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Post-delisting dilemma: Long-term success of *Chrysopsis floridana* introductions depends on continued fire management

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Abstract

The Endangered Species Act aims to recover imperiled species to the point of delisting; however, few plant species have been delisted due to recovery. Critical to many species' recovery criteria are introductions, yet the long-term success of plant introductions is highly variable and their use in delisting decisions is relatively untested. In this study, we examined the dynamics of eight introduced populations of the recently delisted *Chrysopsis floridana* (Florida Golden Aster), a fire-dependent perennial plant endemic to Florida scrub habitats. We analyzed 6 years of demographic data (2017–2022) and 10 years of prescribed fire records to assess population trends, fire management effects, and vital rates related to recruitment, survival, and reproduction. Our results show that population densities with regular fire management increased, while unburned populations generally declined. Prescribed fire increased seedling recruitment and survival rates. We estimated a fire-return interval of approximately 5 years is needed for population persistence, shorter than the 10-year interval used in the delisting decision. Our findings suggest that post-delisting monitoring should prioritize fire management to ensure the long-term success of introduced populations. Our work highlights the need for more robust, specific delisting criteria, particularly regarding the role of introductions, long-term monitoring, and post-introduction habitat management. These insights can inform future recovery plans for fire-dependent species and improve the delisting process for plants under the Endangered Species Act.

KEYWORDS

fire management, Florida scrub, plant introductions, rare plant conservation

1 | INTRODUCTION

The goal of listing species under the United States' Endangered Species Act (ESA) is to aid recovery and prevent extinction, actions that will, ideally, lead to delisting the species (ESA, 1973). Such decisions are made by federal agencies, specifically the US Fish and Wildlife Service (USFWS) and the National Marine Fisheries Service, based on the species meeting certain recovery criteria detailed in the recovery plan. Often, these criteria are

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based both on population performance indicators (e.g., resiliency, redundancy, representation) and the mitigation of threats to the species (e.g., habitat loss, disease) (Evans et al., 2016; Neel & Che-Castaldo, 2013). Of the listed plant species, few have ever been delisted due to recovery (Haines et al., 2021), suggesting that, despite the ESA being over 50 years old, the process of delisting plant species is relatively untested.

Introductions (also referred to as "translocations" or "conservation introductions") are a critical component of many species' recovery plans (e.g., Maschinski et al., 2023; Schemske et al., 1994), yet the success of these introductions is quite variable (Albrecht and Maschinski, 2012; Bellis et al., 2023) and their applicability to satisfying recovery criteria is not clear. The establishment of new populations is one metric by which "success" is measured before a species can subsequently be considered for downlisting or delisting. However, the long-term success of these introduced populations cannot be assessed based on the first few years alone (e.g., Colas et al., 2008; Menges, 2008); the success of an introduction depends on high-quality habitat (Bellis et al., 2023) maintained via appropriate land management. Long-term monitoring, following an introduction as well as subsequent land management, is necessary to inform decision-making regarding the delisting or downlisting of listed species, yet such data are rarely collected (e.g., Godefroid et al., 2011).

The post-delisting monitoring period offers an additional opportunity to confirm that the recovery criteria have been met and, specifically, confirm the success of introductions, the adequacy of management, and the abatement of threats. ESA Section 4(g)(1) requires that USFWS, in cooperation with state agencies, conduct post-delisting monitoring for at least 5 years, after which time the delisting decision may be revisited (ESA, 1973). Furthermore, as new data and analyses become available, federal agencies and their cooperators may use this period to refine management best practices, evaluate population trends and dynamics, and implement additional policies. For example, post-delisting monitoring of animal species has identified new diseases, various genetic concerns, and essential changes in connectivity that affect the species' persistence (Amidon et al., 2021; Janecka et al., 2021). Implementation of post-delisting policies is generally lagging for plants, which have roughly a quarter of the federal or international policies assuring protection post-delisting than vertebrate species (Haines et al., 2021).

In this study, we investigated the dynamics of introduced populations of the recently delisted *Chrysopsis floridana* (Florida Golden Aster, Asteraceae), a short-lived perennial herb endemic to the fire-dependent, xeric scrublands of southwestern Florida, USA. At the time of listing in 1986, the species was known from only nine wild populations across five counties. By the time it was delisted in 2024, the total number of populations had increased to 30 as new wild populations were discovered and efforts to introduce nine populations on protected lands were apparently successful (USFWS, 2024a). Bok Tower Gardens' Rare Plant Conservation Program directed the majority of these introductions, including introducing *C. floridana* seedlings to seven sites across Pinellas, Hardee, and Manatee counties from 2008 to 2013 that were noted in the delisting decision (USFWS, 2024a) and an eighth site in 2019 (Peterson, 2014; Ward et al., 2022) (Figure 1). Of the introduced populations, steady increases from their initial planting through 2015 were observed (Peterson, 2016). USFWS determined the species met the delisting criteria of securing at least 20 geographically distinct populations (USFWS, 1988) based on these 30 known populations, of which 25 were located on protected land and 18 (including six introduced populations) scored as having high or very high resilience (USFWS, 2024a). The resiliency estimates and general population estimates were based on the most recent single-year survey of population estimates available (the oldest survey occurred in 2006) and did not consider population trends.

However, demographic monitoring of the introduced populations suggested that the initial successes were, in some cases, short-lived (Ward et al., 2022). During the 3-year period from 2017 to 2019, introduced *C. floridana* populations varied widely in their population vital rates and relied upon a specific, open microhabitat associated with frequent fire management (i.e., low leaf litter and low shrub cover) (Ward et al., 2022), as fire can promote flowering in *C. floridana* in post-fire years (Lambert & Menges, 1996). In these scrub habitats, periodic fires are essential for maintaining open sand gaps, and populations of *C. floridana*, as well as many other listed herbaceous species, typically rely on continued fire management (e.g., Lambert & Menges, 1996; Menges et al., 2017; Menges & Hawkes, 1998; Quintana-Ascencio et al., 2018; Yates & Broadhurst, 2002). Notably, the 3-year dataset used by Ward et al. (2022) was not sufficient to analyze the effects of fire on population growth or vital rates.

Here, we assessed the performance of eight introduced *C. floridana* populations, populations that were critical toward meeting the delisting criteria, and the adequacy of fire management in maintaining these populations. Specifically, we extend the dataset presented in Ward et al. (2022) and use the resulting six-year dataset to (1) evaluate population trends, (2) estimate a sufficient

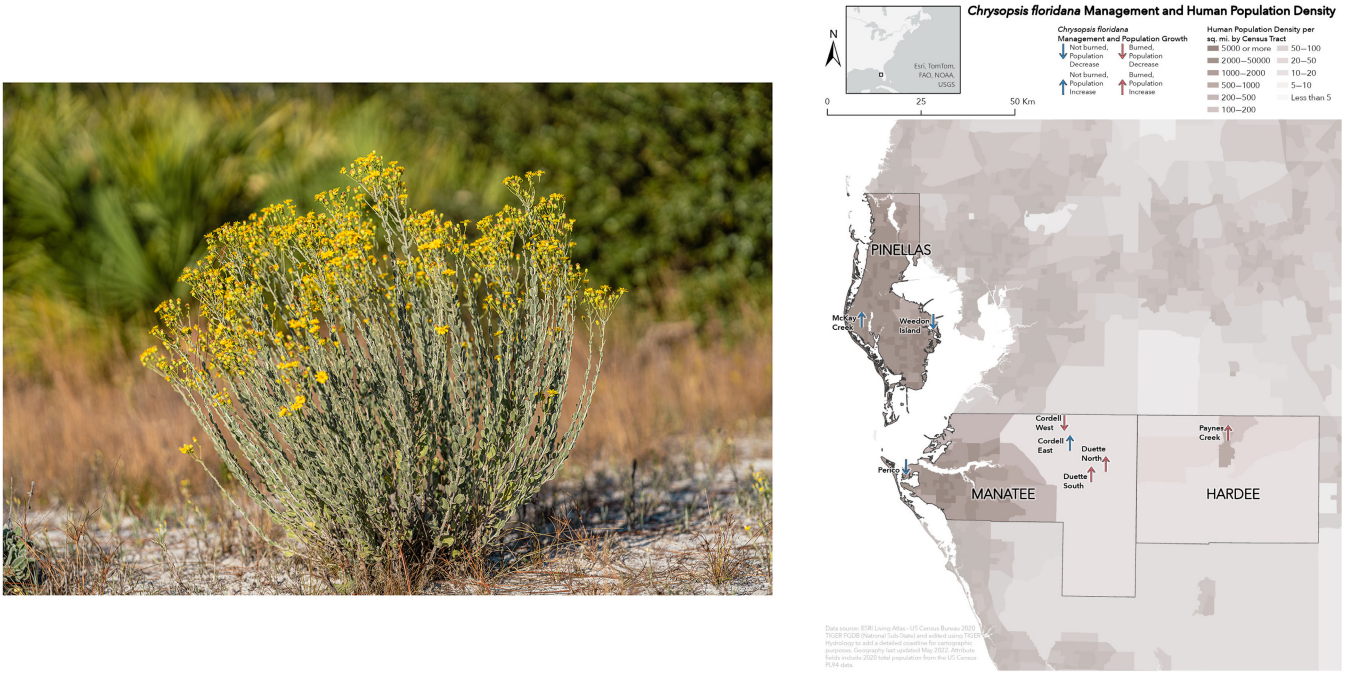

FIGURE 1 (Left) *Chrysopsis floridana* flowering at the Paynes Creek introduction site (Photo Credit: George McKenzie, Jr.). (Right) Population growth of introduced *C. floridana*, fire management, and human population density in three counties of west-central Florida were monitored from 2017 to 2022. The direction of the arrows indicates whether the population of introduced *C. floridana* increased from 2017 to 2022, and the color of the arrows indicates the fire management of the site, with red arrows as burned sites and blue arrows as unburned sites. Human population density per census tract is indicated by the shading of census tracts, with darker shading representing higher population density.

fire return interval, and (3) quantify the effects of fire management on vital rates (recruitment, survival, and reproduction) of introduced *C. floridana* populations. We intend for this work to guide further decision-making during the post-delisting monitoring period, as well as further the broader discussion on how best to utilize introductions in the context of meeting recovery criteria.

2 | METHODS

2.1 | Monitoring

Seven introduced populations of *C. floridana* were established across west-central Florida between 2008 and 2013, with the addition of one introduction site in 2019 (Ward et al., 2022). Introduction sites were selected based on the historical range, the soil series type (Archbold fine sands, St. Lucie fine sands, Lakewood fine sands, Duette fine sands, and Pomello fine sands), habitat suitability, and site management/fire management frequency (Peterson, 2014). Individual plants were germinated at Bok Tower Gardens from seeds collected from multiple wild populations and transplanted as 4–6-month-old seedlings. Initial introduction size (i.e., number of transplants) ranged from 147 to 922 per introduction site (Table 1). At each introduction site, a minimum of 10 randomly chosen plots containing live plants were established. Count-based stage class monitoring (Menges & Gordon, 1996) was conducted at three sites, and demographic monitoring of individuals (Menges & Gordon, 1996) was conducted at five sites (Table 1). Monitoring was conducted in November–December during peak flowering from 2017 to 2022.

At each site with count-based stage class monitoring, we established 10 circular, 4-m diameter plots. Plot centers were randomly chosen within the introduction area. In each plot, we recorded the number of juvenile (non-flowering) and adult (flowering) *C. floridana* plants present. One site (McKay Creek) was not sampled in 2022 due to damage from Hurricane Ian and was only monitored through 2021 (Figure 1).

At each site with demographic monitoring, we established 10–12 2×1 m plots. Individual plants were monitored using numbered aluminum tags and pin flags to track survival and vegetative growth variables. Each year, the survival status of previously marked plants was recorded, and all new plants were given a unique tag. Rosette diameters were measured for all plants, except in cases where adult plants had lost their

TABLE 1 Land management and geographic data for each introduction site For sites that were unburned during the scope of this study (2017–2022), fire return intervals were not calculated.

Fire management classification	Population name	County	Human population per sq mile	Monitoring type	Intro year	Initial intro size	2022 population density (No. of plants/m^2)	Burn year(s)	Fire return interval (years)
Burned	Cordell West	Manatee	12.9	Count-based stage class	2009	545	0.034	2019	10
Burned	Duette North	Manatee	12.9	Demography	2013	147	19.9	2018, 2021	5, 3
Burned	Duette South	Manatee	12.9	Demography	2013	174	17.5	2019, 2021	6, 2
Burned	Paynes Creek	Hardee	193.2	Demography	2011	199	11.21	2011, 2019	8
Unburned	Cordell East	Manatee	12.9	Demography	2008–2009	910	5.75	2011	—
Unburned	McKay Creek	Pinellas	3300.0	Count-based stage class	2009	773	0.99	2013	—
Unburned	Perico	Manatee	1306.5	Count-based stage class	2019	356	0.16	None	—
Unburned	Weedon Island	Pinellas	1032.5	Demography	2012	922	1.04	2013	—

rosette entirely because the rosette of this species can shrink or disappear when the flowering stem emerges. Therefore, we measured the height of flowering stems as an additional growth measurement for reproductive individuals. The total number of flowering heads per plant was recorded for a randomized subset of 20% of the adult plants present. Plants were classified as either seedlings (first-year, non-reproductive plants with basal rosettes), juveniles (second-year plants or older with no reproductive stems), or adults (reproductive plants with at least one flowering stem present). First-year plants that were >5 cm in rosette diameter were assumed to be juveniles that were missed during their seedling stage.

2.2 | Data analysis

All data were analyzed using R v. 4.2.2 (R Core Team, 2023).

Sites that received fire management at any point between 2017 and 2022 were classified as burned, and sites that did not receive fire management during this period were classified as unburned. We estimated the annual population growth rate as r, the intrinsic rate of increase ($ln[N_{t+1}/N_t]$) at the site level for all eight introduction sites (demographic and count-based stage class) for years 2017–2022 and on the plot level for the five introduction sites where demography was monitored. From these r values, we estimated the mean annual growth rate for each site. We then constructed a linear mixed-effect model (LMM) to test the effect of fire management on r at the plot level for demography sites, fitted with crossed random effects of year and plot nested within the population. Based on these trends, we estimated the mean fire return interval by averaging the fire intervals of the sites that were burned and increased in population size (Duette North, Duette South, and Paynes Creek; Table 1). Additionally, we estimated the overall percent change in population density by site as final density-initial density/initial density. We also investigated any effect of the initial introduction size on the logarithm of final population density with an interaction for management using a linear model and an analysis of variance.

For the five introduction sites where demography was monitored, we estimated the following vital rates at the individual level: size (rosette diameter and flowering stem height), reproduction (number of flowers), annual juvenile survival, annual adult survival, and size-dependent transition to flowering. At the plot level, we also considered seedling density (number of new seedlings/m^2/year). For all eight introduction sites, we estimated juvenile and adult density (number of juvenile plants/m^2/year and number of adult plants/m^2/year) at the plot level.

To examine how fire management can impact vital rates over time, we evaluated the role of time-since-fire on the population's plant densities by life stage class (seedlings, juveniles, and adults) at the plot level by constructing a linear mixed-effects model using the lme4 package (Bates et al., 2015). For each life stage class, an LMM was fitted with the crossed random effects of year and either plot nested within a population (individual-level demographic analyses) or a population nested within monitoring type (plot-level juvenile and reproductive adult density analyses). Response variables (i.e., seedling, juvenile, and adult density) were log-transformed. All three LMMs had a fixed effect for time-since-fire (Table 2).

To analyze the effect of fire management on reproduction and growth, we constructed additional LMMs using the lme4 package (Bates et al., 2015). The number of flowering heads (adults only), flowering stem height (adults only), and rosette diameter (juveniles only) from 1 year to the next were all separately analyzed using LMMs with the random nested effects for population, plot, and individual plant, a crossed random effect for year, and the fixed effect of fire management, the previous year's growth metric (i.e., flowering heads/individual plant, flowering stem height, or diameter of the previous year, respectively), and the fire × growth metric (Table 2). The flowering stem height was log-transformed. We also investigated the effect of fire management on the final population density using an analysis of variance.

To analyze the effects of fire management on juvenile and adult survival probability as well as the effects of fire management on the size-dependent transition from juvenile to adult, we constructed binomial generalized LMMs using the lme4 package (Bates et al., 2015). For the analyses of juvenile and adult survival probabilities, random nested effects of population, plot, and individual plant were included, with a crossed random effect for years and a fixed effect of fire management. For analyses of the size-dependent transition from juvenile to reproductive adult stage class, we included the fixed effects of fire management, rosette diameter in the previous year, and their interaction, and the crossed random effects of year and plant nested within plot nested within population (Table 2). The log-odds estimate for juvenile and adult survival were back-transformed to probabilities using the emmeans package (Lenth, 2023), and the exponential of the log-odds estimate of the size-dependent transition model was calculated to estimate the odds of increase.

TABLE 2 Estimated coefficients and analysis of deviance results for linear mixed-effects models for the following response variables: *r* (log of lambda) versus fire management; seedling, juvenile, and adult *Chrysopsis floridana* plant densities versus time-since-fire; year-to-year flowering heads, flowering stem height, and rosette diameter; and generalized linear mixed-effects model with juvenile and adult survival and juvenile transition to adult as response variables versus management (burned or unburned). All response variables were log-transformed except year-year flowering heads and rosette diameter.

	Estimate	SE	χ^2	*p*-value
r				
Intercept	−0.22	0.06		
Fire management (burned)	0.34	0.08	18.51	<.001*
Seedling density				
Intercept	1.07	0.15		
Time-since-fire (years)	−0.12	0.02	20.3	<.001*
Juvenile density				
Intercept	−0.87	0.28		
Time-since-fire (years)	−0.04	0.03	1.3	.236
Adult density				
Intercept	−3.12	0.59		
Time-since-fire (years)	0.07	0.04	2.7	.098
Number of flowering heads per reproductive plant				
Intercept	15.72	18.41		
Fire management (burned)	25.77	23.13	1.5	.225
No. flowering heads in the previous year	0.22	0.14	24.9	<.001*
Burned*No. flowering heads in the previous year	0.04	0.15	0.1	.799
Height				
Intercept	3.00	0.23		
Fire management (burned)	−0.16	0.23	<0.01	.963
Height previous year	0.01	< 0.01	92.45	<.001*
Burned * height in the previous year	<0.01	< 0.01	1.38	.239
Rosette diameter				
Intercept	3.31	0.82		
Fire management (burned)	0.16	0.93	0.2	.696
Diameter previous year	0.72	0.05	1436.10	<.001*
Burned*diameter in the previous year	0.02	0.05	0.18	.675
Juvenile survival				
Intercept	0.49	0.48		
Fire management (burned)	1.56	0.53	8.7	.003*
Adult survival				
Intercept	0.88	0.35		
Fire management (burned)	1.33	0.44	9.4	.002*
Juvenile transition to adult				
Intercept	−2.88	1.24		
Fire management (burned)	0.67	0.78	2.3	.129
Rosette diameter	0.37	0.07	128.73	<.001*
Management*Rosette diameter	0.005	0.08	0.004	.945

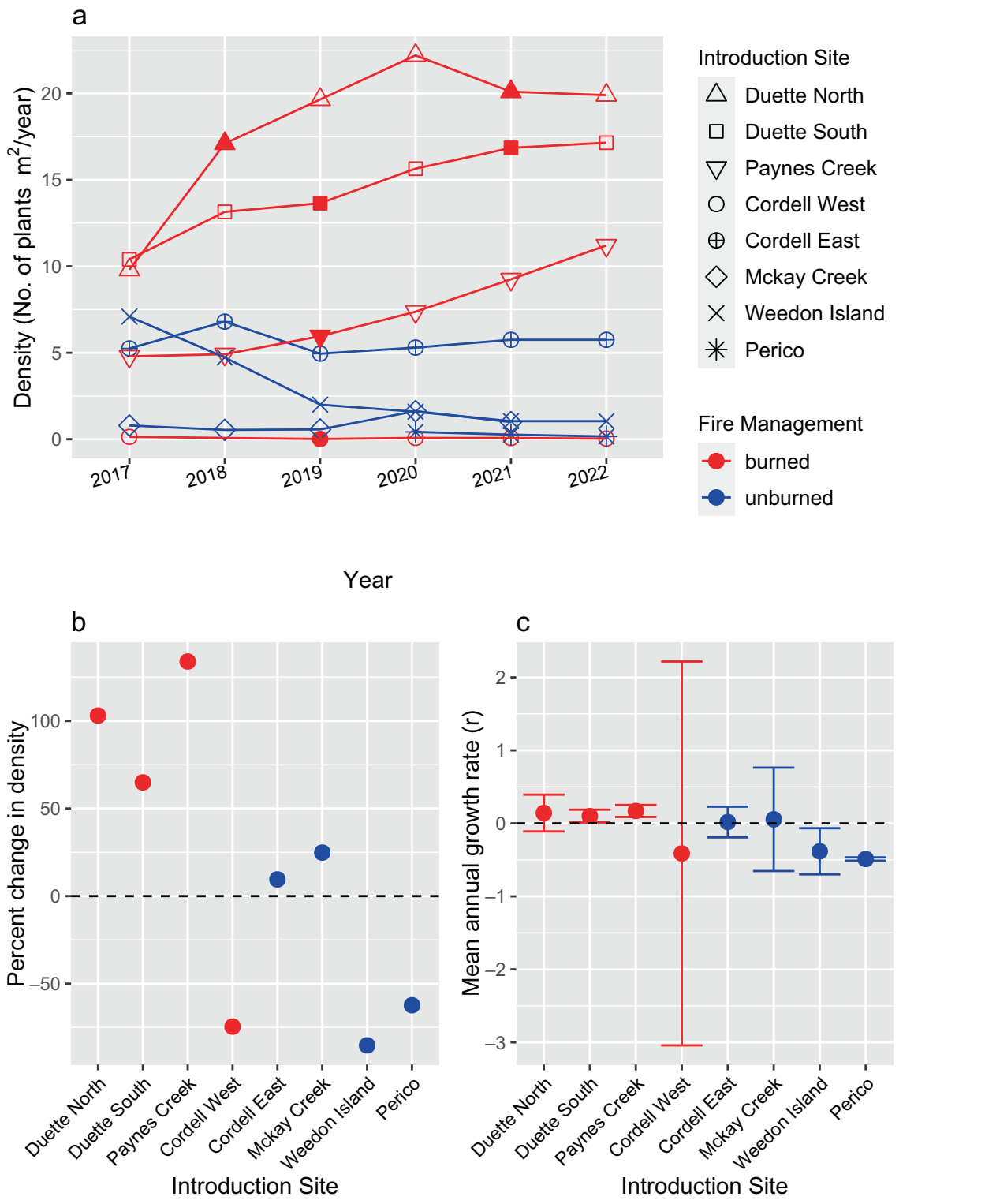

FIGURE 2 Panel (a) shows density (number of plants/m^2/year) over the 6-year period by site; filled-in points indicate the year that the site was burned. Panel (b) shows percent change in density during the study, calculated as (final density-initial density)/initial density. Panel (c) shows the mean annual growth rate (r) of *Chrysopsis floridana* by introduction site for years 2017–2022; error bars represent one standard error of each estimated mean. For all panels, color indicates fire management: Red points are sites that were burned at least once from 2017 to 2022 and blue points are sites that were not burned during this time. Symbols correspond to introduction site names, and filled-in symbols mark the year that the site received fire management.

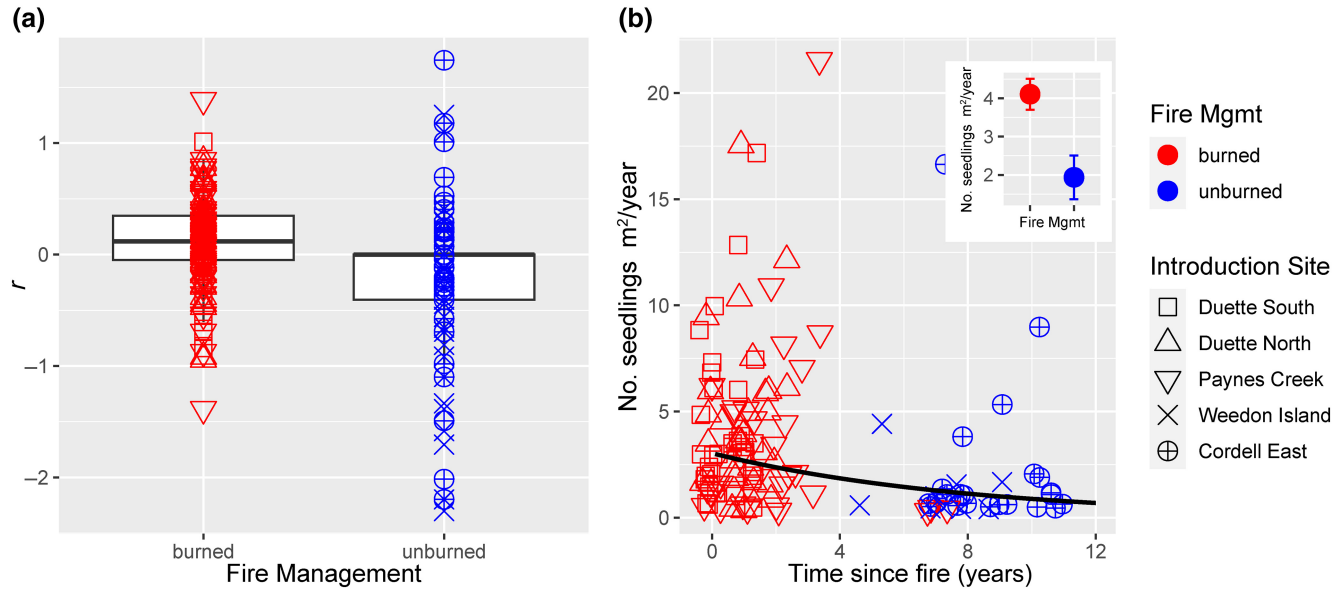

FIGURE 3 (a) Plot-level box plot of annual *r* for the five demographically monitored introduction sites. Symbols indicate the introduction site, and color of points indicate fire management: Red points are sites that were burned at least once from 2017 to 2022, and blue points are sites that were not burned during this time. Table 2 provides the statisticcs. (b) Plot level seedling density versus time-since-fire for the five demographically monitored introduction sites with an inset in the upper right corner showing overall mean seedling density per fire management classification: burned and unburned. The back-transformed regression line is shown.

The terms in all models were evaluated for significance using an analysis of deviance in the car package (Fox & Weisberg, 2019).

3 | RESULTS

3.1 | Population size and fire management

Three introduced sites showed substantial, steady population density increases from 2017 to 2022: Duette South, Duette North, and Paynes Creek, all of which received fire management at least once during the study period (Figure 2a). Populations densities in 2022 were higher for burned populations and, counterintuitively, in introduction sites that received fewer plants (introduction size × management; $F_{1,4} = 76.9$, $p < .001$). Of these three sites that received fire management and showed population density increases, the mean fire-return interval was 4.8 years (SD = 2.38). The fourth burned site (Cordell West) decreased in population growth from 2017 to 2022 (Figure 2b); unlike the other burned sites, this site went 10 years between the population introduction and its first prescribed burn in 2019 (Table 2). In 2022, the last year of our monitoring, the three introduced sites that had been burned showed a >50% increase in population density since 2017 (Figure 2b). Unburned sites, on the other hand, showed plot-level *r*-values .34 lower than those of burned sites, corresponding to a 29% reduction in growth rate ($\chi^2 = 18.51$, df = 1, $p < .001$) (Table 2, Figure 3a). Two of the unburned sites, Weedon Island and Perico, showed overall declines in population density (Figure 2b) and mean *r*-values over the 6-year period (Figure 2c). Cordell East and McKay Creek showed slight increases in population density from 2017 to 2022 (Figure 2b); McKay Creek had highly variable *r*-values, whereas Cordell East's *r*-values were mostly positive (Figure 2c).

LMMs for plant densities (number of plants/m^2) by stage class (seedling, juvenile, and adult) showed a significant negative relationship between time-since-fire and seedling density, with an estimate of 12% lower seedling density for each year a site went unburned ($\chi^2 = 20.3$, df = 1, $p < .001$) (Figure 3b) but did not show a significant relationship of time-since-fire on juvenile ($\chi^2 = 1.3$, df = 1, $p = .236$) or adult density ($\chi^2 = 2.7$, df = 1, $p = .098$) (Table 2).

3.2 | Population vital rates and fire management

Survival rates were generally higher at sites with fire management (Table 2). We found a 61% probability of juvenile survival at unburned sites compared with an 88% probability at burned sites ($\chi^2 = 8.7$, df = 1, $p = .003$). For adults, there was a 70% probability of survival at unburned sites and a 90% probability at burned

sites ($\chi^2 = 9.4$, df = 1, $p = .002$). Juveniles with relatively large diameters were more likely to transition to adults ($\chi^2 = 128.73$, df = 1, $p < .001$), but neither fire management ($\chi^2 = 2.3$, df = 1, $p = .129$) nor the fire management × rosette diameter interaction ($\chi^2 = 0.004$, df = 1, $p = .945$) affected this transition. Similarly, fire management had no effect on year-to-year growth or year-to-year flowering head production.

4 | DISCUSSION

Successful introductions of listed plant species rely on continued habitat management, and long-term monitoring of these introductions is critical to inform decision-making. In this study, we analyzed 6 years of demographic and count-based stage class monitoring of eight introduced populations of the previously federally endangered *C. floridana* and reported three key findings. First, introduced populations at burned sites with fire management generally increased in density over time, while those at unburned sites generally declined. Second, those sites with fire management and growing populations had an average fire-return interval of 4.8 years, a figure that we consider to be an approximate, yet useful estimate of the fire-return interval required for successful management of the species. Third, our analyses of vital rates indicated that fire is particularly important for the recruitment of new seedlings and the survival of juveniles and adults and less so for other vital rates related to growth and flowering. Based on these findings, we conclude that the studied introduced *C. floridana* populations have not all received the fire management required for the long-term success. Critically, these analyses suggest that post-delisting monitoring and decision-making should consider the extent of fire management in addition to multi-year population trends for the long-term outlooks of the introduced populations.

4.1 | Fire management

Based on our research, we suggest that managers apply prescribed fire to populations approximately every 5 years on average. In contrast, the USFWS federal register references a fire-return interval estimate of 10 years based on expert opinion (USFWS, 2024a), a time period that may be too long to support *C. floridana* population persistence. While our estimate is limited because we did not manipulate fire regimes or conduct a formal population viability analysis, our data show that populations experiencing regular fire are generally increasing. Importantly, the only burned population that decreased (Cordell West) went 10 years post-introduction before fire was prescribed, suggesting that this length of fire regime could be too long; benefits of fire for *C. floridana* populations are critical yet also short-lived. Almost all other populations with no fire management during our study period had been burned previously at least once post-introduction (Table 1) but were declining during our study period. Such rapid population surges followed by decline are common for several herbaceous species in the Florida scrub (e.g., Menges & Quintana-Ascencio, 2004; Quintana-Ascencio et al., 2018). Though our dataset is not robust enough to conduct a detailed population analysis (e.g., Menges et al., 2006; Menges & Quintana-Ascencio, 2004), our current estimate is important for directing management in the interim. Given that populations of *C. floridana* seem particularly sensitive to lack of fire, ensuring that fire is prescribed regularly will be critical for population viability and should be considered in the delisting process.

4.2 | Fire and demography

Fire plays an important role in the demography of *C. floridana*, boosting seedling recruitment and increasing the probability of survival of both juveniles and adults, resulting in overall higher rates of population growth. Yet, the importance of fire does not appear to extend to other vital rates, such as rosette or flowering stem growth or number of flowers per plant, which did not differ between sites with and without fire management, suggesting that, once established as seedlings, plants generally produce similar numbers of flowering stems regardless of the fire history. It is possible that other unmeasured metrics of plant performance could also increase in the post-fire environment, such as seed production or seed viability, and future research could be directed toward such work.

An important knowledge gap in our understanding of *C. floridana* is the seed ecology of the species. For example, it is unknown to what extent the species forms a persistent seed bank like other herbaceous scrub species. Other scrub species in the Asteraceae family lack a persistent seed bank (Stephens et al., 2014) and germinate readily. A persistent seed bank can help a fire-dependent population recover after many of the individuals have died, while a lack of a seed bank may require frequent fires for population persistence. In the latter case, a declining population with low seed production may not positively respond to fire, as it has little seed bank to draw from. Such may have been the case for the Cordell

West population, which experienced its first fire 10 years ago post-introduction.

In a broader sense, this work also emphasizes that endemic and endangered plant introductions require ecosystem-specific long-term follow-up management, especially since such plants are often found in unique ecosystems with limited geographical ranges. In the Florida scrub, where plants are adapted to historically regular natural wildfires caused by lightning from thunderstorms, fire suppression leads to increased shrub cover (e.g., *Quercus* spp., *Lyonia* spp.) that reduces the open sand gaps and thus lowers plant species abundance (Menges & Hawkes, 1998, Menges, 2008). From our results, it is evident that regular prescribed fire management supports the long-term success of introduced *C. floridana*.

4.3 | Plant introductions

As a biodiversity conservation strategy, the objective of plant introductions is to create and maintain self-sustaining populations that can survive both long-term and short-term (Monks et al., 2012). However, introductions are rarely monitored over long periods of time (Godefroid et al., 2011), and long-term monitoring is critical for accurate evaluations (e.g., Colas et al., 2008; David et al., 2024; López-Borghesi et al., 2023). To optimize the investments of resource-intensive plant introduction efforts (Simberloff, 2003), we recommend that sites with regular fire management regimens be prioritized as a criterion for introduction site selections of fire-dependent scrub species. Such a decision is not simple in practice; several of the unburned sites in our study had prescribed burning pre- or post-introduction, but a lack of fire in subsequent years has hurt the unburned populations. Of course, whether a site can be managed with fire can vary over time with changes in priorities and resources of decision-makers such as site managers, agency leadership, and other government authorities. Yet, for successful rare plant introductions in any ecosystem, the long-term prospect for success should include ongoing follow-up management as a consideration.

4.4 | USFWS delisting

Our findings question whether the species' persistence is secure in light of the recent delisting decision (USFWS, 2024a). Several of the introduced sites that might have been considered established are, in fact, declining due to lack of fire, and the estimated fire-return interval of 10 years used in the justification for delisting is, based on our findings, likely too high (USFWS, 2024a). Without adequate fire management, these introduced populations might not persist despite their locations on protected lands. Furthermore, while not specifically studied here, wild, unprotected populations are at similar risk of extirpation, particularly if they are left unmanaged, as in the cases of thick, overgrown woody vegetation at sites with wild C. *floridana* populations (Ward et al., 2022). We suggest that additional introductions, continued and frequent fire management, and long-term monitoring during the post-delisting period are needed to ensure the longevity of this species.

The ultimate goal for plant conservation is to improve the outlook for imperiled species, and for federally endangered species in the USA, this goal includes delisting. Yet, the delisting of federally listed plant species rarely occurs—only 36 species total have been delisted (942 species currently listed), of which 25 were delisted because they were determined to have recovered (USFWS, 2024b). Such few instances of delisting could suggest that the process is still relatively untested and could be refined. Our study highlights two outstanding questions where further clarification is needed in the process of delisting. First, how should introduced populations be treated with regard to delisting criteria? Should they be treated differently from wild populations, and, if so, at what point should they qualify as "established"? Given the developing science of determining what introduction success looks like (e.g., Bellis et al., 2023; Menges, 2008), this question is not necessarily straightforward to answer. Second, for species that are management-dependent but otherwise secure, how do we ensure that populations are sufficiently managed after delisting? Further development of federal or state policies may be a fruitful approach, though such policies are generally lacking for delisted plant species (Haines et al., 2021). As more plant species are delisted due to recovery, further development of delisting criteria with regards to introductions and management will benefit the process and, hopefully, protect the species.

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REFERENCES

Albrecht, M. A., & Maschinski, J. (2012). Influence of founder population size, propagule stages, and life history on the survival of reintroduced plant populations. In *Plant Reintroduction in a Changing Climate: Promises and Perils* (pp. 171–188). Island Press-Center for Resource Economics.

Amidon, F. A., Marshall, A. P., Radley, P., & Vanderwerf, E. (2021). Post-delisting monitoring of the Tinian Monarch (Monarcha takatsukasae). Final Report to the US Fish and Wildlife Service.

Bates, D., Maechler, M., Bolker, B., & Walker, S. (2015). Fitting linear mixed-effects models using lme4. *Journal of Statistical Software*, *67*(1), 1–48.

Bellis, J., Osazuwa-Peters, O., Maschinski, J., Keir, M. J., Parsons, E. W., Kaye, T. N., Kunz, M., Possley, J., Menges, E., Smith, S. A., Roth, D., Brewer, D., Brumback, W., Lange, J. J., Niederer, C., Turner-Skoff, J. B., Bontrager, M., Braham, R., Coppoletta, M., ... Albrecht, M. A. (2023). Identifying predictors of translocation success in rare plant species. *Conservation Biology*, *38*, e14190.

Colas, B., Kirchner, F., Riba, M., Olivieri, I., Mignot, A., Imbert, E., Beltrame, C., Carbonell, D., & Fréville, H. (2008). Restoration demography: A 10-year demographic comparison between introduced and natural populations of endemic *Centaurea corymbosa* (Asteraceae). *Journal of Applied Ecology*, *45*, 1468–1476.

David, A. S., Naccarato, A. M., Smith, S. A., Peterson, C. L., Pence, V. C., Crate, S. J., Koontz, S. M., Dole, H. E., & Menges, E. S. (2024). Evaluating long-term introduction success of the Florida scrub endemic *Crotalaria avonensis* (Fabaceae) using a decade of demographic data. *Restoration Ecology*, *32*, e14082.

ESA. (1973). *Endangered species act, public law 205, U.S. Statutes at Large 87* (pp. 884–903). ESA.

Evans, D. M., Che-Castaldo, J. P., Crouse, D., Davis, F. W., Epanchin-Niell, R., Flather, C. H., ... Williams, B. K. (2016). Species recovery in the United States: Increasing the effectiveness of the endangered species act. *Issues in Ecology*, *20*, 1–28.

Fox, J., & Weisberg, S. (2019). *An {R} companion to applied regression* (3rd ed.). Sage. https://socialsciences.mcmaster.ca/jfox/Books/Companion/

Godefroid, S., Piazza, C., Rossi, G., Buord, S., Stevens, A. D., Aguraiuja, R., Cowell, C., Weekley, C. W., Vogg, G., Iriondo, J. M., Johnson, I., Dixon, B., Gordon, D., Magnanon, S., Valentin, B., Bjureke, K., Koopman, R., Vicens, M., Virevaire, M., & Vanderborght, T. (2011). How successful are plant species reintroductions? *Biological Conservation*, *144*(2), 672–682.

Haines, A. M., Leu, M., Costante, D. M., Treakle, T. C., Parenti, C., Miller, J. R. B., & Malcom, J. W. (2021). Benchmark for the ESA: Having a backbone is good for recovery. *Frontiers in Conservation Science*, *2*. https://doi.org/10.3389/fcosc.2021.630490

Janecka, M. J., Janecka, J. E., Haines, A. M., Michaels, A., & Criscione, C. D. (2021). Post-delisting genetic monitoring reveals population subdivision along river and reservoir localities of the endemic Concho water snake (*Nerodia harteri* paucimaculata). *Conservation Genetics*, *22*, 1005–1021.

Lambert, B. B., & Menges, E. S. (1996). The effects of light, soil disturbance and presence of organic litter on the field germination and survival of the Florida goldenaster, *Chrysopsis floridana* Small. *Florida Scientist*, *59*, 121–137.

Lenth, R. (2023). *emmeans: Estimated Marginal Means, aka Least-Squares Means*. R package version 1.8.9. https://CRAN.R-project.org/package=emmeans

López-Borghesi, F., Koontz, S. M., Smith, S. A., Haller Crate, S. J., Quintana-Ascencio, P. F., & Menges, E. S. (2023). Leveraging projection models to evaluate long-term dynamics of scrub mint translocations. *Conservation Science and Practice*, *5*, e12947.

Maschinski, J., Coates, D., Monks, L., Dillon, R., Barrett, S., Possley, J., ... Copp, C. (2023). Rare and threatened plant conservation translocations: Lessons learned and future directions. *Ecological Restoration*, 287–322.

Menges, E. S. (2008). Restoration demography and genetics of plants: When is a translocation successful? *Australian Journal of Botany*, *56*(3), 187–196.

Menges, E. S., Ascencio, P. F. Q., 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.

Menges, E. S., Crate, S. J., & Quintana-Ascencio, P. F. (2017). Dynamics of gaps, vegetation, and plant species with and without fire. *American Journal of Botany*, *104*(12), 1825–1836.

Menges, E. S., & Gordon, D. R. (1996). Three levels of monitoring intensity for rare plant species. *Natural Areas Journal*, *16*(3), 227–237.

Menges, E. S., & Hawkes, C. V. (1998). Interactive effects of fire and microhabitat on plants of Florida scrub. *Ecological Applications*, *8*(4), 935–946.

Menges, E. S., & Quintana-Ascencio, P. F. (2004). Population viability with fire in *Eryngium cuneifolium*: Deciphering a decade of demographic data. *Ecological Monographs*, *74*(1), 79–99.

Monks, L., Coates, D., Bell, T., & Bowles, M. L. (2012). Determining success criteria for reintroductions of threatened long-lived plants. In J. Maschinski & K. E. Haskins (Eds.), *Plant Reintroduction in a Changing Climate: Promises and Perils* (pp. 189–208). Island Press/Center for Resource Economics.

Neel, M. C., & Che-Castaldo, J. P. (2013). Predicting recovery criteria for threatened and endangered plant species on the basis of past abundances and biological traits. *Conservation Biology*, *27*, 385–397.

Peterson, C. L. (2014). An integrated conservation program for the protection of Florida's rare and endangered flora-north and central region. In *Final report for Florida Department of Agriculture and Consumer Service Agreement No. 020160*. Bok Tower Gardens.

Peterson, C. L. (2016). An integrated conservation program for the protection of Florida's rare and endangered flora-north and central region. In *Final report for Florida Department of Agriculture and Consumer Service Agreement No. 22645*. Bok Tower Gardens.

Quintana-Ascencio, P. F., Koontz, S. M., Smith, S. A., Sclater, V. L., David, A. S., & Menges, E. S. (2018). Predicting landscape-level distribution and abundance: Integrating demography, fire, elevation and landscape habitat conFig.Uration. *Journal of Ecology*, *106*(6), 2395–2408.

R Core Team. (2023). *R: A language and environment for statistical Computing*. R Foundation for Statistical Computing. https://www.R-project.org/

Schemske, D. W., Husband, B. C., Ruckelshaus, M. H., Goodwillie, C., Parker, I. M., & Bishop, J. G. (1994). Evaluating approaches to the conservation of rare and endangered plants. *Ecology*, *75*(3), 585–606.

Simberloff, D. (2003). How much information on population biology is needed to manage introduced species? *Conservation Biology*, *17*(1), 83–92.

Stephens, E. L., Tye, M. R., & Quintana-Ascencio, P. F. (2014). Habitat and microsite influence demography of two herbs in intact and degraded scrub. *Population Ecology*, *56*, 447–461.

U.S. Fish and Wildlife Service (USFWS). (1988). *Recovery plan for Florida Golden Aster* (p. 15). U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service (USFWS). (2024a). *Endangered and Threatened Wildlife and Plants; Removal of Chrysopsis floridana (Florida golden aster) From the Federal List of Endangered and Threatened Plants, 89 FR 15763 (proposed June 24, 2021)*. https://www.federalregister.gov/documents/2024/03/05/2024-04278/endangered-and-threatened-wildlife-and-plants-removal-of-chrysopsis-floridana-florida-golden-aster

U.S. Fish and Wildlife Service (USFWS). (2024b). Environmental Conservation Online System. Retrieved September 30 2024, from https://ecos.fws.gov/ecp/

Ward, S. G., Menges, E. S., Charton, K. T., Gonsiska, P. A., & Peterson, C. L. (2022). Using demographic findings to compare wild and translocated populations of Florida goldenaster (*Chrysopsis floridana*) in west-central Florida, USA. *Restoration Ecology*, *30*(5), e13577.

Yates, C. J., & Broadhurst, L. M. (2002). Assessing limitations on population growth in two critically endangered acacia taxa. *Biological Conservation*, *108*, 13–26.

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