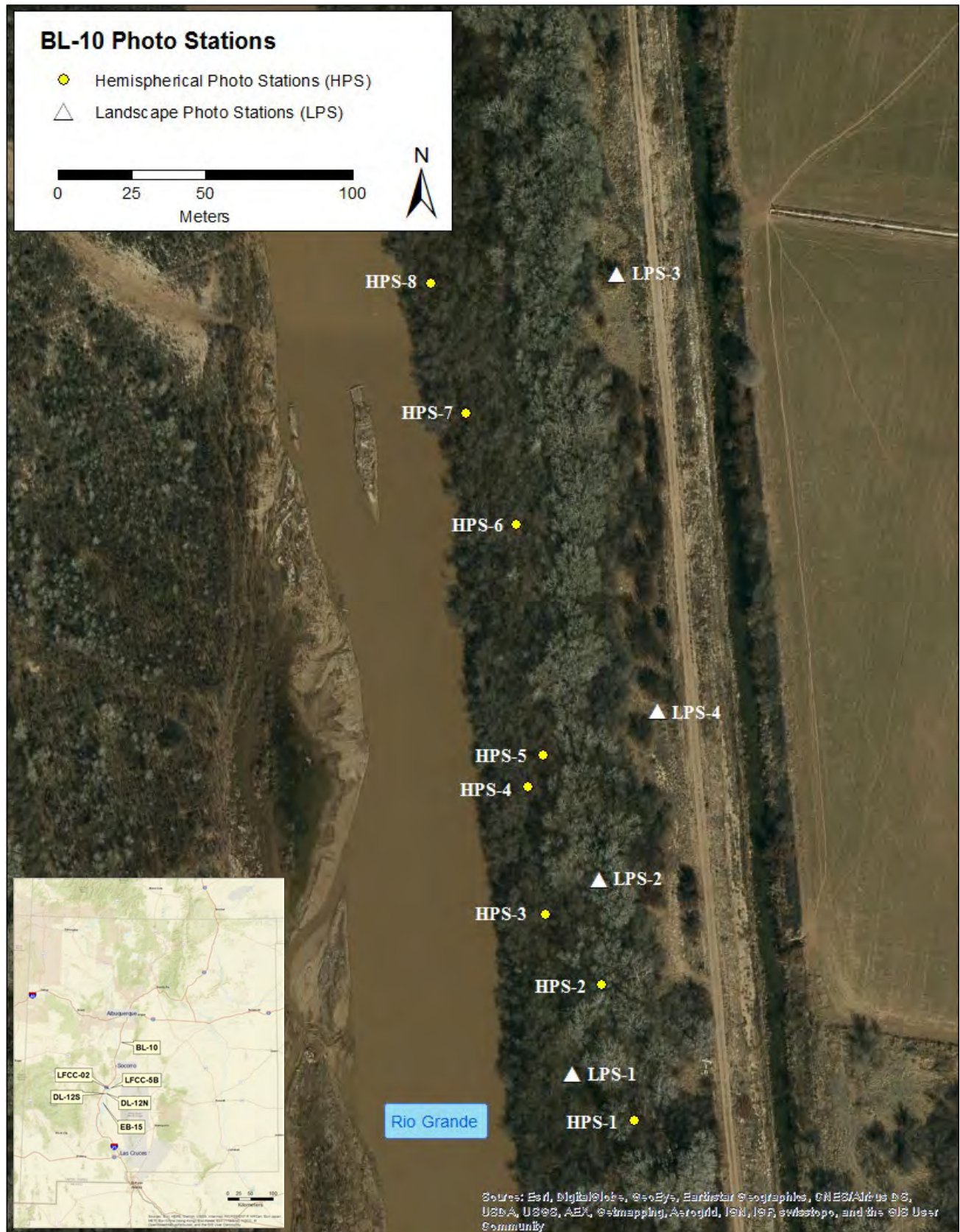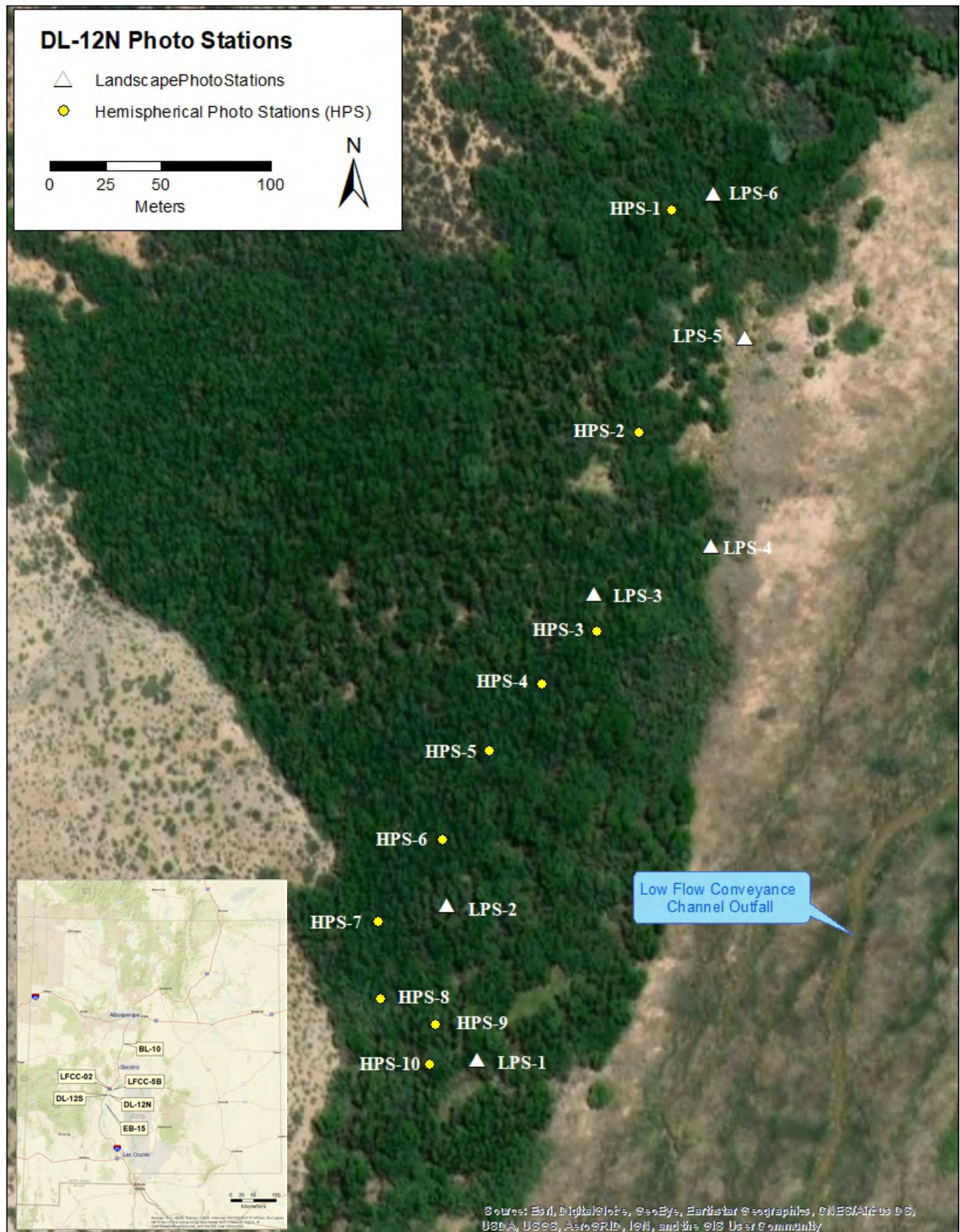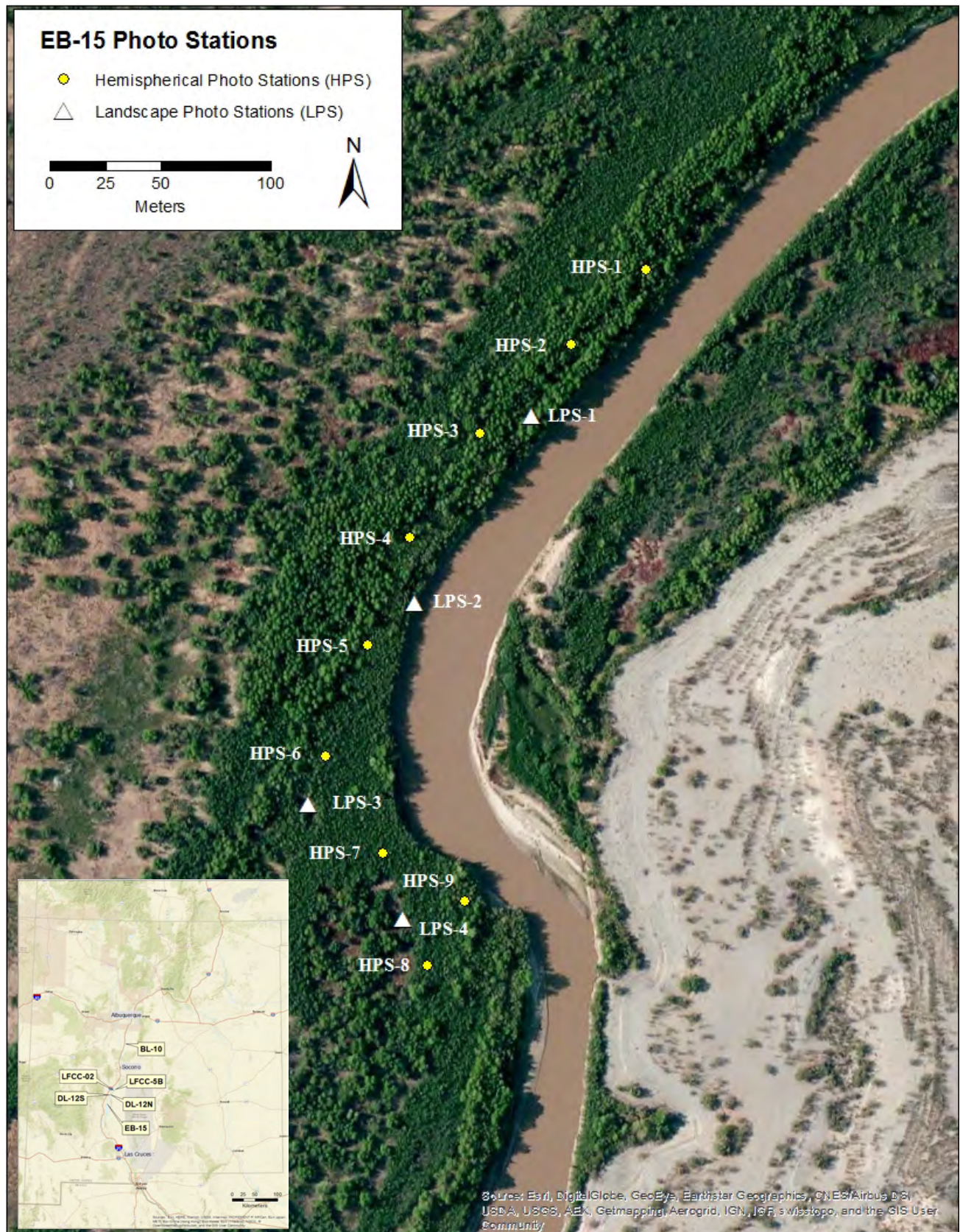**Figure 5.** Hemispherical and landscape photography stations established in 2017 in site EB-15.
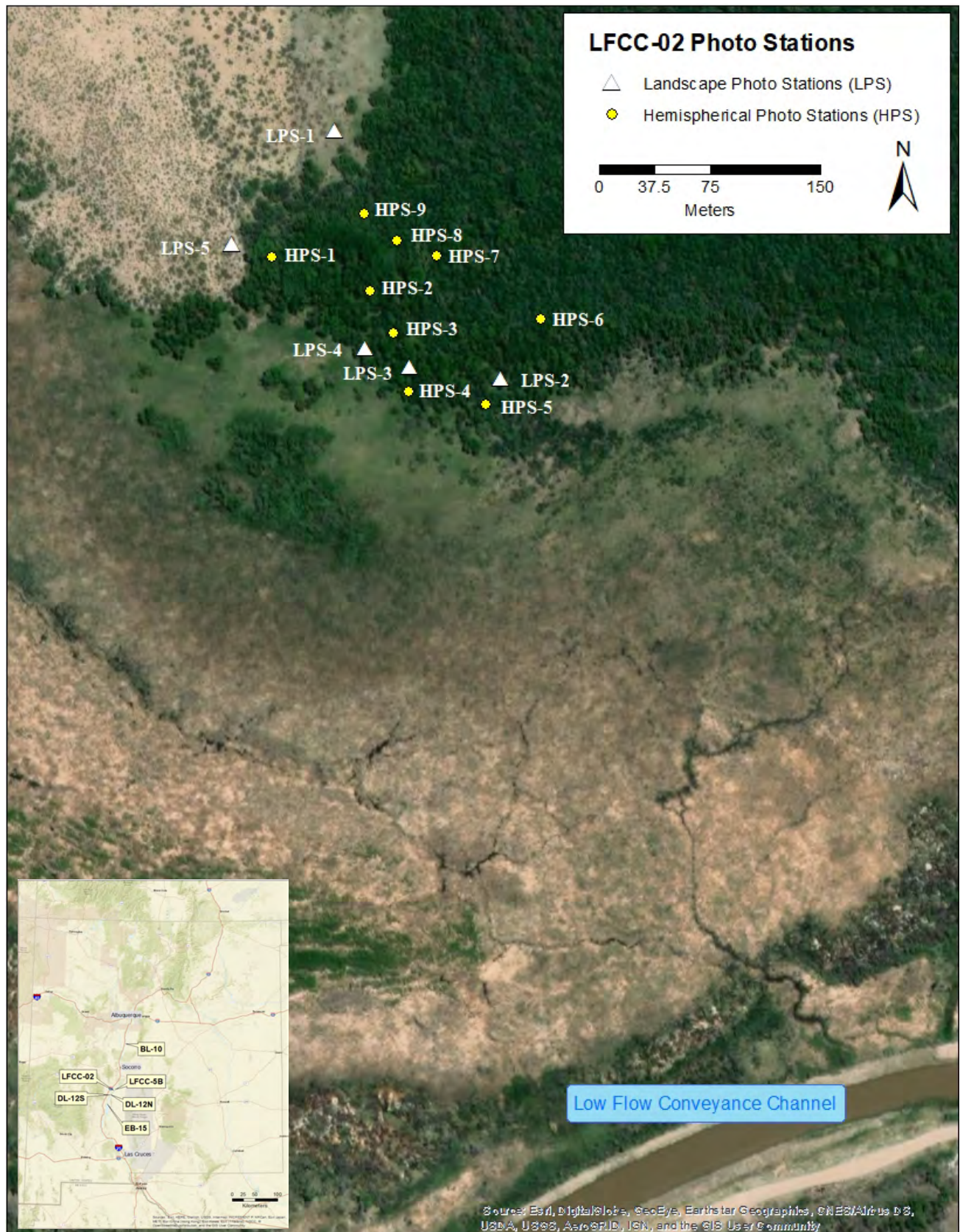
Figure 6. Hemispherical and Landscape photography stations established in 2018 in site LFCC-02.
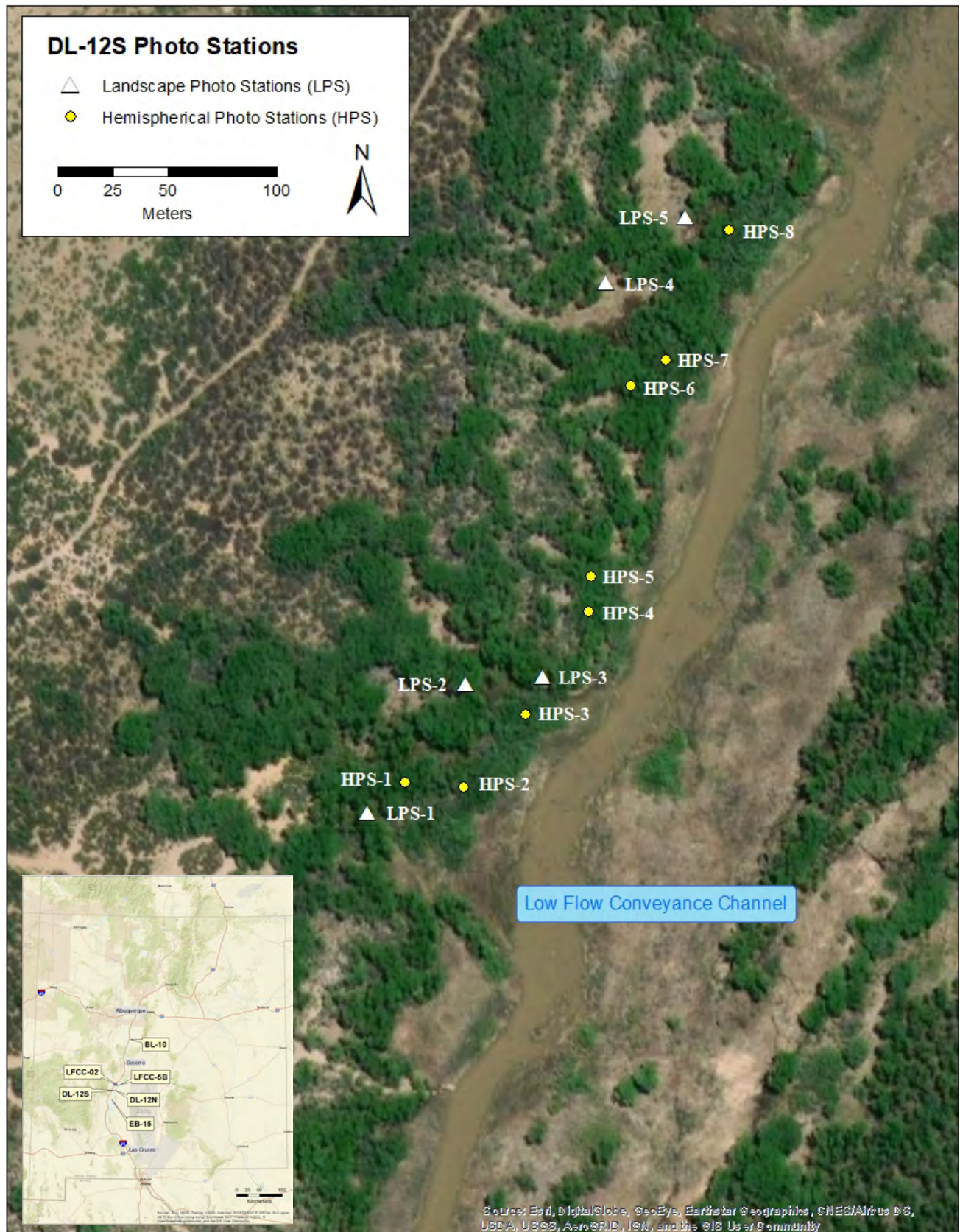
Figure 7. Hemispherical and Landscape photography stations established in 2018 in site DL-12S.

Hemispherical Photography

Hemispherical photographs were taken with a digital camera fitted with a Sigma EX DC 4.5 millimeter (mm) circular fisheye lens aimed upwards towards the vegetation canopy. A circular fisheye lens is a type of wide-angle camera lens that produces an image with a 180 degree angle of view that is projected as a circle within the image frame. Photographs were taken from a fixed point marked by a T-post. All hemispherical photography stations were established directly below a previously active SWFL nest, or within an active SWFL territory (Attachment 1A).

HemiView software (v.2.1; Delta-T Devices 1999) was used to classify each pixel of the hemispherical images as sky or vegetation. This binary classification was repeated three times for each image, with the data analyst blind to the associated numerical result, to ensure repeatability of results. Photographs were reduced to 40 percent of their original radius for analysis, in an effort to focus only on changes in canopy cover and exclude dense woody substrate captured at the periphery of the fisheye image. All photographs were taken before sunrise, after sunset, or on a cloudy day to eliminate glare caused by the reflection of the sun on foliage that would impede the ability of the HemiView software to distinguish between sky and illuminated vegetation.

HemiView quantifies visible sky on a scale from zero to one. Percent canopy closure was calculated from this value by subtracting the amount of visible sky from 1 and multiplying by 100. Canopy cover was then graphically compared among photo stations, survey sites, and years.

Vegetation species composition was visually estimated within a five-meter (m) radius of every hemispherical photo station annually. Relative percent cover (i.e., the proportion of each species within the composition) was estimated as species-specific percentages that were required to sum to 100 percent. This vegetation species composition estimation was added to our sampling to allow the examination of the impacts of tamarisk beetle defoliation within the context of variation in vegetation community composition, as well as to document changes in vegetation community composition over time.

Landscape Photography

Landscape photographs were taken with a digital Canon A620 Powershot camera aimed inwards towards the vegetation community. Photographs were taken in a standardized compass direction at each point to ensure the repeatability of photographs. Photographs were taken from a fixed point marked by a metal tag. Landscape photography stations were established in close proximity to the hemispherical photo stations, and within or on the edge of SWFL breeding territories (Attachment 1B). The purpose of the landscape photography was to visually document changes in vegetation community composition, in association with beetle-induced die-back of tamarisk, as well as to document spatial and temporal variation in defoliation.

Photographs were taken twice at each landscape and hemispherical photography station between May and August in 2015 and 2016, and three times in 2017 through 2019. Photographic sampling was timed to coincide with the SWFL breeding season and the period of tamarisk beetle activity. Starting in 2017, sampling was performed annually in mid-May, mid-June, and late-July, with sampling date standardized within sites across years. Multiple photographs were taken per year in order to enable a comparison of the potential impacts of defoliation by the tamarisk beetle within and among years. Additionally, dormant season hemispherical photographs were taken at all sites except BL-10, DL-12N, and LFCC-5B in March 2017 and at all sites in February 2018 and 2019, in order to provide a basis with which to distinguish woody from vegetative cover. Dormant season photographs are taken annually, as resources allow, in order to document changes in woody substrate.

Microclimate Monitoring

Loss of canopy cover due to defoliation by the tamarisk beetle could impact breeding SWFLs directly, through loss of nest concealment, or indirectly through higher temperatures or lower humidity that lead to higher thermoregulatory demand on adults, offspring, and developing embryos. In an effort to quantify changes in microclimate associated with tamarisk defoliation, Hygrochron™ iButton data loggers were deployed in the four study sites that were active in May 2017. A data logger was affixed at breast height to a tree at all odd-numbered hemispherical photography stations. Hygrochrons™ were programmed to record ambient temperature and percent relative humidity every 2.5 hours for the duration of the SWFL breeding season. The data are downloaded and loggers are redeployed annually.

Vegetation Composition

Repeated defoliation by the tamarisk beetle is predicted to lead to the decline and eventual mortality of tamarisk over time (Dudley 2005). As a result, other plant species would be expected to propagate as resource competition by tamarisk declined, potentially altering the composition of the vegetation community. Whether the habitat remained suitable for breeding flycatchers would be influenced by the nature of the vegetation community changes that occurred. In order to monitor these potential changes, and more comprehensively understand any potential future changes in SWFL habitat suitability, annual estimates of vegetation composition by height class were initiated at all hemispherical photo stations in 2018.

At each hemispherical photo station, a 3m by 3m square was laid out, centered on the photo station post, and marked at the corners with metal stakes to ensure interannual repeatability. An ocular estimate of the percent composition of woody vegetation, invasive herbaceous vegetation, and dead vegetation was made within each of 3 height classes: <1m, 1 to 3m, and 3 to 6m. An individual plant was only included in the layer of its maximum height. Additionally, an ocular estimate of ground cover by height class was made which include all vegetation present within each class, regardless of height, in an effort to estimate height class-specific density.

Results

Hemispherical Photography

Tamarisk beetles were not observed at any hemispherical photo stations in 2015 and were detected at only one site (BL-10) in 2016. Therefore, 2015 and 2016 data (with BL-10 excluded in 2016) were used as baseline "pre-tamarisk beetle colonization" conditions for comparison to future years in data analysis. Additionally, defoliation was not observed in BL-10 in 2017 and canopy cover rebounded to the normal density and growth pattern for that site, providing acceptable baseline canopy cover data for 2017. When EB-15 was added as a study site in 2017, the tamarisk beetle was already present on the first sampling occasion but browning or defoliation was not yet evident. Therefore, the first sampling period of 2017 in EB-15 was included in analyses of pre-defoliation data as an indicator of baseline (i.e., pre-tamarisk beetle) conditions in that site. Furthermore, baseline canopy closure in EB-15 was found to be very similar to LFCC-5B, and therefore the latter is considered to provide a reasonable pre-defoliation reference for EB-15. LFCC-5B was severely burned in the 2017 Tiffany Fire and was thereafter excluded from this study. DL-12S was added to the study in 2018 to serve as a control site. This site is dominated by native vegetation and therefore not likely to be severely impacted by tamarisk beetle defoliation.

Data from 2017 through 2019 were used to investigate the magnitude of change in canopy cover resulting from tamarisk beetle colonization, i.e., "post-tamarisk beetle colonization". Defoliation was observed throughout DL-12N and EB-15 in 2017, but not in BL-10 or LFCC-5B. Defoliation was observed at hemispherical photo stations at all sites except DL-12S and BL-10 in 2018. However, that defoliation occurred later in the breeding season than in previous years. Whereas in 2017 defoliation was already occurring throughout EB-15 in the first monitoring period, defoliation was not observed until the third monitoring period in 2018 and then only in part of the site. Similarly, minor defoliation was observed in DL-12N in the first two monitoring periods of 2018, but severe defoliation was not observed until the third monitoring period. In contrast, severe defoliation was already occurring in the first two monitoring periods of 2017 in that site. In 2019, defoliation was only observed at a few photo stations in the last monitoring period at LFCC-02 (Table 1).

Table 1. Monitoring period* in which defoliation was first observed at hemispherical photo stations by study site annually.

	BL-10	DL-12N	EB-15	LFCC-5B	DL-12S	LFCC-2
2015	None	None	-	-	-	-
2016	3	None	-	None	-	-
2017	None	1	1	None	-	-
2018	None	1	3	1	None	3
2019	None	None	None	None	None	3

*Photo monitoring period 1 = mid-May, 2 = early June, 3 = late July. A dash indicates the site was not monitored that year.

Median site-wide canopy closure prior to colonization by the tamarisk beetle was highest in EB-15 (91%; $n = 9$), LFCC-5B (85%, $n = 16$), and DL-12S (83%, $n = 24$), when summarized across all pre-tamarisk beetle sampling occasions (Figure 8). Median canopy closure in BL-10 (77%; $n = 16$) was similar to DL-12N (76%; $n = 20$) prior to colonization by the tamarisk beetle.

Median site-wide canopy closure after colonization by the tamarisk beetle (2017 through 2019) was approximately 80 to 90 percent in all sites, regardless of whether the tamarisk beetle was present in that site. This represented an apparent increase in observed average canopy closure in DL-12N and BL-10 compared to 2015 and 2016 levels, suggesting that between-year variations in cover in association with defoliation by the tamarisk beetle were confounded by the inherent increase in canopy cover in all sites in 2017. This increase in 2017 may be due to high and persistent river flows compared to previous years. However, a trend towards an overall decrease in canopy cover was observed in all sites in 2019 despite high river flows and the near absence of the tamarisk beetle. Nevertheless, average canopy cover remained within the observed range of natural variation for all sites. LFCC-5B was excluded from this analysis due to the impact of the 2017 Tiffany fire on that site.
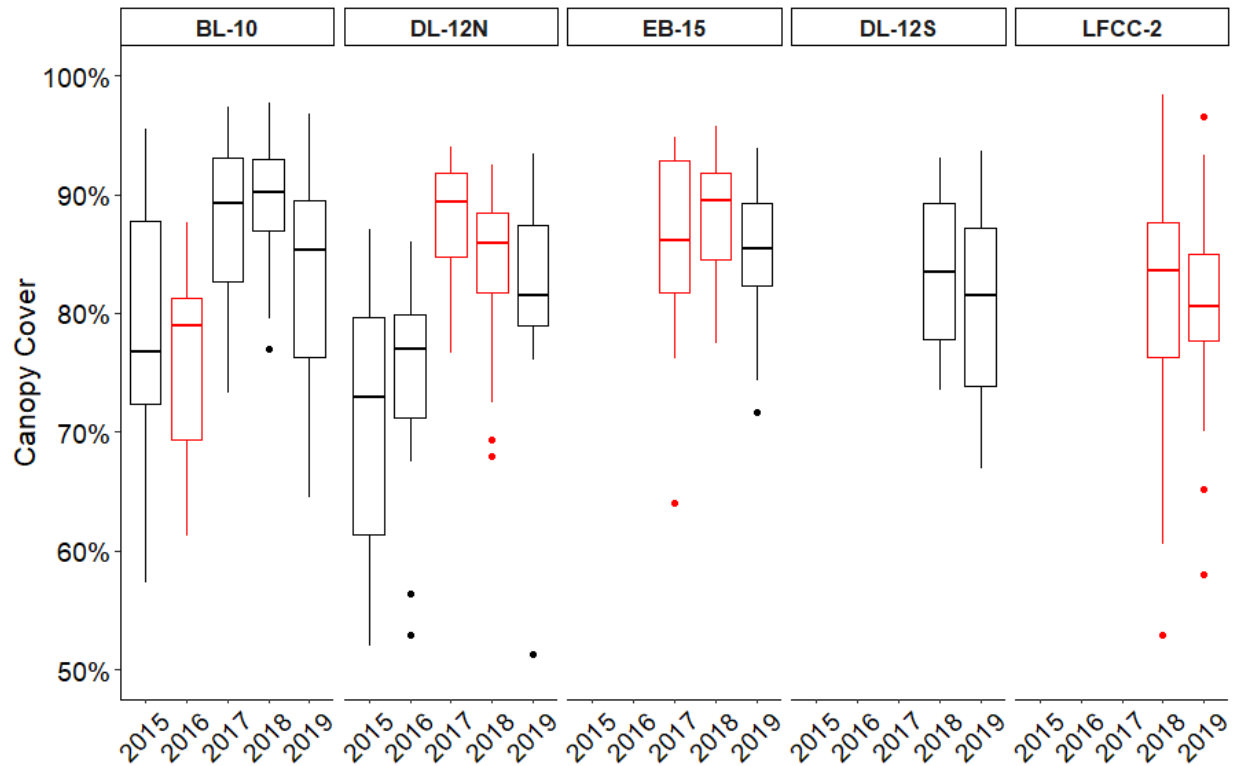
Figure 8. Annual canopy closure by study site.
Red boxplots indicate years in which the site was impacted by the tamarisk beetle. Box centerlines indicate median values, tails indicate minimum and maximum values, and points indicate outliers.

Vegetation composition was estimated at each hemispherical photography station once annually in 2016 through 2018. The median relative percentage of tamarisk within the vegetation community was twice as high in LFCC-02 (90%), DL-12N (95%), and LFCC-5B (100%) as in BL-10 (50%; Figure 9). The majority of the remaining portion of the vegetation community in BL-10 was composed of Russian olive (*Elaeagnus angustifolia*). Tamarisk composed between 5 and 100 percent of the vegetation community at EB-15 photo stations (*Mdn* = 60%). In general, the proportion of the vegetation community composed of tamarisk increased from north to south in this site, such that stations 1 through 4 were less than 50 percent tamarisk, and stations 5 through 9 were greater than 50 percent tamarisk. Vegetation in DL-12S was predominately native (*Mdn* = 0% tamarisk) and therefore serves as a control site in the study.

Figure 9. Percent of vegetation community within 5 m radius of hemispherical photo stations composed of tamarisk.

Hemispherical photos were taken in DL-12N and BL-10 in all five years of this study (Table 1). Canopy cover in the years prior to colonization by the tamarisk beetle followed a similar 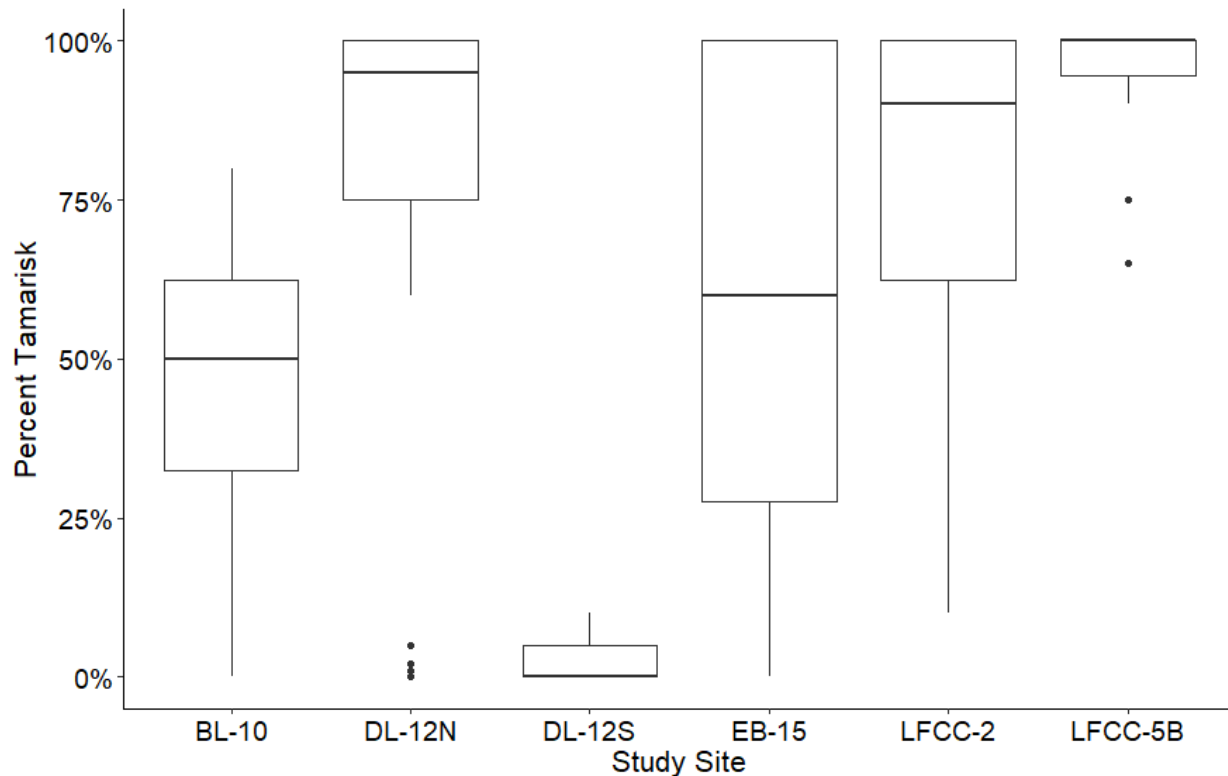trend of increasing canopy cover between May and July, such that canopy cover was highest in June and July, in both sites regardless of between-year variation in absolute cover. However, in the years that each site was defoliated by the tamarisk beetle (BL-10 in 2016, and DL-12N in 2017 and 2018) canopy cover steadily declined from May to July, rather than increasing. BL-10 was not impacted by the tamarisk beetle after 2016 and canopy cover increased or remained stable during those same months, as had been observed prior to tamarisk beetle colonization (Figures 10 and 11). Severe tamarisk foliage death and defoliation was recorded in DL-12N in 2017 and 2018, and a notable decline in canopy cover was observed over the course of the breeding season. Similar patterns were observed in the other study sites, such that canopy cover declined between May and July in LFCC-02 and EB-15 (impacted by the tamarisk beetle; Figures 1 and 2 in Attachment 2) but increased in LFCC-5B and DL-12S (not impacted; Figures 3 and 4 in Attachment 2). Nevertheless, canopy cover remained within the apparent normal range of variation, based on annual observed ranges, for both BL-10 and DL-12N during the years that they were impacted by the tamarisk beetle. Notably, neither site was impacted by the tamarisk beetle in 2019, but canopy cover remained in the lower end of the range observed during the past several seasons.
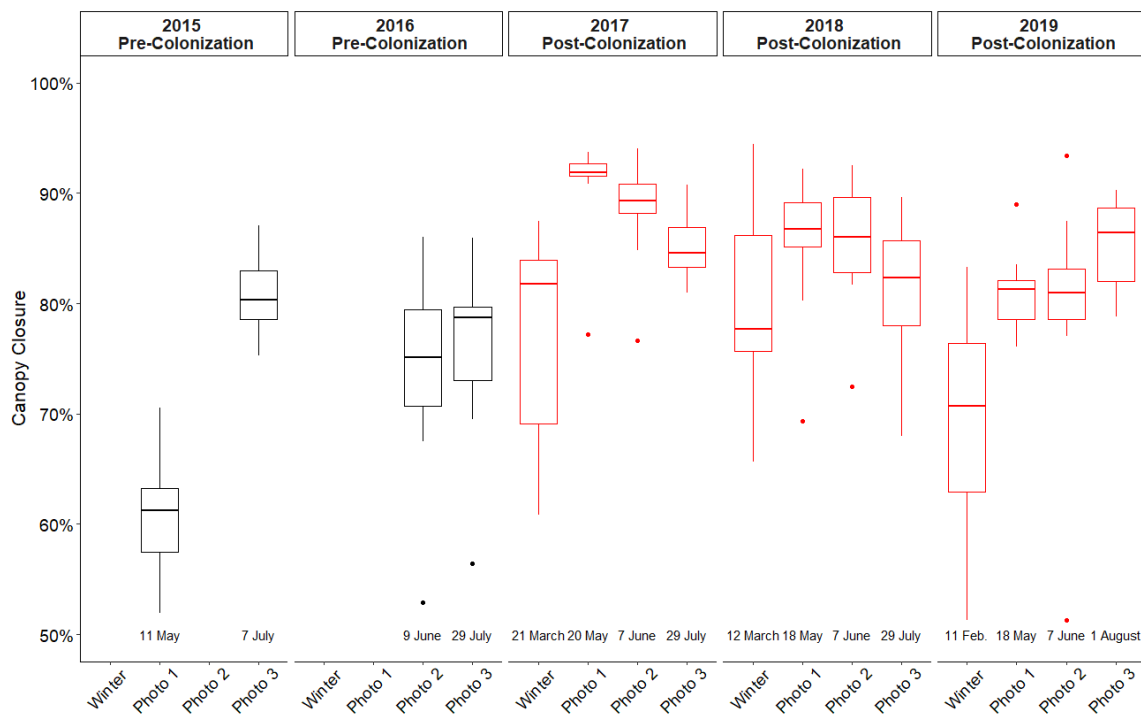
Figure 10. Trends in canopy cover at DL-12N pre- and post-colonization by the tamarisk beetle.
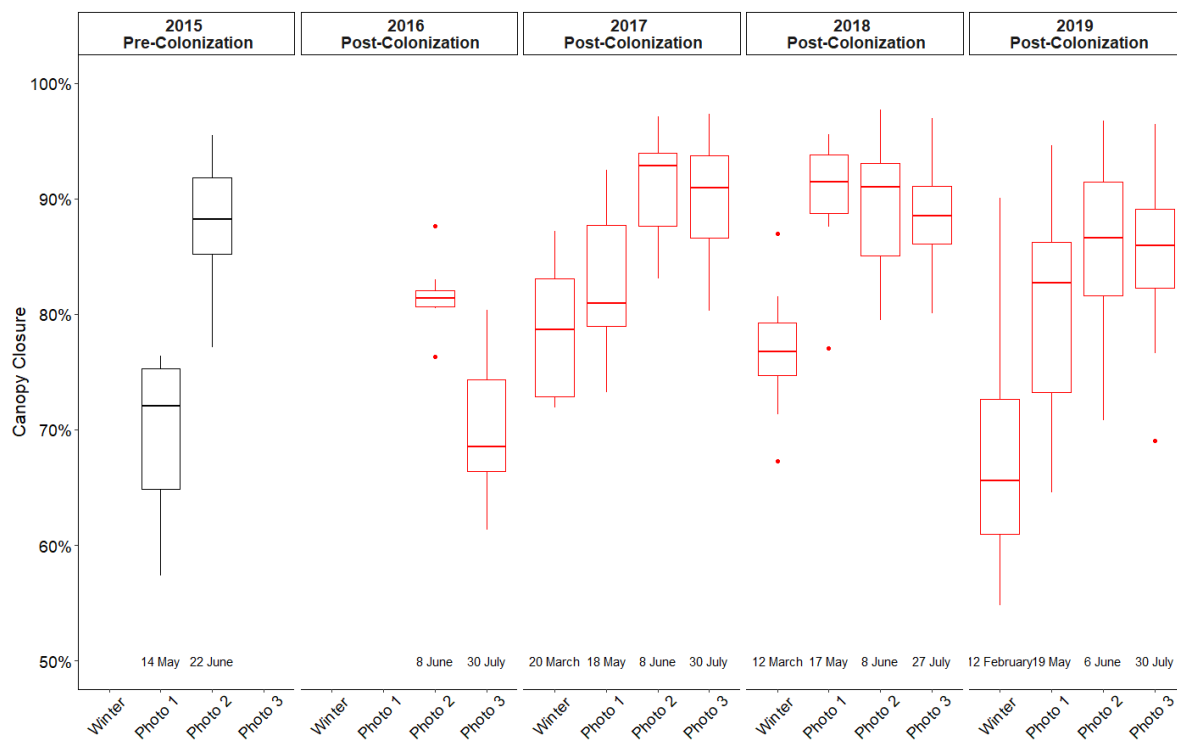

Figure 11. Trends in canopy cover at BL-10 pre- and post-colonization by the tamarisk beetle.
*BL-10 was impacted by the tamarisk beetle in 2016 but recovered to pre-colonization canopy cover in 2017-2019.

The study sites impacted by the tamarisk beetle in 2017 and 2018, DL-12N, EB-15, and LFCC-02, all exhibited extensive foliage death and defoliation. Tamarisk beetle colonization and defoliation occurred later in the SWFL breeding season in 2018 than in 2017. Whereas severe defoliation had occurred throughout much of DL-12N and EB-15 by mid-June in 2017, only moderate levels of defoliation in limited patches of DL-12N and no defoliation in EB-15 were observed at that same time in 2018. Severe defoliation was observed throughout most of DL-12N and LFCC-02 in the third sampling period of 2018 (late July), and in parts of EB-15.

A plot of canopy cover by sample point in LFCC-02 illustrates the loss of cover as the impact of the tamarisk beetle increased throughout the season in 2018 (Figure 12). Sample points are plotted in order of increasing tamarisk dominance. Canopy cover decreased by between 3 and 28 percent in LFCC-02 from early June to late July 2018, a time period when canopy cover increased at non-impacted sites. Indeed, the tamarisk beetle had minimal impact in LFCC-02 in 2019, only just beginning to colonize the site in the final monitoring period, and canopy cover again followed the expected positive seasonal trend (Figures 13 and 14).

Figure 12. Canopy cover at LFCC-02 hemispherical photo stations by 2018 sampling occasion. Subplot titles indicate the percentage of vegetation within a five meter radius of the point comprised of tamarisk, with the point number in parentheses. Error bars represent standard error of canopy cover calculated from three repeat classifications of each hemispherical photo.
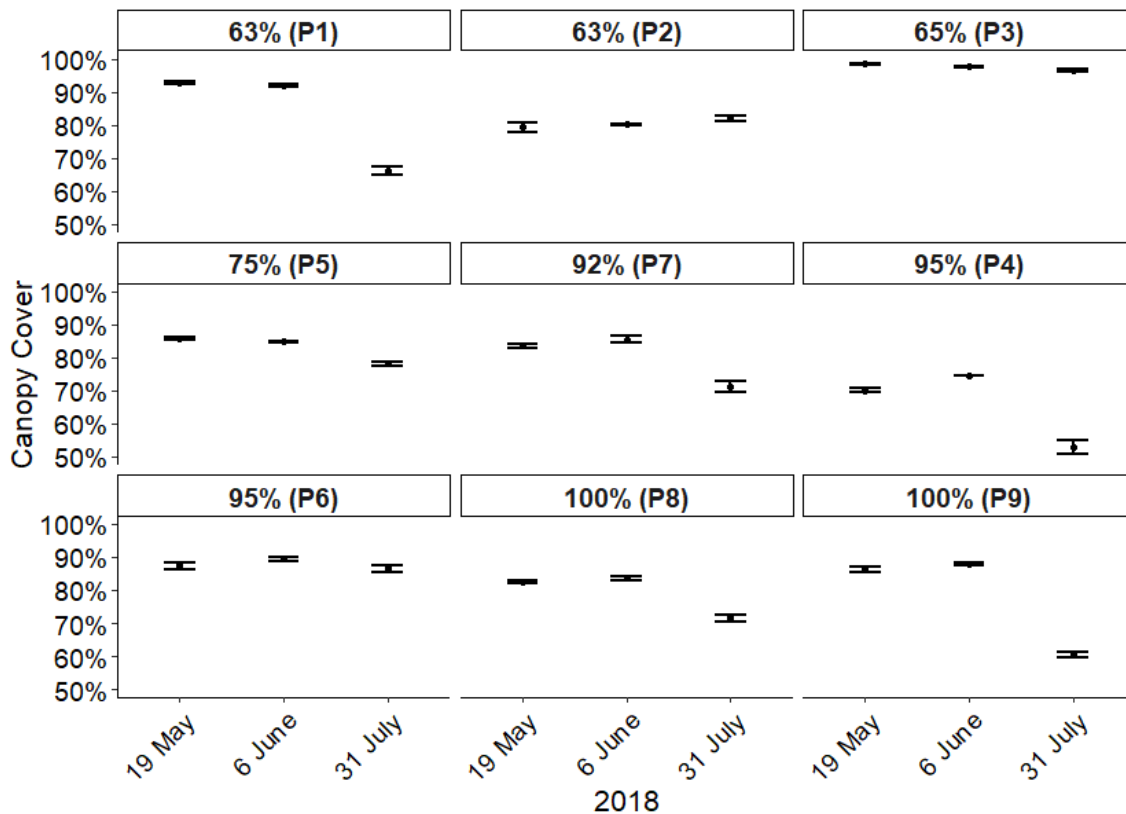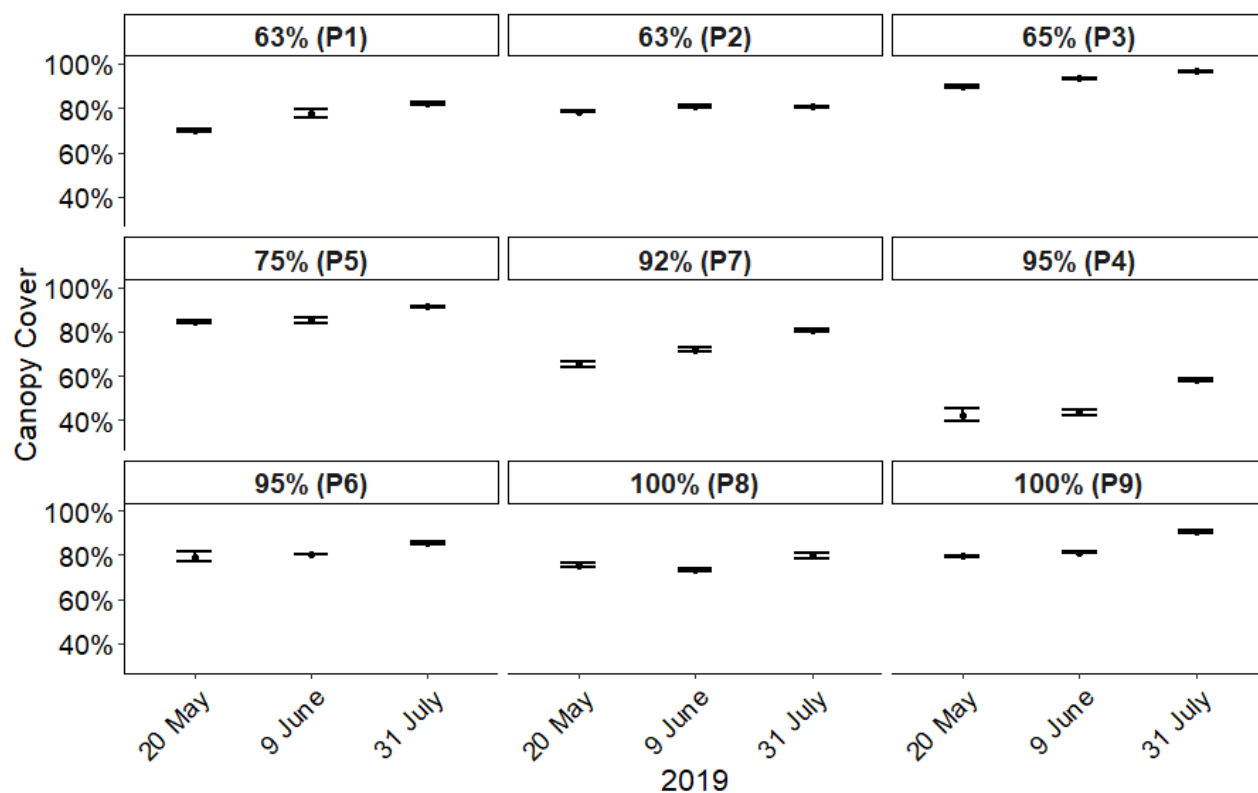
Figure 13. Canopy cover at LFCC-02 hemispherical photo stations by 2019 sampling occasion. Subplot titles indicate the percentage of vegetation within a five meter radius of the point comprised of tamarisk, with the point number in parentheses. Error bars represent standard error of canopy cover calculated from three repeat classifications of each hemispherical photo.
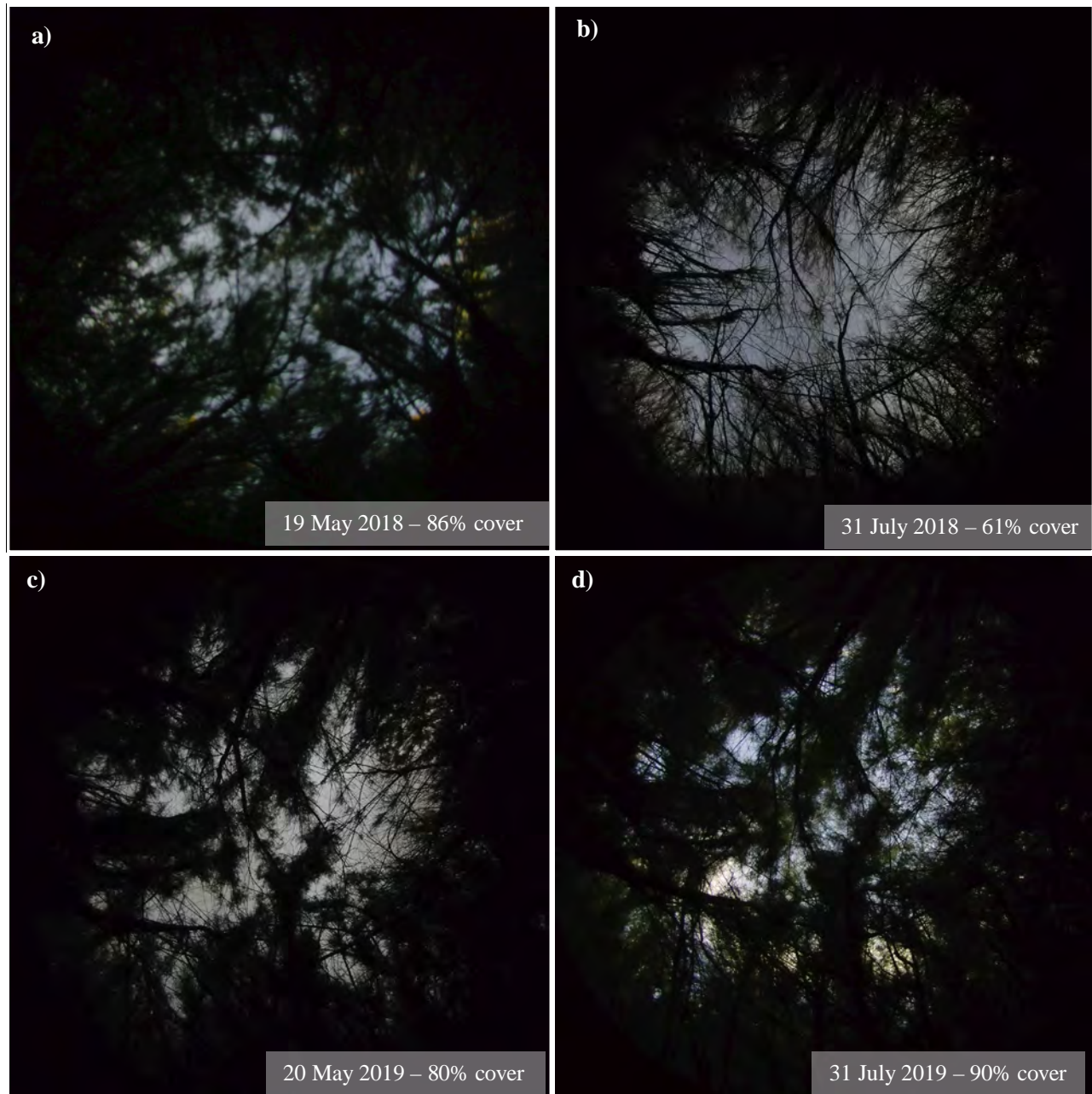
Figure 14. Hemispherical photography at LFCC-02 point 9 in 2018 (a, b) and 2019 (c, d).

Hemispherical photographs were taken prior to the 2018 and 2019 growing seasons (i.e., "winter") at all study sites, and prior to the 2017 growing season in BL-10 and DL-12N. A comparison of annual dormant season canopy cover suggests a possible negative trend in overall woody vegetation in some sites impacted by the tamarisk beetle. Most notably, from 2017 to 2019 median woody cover declined from 79 percent to 65 percent in BL-10, and from 81 percent to 69 percent in DL-12N. Woody cover increased between 2018 and 2019 in DL-12S, the control site that contains predominately native vegetation and is located in close proximity to DL-12N (Figure 15). Continued monitoring is needed to determine whether this downward trend signals the beginning of an overall vegetation decline, due to repeated years of beetle impact, or is simply natural annual variation. The relatively small variation in

canopy cover between the dormant and growing season in all sites suggests that woody mass comprises a great deal of the overall canopy cover in these densely vegetated riparian study sites and decreases in that woody cover can have large impacts on breeding habitat suitability.

Figure 15. Annual changes in dormant season canopy cover by study site.

Landscape Photography

All landscape photographs were taken on the same dates as the hemispherical photographs in each study site. The tamarisk beetle was not recorded in landscape photos at any study site in 2015. Tamarisk browning and defoliation by the tamarisk beetle were recorded at three of four landscape photography stations in BL-10 on the final sampling occasion (30 July) of 2016, but not in any other study site that year. Tamarisk browning and def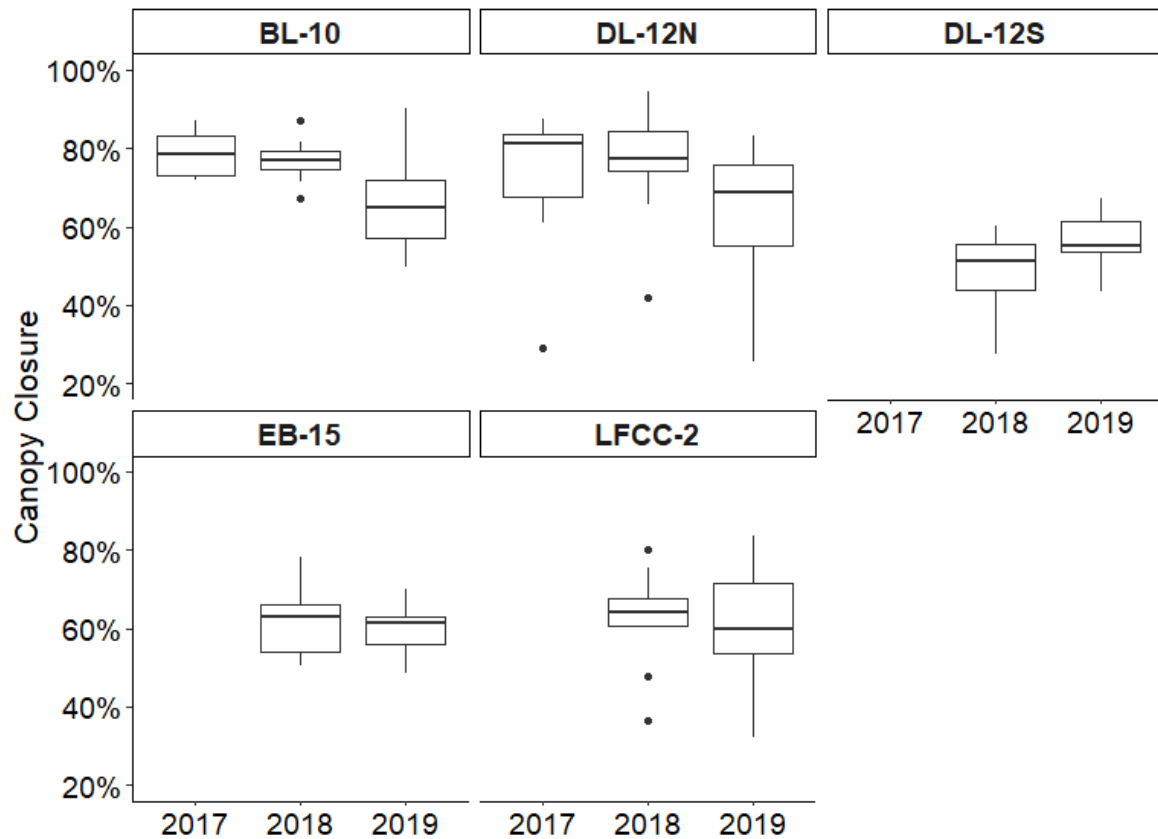oliation was observed at all landscape photography stations in DL-12N ($n =6$) and EB-15 ($n =4$) in 2017, and the tamarisk beetle was present in both sites on all three sampling occasions. Browning and defoliation were likewise observed at landscape photo stations in all three monitoring periods of 2018 in DL-12N, but not until the last monitoring period in EB-15, LFCC-02 and DL-12S.

The tamarisk beetle was present in the interior of DL-12N earlier in the season than in any other site in 2018, and browning and defoliation occurred over a longer time period there than in any other site that year. This was a reversal from 2017, where the earliest colonization occurred in EB-15. Despite not being colonized by the tamarisk beetle during the 2019 breeding season, the cumulative impact of multiple years of tamarisk beetle defoliation was evident in the overall reduction in foliar cover in DL-12N; foliage density in 2019 still did not return to the level observed in May 2017 prior to the arrival of the beetle (Figure 16). The defoliation in DL-12S was minimal and limited to the outer edge of the study site, as the interior of that site was dominated by native vegetation. Indeed, a general pattern was observed in all sites of the tamarisk beetle colonizing the outer edges of the site first, and then moving towards the site interior.
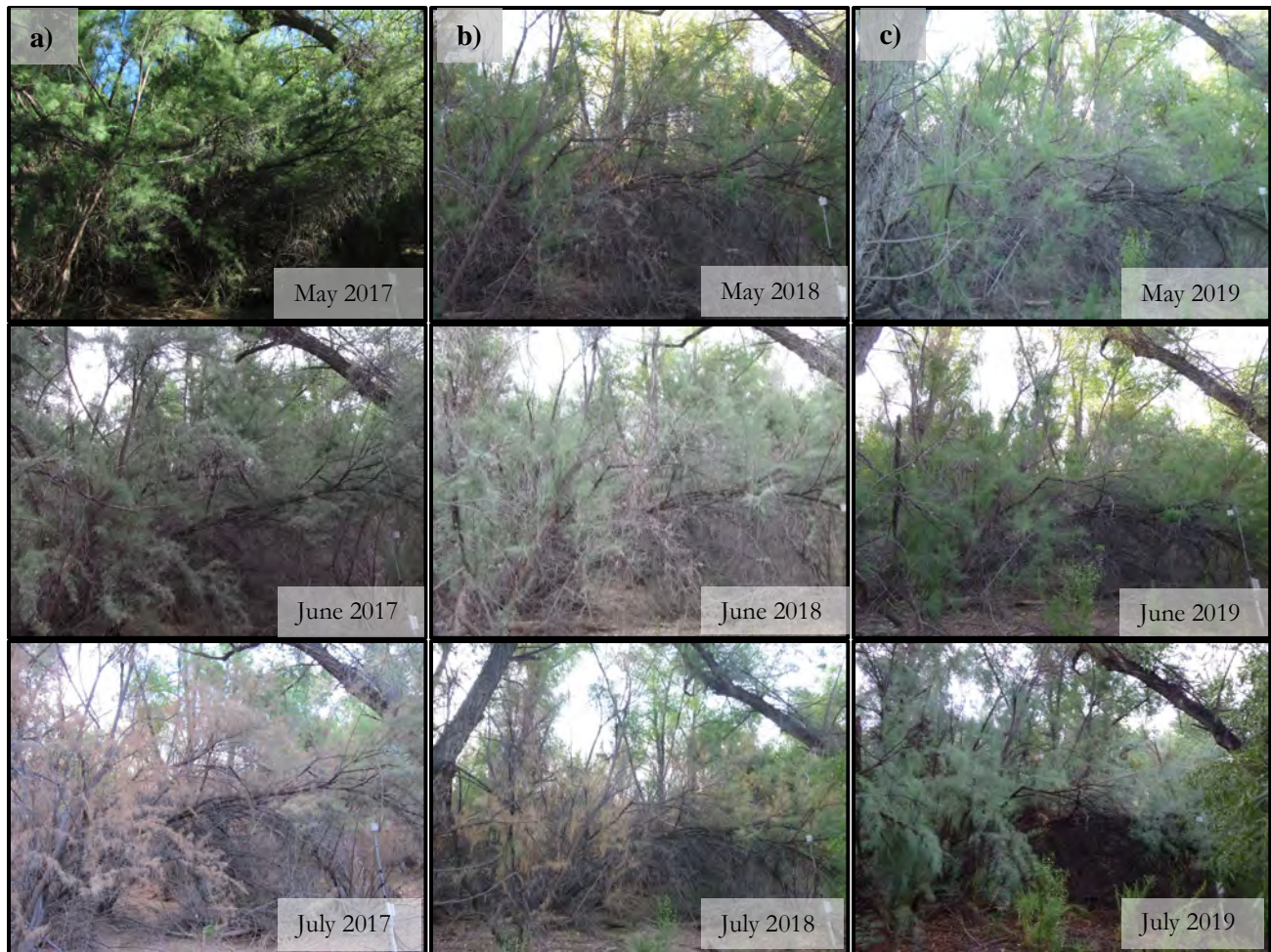
Figure 16. Seasonal progression of tamarisk browning and defoliation in DL-12N in 2017 (column a), 2018 (column b), and 2019 (column c).

Microclimate Monitoring

Although there was minimal tamarisk beetle impact to the study sites during the 2019 monitoring season, the cumulative effect of multiple previous years of defoliation that was suggested by canopy cover data (Figure 15) and landscape photography (Figure 16) was further reflected in microclimate data. DL-12N appears to have experienced the sharpest decline in overall canopy cover over the past several years, including loss of woody mass, and this appears to result in higher and more variable maximum temperatures within the site. The large within-site variations in maximum temperature, and peak temperatures experienced, were even more accentuated in 2019 data than in the first year of defoliation in 2017 (Figure 17). In comparison, EB-15 also experienced temperature extremes and fluctuation in 2017 due to defoliation but that site appeared to recover better and did not experience the same losses in woody mass observed in DL-12N (Figure 15). As a result, maximum temperature extremes were more moderate and within-site temperature fluctuations were minimal in 2019, as would typically be expected in a site not impacted by tamarisk beetle defoliation (Figure 18). Indeed, maximum temperature was considerably higher and more variable in DL-12N than in DL-12S, the control site located approximately one kilometer south of DL-12N (Figure 19).

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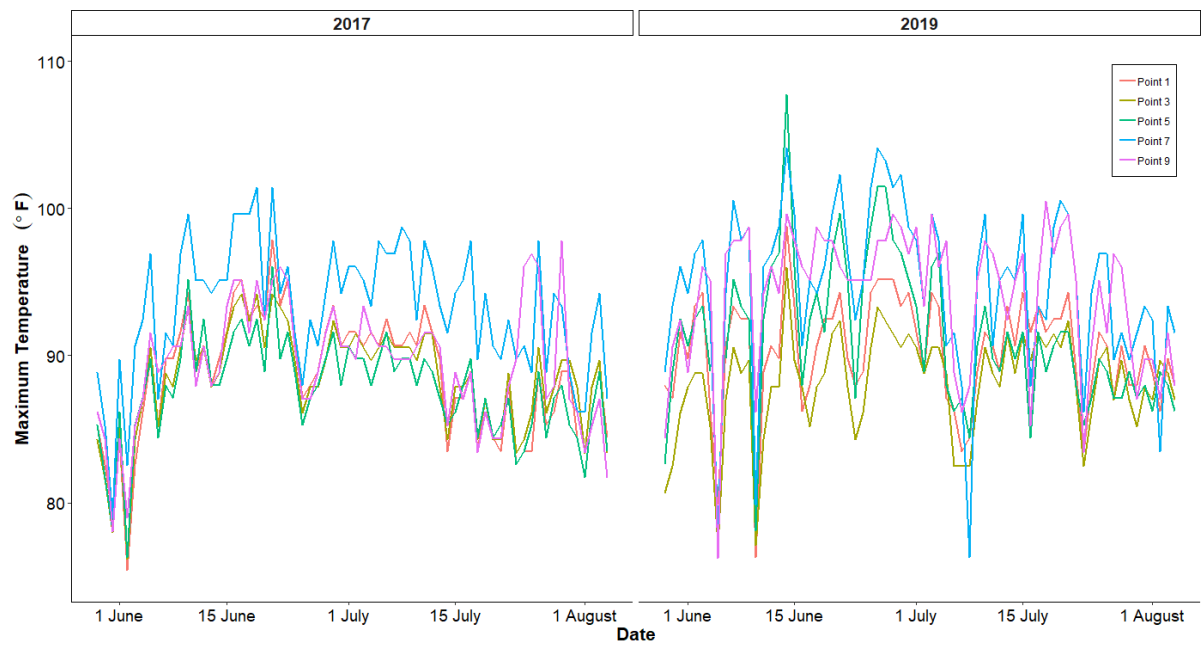
Figure 17. Daily maximum temperature in DL-12N at hemispherical photo stations during the SWFL breeding season.
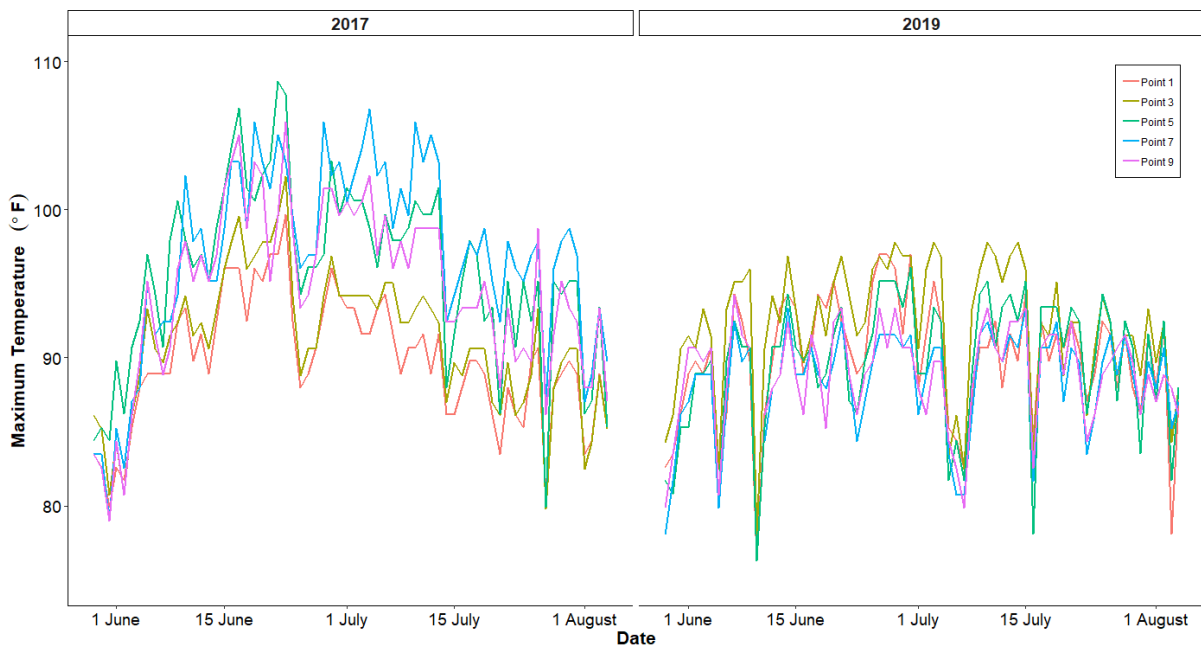
Figure 18. Daily maximum temperature EB-15 at hemispherical photo stations during the SWFL breeding season.
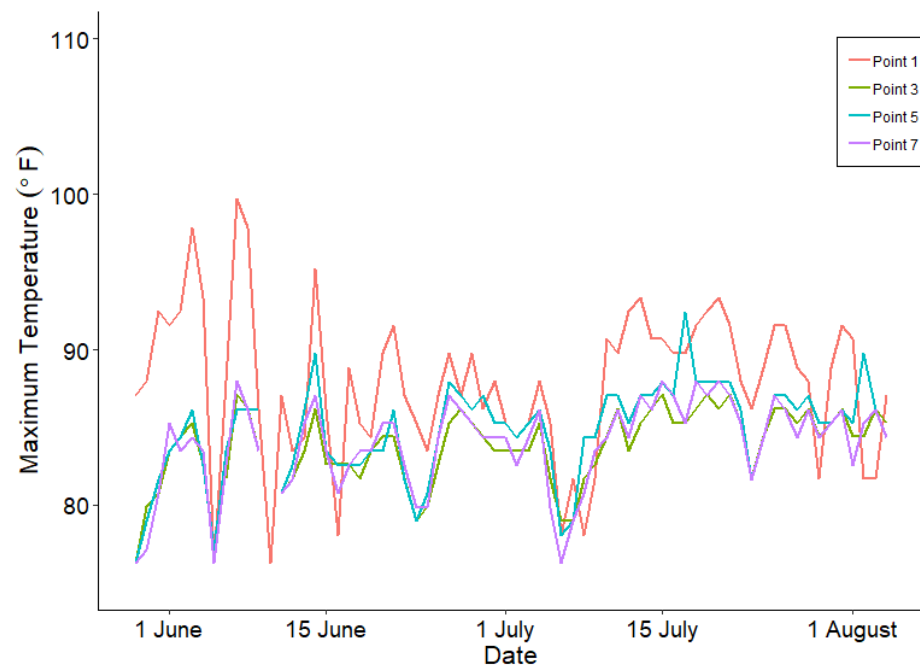
Figure 19. Daily maximum temperature in DL-12S at hemispherical photo stations during the SWFL breeding season.

Similar but less accentuated patterns in percent relative humidity were observed. Minimum percent relative humidity was low and varied considerably between sample points in sites impacted by the tamarisk beetle in 2017 (i.e. DL-12N and EB-15). In 2019, minimum relative humidity reached even lower levels and remained highly variable in DL-12N despite the lack of tamarisk beetle impact during the SWFL breeding and the overall high runoff that occurred (Figure 20). On the other hand, in 2019 relative humidity in EB-15 never decreased to the minimums observed in 2017, and variability was less extreme (Figure 21). Again, this appears to reflect the overall declines in canopy cover observed in DL-12N after multiple years of beetle defoliation.
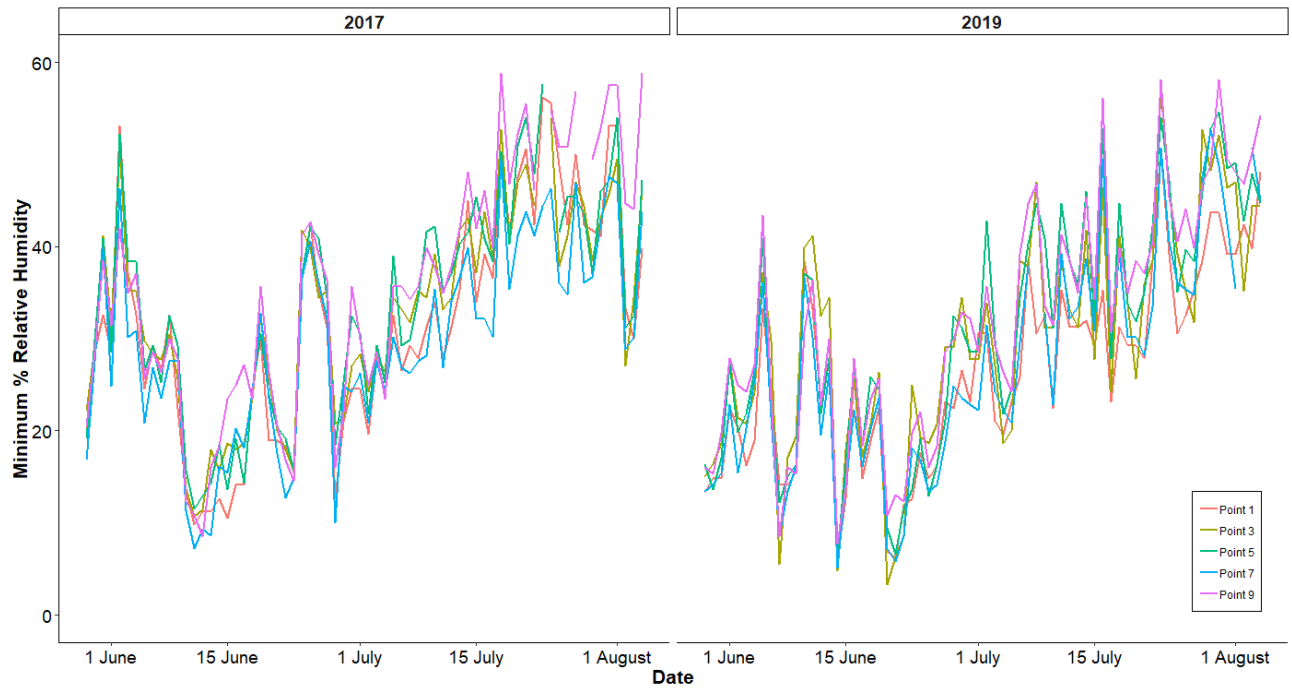
Figure 20. Daily minimum percent relative humidity in DL-12N at hemispherical photo stations during the SWFL breeding season.
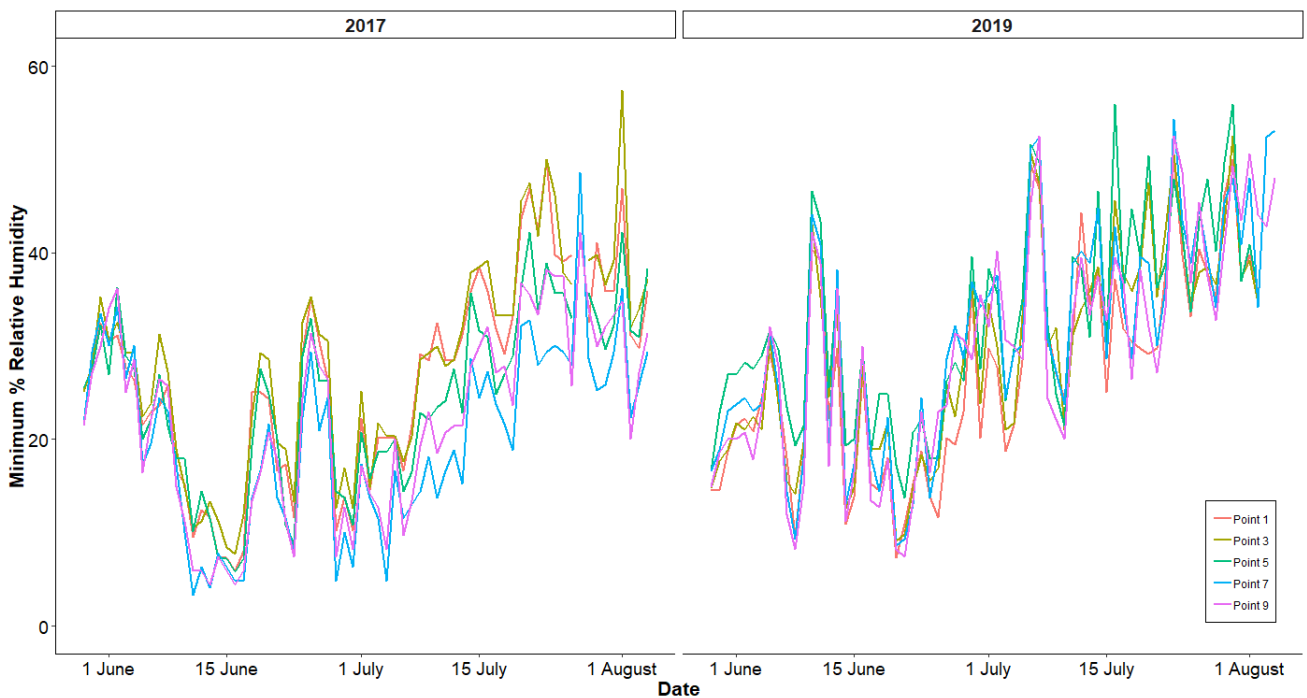
Figure 21. Daily minimum percent relative humidity in EB-15 at hemispherical photo stations during the SWFL breeding season.

Vegetation Composition

Vegetation composition data were collected with the intention of tracking potential vegetation community changes over time. As such, the initial two years of monitoring are only sufficient to summarize current conditions in the study sites; additional years of data will enable the identification of long-term trends. The 3m to 6m height class is most representative of the SWFL nesting habitat layer, whereas the <1m height class is most indicative of new growth. Figure 22 shows the percentage of the 3m to 6m height class composed of salt cedar versus dead vegetation, and the density of vegetation in that height class, at each monitoring site. It would be predicted that multiple years of defoliation in a site may lead to a decline in salt cedar, increase in dead vegetation, and at least an initial decrease in vegetation density.
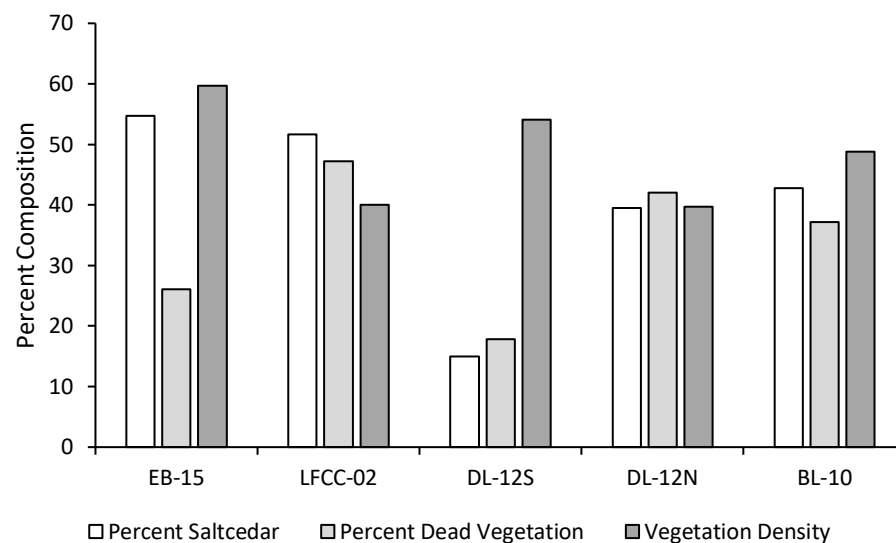
Figure 22. Vegetation composition and density of the 3m to 6m height class by study site in 2018-2019.

Discussion

Tamarisk has become an important component of riparian bird habitat as drought and changes in hydrology have prompted the increased abundance of invasive vegetation and the loss of native vegetation in many riparian areas of the southwestern U.S. Indeed, more than 70 percent of SWFL nests located within the receded pool of Elephant Butte Reservoir, which supports the majority of the Middle Rio Grande population, were constructed in tamarisk for the last three years (Figure 22; Moore *In Press*). This is compared to 70 percent of nests constructed in native willow 15 years ago. Moreover, approximately three quarters of SWFL nests in Elephant Butte Reservoir were in breeding territories that were either dominated by exotic vegetation (primarily tamarisk) or a mix of native and exotic vegetation (Figure 23; Moore *In Press*). The high use of tamarisk by breeding SWFLs raises many questions regarding how defoliation of SWFL habitat by the tamarisk beetle will ultimately influence the recovery of this endangered species and makes early and continuous monitoring of these impacts an important aspect of population management.
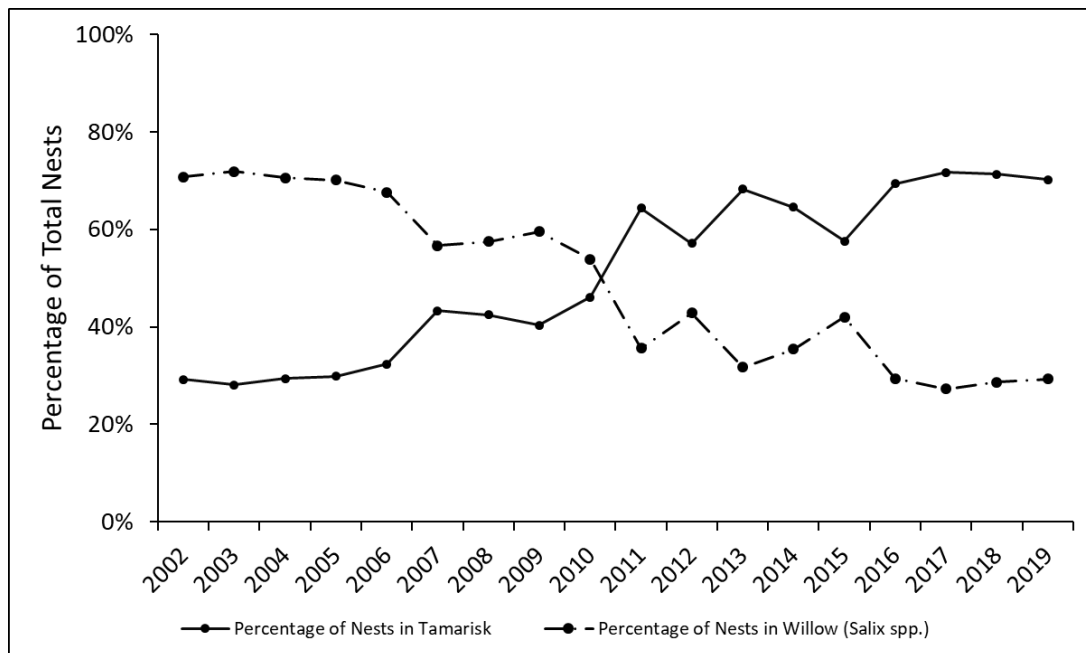
Figure 23. Percentage of SWFL nests constructed in native willow vs. invasive tamarisk. (*n* = 3276, range = 65-270 nests per year)
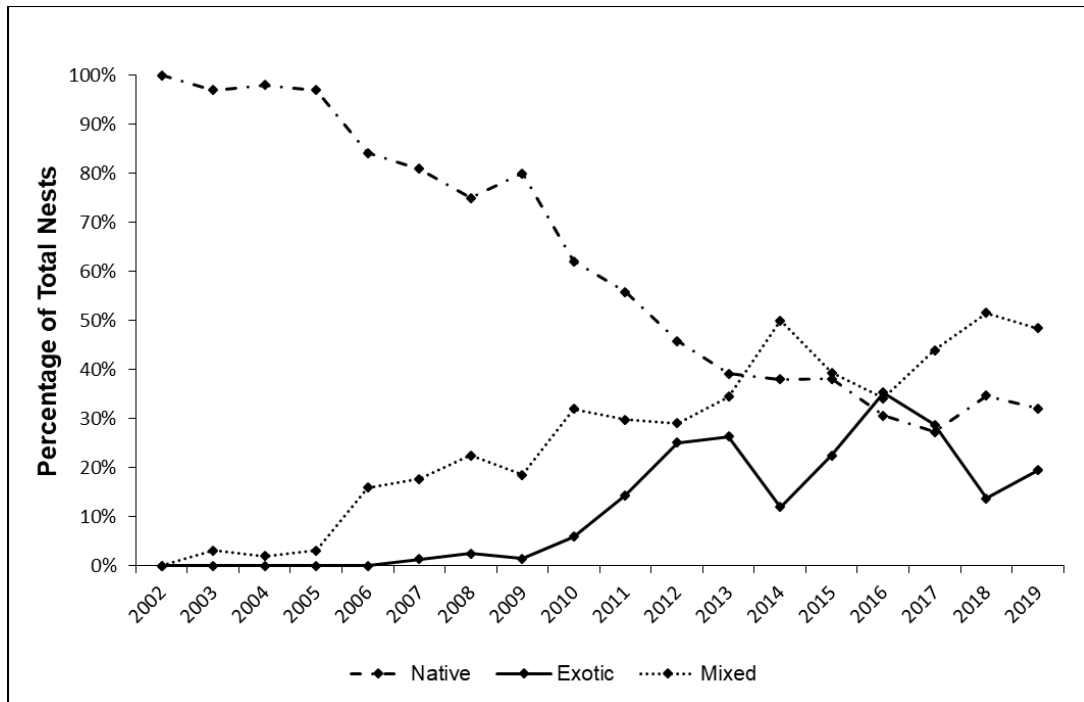
Figure 24. Percentage of SWFL nests constructed in territories dominated by native vs. exotic vs. mixed native and exotic vegetation.
(*n* = 3276, range = 65-270 nests per year)

The tamarisk beetle was documented throughout the entire Rio Grande riparian corridor beginning in 2016 (Johnson pers.comm.; Tamarisk Coalition 2018). The subtropical and northern species of the tamarisk beetle were both found on the Middle Rio Grande, with the two populations converging in the vicinity of San Marcial. The northern tamarisk beetle is known to produce three generations in a single season, and the subtropical species produces four to five generations annually (Johnson pers.comm.). The beetle was still found in disconnected sub-populations rather than continuously throughout the riparian corridor, although the population expanded annually throughout this study. Indeed, the tamarisk beetle was not detected at any of the established photo stations in 2015 and only in one study site (BL-10) in 2016 but appeared at nearly all photo stations in 2017 and 2018. However, in 2019 the tamarisk beetle was again almost entirely absent from the study area.

One of the most notable findings of the 2019 monitoring effort was that despite the lack of direct beetle impact in 2019, the cumulative impact of multiple previous years of defoliation was evident in several study sites. The most striking long-term impacts were recorded in DL-12N, BL-10, and to a lesser extent LFCC-02, where a clear decline in dormant season canopy cover was observed (Figure 15). This result indicates an overall reduction in woody mass due to the physiological stress of repeated and extended defoliation events. Additionally, landscape photography showed an overall decline in vegetation health such that even at the peak of the growing season foliage density did not reach pre-beetle colonization levels (Figure 16). However, hemispherical photography indicated that the observed declines in canopy cover were not outside the natural range of variation for those sites. Indeed, due to overall increases in canopy cover in the 2017 growing season, canopy cover at most locations defoliated by the tamarisk beetle was still higher than it had been in 2015 or 2016 (Figures 10 and 11; Attachment 2).

Nevertheless, it is important to note that hemispherical photography cannot distinguish between live and dead foliage, nor between foliage and woody matter. The tamarisk beetle causes "tamarisk browning" prior to defoliation, such that some foliage dies but remains on the tree. Additionally, hemispherical photos taken in the 2017 through 2019 dormant seasons indicated that the majority of canopy closure in this densely vegetated riparian study system can be attributed to non-foliar vegetation ("winter" samples in graphs in Figures 10 and 11 and Attachment 2). However, the difference between live foliage cover and dead or woody cover is likely meaningful to a breeding bird that depends on that foliar cover for nest concealment and thermoregulation. The tamarisk beetle controls tamarisk by repeatedly defoliating the plant over multiple growing seasons, eventually causing it to lose woody mass and vegetation (Dudley 2005). Multiple years of defoliation may eventually result in canopy cover declines to a level below the 70 to 95 percent range of variation in canopy cover currently observed in the study area. Dense canopy cover is a critical component of SWFL breeding habitat (Stoleson and Finch 2003) and even small decreases in canopy cover can have a strong negative influence on SWFL habitat suitability, nest success, and productivity (Paxton et al. 2011).

Changes in canopy cover can affect breeding flycatchers both directly, through loss of nest concealment, and indirectly through changes in microclimate. In years in which the tamarisk beetle was present in the study area, a notable difference in temperature variation was observed at hemispherical photography stations that were impacted by the tamarisk beetle and those that were not, both between and within study sites, such that temperature was higher at impacted locations (Dillon and Moore 2017, 2018). One of the impacts of an overall, lasting decline in canopy cover as a result of repeated defoliation is that impacted habitat becomes hotter and drier without the foliar cover to create the cool and humid microclimate characteristic of desert riparian areas. This was particularly evident in DL-12N where maximum temperatures in the site remained higher and more variable than sites that had not experienced the same level of past beetle impact. Temperatures in DL-12N were consistently 10 to 15 degrees higher than in DL-12S, the native-dominated control site located only 1 kilometer south, and 5 to 10 degrees warmer than EB-15, another tamarisk-dominated site (Figure 17, 18, 19).

Similar trends in percent relative humidity were observed, such that areas experiencing an overall decline in vegetation health were drier and reached lower relative humidity minimums, than areas that recovered more fully from previous years of defoliation (Figure 20 and 21). This was observed despite the fact that 2019 was a high run-off year and most study sites, including DL-12N, contained standing water. The observed increases in temperature and decreases in relative humidity may be even more accentuated in drier years. Combined, these results suggest that even relatively small losses of canopy cover are leading to hotter and drier flycatcher habitat. Southwestern Willow Flycatchers are a riparian obligate species that rely on the cooler temperatures and increased humidity of desert riparian areas for cover, a reliable food supply, and to satisfy the thermoregulatory requirements of their eggs and offspring. The loss of canopy cover caused by the tamarisk beetle may negatively impact the suitability of current breeding habitat by causing changes in any of these habitat characteristics.

Ultimately, changes in SWFL territory numbers and nest success will be the best indicators of the impacts that the tamarisk beetle has on these populations. If the same study sites are defoliated annually, a gradual decline in nest success and territory numbers might be expected. SWFLs have high site fidelity, so it may take several years of defoliation before adult flycatchers stop establishing breeding territories in an impacted site. Declines in nest success may be observed more quickly. Trends in nesting variables within each study site were plotted for 2014 through 2019 and compared to overall trends in the Middle

Rio Grande during that same time period (Figure 24; Moore *In Press*). Territory numbers and nest productivity in the study sites and the entire Middle Rio Grande were both generally stable between 2014 and 2019 (Figure 24). Data from LFCC-02 suggest a possible downward trend in overall productivity in that site, but additional years of data are needed to distinguish whether that trend is simply natural annual variation in the population.
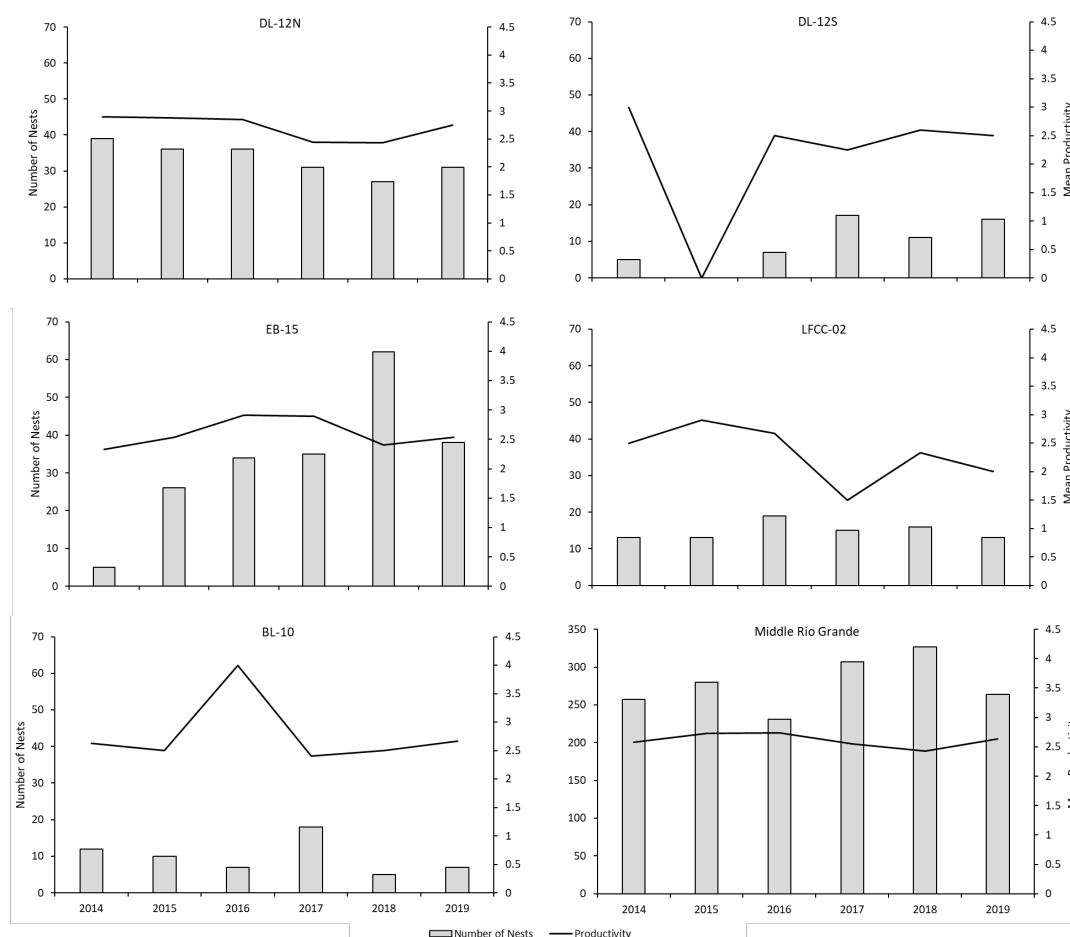
Figure 25. Number of SWFL territories and mean productivity of successful nests by study site, 2014-2019.

Nest success in the monitoring sites in 2019 was approximately 30 to 40 percent, a rate that is average for most songbirds and which mirrored the nest success rate observed in the Middle Rio Grande as a whole (Figure 25; Moore *In Press*). DL-12S, the native-dominated control site, stood out as having a slightly higher nest success rate that appeared to be following an increasing trend, perhaps reflecting the annual increases in habitat extent and suitability in that site. On the other hand, nest success in LFCC-02 has decreased annually for the past several breeding seasons, further suggesting the potential for an overall population decline in that site. The vast majority of nest failures were attributed to nest predation in all sites (Moore *In Press*). One of the most direct expected effects of browning and defoliation by the tamarisk beetle is the loss of nest concealment, which could lead to increased nest predation or parasitism rates. Initial data suggest that defoliation also results in hotter and more arid habitat conditions, which may in turn lead to increases in nest abandonment or decreased hatching success.

Discussion

If the potential effects to SWFL habitat from tamarisk beetle defoliation contributes to or exacerbates low nest success rates on the Middle Rio Grande in the coming years, population declines are likely.
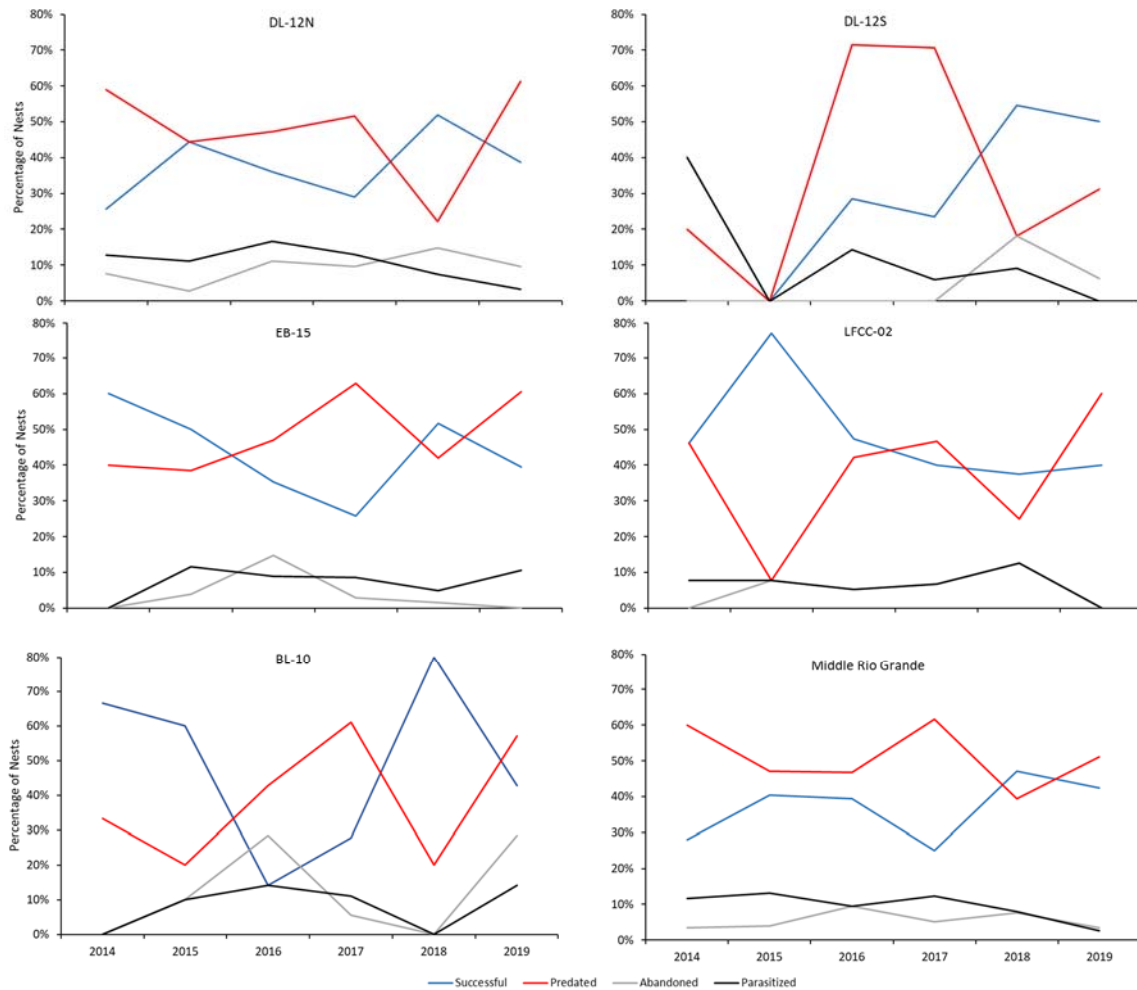
Figure 26. Outcome of SWFL nests by study site, 2014-2019.

Additionally, shifts in nesting substrate may be observed. The percentage of SWFL nests constructed in tamarisk has increased as willow has declined and tamarisk has become dominant in the Middle Rio Grande riparian vegetation community. The vast majority of nests in these study sites, and throughout the Middle Rio Grande, are currently constructed in tamarisk (Figure 26; Moore *In Press*). However, these sites are not monotypic tamarisk and SWFLs are known to nest in coyote willow (*Salix exigua*) as well as Goodding's willow (*Salix gooddingii*), cottonwood (*Populus deltoides*), seep willow (*Baccharis salicifolia*), and Russian olive (*Eleagnus angustifolia*). DL-12S is the notable exception to this trend among the sites in this study, with the majority of the site being dominated by native vegetation. SWFLs may respond to tamarisk beetle defoliation by increasingly using these other species for nesting. Indeed, SWFLs on the Virgin River, UT shifted from nesting primarily in tamarisk to nesting primarily in coyote willow in the years following acute breeding season defoliation by the tamarisk beetle (Edwards 2017). However, shifts in substrate use would still be limited by availability and would likely not be exclusive from declines in nest success or site-specific territory numbers.

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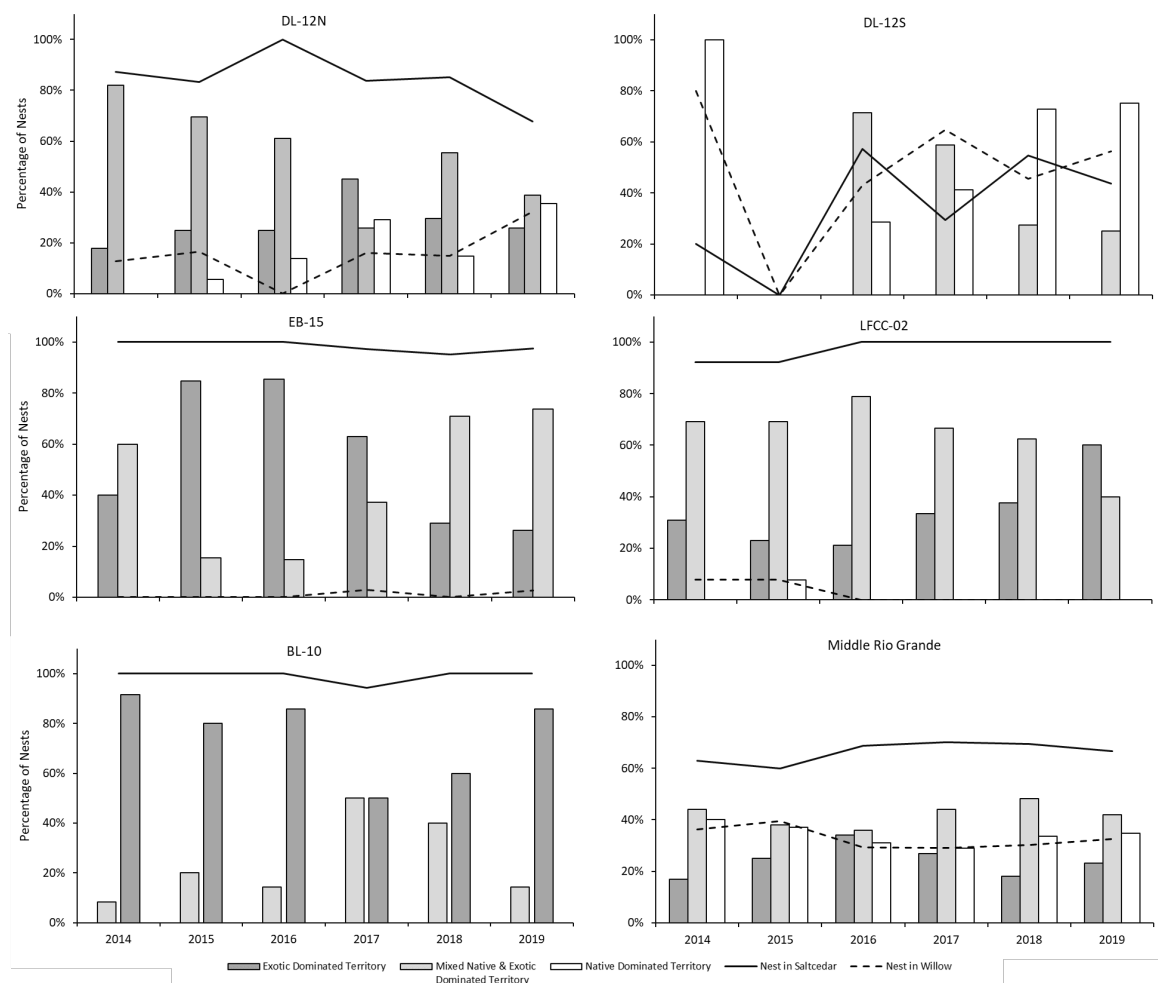
Figure 27. SWFL nesting substrate use and territory dominance by study site, 2014-2019.

The timing of tamarisk beetle defoliation in future years will likely be an important factor in determining the impact that defoliation has on breeding SWFLs. Severe browning and defoliation were not observed in occupied SWFL territories on the Middle Rio Grande until relatively late in the SWFL breeding season (mid- to late July) in 2016 (pers. obs.). In 2017 and 2018 defoliation began early in the season in some sites, but later in the breeding season in others. In 2019 the tamarisk beetle was nearly absent from the study area, with the first evidence of defoliation only observed late in the breeding season at a small number of photo stations. Clearly, interannual movements of the beetle population can be highly unpredictable, but the impacts on SWFL habitat can be severe and lasting. If defoliation occurs after SWFLs establish territories and are in the early stages of nesting, there is the potential to create an ecological trap in which a bird settles in apparently suitable, foliated vegetation only to have that vegetation defoliated shortly after they begin nesting (Paxton et al. 2011). The timing of defoliation on the Middle Rio Grande will be an important determinant of within-year SWFL nest success and productivity. Additionally, the data suggest that repeated defoliation events are beginning to impact overall vegetation health and microclimate conditions in some sites. Repeated defoliation in these areas has the potential to severely negatively impact habitat suitability for breeding SWFLs.

Conclusions

The tamarisk beetle was documented throughout the Rio Grande in the summers of 2017 and 2018 but was nearly absent in 2019. Although Reclamation biologists had only documented tamarisk beetle defoliation in a small number of occupied SWFL breeding territories late in the breeding season in 2016, defoliation was pervasive in two of four study sites in 2017 and four of six study sites in 2018. Declines in canopy cover and increases in ambient temperature and aridity were observed in association with tamarisk beetle colonization of these study sites. Despite the lack of direct defoliation in most areas during the 2019 breeding season, the impact of multiple previous years of defoliation was still evident in the observed long-term declines in vegetation health and increases in the temperature and aridity of breeding habitat. Ultimately, the timing, severity, and extent of tamarisk beetle defoliation will determine the level of impact on breeding SWFLs.

Photographic monitoring of changes in canopy cover and vegetation composition in SWFL habitat will provide an important tool to assess the possible need for more active management of the riparian ecosystem. If changes in SWFL productivity and nest success raise concerns about the species' recovery, such modifications of management strategies may be deemed necessary. For example, efforts to mitigate the impact of tamarisk defoliation on the Virgin River watershed SWFL population led to the development of a collaborative riparian restoration plan (Dudley and Bean 2012). Additionally, overbank flooding of the riparian area during the winter is known to kill tamarisk beetle populations, which are in diapause in the soil at that time. Indeed, overbank flooding in BL-10 in 2017 may explain why the tamarisk beetle was not observed at hemispherical photo stations in that site in 2017 or 2018 despite initial colonization in late summer 2016. It may have also contributed to the general absence of the tamarisk beetle throughout much of the Rio Grande in 2019, a high run-off year in which the river overbanked into the floodplain in many areas. Continued monitoring of beetle-induced changes in riparian vegetation and associated changes in SWFL demographic parameters will provide important data regarding the need, or lack thereof, of these or other management strategies.

Recommendations

- Continue landscape and hemispherical photography and microclimate monitoring at established sampling locations to quantify impacts of the tamarisk beetle on riparian habitat.

- Take hemispherical photographs once annually or bi-annually during the non-growing (winter) season to determine changes in tamarisk foliage density versus woody mass. Winter measurements of canopy cover, after abscission has occurred for this deciduous shrub, when compared to the preceding summer measurements will enable a calculation of the amount of summer canopy cover comprised of foliage versus woody material - a distinction likely to be important for a breeding bird.

- Further characterize the extent of defoliation and associated changes in habitat structure and suitability by quantifying species composition, tree health, percent defoliation, etc. in a fixed radius plot around each hemispherical photography station. Consideration of the potential mechanistic impacts of the tamarisk beetle on flycatcher habitat suitability would be a useful guide in selecting the most important habitat variables to quantify.

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Attachment 1: Photo Station Locations

Attachment 1A: Hemispherical Photography Station Locations

Station Number	Location (UTM NAD 83 Zone 13S)	Distance to Closest Previous Season SWFL Pair (m)
BL-10		
HPS-1	338241, 3824809	79
HPS-2	338230, 3824855	31
HPS-3	338211, 3824879	12
HPS-4	338205, 3824922	13
HPS-5	338210, 3824933	15
HPS-6	338201, 3825011	19
HPS-7	338184, 3825049	6
HPS-8	338172, 3825093	16
DL-12N		
HPS-1	306804, 3717696	17
HPS-2	306789, 3717595	9
HPS-3	306770, 3717505	6
HPS-4	306745, 3717505	6
HPS-5	306721, 3717451	9
HPS-6	306700, 3717411	3
HPS-7	306671, 3717374	14
HPS-8	306672, 3717339	9
HPS-9	306697, 3717327	10
HPS-10	306694, 3717309	9
LFCC-5B		
HPS-1	314963, 3725319	7
HPS-2	314996, 3725102	6
HPS-3	315008, 3725056	52
HPS-4	315032, 3724960	39
HPS-5	315035, 3724872	11
HPS-6	315048, 3724840	13
HPS-7	315057, 3724793	15
HPS-8	315073, 3724697	15
EB-15		
HPS-1	297104, 3691767	5
HPS-2	297070, 3691733	7
HPS-3	297029, 3691693	25
HPS-4	296997, 3691646	14
HPS-5	296978, 3691597	10
HPS-6	296959, 3691547	27

Station Number	Location (UTM NAD 83 Zone 13S)	Distance to Closest Previous Season SWFL Pair (m)
HPS-7	296985, 3691503	14
HPS-8	297005, 3691452	3
HPS-9	297022, 3691481	30
DL-12S		
HPS-1	306294, 3715935	4
HPS-2	306321, 3715933	9
HPS-3	306349, 3715966	4
HPS-4	306378, 3716013	9
HPS-5	306379, 3716029	12
HPS-6	306397, 3716116	18
HPS-7	306413, 3716128	10
HPS-8	306442, 3716187	5
LFCC-02		
HPS-1	310943, 3720684	9
HPS-2	311010, 3720661	5
HPS-3	311026, 3720632	7
HPS-4	311036, 3720593	13
HPS-5	311089, 3720584	10
HPS-6	311126, 3720642	10
HPS-7	311055, 3720685	10
HPS-8	311028, 3720695	2
HPS-9	311006, 3720713	9

Attachment 1B: Landscape Photography Station Locations

Station Number	Location (UTM NAD 83 Zone 13S)	Bearing (degrees)	Distance to Closest Previous Season SWFL Pair (m)
BL-10			
LPS-1	338220, 3824825	20	60
LPS-2	338229, 3824891	260	10
LPS-3	338235, 3825096	170	48
LPS-4	338249, 3824948	180	40
DL-12N			
LPS-1	306716, 3717311	220	18
LPS-2	306702, 3717381	350	13
LPS-3	306769, 3717522	195	17
LPS-4	306822, 3717544	210 & 300	28
LPS-5	306837, 3717638	340 & 250	28
LPS-6	306823, 3717703	270	4
LFCC-5B			
LPS-1	315087, 3724714	215	27
LPS-2	315048, 3724903	130	23
LPS-3	315020, 3725179	175	30
EB-15			
LPS-1	297052, 3691701	310 & 230	25
LPS-2	296999, 3691616	294 & 223	33
LPS-3	296951, 3691525	218 & 166	22
LPS-4	338249, 3824948	260 & 196	8
DL-12S			
LPS-1	306422, 3716193	130	25
LPS-2	306386, 3716163	185	50
LPS-3	306357, 3715983	255	27
LPS-4	306322, 3715980	150	27
LPS-5	306277, 3715921	130	23

Station Number	Location (UTM NAD 83 Zone 13S)	Bearing (degrees)	Distance to Closest Previous Season SWFL Pair (m)
LFCC-02			
LPS-1	310986, 3720770	160	50
LPS-2	311099, 3720601	200	14
LPS-3	311037, 3720609	160	6
LPS-4	311007, 3720622	0	15
LPS-5	310916, 3720693	150	32

Attachment 2: Canopy Cover by Site Pre- & Post-Colonization by the Tamarisk Beetle

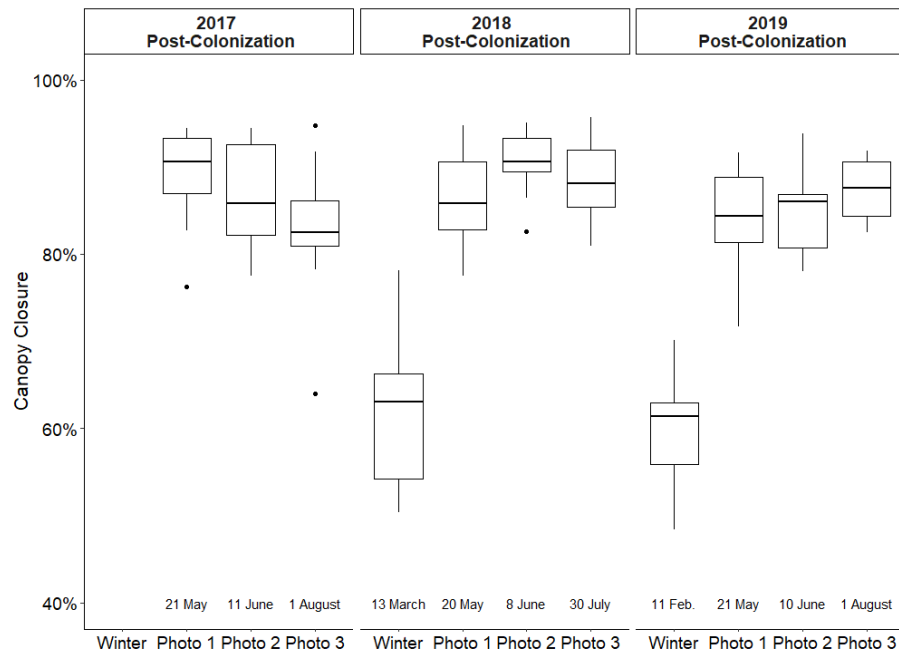**Sites impacted by the tamarisk beetle in 2017/ 2018 breeding season:**

Figure 1. Trends in canopy cover at EB-15 post-colonization by the tamarisk beetle. EB-15 was not sampled prior to 2017.
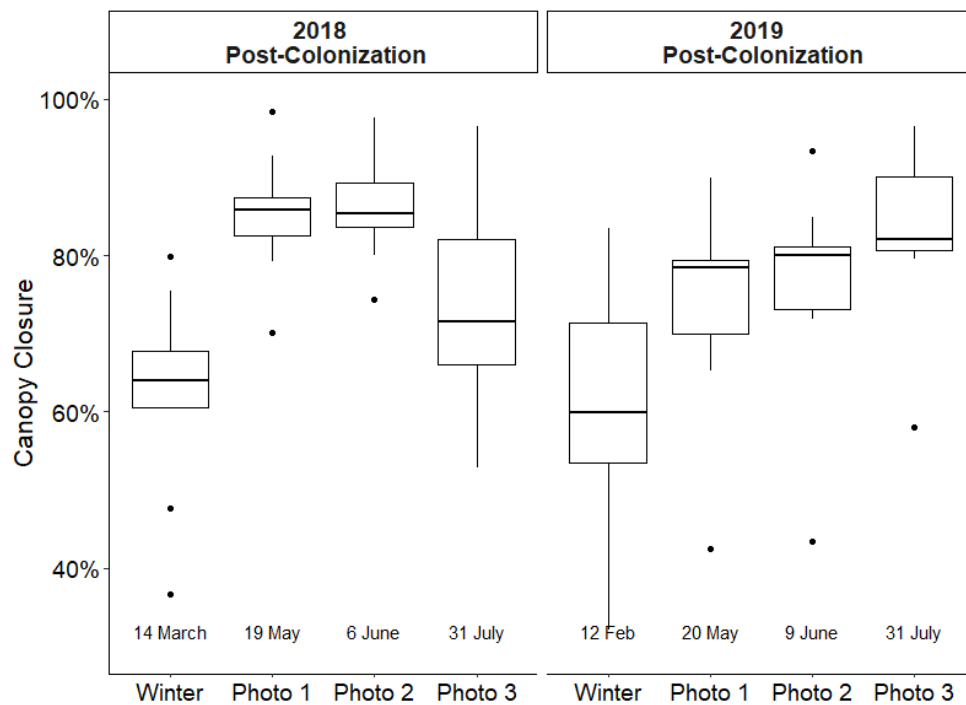

Figure 2. Trends in canopy cover at LFCC-02 post-colonization by the tamarisk beetle.

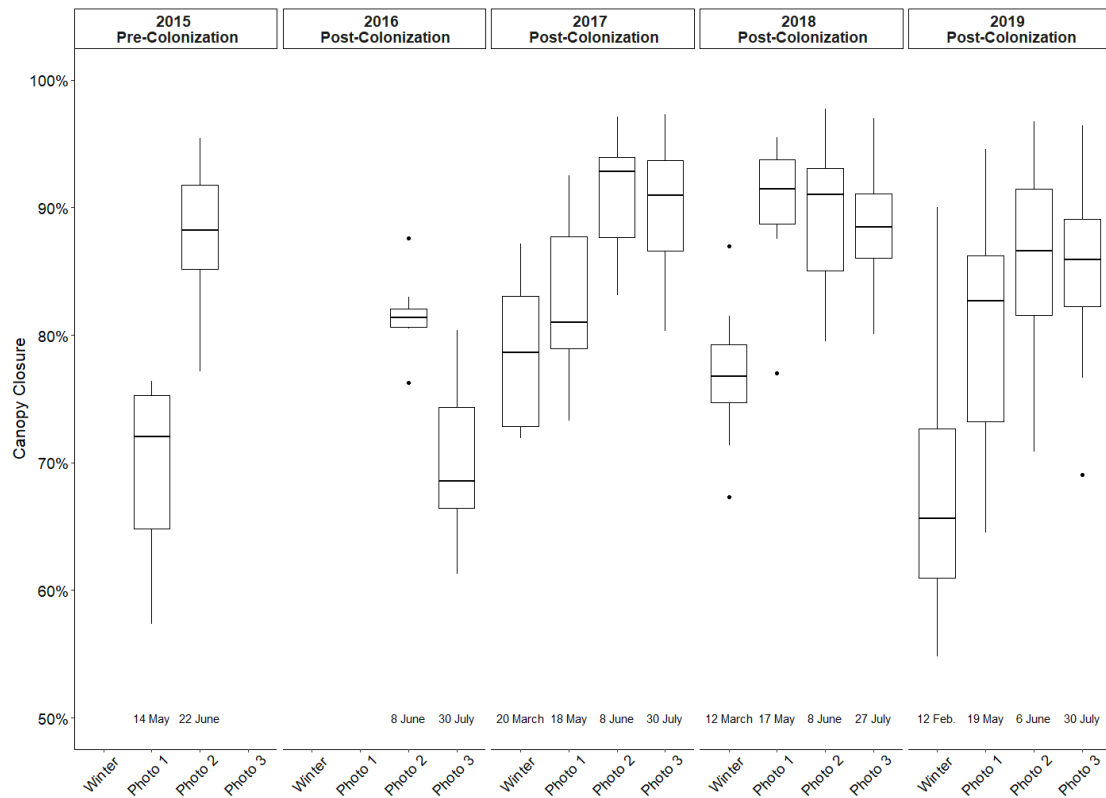**Sites not impacted by the tamarisk beetle in 2017/ 2018 breeding season:**

Figure 3. Trends in canopy cover at BL-10 pre- and post-colonization by the tamarisk beetle.
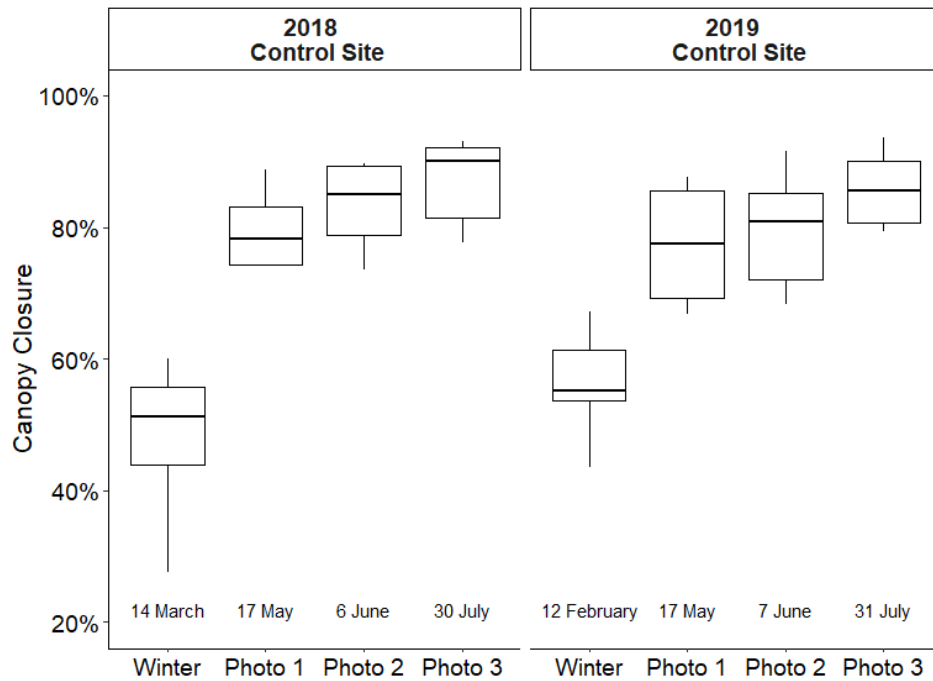

Figure 4. Trends in canopy cover at DL-12S, a native vegetation-dominated control site not impacted by the tamarisk beetle.

PEER REVIEW DOCUMENTATION

PROJECT AND DOCUMENT INFORMATION

Project Name __Photographic Monitoring of Defoliation by the Tamarisk Beetle__ WOID __OA859, F046A__

Document __Photographic Monitoring of Defoliation by the Tamarisk Beetle: Middle Rio Grande from Belen to Elephant Butte Reservoir, NM__

Document Date __January 2020__

Team Leader __Dave Moore, 86-68290, Wildlife Biologist__

Document Author(s)/Preparer(s) __K. Dillon and D. Moore__

Peer Reviewer __Tori Barron, 86-68290__

Peer Reviewer _____

REVIEW REQUIREMENT

Part A: Document Does Not Require Peer Review

Explain _____

Part B: Document Requires Peer Review: SCOPE OF PEER REVIEW

Peer Review restricted to the following Items/Section(s):   Reviewer:

____Complete Document Subject to Review____   __Tori Barron__

REVIEW CERTIFICATION

Peer Reviewer - I have reviewed the assigned Items/Section(s) noted for the above document and believe them to be in accordance with the project requirements, standards of the profession, and Reclamation policy.

Reviewer:____Tori Barron_____   Review Date: __January 23, 2020__   Signature: __TORI BARRON__ Digitally signed by TORI BARRON Date: 2020.01.23 08:02:33 -07'00'

Reviewer:_____   Review Date: _____   Signature: _____

I have discussed the above document and review requirements with the Peer Reviewer and believe that this review is completed, and that the document will meet the requirements of the project.

Team Leader: __Dave Moore_____   Date: __January 23, 2020__   Signature: __STANTON MOORE__ Digitally signed by STANTON MOORE Date: 2020.01.23 11:59:10 -07'00'