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Fire and Invasive Plant Interactions[1]

Deb Stone and Michael Andreu[2]

Introduction

Managing longleaf pine ecosystems requires mimicking natural processes such as fire regimes, and balancing sometimes competing management actions (Katherine Kirkman and Jack 2017, Figure 1). Landscape-scale fire was historically a major driver of both the ecosystem services and the species composition of natural communities in longleaf pine ecosystems (Van Lear et al. 2005; Waldrop, White, and Jones 1992). Florida is home to approximately 1,400 non-native species, of which 81 are considered Category 1 invasive plants by the Florida Invasive Species Council; this problem is compounded by major international ports and various industries like agriculture, horticulture, and the aquarium trade (FISC 2019; Lieurance et al. 2013). Because prescribed fire and invasive species control are two common land-management actions in Florida, it is important that land managers have a solid understanding of their interactions and how they affect the surrounding ecosystems.

Interactions between fire and invasive plants can be roughly categorized into two reciprocal groups: the effects of the invasive plant on fire regime, and the impacts of fire on the invasive plant or other flora. Invasive species can impact fire regimes by altering fuel bed load, moisture content, ignitability, continuity and structure; by changing fire frequency, seasonality or intensity; and by creating conditions in which they are better suited to survive than native species (M. L. Brooks et al. 2004; D'Antonio and Vitousek 1992; Zouhar, Smith, and Sutherland 2008; Mack and Antonio 1998). These impacts often cause feedback loops that continue to alter the fire regime and increase the abundance of the invasive species, creating an invasive plant-fire regime cycle (D'Antonio and Vitousek 1992; M. L. Brooks et al. 2004). Fire can act as a "global herbivore," removing vegetation and providing managers access to either survey for or treat invasive species (Bond and Keeley 2005). Prescribed fire can impact invasive species by increasing light to the groundcover, increasing nutrient supply in a short burst, leaving soil bare or even disturbed such as in fire breaks, and killing or top-killing vegetation (Brooks and Lusk 2008; Evans et al. 2016). These interactions can vary not only by species but also by differences in how fire is applied, such as seasonality or intensity, or by variations in site conditions. For some invasive species, prescribed fire can be a control method when applied at the right time of year or in the right life history stage. However, for other species, fire encourages invasion through increased growth and reduced competition, or by stimulating flowering and seeding or vigorous resprouting (Brooks and Lusk 2008; Gordon 1998; Zouhar, Smith, and Sutherland 2008).

While these interactions complicate the management of natural lands and can drastically alter ecosystem functions and composition, not every invasive species contributes to the cycle or behaves in the same manner. So, understanding what is currently known about how multiple species behave can help land managers make better-informed decisions. This publication will review four invasive species in the southeastern United States that have different interactions with fire: cogongrass (*Imperata cylindrica*), Chinese tallow (*Triadica sebifera*), Old World climbing fern (*Lygodium*

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2. Deb Stone, doctoral candidate; and Michael G. Andreu, associate professor in forest systems and Extension specialist; School of Forestry, Fisheries and Geomatic Sciences, UF/IFAS Extension, Gainesville 32611.

microphyllum), and Japanese climbing fern (*L. japonicum*). These four plant species are widespread in Florida and have large numbers of reports in the Early Detection and Distribution Mapping System (EDDMapS 2021).

Figure 1. Aerial view of a person igniting a prescribed fire along a fire line in a non-native grass-dominated system in Florida.
Credits: Maria Zondervan, SJRWMD

Cogongrass (*Imperata cylindrica*)

Overview

This section provides cursory information on cogongrass. More detailed species information can be found in the Center for Aquatic and Invasive Plants (CAIP) Plant Directory, and treatment information can be found in Integrated Management of Non-Native Plants in Natural Areas of Florida (Enloe et al. 2018).

GEOGRAPHICAL INFORMATION

Native to southeast Asia, cogongrass is often said to be one of the top ten invasive plants in the entire world, as it can be found on every continent except Antarctica and is a major weed in many tropical and subtropical systems (MacDonald 2004). It is problematic in the many natural and managed ecosystems of the southeastern United States, with infestations from Texas up to Tennessee, over to North Carolina and in all Florida counties (EDDMapS 2021).

IDENTIFICATION

Cogongrass is a perennial, rhizomatous grass with leaf blades ½ inch to ¾ inch wide and a white, off-center midrib. New blades are bright green, while senesced leaves can be orange to brown (CAIP 2021). This bright, lime-green color—especially bright when backlit by the sun—combined with quick regrowth helps make cogongrass an easy species to locate a few months after a fire (personal observation, Figure 2). Its inflorescence appears cylindrical and spike-like, with densely packed, white, fluffy seeds (MacDonald 2004).

Figure 2. A field technician ties flagging tape to mark a patch of newly discovered cogongrass (edge marked in yellow) during a post-fire invasive plant survey in a recently burned grassland.
Credits: Deb Stone

BASIC LIFE HISTORY

Cogongrass has several adaptations that make it an excellent competitor: numerous wind-dispersed seeds, a well-developed rhizome system, and adaptations to poor soil conditions, drought, and fire (MacDonald 2004). It tolerates a wide range of growing conditions, easily weathering fluctuations in soil moisture, soil fertility, and available light, and it thrives in disturbed ecosystems (Jose et al. 2002). Initial germination rates can be as high as 98%, but drop quickly to near 0% in just three months in natural settings, and within 11 months when stored indoors (Dozier et al. 1998; MacDonald 2004). It is considered a better competitor than most native species and resprouts quickly after aboveground biomass is removed (Ramsey et al. 2003; Daneshgar and Jose 2009).

How Cogongrass Impacts Fire Ecology

Cogongrass has several well-documented impacts on fire regimes, primarily through increasing fuel loads and fire intensity. Researchers in south Florida found that plots invaded with cogongrass had 1.7 times as much biomass as uninvaded plots, and most of this increase was due to cogongrass litter (Platt and Gottschalk 2001). A study in sandhill showed cogongrass invasion resulted in increased fine fuel loads for taller fuels (approximately five-fold increase for heights above 0.5 m), greater maximum temperatures (~250°C invaded vs. ~175°C uninvaded at 1.5 m), and higher mortality in some longleaf pine juveniles (80% invaded vs. 49% uninvaded for trees 0.5 m to 1 m tall (Lippincott 2000). There is some data to support the theory that cogongrass can shorten fire return intervals and increase fire severity, with recorded temperatures in invaded stands reaching around 450°C compared to just over 300°C in uninvaded stands (Howard 2005; Jose et al. 2002). Other research in Florida has shown that fuel structure can impact fire behavior. Specifically, this research shows that the upright growth of cogongrass leads to shorter heating duration but higher flame heights and rates of spread when compared to horizontally arranged fuels; this can cause more mortality to juvenile trees (Dillon, Hiatt, and Flory 2021, Figure 3). Cogongrass can potentially affect fuel moisture and therefore fire behavior as it is reported to reduce soil moisture by up to half as much in invaded areas compared to uninvaded areas (Jose et al. 2002). These combined effects on the regional fire regime have been quantified by comparing spatial invasives data to fire occurrence and severity data, showing that cogongrass significantly increased fire occurrence and frequency, and also had a positive but less significant effect on fire size (Fusco et al. 2019).

How Cogongrass Responds to Fire

Fire is considered a major factor in cogongrass spread in many parts of the world (Howard 2005). One of cogongrass's main adaptations to fire is its extensive rhizome structure, which accounts for approximately 60% of its biomass and allows it to rapidly resprout after a fire and colonize any available space (Howard 2005; MacDonald 2004). Fire will top-kill cogongrass and remove much of its leaf litter, but it is unlikely that soil temperatures reach the needed level to kill its rhizomes. A study on the effect of different temperatures and durations of fire on cogongrass rhizomes found that the required exposure time for mortality at 65°C was 25 minutes, 5 minutes for 79°C, and 1 minute for 187°C (Bryson, Koger, and Byrd 2007).

The ability to quickly regenerate following fire is bolstered by a flush in needed nutrients during this time. A study in Australia looked at post-fire soil conditions, specifically nitrogen (N) and phosphorus (P), and found that cogongrass in burned areas had lower N:P ratios, that N addition reduced cogongrass growth, and P addition increased it (Butler, Lewis, and Chen 2021). This research indicates that there is a short-term positive effect of fire on cogongrass through the increased P, but long-term fire exclusion might hinder cogongrass growth by increasing soil N. Researchers in India showed nearly 70% higher biomass production during the season immediately post-fire than during the post-monsoon season (Pathak et al. 2018).

Figure 3. Aerially ignited prescribed fire in a cogongrass-infested field showing increased fire spread rate where the spheres (ignition source) landed in cogongrass (white arrow, with yellow arrows pointing to other cogongrass patches).
Credits: Maria Zondervan, SJRWMD

Cogongrass seedlings also benefit from the post-fire environment. Studies have shown that cogongrass seedlings have nearly twice the survival rate in burned areas than in unburned, even though germination rates were comparable between the two (King and Grace 2000). A two-year study showed that fire alone as a treatment method had no net effect on cogongrass coverage, although coverage in the first year was reduced in burned plots compared to unburned plots (Enloe et al. 2013).

Management Implications

Cogongrass can increase fire intensity, causing mortality in seedling and juvenile trees, which affects the composition, structure, and function of forests and savannas (Jose et al. 2002; Cardoso et al. 2018). Cogongrass also has the potential to shorten fire return intervals, which could reduce native species cover; but complete fire exclusion results in similar outcomes and therefore isn't a viable management choice in many fire-dependent systems (Howard 2005). While studies have also shown that burning alone doesn't impact cogongrass and can even encourage it, burning can be used as part of a multi-pronged approach combining chemical and other mechanical control methods, with herbicide treatment reducing cogongrass coverage to less than 4% by the end of one study in both burned and unburned plots (Enloe et al. 2013). Revegetation with native species is recommended for larger infestations. Without revegetation, cogongrass will re-invade, or a secondary invasion could occur.

Due to these interactions with fire and the resulting ecological impacts, one of the most important actions land managers can take is to find cogongrass infestations when they are small. Cogongrass's quick resprouting response to fire (Howard 2005; MacDonald 2004) means that it is easier to spot during surveys after a prescribed fire, during which time traversing through the area is also easier. With dormant-season or early growing-season fires, the window to easily observe newly sprouted cogongrass is about 12 weeks post-fire. Native species will respond more quickly after later growing-season burns, so those burns may have a better detection window that only lasts 4 to 6 weeks post-fire (Ramsey et al. 2003; Daneshgar and Jose 2009, personal observation).

The positive feedbacks between fire and cogongrass can help one prioritize areas for treatment. Because of its quick vegetative response to fire, and the increased chance of seedling survival post-fire, cogongrass has the ability to expand more rapidly in frequently burned areas than in less frequently burned ones (Butler, Lewis, and Chen 2021; Pathak et al. 2018; King and Grace 2000). Therefore, areas that are burned more frequently should be prioritized for treatment.

Coordinating burn timing and frequency around cogongrass management is one of the best tools to reduce cogongrass spread. Given these positive effects, prescribed fire can easily add fuel to the proverbial invasion fire and increase the spread rate, especially if prescribed fire is not applied in a coordinated and well-planned manner. If cogongrass is present on the landscape, particularly if grows in multiple, small, scattered patches in otherwise high-quality habitat, the best option is putting a temporary hold on applying fire until the cogongrass is removed or greatly reduced. Native species have a better chance of persisting against several years of fire suppression than they do of competing against fire-charged cogongrass.

When fire must be applied, burn at the longer end of the fire return interval for the habitat type, and try to burn well into the growing season, avoiding winter burns as much as possible. This strategy minimizes the positive effect on cogongrass, reduces the chance of cogongrass seed being around the burn zone to take advantage of less competition and increased nutrients, and allows for native species to make a quicker recovery and compete better for the available resources (Ramsey et al. 2003; Daneshgar and Jose 2009; King and Grace 2000).

The reported increases in cogongrass seedling survival in burned areas should influence treatment and burning schedules. For example, land managers could plan herbicide treatments around a burn schedule to minimize the nearby presence of potential seed sources or schedule prescribed burns outside of the season of high seed production (King and Grace 2000). Given that cogongrass seedling survival is better in newly burned areas, managers should not burn zones with cogongrass in the area when viable seeds are present, which is likely late winter and through early growing season based on typical flowering times and seed longevity (Dozier et al. 1998; MacDonald 2004). The best way to avoid burning during seed-viability season is to make sure that zones on the next year's burn plan—and the zones adjacent to them—are treated before they flower. If zones cannot be pre-treated, managers should consider postponing the burn until at least three to four months after seed set, as most seed will have lost viability by then.

Take-Home Points:

- Top survey priority: Survey for new infestations 4 to 12 weeks after fire (use the shorter window for burns later in the growing season).
- Top treatment priority: Administer follow-up/spot treatments in regularly burned areas.
- If you have lots of small, scattered patches (especially if in otherwise good habitat), hold off on fire until you can get the cogongrass under control.
- If complete fire exclusion isn't an option, burn at the longest fire return interval you can for the habitat until you get the cogongrass under control.
- Burn later in the growing season for better native response.
- Don't burn an area if/when a cogongrass seed source is nearby. Plan herbicide treatments around your burn schedule to minimize the nearby presence of potential seed sources. This means thinking about a year ahead. Alternatively, schedule prescribed burns at least 3 months after the season of high seed production. (Think about what's across the fire line, too.)

Chinese tallow (*Triadica sebifera*)

Overview

This section provides cursory information on Chinese tallow. More detailed species information can be found in the Center for Aquatic and Invasive Plants (CAIP) Plant Directory, and treatment information can be found in Integrated Management of Non-Native Plants in Natural Areas of Florida (Enloe et al. 2018).

GEOGRAPHICAL INFORMATION

Chinese tallow, a native species in eastern Asia, is a popular ornamental tree in the southeastern United States due mainly to its fall foliage, distinctive seeds, and low maintenance needs. Chinese tallow is documented throughout the southeast from Texas over to North Carolina, and throughout most of Florida with the exception of six southern counties. Outlier populations occur in California, Kentucky, Tennessee, and Wisconsin, although some may be ornamental rather than established (EDDMapS 2021).

IDENTIFICATION

Chinese tallow typically reaches about 20 feet tall but can grow as tall as 40 or 50 feet. Its leaves have fairly distinct acuminate tips and turn orange red in the fall. The seeds are covered in a white waxy coating, resembling popcorn and giving it one of its other common names, the popcorn tree (CAIP 2021).

BASIC LIFE HISTORY

Chinese tallow has been shown to grow in many different habitats, such as bottomland forest, maritime forest, coastal prairie, or slash pine forest, indicating tolerance of a wide variety of site conditions (Bruce, Cameron, and Harcombe 1995; CAIP 2021; Smith, Nicholas, and Zedaker 1997; Tian et al. 2017). Within the first two years, Chinese tallow can grow to 2.8 m tall from seed and more than 5.5 m as resprouts (Herbert W Scheld and Cowles 1981). This species is also a prolific seed producer. An individual tree can begin flowering when it is only three feet tall, and its seeds are dispersed by wildlife, water, and humans, making Chinese tallow a very competitive invasive species (Godfrey 1988; H. W. Scheld et al. 1984).

How Chinese Tallow Impacts Fire Ecology

Chinese tallow mainly impacts fire by altering the fuel bed structure and moisture content, decreasing the chance that fuels will ignite and carry fire (Meyer 2011; Zouhar et al. 2008). One study found that Chinese tallow reached its maximum percent cover of approximately 60% by the 6- to 10-year age class, which coincided with a 50% decline in the percent cover of grasses and forbs (Bruce et al. 1995). These herbaceous fine fuels are some of the major carriers of fire, so Chinese tallow infestations can impact the spatial footprint of fire, and eventually might impact the fire frequency of the area by continually reducing fire spread.

How Chinese Tallow Responds To Fire

Chinese tallow's main adaptations to fire are thicker bark, rapid re-growth in the season following fire, and root sprouting (Grace 1998; Meyer 2011). One study found that 100% of trees re-sprouted following growing-season fire, but 70% of trees less than 2 meters tall were dead or top-killed by the end of the next growing season (Grace 1998). This same study found that dormant-season fires did not have a significant effect on Chinese tallow trees. Fire has been reported to kill Chinese tallow trees less than 1 inch in diameter, but mature trees typically survive prescribed fires (Meyer 2011).

Studies on the effects of fire on germination of Chinese tallow have yielded varied results. One study found higher germination rates in a regularly burned area, and the authors proposed that this is likely due to the removal of litter (indirect effect) and encouragement of germination (direct effect) of *T. sebifera* seeds (Samuels 2004, Figure 4). However, another study reported decreased germination immediately after a fire (7.4% in unburned plots versus 1% in burned plots) (Burns and Miller 2004). A study in North Carolina found no survival in Chinese tallow seedings at 2 years in burned plots, whereas unburned plots averaged 43% survival at 2 years and 8% at 42 months (Just, Hohmann, and Hoffmann 2017).

Figure 4. Chinese tallow seedlings (some with white arrows) grow scattered among grasses in a clearing created from treating mature trees, which are now fallen logs.
Credits: Deb Stone

Several studies have examined interactions between fire and Chinese tallow at a larger scale and have found that the effects of fire vary based on local conditions and the fire return interval. Invasion across Texas, Louisiana, Mississippi, and Alabama has been reported to be more likely in fire-damaged stands (defined as fire within the last five years with 25% or more of the trees being damaged), although the effect was statistically significant only in areas with very high invasion rates (Gan et al. 2009). Similar results were reported in Mississippi, with more Chinese tallow seedlings found around snags than live trees, as well as an increased invasion chance with lower canopy cover, lower shrub cover (<50%), slash pine overstory, and closer proximity to roads or trails (Fan 2018; Cheng et al. 2021). The authors suggest that fire promotes wildlife usage, which in turn may promote seed germination and seedling recruitment, especially in areas with nearby Chinese tallow infestations or human-mediated disturbance. Another study found that the presence of seed trees was the main determining factor on the direction of fire's impact on Chinese tallow invasion. A longer time since fire decreased invasion risk if seed trees were farther than 300 meters away but actually increased invasion risk when seed trees were closer (Yang et al. 2019). The authors suggest that fire has a two-fold effect on Chinese tallow in that it can kill seedlings but at the same time promote seedling growth by reducing competition. This study also found alarge increase in spread after 9–10 years of establishment, which is similar to the Bruce et al. study (1995) that found Chinese tallow reached its maximum cover class in the 6- to 10-year age class.

Management Implications

For Chinese tallow, the seasonality of fire has important implications for controlling young trees or new invasions, so managers should use a well-timed prescribed fire to limit the spread of satellite populations. The literature on fire's effects on germination is mixed, but there is broad consensus that the impact to larger trees is minimal in most prescribed fire scenarios. Chinese tallow can have negative feedbacks on fire, so chemical or mechanical control of larger infestations is likely necessary to help reinstate a more natural fire regime.

When dealing with larger infestations, research indicates that combining fire, herbicide, and mechanical treatment can provide good control, with a specific recommendation of mechanical treatment in spring, foliar treatment in fall, and growing-season fire in the second year,which can reduce Chinese tallow trees per acre by >95% (Pile et al. 2017b).

Integrating all the information on Chinese tallow spread across the landscape, it appears that it only takes about two years for Chinese tallow individuals to escape the fire trap (repeated top-kill and resprouting from fire), but it takes six to ten years before community-level effects start to take place (Grady and Hoffmann 2012; Hoffmann et al. 2020). The larger trees are much less likely to be controlled with fire, so managers can limit Chinese tallow's impacts and have a better chance of using fire for control if they catch an infestation before it is more than 5 years old (Grace 1998; Meyer 2011). To catch infestations while they are young, land managers need to survey suitable fire-dependent habitat at least every 4–5 years. Areas that are more likely to be invaded—highly disturbed areas, areas with more snags, lower canopy cover, or lower shrub cover—should be the highest priority for surveys, as well as areas frequented by fruit-dispersing wildlife (Fan 2018; Cheng et al. 2021).

Studies on Chinese tallow and fire have shown varying effects on Chinese tallow germination as well as spread across the landscape. Given that growing season fires can control small trees (less than 2 m tall or 1 in diameter at breast height [DBH]) and that Chinese tallow can grow past these size limits in as few as two years, managers with minor infestations should burn at the lower end of the fire return interval for that natural community and target windows when they can safely burn hot without damaging natives (Grace 1998; Meyer 2011; Herbert W Scheld and Cowles 1981; Gan et al. 2009).Catching an infestation in the early stages allows managers to use a growing-season fire to kill many of the smaller trees, resulting in less need for herbicide as well as easier access for treatment post-fire to target any larger individuals (Grace 1998, Pile et al 2017a).

However, many trees may coppice in the first season post-fire and then succumb to stress later on, so managers should wait until the second season post-fire to treat smaller individuals; waiting can save herbicide and reduce treatment costs (Grace 1998). A study by Yang et al. found that when seed trees are nearby, recently burned areas are at a decreased invasion risk compared to areas with a longer time since fire, which indicates that frequent fires effectively control Chinese tallow seedlings, thereby reducing Chinese tallow spread across an already invaded landscape. Other results in this study suggest that fire return intervals of less than three years would kill most if not all Chinese tallow seedlings and reduce Chinese tallow spread across fire-dependent habitats (Yang et al. 2019).

Take-Home Points:

- Survey at least every 5 years to increase the chance of catching new infestations at a size when fire can provide control.
- Top survey priority: Areas with human-mediated disturbances or extreme natural disturbances, areas where seed-dispersing birds congregate, and areas with lower shrub cover.
- Top treatment priority: target seed trees (>4 years old) and the area ~300 meters around them.
- Burn at the lower end of your fire return interval for the natural community, less than every 3 years if possible,

and try to target windows where you can safely burn hot without damaging natives.

- Use growing-season fire to kill smaller plants (less than 2 m tall or 1 in DBH).
- Be patient when you see post-fire coppicing the first year after fire, but follow up with treatment the second year after fire.

Climbing ferns (*Lygodium* spp.)

Overview

This section provides cursory information on climbing ferns. More detailed species information can be found in the Center for Aquatic and Invasive Plants (CAIP) Plant Directory, and treatment information can be found in Integrated Management of Non-Native Plants in Natural Areas of Florida (Enloe et al. 2018).

GEOGRAPHICAL INFORMATION

There are two invasive species of climbing fern found in Florida: Old World climbing fern (*L. microphyllum*) and Japanese climbing fern (*L. japonicum*). Old World climbing fern is native to the tropical and subtropical regions of Africa, Asia, and Oceania; and Japanese climbing fern is native to temperate and tropical eastern Asia, Australia, and the East Indies (Pemberton et al. 1998; Van Loan 2006). Old World climbing fern has become a major invasive species in the wetlands of south Florida and is slowly moving northward into central and northern Florida, with occurrences documented as far north as Duval and Bradford Counties (EDDMapS 2021). Japanese climbing fern has been documented across the southeastern United States, from Texas to South Carolina, including all but one county in Florida, and outliers have been observed in Tennessee, Kentucky, North Carolina, and Hawaii (EDDMapS 2021).

IDENTIFICATION

Old World climbing fern fronds can grow to nearly 100 feet long, with once compound leafy branches with a general oblong shape along the main stem. Japanese climbing fern leaves are compound with a general triangular shape and lacy appearance. Fertile leaflets for both species have a frilly appearance due to the sporangia on the margins (CAIP 2021).

BASIC LIFE HISTORY

Both Old World climbing fern (*L. microphyllum*) and Japanese climbing fern (*L. japonicum*) are true ferns that climb and create large, complex vine mats (Figure 5). Old World climbing fern can grow over trees and form thick horizontal mats, shading out any vegetation below them (Pemberton et al. 1998). Old World climbing fern spores have been shown to have germination rates averaging 90% in soil and 85% in water, but with lower rates in leaf litter, especially leaf litter containing wax myrtle (*Myrica sebifera*) (Call, Brandt, and DeAngelis 2007). In the control group for cold-tolerance germination tests, approximately 42% of Old World climbing fern spores germinated, while 50% of Japanese climbing fern spores germinated; freezing temperatures (-2.2°C) decreased Old World climbing fern survival with exposure times longer than 1 hour (Hutchinson and Langeland 2014). Japanese climbing fern is reported to have peak spore release in October (Van Loan 2006).

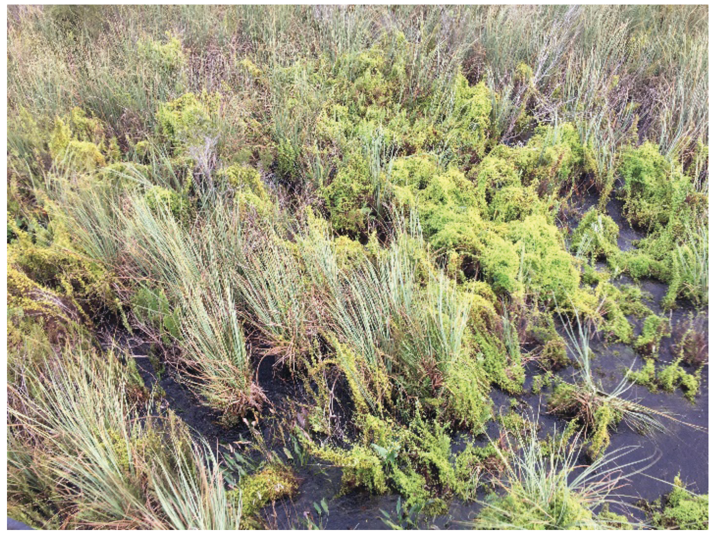

Figure 5. Old World climbing fern vines grow over grasses and open water patches in a fire-maintained marsh system.
Credits: Deb Stone

Japanese climbing fern inhabits drier areas than Old World climbing fern, and while it does not tend to smother trees or form mats of the same thickness, it has similar effects in the systems it invades. This reduced effect may be due in part to freezing temperatures that can occur in much of its current range (Van Loan 2006).

How Climbing Ferns Impact Fire Ecology

The main impact to fire of both climbing fern species is a change to vertical fuel continuity because the ferns provide ladder fuels up into the tree canopy, often allowing fire to spread into systems that have characteristics that would otherwise exclude fire, such as cypress domes or tree islands (Langeland, Enloe, and Hutchinson 2016; Minogue et al. 2009; O'Brien et al. 2010, Figure 6). However, chemical analysis has indicated that the lipids in the spores are very flammable, which could also help introduce fire to these habitats (Hutchinson and Langeland 2010).

Figure 6. Old World climbing fern infestation climbing into the treetops and altering fuel structure. Open areas contain frost-killed vines that are a rusty brown (white arrows), while bright green vines climb up many tree trunks (yellow arrows).
Credits: Deb Stone

How Climbing Ferns Respond to Fire

Both climbing fern species have been reported to be top killed by fire, but effects can vary between the two species. One study examined the distribution of Japanese climbing fern in disturbed and undisturbed pine savannas in Louisiana, and on the effect of fire on regrowth. They found that while growth was slightly greater in the year immediately after fire, there was no significant difference between the two years (Leichty, Carmichael, and Platt 2011). Another research project tested the effect of fine fuel loads (unaltered, reduced, or increased) on survival of Japanese climbing fern over 12 months post-fire. Six months post-treatment, increased fuel loads had slightly lower numbers of fronds while reduced fuel loads had nearly double the number of fronds; however, by 12 months no significant differences between treatments remained (Carmichael 2012).

For Old World climbing fern, one study found that fire killed 25% of plants and decreased growth rates for up to 18 months, while clipping alone had lower control rates that only lasted 6 months (Richards, Sebesta, and Taylor 2020). Another study reported that Old World climbing fern takes 12–24 months to recover following fire; but following up with annual herbicide treatment reduced cover by up to 99% after three years (Hutchinson 2010). Stocker et al. (2008) found that fire reduced the amount of herbicide needed for treatment by about one-half but did not reduce treatment time; and also that visits every two months were not more effective than those at 6-month intervals. Using fire after herbicide treatment has also been studied in freshwater marshes, with results indicating that treatment with metsulfuron methyl followed by fire resulted in increased understory plant species richness and diversity but that most of these increases came from non-native species; similar but non-significant results were observed in a hydric hammock (Hutchinson and Langeland 2010).

Many managers are interested in understanding the effects of fire on the spores of Old World climbing fern. Research examining the effect of different temperatures on the viability of Old World climbing fern spores found that all extreme temperature treatments of 100°C for either 5 or 30 seconds, or 300°C for 5 seconds did not germinate (a maximum of 0.23%), indicating that temperatures common in prescribed fire would effectively kill spores (Sebesta, Richards, and Taylor 2016).

Several biocontrol agents exist for Old World climbing fern, and recent research has shown that they can re-colonize burned climbing fern stands within five to nine months post fire, but also that Old World climbing fern could reach its original percent cover within just five months (David et al. 2020). The authors suggest that leaving refugia of climbing fern (and hence the biocontrol agents) could help with post-burn recolonization and therefore provide a reduction in overall growth rates, but this has not yet been demonstrated, and more research is needed.

Management Implications

The presence of either climbing fern species in a burn unit can have serious implications for prescribed burning because the ferns easily carry fire to the canopy or into fire-sensitive communities, which often results in drastic changes to structure and composition.

Recent research on spore viability after heat exposure has indicated that convection currents from fire may be less of a threat as a vector than most managers previously expected; however, the firebrands produced in such a situation can be a major fire hazard by spreading the fire outside of the planned burn unit (Sebesta, Richards, and Taylor 2016). Also, potential "spore rain" may be common enough that local removal of some propagules won't influence overall germination probability, especially in areas that are already heavily invaded. Fire following a treatment can increase diversity in some habitats, which could provide competition to help reduce the density of climbing fern (Hutchinson and Langeland 2010).

Fire can provide short-term reductions in growth and reduced coverage of both climbing fern species. However Japanese climbing fern recovers more quickly, resulting in a shorter window of around 6 to 9 months to get benefits from its top kill (Carmichael 2012). Studies on Old World climbing fern indicate its window for control is 12 to 18 months post-fire, although one study observed full recovery in just 5 months (Richards, Sebesta, and Taylor 2020; Hutchinson 2010; David et al. 2020).

Because increased fuel loading has been shown to provide a greater short-term decrease in Japanese climbing fern, and because heat kills Old World climbing fern spores, it's reasonable to assume that hotter fires—*when safe*—should provide greater control of existing climbing ferns (Carmichael 2012; Sebesta, Richards, and Taylor 2016). But as pointed out before, this effect is short lived and must be followed up with herbicide (within 6 to 12 months) for better long-term control. Studies have also shown that herbicide treatment after fire greatly reduces climbing fern coverage (Hutchinson 2010; Stocker et al. 2008). Finally, recent field research has indicated that a treatment return interval of less than two years is needed to reduce Old World climbing fern (Dietz et al. 2020).

Take-Home Points:

- Fire may provide short-term reductions in growth of an individual plant; in general Japanese climbing fern will recover in less than one year and Old World climbing fern generally in less than one year but possibly up to two years.
- Use hotter burns if it is safe to do so and only if ladder fuels aren't a concern (e.g., natural community type is tree-limited, or the infestation is small enough).
- Follow up fire with herbicide treatment within the first year (preferably within 6 months) to maximize impact, and continue with treatments at least every one to one and a half years.

Conclusion

These are just a few of the species that land managers in Florida must consider before applying prescribed fire. Information on additional species can be found in the Fire Effects information System database (FEIS; https://www.feis-crs.org/feis/), and general guidelines for determining when fire may help control invasive species can be found from the Southern Fire Exchange (Fill and Crandall 2019).

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