Using disturbance to restore Bartram's hairstreak habitat on Big Pine Key
Final Report Submitted to Florida Keys Wildlife Refuges, May 2018
Grant Agreement #F14AP00922

Prepared by Erica Henry, Nick Haddad, and Martha Reiskind
North Carolina State University, Department of Applied Ecology
Primary contact: erica_henry@ncsu.edu

EXECUTIVE SUMMARY

Alteration of historic disturbance regimes is one of the biggest threats to species' persistence. This is particularly true for species adapted to frequent, low-intensity fires, such as Bartram's scrub-hairstreak and its sole hostplant, pineland croton. When restoration via prescribed fire is not an option, surrogate methods of clearing vegetation are needed. We use an experimental study of mechanical clearing to test how mechanical clearing affects survival and recruitment rates and population growth rate of a rare, endemic plant *Croton linearis*. We implemented mechanical understory removal treatments to mimic prescribed fire in two pine rockland units on Big Pine Key. In these treatments and in no-treatment controls we measured croton survival and recruitment as well as presence of caterpillar herbivory on croton plants.

We found that survival rates one-year post-disturbance were similar across, mechanical clearing and control plots. Two-years post treatment, however, there was a trend toward increased survival in mechanically cleared plots, while survival rates continued to decline in controls. Mechanical treatments stimulated a flush of recruitment in the first-year post disturbance but not in the second year, and we saw very little recruitment in control plots in either year. Even with this flush of recruitment, vital rates combined to produce shrinking populations in both mechanically cleared and control plots. However, new recruits survived at slightly higher rates than older, established plants following mechanical clearing. Our results demonstrate that with no action the croton population on Big Pine Key is declining by 20-30% annually. Without croton, there will be no Bartram's scrub-hairstreaks on Big Pine Key. Therefore, increasing the number of host plants across the island should be the number one management priority. This will require disturbance. Because mechanical clearing increases recruitment, and those recruits are more likely to survive than other plants, it is a better alternative than no action.

TABLE OF CONTENTS

INTRODUCTION **3**
OBJECTIVES **4**
PROJECT SITE AND STUDY SPECIES **5**
METHODS **6**
Mechanical clearing experiment 6
Butterfly monitoring 7
Statistical analyses 7
RESULTS **8**
DISCUSSION/MANAGEMENT RECOMMENDATIONS **10**
TAKE HOME POINTS **14**
ACCOMPLISHMENTS VS. GOALS AND OBJECTIVES **14**
REFERENCES **16**
FIGURES **18**

INTRODUCTION

Fire adapted ecosystems are among the world's most endangered (Noss, LaRoe, & Scott, 1995). Loss of fire-dependent ecosystems is highest near human development, as people restrict or eliminate fire near their homes, cities, and farms. Loss of natural fire regimes remains a threat even within protected areas. In many cases, this is a legacy of historical management, in which fire was viewed as a neutral or even negative process in maintaining habitat. Without fire, however, succession fundamentally alters the competitive landscape for plants and animals, changing habitat structure and species composition, ultimately leading to extirpation of species adapted to historic fire regimes (Bond & Keeley, 2005; Leach & Givnish, 1996; Ratajczak, Nippert, Hartman, & Ocheltree, 2011).The preferred management strategy in these systems is prescribed fire. However, lighting fires can be difficult due to high costs, narrow prescription windows, and the threat of escape (Miller & Wade, 2003). This results in the need for habitat management that replicates the effects of fire such as mechanical understory removal. Mechanical methods mimic the effects of fire on vegetation structure but have different effects on below ground processes and litter dynamics (Moghaddas & Stephens, 2007; Stephens et al., 2012; Youngblood, Wright, Ottmar, & McIver, 2008); differences that could result in altered population dynamics for fire-adapted species. We test how prescribed fire and a surrogate, mechanical understory removal, affect survival, recruitment, and population growth rates of a rare plant.

The success of fire management and its surrogates, in terms of rare species restoration, will be determined by the effects a management strategy has on population size and growth rates of target species. Efforts to understand how fires and their surrogate management strategies affect populations should, therefore, focus on estimating responses of vital rates. The differential effects of restoration strategies have been examined in many fire prone systems, however, these comparisons focus on community responses or changes in species cover rather than individual vital rates (Perchemlides, Muir, & Hosten, 2008; Steen, Conner, et al., 2013a; Steen, Smith, et al., 2013b). Studies that do focus on vital rates generally only examine the effects of fire and do not compare fire to other management strategies (Liu, Menges, & Quintana-Ascencio, 2005; Menges, 2007), but see Gross et al. 1998). The advantage of focusing

on vital rates is that it allows for estimation of population growth rates, and therefore, of the future trajectory of the population. This information is especially important for land managers tasked with the job of ensuring the survival of threatened and endangered plants and animals. Understanding the likely trajectory of a population following a specific management action feeds into the decision-making process. With this information, land managers can consider the full range of costs, benefits, and risks when deciding between different management strategies (van Wilgen, 2013).

We test the effects of fire and mechanical restoration treatments on vital rates of pineland croton (*Croton linearis* [Euphorbiaceae], hereafter referred to as croton), a rare plant endemic to fire-adapted pine rocklands in South Florida. Although not listed as endangered itself, croton is the only host for two endangered butterflies, Bartram's scrub-hairstreak (*Strymon acis bartrami*) and Florida leafwing (*Anaea troglodyta*). After habitat loss, loss of disturbance regime is the primary reason for declines in these species. Most of the remaining pine rockland fragments are surrounded by people in Miami-Dade county and on Big Pine Key, making fire difficult to implement and mechanical treatments an attractive alternative. Working in an experimental study of mechanical clearing and an observational study of fire effects, we ask: do fire and mechanical clearing result in similar survival and recruitment responses and population growth rates? This research is essential for understanding population level responses of this rare plant to habitat restoration and has implications for restoration of other species, specifically the butterflies that rely on croton.

OBJECTIVES

The overall objective of this project was to restore habitat for Bartram's scrub-hairstreaks on Big Pine Key. The specific objectives of our monitoring work were focused on understanding 1) the demographic response of croton populations to different types of disturbance-based management, and 2) how quickly restored areas are colonized by Bartram's scrub-hairstreaks. This report describes our efforts to accomplish these objectives. Because fires were never lit, we could not compare the effects of mechanical clearing to prescribed fire on the island, but we did compare mechanical treatments to no-treatment controls. When

initially proposed, we outlined additional objectives focused on how other factors, such as potential predator (ant) densities and key deer populations, interact with restoration efforts to affect the success of these strategies. These objectives were not accomplished, and reasons why are detailed at the end of this report.

PROJECT SITE AND STUDY SPECIES

Pine rocklands are found only in south Florida, The Bahamas, and Turks and Caicos where they grow directly out of the limestone substrate. These forests are characterized by open stands of South Florida slash pines (*Pinus ellioti* var. densa) and an open shrub/palm subcanopy, below which a rich herbaceous community thrives. In south Florida, pine rocklands once extended along the Miami rock ridge, from North Miami Beach to Everglades National Park, and a few islands in the Florida Keys. Ninety percent of the pine rockland on mainland south Florida and much of the habitat in the Keys has been cleared for development in the last 100 years. The largest remaining pine rocklands are protected within Everglades National Park, Navy Wells – a Miami-Dade County Natural Area, and the National Key Deer Refuge on Big Pine Key. Given this large decline of pine rockland habitat, croton populations have declined and become increasingly fragmented. Pine rockland habitat loss affects not only croton, but the butterflies that rely on it, Bartram's scrub-hairstreak and Florida leafwing. Declines in these butterfly populations are closely linked with the fragmentation and loss of croton populations in the landscape.

Remaining pine rocklands are vulnerable to lack of disturbance and subsequent forest succession, which excludes croton, this is true even within protected areas. Historically, pine rocklands burned frequently, maintaining the forest in an early successional state. Croton is well adapted to fire; based on anecdotal data, it appears to both seed and re-sprout following fires (C. Anderson unpublished data). Without fire, pine rocklands convert to hardwood hammocks, and croton eventually disappears from the ecosystem. Since 1951, croton has gone from being one of the 10 most common plants in pine rocklands on Big Pine Key (Alexander 1972) to covering less than 0.01% of pine rocklands in 2014 (Bradley 2009); this decline has occurred simultaneous with decline in fire across the island.

Fire is the preferred management strategy for maintaining croton and habitat for Bartram's scrub-hairstreaks and Florida leafwings; however, there exist multiple obstacles to implementing prescribed fire, especially at the wildland-urban interface and in instances where fuels have accumulated to levels that make it unsafe to burn. In these areas, mechanical clearing through mowing, brush hogging, etc. has been proposed as a possible habitat maintenance strategy. This is motivated by anecdotal observations of croton plants sprouting and thriving in newly cleared fire breaks. While these observations are encouraging, evidence-based data are necessary for making informed management decisions.

METHODS

Mechanical clearing experiment

To test the effects of mechanical clearing on croton demography, we implemented a clearing experiment on Big Pine Key. In July 2015, we established four, half-hectare replicates each of two treatments, mechanical clearing and a no treatment control. Two of these blocks were located in the central pine rocklands of Big Pine Key, and two were on the south end of the island, we will refer to these as the South and Central Units. US Fish and Wildlife Service fire staff implemented clearing treatments by haphazardly driving a skid steer with a masticating head through the treatment unit. The prescription for the treatment was to clear and mulch the understory vegetation in a mosaic pattern covering 50% of the plot with the goal of mimicking the mosaic burn patterns that might result from a prescribed fire.

Prior to treatments, we tagged all croton plants in 15 subplots in each replicate. To ensure that we tagged croton plants across the replicate, we identified croton sampling plots by establishing a grid of parallel 50-meter transects spaced 10 meters apart in all treatment units (total of 10 transects). In each unit, we randomly selected five transects and established three, permanent 2.5m-radius subplots on each one, in the middle and each end of each transect. We marked subplots by hammering a 25cm galvanized spike at the subplot center and attached a numbered, metal plant tag to identify the subplot number. In each subplot, we tagged all croton plants by wiring metal plant tags around the base of each plant. If we did not reach our goal of 30 plants tagged in the first five transects, we randomly sampled additional transects

until we reached at least 30 plants. We tagged plants in 5-10 subplots in each unit, depending on croton density.

We first tagged plants in July 2015 and implemented mechanical treatments in August of the same year. We re-sampled the croton plots in July 2016 and July 2017. Each time we re-sampled we located all previously tagged plants and recorded if they were alive or dead. We also tagged new seedlings as well as plants that we missed in the previous round of sampling. New seedlings were easily distinguished from missed plants by their orange-colored stems that were not noticeably woody. From these data, we estimated annual survival and recruitment in the two years following mechanical treatments. To quantify recruitment, we divided the number of seedlings tagged after the disturbance by the number of tagged plants in the year prior to disturbance. We also used these data to compare treatment and block effects on reproductive status of plants.

Butterfly monitoring

To monitor Bartram's hairstreak larval presence, we surveyed all tagged plants every six months (Feb and July of each year) recording whether or not we observed larval feeding damage. We also concurrently conducted monthly adult surveys in Mechanical and Control treatment units on Big Pine Key (Henry & Anderson, 2016).

Statistical analyses

All statistical analyses were done in R version 3.3.1 (R Core Team, 2016).

Survival: To test for differences in overall survival, we used a generalized linear model with a binomial response variable and fixed effects of treatment (mechanical, control), year (one year and two years post treatment), and unit using the glm function and Anova function from the car package (Fox & Weisberg 2011).

Recruitment: We tested for differences in recruitment between the two treatments with a linear model, using our standardized recruitment estimates as the response variable, and post-hoc tests with the TukeyHSD function.

Recruit vs. established plant survival: We compared survival of new recruits to survival of previously established plants in the second year post-treatment in our mechanical experiment. To do this we used a two-way ANOVA with main effects of age (recruit vs. established) and treatment (mechanical vs. control).

Population growth rate: We calculated count-based population growth rates for the two years following treatments as N(t+1)/N(t). We then used a linear model and post-hoc Tukey test to test for differences between population growth rates across treatments and years.

Reproductive plants: We compared the number of reproductive plants (those for which sex could be determined based on presence of fruits and/or flowers) across unit, treatment, and year. To do this, we used a generalized linear model with a binomial response variable and block, treatment, and year as fixed effects. We used the glm function and Anova function from the car package (Fox & Weisberg 2011).

Butterfly monitoring: We did not conduct statistical analyses of these data due to our low sample sizes. Summary results are described below.

RESULTS

We tagged a total of 296 croton plants, 161 in control plots and 135 in mechanically cleared plots.

Overall survival by treatment depended on year (interaction: Wald's chi-square=24.74, df=2, $p<0.0001$). In the first year post-treatment, survival was similar across all treatment types. In the second year post-treatment, survival in the disturbance treatments increased, while survival in control plots showed the opposite trend (Figure 1a). Although not significant, there is a trend of higher survival in mechanically cleared plots than controls in the south unit (Figure 3a).

Recruitment was highest in mechanically cleared plots one year post-treatment, but did not differ between treatments in year two (interaction: $F_{2,8}=0.16$, $p=0.026$, Figure 1b). Survival of new recruits was not different from survival of already established plants (Wald's chi-square=0.059, df=1, $p=0.81$). However, there was a trend in mechanically cleared plots of survival being higher for new recruits (0.85±0.084) than previously established plants

(0.77±0.059), and a trend suggesting that recruit survival was higher in mechanically cleared plots (0.85±0.084) than controls (0.64±0.17). The interaction between plant age and treatment was not significant (Wald's chi-square=1.12, df=1, p=0.29; Figure 2).

Differences in survival and recruitment across treatments combined to produce differences in population growth rates. There was a significant year effect ($F_{2,14}$=4.33, p=0.033), with population growth rate in the first year post-treatment significantly higher than the second (Figure 1c). Growth rates were slightly higher in mechanically cleared plots than controls, although the treatment effect was marginally significant ($F_{2,14}$=0.120, p=0.11). Mean growth rates were above 1 in mechanical treatments one year post treatment, but fell back below 1 in in these plots two years post-clearing.

There were notable differences in the number of reproductive plants between the two units on Big Pine Key, with almost twice as many plants having fruits and/or flowers in the south unit than in the center of the island. There were significant Unit*treatment and Unit*year interactions (Unit*TX: Wald's chi-square=8.239, df=1, p=0.004, Unit*YR: Wald's chi-square=6.587, df=1, p=0.037, Figure 3b). The unit*treatment interaction reveals that there was a difference in treatment effect in the two units, there was no treatment effect on the percentage of reproductive plants in the center unit, but mechanical clearing increased the number of reproductive plants in the south unit. The unit*year interaction highlights that the percentage of reproductive plants declined in each year in the central unit, whereas in the south unit the percentage of reproductive plants declined in the year following mechanical clearing but rebounded to pre-treatment levels in the second year post-treatment.

We conducted butterfly surveys monthly throughout the duration of this project following Henry and Anderson (2016) protocol. An adult Bartram's scrub-hairstreak has not been detected in a survey since Feb 2016. We observed additional adult butterflies periodically in 2016 and early 2017, just not on a survey transect. However the last adult Bartram's scrub-hairstreak to be detected on Big Pine Key was in May 2017. In our pre-treatment survey, we observed Bartram's herbivory on 15-25% of the plants (Figure 4). Six months post-treatment, this proportion had not changed indicating that both treatment and control plots were occupied by Bartram's scrub-hairstreaks within six months of treatment. In our one year post-

treatment survey, the proportion of plants with herbivory declined in all locations, in fact, throughout the duration of this project, the proportion of plants with noticeable herbivory declined. Although we did not ever detect Bartram's hairstreak caterpillars on tagged plants, we did find three Bartram's caterpillars on plants in mechanically cleared plots in May 2016. At least one of these was found on a plant that had germinated after clearing (Figure 5).

DISCUSSION/MANAGEMENT RECOMMENDATIONS

Without restoration, croton populations on Big Pine Key are shrinking at a rate of 20-30% per year. We demonstrated that mechanical surrogates for fire are a useful restoration strategy. Mechanical understory removal resulted in an initial decline in survival in the first year following treatments, but then an increase in survival rates in the second year (Figure 1a). Control plots, on the other hand continued to steadily decline. Mechanical treatments resulted in a flush of recruitment, that did not occur in controls (Figure 1b, Figure 7). These two rates resulted in population growth rates that increased above 1 in the first year following treatments, effectively reversing the population decline in control plots, but in year two, populations in cleared plots were again declining like controls (growth rates <1). However, it is important to note that new recruits had slightly higher survival than established plants in both cleared and control plots (Figure 2). If this trend continues, these new recruits could have a positive impact on long term population persistence (Pascarella & Horvitz, 1998). Since there was no recruitment and low overall survival in control plots in both years, mechanical clearing is better than no restoration at all.

Like many shrubs in fire-adapted landscapes, croton relies heavily on re-sprouting post disturbance to maintain extant populations. Seventy percent of croton plants survived and/or re-sprouted following mechanical clearing (Figure 6 & 7). Although we documented post-treatment recruitment, it only occurred in plots where croton was present prior to treatment, suggesting that croton seeds do not disperse far from natal patches (as far as we know, croton dispersal is limited to dehiscence, although animals may disperse seeds further). These results highlight the importance of promoting the survival of existing croton patches. Once patches are extirpated, they are unlikely to be re-colonized rapidly.

The question remains, how do the effects of mechanical clearing treatments compare to prescribed fire? To answer this question, Henry (2018) analyzed data from Big Pine Key experiment along with results from prescribed fire monitoring at Everglades National Park. She found that both survival and population growth rates two years post treatment were much higher following prescribed fires than after mechanical clearing. In fact, every plant that survived the prescribed fire was still alive in the second year. The same cannot be said for plants that survived mechanical treatments. The increased survival rates two years post-fire resulted in stable, not declining, populations. These results demonstrate that although mechanical clearing is a good surrogate for fire, it does not totally mimic the beneficial effects of fire (Henry 2018).

Disturbance history plays a role in how well a population may respond to either prescribed fire or mechanical understory removal. The center unit on Big Pine Key last burned 15 years prior to this project in 2000, and the south unit last burned in 2010. This variability in time since previous disturbance could underlie the differences in survival we found across units. Although not significant, the differences between the south and center units on Big Pine suggest that plants that were more recently disturbed are more likely to respond positively to mechanical disturbance than plants that are no longer reproductive and already senescing (eg. Figure 6). This difference in disturbance history also likely drives the differences we observed in the reproductive status of plants between the two units on Big Pine Key. The proportion of plants that were reproductive in the south block was almost twice the proportion in the center block. The fact that only ~30% of the croton plants are reproductive in the center block signifies that 15 years without disturbance may be too long to maintain viable croton populations. Our data reveal a shrinking population of senescing plants in the center unit, regardless of disturbance treatment. In the south block, on the other hand, ~60% of the plants are reproductive. In this block, the number of reproductive plants actually increased in the second year post-treatment. These patterns of reproduction and survival suggest that plants that are disturbed more frequently may respond more positively to disturbance than those that have not been disturbed in over a decade.

Too frequent and too long of a disturbance return interval leads to declining populations of disturbance-adapted species (Altwegg, De Klerk, & Midgley, 2014). To understand the optimal disturbance return interval, we would ideally estimate all vital rates in populations at different years post-disturbance and then simulate the effects of varying disturbance return intervals on long-term population growth rates (Gross, Lockwood, Frost, & Morris, 1998; Liu et al., 2005; Menges & Quintana-Ascencio, 2004; Menges, Quintana-Ascencio, Weekley, & Gaoue, 2006). For croton, this will require more information than we currently have. Based on the population declines we measured in our control plots on Big Pine Key that have not burned in 15 years, we can infer that a 15 year disturbance return interval is too long to maintain growing croton populations. In addition to increasing competition, long fire return intervals increase the threat of catastrophic wildfires. Unlike the frequent, low intensity fires that native species are adapted to, these catastrophic fires are generally much hotter and may result in population declines or even extinction. Hot fires can kill seeds in the seedbank and consume below ground resource stores thus limiting the probability that fire adapted plants survive (Moreno & Oechel, 1993; Odion & Davis, 2000; Pausas & Keeley, 2014). The catastrophic fires that result from fire suppression and long fire return intervals also have the potential to threaten homes and other valuable infrastructure. Understanding optimal fire return intervals is therefore important for both maintaining rare species populations and reducing threats to human development.

We measured a plant's response to mechanical clearing, but the question remains: what happens when disturbances interact? Mechanical clearing has been proposed as a pre-fire treatment in this system to reduce vertical fuels and therefore the risk of prescribed fire escape. What we do not yet understand is how this would affect the responses that we measured. Given that we saw very little recruitment in the second year following clearing, we might hypothesize that the two disturbances in sequence act antagonistically, with the first disturbance, mechanical clearing, stimulating germination and thus depleting the seed bank such that there are few seeds left to germinate following the second disturbance, prescribed fire. Additionally, mechanical clearing is likely to alter fire behavior since it concentrates fuels on the ground, thus increasing ground fire temperature and duration. This could have the unintended consequence of killing seeds present in the seedbank (Busse, Hubbert, Fiddler,

Shestak, & Powers, 2005) or burning up the underground resource stores necessary for re-sprouting post-fire. In this case, the two responses again act antagonistically by reducing recruitment and/or survival. Conversely, if the prescribed fires occurred before seeds germinate, they could act synergistically with the mechanical clearing, further enhancing germination rates by increasing the amount of bare ground and providing additional chemical cues that may be necessary to stimulate germination. The fires that follow mechanical treatments might also promote future survival of new recruits and re-sprouting plants by more effectively reducing competition.

Understanding host-plant responses to restoration actions is a first step in understanding how habitat restoration affects butterfly populations. This is especially true for Bartram's scrub-hairstreak on Big Pine Key, whose population is limited by the number, distribution, and quality of croton plants. Currently, Bartram's scrub-hairstreak populations on the island are too small to be able to show that restoration actions directly increase numbers in a statistically rigorous fashion. However, our larval herbivory data demonstrated that female butterflies laid eggs on plants in mechanically cleared plots within six months of treatment. Anecdotal observations of Bartram's scrub-hairstreak caterpillars feeding on croton plants that sprouted post-mechanical clearing are encouraging (Figure 5). It appears that, at least in some cases, it doesn't take long for new plants to provide resources necessary to support a developing caterpillar. Ideally, we would be able to assess how egg and larval and adult survival rates differ across treatments. However, at this point, there just are not enough butterflies on the island to be able to find enough eggs and caterpillars and adults to measure survival rates.

Bartram's scrub-hairstreaks have steadily declined on Big Pine Key since Henry and Anderson's surveys in 2012 and 2013 (2016). In 2013 they detected 11 butterflies on a single island-wide survey. The last butterfly to be detected in the same survey was in February 2016. This decline in the number of butterflies follows a decline in croton on the island. Our data document a croton population that is declining at a rate of 20-30% annually. This has been ongoing for at least the past 5 years resulting in significant croton loss on Big Pine Key. If declines continue at this rate, croton could potentially disappear from the island in the next five

years. Therefore, increasing the number of host plants across the island should be the number one management priority. This will require disturbance, ideally fire.

We have shown that mechanical disturbance is a better alternative than no action. Our experimental test of the demographic effects of mechanical clearing on a rare plant provides a framework for understanding the mechanisms underlying population level responses to this restoration strategy. These mechanisms are important to include into predictions of species responses to global change (Buckley et al., 2010; Gilman, Urban, Tewksbury, Gilchrist, & Holt, 2010). In the face of changing climate, it may become even more difficult to meet the environmental conditions necessary for prescribed burning (Terando et al. in prep), therefore other habitat maintenance and restoration strategies may become increasingly important in fire adapted systems.

TAKE HOME POINTS

- Without disturbance, croton population on Big Pine Key is declining at a rate of 20-30% per year.
- Mechanical understory removal does not increase croton survival relative to no treatment but does promote germination of seeds from the seedbank.
- Mechanical understory removal may have more beneficial effects on croton survival than those we measured if used in areas with healthier (more recently disturbed) croton plants.
- Female Bartram's scrub-hairstreaks will lay eggs in mechanically cleared areas within six months of treatment.
- Understanding optimal disturbance return interval is next step in refining croton management strategies, but disturbance return interval >15 years are likely to result in declining croton populations.

ACCOMPLISHMENTS VS. GOALS AND OBJECTIVES

Our original proposal outlined the following five objectives:

1. Determine the demographic response of croton populations to disturbance treatments

2. Document how soon after treatment restored areas are colonized by Bartram's scrub-hairstreaks.
3. Determine how the predatory ant community is altered by disturbance treatment – both composition and relative abundance
4. Measure whether or not butterfly larval survival differs between disturbance treatments
5. Determine if Key deer browsing affects croton establishment

We were successful in accomplishing objectives 1 and 2 (as reported above), but did not accomplish objectives 3, 4, and 5. The primary reason objectives 3 and 4 were not met, is that fire treatments were never implemented on Big Pine Key. Our initial plan was to initiate ant community (objective 3) and larval survival experiments (objective 4) once both mechanical and fire treatments were implemented, since fires were never lit on Big Pine Key, we never conducted these experiments. Instead, we collaborated with fire effects staff at Everglades National Park who had collected multiple years' worth of croton survival and recruitment data that could be included in our analyses. We did not undertake objective 5 due to sample size constraints. Once we completed the initial round of croton tagging, we realized that there were fewer croton plants (and therefore croton plots) in some blocks than we originally anticipated. Because of this, we were concerned that if we further subset these plots into fenced and unfenced treatments, we would lose any ability to detect treatment effects on the island. In reality, pine rockland restoration on Big Pine Key will occur in the presence of the Key Deer, fencing large areas of pine rockland is logistically unfeasible. Therefore, we abandoned this objective of the study in favor of having the statistical power necessary to detect effects of restoration treatments in the presence of the deer.

REFERENCES

Alexander TR, Dickson JH III. 1970. Vegetational changes in the National Key Deer Refuge-II. Quarterly Journal of the Florida Academy of Sciences 32(2):81-89.

Altwegg, R., De Klerk, H. M., & Midgley, G. F. (2014). Fire-mediated disruptive selection can explain the reseeder–resprouter dichotomy in Mediterranean-type vegetation. *Oecologia*, *177*(2), 367–377. http://doi.org/10.1007/s00442-014-3112-6

Bond, W., & Keeley, J. (2005). Fire as a global "herbivore": the ecology and evolution of flammable ecosystems. *Trends in Ecology and Evolution*, *20*(7), 387–394. http://doi.org/10.1016/j.tree.2005.04.025

Buckley, L. B., Urban, M. C., Angilletta, M. J., Crozier, L. G., Rissler, L. J., & Sears, M. W. (2010). Can mechanism inform species' distribution models? *Ecology Letters*, *13*(8), 1041–1054. http://doi.org/10.1111/j.1461-0248.2010.01479.x

Busse, M. D., Hubbert, K. R., Fiddler, G. O., Shestak, C., & Powers, R. F. (2005). Lethal soil temperatures during burning of masticated forest residues. *International Journal of Wildland Fire*, *14*, 267–276. http://doi.org/10.1071/WF04062

Gilman, S. E., Urban, M. C., Tewksbury, J., Gilchrist, G. W., & Holt, R. D. (2010). A framework for community interactions under climate change. *Trends in Ecology and Evolution*, *25*(6), 325–331. http://doi.org/10.1016/j.tree.2010.03.002

Gross, K., Lockwood, J. R., III, Frost, C. C., & Morris, W. F. (1998). Modeling controlled burning and trampling reduction for conservation of Hudsonia montana. *Conservation Biology*.

Henry, E. H., & Anderson, C. T. (2016). Abundance estimates to inform butterfly management: double-observer versus distance sampling. *Journal of Insect Conservation*, *20*, 505–514.

Leach, M. K., & Givnish, T. J. (1996). Ecological Determinants of Species Loss in Remnant Prairies. *Science*, *273*(5281), 1555–1558. http://doi.org/10.1126/science.273.5281.1555

Liu, H., Menges, E. S., & Quintana-Ascencio, P. F. (2005). Population viability analyses of *Chamaecrista keyensis*: effects of fire season and frequency. *Ecological Applications*, *15*(1), 210–221. http://doi.org/10.1890/03-5382

Menges, E. S. (2007). Integrating demography and fire management: an example from Florida scrub. *Australian Journal of Botany*, *55*(3), 261–272. http://doi.org/10.1071/BT06020

Menges, E. S., & Quintana-Ascencio, P. F. (2004). Population viability with fire in *Eryngium cuneifolium*: deciphering a decade of demographic data. *Ecological Monographs*. http://doi.org/10.2307/4539046

Menges, E. S., Quintana-Ascencio, P. F., Weekley, C. W., & Gaoue, O. G. (2006). Population viability analysis and fire return intervals for an endemic Florida scrub mint. *Biological Conservation*, *127*(1), 115–127. http://doi.org/10.1016/j.biocon.2005.08.002

Miller, S. R., & Wade, D. (2003). Re-introducing fire at the urban/wild-land interface: planning for success. *Forestry: an International Journal of Forest Research*, *76*(2), 253–260. http://doi.org/10.1093/forestry/76.2.253

Moghaddas, E. E. Y., & Stephens, S. L. (2007). Thinning, burning, and thin-burn fuel treatment effects on soil properties in a Sierra Nevada mixed-conifer forest. *Forest Ecology and Management*, *250*(3), 156–166. http://doi.org/10.1016/j.foreco.2007.05.011

Moreno, J. M., & Oechel, W. C. (1993). Demography of *Adenostoma fasciculatum* after fires of different intensities in southern California chaparral. *Oecologia*, *96*(1), 95–101. http://doi.org/10.1007/BF00318035

Noss, R. F., LaRoe, E. T., III, & Scott, J. M. (1995). *Endangered Ecosystems of the United States: A preliminary assessment of loss and degradation.* (No. 28). Washington, DC.

Odion, D. C., & Davis, F. W. (2000). Fire, soil heating, and the formation of vegetation patterns in chaparral. *Ecological Monographs*, *70*(1), 149–169. http://doi.org/10.1890/0012-9615(2000)070[0149:FSHATF]2.0.CO;2

Pascarella, J. B., & Horvitz, C. C. (1998). Hurricane distrubance and the population dynamics of a tropical understory shrub: megamatix elasticity analysis. *Ecology*, *79*(2), 547–563. http://doi.org/10.1890/0012-9658(1998)079[0547:HDATPD]2.0.CO;2

Pausas, J. G., & Keeley, J. E. (2014). Evolutionary ecology of resprouting and seeding in fire-prone ecosystems. *New Phytologist*, *204*(1), 55–65. http://doi.org/10.1111/nph.12921

Perchemlides, K. A., Muir, P. S., & Hosten, P. E. (2008). Responses of Chaparral and Oak Woodland Plant Communities to Fuel-Reduction Thinning in Southwestern Oregon. *Rangeland Ecology and Management*, *61*(1), 98–109. http://doi.org/10.2111/07-026R1.1

Ratajczak, Z., Nippert, J. B., Hartman, J. C., & Ocheltree, T. W. (2011). Positive feedbacks amplify rates of woody encroachment in mesic tallgrass prairie. *Ecosphere*, *2*(11), 1–14. http://doi.org/10.1890/ES11-00212.1

Steen, D. A., Conner, L. M., Smith, L. L., Provencher, L., Hiers, J. K., Pokswinski, S., et al. (2013a). Bird assemblage response to restoration of fire-suppressed longleaf pine sandhills. *Ecological Applications*, *23*(1), 134–147. http://doi.org/10.1890/12-0197.1

Steen, D. A., Smith, L. L., Conner, L. M., Litt, A. R., Provencher, L., Hiers, J. K., et al. (2013b). Reptile assemblage response to restoration of fire-suppressed longleaf pine sandhills. *Ecological Applications*, *23*(1), 148–158. http://doi.org/10.1890/12-0198.1

Stephens, S. L., McIver, J. D., Boerner, R. E. J., Fettig, C. J., Fontaine, J. B., Hartsough, B. R., et al. (2012). The Effects of Forest Fuel-Reduction Treatments in the United States. *BioScience*, *62*(6), 549–560. http://doi.org/10.1525/bio.2012.62.6.6

van Wilgen, B. W. (2013). Fire management in species-rich Cape fynbos shrublands. *Frontiers in Ecology and the Environment*, *11*(s1), e35–e44. http://doi.org/10.1890/120137

Youngblood, A., Wright, C. S., Ottmar, R. D., & McIver, J. D. (2008). Changes in fuelbed characteristics and resulting fire potentials after fuel reduction treatments in dry forests of the Blue Mountains, northeastern Oregon. *Forest Ecology and Management*, *255*(8-9), 3151–3169. http://doi.org/10.1016/j.foreco.2007.09.032

FIGURES

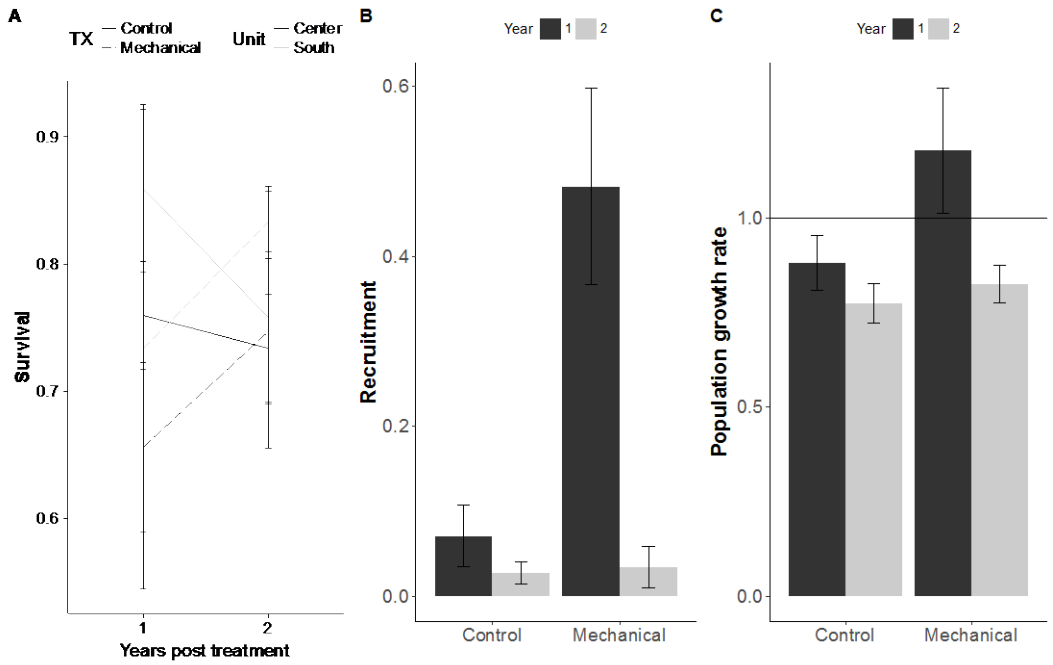

Figure 1. A) Percent survival following mechanical clearing (dashed line), and no treatment control (solid line) in the south and center units on Big Pine Key. B) Standardized recruitment and C) population growth rates in year one (black bars) and year two (grey bars) post-treatment. Error bars are standard error in all panels. Population growth rates >1 indicate growing population.

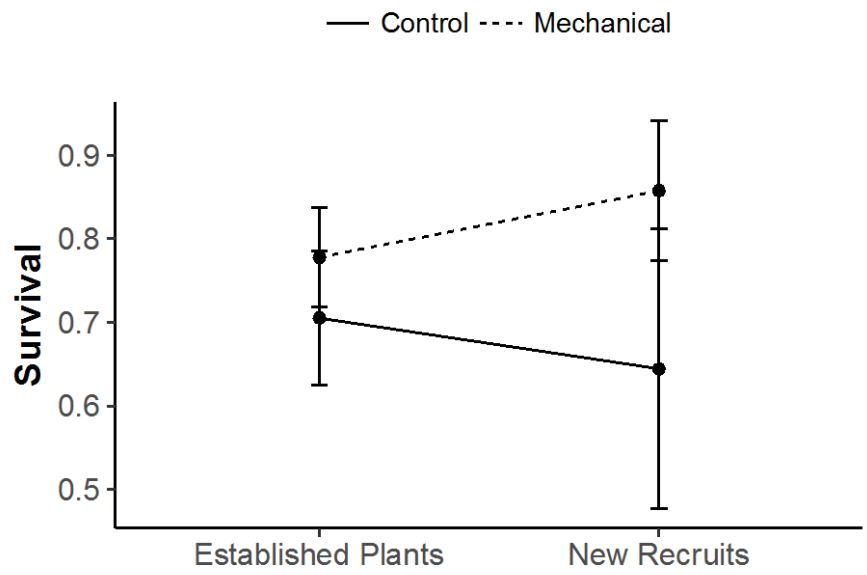

Figure 2. Survival of previously established plants and new recruits in mechanically cleared plots (dotted line) and controls (solid line).

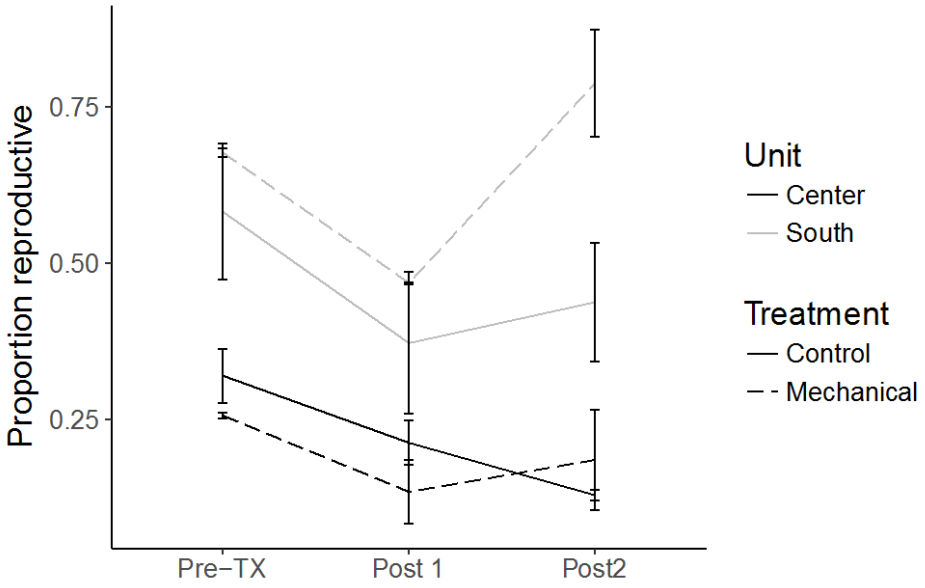

Figure 3. Proportion reproductive plants in each treatment and unit. Reproductive plants are those with fruits and/or flowers present at the time of sampling. Lines are means, error bars represent standard error.

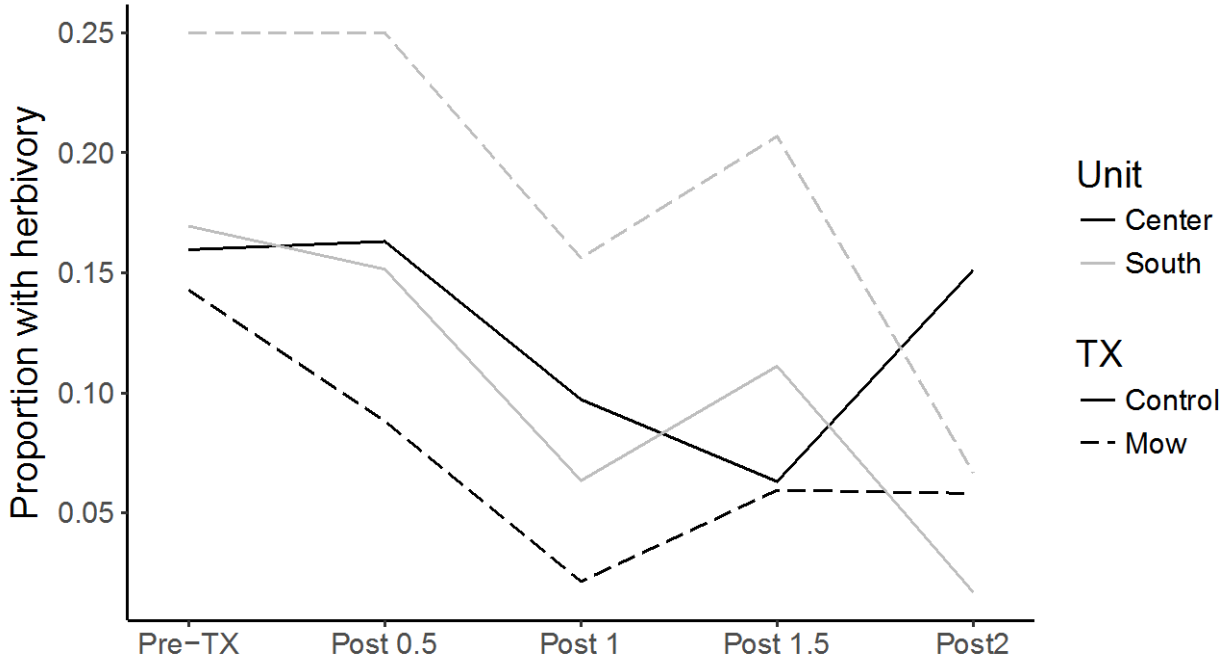

Figure 4. Proportion of tagged plants with observed Bartram's scrub-hairstreak herbivory.

Figure 5. Caterpillar (at yellow arrow) and its feeding damage (in yellow box) on May 23, 2016. This plant was new recruit following mechanical treatments that occurred in July 2015.

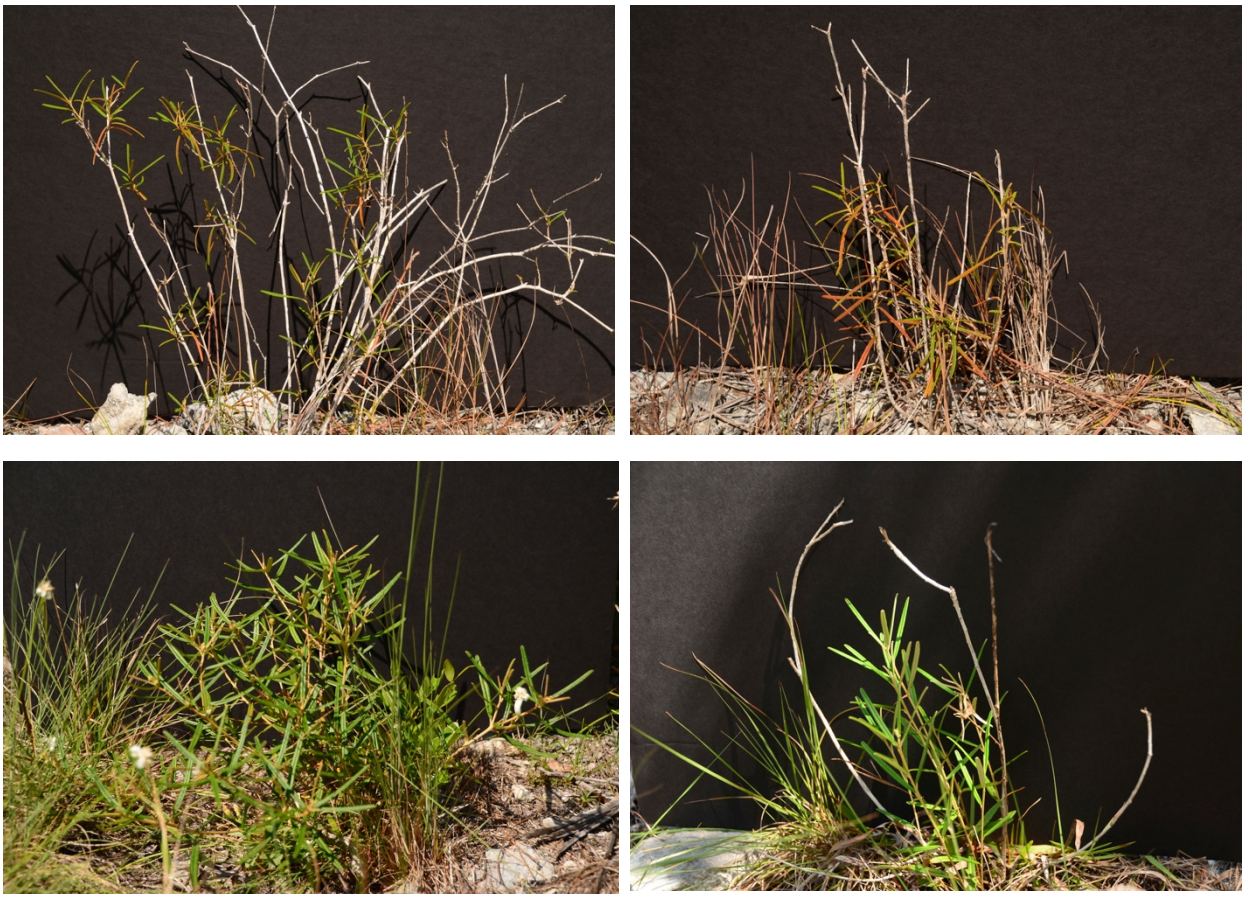

Figure 6. Photos of plants representative of plants in the center unit that are senescing (top row), and plants that re-sprouted post mechanical clearing. Note differences in color and apparent health.

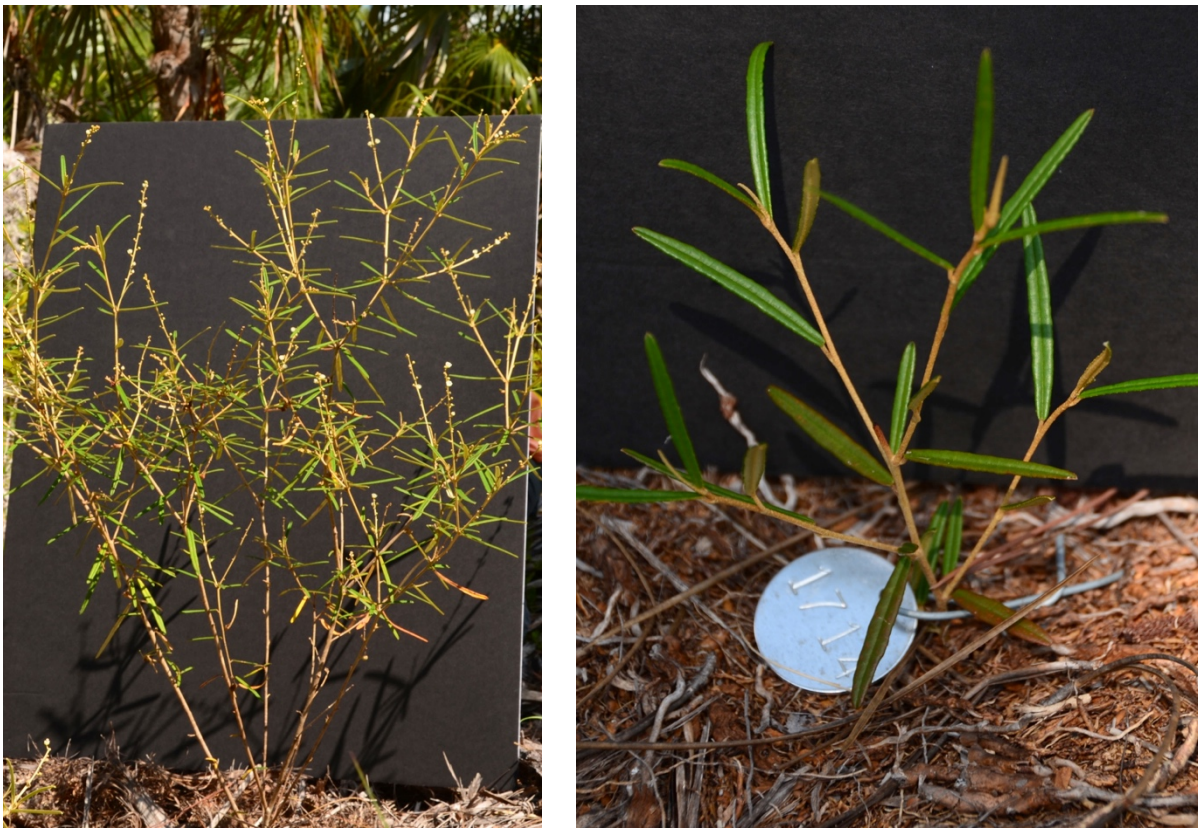

Figure 7. Champion post-mechanical re-sprouting plant (left) and example of new recruit post-mechanical treatments (right).