Impacts of Tamarisk Leaf Beetle Defoliation and Interactive Effects of Tree Removal on Saltcedar Productivity, Reproductive Effort, and Branch Mortality

Kim Eichhorst, Matt Leister, Ara Winter, and Missy Bacigalupa

University of New Mexico, Department of Biology
Bosque Ecosystem Monitoring Program (BEMP)

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Executive Summary

Since its introduction in the 1800s, saltcedar (*Tamarix* sp.) has become a dominant woody species in riparian corridors of the southwestern United States, where it forms dense stands that alter ecosystem structure, fire potential, and habitat suitability for native plants and animals. Management efforts to control saltcedar have included mechanical removal, herbicide treatment, and the introduction of the tamarisk leaf beetle (*Diorhabda* spp.; TLB) as a method of biological control. While TLBs have become widespread throughout the Middle Rio Grande (MRG), the direct effects of beetle activity on saltcedar productivity and survivability and how those effects interact with active saltcedar removal remain poorly understood.

This white paper evaluates long-term TLB monitoring data from the Bosque Ecosystem Monitoring Program (BEMP) to examine relationships between TLB abundance, TLB defoliation levels, saltcedar productivity, and twenty years of management across multiple sites in the MRG. Sites include both areas where saltcedar has been removed and areas where saltcedar remains relatively unmanaged but is subject to repeated defoliation by TLB. Indicators examined include TLB abundance, saltcedar cover, saltcedar leaf fall (used as a proxy for productivity), saltcedar reproductive effort, and percent defoliation.

Preliminary analyses suggest that increases in TLB abundance are associated with measurable, but small, reductions in saltcedar productivity. Across monitored sites, a 1% increase in TLB adult abundance corresponds to an approximate 0.08% decrease in saltcedar productivity. When allowing for a delayed response in tamarisk productivity to changes in beetle abundance, this effect becomes more clear: including lag effects increases the impact of a 1% increase in TLB to a 3% decrease in saltcedar productivity. A 1% increase in defoliation is associated with a 0.14% decrease in saltcedar litterfall in the same year. Estimates of percent defoliation are indicative of cumulative TLB impact and do not rely on capturing TLB abundance at their peak.

However, BEMP data indicate that TLB abundance is not strongly correlated with saltcedar cover, with comparably high beetle abundance occurring at sites where saltcedar cover has been greatly reduced through management strategies. This suggests that beetle populations remain active at these sites despite changes in host plant density and that annual population numbers in an area oscillate independent of saltcedar density.

Evidence from long-term monitoring also suggests that saltcedar management combined with TLB presence may produce minimal additive effects on saltcedar productivity in some areas. Sites with a history of removal often show larger deviations in leaf fall and canopy productivity compared to sites influenced by the beetle alone. Strong yearly variation in beetle populations and vegetation responses highlight the importance of long-term datasets when evaluating outcomes of biological control.

Continued monitoring of TLB and saltcedar response will be necessary to determine whether reductions in saltcedar productivity result in lasting changes in saltcedar density. Understanding

how biological control interacts with active vegetation management is crucial for developing effective riparian restoration strategies along the MRG.

Introduction

Saltcedar, or tamarisk (*Tamarix* spp.), was introduced to the US from Eurasia in the late 1800's as an ornamental plant (Sher, 2013). Later, it was planted by the U.S. Army Corps of Engineers along streams and rivers, including the Rio Grande, for bank stabilization and as a windbreak (Sher, 2013). It then spread rapidly along rivers in the Southwest, creating monotypic stands. River regulation efforts, primarily implemented to reduce flooding and control river flows, accelerated the displacement of native species. While cottonwoods and willows rely on spring overbank flooding for successful germination and seedling survival, saltcedar is not restricted to the strict timing of high spring flows for germination, allowing it to outcompete these native species under the new reduced flooding regime. Over time, saltcedar became the second-most dominant woody species in river systems across the southwestern United States (Kennard et al., 2016). This success raised multiple concerns, starting with saltcedar's potential for high water consumption in water-limited ecosystems (Cleverly 2013). Additionally, saltcedar has the ability to take up and exude salts onto the soil surface, inhibiting native plant germination. It is a known ladder fuel that tends to form thickets, supporting hot, intense fires. Following fires, saltcedar is able to quickly resprout, growing rapidly and flowering within months, unlike native species adapted to the disturbance of flooding rather than fire (Stuever, 1997; Eichhorst et al., 2012). Saltcedar negatively impacts native species and habitat quality, particularly by depositing salt through leaf drop, competing for water and space, and forming dense understory growth. Bateman et al. (2008) suggested native lizard diversity is lower in monotypic saltcedar stands when compared to mixed stands. Similarly, Brand et al. (2008) found lower bird diversity in monotypic tamarisk stands, and Ellis (1995) found more unique bird species in cottonwood stands than in tamarisk, although birds can and do use tamarisk for nesting and foraging.

Concerns about saltcedar abundance, spread, and impacts led to increased public scrutiny and focused research in the mid-1900's. Concurrent management efforts focused on controlling or eradicating saltcedar from riparian sites. Since then, saltcedar removal projects have been implemented by various federal, state, and local agencies along with private landowners with varying success. These management efforts have been tested and applied for decades and include strategies such as mechanical removal, herbicide application, fire, flooding, and varying combinations, most commonly mechanical removal and herbicide (Taylor and McDaniel 1998). These strategies tend to be expensive and are not always successful. Saltcedar is highly resilient to mechanical cutting, herbicide, fire, and flooding; cut stems often resprout, and herbicide-treated stems may recover if not re-treated. By the 2000s, the difficulties of saltcedar control led to consideration of bringing in biocontrol agents to suppress saltcedar.

Following years of discussion and trials, the tamarisk leaf beetle (TLB) (*Diorhabda* spp.) was released in Nevada, Utah, Colorado, and Wyoming in 2001 and Texas in 2004 as a biological control agent for saltcedar. Referencing its native range, the tamarisk leaf beetle species released in Nevada, Utah, and Colorado was not expected to survive below the 38th parallel,

which coincides with Southern Colorado and Southern Utah. This was an important consideration, as below this parallel the endangered Southwestern Willow Flycatcher (*Empidonax traillii extimus*) had been shown to use saltcedar for nesting and foraging. These early concerns about the negative impact of TLB on SWFL successfully nesting in saltcedar resulted in the USDA initially excluding New Mexico, Arizona, and California from TLB introductions. New Mexico later obtained permits for release at five sites after 2005. *D. elongata* was released at one site in Artesia, New Mexico in 2005 and *D. carinulata* was released at two sites in New Mexico. Continuing concerns and lawsuits resulted in the cancellation of release programs in 2010 in all states.

Novel conditions released northern-released TLB species from the constraints of its native territory and the beetle quickly expanded south of the 38th parallel. The range of the TLB species released in Texas and southern New Mexico expanded northward, meeting with that of the northern-released species. Although not known to hybridize in their native Eurasian range, intermediate morphotypes of the species were found in south-central New Mexico (Knutson et al. 2019), and hybridization was later confirmed by genetic analysis (Stahlke et al., 2021). In response to this unexpected expansion, the Tamarisk Coalition was formed in 2007 to help land managers manage saltcedar and monitor the expansion of TLB along with other agencies. The Bosque Ecosystem Monitoring Program (BEMP) began sampling for TLB in the Middle Rio Grande in 2013 and continues to monitor TLB annually.

TLB population densities and defoliation potential decreased in NM with highest populations and defoliation persisting for only 3-4 years after release, with current populations persisting along the MRG but not in sufficient densities for large area defoliation (Knutson et al., 2019).

This white paper uses long-term data collected by BEMP to evaluate relationships between TLB abundance, defoliation of saltcedar, saltcedar productivity, and management history across sites in the MRG. Sites include both areas where saltcedar has been removed and areas where saltcedar remains relatively unmanaged, but is subject to repeated defoliation by TLB.

Study Area

Monitoring sites included in the analysis are located along the middle and lower Rio Grande in New Mexico, spanning from Santo Domingo (Kewa) Pueblo in the north to Mesilla Valley State Park in the south. In total, 28 long-term BEMP sites, one short-term site, and three Greater Rio Grande Watershed Alliance (GRGWA) sites were included in the analyses, representing a 150-mile riparian corridor. Not all sites were monitored in all years. Sites occurred within the riparian corridor of the Rio Grande, which is historically characterized primarily by Cottonwood-dominated bosque with mixed native and non-native understory. The Rio Grande in this area is highly regulated by dams, levees, and irrigation and clear water ditches. These modifications have altered historic flooding events that support the recruitment of native cottonwood and willow communities. Many portions of the bosque now contain older senescing cottonwoods and dense stands of exotic vegetation including saltcedar.

Tamarisk Leaf Beetle Monitoring

BEMP began monitoring TLB populations at 22 sites in 2013 to track the abundance and movement of these species through the MRG. Sites were monitored monthly May through August, with a subset of sites monitored in September to detect late season activity. By 2015, BEMP reduced TLB monitoring to 16 sites. Due to exotic removal projects, fires, and funding shifts over the years, there are now 9 BEMP sites and one GRGWA site monitored for TLB, strategically located to maximize coverage in the MRG.

At each site, five saltcedar trees are selected for repeat monitoring. Trees are sampled using a sweepnet to quantify TLB presence and abundance. For each tree, counts of each TLB life stage are recorded, as well as counts of other arthropods suspected to feed on saltcedar or predate on saltcedar herbivores, such as spiders, ants, and ladybugs. In addition to arthropod collections, estimates of percent green foliage, percent yellow defoliation (attributed to leafhopper activity), percent brown defoliation (attributed to TLB activity), percent refoliation, percent flowering, and percent dead branches are also recorded.

This monitoring allows for the tracking of both TLB populations and the saltcedar response to herbivory across the growing season.

Saltcedar Productivity and Reproductive Monitoring

Plant productivity is assessed through monthly leaf litter collections from ten 1 m² litterfall tubs per site. Leaf fall, woody branches, and reproductive material are collected. Leaves and reproductive parts are dried, sorted by species, and weighed. These data are used as a proxy for plant productivity, branch loss, and reproductive effort, as well as to track phenologic shifts.

This dataset provides long-term measures of saltcedar abundance, productivity, phenologic shifts, and changes in reproductive effort across the MRG. Litterfall is reported in g/m^2.

Site Management History

Several monitoring sites have experienced saltcedar management and removal by GRGWA and other managers. These efforts provide an opportunity to compare sites influenced primarily by TLB and those with additional saltcedar treatment.

At the Valencia Clear BEMP site, saltcedar was treated using cut-stump methods followed by herbicide in 2003, with a follow-up clearing in 2008 and 2012. At the Reynolds Cleared BEMP site, saltcedar was also cleared in 2003, and retreated using cut-stump methods followed by herbicide treatment in spring 2012. Reynolds Forest BEMP site was treated in 2012 using cut-stump methods, followed by herbicide treatment. The Rio Abajo site remained untreated

until prior to the 2023 TLB collections when the Valencia Soil and Water Conservation District cleared much of the exotics in the area; no evidence of herbicide treatment was noted.

In addition to the GRGWA projects, other BEMP sites have undergone exotic removal over the years. These sites have allowed for a more in depth modeling of the interaction between exotic removal and TLB presence. Some sites had exotic tree removal before site establishment; many of BEMP's sites have had some sort of exotic removal or clearing treatment (Table 1). Valencia Cleared and Reynolds Cleared were cleared in 2003, just before sites were installed, and later had follow-up clearing treatments. Valencia Forest was cleared following the 2007 fire. Lemitar had saltcedar removal prior to the 2024 collection and TLB monitoring continued for one year post-treatment.

These management histories allow a comparison between sites with saltcedar treatment and relatively untreated sites. All sites are subject to repeated defoliation by TLB.

Table 1. Exotic removal / clearing events at sites monitored for TLB.

Site name	Year of exotic removal/clearing
Santo Domingo	2007
Santa Ana	1998
Sandia	2012
Badger	2017
Diversion	2003
Minnow	2013
Calabacillas	2004, 2009, 2023
Alameda	2004
Montano	2004 (fire in 2003)
Rio Grande Nature Center	2004
Route 66	2008
BioPark	2007, 2024
AOP	2003
Bosque Farms	pre 2009
Reynolds Forest	2012, 2025
Reynolds Cleared	2003, 2012
Valencia Cleared	2003, 2008, 2012
Valencia Forest	2008, 2009 (fire in 2007)
Crawford	2008
Lemitar	2024
Mesilla	2011

Analyses

We used TLB adult beetle abundance, percent defoliation, percent dead branches, percent flowering, monthly and annual leaf fall, monthly and annual reproductive parts litterfall, and site history (clearing/exotic removal and fires) to assess the impacts of TLB on riparian sites in the Middle Rio Grande Valley in New Mexico as well as the interactions between TLB and management practices on saltcedar productivity and reproductive effort. Bayesian multilevel log-log with site interaction models were used to analyze these relationships.

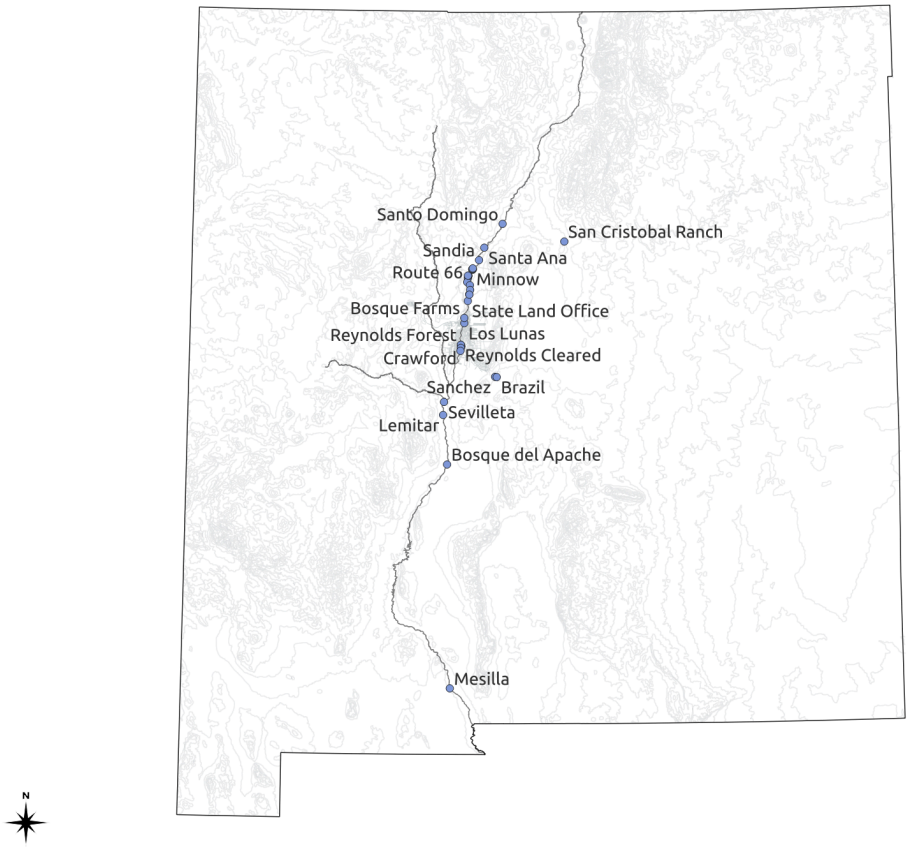

Figure 1. Tamarisk leaf beetle monitoring sites. Not all sites have been monitored in all years. BEMP sites that have been monitored each year starting in 2013 include: Diversion, AOP, Rt 66, Bosque Farms, Reynolds Forest, Valencia Cleared, Crawford, Sevilleta, and Bosque del Apache.

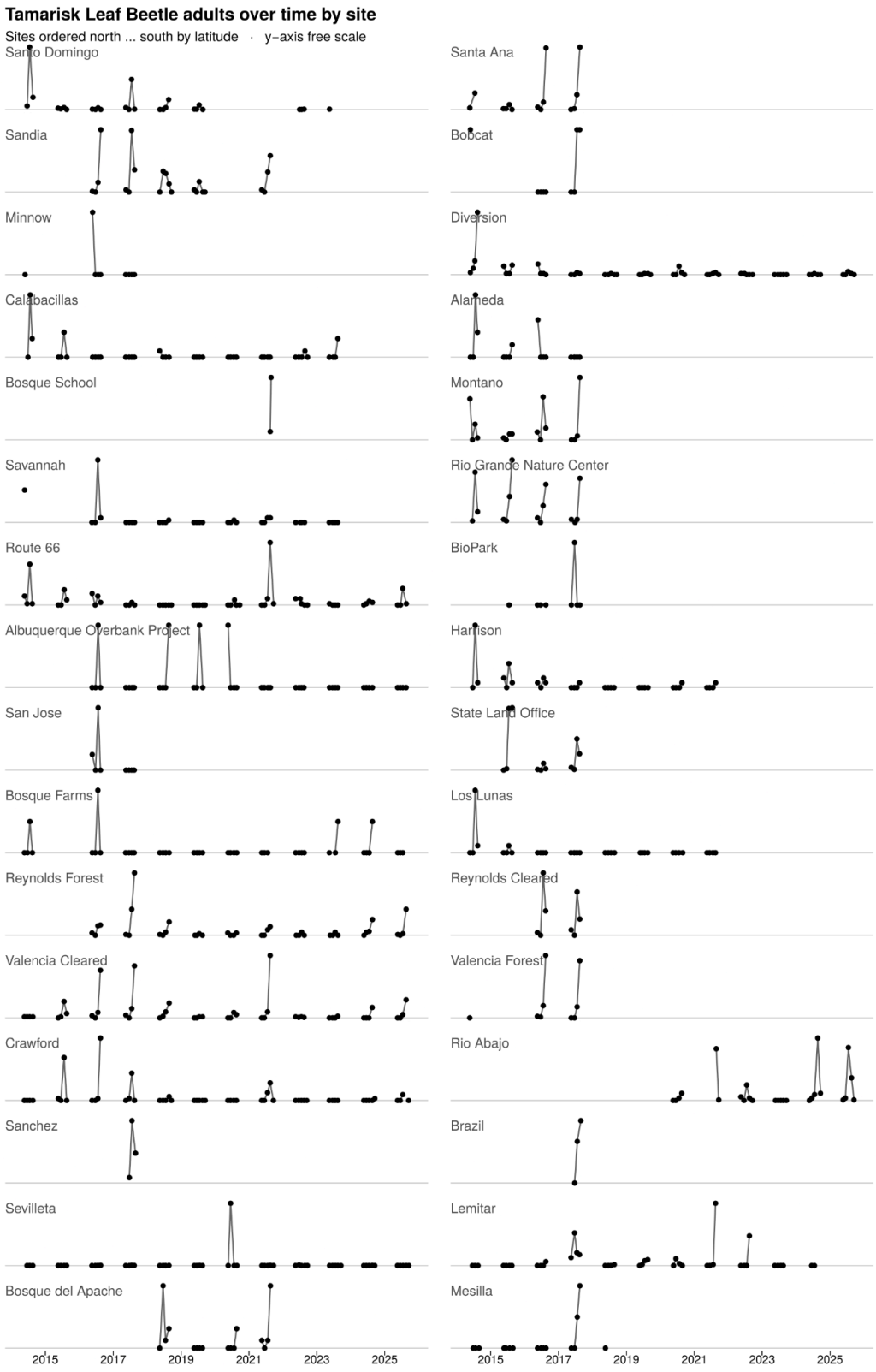

Figure 2a. Sparkline plot (with exact values removed). Showing sums of TLB adults across sites. Not all sites were monitored each year.

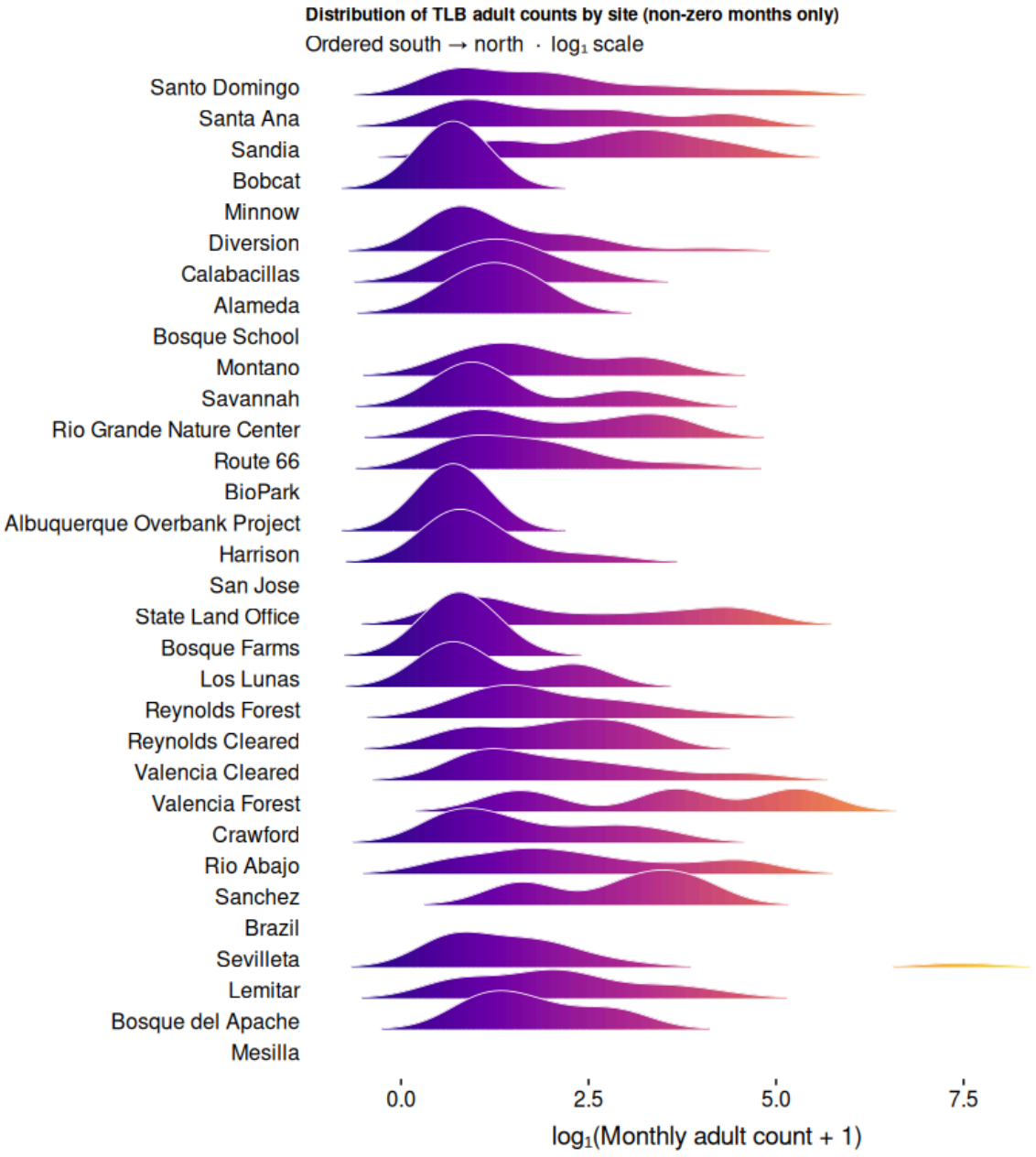

Figure 2b. Ridgeline plot showing all the monthly TLB adult counts recorded at that site across the entire study period. This plot shows the most common non-zero count observed (indicated by the peak); high variance sites have fat, flat ridges; outlier outbreak months pull the tail toward high $\log_1$ values.

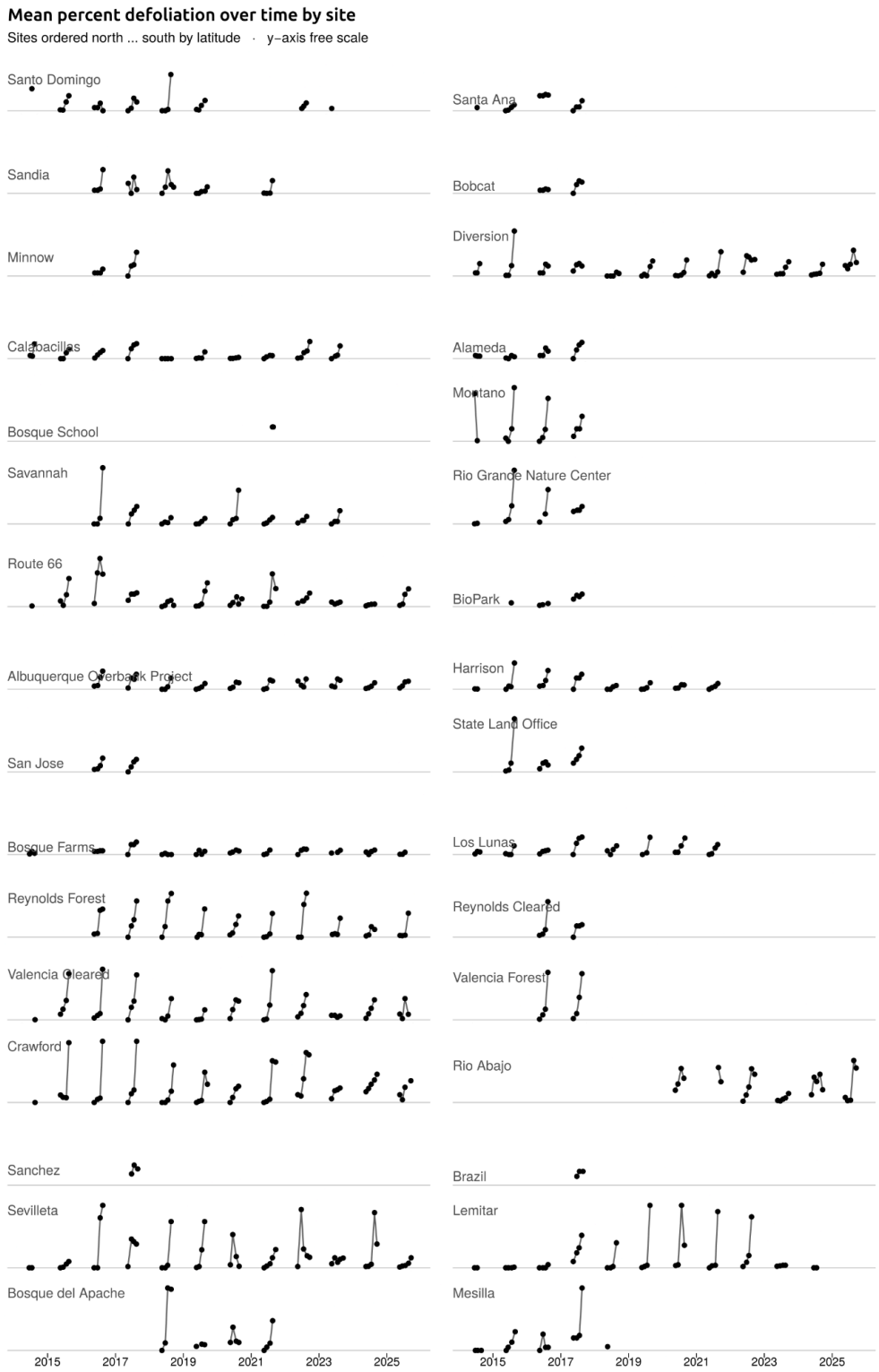

Figure 3. Monthly mean percent defoliation (visual estimate) of saltcedar trees over time. Percent defoliation is a visual indication of herbivory from TLB and leafhoppers and is apparent even after TLB counts have dropped or are no longer present.

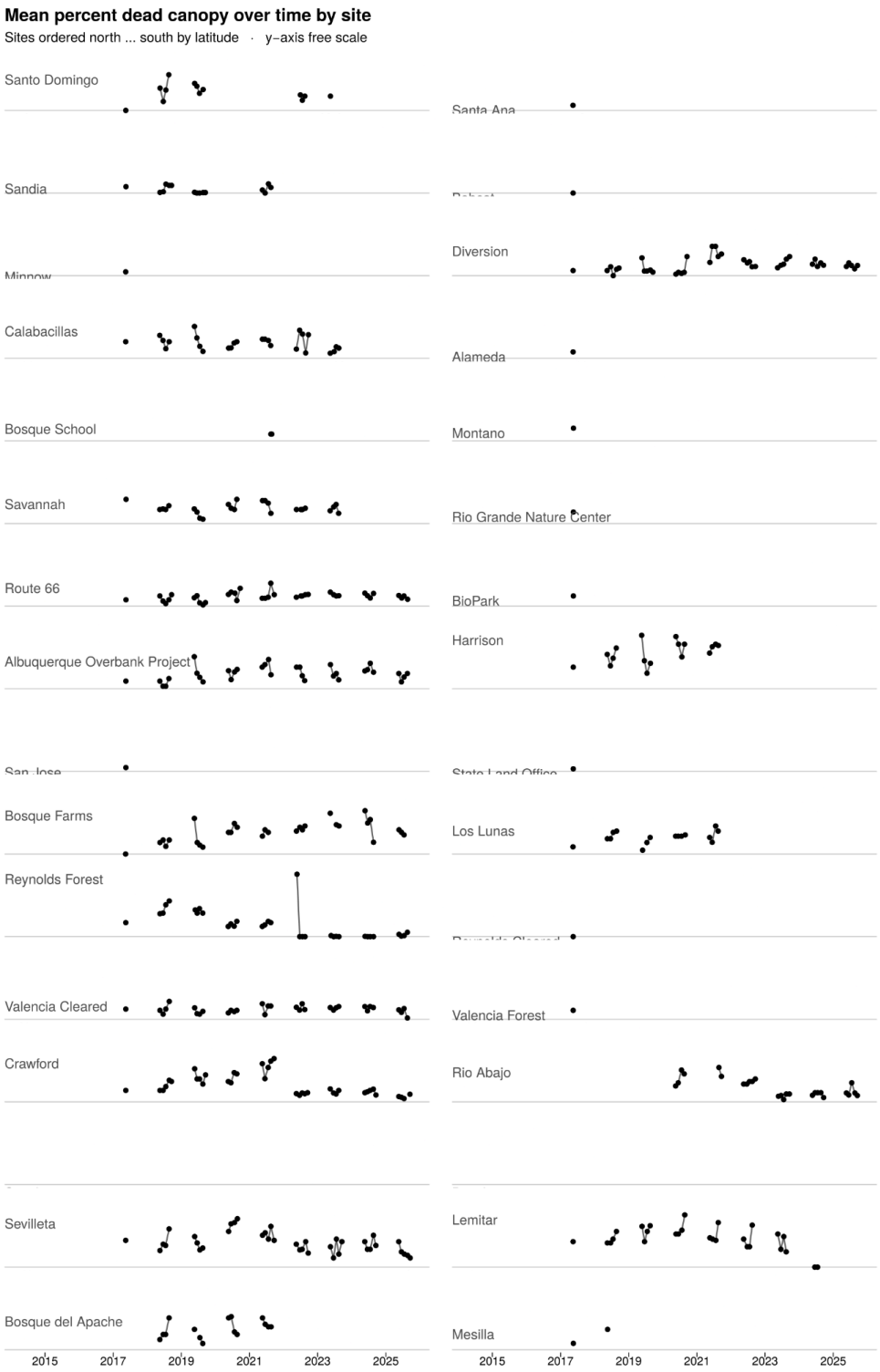

Figure 4. Percent dead branches over time at TLB sampling sites. Branch mortality was likely increased at some sites from repeated sweep net sampling. Sharp drops in percent dead branches (AOP, Diversion, Bosque Farms, Crawford, Sevilleta, Lemitar) indicate a shift to sampling new trees at the site and not plant recovery. The sharp decline seen at Reynolds Forest in 2022 was due to the Big Hole Fire; at Rio Abajo due to clearing of exotic trees.

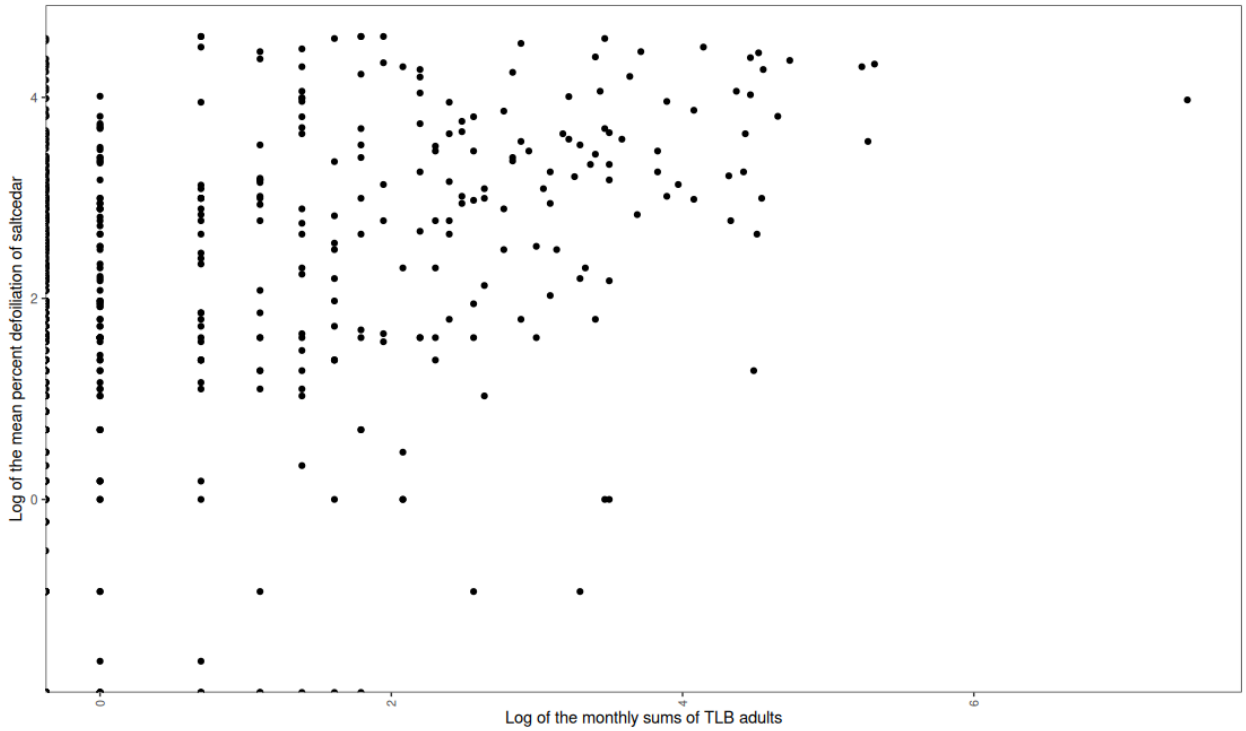

Figure 5. Log of monthly TLB adult sums vs log of mean percent defoliation shows relative changes in defoliation in relation to TLB counts. This indicates higher loads of TLB resulting in increased defoliation levels, even when high abundance isn't fully captured by monthly sampling.

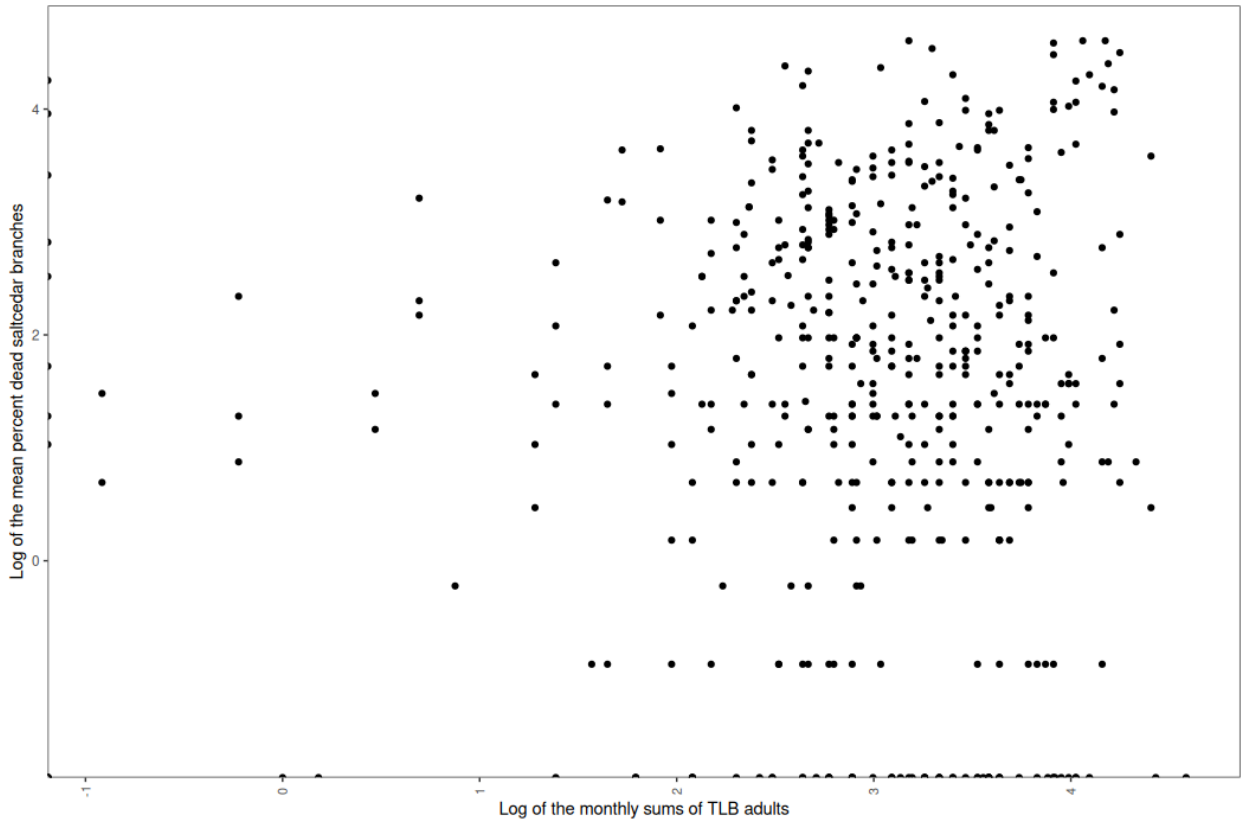

Figure 6. Log of monthly TLB sums vs log mean percent dead branches. This indicates higher branch mortality with higher TLB counts, though not all branch mortality can be attributed to TLB.

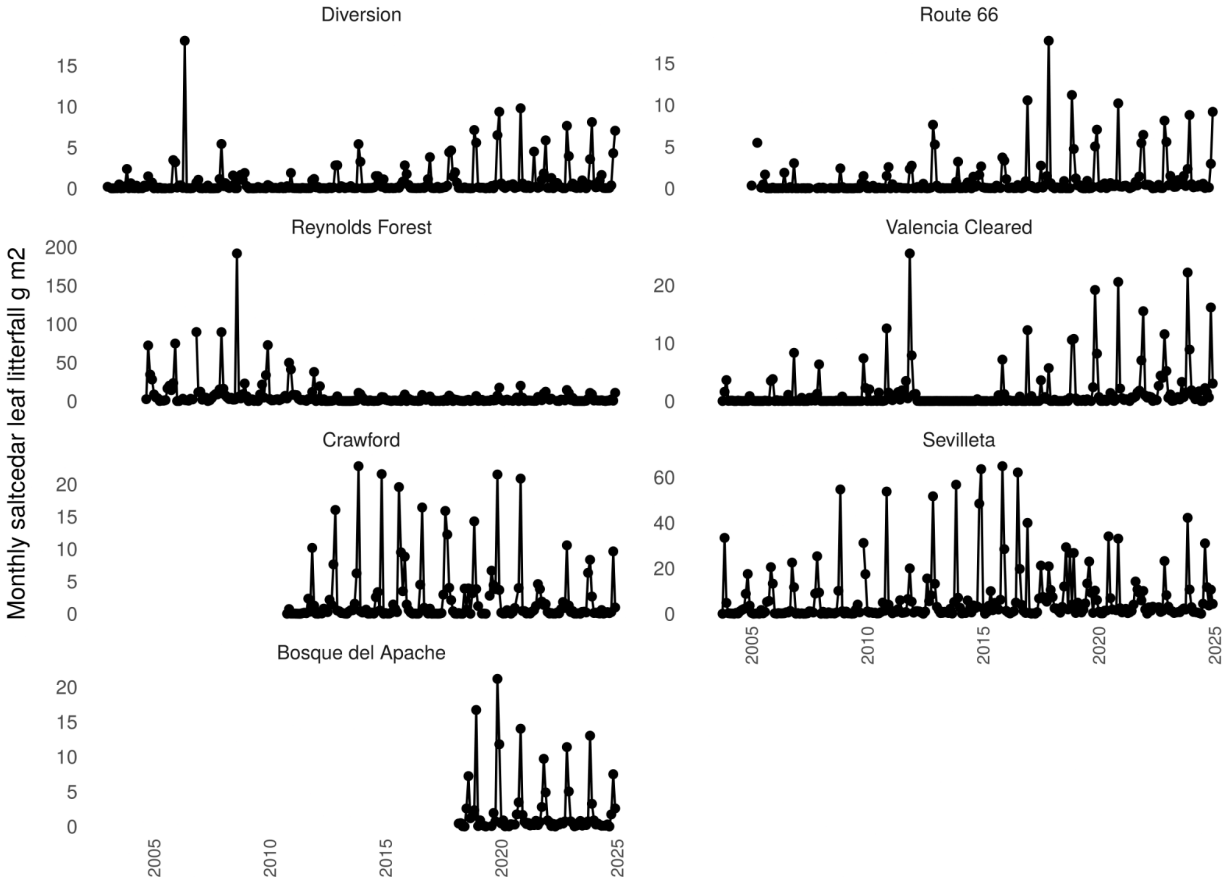

Figure 7. Monthly saltcedar leaf fall at the nine BEMP sites consistently monitored for TLB. The y-axes are on different scales for each site to better show the data.

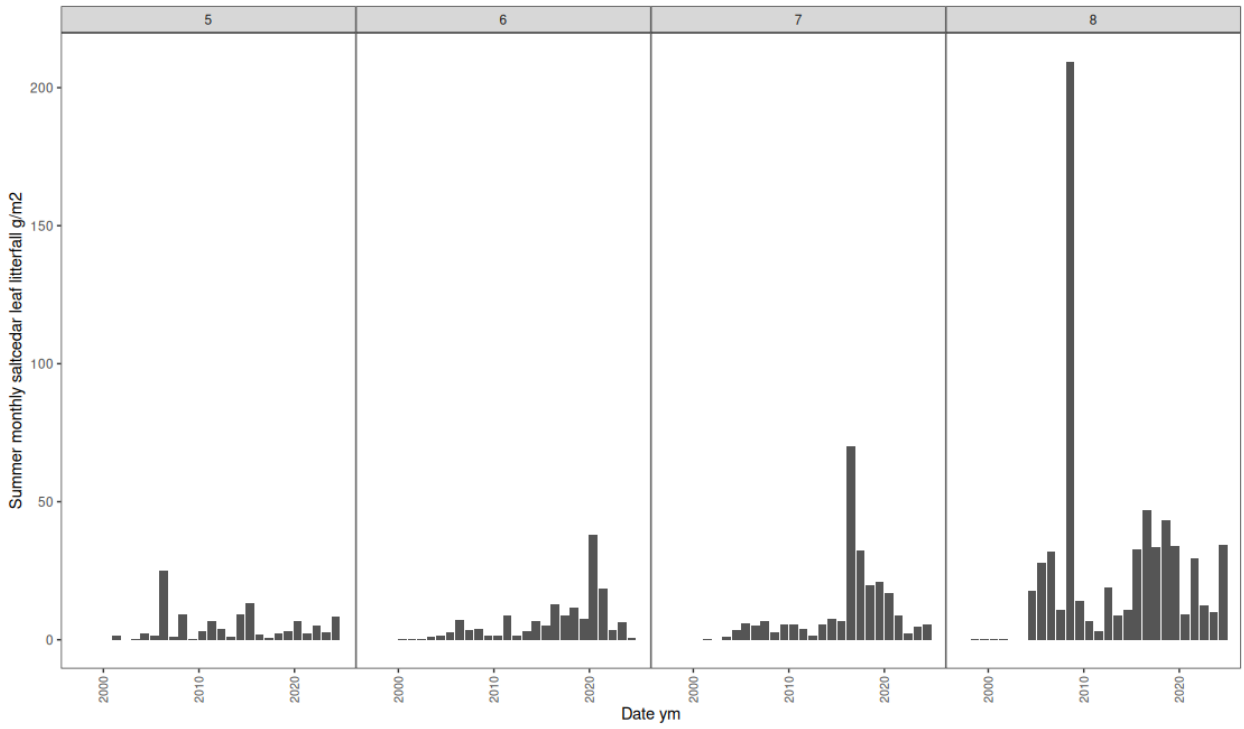

Figure 8. Summer saltcedar litterfall across years, grouped by month (May - August: 5-8), showing an increase in summer leaf fall following TLB presence.

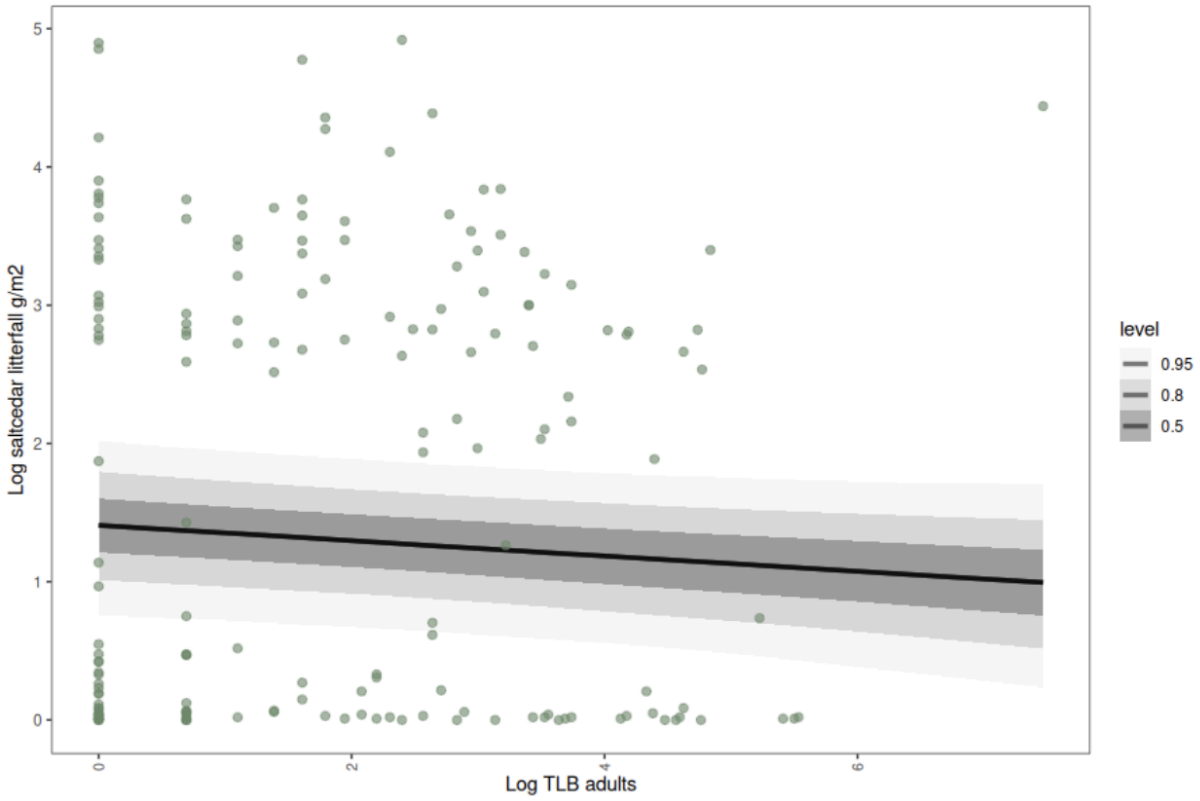

Figure 9. Log-transformed saltcedar leaf fall vs. log-transformed TLB adults showing a slight decline in productivity with increasing TLB adult abundance.

Preliminary analyses suggest that increases in TLB abundance are associated with measurable, but small, reductions in saltcedar productivity. Bayesian models show that a 1% increase in TLB adult abundance corresponds to an approximate 0.08% decrease in saltcedar productivity. Lag effects increase this to a 3% decrease in saltcedar productivity, as impacts of the beetle compound across years. When percent defoliation is used in the models instead of TLB adult abundance, a 1% increase in percent defoliation is associated with a 0.14% decrease in saltcedar litterfall in the same year. Percent defoliation is most likely a better predictor of the impact of TLB and a better indicator of the actual TLB abundance than the adult TLB counts. This is because TLB monitoring occurs once each month, and the sampling period does not always coincide with the peak TLB numbers, resulting in high defoliation alongside small TLB abundance counts. BEMP monitored one short-term site, Bosque School, in weekly intervals during 2021 in order to better capture TLB population dynamics. At this site, peak abundance in August occurred as two nonconsecutive spikes separated by a week of lower abundance, leading to higher percent defoliation but inconsistent beetle counts.

Branch die-off has been recorded during TLB monitoring (Figure 4), but may be partly due to repetitive long-term sampling during the summer months (some samplers are more aggressive

in their sweeping than others). Drought and heat stress may also contribute to branch mortality. Only the Santa Ana site has had apparently natural saltcedar mortality, which co-occurred with widespread mortality of other native and non-native trees, making this mortality event not directly attributable to TLB. Other sites have had saltcedar mortality through effective mechanical removal.

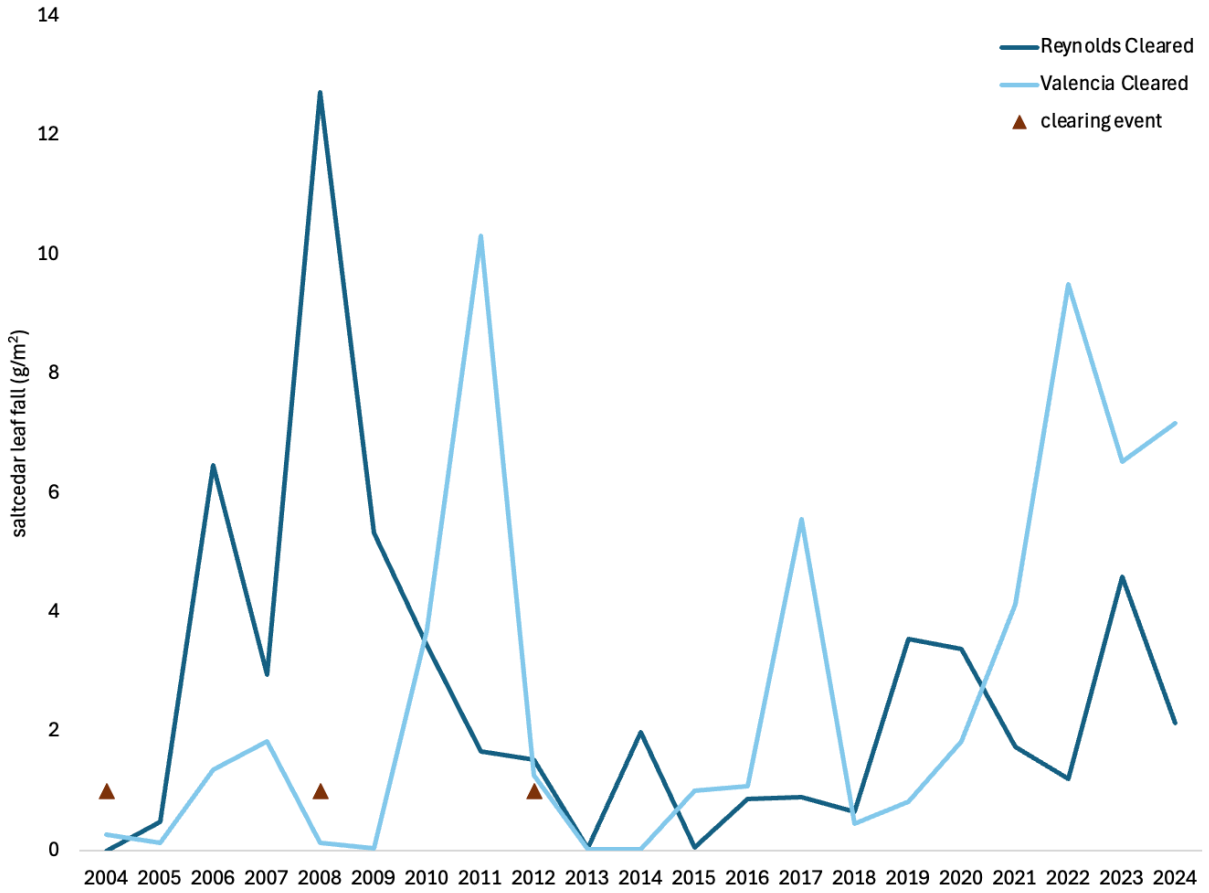

Figure 10. Saltcedar leaf fall at VC and RC. Both sites were cleared in 2003. VC had follow-up clearing in 2008 and RC had follow-up clearing in 2012. Clearing events are indicated by brown triangles. TLB were first recorded in Belen in 2013 with high TLB abundances not occurring until 2015.

There are two sites in Belen that were part of GRGWA projects, Valencia Cleared (VC) and Reynolds Cleared (RC). Both were cleared in 2003 (before TLB were present) and again in 2012 (likely after TLB had arrived). VC was spot treated again in 2008. Following the 2003 clearing, saltcedar productivity recovered quickly at both sites (Figure 10). Saltcedar productivity recovered quickly following the 2008 secondary clearing in VC as well. Clearing in 2012 is considered “post-TLB”, as the beetles were recorded in small numbers in 2013, and saltcedar productivity from 2013-2020 showed recovery that was slightly less than pre-TLB recovery levels. There are potential additive impacts of clearing and presence of TLB on post-clearing recovery of saltcedar. Additionally, this also likely shows the additive impacts of TLB and a

second clearing event within a site. Saltcedar recovery is still apparent, even with multiple clearing events and herbivory pressure from TLB.

Ji et al. (2017) found that TLB dispersal is influenced by saltcedar density and that TLB abundance is positively influenced by width of the riparian area and saltcedar abundance; however, BEMP data do not show strong correlation between TLB numbers and saltcedar cover. While sites with dense saltcedar stands do support high TLB abundance in some years, populations oscillate between years resulting in thick saltcedar stands supporting low TLB densities. In addition, sites with low saltcedar cover, like Valencia Cleared, Reynolds Forest post-clearing, and Rt 66, have supported very high populations of TLB at the tree level. This suggests that once the population is established, TLB can stay in an area even after saltcedar removal. Even in 2025, after 15 years of TLB presence,the highest TLB counts do not always correspond to the densest stands of saltcedar.

Most sites had a detectable response of saltcedar productivity and/or reproductive effort following initial spikes in TLB abundance (changes in productivity are not always detected when TLB initially arrive in low numbers) but the response is not always maintained through time. Diversion (in Albuquerque) and Sevilleta, near Socorro, are good examples that highlight this response.

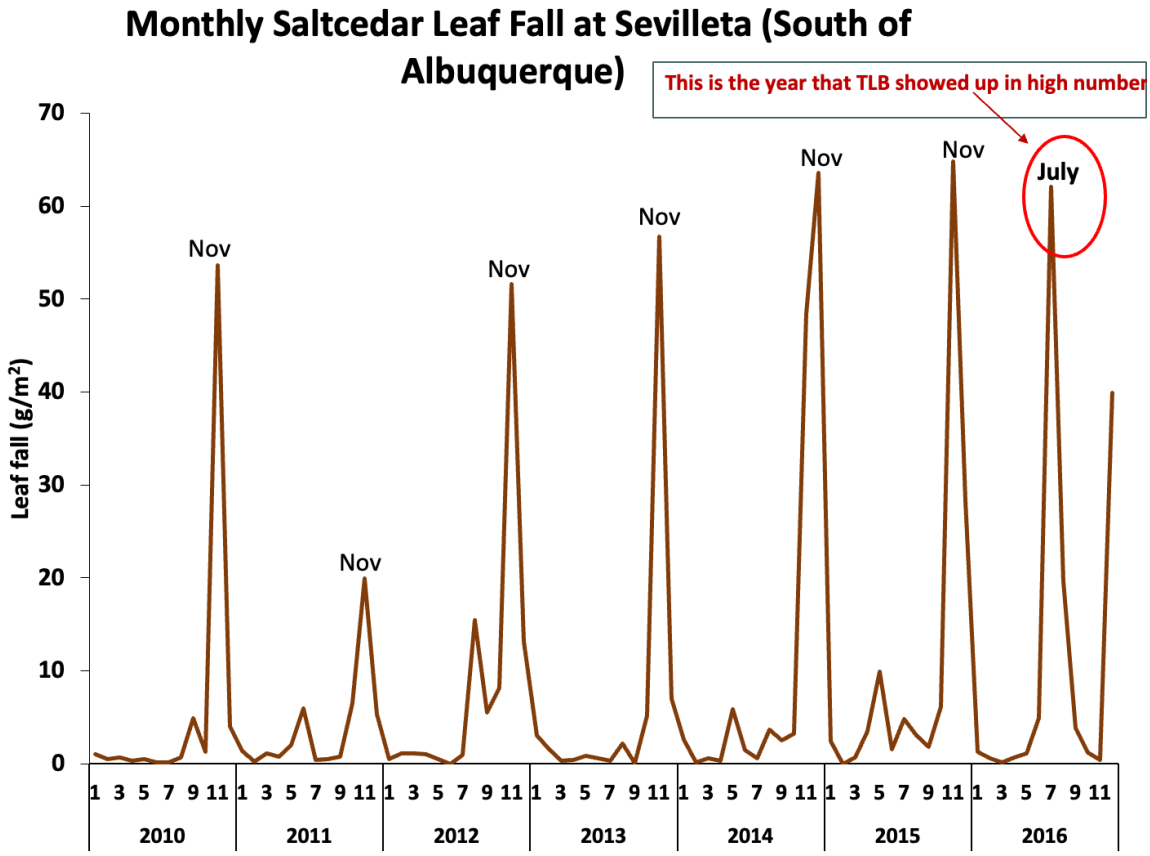

Figure 11. Monthly saltcedar leaf fall at Sevilleta.

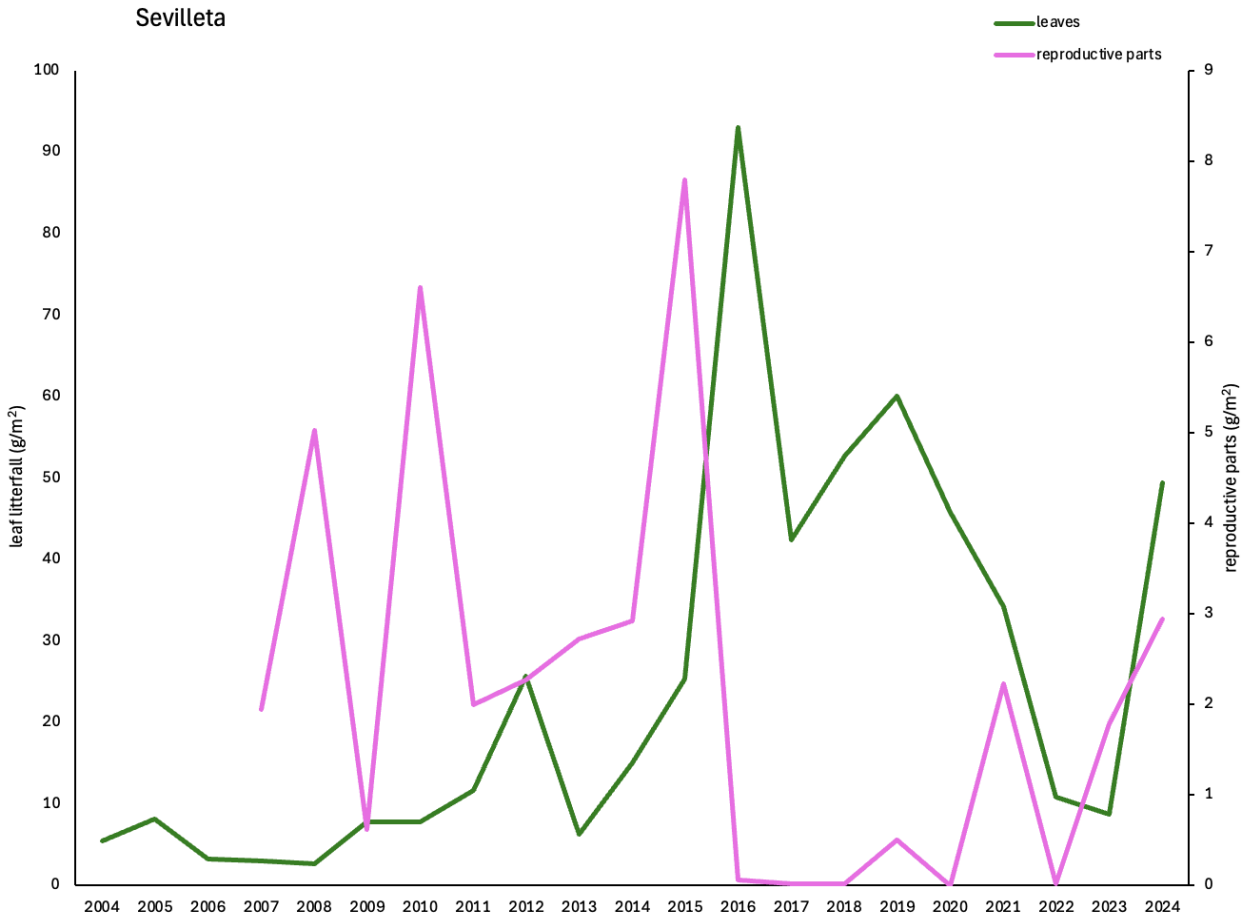

Figure 12. Sum annual leaf fall and reproductive part litterfall at Sevilleta. Reproductive parts are shown on the secondary y-axis as the values are an order of magnitude lower than the leaf fall values.

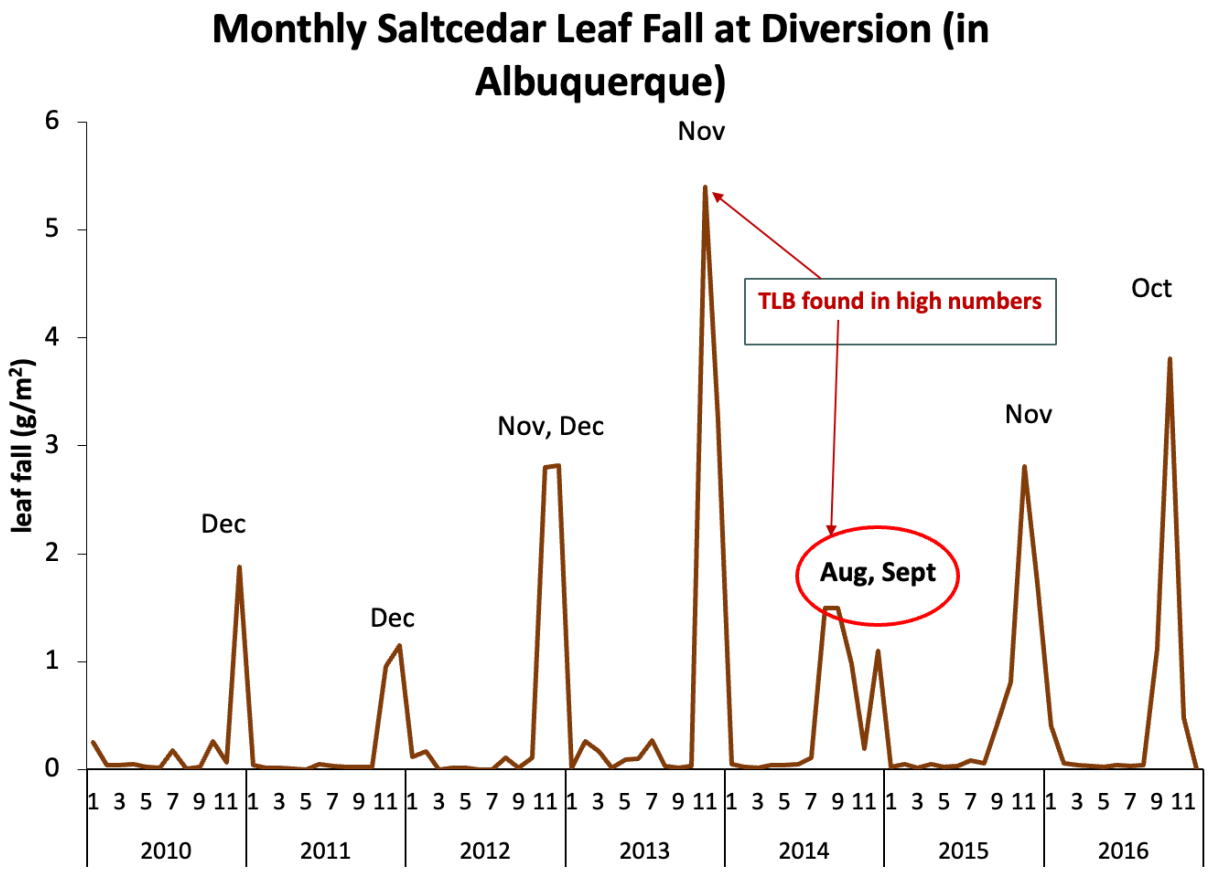

Figure 13. (From TLB presentations and previous report). Monthly saltcedar leaf fall at Diversion.

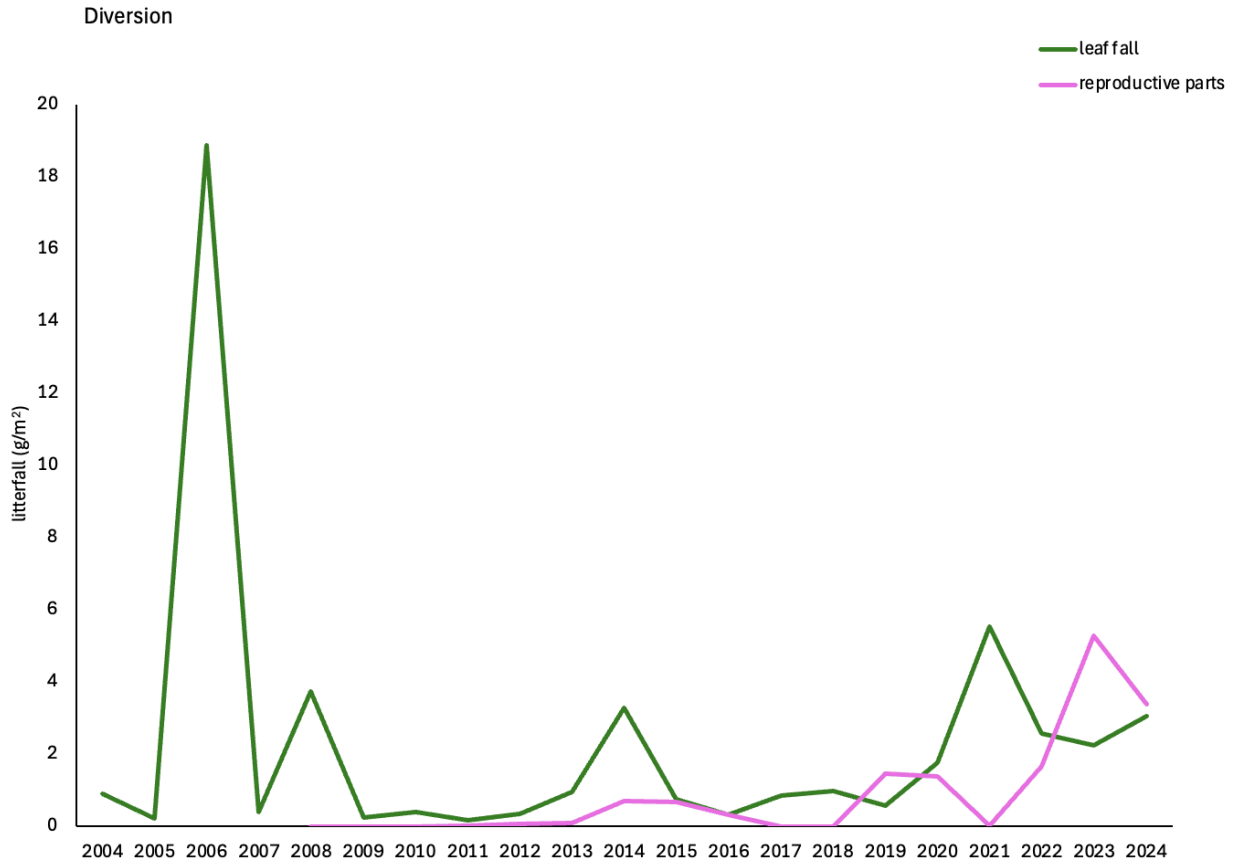

Figure 14. Sum annual leaf fall and reproductive part litterfall at Diversion.

Saltcedar leaf fall and reproductive litterfall at the Sevilleta BEMP site show slow to rapid increases in productivity and high variability in reproductive effort until 2016 when the TLB was first recorded in high abundance at the site (Figure 11). That year, there was an early leaf drop in July, followed by another large leaf drop in November, resulting in higher annual leaf production levels that year (Figures 11 and 12). Conversely, reproductive effort that year was extremely low. This suggests that the initial response to high defoliation was to allocate resources to refoliation at the expense of reproduction. Similarly, at the Diversion site there were two distinct leaf drops (August-September and again in November) in 2014, when large numbers of TLB were first recorded (Figure 13). This was also a year of higher than normal leaf production (Figure 14). In contrast, the reproductive effort has remained largely unaffected (Figure 14).

Changes in productivity, as measured through leaf fall, indicate that there may be a small additive effect of clearing and TLB presence. Clearing events in 2006 and 2012 in Belen predate the arrival of TLB, although TLB were recorded in low numbers in 2013.

We used both the monthly percent flowering on each tree during TLB monitoring as well as the biomass of reproductive litterfall each month to assess TLB impacts on saltcedar reproductive effort. Bayesian multilevel models showed a mean decrease in reproductive biomass of -0.03

g/m^2 with increasing TLB, but we can't confidently say TLB suppresses reproductive litterfall in the same year. There was also no clear lagged effect of TLB on reproduction and there is no clear directional trend in reproductive litterfall over time. Saltcedar reproduces relatively consistently across sites regardless of local conditions, with less site-to-site variation in reproductive litterfall than in leaf fall. However, variability in reproduction has greater implications to population dynamics than variability in leaf fall.. Reproductive effort in saltcedar is much more sensitive to annual conditions than TLB abundance, suggesting saltcedar reproduction is more likely driven by climate factors such as temperature, precipitation, or flood timing, rather than by TLB. Finally, sites with higher baseline reproductive litterfall show a stronger negative TLB slope. In plain language — TLB most strongly suppresses reproduction at sites where reproduction is high to begin with.

Management Strategies

Current monitoring results suggest that TLB alone does not substantially reduce saltcedar cover across studied sites, largely due to strong year-to-year fluctuations in beetle abundance and the ability of saltcedar to refoliate following defoliation events. There is a high degree of variability between sites. However, reductions in saltcedar productivity associated with TLB activity indicate the beetle presence may weaken saltcedar stands over time.

Reductions in productivity at sites with TLB activity and a history of saltcedar removal are stronger than reductions in productivity at sites influenced by TLB alone, suggesting that biological control may complement traditional management approaches. TLB presence may function as a long-term stressor on saltcedar populations that enhances the effectiveness of clearing efforts; however, clearing efforts are still needed to manage saltcedar presence.

For land managers, these results indicate that TLB presence should not be viewed as a replacement for active saltcedar management but rather as a process to help suppress saltcedar recovery following treatment. Continued monitoring is essential for evaluating whether these combined effects lead to a long-term change in vegetation structure and improved habitat for native plants and animals.

Conclusion

Overall, the largest impact on saltcedar productivity is seen following clearing and fire events. There is a small but clear impact on saltcedar productivity from TLB defoliation, especially over time, but this has not been as dramatic in New Mexico as what has been reported elsewhere. Refoliation in the first year or years following beetle or leafhopper defoliation can be as high or higher than initial leaf production , but in subsequent years, refoliation and total leaf production are both diminished. Leaf biomass is more heavily impacted than flowering and reproductive litterfall. TLB density at the tree level is not necessarily linked to saltcedar stand density, even though dense sections of saltcedar provide more habitat. Long-term trends indicate an additive effect of stressors on saltcedar productivity, which includes combinations of clearing, fire,

drought, and herbivory. No saltcedar mortality has been directly attributed to TLB defoliation at BEMP sites.

More data are needed to determine if saltcedar defoliation contributes to long-term competitive release on perennials like coyote willow, or other broad ecosystem impacts. New Bayesian models will incorporate the impacts of TLB and native vegetation community response. These models will also address the interaction of climate variables and TLB impact.

TLB abundance is likely not well captured with monthly monitoring, as peak abundance can occur and then drop within a week. Percent defoliation, a subjective measurement, is better at capturing the impact of TLB abundance. Moving forward, we recommend weekly sampling at key locations and use of non-active sampling methods, like sticky traps, in order to better capture beetle numbers.

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