ATTACHMENT 5

Biological Information on Covered Species and Special Status Plants

Abbreviations/Acronyms

Act	Endangered Species Act of 1973
APAFR	Avon Park Air Force Range
ATV	All-terrain Vehicle
BCNP	Big Cypress National Preserve
BSHB	Bartram’s scrub-hairstreak butterfly
CFR	Code of Federal Regulations
CH	Critical Habitat
CRC	Coral Reef Commons Project
DERM	Miami-Dade Department of Environmental Resource Management
EEL	Environmentally Endangered Lands
ENP	Everglades National Park\
FBC	Florida Bat Conservancy
FDOT	Florida Department of Transportation
FNAI	Florida Natural Area Inventory
FPNWR	Florida Panther National Wildlife Refuge
FR	Federal Register
FTBG	Fairchild Tropical Botanical Garden
FWC	Florida Fish and Wildlife Conservation Commission
GA DNR	Georgia Department of Natural Resources
GTC	Gopher Tortoise Council
HCP	Habitat Conservation Plan
Indigo Snake	Eastern indigo snake
IRC	Institute for Regional Conservation
JDSP	Jonathan Dickinson State Park
Leafwing	Florida leafwing butterfly
LTDS	Line Transect Distance Sampling
MDC	Miami-Dade County
MVP	Minimal viable population
NAM	Natural Areas Management
NCSU	North Carolina State University
NFC	Natural Forest Community
NKDR	National Key Deer Refuge
NPS	National Park Service
Service	United States Fish and Wildlife Service
SOCSOUTH	United States Army Special Operations Command Center South
SWP	Seminole Wayside Park
TNC	The Nature Conservancy
UM	University of Miami
USCG	United States Coast Guard
USDA	United States Department of Agriculture
US Highway 1	US 1
WMA	Wildlife Management Area

Attachment 5 - Biological Information on Covered Species and Special Status Plants

Bartram's Scrub-Hairstreak Butterfly (endangered)

Legal Status: The U.S. Fish and Wildlife Service (Service) listed the Bartram's scrub-hairstreak butterfly (*Strymon acis bartrami*; BSHB) as an endangered species under the Endangered Species Act of 1973, as amended (Act) (87 Stat. 884; 16 U.S.C. 1531 *et seq.*) on August 12, 2014 (79 FR 47221). Critical habitat (CH) was designated at the same time on August 12, 2014 (79 FR 47179) within Miami-Dade (MDC) and Monroe Counties.

Life History/Population Dynamics: Pineland croton (*Croton linearis*, formerly referred to as *C. cascarilla*) is the only known hostplant for the BSHB (Minno and Emmel 1993; Smith *et al.* 1994). However, other related scrubhairstreak species, such as the Martial scrub-hairstreak (*Strymon martialis*), while having preference for bay cedar as a larval hostplant, have recently been documented using nickerbean (*Caesalpinia spp.*) in the Florida Keys (Daniels *et al.* 2005). Similarly, the mallow scrubhairstreak (*Strymon istapa*) has also been shown to use a variety of host sources in southern Florida. While the BSHB has been consistently documented to use pineland croton, further natural history studies may indicate the subspecies' use of additional pine rockland plants for larval development. The Bartram's scrub-hairstreak is rarely encountered more than 5 meters (m) (16.4 feet [ft]) from its host plant–pine rockland interface (Schwartz 1987; Worth *et al.* 1996; Salvato and Salvato 2008). Worth *et al.* (1996) and Salvato and Hennessey (2004) indicate that the BSHB may have limited dispersal abilities. However, while the hairstreak is often described as sedentary, the need to evade natural disturbance (fires, storms) and subsequently recolonize suggests that adult hairstreaks, perhaps as a function of age, sex, or density, are adapted for effective dispersal throughout the pine rockland habitat and associated ecosystems. Eggs are laid singly on the flowering racemes of pineland croton (Worth *et al.* 1996; Salvato and Hennessey 2004). Hennessey and Habeck (1991) observed a female oviposit three eggs over the course of 5 minutes. This long duration of oviposition likely enables females to serve as one of the major pollinating species for the host plant (Salvato 2003). First and second instars remain well camouflaged amongst the white croton flowers, while the greenish later stages occur more on the leaves.

The BSHB is most often observed visiting pineland croton flowers for nectar, but has also been observed using the flowers of other species, including: pine acacia (*Acacia pinetorum*), Spanish needles (*Bidens bipinnata*), saw palmetto (*Serenoa repens*), button sage (*Lantana involucrata*), Bloggett's swallowwort (*Cynanchum blodgettii*), Everglades Key false buttonwood (*Spermacoce terminalis*), locustberry (*Byrsonima lucida*), and starrush whitetop (*Rhynchospora colorata*) (Minno and Emmel 1993; Worth *et al.* 1996; Calhoun *et al.* 2000; Salvato and Hennessey 2004; Salvato and Salvato 2008; Anderson 2010).

The BSHB has been observed during every month on Big Pine Key and in ENP; however, the exact number of broods appears to vary sporadically from year to year (Salvato and Hennessey 2004; Salvato and Salvato 2010a). Salvato and Salvato (2010a) indicated the hairstreak is most abundant in the spring and early summer, throughout its range. However, on Big Pine Key, the subspecies is often uncommon during the fall and early winter (Salvato 1999; Anderson and Henry 2015; Salvato 2015).

In ENP, Salvato and Salvato (2010a) and Salvato (2015) have encountered as many as 6.3 adult BSHBs per hectare (ha) (2.5 per acre [ac]) annually from 1999 to 2015, based on monthly surveys in Long Pine Key. Ongoing surveys conducted by ENP staff from 2005 to present have encountered a total of approximately 24 and 30 hairstreak adults and larvae, respectively, throughout Long Pine Key (Land 2012; Sadle 2013).

Additional pine rockland habitat fragments within MDC that are known to maintain small, localized populations of pineland croton and sporadic occurrences of BSHB, based on limited survey work, include: Navy Wells Pineland Preserve, Camp Owaissa Bauer (owned and managed by MDC), and several parcels within the Richmond Pine Rocklands, including: Larry and Penny Thompson Memorial Park, Zoo Miami Preserve, Martinez Pineland Park, and U.S. Coast Guard lands in Homestead (Minno and Minno 2009; Possley 2010). Adult butterflies have also been observed within Zoo Miami (Cook 2013).

In the lower Florida Keys ongoing surveys by Salvato (2015) indicate the average number of adult BSHBs recorded annually on Big Pine Key has declined considerably, from a high of 19.3 per ha (7.7 per ac) in 1999, to a low of less than 1 per ha (0.3 per ac) in 2011, based on monthly (1999–2006) or quarterly (2007 to 2012) surveys.

Since early 2012, North Carolina State University (NCSU) personnel have collaborated with the Service on techniques to improve detection probabilities, estimate abundances, and measure vegetation characteristics associated with butterfly populations on the National Key Deer Refuge (NKDR) (Henry and Haddad 2013). These studies have documented a mean monthly count across sites ranging from 0.0 to 2.8 ± 0.33 adult BSHBs per ha (Anderson 2012b). During 2013, using these survey techniques, NKDR documented a peak abundance of 159 adults in the early summer months (Anderson 2014).

There were few or no observations of the BSHB on Big Pine Key in 2014 (n = 0) and 2015 (n = 1), by NKDR staff or Salvato (Salvato 2015, Breaux 2015).

<u>Species Critical Habitat Description</u>: CH for BSHB butterfly was designated in Miami-Dade and Monroe Counties, consisting of 4,670 ha (11,539 ac) in seven units. Five of the seven CH units are currently occupied by the BSHB (79 FR 47179).

<u>Status and Distribution</u>: The BSHB is endemic to South Florida including the lower Florida Keys. The butterfly was locally common within pine rockland habitat that once occurred within MDC and Monroe County and were less common and sporadic within croton-bearing pinelands in Collier, Palm Beach, and Broward Counties (Kimball 1965; Baggett 1982; Schwartz 1987; Hennessey and Habeck 1991; Minno and Emmel 1994; Smith *et al.* 1994; Emmel *et al.* 1995; Worth *et al.* 1996; Schwarz *et al.* 1996; Salvato and Hennessey 2004).

Populations of the BSHB have become increasingly localized as pine rockland habitat has been lost or altered through anthropogenic activity (Baggett 1982; Hennessey and Habeck 1991; Schwarz *et al.* 1996; Salvato and Hennessey 2004; Salvato and Salvato 2010a). Recent surveys and natural history studies (Salvato 1999, 2001, 2003; Salvato and Hennessey 2004; Minno and Minno 2009; Salvato and Salvato 2010a; Anderson 2012a; Land 2012; Maschinski *et al.* 2013; Cook 2013) indicate that the BSHB is extant on Big Pine Key (Monroe County), in the Long

Pine Key Region of ENP, and locally within pine rockland habitat fragments on mainland MDC, particularly those adjacent to ENP, such as Navy Wells Pineland Preserve and the Richmond Pine Rocklands. The BSHB butterfly is currently known to occur at Long Pine Key within ENP as well as several of the larger pine rockland fragments outside ENP including Navy Wells Pineland Preserve, Camp Owaissa Bauer, and the Richmond Pinelands (Larry and Penny Thompson Park, Martinez Pineland, and the Miami Metrozoo) adjacent and south of CRC (Coral Reef Commons Project).

Threats: The BSHB has experienced substantial destruction, modification, and curtailment of its habitat and range. The pine rockland community of South Florida , on which the butterfly and its' hostplant depend, is critically imperiled globally (Florida Natural Areas Inventory [FNAI] 2015). Destruction of the pine rockland habitat for economic development has reduced this habitat community by 98 percent on mainland South Florida outside of ENP (O'Brien 1998). However, any unknown extant populations of the butterfly or suitable habitat that may occur on private land or non-conservation public land, such as within the Richmond Pine Rocklands, are vulnerable to habitat loss.

Similarly, most of the pine rockland habitat within the Florida Keys have been impacted or destroyed for residential and commercial development (Hodges and Bradley 2006). All vacant land in the Florida Keys is projected to be developed, including lands currently inaccessible for development, such as islands not attached to the Overseas Highway (US Highway 1 [US 1]) (Zwick and Carr 2006). During 2006, Monroe County implemented a Habitat Conservation Plan (HCP) for Big Pine and No Name Keys. Subsequently, development on these islands has to meet the requirements of the HCP with the resulting pace of development changed accordingly. Furthermore, in order to fulfill the HCP's mitigation a requirement, Monroe County has been actively acquiring parcels of high-quality habitat for listed species and managing them for conservation, including pine rockland habitat occupied by the BSHB on Big Pine Key. However, land development pressure and habitat losses may resume when the HCP expires in 2023. If the HCP is not renewed, residential or commercial development could increase to pre-HCP levels. Consequently, remaining extant BSHB and pine rockland habitat fragments could be at risk to habitat loss and modification.

The threat of habitat destruction or modification is further exacerbated by a lack of adequate fire management (Salvato and Salvato 2010a, 2010b). Historically, lightning-induced fires were a vital component in maintaining native vegetation within the pine rockland ecosystem, including pineland croton (Loope and Dunevitz 1981; Slocum *et al.* 2003; Snyder *et al.* 2005; Salvato and Salvato 2010a). Resprouting after burns is the primary mechanism allowing for the persistence of perennial shrubs, including pineland croton, in pine habitat (Olson and Platt 1995). Without fire, successional climax from tropical pineland to hardwood hammock is rapid, and displacement of native species by invasive nonnative plants often occurs.

Cyclic and alternating treatment of burn units may benefit the BSHB throughout Long Pine Key (Salvato and Salvato 2010a). The influence of prescribed burns on the status and distribution of the BSHB and pineland croton is being evaluated. The BSHB is rarely encountered more than 5 m (16.4 ft) from its hostplant (Schwartz 1987; Worth *et al.* 1996; Salvato and Salvato 2008). Salvato and Hennessey (2004) and Salvato and Salvato (2010a) indicate that, if the BSHB is unable to disperse adequately during fire events, then only adults at the periphery of burned areas are likely to escape to adjacent pine rockland habitat. Ideally, as a result of cyclic burns and multiyear treatment intervals, the BSHBs will move from burned locations to adjacent refugia (*i.e.*, unburned areas of croton hostplant) and then back to the recently burned area in numbers equal to or greater than before the fire. Over the past decade, the BSHB appears to have benefited from prescribed burns within Long Pine Key, with population densities greater than those recorded in any previous studies (Salvato and Salvato 2010a), and this trend has continued subsequently (Land 2011, 2012, Salvato 2012).

Outside of ENP, MDC has implemented various conservation measures, such as burning in a mosaic pattern and on a small scale, during prescribed burns in order to protect the butterflies (Maguire 2010). MDC Parks and Recreation staff has burned several of their conservation lands on a fire-return interval of approximately 3 to 7 years. As a result, the BSHB has retained populations within many of these County-managed conservation lands.

Recent natural or prescribed burn activity on Big Pine Key and adjacent islands within NKDR appears to be insufficient to prevent loss of pine rockland habitat (Carlson *et al.* 1993; Bergh and Wisby 1996; O'Brien 1998; Snyder *et al.* 2005; Bradley and Saha 2009; Saha *et al.* 2011). As a result, many of the pine rockland habitat parcels, across NKDR are being compromised by succession to hardwood hammock (Bradley and Saha 2009; Saha *et al.* 2011). Pineland croton, which was historically documented from No Name and Little Pine Keys (Dickson 1955; Hennessey and Habeck 1991; Carlson *et al.* 1993), is now absent from these locations (Emmel *et al.* 1995; Salvato and Salvato 2010b).

Fire management of pine rockland habitat in NKDR is hampered by the pattern of land ownership and development; residential and commercial properties are embedded within or in close proximity to pine rockland habitat (Snyder *et al.* 2005; Anderson 2012a). As a result, hand or mechanical vegetation management may be necessary at select locations on Big Pine Key (Emmel *et al.* 1995; Minno 2009, Service 2010) to maintain or restore pine rocklands. However, mechanical treatments may not provide the same ecological benefits as fire, therefore NKDR continues to focus efforts on conducting prescribed burns where possible (Anderson 2012a).

Efforts to control salt marsh mosquitoes (*Aedes taeniorhynchus*), among others, have increased as human activity and population have increased in South Florida. To control mosquito populations, second-generation organophosphate (naled) and pyrethroid (permethrin) adulticides are applied using both aerial and ground-based methods by mosquito control districts throughout South Florida. The use of such pesticides to control mosquitoes presents a potential risk to nontarget species, including the BSHB.

The Long Pine Key region of ENP is not treated with pesticides for mosquito control. Outside of the ENP, occupied butterfly habitat within MDC remains vulnerable to the effects of adulticide applications. However, use of mosquito control pesticides within MDC pine rockland habitat areas is limited.

On Big Pine Key, Salvato (2001) and Hennessey *et al.* (1992) suggested declines in populations of the BSHB were partly attributable to mosquito control chemical applications. Specifically, Salvato (2001) noted that butterflies, such as the BSHB, were particularly vulnerable to truck applications based on their tendency to roost within low-lying vegetation (including along roadsides), an area with maximal exposure to ground-based treatments.

MDC and the Florida Key Mosquito Control District coordinate annually with the Service in order avoid or minimize any impacts to pine rockland and butterfly habitat. In addition, extensive no spray and buffer zones have been established around BSHB CH both on Big Pine Key and throughout MDC.

Eastern Indigo Snake (threatened)

Legal Status: The Service listed the eastern subspecies of indigo snake (*Drymarchon corais couperi*) as threatened under the Act on January 31, 1978 (43 FR 4026 4029). The State of Florida recognizes the eastern indigo snake as Federally-designated Threatened. There is no designated CH.

Life History/Population Dynamics: The eastern indigo snake (indigo snake) is an apex predator among snakes, eating any vertebrate it can overpower, especially other snakes (Keegan 1944; Belson 2000; Ernst and Ernst 2003, Stevenson *et al.* 2010). It is a generalized predator immune to the toxins of the venomous snakes it encounters and is only limited by its gape and ability to overpower its prey. Food items include fish, frogs, toads, snakes, lizards, turtles, turtle eggs, small alligators, birds, and small mammals (Keegan 1944; Babis 1949; Kochman 1978; Steiner *et al.* 1983).

In south-central Florida, indigo snake breeding extends from June to January, egg-laying occurs from April to July, and hatching occurs during mid-summer to early fall (Layne and Steiner 1996). Young hatch approximately 3 months after egg-laying and there is no evidence of parental care. Indigo snakes in captivity take 3 to 4 years to reach sexual maturity (Speake and Smith 1987). It is possible female indigo snakes can store sperm and delay fertilization of eggs for significant periods of time or are parthenogenetic (Carson 1945). Carson (1945) concluded that sperm storage and delayed fertilization were the most likely explanation for the fertile eggs produced by an indigo snake that he had kept in captivity for more than 4 years. However, there have been several recent reports pathogenesis in other snakes, so it is possible sperm storage may not explain Carson’s (1945) example (Moler 1998). There is no information on indigo snake lifespan in the wild, although one captive individual survived 25 years, 11 months (Shaw 1959).

Indigo snakes are active and spend a great deal of time foraging for food and searching for mates within their territories, with most activity occurring in the summer and fall (Speake and Smith 1987; Moler 1985a). Adult males have larger home ranges than adult females and juveniles; their home ranges average 554 ac, reducing to 390 ac in the summer (Moler 1985b). In contrast, a gravid female may use from 3.5 to 106 ac (Speake and Smith 1987). In Florida, home ranges for females and males range from 5 to 371 ac and 4 to 805 ac, respectively (Smith and Dyer 2003). At Archbold Biological Station, the average home range size for females was determined to be 47 ac, and overlapping male home range size determined to be 185 ac (Layne and Steiner 1996).

Due to their use of subterranean refugia and frequent long-distance dispersal, detectability of indigo snakes is low and estimates of mortality difficult (Hyslop *et al.* 2012). Consequently, the exact size and viability of the range wide population is unknown (Service 2008). However, there is no information indicating the range of indigo snake has expanded or retracted, so it's presumed the population is stable.

Status and Distribution: Historically, the indigo snake occurred throughout Florida and in the coastal plain of Georgia, Alabama, and Mississippi (Loding 1922, Haltom 1931, Carr 1940, Cook 1954, Diemer and Speake 1983, Lohoefener and Altig 1983, Moler 1985a). Most, if not all, of the remaining viable populations of the indigo snake occur in Georgia and Florida (Service 2008).

Indigo snakes are habitat generalists; they will use everything from the pristine uplands and wetlands to highly disturbed residential areas (Bolt 2006). Even though the action area is in South Florida where gopher tortoise burrows are not widespread, indigo snakes will use a variety of den sites, such as other holes or burrows, tree stumps, root masses, and piles of yard or construction debris (Bolt 2006).

Reptile surveys were conducted in pine rockland habitat in six county parks in MDC in 1996-97 and in 2001, but no indigo snakes were detected (Enge *et al.* 2004). Enge *et al.* (2004) stated these parks may not have supported the prey base needed for large snakes. Indigo snakes were reported from pineland habitat in Long Pine Key, ENP (Dalrymple *et al.* 1991). Staff at the Miami Metrozoo have observed one indigo snake in the property's undeveloped pinelands within the last 10 years (Conners 2002, as cited in Enge *et al.* 2004).

Threats: Throughout the indigo snake's range expanding urban areas are creating barriers to the dispersal of individuals and gene flow between populations, and habitat loss and degradation are a threat to the species (Lawler 1977, Moler 1985b). In northern areas of its range in Georgia and peninsular Florida the species is impacted by a decline in longleaf pine forests, gopher tortoises, and gopher tortoise habitat (Van Lear *et al.* 2005). In central and southern Florida the indigo snake is less dependent on any one habitat type, but does avoid developed areas (Lawler 1977, Moler 1985a, Hyslop 2007). Throughout Florida, developed areas are expanding rapidly with population growth at the expense of wildlife habitat (Cerulean 2008).

At the time of listing, other threats to the indigo snake included commercial collection for the pet trade and mortality during the gassing of gopher tortoise burrows by individuals attempting to drive rattlesnakes out for collection (43 FR 4026 4029). Since their listing additional potential threats to the species have expanded to include disease, road mortality, kills of indigo snakes by land owners and pets, and ATV use in gopher tortoise habitat (Service 2008).

Florida Bonneted Bat (endangered)

Legal Status: The Service proposed to list the Florida bonneted bat (*Eumops floridanus*) under the Act on October 4, 2012 (Service 2012a). The final listing determination published on October 2, 2013, and became effective November 1, 2013 (Service 2013b)(78 FR 61003 61043). No CH for this species has been designated.

Life History/Population Dynamics: Relatively little is known about the Florida bonneted bat's life history and its lifespan is not known. Based upon the work of Wilkinson and South (2002), Gore *et al.* (2010) inferred a lifespan of 10 to 20 years for the Florida bonneted bat, with an average generation time of 5 to 10 years.

The Florida bonneted bat has a fairly extensive breeding season during summer months (Timm and Genoways 2004). The maternity season for most bat species in Florida occurs from mid-April through mid-August (Marks and Marks 2008a). The Florida bonneted bat is a subtropical species, and limited data suggest the species may be aseasonally polyestrous (having more than one period of estrous in a year, although no females have been documented as pregnant multiple times in a given year) (Timm and Genoways 2004; FBC [Florida Bat Conservancy] 2005; Ober *et al*. 2016). Recent studies at Babcock-Webb Wildlife Management Area (WMA) in 2014and 2015 have helped document pregnant bonneted bats in April and August, with males displaying reproductively active status (open gular glands) in April, Auguest and December (Ober *et al*. 2016). Pups were also observed within a known natural roost at Avon Park Air Force Range (APAFR) in mid-October (Scofield 2014; Halupa 2014b). The full extent of the maternity season is not well understood, but is a time of particular sensitivity, with increased energy demands and risks as females leave young in roosts while making multiple foraging excursions to support lactation (Kurta *et al.* 1989; Kurta *et al.* 1990; Kunz *et al.* 1995; Marks and Marks 2008a; Ober 2014a). Exploitation of insects in patches that yield high energy returns for pregnancy and lactation is important (Kunz *et al.* 1995). Reduced insect populations in urban areas may make it difficult for females to successfully raise offspring to maturity (Kurta *et al.* 1990; Kurta and Teramino 1992). Information on reproduction and demography is sparse. The Florida bonneted bat has low fecundity; litter size is one (FBC 2005; Timm and Arroyo-Cabrales 2008).

At present, only three active, natural roost sites are known, and only limited information on historical sites is available. Based upon limited information, the species roosts singly, or in colonies consisting of a male and several females (Belwood 1992). G.T. Hubbell believed individuals in Miami roosted singly (Belwood 1992). However, Belwood (1981) suggested a colony, consisting of seven females and one male using a longleaf pine cavity as a roost site in Punta Gorda, was a harem group, based on its sex ratio. Belwood (1981; 1992) suggested this behavior has been recorded in a few bat species and such social groupings may be facilitated by roosting in tree cavities, which can be defended from other males (Morrison 1979). Few details are available for the composition of the currently known active natural roosts. At APAFR, approximately 22 bats emerged from the roost in October 2014, with several others including young remaining at the roost after emergence (Halupa 2014b). At Florida Panther National Wildlife Refuge (FPNWR), 12 bats emerged from the roost tree on 2 consecutive nights in July 2015, with others, possibly pups, remaining in the roost after emergence (Braun de Torrez 2015). At Big Cypress National Preserve (BCNP), 11 bats emerged from a natural roost in December 2015 (Arwood 2015).

Information on roosting habits from artificial structures is also limited. The Florida bonneted bat colony using bat houses on private property in Lee County consisted of 8 to 25 individuals, including one albino (Trokey 2006a, 2006b; 2008a, 2008b; 2012). Sex ratio is not known. Some movement between the houses has been observed (Trokey 2006a). Periodic simultaneous counts taken at bat houses at Babcock-Webb WMA and recent research suggest that use fluctuates among the seven roost sites (artificial structures) (Ober et al. 2016). Simultaneous counts taken at bat houses at emergence from 2012 to 2014, indicated that Florida bonneted bat roosts are generally small, occupied by 1-14 individuals, except for one location which generally supports 25-44 individuals among two houses (Myers 2013, 2014).

The Florida bonneted bat is active year-round and does not have periods of hibernation or torpor. The species is not migratory, but there might have been seasonal shifts in roosting sites (Timm and Genoways 2004). Belwood (1992) reported that, prior to 1967; G.T. Hubbell routinely obtained several individuals per year collected during the winter from people's houses. Precise foraging and roosting habits and long-term requirements are unknown (Belwood 1992). Active year-round, the species is likely dependent upon a constant and sufficient food supply, consisting of insects, to maintain its generally high metabolism. Based upon limited information, Florida bonneted bats feed on flying insects of the following orders: Coleoptera (beetles), Diptera (true flies), Hemiptera (true bugs), and Lepidoptera (moths) (Belwood 1981; Belwood 1992; FBC 2005). An analysis of bat guano (droppings) from the colony using the pine flatwoods in Punta Gorda indicated the sample (by volume) contained coleopterans (55 percent), dipterans (15 percent), and hemipterans (10 percent) (Belwood 1981; Belwood 1992). More recent analyses of bat guano collected from occupied bat houses at Babcock-Webb WMA indicated that the samples contained high percentages of Lepidoptera and Coleoptera (Marks 2013).

Molossids, in general, seem adapted to fast flight in open areas (Vaughan 1966). Various morphological characteristics (*e.g.*, narrow wings, high wing-aspect ratios [ratio of wing length to its breadth]) make *Eumops* well-adapted for efficient, rapid, and prolonged flight in open areas (Findley *et al.* 1972; Freeman 1981; Norberg and Rayner 1987; Vaughan 1959, as cited in Best *et al.* 1997). Barbour and Davis (1969) noted that the species flies faster than smaller bats, but cannot maneuver as well in small spaces. Belwood (1992) stated *E. glaucinus* is "capable of long, straight, and sustained flight," which should allow individuals to travel large distances. Norberg and Rayner (1987) attributed long distance flights of Brazilian free-tailed bats to their high wing-aspect ratios, with that species capable of traveling 65 kilometers (km) (40 miles [mi]) from its roosting site to its foraging areas (Barbour and Davis 1969). Like other molossids, the Florida bonneted bat's morphological characteristics make it capable of dispersing large distances and generally adapted for low cost, swift, long distance travel from roost site to foraging areas (Norberg and Rayner 1987; Gillies 2012; Ober 2012). Given this, it seems likely that foraging areas may be located fairly long distances from roost sites (Ober 2012). Nonetheless, average foraging distances for the Florida bonneted bat are not known (Marks 2012). Although the species can fly long distances, it likely does not travel farther than necessary to acquire food needed for survival (Marks 2012).

Bonneted bats are "fast hawking" bats that rely on speed and agility to catch target insects in the absence of background clutter, such as dense vegetation (Simmons *et al.* 1979; Belwood 1992; Best *et al.* 1997). Foraging in open spaces, these bats use echolocation to detect prey at relatively long range, roughly 3 to 5 m (10 to 16 ft) (Belwood 1992). Based upon information from G.T. Hubbell, Belwood (1992) indicated that individuals leave roosts to forage after dark, seldom occur below 10 m (33 ft) in the air, and produce loud, audible calls when flying; calls are easily recognized by some humans (Belwood 1992; Best *et al.* 1997; Marks and Marks 2008a).

Relatively little is known of the ecology of the Florida bonneted bat, and long-term habitat requirements are poorly understood (Robson 1989; Robson *et al.* 1989; Belwood 1992; Timm and Genoways 2004). Habitat for the Florida bonneted bat mainly consists of foraging areas and roosting sites, including artificial structures. At present, only three active, natural roost sites are known, and only limited information on historical sites is available.

More recently, radio-telemetry studies in natural environments are being used as part of studies to close data gaps on the Florida bonneted bat to better understand the species and its relationship and response to fire (Ober and McCleery 2012; Bailey 2013; Ober 2013). Results from a limited pilot study testing the tolerance and effectiveness of transmitters on three female Florida bonneted bats at Babcock-Webb WMA in December 2014 indicated that individuals foraged several miles from their roosts (Ober 2014b; Braun de Torrez 2014).

Little information exists on historical population levels. The Florida bonneted bat was considered common in the Miami-Coral Gables area because of regular collection of specimens from 1951 to 1965 (Robson 1989; Belwood 1992). Jennings (1958) indicated the species was not abundant; noting a total of 20 individuals had been taken from 1936 to 1958. Prior to 1967, G.T. Hubbell regularly heard loud, distinctive calls at night as the bats foraged above buildings in the Miami area (Timm 2012a), and he routinely obtained several individuals per year that were collected from people's houses (Belwood 1992). Barbour and Davis (1969) indicated that, on average, about two individuals per year were brought to the Crandon Park Zoo in Miami, due to injuries, but no time period was specified.

Unpublished data from a survey of 100 pest control companies in 1982 on the southeastern coast of Florida showed that requests to remove "nuisance" bats from this area all but ceased beginning in the 1960s (Belwood 1992), indicating a sharp decline in bats in general. Timm and Genoways (2004) found only three records of Florida bonneted bats in the greater Miami area after 1965. The colony found near Punta Gorda in 1979 appeared to be the only recorded occurrence since 1967 (Belwood 1981). A 6-week field trip in 1980 to locate other occurrences was unsuccessful and led to the belief this species was ''probably extinct in Florida'' (Belwood 1992). No new evidence of this species was found from 1979 until 1988 when Robson *et al.* (1989) found a pregnant female in Coral Gables (Robson 1989).

Timm and Genoways (2004) surmised the Florida bonneted bat may have been uncommon for several decades, based upon the work of previous researchers (Barbour 1945, as cited in Timm and Genoways 2004; Jennings 1958; Layne 1974), who noted the scarcity of bats in southern Florida. Owre (1978) observed fewer than a dozen individuals in roughly 25 years and noted few mammalogists had success in finding the species. Robson (1989) indicated the decline of specimens and sightings in the mid-1960s is reflected in the museum record and noted the 1950s and 1960s was a period of rapid growth in the Miami area. Robson (1989) suggested the resulting disturbance and destruction of native habitat may have flushed a large number of specimens out of established roosts, resulting in a high collection rate. A status survey conducted in 1989, encompassing 25 sites within natural areas within a nine county area, found no new evidence of this species (Robson 1989).

In his independent review of the Florida Fish and Wildlife Conservation Commission's (FWC) Biological Status Report, Ted Fleming, Emeritus Professor of biology at University of Miami, noted anecdotal evidence from the 1950s and 1960s suggests this species was more common along Florida's southeast coast compared with the present (FWC 2011b). Fleming stated, "There can be no doubt that *E. floridanus* is an uncommon bat throughout its very small range. Its audible echolocation calls are distinctive and easily recognized, making it relatively easy to survey in the field" (FWC 2011b). He also stated he does not doubt the total State population numbers "in the hundreds or low thousands" (FWC 2011b).

Similarly, in response to a request for information as part of the Service's annual Candidate Notice of Review, Robert Timm (2012), Curator of Mammals at Department of Ecology and Evolutionary Biology and Biodiversity Institute at the University of Kansas, indicated that numbers are low, in his view, as documented by survey attempts. "*Eumops* are very obvious bats where they occur because of their large size and distinctive calls. Given the efforts to locate them throughout southern Florida, if they were there in any significant numbers, they would have been located" (Timm 2012).

Results of the 2006-2007 range-wide survey suggested that the Florida bonneted bat is a rare species with limited range and low abundance (Marks and Marks 2008a). Based upon results of both the range-wide study and survey of select public lands, the species was found at 12 locations (Marks and Marks 2008b), but the number and status of the bat at each location are unknown. Based upon the small number of locations where calls were recorded, the low numbers of calls recorded at each location, and the fact that the species forms small colonies, Marks and Marks (2008a) stated that it is possible that the entire population of Florida bonneted bats may number less than a few hundred individuals.

Results of the 2010 to 2012 surveys and additional surveys by other researchers identified new occurrences within the established range (*i.e.*, within Miami area, areas of ENP and BCNP) (Snow 2011a, 2011b, 2012; Arwood 2012; Marks and Marks 2012), however, not in sufficient numbers to alter previous population estimates. In their 2012 report on the status of the species, Marks and Marks (2012) provided an updated estimation of population size, based upon 120 nights of surveys at 96 locations within peninsular Florida, results of other known surveys, and personal communications with others involved in Florida bonneted bat work. Based upon an average colony size of 11 and an estimated 26 colonies within the species' range, researchers estimated the total Florida bonneted bat population at 286 bats at that time (Marks and Marks 2012). Since that time, the discovery of the three natural roost sites and the continuation of additional research at occupied bat houses has provided opportunities for more quantitative emergence counts. This suggests that previous estimates of hundreds to a few thousand might be more representative of population size.

In summary, we cannot accurately estimate population size at this time. This is in part because so few roosts are known, roost switching can occur, emergence counts are not conducted simultaneously (or even at the same time of year), and precise counts are difficult to obtain due to environmental conditions and the propensity for some individuals to remain within roosts during counts.

Status and Distribution: Based upon available data and information, the Florida bonneted bat occurs within a restricted range and in apparent low abundance (Marks and Marks 2008a; 2012; Timm and Arroyo-Cabrales 2008; FWC 2011a; FWC 2011b; Timm 2012b). Actual population size is not known, and no population viability analyses are available (FWC 2011a; 2013; Bohn, 2012).However, population size is thought to be less than that needed for optimum viability (Timm and Arroyo-Cabrales 2008; Bohn 2012).As part of their evaluation of listing criteria for the species, Gore *et al.* (2010) found the extent of occurrence appears to have declined on the east coast, but trends on the west coast could not be inferred due to limited information.

Records indicating historical range are limited. Morgan (1991) indicated that *E. glaucinus* had been identified from four late Pleistocene (approximately 11,700 years ago) and Holocene (time period beginning 10,000 years ago) fossil sites in the southern half of the Florida peninsula. Late Pleistocene remains are known from Melbourne, Brevard County, and Monkey Jungle Hammock in MDC (Allen 1932; Martin 1977, as cited in Belwood 1981 and Timm and Genoways 2004; Morgan 1991). Holocene remains are known from Vero Beach, Indian River County (Ray 1958; Martin 1977; and Morgan 1985, 2002, as cited in Timm and Genoways 2004; Morgan 1991), and also Monkey Jungle Hammock (Morgan1991). The largest fossil sample (9 specimens) was reported from the Holocene stratum at Vero Beach (Morgan 1985, as cited in Morgan 1991). The fossil records from Brevard County and Indian River County are considerably farther north than where living individuals have typically been recorded (Timm and Genoways 2004; Marks and Marks 2008b).

Most of the historical records and sightings for this species are several decades old from the cities of Coral Gables and Miami in extreme southeastern Florida, where the species was once believed to be common (Belwood 1992; Timm and Genoways 2004; Timm and Arroyo-Cabrales 2008). G.T. Hubbell also reported a female with young from Fort Lauderdale in Broward County; all of his sightings of Florida bonneted bats were near human dwellings (Belwood 1992). Prior to 1967, G.T. Hubbell regularly heard loud, distinctive calls at night as the bats foraged above buildings, and he routinely obtained several individuals per year that were collected during the winter months from people's houses (Belwood 1992). Other early literature also mentioned Fort Lauderdale as an area where the species occurred (Barbour and Davis1969; Belwood 1992). However, in their comprehensive review, none of the specimens examined by Timm and Genoways (2004) were from Broward County. Belwood (1981) found a colony in Punta Gorda; however, the longleaf pine in which the bats roosted was felled during highway construction. Recent specimens are only known from extreme southern and southwestern Florida, including MDC on the east coast and Charlotte, Collier, and Lee Counties on the Gulf coast (Timm and Genoways 2004).

Endemic to Florida, the Florida bonneted bat has one of the most restricted distributions of any species of bat in the New World (Belwood 1992; Timm and Genoways 2004). Although numerous acoustical surveys for the Florida bonneted bat have been conducted in the past decade by various parties, the best scientific information indicates that the species exists only within a very restricted range, confined to South Florida (Timm and Genoways 2004; Marks and Marks 2008a, 2012).

Based upon available information, the Florida bonneted bat appears to be restricted to south, southwest, and south-central Florida. The core range may primarily consist of habitat within

Charlotte, Lee, Collier, Monroe, and Miami-Dade Counties. Recent data also indicate use of portions of Okeechobee, Polk, DeSoto, Hendry, and Broward Counties and possible use of areas within Glades and Highlands Counties.

The Florida bonneted bat appears to be restricted to south and southwest Florida. The core range may primarily consist of habitat within Charlotte, Lee, Collier, Monroe, and Miami-Dade Counties. Recent data also confirm use of portions of Okeechobee and Polk counties and possible use of areas within Glades County. Within MDC, the Florida bonneted bat has been documented at the Richmond Pinelands (Larry and Penny Thompson Memorial Park and the Miami Metrozoo) and within the proposed action area. The species has been documented at the Miami Metrozoo within an urban public park (Ridgley 2012; Marks and Marks 2012). A dead specimen was found on the Miami Metrozoo grounds at the Asian Elephant barn in 2004 (Marks and Marks 2008a). A single call was recorded by Florida Bat Conservancy outside the same enclosure in September 2011 (Ridgley 2012; Marks and Marks 2012). Florida bonneted bat calls have also been recorded at Larry and Penny Thompson Memorial Park (Ridgley 2013).The population is estimated to be in hundreds to low thousands (Marks and Marks 2008a; FWC 2011a, 2011b).

Threats: Habitat loss and alteration in forested and urban areas are substantial threats to the Florida bonneted bat (Belwood 1992). In natural areas, this species may be impacted when forests are converted to other uses or when old trees with cavities are removed (Belwood 1992). In urban settings, this species may be impacted when buildings with suitable roosts are demolished (Robson 1989) or when structures are modified to exclude bats. Small population size, restricted range, low fecundity, and few and isolated occurrences are considerable on-going threats. This species is also vulnerable to prolonged extreme cold weather events. The cold spell experienced in Florida in early 2010 may have caused a decline in the Florida bonneted bat population. A colony in Lee County once included approximately 20 to 24 individuals in two houses (Trokey 2008a, 2008b), but only 9 remained after the prolonged cold temperatures in early 2010 (Trokey 2010a, 2010b).

Florida Leafwing Butterfly (endangered)

Legal Status: The Service listed the Florida leafwing butterfly (*Anaea troglodyta floridalis*) as an endangered species under the Act on August 12, 2014 (79 FR 47221). CH was designated at the same time on August 12, 2014 (79 FR 47179) within Miami-Dade and Monroe Counties.

Life History/Population Dynamics: Numerous authors have observed and documented the behavior and natural history of the Florida leafwing (Matteson 1930; Lenczewski 1980; Pyle 1981; Baggett 1982; Opler and Krizek 1984; Schwartz 1987; Hennessey and Habeck 1991; Smith *et al.* 1994; Worth *et al.* 1996; Salvato 1999; Salvato and Hennessey 2003; Salvato and Salvato 2008, 2010a). Adults are rapid, wary fliers and have strong flight abilities and are able to disperse over large areas. The Florida leafwing is multivoltine (*i.e.*, produces multiple generations per year), with an entire life cycle of about 2 to 3 months (Hennessey and Habeck 1991) and maintains continuous broods throughout the year (Salvato 1999).

The immature stages of the Florida leafwing feed, exclusively on pineland croton (*Euphorbiaceae*), for larval development. Eggs are spherical and light cream-yellow in color

(Worth *et al.* 1996). Females lay eggs singly on both the upper and lower surface of the host (croton plant) leaves, normally on developing racemes (flowers) (Baggett 1982; Hennessey and Habeck 1991; Worth *et al.* 1996; Salvato 1999; Minno *et al.* 2005). Worth *et al.* (1996) and Salvato (1999) visually estimated that females may fly more than 30 meters (m) (98 ft) in search of a suitable host plant.

In the Everglades, ongoing surveys conducted by Salvato (2015) from 2009 to 2015 have recorded an average abundance of 1 to 3 adult Florida leafwings per ha (1 per ac), within Long Pine Key. In addition, surveys conducted by ENP staff from 2005 to present have encountered a total of approximately 34 and 216 leafwing adults and larvae, respectively, throughout Long Pine Key (Land 2012; Sadle 2013a).

No leafwings have been documented on Big Pine Key in the Florida Keys since 2006 (Salvato and Salvato 2010b). On the mainland, Salvato (2012) has found that the extant leafwing population within ENP is maintained at several hundred individuals or fewer, although numbers vary greatly depending upon season and other factors. However, Minno (2009) estimated the extant leafwing population size at less than 100 at any given period.

Ongoing natural history studies of the leafwing by Salvato and Salvato (Salvato 2015) and Sadle (2013a) designed to evaluate mortality factors amongst the butterfly's immature stages have identified a suite of predators, parasitoids, and pathogens that may substantially influence annual variability.

The Florida leafwing is endemic to South Florida including the lower Florida Keys. The butterfly was locally common within pine rockland habitat that once occurred within Miami-Dade and Monroe Counties and less common and sporadic within croton-bearing pinelands in Collier, Martin, Palm Beach, and Broward Counties (Kimball 1965; Baggett 1982; Minno and Emmel 1994; Smith *et al.* 1994; Salvato 1999; Salvato and Hennessey 2003).

Populations of Florida leafwing have become increasingly localized as pine rockland habitat has been lost or altered through anthropogenic activity (Lenczewski 1980; Baggett 1982; Hennessey and Habeck 1991; Schwarz *et al.* 1996; Salvato and Hennessey 2003; Salvato and Salvato 2010a).

In the lower Florida Keys (Monroe County), the Florida leafwing has not been reported from Big Pine Key since 2006 (Minno and Minno 2009; Salvato and Salvato 2010b).

At present the Florida leafwing is extant only within Long Pine Key (Salvato and Salvato 2010a). The subspecies is sporadically encountered (as strays) within the pine rockland fragments adjacent to ENP (Salvato and Salvato 2010a). However, breeding Florida leafwing populations have not been documented in pine rockland fragments adjacent to ENP for the past 25 years.

The Florida leafwing butterfly is currently known to occur only at Long Pine Key within ENP in MDC (Salvato and Hennessey 2003; Salvato and Salvato 2008). Until recently the species was also known to occur in several pine rockland fragments outside of ENP, as well the lower Florida Keys (Salvato and Hennessey 2003). However, Salvato and Hennessey (2003) and Salvato (2008) have generally failed to observe the Florida leafwing butterfly in these or other relict (surviving remnant) pine rockland areas outside ENP. During June 2007, one adult leafwing was observed within Navy Wells, relatively close to the proposed action area (Salvato and Salvato 2008); however, no evidence of larval activity was encountered suggesting this observation may have been a stray occurrence. In addition, no leafwing have been recorded outside of ENP since that time. Breeding Florida leafwing butterfly populations have not been documented in pine rockland fragments adjacent to ENP for the past 25 years, 20 mi southwest of the CRC.

<u>Species Critical Habitat Description</u>: CH for the Florida leafwing butterfly was designated in Miami-Dade and Monroe Counties, consisting of 4,273 ha (10,561 ac) in four units. One of the four CH units is currently occupied by the leafwing (79 FR 47179).

<u>Status and Distribution</u>: The Florida leafwing is endemic to South Florida including the lower Florida Keys. The butterfly was locally common within pine rockland habitat that once occurred within Miami-Dade and Monroe Counties and less common and sporadic within croton-bearing pinelands in Collier, Martin, Palm Beach, and Broward Counties (Kimball 1965; Baggett 1982; Minno and Emmel 1994; Smith *et al.* 1994; Salvato 1999; Salvato and Hennessey 2003). Populations of Florida leafwing have become increasingly localized as pine rockland habitat has been lost or altered through anthropogenic activity (Lenczewski 1980; Baggett 1982; Hennessey and Habeck 1991; Schwarz *et al.* 1996; Salvato and Hennessey 2003; Salvato and Salvato 2010a). In the lower Florida Keys (Monroe County), the Florida leafwing has not been reported from Big Pine Key since 2006 (Minno and Minno 2009; Salvato and Salvato 2010b). At present the Florida leafwing is extant only within Long Pine Key (Salvato and Salvato 2010a). The subspecies is sporadically encountered (as strays) within the pine rockland fragments adjacent to ENP (Salvato and Salvato 2010a). However, breeding Florida leafwing populations have not been documented in pine rockland fragments adjacent to ENP for the past 25 years.

<u>Threats</u>: The Florida leafwing has been extirpated (no longer in existence) from nearly 96 percent of its historical range; the only known extant population occurs within ENP in MDC. Threats of habitat loss and fragmentation, including climatic change, poaching, parasitism and predation, and small population size, restricted range, and influence of chemical pesticides used for mosquito control, still exist for the only remaining population. Because there is only one small extant population of this butterfly, and limited law enforcement, collection has and continues to be a significant threat to this butterfly. Existing regulatory mechanisms are inadequate to reduce these threats. The leafwing may be impacted when pine rocklands are converted to other uses or when lack of fire causes the conversion to habitats that are unsuitable for this butterfly. Because the remaining population is isolated and the butterfly has a limited ability to recolonize historically occupied habitats that are now highly fragmented, it is vulnerable to natural or human-caused changes in its habitats. As a result, impacts from increasing threats, singly or in combination, are likely to result in the extinction of the butterfly as there is no redundancy of populations (Service 2013c).

Gopher Tortoise (candidate)

<u>Legal Status</u>: The Service announced candidate status for the eastern subspecies of gopher tortoise (*Gopherus Polyphemus*) on July 27, 2011. The State of Florida recognizes the gopher tortoise as Threatened. There is no designated CH.

<u>Life History/Population Dynamics</u>: The gopher tortoise is the only tortoise (family Testudinidae) east of the Mississippi River; one of five species in the genus in North America. It is larger than any of the other terrestrial Gopherus turtles in this region, with a domed, dark brown to grayish-black carapace (upper shell), and is typically 10 to 12 in (inches) (25.4 to 30.5 centimeters [cm]) long (Ashton and Ashton 2008). The plastron (lower shell) is yellowish and hingeless. A fossorial species, its hind feet are elephantine or stumpy, and the forelimbs are shovel-like, with claws used for digging. In comparison to females, males are generally smaller; with a larger gland under the chin, a longer gular (throat) projection, and more concave (curved in) plastron. Hatchlings are up to 2 in (5 cm) in length, with a somewhat soft, yellow-orange shell.

Gopher tortoises require well-drained, sandy soils for burrowing and nest construction, an abundance of herbaceous ground cover for food, and a generally open canopy that allows sunlight to reach the forest floor (Landers 1980; Auffenberg and Franz 1982). Longleaf pine and oak uplands, xeric hammock, xeric Florida scrub, maritime scrub, and ruderal (disturbed) habitat most often provide the conditions necessary to support gopher tortoises (Auffenberg and Franz 1982). Ruderal (*i.e.*, disturbed or atypical) habitats include roadsides and utility rights-of-way, grove/forest edges, fencerows, and clearing edges. In the western range, soils contain more silt, and xeric (dry) conditions are less common west of the Florida panhandle (Craul *et al.* 2005). Ground cover in this Coastal Plains area can be separated into two general regions with the division in the central part of southern Alabama and northwest Florida. To the west, bluestem (*Andropogon and Schizachyrium spp*.) and panicum (*Panicum spp*.) grasses predominate; to the east, wiregrass (*Aristida stricta*) is most common (Boyer 1990). However, gopher tortoises do not necessarily respond to specific plants but rather the physical characteristics and structure of habitat (Diemer 1986). Historic gopher tortoise habitats were open pine forests, savannahs, and xeric grasslands that covered the coastal plain from Mexico and Texas to Florida.

Gopher tortoises have a well-defined activity range where all feeding and reproduction take place and that is limited by the amount of herbaceous ground cover (Auffenberg and Iverson 1979). Tortoises are herbivores eating mainly grasses, plants, fallen flowers, fruits, and leaves. Gopher tortoises prefer grassy, open-canopy microhabitats (Boglioli *et al.* 2000), and their population density directly relates to the density of herbaceous biomass (Auffenberg and Iverson 1979; Landers and Speake 1980; Wright 1982;) and a lack of canopy (Breininger *et al.* 1994; Boglioli *et al.* 2000). Grasses and grass–like plants are important in gopher tortoise diets (Auffenberg and Iverson 1979; Landers 1980; Garner and Landers 1981; Wright 1982; Macdonald and Mushinsky 1988; Mushinsky *et al.* 2006; Birkhead *et al.* 2005).
A lack of vegetative diversity may negatively impact the long-term sustainability of gopher tortoise populations (Ashton and Ashton 2008).

Gopher tortoises require a sparse canopy and litter-free ground not only for feeding, but also for nesting (Landers and Speake 1980). In Florida, McCoy and Mushinsky (1995) found that the number of active burrows per tortoise was lower where canopy cover was high. Females require almost full sunlight for nesting (Landers and Buckner 1981) because eggs are often laid in the burrow apron or other sunny spot and require the warmth of the sun for appropriate incubation (Landers and Speake 1980). At one site in southwest Georgia, Boglioli *et al.* (2000) found most tortoises in areas with 30 percent or less canopy cover. Diemer (1992) found that ecotones (areas on the edges of landscapes) created by clearing were also favored by tortoises in North

Florida. When canopies become too dense, usually due to fire suppression, tortoises tend to move into ruderal habitats such as roadsides and rights-of-way with more herbaceous ground cover, lower tree cover, and significant sun exposure (Garner and Landers 1981; McCoy *et al.* 1993; Baskaran *et al.* 2006). In Georgia, Hermann *et al.* (2002) found that open pine areas (*e.g.*, pine forests with canopies that allow light to penetrate to the forest floor) were more likely to have burrows, support higher burrow densities, and have more burrows used by large, adult tortoises than closed-canopy forests. Historically, open-canopied pine forests were maintained by frequent, lightning-generated fires, with peak lightning ignition occurring in late spring to early summer (Knapp *et al.* 2009). The burrows of a gopher tortoise are the habitat and center of normal feeding, breeding, and sheltering activity. Gopher tortoises can excavate many burrows over their lifetime, and often use several each year. Burrows typically extend 15-25 ft (4.6 to 7.6 m), with a record burrow measuring 67 ft (20.5 m; Ashton and Ashton 2008), can be up to 12 ft (3.7 m) deep, and provide shelter from predators, winter cold, fire, and summer heat. Tortoises spend most of their time within burrows and emerge during the day to bask in sunlight, to feed, and reproduce. Tortoises breed from March through October (McRae *et al.* 1981; Wright 1982; Eubanks *et al.* 2002), but females do not reproduce every year (estimated at 80 to 85 percent; Smith *et al.* 1997). Females excavate a shallow nest to lay and bury eggs, typically between early May and late June, and usually in the apron of soil at the mouth of the burrow. Range-wide, average clutch size varies from about 4 to 10 eggs per clutch, and incubation lasts 85 to 100 days.

Home range size and movements increase with age and body size, and home range area tends to vary with habitat quality, becoming larger in areas of poor habitat (Auffenberg and Iverson 1979, p. 558). Males typically have larger home ranges than females. Mean home ranges of individual tortoises in Alabama, Florida, and Georgia have varied from 1.3-5.2 ac (3.2-2.2 ha) for males and 0.2-2.5 ac (0.09-1.0 ha) for females (McRae *et al.* 1981; Diemer 1992; Tuma 1996; Eubanks *et al.* 2002).

The current range for the eastern (candidate) population of the gopher tortoise aligns with the historic range which includes Alabama (east of the Tombigbee and Mobile Rivers), Florida, Georgia, and South Carolina. The core of the current distribution of the gopher tortoise in the eastern portion of its range includes central and north Florida and southern Georgia.

Due to discrepancies in historical data collection (described below), we have recommended that surveys be performed using Line Transect Distance Sampling (LTDS) when possible and applicable, as this method is the most statistically reliable to assess accurate measurements of tortoise populations (Smith *et al.* 2009). Surveys using this methodology are currently ongoing across all states within the candidate range of the tortoise and are providing more comprehensive data on the status of the species. For instance, the State of Georgia has the most comprehensive gopher tortoise survey effort to-date, both on public and private lands. Georgia Department of Natural Resources (GA DNR) has estimated that surveys have been contracted and/or completed on at least 52 individual properties statewide, and estimates at least 120 tortoise populations that meet the size and demographic requirements of a viable population.

<u>Status and Distribution</u>: A wide variety of information is available on the number and density of gopher tortoises and their burrows throughout their range. These data are the result of numerous surveys/censuses using a variety of methodologies ranging from one-time censuses to repeated

surveys over several decades. In the past, the diversity of data has posed a challenge when trying to evaluate the status of the species from a landscape perspective. For example, in geographic areas where we had more data, we had higher confidence in drawing conclusions about the status of those populations. In other areas, where there is little or no data, our confidence in assessing the status of tortoises is lower. Because of disparities in the type of data collected, methodologies in collecting data, and differences in the scope of studies, it is not possible to simply combine datasets to evaluate the status of the gopher tortoise. In order to address the issue of incompatible data from various survey methodologies, we have recommended that surveys be performed using LTDS when possible and applicable, as this method is the most statistically reliable to assess accurate measurements of tortoise populations (Smith *et al.* 2009).

The gopher tortoise is more widespread and abundant in parts of the eastern portion of its range, in particular southern Georgia and central and northern Florida; these areas have been designated as the "central" portion of the tortoise's geographic extent previously in the literature (Tuberville *et al.* 2009). Although most state-wide estimates of gopher tortoise abundance have not been calculated directly from survey results, some estimates have been made based on available habitat and extrapolation of existing population data. These estimates include approximately 785,000 in Florida (FWC 2012); 250,000 in Georgia (Elliott 2013); and 30,000 to 130,000 in Alabama (Guyer *et al.* 2011). Many surveys indicate that tortoise populations often occur in fragmented and degraded habitat, and densities of individuals are low within populations; however, there are also many populations of tortoises in the eastern portion of the range that appear to be sufficiently large enough to persist long-term if proper management and protections are secured (Service 2011a).

Presently there is an effort to define the characteristics of a viable gopher tortoise population, and identify the locations of those populations in order to assist with developing conservation priorities. This effort will also assist with determining population targets across the range; that is, how many tortoises (and populations) might each state have had historically, and where were they concentrated. All states in the candidate range of the tortoise are evaluating their current populations, in order to have a more thorough understanding of: the status of the species; areas with the highest potential for expansion or connection between populations; areas where recruitment of young tortoises seems to be highest; populations necessary to maintain the genetic viability of the species; and identifying populations most susceptible to fragmentation or pressure from urbanization. The Gopher Tortoise Council (GTC) has prepared a document detailing the characteristics of a minimum viable population (MVP), as well as the definitions of smaller support populations that are not presently viable. An MVP has been described as a demographically stable population with at least 250 adult tortoises, at a density of no less than 0.4 tortoises/hectare (approximately one tortoise for every 6 ac), on at least 100 ha (250 ac) of well-managed, suitable habitat (GTC 2014). These populations should have a sex ratio approaching 1:1, and have evidence of active burrows representing all age classes. However, a full assessment of viability must also include determinations that appropriate habitat management and land protection have been secured long-term. Initial evaluations of the number of large, potentially-viable populations in Alabama, Florida, and South Carolina are currently underway.

Threats: The primary threat to the gopher tortoise is from habitat destruction and modification in the form of conversion of native pine forests to intensively managed silvicultural pine forests, urban development, and habitat degradation due to lack of fire management. Overutilization for commercial, recreational, scientific, or educational purposes resulting from ongoing rattlesnake roundups are likely to continue to threaten the gopher tortoise now and into the future in the vicinity of roundup events (Service 2011a). Disease is expected to become more problematic for gopher tortoises as additional habitat is lost and fragmentation increases. Stressors are likely to elevate risks of tortoises to upper respiratory tract disease, but these effects will likely be localized (Service 2011a).

Miami Tiger Beetle (endangered)

Legal Status: The Service listed the Miami tiger beetle (*Cicindelidia floridana*) as endangered under the Act on October 5, 2016 (81 FR 68985). Proposed CH is expected later in 2017.

Life History/Population Dynamics: In tiger beetles, the adult female determines the habitat and microhabitat of the larva by the selection of an oviposition (egg-laying) site (Knisley and Schultz 1997). Generally, the same microhabitats are occupied by both larvae and adults. Females will often touch the soil with the antennae, bite it, and even dig trial holes, possibly to determine the suitable soil characteristics (Willis 1967) before placing a single egg into a shallow oviposition burrow (1 to 2 cm [0.39 to 0.79 in]) dug into the soil with the ovipositor. The egg hatches, apparently after sufficient soil wetting, and the first instar larvae digs a burrow at the site of oviposition. Development in tiger beetles includes three larval instars followed by a pupal and adult stage. In most species of tiger beetles, development requires 2 years, but can range from 1 to 4 or more years depending on climate and food availability. The life cycle of most tiger beetles in the United States follows either a summer or spring-fall adult activity pattern (Knisley and Schultz 1997). These life cycle patterns all indicate the length of the adult flight season is typically 2 to 3 months, but the life span of individual adults is likely to be less.

Based on available information, the Miami tiger beetle appears to have only limited dispersal abilities. Among tiger beetles there is a general trend of decreasing flight distance with decreasing body size (Knisley and Hill 1996). The Miami tiger beetle is one of the smallest tiger beetles (less than half an inch in length); it is likely to be a weak flier based on its size and the limited flight distance of the closely related Highlands tiger beetle (*Cicindelidia highlandensis*) (usually flying only 5-10 m [16.4-32.8 ft] and no more than 150 m [490 ft]) (Knisley and Hill 2013).

As a group, tiger beetles occupy ephemeral habitats where local extinction from habitat loss or degradation is common, so dispersal to establish new populations in distant habitat patches is a likely survival strategy for most species (Knisley 2015b). Limited dispersal capabilities and other constraints (few populations, limited numbers, and barriers created by intervening unsuitable habitat), however, can disrupt otherwise normal metapopulations dynamics and contribute to imperilment.

The Miami tiger beetle is vulnerable to extinction due to its severely reduced range, the fact that only two small populations remain, and the species' relative isolation. Demographic stochasticity refers to random variability in survival or reproduction among individuals within a population (Shaffer 1981). Demographic stochasticity can have a significant impact on population viability for populations that are small, have low fecundity, and are short-lived. In small populations, reduced reproduction or die-offs of a certain age-class will have a significant effect on the whole population. Although of only minor consequence to large populations, this randomly occurring variation in individuals becomes an important issue for small populations. Environmental stochasticity is the variation in birth and death rates from one season to the next in response to weather, disease, competition, predation, or other factors external to the population (Shaffer 1981). For example, drought or predation, in combination with a low population year, could result in extirpation. The origin of the environmental stochastic event can be natural or human-caused.

In general, tiger beetles that have been regularly monitored consistently exhibit extreme fluctuations in population size, often apparently due to climatic or other habitat factors that affect recruitment, population growth, and other population parameters. In 20 or more years of monitoring, most populations of the northeastern beach (*Cicindela dorsalis dorsalis*) and puritan tiger beetles (*Cicindela puritan*) have exhibited 2 to 5 or more fold differences in abundance (Knisley 2012). Annual population estimates of the Coral Pink Sand Dunes tiger beetle (*Cicindela albissima*) have ranged from fewer than 600 to nearly 3,000 adults over a 22-year period (Gowan and Knisley 2014). The Miami tiger beetle has not been monitored as extensively as these species, but in areas where Miami tiger beetles were repeatedly surveyed, researchers found fluctuations that were several fold in numbers (Knisley 2015a). While these fluctuations appear to be the norm for populations of tiger beetles (and most insects), the causes and effects are not well known. Among the suggested causes of these population trends are annual rainfall patterns for the Coral Pink Sand Dunes tiger beetle (Knisley and Hill 2001, Gowan and Knisley 2014), and shoreline erosion from storms for the northeastern beach and puritan tiger beetles (Knisley 2011b). As a result of these fluctuations, many tiger beetle populations will experience episodic low numbers (bottlenecks) or even local extinction from genetic decline, the Allee effect, or other factors. Population viability analyses for these other species (northeastern beach, puritan, and Coral Pink Sand Dunes tiger beetles) determined that stochasticity, specifically the fluctuations in population size, was the main factor accounting for the high risk of extinction (Gowan and Knisley 2001, 2005, Knisley and Gowan 2009). Given that the Miami tiger beetle is only known from two remaining populations with few adult individuals and separated by substantial urban development, any significant decrease in the population size could easily result in extinction of the species.

<u>Status and Distribution</u>: Several studies comparing various methods for estimating adult tiger beetle abundance have found numbers present at a site are typically 2 to 3 times higher than that produced by the visual index count (Knisley and Schultz 1997, Knisley 2009, Knisley and Hill 2013). Numbers are underestimated because tiger beetles are elusive, and some may fly off before being detected while other may be obscured by vegetation in some parts of the survey area. Even in defined linear habitats like narrow shorelines where there is no vegetation and high visibility, index counts produce estimates that are 2 to 3 times lower than the numbers present (Knisley and Schultz 1997).

Information on the Richmond population size is limited because survey data are inconsistent, and some sites are difficult to access due to permitting, security, and liability concerns. Of the occupied sites, the most thoroughly surveyed site for adult and larval Miami tiger beetles is in Zoo Miami, with over 35 survey dates from 2008 to 2016 (Knisley 2015a, Mays and Cook 2015, Cook 2016). Recent surveys within the Richmond site have extended the known distribution of

Miami tiger beetles to Larry and Penny Thompson Park (Mays and Cook 2015, Cook 2016), which is adjacent to the Zoo Miami and the University of Miami (UM) Richmond Campus properties. In total 34 adult Miami tiger beetles have been observed at Larry and Penny Thompson Park. Adult beetle surveys at the UM Richmond Campus (also known as CSTARS) and USCG properties have been infrequent, and access was not permitted in 2012 through early summer of 2014. In October 2014, access to both the UM Richmond Campus and United States Coast Guard (USCG) properties was permitted and no beetles were observed during October 2014 surveys. Surveys in 2016 observed both adult and larval Miami tiger beetles throughout the cleared antenna fields at the USCG property (Hazelton 2016, Maguire 2017).

Raw index counts found adults in four areas (Zoo A, Zoo B, Zoo C, and Zoo D) of the Zoo Miami parcel. Two of these patches (Zoo C and Zoo D) had fewer than 10 adults during several surveys at each. Zoo A, the more northern site where adults were first discovered, had peak counts of 17 and 22 adults in 2008 and 2009, but declined to 0 and 2 adults in six surveys from 2011 to 2014, despite thorough searches on several dates throughout the peak of the adult flight season (Knisley 2015a). Zoo B, located south of Zoo A, had peak counts of 17 and 20 adults from 2008 to 2009, 36 to 42 adults from 2011 to 2012, and 13 and 18 adults in 2014 (Knisley 2015a). These surveys at Zoo A and Zoo B also recorded the number of suitable habitat patches (occupied and unoccupied). Surveys between 2008 and 2014 documented a decline in both occupied and unoccupied open habitat patches. Knisley (2015a) documented a decrease at Zoo A from 7 occupied of 23 patches in 2008, to 1 occupied of 13 patches in 2014. At Zoo B, there was a decrease from 19 occupied of 26 patches in 2008, to 7 occupied of 13 patches in 2014 (Knisley 2015a). Knisley (2015a) suggested this decline in occupied and unoccupied patches is likely the result of the vegetation that he observed encroaching into the open areas that are required by the beetle.

A peak season raw count at UM Richmond Campus on August 20, 2010 produced 38 adults in 11 scattered habitat patches with 1 to 9 adults per patch, mostly in the western portion of the site (Knisley 2015a). Three surveys at the USCG included only a portion of the potential habitat and produced raw adult counts of two, four, and two adults in three separate patches from 2009, 2010, and 2011, respectively (Knisley 2015a). Additional surveys of the UM Richmond Campus and USCG properties on October 14 to 15, 2014, surveyed areas where adults were found in previous surveys and some new areas; however, no adults were observed. Observations made during surveys indicated that habitat patches that previously supported adults seemed smaller due to increased vegetation growth, and consequently these patches appeared less suitable for the beetle than in the earlier surveys (Knisley 2015a). In addition, recent surveys at Zoo Miami and USCG have identified adult and larval MTBs in open areas that are maintained, either by mowing (*e.g.*, cleared fields, grassy parking areas, open areas around existing structures, canal banks) or vehicles or pedestrian use (*e.g.*, dirt or gravel roads) (Knisley 2014; Hazelton 2016).

Surveys of adult numbers over the years, especially the frequent surveys in 2009, did not indicate a bimodal adult activity pattern (Knisley 2015a). Knisley (2015a) suggests that actual numbers of adult Miami tiger beetles could be 2 to 3 times higher than indicated by the raw index counts. Several studies comparing methods for estimating population size of several tiger beetle species, including the Highlands tiger beetles, found total number present were usually more than two times that indicated by the index counts (Knisley and Hill 2013). The underestimates from raw index counts are likely to be comparable or greater for the Miami tiger beetle because of its small sized and occurrence in small open patches where individuals can be obscured by vegetation around the edges, making detection especially difficult (Knisley 2015a).

Surveys for larvae at the Zoo Miami parcel (Zoos A and B) were conducted in several years during January when lower temperatures would result in a higher level of larval activity and open burrows (Knisley and Hill 2013) (see Table 2 in *Supporting Documents* on *http://www.regulations.gov*). The January 2010 survey produced a count of 63 larval burrows, including 5 first instars, 36 second instars, and 22 third instars (Knisley 2013). All burrows were in the same bare sandy patches where adults were found. In March 2010, a follow-up survey indicated most second instar larvae had progressed to the third instar (Knisley 2015a). Additional surveys to determine larval distribution and relative abundance during January or February in subsequent years detected fewer larvae in section Zoo B: 5 larvae in 2011, 3 larvae in 2012, 3 and 5 larvae in 2013, 3 larvae in 2014, and 15 larvae in 2015 (Knisley 2013; Knisley 2015c). The reason for this decline in larval numbers (*i.e.*, from 63 in 2010, to 15 or fewer in each survey year from 2011 to 2015) is unknown. Possible explanations are that fewer larvae were present because of reduced recruitment by adults from 2010 to 2014, increased difficulty in detecting larval burrows that were present due to vegetation growth and leaf litter, environmental factors (*e.g.*, temperature, precipitation, predators), or a combination of these factors (Knisley 2015a). Larvae, like adults, also require open patches free from vegetation encroachment to complete their development. The January 2015 survey observed vegetation encroachment, as indicated by several of the numbered tags marking larval burrows in open patches in 2010 covered by plant growth and leaf litter (Knisley 2015c). No larvae were observed in the January 2015 survey of Zoo A (Knisley 2015c). Knisley (2015d) reported that the area had been recently burned (mid-November) and low vegetation was absent, resulting in mostly bare ground with extensive pine needle coverage.

Surveys for the beetle’s presence outside of its currently known occupied range found no Miami tiger beetles at a total of 42 sites (17 pine rocklands sites and 25 scrub sites) throughout Miami-Dade, Broward, Palm Beach, and Martin Counties (Knisley 2015a, Mays and Cook 2015). The absence of the Miami tiger beetle from sites north of Miami-Dade was probably because it never ranged beyond pine rockland habitat of MDC and into scrub habitats to the north (Knisley 2015a). Sites without the Miami tiger beetle in MDC mostly had vegetation that was too dense and were lacking the open patches of sandy soil that are needed by adults for oviposition and larval habitat (Knisley 2015a).

The Miami tiger beetle is extremely rare and only known to occur in two separate locations within pine rockland habitat in MDC (Richmond Pine Rocklands - Zoo Miami, Larry and Penny Thompson Park, UM Richmond Campus, and USCG; and a more recently identified location within 5 km of the Richmond population). It is considered as one of the two tiger beetles in the United States most in danger of extinction (Knisley *et al.* 2014). The Miami tiger beetle is currently ranked S1 and G1 by the FNAI (2016), meaning it is critically imperiled globally because of extreme rarity or because of extreme vulnerability to extinction due to some natural or manmade factor. The overall population size of the Miami tiger beetle is exceptionally small and viability is uncertain. Based upon the index count data to date, it appears that the two populations exist in extremely low numbers (Knisley 2015a, Mays and Cook 2015, Cook 2016).

Threats: The Miami tiger beetle is threatened by habitat loss and modification caused by changes in land use and inadequate land management, including the lack of prescribed burns and vegetation (native and nonnative) encroachment (discussed separately below). Habitat loss and modification are expected to continue and increase, affecting any populations on private lands as well as those on protected lands that depend on management actions (*i.e.*, prescribed fire) where these actions could be precluded by surrounding development.

The Miami tiger beetle has experienced substantial destruction, modification, and curtailment of its habitat and range (Brzoska *et al.* 2011, Knisley 2013, Knisley 2015a). The pine rockland community of South Florida, on which the beetle depends, is critically imperiled globally (FNAI 2013). Destruction of the pinelands for economic development has reduced this habitat by 90 percent on mainland South Florida (O'Brien 1998). Outside of ENP, only about 1 percent of the Miami Rock Ridge pinelands have escaped clearing, and much of what is left is in small remnant blocks isolated from other natural areas (Herndon 1998).

The two known populations of the Miami tiger beetle occur within the Richmond Pine Rocklands, on parcels of publicly or privately owned lands that are partially developed, yet retain some undeveloped pine rockland habitat. In the 1940s, the Naval Air Station Richmond was built largely on what is currently the Zoo Miami parcel. Much of the currently occupied Miami tiger beetle habitat on the Zoo Miami parcel was scraped for the creation of runways and blimp hangars (Service 2015). The fact that this formerly scraped pine rockland area now provides suitable habitat for the Miami tiger beetle demonstrates the restoration potential of disturbed pine rockland habitat (Possley 2015).

Any current known or unknown, extant Miami tiger beetle populations or potentially suitable habitat that may occur on private lands or non-conservation public lands, such as elsewhere within the Richmond Pine Rocklands or surrounding pine rocklands, are vulnerable to habitat loss. MDC leads the State in gross urban density at 15.45 people per ac (Zwick and Carr 2006), and development and human population growth are expected to continue in the future. By 2025, MDC is predicted to exceed a population size of over 3 million people (Zwick and Carr 2006). This predicted economic and population growth will further increase demands for land, water, and other resources, which will undoubtedly impact the survival and recovery of the Miami tiger beetle.

The threat of habitat destruction or modification is further exacerbated by a lack of adequate fire management (Brzoska *et al.* 2011, Knisley 2013, Knisley 2015a). Historically, lightning-induced fires were a vital component in maintaining native vegetation within the pine rockland ecosystem, as well as for opening patches in the vegetation required by the beetles (Loope and Dunevitz 1981, Slocum *et al.* 2003, Snyder *et al.* 2005, Knisley 2011a). Open patches in the landscape, which allow for ample sunlight for thermoregulation, are necessary for Miami tiger beetles to perform their normal activities, such as foraging, mating, and oviposition (Knisley 2011a). Larvae also require these open patches to complete their development free from vegetation encroachment. Without fire, successional change from tropical pineland to hardwood hammock is rapid, and displacement of native plants by invasive, nonnative plants often occurs, resulting in vegetation overgrowth and litter accumulation in the open, bare, sandy patches that are necessary for the Miami tiger beetle.

In the absence of fire, pine rockland will succeed to tropical hardwood hammock in 20 to 30 years, as thick duff layer accumulates and eventually results in the appearance of humic soils rather than mineral soils (Alexander 1967, Wade *et al.* 1980, Loope and Dunevitz 1981, Snyder *et al.* 1990).

MDC has implemented various conservation measures, such as burning in a mosaic pattern and on a small scale, during prescribed burns; to help conserve the Miami tiger beetles and other imperiled species and their habitats (Maguire 2010). MDC Parks and Recreation staff has burned several of its conservation lands on fire return intervals of approximately 3 to 7 years. However, implementation of the county's prescribed fire program has been hampered by a shortage of resources, logistical difficulties, smoke management, and public concern related to burning next to residential areas (Snyder *et al.* 2005, FNAI 2010). Many homes and other developments have been built in a mosaic of pine rockland, so the use of prescribed fire in many places has become complicated because of potential danger to structures and smoke generated from the burns. The risk of liability and limited staff in MDC have hindered prescribed fire efforts (URS 2007). Nonprofit organizations, such as the Institute for Regional Conservation, have faced similar challenges in conducting prescribed burns, due to difficulties with permitting and obtaining the necessary permissions, as well as hazard insurance limitations (Bradley and Gann 2008; Gann 2013). Few private landowners have the means or desire to implement prescribed fire on their property, and doing so in a fragmented urban environment is logistically difficult and costly (Bradley and Gann 2005).
Lack of management has resulted in rapid habitat decline on most of the small pine rockland fragments, with the disappearance of federally listed and candidate species where they once occurred (Bradley and Gann 2005).

Despite efforts to use prescribed fire as a management tool in pine rockland habitat, sites with the Miami tiger beetle are not burned as frequently as needed to maintain suitable beetle habitat. Most of the occupied beetle habitat at MDC's Zoo Miami parcel was last burned in January and October of 2007; by 2010, there was noticeable vegetation encroachment into suitable habitat patches (Knisley 2011a). The northern portion (Zoo A) of the Zoo Miami site was burned in November 2014 (Knisley 2015c). Several occupied locations at the UM Richmond Campus parcel were burned in 2010, but four other locations at UM Richmond Campus were last burned in 2004 and 2006 (Knisley 2011a). No recent burns are believed to have occurred at the USCG parcel (Knisley 2011a). The decline in adult numbers at the two primary Zoo Miami patches (A and B) in 2014 surveys, and the few larvae found there in recent years, may be a result of the observed loss of bare open patches (Knisley 2015a, Knisley 2015c). Surveys of the UM Richmond Campus and USCG parcels in 2014 found similar loss of open patches from encroaching vegetation (Knisley 2015a).

Alternatives to prescribed fire, such as mechanical removal of woody vegetation are not as ecologically effective as fire. Mechanical treatments do not replicate fire's ability to recycle nutrients to the soil, a process that is critical to many pine rockland species (URS 2007). To prevent organic soils from developing, uprooted woody debris requires removal, which adds to the required labor. The use of mechanical equipment can also damage soils and inadvertently include the removal or trampling of other non-target species or CH (URS 2007).

Nonnative plants have significantly affected pine rocklands (Bradley and Gann 1999, Bradley and Gann 2005, Bradley and van der Heiden 2013). As a result of human activities, at least 277 taxa of nonnative plants have invaded pine rocklands throughout South Florida (Service 1999). *Neyraudia neyraudiana* (Burma reed) and *Schinus terebinthifolius* (Brazilian pepper), which have the ability to rapidly invade open areas, threaten the habitat needs of the Miami tiger beetle (Bradley and Gann 1999). *S. terebinthifolius*, a nonnative tree, is the most widespread and one of the most invasive species. It forms dense thickets of tangled, woody stems that completely shade out and displace native vegetation (Loflin 1991, Langeland and Craddock Burks 1998). *Acacia auriculiformis* (earleaf acacia), *Melinis repens (*natal grass), *Lantana camara* (shrub verbena), and *Albizia lebbeck* (tongue tree) are some of the other nonnative species in pine rocklands. More species of nonnative plants could become problems in the future, such as *Lygodium microphyllum* (Old World climbing fern), which is a serious threat throughout South Florida.

Nonnative, invasive plants compete with native plants for space, light, water, and nutrients, and make habitat conditions unsuitable for the Miami tiger beetle, which responds positively to open conditions. Invasive nonnatives also affect the characteristics of a fire when it does occur. Historically, pine rocklands had an open, low understory where natural fires remained patchy with low temperature intensity. Dense infestations of *Neyraudia neyraudiana* and *Schinus terebinthifolius* cause higher fire temperatures and longer burning periods. With the presence of invasive, nonnative species, it is uncertain how fire, even under a managed situation, will affect habitat conditions or Miami tiger beetles.

Management of nonnative, invasive plants in pine rocklands in MDC is further complicated because the vast majority of pine rocklands are small, fragmented areas bordered by urban development. Fragmentation results in an increased proportion of "edge" habitat, which in turn has a variety of effects, including changes in microclimate and community structure at various distances from the edge (Margules and Pressey 2000); altered spatial distribution of fire (greater fire frequency in areas nearer the edge) (Cochrane 2001); and increased pressure from nonnative, invasive plants and animals that may out-compete or disturb native plant populations. Additionally, areas near managed pine rockland that contains nonnative species can act as a seed source of nonnatives, allowing them to continue to invade the surrounding pine rockland (Bradley and Gann 1999).

Rim Rock Crowned Snake (petitioned)

Legal Status: The Service received a petition for listing the rim rock crowned snake (*Tantilla oolitica*) as endangered or threatened under the Act on July 11, 2012. The State of Florida recognizes the rim rock crowned snake as State-designated Threatened. CH has not been designated at this time

Species and Critical Habitat Description: There are three species of burrowing snakes belonging to the genus *Tantilla* that occur in Florida, including the rim rock crowned snake. The rim rock crowned snake is non-venomous and can reach a length of up to 11.5 in (29 cm) (Ernst and Ernst 2003). It has a tan to beige dorsum (back) and pinkish-white to cream-colored belly (Florida Fish and Wildlife Conservation Commission (FWC) 2013). Its head and neck are dark brown or black, and its scales are smooth (Florida Natural Areas Inventory (FNAI 2001). Individuals from the Keys may have a pale neckband that is not present in mainland snakes (Porras and Wilson 1979).

<u>Life History/Population Dynamics</u>: The rim rock crowned snake is a very rare species with only 12 individuals recorded between 1991 and 2009 (Hines and Bradley 2009). They inhabit pine rockland and tropical hardwood hammocks, as well as human-altered habitats such as roadsides, vacant lots, and pastures with shrubby growth and pines (Duellman and Schwarz 1958, Campbell and Moler 1992, Hines and Bradley 2009). They can be found in holes and depressions in the oolitic limestone (formed by calcium carbonate), but they can also be found periodically in rotten logs and under rocks and trash (Enge *et al.* 2003, Campbell and Moler 1992). It is not known what the rim rock crowned snake eats; however, if similar to other members of the genus *Tantilla*, it likely feeds on insects and other small invertebrates (Ernst and Ernst 2003).

Much of the life history of this species is unknown because of its secretive, fossorial nature. Based upon reproductive ecology and longevity of the similar southeastern crowned snake (*Tantilla coronata*), the rim rock crowned snake likely reaches reproductive maturity at 2 years of age and lays up to six eggs per year (three eggs per clutch; two clutches per year), with a lifespan of at least 5 years in the wild (Todd *et al.* 2008, Ernst and Ernst 2003).

Surveys conducted by Enge *et al.* (2004) in six MDC parks failed to locate any rim rock crowned snakes. Staff at Zoo Miami began herpetological trapping efforts with coverboards and drift fence arrays with pitfall traps in 2004, and, despite 1,640 hours of trapping effort, they found only one individual in a pitfall trap in August 2009 (FWC 2011). Hines (2011) deployed 84 coverboards over a 3-year period and only located two individuals. Because of the difficulty of finding this species, population size estimates are nearly impossible to determine; however numbers observed over the 10 years of her study were similar to those from the previous 10-year period (Hines 2011).

Based upon observations over the years, the species appears to be able to adapt to a multitude of habitats, including rockland habitats, dump sites, urban and agricultural landscapes, and hammock habitat with closed canopy and loose, dark, moist soil (Hines and Bradley 2009). A sighting record on Big Pine Key was documented at a pineland-hammock ecotone where construction and household waste (*i.e.*, carpet, old plywood boards, etc.) provided dependable moisture in an otherwise dry landscape (Hines and Bradley 2009). Hines and Bradley (2009) also documented observations at Barnacle Historic State Park, which is comprised of fewer than 4 ac of hammock habitat. This record suggests that a large expanse of habitat may not be necessary for survival, most likely because home range sizes appear to be small.

<u>Status and Distribution</u>: Very few sightings have been reported for the elusive rim rock crowned snake. Hines and Bradley (2009) interviewed observers (professional herpetologists and hobbyists) and found that many had never seen a rim rock crowned snake despite extensive searching. They found that the number of observations has been highest in the last two 20-year periods but only averages 10 individuals per 20-year period. Only six observations were documented from 1930 to 1950; likewise, six were documented from 1951 to 1970 (Hines and Bradley 2009). From 1971 to 1990, 18 were reported, while only 12 were observed between 1991 and 2009 (Hines and Bradley 2009). This increase in documented observations may be the

result of greater exposure of residents to this species due to urban expansion and human population growth in South Florida. Because only two individuals were documented during extensive searches and there are no areas known to consistently support them, it's unlikely that the snake population is experiencing growth (Hines and Bradley 2009).

Rim rock crowned snakes inhabit pine rockland and tropical hardwood hammocks near fresh water and, of the more than 40 species of this genus extending from the southeastern United States down to northern Argentina into South America, the rim rock crowned snake has the most limited distribution (Wilson 1982, Scott 2004). Its geographic range is confined to the southern tip of Florida, including the Florida Keys (FWC 2013). Limited to the Miami Rock Ridge in southeastern MDC and southern Monroe County, this species has been impacted by rapid human growth and urbanization (Hines and Bradley 2009). Rim rock crowned snakes are known from various mainland locations in Miami, including Brownsville, Coconut Grove, Coral G ables, Cutler, Cutler Ridge, Kendall, Leisure City, North Miami, and Perrine (Duellman and Schwartz 1958, FNAI 2001). The species is also known to occur in the Upper and Middle Florida Keys with only one confirmed occurrence in the Lower Florida Keys (Campbell and Moler 1992, Yirka *et al.* 2010), and Zoo Miami (FWC 2011).

Threats: The primary threat to the rim rock crowned snake is the fragmentation and degradation of their habitat (FWC 2011). The proximity of the species' habitat to coastal Florida carries the increasing threat of damage and loss from hurricanes, tropical storms, and sea level rise due to global climate change.

White-crowned pigeon (State threatened)

The following species review has been assembled from the FWC's information on the white-crowned pigeon including their web-site, http://myfwc.com/media/2211508/White-Crowned-Pigeon.pdf and November 2013 and Species Action Plan http://myfwc.com/media/2738292/White-Crowned-Pigeon-Species-Action-Plan-Final-Draft.pdf.

Legal Status: The white-crowned pigeon (*Patagioenas leucocephala*) is designated as threatened by the State of Florida.

Species and Critical Habitat Description:

The white-crowned pigeon is a medium size member of the genus *Patagioenas*. This pigeon species can reach a length of 14 inches (35.6 centimeters) long, with a wingspan of 23 inches (58.4 centimeters). White-crowned pigeons have a white head and a gray body with green feathers on the dorsal side of the neck (The Cornell of Ornithology 2011). White-crowned pigeons are very arboreal and are rarely seen on the ground (Bancroft and Bowman 2001). They are extremely skittish and easily flushed from both nesting and foraging areas (Bancroft 1996, Bancroft and Bowman 2001).

The white-crowned pigeon is a subtropical frugivorous species occurring in low-lying forest habitats with ample fruiting trees. In south Florida and the Florida Keys, white-crowned pigeons feed primarily on the fruits of hardwood trees and shrubs in deciduous seasonal forests, mostly in tropical hardwood hammocks. In Florida, two specific habitat types are essential for the survival of the white-crowned pigeon: mangrove islands for breeding and tropical hardwood hammock for foraging. Because the white-crowned pigeon is not currently listed under the Act, there is no critical habitat designated for the species.

Life History/Population Dynamics:

White-crowned pigeons breed in south Florida from May to early September where they commonly nest semi-colonially on tidally inundated mangrove islands, which provide some protection from predators such as raccoons (Bancroft and Bowman 2001). Males are semi-territorial on nesting sites and assist females with nest construction (Wiley and Wiley 1979). One to 3 eggs are laid, with 2 being the most common. In the daytime, males care for the eggs and nestlings; females care for them at night. The eggs hatch in 13 to 14 days, usually on successive days. The young will leave the nest after 16 to 22 days, though they may remain in the vicinity of the nest for up to 40 days after hatching (Bancroft 1996). Juveniles will disperse from the nesting islands after 26 to 45 days from hatching (Strong and Bancroft 1994).

During nesting, adult white-crowned pigeons produce liquid from their crops, referred to as crop-milk, which they feed to their young. The crop-milk is high in protein and lipids and provides 100% of the chick's diet for the first 2 days. The adults then begin supplementing the crop-milk with fruit, gradually increasing the proportion of fruit to milk as the chicks mature. Adults may continue to provide some crop-milk to the young during the entire nestling period (Bancroft and Bowman 1994).

White-crowned pigeons are obligate frugivores and travel daily from their nesting location in the mangroves to forage in tropical hardwood hammocks and, to a lesser extent, pine rocklands that contain an understory of fruit-bearing trees and shrubs (Bancroft and Bowman 2001). They will also feed heavily on individual fruit-bearing trees in small, vegetated patches in suburban environments where a quality food source is present (Meyer and Wilmers 2006). White-crowned pigeons are strong and fast in flight and are known to travel daily in straight flight-lines, sometimes over 50 kilometers (31 miles), for foraging. They can fly more than 150 kilometers (93 miles) over the water during migration, generally flying low until reaching land.

Status and Distribution:

The white-crowned pigeon's distribution in the United States is restricted to southern Florida. This species also occurs in the Bahamas, Greater and Lesser Antilles, and the Caribbean coast of southeastern Mexico and Central America. The breeding range for the species is centered on the Bahamas and Greater Antilles, although populations extend into southern Florida, the Lesser Antilles, and the Caribbean coast of southeastern Mexico and Central America (Arendt *et al*. 1979, Bancroft and Bowman 2001, Bent 1932, Goodwin 1983, Wiley 1979). Its breeding range in the United States is restricted to Florida Bay, Barnes Sound, Card Sound, Biscayne Bay, and the Florida Keys, although a few individuals probably nest inland in Monroe and Miami-Dade counties (Bancroft and Bowman 2001, FWC 2003, National Park Service 2011, Strong *et al*. 1991).

Distributions of local populations in south Florida are influenced by temporal forage availability and level of habitat fragmentation (Bancroft and Bowman 1994). In 1987 through 1989, Strong *et al*. (1991) found that 52% of mangrove keys supported white-crowned pigeon nests in Florida

Bay. More islands close to the mainline Florida Keys supported nests than did islands closer to peninsular Florida, with the highest nest densities in the center of Florida Bay (Strong *et al*. 1991).

The Biological Status Review estimated the species' range in Florida, or total extent of occurrence, at <5,000 kilometers^2 (<1,930.5 miles^2) and noted that a large majority of this area was open water; the land area actually occupied by white-crowned pigeons in Florida is probably <1,250 kilometers^2 (<482.6 miles^2).

Some white-crowned pigeons that breed in Florida overwinter there, while most of the population migrates south or southeast for the winter (Bancroft 1996, Robertson and Woolfenden 1992). Studies suggest that 80% to 90% of white-crowned pigeons breeding in Florida winter in the Bahamas and the Caribbean (Bancroft and Bowman 1994, Bancroft and Bowman 2001, Meyer and Wilmers 2006).

<u>Threats</u>:

During the late 1800s and early 1900s, the white-crowned pigeon was threatened by overhunting. However, conservation laws helped the population recover in the United States. In the Caribbean, the white-crowned pigeon continues to be overhunted. Since white-crowned pigeons are restricted to low-lying areas, the main threat to the white-crowned pigeon presently is habitat degradation and deforestation (BirdLife International 2008). White-crowned pigeons also face threats to their food supply as tropical hammocks continue to be destroyed in the Keys (Florida Natural Areas Inventory 2001). Bancroft and Bowman (2001) list hunting and harvesting, pesticides and other contaminants, collisions with structures or objects, degradation of habitat, and human disturbance impacts as primary threats to white-crowned pigeons. Nest predation by raccoons (*Procyon lotor*) and other predators is also a documented threat (Strong *et al.* 1991).

Furthermore, the Florida white-crowned pigeon population is contained within Monroe and Miami-Dade counties, where it is vulnerable to hurricane events, both because of its geographic location and its restricted range. Historical storm records corroborate the vulnerability of these geographic locations. Increasing frequency of severe tropical storms and hurricanes (Webster *et al.* 2006) as well as sea level rise due to climate change are expected to degrade and reduce the available nesting habitat for the species. In addition, essential foraging habitat continues to decline. For example, the area of tropical hardwood hammocks in the upper Florida Keys declined by 31% between 1991 and 2004 (Karim and Main 2009), primarily due to development. Nest numbers and productivity are strongly correlated to the food supply (Bancroft and Bowman 2001).

Blodgett's Silverbush (threatened)

<u>Legal Status</u>: The Service listed the Blodgett's silverbush (*Argythamnia blodgettii*) as threatened on September 29, 2016 (81 FR 66842 66865) under the Act. CH has not been designated for this species.

<u>Life history and Population Dynamics</u>: Blodgett's silverbush first became a candidate on October 25, 1999 and is currently listed as threatened. The following discussion is summarized

from the most recent species assessment (Service 2012b) and from recent research publications and monitoring reports. "*A. blodgettii* is an erect suffrutescent perennial 1-6 decimeters (dm) tall, the stems and leaves covered with bifurcate hairs; leaves entire, oval to elliptic, sometimes slightly spatulate, 1.5-4 cm long, often colored a distinctive metallic bluish green, distinctly 3-nerved; staminate calyx 7-8 millimeter (mm) wide; sepals are lanceolate; petals broadly elliptic, shorter than sepals; pistillate sepals lanceolate to linear-lanceolate; petals broadly elliptic, shorter than sepals; pistillate sepals lanceolate to linear-lanceolate, 5-6 mm long; capsule 4-5 mm wide (Adapted from Small 1933)" (Bradley and Gann 1999). Reproduction is sexual; flowering and fruiting apparently takes place throughout the year (Bradley and Gann 1999).

On the mainland, Blodgett's silverbush grows in pine rockland and edges of rockland hammock (Bradley and Gann 1999). In the Keys, this species grows in pine rockland, rockland hammock, coastal berm and on roadsides, especially in sunny gaps or edges (Bradley and Gann 1999). Bradley and Gann (1999) stated "*A. blodgettii* is primarily a plant of open sunny areas in pine rockland, edges of rockland hammock, edges of coastal berm, and sometimes disturbed areas in close proximity to a natural area. Plants can be found growing from crevices on oolitic or Key Largo limestone or on sand. The pine rockland habitat where it occurs in MDC and the Florida Keys requires periodic fire to maintain an open, sunny understory with a minimum amount of hardwoods." Bradley and Gann (1999) indicated this species does tolerate some degree of human-induced disturbance. It can often be found along disturbed edges of pine rockland, rockland hammock, and coastal berm, or in completely scarified pine rockland (Bradley and Gann 1999).

In the Keys, Blodgett's silverbush is extant on nine islands, with three others of uncertain status (Hodges and Bradley 2006). The largest population surveyed is on Big Munson Island and is estimated to be 8,000-9,000 plants (Hodges and Bradley 2006). The population size in the Keys, excluding Big Pine, is estimated to be approximately 11,000 plants (Hodges and Bradley 2006). Occurrences on Big Pine Key vary by location and are shown below (Hodges and Bradley 2006). According to data from the Institute for Regional Conservation (IRC), the estimated population of Blodgett's silverbush in MDC is 375-13,650 plants (*i.e.*, total of low and high estimates from $\log_{10}$ scale) (Bradley 2007); however, this may be an overestimate of the actual population size because it was based upon a $\log_{10}$ scale. In ENP, the current estimated population size is 1,000 plants (Sadle 2008a, 2010).

<u>Status and Distribution</u>: *A. blodgettii* historically occurred from central and southern MDC from Brickell Hammock (latitude ca. 25° 45.9') to southwestern Long Pine Key in ENP (latitude ca. 25° 24.2'), and throughout the Florida Keys (Monroe County and MDC) from Totten Key (latitude 25° 22.95') south to Key West (latitude 24° 32.52')" (Bradley and Gann 1999). Based upon Hodges and Bradley (2006) and data from IRC (Bradley 2007), Blodgett's silverbush has been extirpated from the sites in Table 1 (Bradley and Gann 1999).

Table 1. Extirpated occurrences of Blodgett's silverbush (Bradley and Gann 1999).

Site	Owner	County	Last Observed	Cause
Brickell Hammock	Unknown	Miami-Dade	1937	Developed
Caribbean Park	Miami-Dade County	Miami-Dade	1998	Developed
Coconut Grove	Unknown	Miami-Dade	1901	Developed
Coral Gables Area	Unknown	Miami-Dade	1967	Developed
Fuchs Hammock	Miami-Dade County	Miami-Dade	1991	Developed, fire suppression
Key West	Unknown	Monroe	1965	Developed
Key West Cemetery	Public	Monroe	1965	Unknown
Miller and 72 Ave.	Unknown	Miami-Dade	1975	Developed
North Key Largo	Various	Monroe	1977	Unknown
Orchid Jungle	Miami-Dade County	Miami-Dade	1930	Unknown (development, fire suppression, exotic pest plants likely)
Palms Woodlawn Cemetery	Private	Miami-Dade	1992	Developed
S. of Miami River	Unknown	Miami-Dade	1913	Developed
Stock Island	Private	Monroe	1981	Developed
Totten Key (Biscayne National Park)	National Park Service	Miami-Dade	1904	Unknown
Vaca Key	Unknown	Monroe	1909	Developed
NFC #317	Private	Miami-Dade	unknown	Developed

A. blodgettii is currently known from central MDC from Coral Gables (latitude 25° 43.45') and southern MDC to southwestern Long Pine Key in ENP (latitude 25° 24.2'), and the Florida Keys from Windley Key (latitude 24° 57.08') southwest to Big Pine Key (latitude 24° 38.52')" (Bradley and Gann 1999). Although the total extent of the former range is unknown, approximately 12 mi (19 km) of habitat has been lost near the northern end of the range in MDC and 43 mi (69 km) has been lost in Monroe County (Bradley and Gann 1999). More recently, Hodges and Bradley (2006) indicated that species' verified range extends from MDC to Boca Chica Key.

Based upon Bradley and Gann (1999), Hodges and Bradley (2006), and data from IRC (Bradley 2007), Blodgett's silverbush is extant at the sites in Table 2. However, the species may be extirpated from the Charles Deering Estate, the Epmore Drive Pineland fragment, the Old Dixie Pineland, and S.W. 184 Street and 83 Avenue (Bradley 2007). The NFC (Natural Forest Community) #317 site has been destroyed (Bradley 2007). Indefinite occurrences (those which have not been verified lately) in Monroe County include Key West Golf Course, Boot Key, and Long Key State Park (Hodges and Bradley 2006). Indefinite occurrences in MDC are between Coconut Grove and Cutler, and between Cutler and Longview Camp (Bradley 2007).

Table 2. Extant occurrences of Blodgett's silverbush (Bradley and Gann 1999; Hodges and Bradley 2006; Sadle 2007, 2008a, 2008b, 2010, 2011).

Site	Owner	County	Population Size	Threats
Big Munson Island	The Boy Scouts of America	Monroe	1,001-10,000	exotic plants, possible development in future (non-imminent)
Big Pine Key, Cactus Hammock and Long Beach coastal berm	National Key Deer Refuge	Monroe	1,000 – 10,000 (approximately 2,000)	fire suppression, storm surge, exotic plants, trail maintenance
Big Pine Key, Koehn's subdivision	National Key Deer Refuge (in part)	Monroe	101 – 1,000 (approximately 200)	fire suppression, exotic plants, road maintenance, illegal dumping, paving, infrastructure projects, herbicide spraying
Big Pine Key, Watson's Hammock	National Key Deer Refuge	Monroe	2	fire suppression, hot fires, other natural disturbance events, exotic plants
Blue Heron Hammocks, Vaca Key	Florida Fish and Wildlife Conservation Commission	Monroe	11-100	road maintenance, exotic plants, infrastructure, herbicide spraying
Boot Key	private	Monroe	11-100	development
Camp Owaissa Bauer	Miami-Dade County	Miami-Dade	101-1,000	fire suppression, exotic plants
Castellow Hammock	Miami-Dade County	Miami-Dade	11-100	fire suppression, exotic plants
Charles Deering Estate**	Miami-Dade County	Miami-Dade	11-100	fire suppression, exotic plants, fence line maintenance
Country Ridge Estates	private	Miami-Dade	11-100	exotic plants
Epmore Drive Pineland Fragment**	private	Miami-Dade	2-10	development, exotic plants
ENP, Deer Hammock Area (pine block A) and adjacent pine block B	National Park Service	Miami-Dade	1,000	Brazilian pepper
Fuchs Hammock Addition	Miami-Dade County	Miami-Dade	2-10	
Gifford Arboretum Pineland	private	Miami-Dade	2-10	development, exotic plants
Key Largo, Dove Creek Hammocks	Florida Fish and Wildlife Conservation Commission	Monroe	11 - 100	road construction, mowing, exotic plants
Key West Naval Air Station, Boca Chica Key	Department of Defense	Monroe	1,001 – 10,000 (approximately 1,200)	lead tree (*Leucaena leucocephala*), maintenance activities, development, dumping of toxic substances, opening of new roads
Larry and Penny Thompson Park and adjacent properties	Miami-Dade County	Miami-Dade	1,001-10,000	development, fire suppression, exotic plants
Lignumvitae Key Botanical State Park, Lignum Vitae Key	Department of Environmental Protection	Monroe	11-100	maintenance activities, exotic plants
Lignumvitae Key Botanical State Park, Klopp Tract, Matecumbe Key	Department of Environmental Protection	Monroe	11-100	general disturbance, weedy and exotic plants
Ned Glenn Nature Preserve	Miami-Dade County	Miami-Dade	11-100	fire suppression, exotic plants
Old Dixie Pineland (= Keg South Pineland)**	private	Miami-Dade	11-100	development, fire suppression, exotic plants
Owaissa Bauer Addition	Miami-Dade County	Miami-Dade	100-1,000	fire suppression, exotic plants
Pine Ridge Sanctuary	private preserve	Miami-Dade	2-10	exotic plants
Snake Creek Hammock, Plantation Key	Florida Fish and Wildlife Conservation Commission	Monroe	101-1,000	exotic plants, maintenance activities
S.W. 184 St. and 83 Ave.**	private	Miami-Dade	11-100	development, fire suppression, exotic plants
Windley Key Fossil Reef State Geological Site	Florida	Monroe	11-100	maintenance activities, exotic plants

The finding of a new, small occurrence approximately 0.9 mi (1.5 km) from the Deer Hammock site (yet within Pine Block B) suggests that the range within ENP is larger than originally thought (Sadle 2010).

The current range for Blodgett's silverbush includes MDC and the Keys. According to data from IRC, the estimated population of Blodgett's silverbush in MDC is 375-13,650 plants (*i.e.*, total of low and high estimates from log10 scale) (Bradley 2007); however, this may be an overestimate of the actual population size because it was based upon a log10 scale. "Blodgett's silverbush is currently known from central MDC from Coral Gables (latitude 25° 43.45') and southern MDC to southwestern Long Pine Key in ENP (latitude 25° 24.2'). The range in MDC has contracted approximately 12 mi, all at the northern end of its range, the heaviest developed portion of MDC" (Bradley and Gann 1999).Based upon Hodges and Bradley (2006) and data from IRC (Bradley 2007), Blodgett's silverbush has been extirpated from at least 10 former sites within

MDC. However, several sites relatively close to the proposed action area retain Blodgett's silverbush populations: Camp Owaissa Bauer, the Deering Estate (population potentially extirpated), Fuchs Hammock Addition, Richmond Pinelands (Larry and Penny Thompson Park and adjacent properties), Ned Glenn Pineland, and Owaissa Bauer Addition.

Threats: The primary threat to the Blodgett's silverbush is development leading to the destruction and modification of its habitat. Continued habitat degradation through fire suppression and the spread of exotic vegetation pose significant risks to the extant populations remaining.

Carter's Small-flowered Flax (endangered)

Legal Status: The Service listed the Carter's small-flowered flax (*Linum carteri* var. *carteri*) as endangered on September 4, 2014 (79 FR 52567 52575) under the Act. CH was designated on August 17, 2015 (80 FR 49845 49886) for approximately 2,605 ac (1,054 ha) in MDC, Florida.

Life history and Population Dynamics: Carter's small-flowered flax is an annual or short-lived perennial herb that is endemic to MDC, where it grows in pine rockland, particularly disturbed pine rocklands (Bradley and Gann 1999). Bradley and Gann (1999) described the species as follows, "Stems erect 23 to 36 cm tall, commonly branched near the base, puberulent; leaves slender, 18-26 mm long, 0.8 to 1.2 mm wide, entire, alternate, closely overlapping at the base of the plant, more distant above; stipules with paired dark glands; inflorescence an ascending or spreading cyme; pedicels 4.5 to 9 mm long in fruit; sepals lanceolate, short-awned, glandular toothed, 3-nerved; petals orange yellow, broadly obovate, 9 to 17 mm long, quickly deciduous; fruit straw-colored, ovoid, 4.1 to 4.6 mm long, 3.4 to 3.7 mm diameter, dehiscing into five two-seeded segments; seeds narrowly ovoid-elliptic, 2.3 to 2.8 mm long,1 to 1.3 mm wide. (Adapted from Rogers 1963 and 1968). In habit and flower the plant closely resembles *Piriqueta caroliniana* (pitted stripeseed) in the Turneraceae."

Carter's small-flowered flax is found in pine rocklands, particularly those that are scarified or have undergone some sort of soil disturbance (*e.g.*, firebreaks, canal banks, edges of railway beds) (Bradley and Gann 1999). None of the known occurrences are from a completely undisturbed pine rockland (Bradley and Gann 1999). Bradley and Gann (1999) indicated that all documented occurrences are within scarified pine rocklands, in disturbed areas adjacent to or within pine rocklands, or in completely disturbed areas. This species does not tolerate shading or litter accumulation, and therefore may have been excluded from much of its former habitat by fire suppression (Bradley and Gann 1999).

The reproductive ecology and biology of this taxon is not well understood, but reproduction is sexual (Bradley and Gann 1999). The magnitude and frequency of seed production is unknown; some fruits dehisce in a characteristic 5-parted star pattern, while others never dehisce (Fellows 2002).

Maschinski and Walters (2008) studied in situ germination and growth-to-maturity of plants growing in the wild at two sites, measuring height, number of branches, number of buds, flowers, and fruit of 32 seedlings. Of the total 32 seedlings tracked, only 6 set fruit (Maschinski and Walters 2008). The mean time to set first bud was 197 ± 2.4 days, while mean time to first fruit set was 226 ± 2.3 days (Maschinski and Walters 2008). The 226-day growth- to-maturity enables Carter's small-flowered flax to contribute seeds to a next generation in a relatively short period (Maschinski and Walters 2008). Once mature, individuals may live one to several years producing multiple fruits (Maschinski and Walters 2008). Growth-to maturity may be influenced by season of germination; seeds germinating in the summer may grow to maturity more rapidly than seedlings that germinate in the fall or winter (Maschinski and Walters 2008). Carter's small-flowered flax is capable of flowering throughout the year, but tends to have the most abundant flowering and fruiting following rain (Maschinski and Walters 2008).

Carter's small-flowered flax has typical behavior for an early successional species (Maschinski 2006). In a recent study to examine population viability in response to disturbance, long-term demography studies were initiated at disturbed and undisturbed sites in MDC (Maschinski 2006; Maschinski and Walters 2007). These studies indicated Carter's small-flowered flax occurred in higher densities at a mowed site where competition with other plants was decreased. However, mowing can also eliminate reproduction entirely in very young plants or delay reproductive maturation (Maschinski and Walters 2007). Disturbance from mowing was found to result in higher mortality, but greater fruit production (Maschinski 2006). Because mowing had been a repeated pressure on one population for more than 50 years, it is possible that mowing is also selecting for plants that can grow and reproduce more rapidly than the disturbed site plants (Maschinski 2006). This work confirms, to a degree, the recommendation by Bradley and Gann (1999) that "periodic mowing in these areas may partially replace fires, maintaining an open, shrub-free understory."

Preliminary models indicated that population viability was greatly affected by reproduction and whether there is a persistent seed bank (Maschinski 2006; Maschinski and Walters 2007). Fruiting was variable across years and sites, such that there was no clear effect of mowing on fruit production (Maschinski 2006; Maschinski and Walters 2007). Seedlings and juveniles (non-reproductive) had a higher probability of survival to adult stage at the undisturbed site than at the mown site; however, the mown site had higher reproduction than the undisturbed site (Maschinski 2006). Models indicate that transitions from seedling to adult and adult reproduction greatly influence population trajectories (Maschinski and Walters 2007). Increasing these vital rates is critical to improving population persistence (Maschinski and Walters 2007). Year-to-year variation was found to be extremely high across populations and subject to the unpredictability of weather (Maschinski and Walters 2007). Continued monitoring is needed to determine whether disturbance regime has a persistent impact on life history (Maschinski 2006).

Bradley and Gann (1999) estimated that the total population size was 101 to 1,000 plants (based on a $\log_{10}$ scale) and that the population may be declining. Based on the latest available data, the total population size is estimated to be between 318 to 2,615 individuals, although a better estimate of the upper range may be 2,215 if all populations on private lands have been extirpated. Maschinski *et al.* (2003 and 2004) noted that this short-lived perennial has widely fluctuating numbers of individuals. Development, exotic plants, mountain biking, modification to fire regime, mechanical disturbance, and herbicide use were cited as threats (Bradley and Gann 1999). Bradley and Gann (1999) stated that this taxon is in severe danger of extinction since most of the occurrences were not on conservation lands (at that time). Bradley and Gann (1999) also indicated that the conservation lands where this species occurs contained only a few dozen plants combined, one of which was damaged by maintenance crews. Since 1999, data from IRC and Fairchild Tropical Botanical Garden (FTBG) indicate that at least three additional occurrences (on private lands) have likely been extirpated since most of those sites were destroyed (Cocoplum Development, Old Dixie Pineland [=Keg South Pineland], and Ponce and Riviera Pineland) (Bradley 2007; Possley 2012). However, populations at the Rockdale Pineland Preserve and the United States Department of Agriculture (USDA) Subtropical Horticulture Research Station were found to have more individuals than previously estimated (Bradley 2007; Possley 2012), and a new occurrence was discovered (Montgomery Foundation) (Maschinski 2006).

Status and Distribution: John Kunkel Small and Joel J. Carter first collected this species in 1903 between Coconut Grove and Cutler; Small described it as a new species in 1905 (Gann *et al.* 2002). Bradley and Gann (1999) indicated it has been found at many widespread locations, from the Coconut Grove area of Miami (latitude 25° 43.8') to southern MDC, terminating near SW 280 Street (latitude 25° 30.4'), a range of about 24 mi (39 km) (Bradley and Gann 1999). Since 1903, Carter's small-flowered flax has been found in pine rocklands from as far north as the Brickell Hammock area to as far south as the Naranja area (Gann *et al.* 2002).

Austin *et al.* (1980) mapped 17 stations for Carter's small-flowered flax. Most of those stations are likely to be historic (the report's format did not allow the authors to clearly note where plants had been found during field work). Bradley and Gann (1999) believe several occurrences represented misidentifications—that the plants were either sand flax or Small's flax (*Linum carteri* var. *smallii)*. For example, a previous report of the plant occurring at Homestead Air Reserve Base site is now considered to be erroneous (Bradley 2008). Based upon data from IRC, Carter's small-flowered flax is extirpated from Brickell Hammock (owner unknown) due to development, Charles Deering Estate (owned by MDC) for unknown reasons, and the Red Road and 114 Terrace locations (private land) due to development (Bradley 2007). Austin *et al.* (1980) noted that there were four historical sites for this species in a study of southern Florida, including NKDR and Great White Heron National Wildlife Refuge. However, in 1980, Austin *et al.* (1980) found only one site remaining, representing a 75 percent reduction in number of sites, and attributed the reduction to urbanization. Gann *et al.* (2002) indicated most of its habitat has been destroyed.

Carter's small-flowered flax is currently found from R. Hardy Matheson Preserve (near Pinecrest) southwest to Naranja/Modello, with a distance of approximately 27.3 km (17 mi) between the farthest locations. The apparent reduction in its historic range (11.2 km [7.2 mi]; 30 percent) has occurred entirely in the northern portion, between Pinecrest and Coconut Grove, primarily due to urban development. Similarly, much of the habitat within the variety's current range has been destroyed (Gann *et al.* 2002). At least five known populations have been extirpated including: Brickell Hammock (site developed; last observation in 1911); Red Road/114 Terrace (site developed; last observation in 1969); Deering Estate at Cutler (not sighted since 1980s; unknown reason); Ponce and Riviera Pineland (site developed in 2004); and Cocoplum Development (site developed in 2005) (Bradley 2007; Bradley and van der Heiden 2013). Bradley and Gann (1999) described nine known populations (only three of these occurring on conservation lands) with an estimated total population of 100-1,000 individuals; its status was thought to be possibly declining. Maschinski *et al.* (2004) estimated the total population to be 10,300 plants across eight populations in 2003, with one population sustaining the vast majority (approximately 10,000 individuals). Carter's small-flowered flax was not found during a 2-year project intended to survey and map nonnative and rare plants along Florida Department of Transportation (FDOT) right-of-ways within MDC (Gordon *et al.* 2007).

In 2012, Bradley and van der Heiden (2013) conducted a status survey for Carter's small-flowered flax to include extant occurrences, historic locations, and new survey stations. Because they had previously conducted a comprehensive survey of all pine rockland habitat in 2004-2005 (during which, Carter's small-flowered flax was not found on any new sites), this habitat was excluded from new surveys. Canals within urban MDC that intersected with the pine rockland soils of the Miami Rock Ridge were surveyed, as were additional disturbed sites with remnant native vegetation in close proximity to existing sites. Carter's small-flowered flax was found at seven locations containing approximately 1,313 individuals; populations ranged in size from a single plant to 700 plants, with a median of 18 plants (Table 3; Bradley and van der Heiden 2013). One occurrence (at Gifford Arboretum Pineland), which had not been observed since the 1990s but whose habitat was still extant, was deemed "Historical" and may reappear there (Bradley and van der Heiden 2013). Of the seven extant occurrences, five populations are on publicly owned lands but only three of these are managed for the conservation of natural resources (Table 3). Four of the populations occur near the north end of the variety's range (near R. Hardy Matheson Preserve) and three occur near the south end (near Camp Owaissa Bauer), with an approximately 16 km (10 mi) gap between the closest populations of these groups. Within each grouping, populations are approximately 1.3-4.3 km (0.8-2.7 mi) apart.

Because this variety is known to be a short-lived perennial with widely fluctuating numbers of individuals (Maschinski *et al.*2003, 2004), as well as being difficult to find when not in flower, we include an estimate of population range using the logarithmic scale (Table 3) to account for these characteristics and to provide a comparison to the previous total population estimates. Using the logarithmic scale, the total population estimate is 337-3,310 plants. However, it should be noted that most 2012 observations were at the low end of the corresponding logarithmic range such that the resulting high end for the total population estimate may be a gross overestimate of the actual population. Based strictly on 2012 observations, the total population estimate may be closer to 1,300 individuals. Comparing these estimates to the 1999 and 2003 population estimates generally supports the boom-and-bust nature of Carter's small-flowered flax, although the decline since 2004 could also potentially indicate a declining trend for the largest occurrence.

The species was not found during a two-year project intended to survey and map exotic and rare plants along FDOT right-of-ways within Miami-Dade and Monroe counties (Gordon *et al.* 2007).

Carter's small-flowered flax was recently found at seven locations containing approximately 1,313 individuals; populations ranged in size from a single plant to 700 plants, with a median of 18 plants (Bradley and van der Heiden 2013). Four of the populations occur near the north end of the variety's range (near R. Hardy Matheson Preserve) and three occur near the south end (near Camp Owaissa Bauer), with an approximately 16 km (10 mi) gap between the closest populations of these groups. This variety occurs on the Owaissa Bauer Addition (11-100 plants), R. Hardy Matheson (101-1,000 plants), and the Rockdale Pineland (101-1,000 plants) in MDC. R. Hardy Matheson and the Rockdale Pineland appear to contain two of the three largest occurrences of the subspecies.

Table 3. Extant and Historical Populations of Carter's Small-Flowered Flax

POPULATION (NFC # if applicable (P-#))	**OWNERSHIP (* denotes lands managed for conservation)**	**POPULATION RANGE (Est. No. of plants in 2012)** [1]
Extant: Population status confirmed in 2012 surveys conducted by IRC		
C-103 Canal	State of Florida – South Florida Water Management District	1-10 (1)
Camp Owaissa Bauer Addition (P-255.4)	State of Florida – managed by MDC*	11-100 (13)
Chapman Field, USDA Subtropical Horticultural Research Station (portions are P-63)	Federal – U.S. Department of Agriculture	101-1000 (700)
Montgomery Botanical Center	Private – Montgomery Botanical Center	11-100 (12)
Old Dixie Pineland	Private	11-100 (18)
R. Hardy Matheson Preserve (H-634)	State of Florida – managed by MDC*	101-1000 (374)
Rockdale Pineland (P-52)	MDC*	101-1000 (195)
Historical: Population not observed for > 10 years, but habitat extant		
Gifford Arboretum Pineland	Private	0

[1] Source for number of plants is Bradley and van der Heiden (2013)

Threats: The number of known populations of Carter's small-flowered flax has decreased by nearly 50 percent in recent years, and extant populations are small and isolated. Of the remaining species' occurrences, four are on conservation lands; three of these have approximately 100 individuals or fewer. Another site is owned by the U.S. government, but the site is not managed for conservation. On private lands, this species is threatened by on-going urban development (NatureServe 2012), and habitat destruction is a major threat (Gann *et al.* 2002) as demonstrated by the recent probable extirpations of at least three populations on private lands. The Service has determined that the threats to Carter's small-flowered flax consist primarily of habitat loss and modification through urban and agricultural development, fire suppression, proliferation of nonnative invasive plants, and sea level rise.

Threats described under habitat loss, fragmentation, and degradation resulting from development, fire suppression, and competition from nonnative invasive plants are believed to be the primary drivers in the historic and recent declines of Carter's small-flowered flax and has also been threatened by anthropogenic disturbances which threaten populations in disturbed habitats such as firebreaks and road rights-of-way, and both taxa are suspected to be negatively affected by threats related to small, isolated populations. All of these threats are expected to continue to impact populations of these taxa in the future. Current local, State, and Federal regulatory mechanisms are inadequate to protect these taxa from taking and habitat loss. Despite the existing regulatory mechanisms, Carter's small-flowered flax continue to decline. Remaining habitats are fragmented. Climatic changes, including sea-level rise, are long-term threats that will further reduce the extent of habitat. Most occurrences are in low-lying areas and will likely be affected by rising sea level.

Carter's small-flowered flax is vulnerable to natural disturbances, such as hurricanes, tropical storms, and storm surges. Due to the few remaining occurrences within a restricted range and the small and isolated populations, this species is vulnerable to environmental (catastrophic hurricanes), demographic (potential episodes of poor reproduction), and genetic (potential inbreeding depression) threats. This species exists in such small numbers at so few sites, that it may be difficult to develop and maintain viable occurrences on the available conservation lands. Viable plant populations for small, short-lived herbs may consist of tens of thousands of plants. Although no population viability analysis has been conducted for this plant, indications are that existing occurrences are at best marginal, and it is possible that none are truly viable. Lack of dispersal between occurrences may also be a threat (Fellows *et al.* 2004; Service 2013a).

Crenulate Lead-Plant (endangered)

Legal Status: The Service listed the crenulate lead-plant (*Amorpha crenulata*) as endangered on July 18, 1985 (50 FR 29345 29349) under the Act. No CH has been designated for the crenulate lead-plant.

Life history and Population Dynamics: The crenulate lead-plant is a rhizomatous, perennial, deciduous shrub that inhabits marl prairies and wet pine rocklands in a small area of MDC. This pine rockland community is maintained by periodic fires. Also known as the Miami lead-plant, crenulate lead-plant grows to 1.5 m in height and is endemic to MDC, Florida (FDOT 1997). The branches of this plant are red/purple, and contain 25 to 33 leaflets borne on leaves that are 0 to 15 cm long, with petioles 1 cm long or less. The crenulate leaflets are gray and green above, paler and glandular dotted below, and 5 to 11 cm long. The racemes are terminal, 15 to 20 cm long, solitary, or in clusters of two to three. The 8-mm long flowers are held in loose clusters. The calyx is dark green or purplish, 3.2 to 4.0 mm long with the upper half glandular dotted. The showy white standard flower is 5.2 mm long and 4.2 mm wide with long exerted stamens. The fruit is 6 to 11 mm long, laterally compressed, and glandular dotted on the upper two-thirds. The seeds produced in the fruit are 5 mm long and compressed.

Not much is known of the life history of crenulate lead-plant. The plants are long-lived, but little to no recruitment occurs in populations in a typical year (Fisher 2000). Plants show little to no growth and flower primarily following human disturbance. Several species of native solitary bees, such as *Dianthidium curvatum floridens* and non-native honeybees, *Apis mellifora,* pollinate the flowers (Koptur 2006). Shoots of these woody plants die back to the root stock following fire or other disturbance, and, therefore, age of the plant may not be strongly correlated with size (Fisher 2000). Crenulate lead-plant is semi-deciduous, with about 70 percent of plants losing most or all leaves between December and February. New sprouts, when observed, have been identified as primarily adventitious roots (FDOT 1997). In addition, the viability of germplasm is not known (FDOT 1997). Fisher (2000) reported this species is relatively easy to cultivate, indicating the lack of reproduction in the wild may not be due to a lack of viable seeds.

Maschinski *et al.* (2005) reported low recruitment rates may be due to the depth of the duff layer and to hydrologic influences. A propagation protocol has recently been developed for conservation purposes (Roncal *et al.* 2006).

The crenulate lead-plant occurs in plant communities that were historically associated with seasonally hydrated soils and frequent burning, including wet pinelands, transverse glades, and hammock edges. It can be found growing in poorly-drained Opalocka sands within pine rocklands or in wet prairies with Opalocka-rock outcrop complex soils. It requires open sun to partial shade. The type specimen (Small and Wilson #1898) describes the primary habitat type for crenulate lead-plant as hammock (MDC Department of Environmental Resource Management [DERM] 1993). No recent collections have been seen from within hardwood hammocks. Many of Small's specimen labels were pre-printed with habitat data and some species were collected and labeled as occurring in hammocks that were actually collected in habitat types outside of hammocks. It is possible crenulate lead-plant was never collected in hammocks.

The pine rocklands where the crenulate lead-plant occurs are characterized by a canopy of slash pine (*Pinus elliottii* var. *densa*), a shrub canopy of saw palmetto (*Serenoa repens*), wax myrtle (*Myrica cerifera*), poison wood (*Metopium toxiferum*), and willow bustic (*Sideroxylon salicifolium*). Common herbaceous associates include crimson bluestem (*Schizachyrium sanguineum* var. *sanguineum)*, wire bluestem (*S. gracile)*, scaleleaf aster (*Aster adnatus)*, and bastard copperleaf (*Acalypha chamaedrifolia).* Other typical species associates of crenulate lead-plant include cabbage palm (*Sabal palmetto*), southern sumac (*Rhus copallina* var. *leucantha*), bluestem (*Schizachyrium rhizomatum*), wild-petunia (*Ruellia succulenta*), gulfdune paspalum (*Paspalum monostachyum)*, and blueheart (*Buchnera americana*).

<u>Status and Distribution</u>: Vegetative communities within the historic range of crenulate lead-plant have been almost entirely eliminated by agricultural, urban, and commercial development. The transverse glades where crenulate lead-plant occurs were among the first areas in MDC to be farmed, because their marl soils were better suited to conversion to farmland than the limestone rock of the adjacent pinelands. By 1984, 98 to 99 percent of MDC pine rocklands had been destroyed, and development continues today.

The crenulate lead-plant was known from a 20-mi^2 area from Coral Gables to Kendall, MDC (DERM 1993). Its historic range was only slightly greater, extending south to Cutler (based on an entry of *Amorpha caroliniana* on an unpublished plant list by John Kunkol Small of Addison Hammock), and north to the Little River in northeast MDC. This range encompasses an area 5 mi east to west and 12 mi north to south. The crenulate lead-plant is currently known from six sites, four of which contain natural populations and two contain re-introduced populations (Roncal *et al.* 2006). The two largest natural populations showed a slight increase in numbers of individuals in 2012, of which one site had particularly high seedling recruitment (Maschinski *et al.* 2012). However, within the last 10 years, four additional natural populations were lost to urban development, leaving the total population size at less than 2,000 individuals (Roncal *et al.* 2006).

The current range of this species, limited to four natural sites and two introduced sites, is almost fully confined within MDC. The sites in the action proposed for management that contain natural populations of the plant are A.D. Barnes Park (22 ac), and Tropical Park (5 ac). There were 208 plants documented at A.D. Barnes Park and 130 plants recorded in Tropical Park from 2012 surveys (Maschinski *et al.* 2012). These are the two largest natural populations. R. Hardy Matheson contained one of the four natural populations, but the most recent survey in 2010 found no plants (Maschinski *et al.* 2012). The only other natural populations of this species, at Coral Pines Park (Pinecrest), is very small (5 plants; Maschinski *et al.* 2012) approximately 5 mi northeast of the proposed action area. The Deering Estate has one documented introduced population (67 plants in 2011; Maschinski *et al.* 2011) and the other introduced population at Luis Martinez Army Reserve in the Richmond Pinelands (215 plants in 2012; Maschinski *et al.* 2012), immediately adjacent to the proposed action area.

Threats: Crenulate lead-plant was listed as endangered because of the loss of pine rockland habitat from residential and commercial development. In addition, fire suppression, invasion by exotic plant species, and drainage threaten the survival of the crenulate lead-plant. Flowering and seed production may not occur as a result of these disruptions. A newly recognized potential threat to trees and shrubs in South Florida is lobate lac scale (*Paratachardina lobata lobata*), an invasive scale insect. It was discovered on some of the crenulate lead-plants at one of the sites in November 2004 (Maschinski *et al.* 2005). Since that time, it has not appeared to be a threat to crenulate lead-plant.

Deltoid Spurge (endangered)

Legal Status: The Service listed the deltoid spurge (*Chamaesyce deltoidea* ssp. *deltoidea*) as endangered on July 18, 1985 (50 FR 29345 29349) under the Act. No CH has been designated for the deltoid spurge.

Life history and Population Dynamics: Deltoid spurge, a member of the *Euphorbiaceae* (spurge family), is an herbaceous, prostrate to barely ascending plant forming small mats to a few decimeters in diameter. The thin, wiry stems extend from a central woody taproot. Leaves are deltoid to ovate in shape, opposite, and up to 5 mm (0.2 in) long. Flowers are unisexual; male and female flowers are arranged in a cuplike structure (cyathium). The 3-seeded fruits are 1 to 2 mm (0.04 to 0.08 in) wide; seeds measure about 1 mm (0.04 in) wide. The density and distribution of hairs on the stems, leaves, and capsules distinguish varieties *deltoidea* and *adhaerens*. Variety *deltoidea* is essentially hairless; *adhaerens* is fairly hairy.

The deltoid spurge tends to occur in areas with an open shrub canopy, exposed limestone (oolite), and minimal litter (pine needles, leaves, and other organic materials). It is most often found growing at the edges of sand pockets with plants growing both in sand (sometimes in association with the endangered tiny polygala) and on oolitic limestone. The soils in which it grows are classified as Opalocka-Rock Outcrop soils. The subspecies *C. deltoidea* ssp. *adhaerens* occurs in fine, reddish sandy loam over limestone. Dense colonies are sometimes found in pinelands that have undergone a slight mechanical disturbance, where little or no topsoil is formed and where productivity is low. The shrub canopy in this disturbed habitat is often poorly developed providing high light levels and low organic litter accumulation rates. The pine rocklands are often considered a fire subclimax, and are maintained with periodic fires (3 to 7 years). These periodic fires keep the shrub canopy down and eliminate the litter accumulations.

Studies into the life history of the deltoid spurge have only recently begun, and little is known about its reproduction. It is a perennial that flowers from April through November, peaking in July. Its extensive root system gives evidence it is a long-lived plant (DERM 1993). The reproductive ecology in *Chamaesyce* has been poorly studied, but it is known to be highly variable (Ehrenfeld 1976, 1979; Webster 1967). Some species are completely reliant on insects for pollination and seed production while others are self-pollinating. Pollinators may include bees, flies, ants, and wasps (Ehrenfeld 1979). Seed capsules of many Euphorbiaceae are explosively dehiscent, ejecting seeds a short distance from the parent plant. The seeds of some species are dispersed by ants (Pemberton 1988).

Current estimates of the number of individuals have not been obtained for the entire population, and population trends are not well understood. The NAM (Natural Areas Management) staff of MDC have reported plants on some of their sites have significantly declined with one site having only three plants, another having two populations containing no more than one or two plants, and a third site having only two distinct colonies remaining after reporting an abundance of plants in the late 1980s (Maguire 2006 in litt.). In a study conducted in three plots located in the northern Biscayne pinelands, Herndon (2002) noted populations occur in small, dense, widely-separated clusters of 50 to 200 individuals. Population sizes varied 10 to 50 percent annually but no general decrease in population size was reported. He estimated 800 to 8,000 plants occurred in each population at the Deering Estate pinelands and Larry and Penny Thompson Park.

Annual recruitment rates range from 0.0 to 0.2 and mortality rates range from 0 to 0.39 (Herndon 2002). Survival in three study plots over the 3-year study period was 41, 46, and 65 percent. Low seed germination rates were detected in both greenhouse conditions and field assessments, and seed production varied seasonally by rainfall amount. While Herndon's (2002) study evaluated parameters such as population size, recruitment, survival, and mortality, other information such as growth and reproductive characteristics are necessary for population modeling. A research project conducted at Larry and Penny Thompson Park in 1992 compared the growth rates of this subspecies in burned versus unburned plots (DERM 1993). Data on plant size and flower density was collected in each plot, and results indicated that plants respond to fire by allocating energy towards vegetative recovery immediately after fire, rather than to flowering.

Status and Distribution: Deltoid spurge is a MDC endemic that was historically known to occur in pine rocklands of the Miami rock ridge from the Goulds area north to the center of the city of Miami. The northern portion of its range has been completely modified by urban expansion. In 1992-93, deltoid spurge plants were known to occur on 18 sites, including the Richmond pine rocklands classified as one site where several thousand individuals were recorded (DERM 1993). Seven of these sites were owned by MDC, and eight others were proposed for acquisition. According to recent updates, five sites located on private lands have been developed (Maschinski 2005 in litt.).

Results of a project to map the remaining pine rockland habitat in 2006 reported deltoid spurge occurred on 11 public sites (IRC 2006). Currently, the species is known to remain on 14 public lands (12 County, 1 State, and 1 Federal) and an undetermined number of private lands from southern Miami to Homestead (Bradley 2010). Even though the majority of the populations occur on public lands, they are fragmented, and habitat degradation continues to affect the extant populations. Because of habitat modification due to urban expansion in the northern portion of the range, deltoid spurge is now known only from south of Miami to the Homestead area. Its limited distribution renders the spurge vulnerable to random natural or human induced events, such as hurricanes and encroachment of invasive exotic species (IRC 2006). The current number of individuals in wild populations is not known, therefore, trend analysis is not available. Although some demographic information is available for deltoid spurge, additional long-term research will be necessary to develop accurate population models.

Deltoid spurge is a MDC endemic that was historically known to occur in pine rocklands of the Miami rock ridge from the Goulds area north to the center of the city of Miami. Currently the species is known to remain on 14 public lands (12 County, 1 State, 1 Federal) and an undetermined number of private lands from southern Miami to Homestead (Bradley 2010). Deltoid spurge occurs within the Richmond Pine Rocklands (Woodmansee 2014) in addition to 9 other sites within MDC: Bill Sadowski Park, the Deering Estate, Ludlum Pineland, Ned Glenn Pineland, Pine Shore, Quail Roost, Rockdale Pineland, Ron Ehman Park, and Trinity Pineland. The current number of individuals in wild populations is not known.

Threats: Continued habitat loss and fragmentation threaten the existence of deltoid spurge, and less than 2 percent of the original acreage of pine rockland habitat remains (Maschinski *et al.* 2002). Populations on private sites remain threatened with destruction or habitat modification due to improper or lack of management. Modification of pine rockland habitat on protected lands is also of concern (Maschinski *et al.* 2008). There is an ongoing effort to conduct prescribed burns at the publicly-owned sites. Management of these small preserves is difficult because exotic plants are present within and near the properties. Habitat degradation on these sites continues to be a moderate threat because vegetation restoration and management programs are costly and depend upon availability of funding (Service 2006b).

Everglades Bully (proposed threatened)

Legal Status: The Service proposed to list the Everglades bully (*Sideroxylon reclinatum* ssp. *austrofloridense*) as threatened under the Act on October 11, 2016 (81 FR 70282 70308). CH has not been designated for this species.

Life history and Population Dynamics: Everglades bully is a decumbent or upright shrub, 3-6 ft (1-2 m) tall. The branches are smooth, slightly geniculate, and somewhat spiny. Leaves are thin, obovate or ovate, 0.8-2 in (2-5 cm) long, evergreen, oblanceolate, and fuzzy on their undersides. The flowers are in axillary cymes (Long and Lakela 1971). Everglades bully is distinguished from the other two subspecies of *S. reclinatum* in Florida by its leaves, which are persistently pubescent (fuzzy) on their undersides, rather than smooth or pubescent only along the midvein (Wunderlin and Hansen 2003).

Everglades bully is restricted to pinelands with tropical understory vegetation on limestone rock (pine rocklands), mostly in the Long Pine Key area of ENP, which is an area of pine rockland surrounded by wetlands. In ENP and BCNP, Everglades bully is found in pinelands, pineland/prairie ecotones, and prairies (Gann *et al.* 2006; Bradley *et al.* 2013). Plants are found in low elevation pinelands and pineland/marl prairie ecotones that flood each summer (Gann *et al.* 2006; Bradley *et al.* 2013). Bradley *et al.* (2013) conducted surveys in the Gum Slough region of Lostman's Pines in BCNP and reported finding the subspecies to have distribution within the study area.

In 2005, IRC reported that more than 10,000 plants were found in surveys of Long Pine Key (Bradley 2005). The baseline abundance estimate at Long Pine Key based on a $\log_{10}$ abundance estimate is 10,000-100,000 plants (Gann *et al.* 2006). Gann *et al.* (2006) found 14 occurrences of this species recorded at 149 stations. Bradley *et al.* (2013) conducted surveys in the Gum Slough region of Lostman's Pines in BCNP and reported finding Everglades bully to have limited distribution within the study area. A total of 17 plants (representing 0.2 plants per ha) were counted within pinelands plots (n = 3), that were associated with sawgrass and hardwood habitats (Bradley *et al.* 2013).

FTBG tagged 41 groups of plants, each group consisting of 1 to 6 individuals, for a total of approximately 73 individuals at Larry and Penny Thompson Park (Possley and McSweeney 2005). This is probably the largest population outside of Long Pine Key. Estimated population sizes for the other occurrences are noted in Table 4 (Hodges and Bradley 2006; Gann *et al.* 2006; Bradley 2007; Possley 2011a, 2011b).

<u>Status and Distribution</u>: The rounded global status of Everglades bully is T1, critically imperiled (NatureServe 2010). NatureServe (2010) indicates this taxon is a narrow, endemic subspecies occurring in sensitive and highly fragmented pine rocklands of southern Florida. FNAI considers Everglades bully to have a global rank of G4G5T1, meaning the species as a whole is "apparently" or "demonstrably secure globally," but the subspecies is "critically imperiled globally" (FNAI 2011). Everglades bully was considered to be critically imperiled by IRC; however, based upon data collected in the first year of their study, IRC down-ranked this species to imperiled (Gann *et al.* 2006; Gann *et al.* 2001-2010). Everglades bully is not listed by the State.

Everglades bully was long considered to be restricted to the tropical pinelands of MDC. Gann *et al.* (2002) provided a history of collections: Everglades bully was first documented at Camp Jackson near what is now the main entrance to ENP. It has been collected several times (starting in 1852) at Long Pine Key. The species has been observed in pinelands east of ENP, the Nixon-Lewis Hammock (where the pinelands have since been destroyed), privately-owned Grant Hammock, and privately-owned Pine Ridge Sanctuary.

In Monroe County, this species is found only on the mainland (Hodges and Bradley 2006). Hodges and Bradley (2006) stated that if it had occurred in the Florida Keys, the most likely locations would have been pine rocklands on Key Largo, Big Pine Key, Cudjoe Key or Lower Sugarloaf Key, all of which were surveyed for this species. Hodges and Bradley (2006) indicated that most of the sites on Key Largo have been developed. There have been no records of this taxon ever being collected there.

Everglades bully is extant at 11 sites (Table 4). One population occurs locally at BCNP along the edges of Gum Slough within Lostman's Pines area (south of Loop Road), on the mainland portion of Monroe County (Bradley *et al.* 2013). The largest population is at Long Pine Key within ENP in MDC (Hodges and Bradley 2006; Gann *et al.* 2006). New occurrences within ENP are expected to be found as work continues to establish the limits of this species' habitat requirements. Everglades bully appears to have a much wider range than previously thought (Gann *et al.* 2006).

One occurrence is located at Larry and Penny Thompson Park in the Richmond Pinelands adjacent to the Metrozoo in MDC (Gann *et al.* 2002; Possley and McSweeney 2005). This plant occurs at the privately-owned Pine Ridge Sanctuary in MDC and possibly at a few non-protected pinelands, such as Grant Hammock (Gann *et al.* 2002). In 2007, Bradley (2007) reported small occurrences in MDC at the following locations: Lucille Hammock, South Dade Wetlands, NFC #P-300, and NFC #P-310. More recently, Possley (Possley 2011a) found two plants at Quail Roost Pineland, an area that was formerly very overgrown, but was treated for manual hardwood reduction in 2007 and then burned in 2009.

Possley (2011b) reported populations from Navy Wells Pineland Preserve (four plants) and Sunny Palms Pinelands (two plants), both areas are MDC conservation lands. Everglades bully is extant at 11 sites in Monroe and Miami-Dade Counties (Bradley *et al.* 2013). This subspecies occurs within the Richmond Pinelands at Larry and Penny Thompson Park adjacent to the Metrozoo (73 plants; Gann *et al.* 2002; Possley and McSweeney 2005). Possley (2011a) found two plants at Quail Roost Pineland, an area that was formerly very overgrown, but was treated for manual hardwood reduction in 2007 and then burned in 2009. Possley (2011b) reported populations from Navy Wells Pineland Preserve (four plants) and Sunny Palms Pinelands (two plants); both areas are MDC conservation lands.

Table 4: Extant occurrences of Everglades bully (Hodges and Bradley 2006; Gann *et al.* 2006; Bradley 2007; Possley 2011a, 2011b; Sadle 2011; Bradley *et al.* 2013).

Site	**Owner**	**County**	**Estimated abundance**	**Threats (site specific only)**
Long Pine Key, ENP	NPS	Miami-Dade	10K – 100K	Sea level rise, exotic plants, fire suppression, hydrologic alterations
Big Cypress National Preserve	NPS	Monroe	17	Sea level rise, exotic plants, fire suppression, hydrologic alterations
Larry and Penny Thompson Park	MDC	Miami-Dade	Approx. 73	Sea level rise, exotic plants, fire suppression, hydrologic alterations
Navy Wells Pineland Preserve	MDC	Miami-Dade	4	Sea level rise, exotic plants, fire suppression, hydrologic alterations
Sunny Palms Pineland	MDC	Miami-Dade	2	Sea level rise, exotic plants, fire suppression, hydrologic alterations
Pine Ridge	private	Miami-	Unknown	Sea level rise, development,

Sanctuary		Dade		fire suppression, exotic plants
Lucille Hammock	MDC	Miami-Dade	11 - 100	Sea level rise, exotic plants, fire suppression
South Dade Wetlands	Partially acquired by MDC	Miami-Dade	Unknown	Sea level rise, exotic plants, fire suppression
NFC #P-300	private	Miami-Dade	2 - 10	Sea level rise, development, fire suppression, exotic plants
NFC #P-310	private	Miami-Dade	11 - 100	Sea level rise, development, fire suppression, exotic plants
Quail Roost Pineland	Miami-Dade EEL Preserve	Miami-Dade	2	Sea level rise, fire suppression, exotic plants

Threats: The MDC pine rocklands have largely been destroyed by residential, commercial, and urban development and agriculture. Pine rocklands in the county (including patches of marl prairie) have been reduced to about 11 percent of their former extent (Kernan and Bradley 1996, p. 2). Of the estimated historical extent of 182,780 ac (74,000 ha), only 20,106 ac (8,140 ha) of pine rocklands remained in 1996. Outside of ENP, only about one percent of the Miami Pine Rock Ridge pinelands remain and much of what is left is in small remaining blocks isolated from other natural areas (Herndon 1998, p. 1).

Habitat loss continues to occur in the species range and most remaining suitable habitat has been negatively altered by human activity. MDC has developed a network of small public conservation lands and has encouraged conservation of natural vegetation on private land. The County's actions may have averted extirpation of this and other pineland plants. As a result, some opportunities exist to conserve this plant on private land in MDC, but there is little opportunity to acquire more conservation lands. Conservation of privately owned pine rocklands in MDC is largely a matter of County government cooperation with private landowners and the County offers incentives for landowners to maintain their natural forest communities.

Everglades bully habitat at Long Pine Key in ENP (*e.g.*, pinelands, pineland/prairie ecotones, and prairies [Gann *et al.* 2006, p. 12]) and BCNP are, for the most part, protected. The largest population is essentially protected from habitat loss due to development or agriculture; however impacts from sea level rise, hydrological changes, and other natural and anthropogenic factors may still affect this species despite its protection on public conservation lands. Any occurrences and suitable habitat remaining on private land are threatened by habitat loss and degradation, and threats are expected to continue with increases in Florida's human population.

Fire suppression is a significant threat to Everglades bully (Gann *et al.* 2002, p. 527). Fire maintains the pine rockland community. Under natural conditions, lightning fires typically occurred at 3 to 7- year intervals or more frequently in marl prairies. With fire suppression, hardwoods eventually invade pine rocklands and shade out understory species. Fire suppression has reduced the size of the areas that do burn and habitat fragmentation has prevented fire from moving across the landscape in a natural way. Thus, many pine rockland communities are becoming tropical hardwood hammocks.

Exotic species have altered the type of fire that occurs in pine rocklands. Historically, pine rocklands had an open, low understory where natural fires remained patchy, with relatively low temperatures, thus sparing many native grasses and shrubs. Dense exotic plant growth can create higher temperatures and longer burning periods. Pine rockland plants cannot tolerate these extreme conditions. As a result, the native plants may have to be conserved by removing exotics through methods other than burning. One such method, hand chopping followed by spot treatment, is labor intensive and very costly. Pinelands in MDC outside of ENP are kept intact only by constant maintenance, including removal of exotic plants such as *Neyraudia reynaudiana* (Burmareed), *Schinus terebinthifolius* (Brazilian pepper), and others, use of prescribed fires, and prevention or cleanup of dumped trash.

Florida Brickell-bush (endangered)

Legal Status: The Service listed the Florida brickell-bush (*Brickellia mosieri*) as endangered on October 6, 2014 (79 FR 52567 52575) under the Act. CH was designated for approximately 2,646 ac (1,071 ha) in MDC, Florida on August 17, 2015 (80 FR 49845 49886).

Life history and Population Dynamics: Florida brickell-bush is a perennial herb 1 to 3.5 ft (0.3 to 1.1 m) tall, slender, erect, and branching (Chafin 2000). Leaves are 0.4 to 1.2 in (1 to 3 cm) long, alternate, narrow, linear, thick, usually spreading or curved downward, entire or slightly toothed, resin-dotted (Chafin 2000). The flower heads are in loose, open clusters at the ends of branches (Chafin 2000). Disk flowers are white in small, dense heads surrounded by hairy, slightly ribbed bracts; there are no ray flowers, although long style branches (white, sometimes brown) may appear to be rays (Chafin 2000). Reproduction is sexual, pollinators and dispersers are unknown (Bradley and Gann 1999). Flowering takes place primarily in the fall (August to October), but individuals may be found in flower during most of the year (Bradley and Gann 1999).

Bradley and Gann (1999) stated that Florida brickell-bush is “found exclusively in pine rocklands. It tolerates only minor amounts of disturbance. The pine rockland habitat where it occurs in MDC requires periodic fires to maintain an open sunny understory with a minimum amount of hardwoods. It tends to occur in areas within open shrub canopy and exposed limestone with minimal organic litter (pine needles, leaves, and other organic materials). Some populations are found at relatively high elevations (3 to 4 m), one occurrence is in a low elevation pine rockland very close to a marl prairie (2 to 3 m). The pine rockland which contains this occurrence may have flooded periodically during the summer wet season. Periodic fires are extremely important in maintaining this ecosystem. The natural fire regime was probably 3 to 7 years, with most fires occurring at the beginning of the wet season in spring and early summer. These periodic fires keep the shrub canopy low and reduce litter accumulation.”

Larry and Penny Thompson Park has the only large population. Based upon data from IRC, Keith Bradley (2007) had estimated 1,001-10,000 individuals at this location. More recently, based upon data from FTBG, Jennifer Possley (2008) had estimated the population size at 1,000-

1,500 individuals, noting that 200 plants were found in a survey covering approximately 10 percent of the Park. Bradley and Gann (1999) indicated that this species rarely occurs in great abundance; most populations are very sparse, containing a low density of plants.

Bradley and Gann (1999) estimated populations using a logarithmic scale. On that scale, the total population of Florida brickell-bush was estimated at 1,001 to 10,000 plants, with the exact number probably between 5,000 and 7,000 plants (Bradley and Gann 1999). Based on the latest available data, the lower range may be closer to approximately 1,550 individuals. Bradley and Gann (1999) also stated the population was probably declining because "private sites where this plant occurs are either not being managed or are being developed. Populations on public lands are also being impacted."

Status and Distribution: Florida brickell-bush is "endemic to MDC on the Miami Rock Ridge. It was historically distributed from central and southern MDC from South Miami (latitude ca. 25º 42.5') to Florida City (latitude 25º 26.0'). This is a range of approximately 22.5 mi along the Miami Rock Ridge. Herbarium specimens have not been studied from the New York Botanical Garden, so the full extent of its historic range is unknown" (Bradley and Gann 1999). Bradley and Gann (1999) provided a list of herbarium specimens and other records for this plant that do not give precise or accurate location information. In these cases, the localities have almost certainly been destroyed because they were located in MDC. Bradley and Gann (1999) indicated this species was extirpated from two privately owned sites (Palms Woodlawn Cemetery, and Sunset Drive and 71 Court) in 1968 and 1992, due to development. Bradley (2007) also confirmed the more recent extirpation of another population at a privately owned site (Turnpike Extension and 93rd Terrace) due to development.

Florida brickell-bush is currently distributed from central and southern MDC from SW 120 Street (latitude ca. 25" 39.4) to Florida City (latitude ca. 25" 26.0), suggesting its historic range has contracted at least 4.8 km (3 mi; more than 13 percent) (Bradley and Gann 1999). At least 9 known populations on private lands have been extirpated including: Sunset Drive and 71 Court (site developed; last observation in 1968); Palms Woodlawn Cemetery (site developed; last observation in 1992); Turnpike Extension and 93rd Terrace (site destroyed; confirmed extirpated in 2007); plus at least 6 of 18 undated occurrences reported by Alan Herndon (Bradley and Gann 1999; Bradley 2007). In addition, several of Herndon's 18 sites experienced impacts to habitat through disturbance or invasion by nonnative plants or dense hardwoods, and Florida brickell-bush may no longer occur at these sites (Bradley and Gann 1999).

The number of extant occurrences of this species is somewhat uncertain due to the lack of complete and recent survey information, which is primarily a function of the number of populations which occur on private lands, making them difficult to survey. In addition, Florida brickell-bush can be extremely difficult to identify when not in flower, making it difficult to confidently determine when a population has been extirpated. The most complete survey which included the species was the 2004–2005 mapping by IRC of natural forest communities (NFCs; pinelands and hardwoods) in MDC outside of ENP. IRC mapped both public and private NFCs where the county government obtained landowner permission or determined it was not necessary. This survey found Florida brickell-bush on six privately owned parcels, including on the UM Richmond Campus (formerly the U.S. Naval Observatory). Surveys of populations on public lands, specifically those owned or managed by the County, occur more commonly and provide a more detailed assessment of the species' status on selected preserves. Florida brickell-bush was not found during a 2-year project intended to survey and map nonnative and rare plants along FDOT right-of-ways within MDC (Gordon *et al.* 2007).

Based on the best available data, we classified those occurrences of Florida brickell-bush which have not been confirmed extirpated as either extant (status confirmed within the last 10 years), possibly extant (reliable data are greater than 10 years but less than 15 years old, habitat is still extant), or unknown/historical (observation does not include sufficient detail and/or data are more than 15 years old, habitat is still extant) (Table 5). Using this classification, populations of Florida brickell-bush are believed to occur on at least 17 (extant or presumed extant) sites, and may possibly occur on up to another 5 (possibly extant) sites although most of these latter sites have been searched in recent years without the species being found. Florida brickell-bush may also occur at three historical sites, but additional information would be needed to confirm at this time. Of the 17 extant occurrences, 9 occur on public conservation lands, 3 occur on private lands managed for conservation, and 5 occur on private lands with unknown management (Table 5). Four of the populations on public conservation lands, including two of the three large (>100 plants) monitored populations, occur adjacent to one another in the Richmond Pineland Complex.

Bradley and Gann (1999) estimated population size using a logarithmic scale. On that scale, the total population of the species in 1999 was estimated at 1,001–10,000 plants (with the exact number probably between 5,000 and 7,000 plants), and was thought to be declining (Bradley and Gann 1999). Since that time, the estimate for the largest population (Larry and Penny Thompson Park, 1,001–10,000 plants in 1999) has decreased to 101–1,000 plants, with adjacent areas (University of Miami, Metrozoo, Martinez Pineland) estimated to hold another 112–1,100 plants combined (Possley 2013b, 2013c). Additional plants are suspected to occur on adjacent privately owned parcels in the Richmond Pineland Complex (Possley 2013a). The only other monitored population estimated to be composed of greater than 100 plants occurs on the Navy Wells Pineland Preserve, located approximately 20 km (12.5 mi) southwest at the southern end of species current range. Another large population was observed on a private parcel situated between Navy Wells and the Richmond Pinelands, however this property has not been surveyed since 2004. Smaller populations occur on pine rockland fragments spread across the landscape, most no more than approximately 3.2 km (2 mi) from their nearest neighboring population – the major exception to this is a 7.2-km (4.5-mi) gap between the populations on Quail Roost Pineland and Camp Owaissa Bauer. Based on the 17 populations considered to be extant, the current total population estimate is between 515 and 4,935 plants, although the actual number of individuals is probably closer to 2,150-3,700. Based on current estimates, the total population of Florida brickell-bush has apparently declined by approximately 50 percent since 1999.

Table 5. Extant and recent (presence still possible) occurrences of Florida brickell-bush.

POPULATION **NFC # if applicable (P-#)**	**OWNERSHIP** *** denotes lands managed for conservation**	**POPULATION RANGE** **No. plants and year if available**
Extant: Regularly monitored populations – Status confirmed within last 5 years		
Navy Wells Pineland Preserve (P-415)	MDC*	101-1,000 (272 in 2009) [1]
Nixon Smiley Pineland Preserve (P-370)	MDC*	28 (2017)[2]
Pine Shore Pineland Preserve (P-48)	MDC*	11-100 (77-118 in 2009) [1]
Quail Roost Pineland (P-144)	State of Florida – managed by MDC*	11-100 (23 in 2011) [1]
Richmond Pinelands Complex – Larry and Penny Thompson Park (P-391)	MDC*	101-1,000 (815 in 2008) [1]
Richmond Pinelands Complex – Miami MetroZoo (P-391)	MDC*	101-1,000 (742 in 2009) [1]
Rockdale Pineland (P-52)	State of Florida – managed by MDC*	1-10 (5 in 2010) [1]
Ron Ehman Park	MDC*	11-100 (31-45 in 2011) [1]
West Biscayne Pineland (P-295)	State of Florida – managed by MDC*	11-100 (15-150 in 2008) [1]
Presumed Extant: Populations not regularly monitored – Status confirmed within last 10 years		
P-132	Private	1-10 [3]
P-295	Private	101-1000 [3]
P-297	Private	11-100 [3]
P-316	Private	11-100 [3]
P-365	Private	11-100 [3]
Pine Ridge Sanctuary (P-310)	Private*	11-100 [4]
Porter Russell Pineland Preserve (P-160)	Private – Tropical Audubon Society*	10-15 [5]
Richmond Pinelands Complex – Martinez Pineland (P-391)	MDC*	Unknown (previously grouped with Larry and Penny Thompson Park)
Richmond Pinelands Complex – University of Miami, Richmond Campus (P-391)	Private – University of Miami*	11-100[3]

Possibly Extant: Habitat extant but status last confirmed 10-15 years ago		
Camp Choee (P-397)	Private*	11-100 [6]
Camp Owaissa Bauer (H-681)	MDC*	11-100 [6]
Panther Pineland (P-338)	Private	11-100 [6]
Seminole Wayside Park (P-365)	MDC*	11-100 [6]
Tamiami Pinelands Complex Addition (P-6.00)	State of Florida – managed by MDC*	10-100 [6]
Unknown/Historical: Habitat extant but records regarding occurrence are limited and/or >15 years old		
Ingram Pineland (P-360)	State of Florida – managed by MDC*	Unknown [7]
Navy Wells #2 (P-329)	MDC – School Board	Unknown [8]

[1] Possley 2013a, 2013b, 2013c; [2]Lange 2017; [3]Bradley and Gann 2005; [4]Glancy 2013; [5]Bradley 2008; [6]Bradley and Gann 1999; [7]Included in a 2005 plant list by IRC, but no estimate provided; [8] FNAI Element Occurrence #7, dated 9/5/1995

Alan Herndon had reported 18 occurrences in an undated report (Bradley and Gann 1999). Six of Herndon's occurrences have been developed and several additional sites have been disturbed or, because of lack of management, the sites are now dominated by exotic plants and/or dense hardwoods (Bradley and Gann 1999). Florida brickell-bush may no longer occur at some of these sites (Bradley and Gann 1999). IRC mapped all of the public and many private pinelands in MDC outside of ENP in 2004. They found no new sites for this plant, other than at the Porter Russell Preserve. Data from IRC from 2007 indicates that 21 other locations have an undetermined status (*i.e.*, the area was surveyed, but the plant was not observed by IRC) (Bradley 2007). Additional survey work at these locations (all private land) would be needed to determine presence. The species was not found during a 2-year project intended to survey and map exotic and rare plants along FDOT right-of-ways within MDC (Gordon *et al.* 2007).

All of the extant sites where this species is known to occur are within MDC including those documented in the Richmond Pine Rocklands (Larry and Penny Thompson Park and Miami Metrozoo). There are also three of five populations where the habitat remains extant but the species status was last confirmed 10 to 15 years ago that are part of the action area in both projects (Camp Owaissa Bauer, Seminole Wayside Park [SWP], and Tamiami Pinelands Complex Addition). The majority of the population is growing on the extant sites referenced above.

Threats: Nearly all of the pine rockland habitat within the narrow range of Florida brickell-bush has been urbanized, converted to agricultural use, or degraded, so that the original low understory has been replaced by hardwoods or exotic plants. Based upon available data, there are 16 extant occurrences of Florida brickell-bush in remnants of its former pine rockland habitat in MDC (Bradley and Gann 1999; Bradley 2007). Only one occurrence of more than 100 individuals is known to exist. Essentially all remaining occurrences are small and isolated. The Service has determined that the threats to Florida brickell-bush consist primarily of habitat loss and modification through urban and agricultural development, fire suppression, proliferation of nonnative invasive plants, and sea level rise. Threats described under habitat loss, fragmentation, and degradation resulting from development, fire suppression, and competition from nonnative invasive plants are believed to be the primary drivers in the historic and recent declines of Florida brickell-bush. This species has also been threatened by anthropogenic disturbances which threaten populations in disturbed habitats such as firebreaks and road rights-of-way, and this taxa is suspected to be negatively affected by threats related to small, isolated populations. All of these threats are expected to continue to impact populations of these taxa in the future. Current local, State, and Federal regulatory mechanisms are inadequate to protect these taxa from taking and habitat loss. Despite the existing regulatory mechanisms, Florida brickell-bush continue to decline.

This species is threatened by habitat loss, which is exacerbated by habitat degradation due to fire suppression, modification of fire regime, the difficulty of applying prescribed fire to pine rocklands, and threats from exotic plants (Bradley and Gann 1999; NatureServe 2012). Remaining habitats are fragmented, and populations which occur on private lands are threatened by development and further fragmentation. Climatic changes, including sea-level rise, are long term threats that will further reduce the extent of habitat. Florida brickell-bush is vulnerable to natural disturbances, such as hurricanes, tropical storms, and storm surges. Due to its restricted range and the small sizes of most isolated occurrences, this species is vulnerable to environmental (catastrophic hurricanes), demographic (potential episodes of poor reproduction), and genetic (potential inbreeding depression) threats.

Florida Bristle Fern (endangered)

Legal Status: The Service listed the Florida bristle fern (*Trichomanes punctatum* ssp. *floridanum*) as endangered on October 6, 2015 (80 FR 60439 60465) under the Act. CH has not been designated but is tentatively scheduled to be proposed late in 2016 or early 2017.

Life history and Population Dynamics: It is a very small, mat-forming fern, superficially resembling some liverwort species.Wunderlin and Hansen (2000) described it as “Stem long-creeping, mat forming, the trichomes (hairlike or bristlelike outgrowth) brownish black, of 2 types, 2-celled glandular and elongate rhizoidlike ones; roots absent. Leaves separated, the petiole 0.1-2 cm long, usually shorter than the blade, pubescent above and below with trichomes like those of the stem but shorter, with stellate (star-shaped) trichomes few and distal on the winged upper part, the blade flabellate (fan-shaped), round, narrowly oblanceolate to nearly linear, entire or irregularly lobed at the apex, 0.5-2 cm long, 0.2-1.1 cm wide, the midrib wanting or less than half the blade length, the apex rounded to obtuse, the base narrowly cuneate (wedge-shaped), the margin entire to irregularly and flabellately lobed, lobes oblong and blunt to obscurely deltoid, frequently resembling proliferous outgrowths distally, with marginal black stellate trichomes, with 2-celled glandular trichomes on the veins, false veins few, the true veins not enlarged at their apex. Involucres (a cup-shaped structure which houses the spore-bearing organs) 1.5-2 mm long, 1-6 at the blade apex, immersed for half or more of their length to fully so, the lips distinct from the blade tissue, inconspicuously dark-margined, the receptacle included or exserted to less than about half the involucre length.”

Florida bristle fern is always associated with shaded limestone outcrops. Plants usually grow on bare limestone, but are occasionally found on tree roots growing on limestone. In MDC, it has been found exclusively in oolitic (composed of minute rounded concretions resembling fish eggs) limestone solution holes and rocky outcrops in rockland hammocks. Solution holes are formed by dissolution of subsurface limestone followed by a collapse above (Snyder *et al.* 1990). Solution holes vary in size, from shallow holes less than 0.5 m (1.6 ft) deep to those that cover over 100 square meters (sq m) (1,076 square feet [sq ft]), and are several meters deep. The bottoms of most solution holes are filled with deep organic soils. Deeper solution holes penetrate the water table and have (at least historically) standing water for part of the year. Humidity levels are higher in and around the solution holes because of standing water and moisture retained in the organic soils.

The canopy cover is typically very dense where Florida bristle fern occurs, and consists of a mix of temperate and tropical hardwood trees including lancewood (*Ocotea coriacea*), pigeon plum (*Coccoloba diversifolia*), live oak, paradise tree (*Simarouba glauca*), strangler fig (*Ficus aurea*), and mastic (*Sideroxylon foetidissimum*) (Bradley 2007). Many tropical, epipetric plant species are associated with solution holes in rockland hammocks. Soils at the MDC sites are classified as Matecumbe Muck (http://www.fgdl.org/). In Sumter County, the plants occur in a mesic/hydric hammock on limestone boulders 1 - 2 m (3.3 - 6.6 ft) tall, under a canopy of live oak, cabbage palm, and American hornbeam (*Carpinus caroliniana*) (Werner 2007). Florida bristle fern grows on boulders with tall, horizontal faces with other rare fern species (*e.g.*, hemlock spleenwort [*Asplenium cristatum*], and widespread polypody [*Pecluma dispersa*]). The hammocks where it has been found are surrounded by a mosaic of wetlands. Soils at the Sumter County station are classified as Mabel Fine Sand, bouldery subsurface (http://www.fgdl.org/).

Little is known about the life history of this taxon, or for members of the genus in general. Like all ferns, Florida bristle fern has two life history stages, a gametophyte stage and a sporophyte stage. All populations that have been reported have been in the sporophyte stage. The initial stage, after a spore germinates, is the gametophyte stage. The gametophyte contains separate sperm and egg producing structures. In the presence of water or moisture, sperm reach the eggs for fertilization. Fertilized eggs, under the proper conditions, develop into sporophytes – the typical form most ferns are observed in. The sporophytes produce spores which in turn can germinate to produce new gametophytes (Nelson 2000). Reproduction may also occur in two other ways. Plants may reproduce by division, when rhizomes break, forming clones of the parent plant. They may also reproduce with the production of gemmae, propagules produced by gametophytes, which can grow into new gametophytes of the same genotype (Hill 2003).

Spores have been recorded in October (Possley 2007), but plants probably produce spores during much of the summer wet season. During the dry season, sporophytes have been observed to desiccate, and probably do not produce spores. For Florida bristle fern, the reproductive requirements, such as moisture levels, needed for each stage of its life history are unknown. Data is needed on longevity, growth rates, recruitment rates, dispersal methods, and genetic variation.

Because Florida bristle fern grows in dense mats and is rhizomatous, it is difficult, if not impossible, to accurately count individual plants. This difficulty has been encountered in other *Trichomanes* species, such as Appalachian bristle fern (*Trichomanes boscianum*) (Hill 2003). In MDC the taxon occurs at four sites in eight solution holes and several smaller holes and rocky outcroppings (Possley 2008, 2011). Possley has estimated that individual colonies cover from 30 square centimeters (sq cm) (4.7 square inches [sq in]) to a maximum of 400 sq cm (62 sq in) on the walls of solution holes. The total area covered by the colonies at the eight solution holes is roughly 1620 sq cm (251.1 sq in). There are probably less than 500 total plants, and many plants may be genetically identical, since new plants can arise from broken rhizomes (Possley 2011). In Sumter County, the single small colony grows on five or six boulders and covers approximately 0.3 sq m (3.0 sq ft) (Werner 2007). There are probably fewer than 1,000 total plants in existence, but this may be a large overestimate of the actual number (Bradley 2007).

Status and Distribution: FNAI considers the State status of the Florida bristle fern to be S1, "critically imperiled in Florida because of extreme rarity (five or fewer occurrences or less than 1000 individuals) or because of extreme vulnerability to extinction due to some natural or man-made factor" (FNAI 2011). NatureServe (2010) gives its global short-term trend as declining with a rounded global status of T1, critically imperiled, due to extreme rarity and threats from drainage, conversion of habitat, and exotic plants. The IRC considers its status as "critically imperiled" (Gann *et al.* 2001-2008). The Florida bristle fern is listed as endangered by the State.

The historical range of Florida bristle fern included southern (MDC) and central (Sumter County) Florida. In MDC it occurred historically in at least 12 hammocks (Castellow, Cox, Fuchs, Hattie Bauer, Meissner, Modello area, Nixon-Lewis, Ross, Royal Palm, Shields, Silver Palm, Snapper Creek area) (Gann *et al.* 2002). The range extended from Royal Palm Hammock (now in ENP) at its southern limit, north to at least Snapper Creek, and possibly further north into the Miami area (Gann *et al.* 2002). This is a range of at least 45 km (28 mi).

John Kunkel Small called attention to the demise of this taxon because of habitat destruction in 1938 (Small 1938). Sites that have been destroyed include a station (study location) near the City of Miami, the Snapper Creek area, a hammock near Modello (in southern MDC near the intersection of US 1 and S.W. 288 Street), Shields Hammock, and a hammock near Longview Camp (between Florida City and ENP). Some other hammocks still exist where the taxon formerly occurred. These include Cox Hammock (privately-owned Monkey Jungle tourist attraction) where it was last seen in 1989, Silver Palm Hammock (preserve owned by MDC) where it was last seen around 1980, Nixon-Lewis Hammock (privately-owned, disturbed, and mostly destroyed) where it was collected in 1915, and Royal Palm Hammock (in ENP) where it was last reported in 1917 or earlier (Gann *et al.* 2002). It has also been reported for the Deering Estate at Cutler and Matheson Hammock Park, both MDC Parks, but these reports were never confirmed (Gann *et al.* 2002).

In Sumter County, Florida bristle fern has been documented to occur only in a small area (Wunderlin and Hansen 2000). All of the known collections are from the vicinity of the town of Wahoo. However, most herbarium label data are imprecise. Essentially all verified collections have been made from the area just north of Wahoo, which is east of the Withlacoochee River. The only known population in Sumter County still occurs in this area and is approximately 2 km (1.2 mi) north of Wahoo.

Two specimens have label data that indicate that the specimens were not collected north of Wahoo, but the label data on both of these are suspect. One specimen in 1963, Lakela #26474 (University of South Florida herbarium), was collected at "Indian Field Ledges west of Withlacoochee River off #48." If this label data are correct, this station was about 6.0 - 6.5 km (3.7 – 4.0 mi) to the west of Wahoo. The statement that it was west of the river may be in error, as Darling (1961) stated that the Indian Field Ledges are north of Wahoo, a locality east of the river. Another specimen collected in 1939 (three years after its discovery in Florida, when it was thought to be *T. sphenoides*) has the label data "south of Floral City, FL. This is the only known station in the United States." It was collected by J.B. McFarlin (Florida State University herbarium). Wahoo is approximately 11.3 km (7.0 mi) southeast of Floral City. The label data may be incorrect and probably refer to the population in the Wahoo area. Because of the new report of the taxon from that area, McFarlin probably collected at the same locality where the taxon was found in 1936 and incorrectly recorded the direction from Floral City as south instead of southeast. The specimen has led to reports of the taxon in Citrus County (Wherry 1964; Nelson 2000).

There are currently five, and possibly six, extant occurrences of Florida bristle fern (Gann *et al.* 2002), four in MDC and two in Sumter County (Table 6). The Sumter County occurrences are approximately 400 km (249 mi) north of those in MDC.

In MDC, Florida bristle fern is known from Meissner Hammock in two solution holes (Bradley 2009), from Fuchs Hammock Preserve in three solution holes, and from Castellow Hammock Park in two large solution holes and several smaller holes and rocky outcroppings (Possley 2008). Fuchs and Meissner Hammocks are immediately adjacent to each other, and Castellow Hammock Park is 10.5 km (6.5 mi) to the northeast. During 2011, eight small patches of Florida bristle fern were re-discovered at Hattie Bauer Hammock. Seven of the these patches occurred within a single solution hole, the eighth patch was found a few meters away from the hole J. Possley (2011). Hattie Bauer Hammock is 2.5 mi south of Castellow Hammock and approximately 5 mi northeast of Fuchs and Meissner Hammocks. In Sumter County, it is known from one colony in the Withlacoochee State Forest's Jumper Creek Tract, north of Wahoo. Another occurrence consisting of two colonies on private land just south of the State Forest may be extirpated.

While no comprehensive status survey has been conducted, rockland hammocks in MDC with suitable habitat have been extensively explored, including sites where it was formerly found. It is unlikely that additional surveys will reveal new occurrences in MDC. However, it is possible Florida bristle fern occurs at some of the hammocks or hammock fragments that remain intact. It is possible three or four hammocks may be sufficiently intact to support the species (Bradley 2009). Attempts to relocate the taxon in Royal Palm Hammock in ENP have not been successful (Gann *et al.* 2006; Sadle 2008a), and additional surveys there are not expected to be successful (Sadle 2008b). It could not be found in surveys of Silver Palm Hammock in the late 1990s and early 2000s (Gann *et al.* 2002). It could not be found in Nixon-Lewis Hammock in 2004, although what remains of the hammock is so disturbed that finding it was extremely unlikely (Bradley 2007). Extensive surveys have not been undertaken at Cox Hammock, and the species may persist there (Bradley 2008).

Also, new locations could be encountered in Sumter County. The soil type of the known occurrence in Sumter County covers 3,652 ac (1,478 ha), and these areas have not been systematically surveyed. In August 2007, a boulder field in the Withlachoochee State Forest's Jumper Creek Tract called the Indian Fields was explored without success (Werner 2007). The hammocks in the vicinity of the known colony have also been searched without finding additional colonies (Werner 2007).

There are currently five, and possibly six, extant occurrences of Florida bristle fern (Gann *et al.* 2002), four in MDC and two in Sumter County (Table 6). Within the action area for the FWC project in MDC, Florida bristle fern is known from Fuchs Hammock Preserve in three solution holes (Possley 2008). While no comprehensive status survey has been conducted, rockland hammocks in MDC with suitable habitat have been extensively explored, including sites where it was formerly found. It is unlikely that additional surveys will reveal new occurrences in MDC. However, it is possible that Florida bristle fern occurs at some of the hammocks or hammock fragments that remain intact. It is possible that three or four hammocks may be sufficiently intact to support the species (Bradley 2009).

Table 6. Summary of known, extant occurrences of Florida bristle fern. Data are from Gann *et al.* (2002), K. Bradley (2009), and J. Possley (2008, 2011).

County	*Location*	*Ownership*	*# of colonies*	*Status*
Miami-Dade	Meissner Hammock[1]	Public	2	Extant
	Fuchs Hammock Preserve[2]	Public	3	Extant
	Castellow Hammock Park[3]	Public	2+	Extant
	Hattie Bauer Hammock[4]	Public	1	Extant
Sumter	Withlacoochee State Forest's Jumper Creek Tract[5]	Public	1	Extant
	Private land south of Jumper Creek Tract[6]	Private	2	Unknown

Threats: Habitat modification and destruction, caused by human population growth and development, agricultural conversion, regional drainage, and canal installation, have impacted the range and abundance of *Trichomanes punctatum* ssp. *floridanum.* Secondary effects from hydrology and canopy changes have resulted in changes in humidity, temperature, and existing water levels; loss of natural vegetation; and habitat fragmentation. The modification and destruction of habitat where *T.p.* ssp. *floridanum* was once found has been extreme.

Florida Pineland Crabgrass (proposed threatened)

Legal Status: Florida pineland crabgrass (*Digitaria pauciflora*) first became a candidate on October 25, 1999, and was proposed to be listed by the Service as threatened on October 11, 2016 (81 FR 70282 70308). No CH has been designated for Florida pineland crabgrass.

Life History and Population Dynamics: Florida pineland crabgrass is a rhizomatous perennial; sheath auricles ca. 1.5 mm long; sheaths hairy (becoming glabrous with age); ligule 1.5 to 2 mm long; leaf blades flexuous or twisted, spreading, 7 to 18 cm long, 1 to 2.2 mm wide, hairy on both surfaces (becoming glabrous with age); main axis of the inflorescence 10 to 80 mm long, primary branches 2 to 8, appressed or spreading from the main axis, ca. 0.3 mm wide; pedicels 2 to 3 mm long, 0.7 to 0.9 mm wide; spikelets 30 to 60 on a primary branch, lanceolate, 2.7 to 3 mm long, 0.7 to 0.9 mm wide; first glume often present; second glume the same length as spikelet, usually 7-nerved, glabrous, acuminate to acute; lemma of lower floret 7-nerved, acuminate to acute, glabrous; upper floret the same length as the lower floret; lemma of the upper floret becoming purple, acuminate to acute (Adapted from Webster and Hatch 1990; Bradley and Gann 1999).

The reproductive biology and ecology has not been studied, but reproduction is sexual (Bradley and Gann 1999). This species fruits in the fall (Wendelberger and Maschinski 2006). The species occurs most commonly along the ecotone between pine rockland and marl prairie habitats, but do overlap somewhat into both of these ecosystems (Bradley and Gann 1999). The soil where it occurred at the Richmond Pine Rocklands has been classified as Biscayne marl, drained (USDA 1996). These habitats, particularly marl prairie, do flood for 1 to several months every year in the wet season. Gann *et al.* (2006) described the major habitat types for Florida pineland crabgrass at Long Pine Key to consist of pineland / prairie ecotones and prairies. Gann *et al.* (2006) indicates this species is associated with low elevation pinelands and pineland / marl prairie ecotones that flood each summer.

<u>Status and Distribution</u>: The historical distribution included central and southern MDC along the Miami Rock Ridge, from the south Miami area (latitude 25° 42.5') to Long Pine Key (latitude 25° 20.5'), a range of approximately 42 mi (67.6 km). J. K. Small and J. J. Carter (No. 916, NY) collected Florida pineland crabgrass in pinelands near the homestead road, between Cutler and Longview Camp, Florida, Nov. 9-12, 1903" (Bradley and Gann 1999). The 1903 Eaton collections from "Jenkins to Everglades" were possibly from the same collecting trip.

Bradley and Gann (1999) stated after a few collections in the beginning of the century, this species seemed to disappear. After a 1936 collection, it was not found again until 1973 in ENP near Osteen Hammock on Long Pine Key (Avery 1983 as cited in Bradley and Gann 1999). Since that time it had been documented many times in Long Pine Key. In 1995, a single plant was discovered in a small marl prairie on the grounds of the Luis Martinez U.S. Army Reserve Center in the Richmond Pine Rocklands in MDC; however, this plant has since disappeared (Herndon 1998; Bradley and Gann 1999). Based on data from IRC, this occurrence was last observed in 1997 and is considered extirpated due to decreased hydroperiod (Bradley 2007; IRC 2009). This species was extirpated from its historical range on the Miami Rock Ridge by drainage and development (FNAI 2007). Prior to its discovery in BCNP in 2003, the range of this species was thought to have contracted by approximately 29 mi (46.7 km) (Bradley and Gann 1999).

Wipff (2004) noted Florida pineland crabgrass is known only from the type collection, which was collected in pinelands of MDC, Florida. Wipff apparently did not have access to more recent collections, although the distribution map cites "reliable reports" from mainland Monroe and Collier Counties. The source of these reports is unknown. Wunderlin and Hansen (2004) report it only from MDC.

Florida pineland crabgrass is currently known from the Long Pine Key area of ENP (Bradley and Gann 1999; Gann *et al.* 2006) and from BCNP (Table 7) (Bradley *et al.* 2013). Citing Avery, Bradley and Gann (1999) indicated that this species occurred in an area of ENP "stretching from near the park entrance (just east of Long Pine Key), southwest to the Mahogany Hammock turnoff at the western end of Long Pine Key", an area of about 31 mi^2 (8,000 hectares [ha]). Prior to research by Gann *et al.* (2006), this species was known from the following locations within Long Pine Key: Hole-in-the Donut, Pine Blocks A, C, D, H. Follow-up surveys of historical locations yielded two additional extant occurrences of this species in the Hole-in-the-Donut (Gann *et al.* 2006). In addition, Jimi Sadle, botanist at ENP, located the species at Pine Blocks SW2, B, and F2 (Sadle 2010). Gann *et al.* (2006) also expect to find new occurrences of Florida pineland crabgrass within ENP as work continues to establish the limits of this species' habitat requirements. Florida pineland crabgrass appears to have a much wider range than previously thought (Gann *et al.* 2006).

In 2003, Keith Bradley (2005) discovered this species south of Loop Road in BCNP in Monroe County. This finding is a significant discovery, since it is the first occurrence of this narrow endemic documented outside of the Miami Rock Ridge / Everglades area (FNAI 2007). Prior to this discovery, the only extant population was on Long Pine Key (FNAI 2007). IRC and FTBG have initiated surveys of the general area around Gum Slough, south of Loop Road (Bradley 2007). Funding became available for a full survey in 2009, and a full survey within BCNP began in 2011 (Bradley 2009). Until this study is complete, the most accurate rangewide estimate is 1,000-10,000 individuals at Long Pine Key (Gann *et al.* 2006) and >10,000 individuals within BCNP (Bradley 2007).There is also some potential for the species to still occur on remaining unsurveyed pine rockland fragments within MDC.

Florida pineland crabgrass appears to have a much wider range than previously thought (Gann *et al.* 2006) and ongoing studies within the action area are expected to find additional populations.

Table 7. Extant occurrences and population estimates of Florida pineland crabgrass (Gann *et al.* 2006; Bradley 2007; Sadle 2010, 2011).

Site	**Owner**	**Population Size**	**Threats**
ENP	NPS	1,000-10,000	hydrologic changes (possible), exotic plants
BCNP	NPS	> 10,000	exotic plants, fire suppression

Threats: Habitat loss continues to occur in this species historical range and most remaining suitable habitat has been negatively altered by human activity. Pine rocklands within MDC have largely been destroyed by residential, commercial, and urban development and agriculture. Pine rocklands in the county (including patches of marl prairie) have been reduced to about 11 percent of their former extent (Kernan and Bradley 1996, p. 2). Of the estimated historical extent of 182,780 ac (74,000 ha), only 20,106 ac (8,140 ha) of pine rocklands remained in 1996. Outside of ENP, only about 1 percent of the Miami Pine Rock Ridge pinelands remain and much of what is left is in small remaining blocks isolated from other natural areas (Herndon 1998, p. 1).

Florida pineland crabgrass habitat at Long Pine Key in ENP (*e.g.*, pineland/prairie ecotones and prairies [Gann *et al.* 2006, p. 12]) and BCNP are, for the most part, protected. The largest and only known populations are, therefore, essentially protected from habitat loss due to development or agriculture. Effects from hydrological changes and other natural and anthropogenic factors, however, may still affect this species.

Fire maintains the pine rockland community. Under natural conditions, lightning fires typically occurred at 3 to 7- year intervals, or more frequently in marl prairies. With fire suppression, hardwoods eventually invade pine rocklands and shade out Florida pineland crabgrass (Bradley and Gann 1999, p. 50). Fire suppression outside of ENP has reduced the size of the areas that do burn and habitat fragmentation has prevented fire from moving across the landscape in a natural way. Thus, many pine rockland communities are becoming tropical hardwood hammocks. While application of prescribed fire is difficult in the urban pine rockland fragments in MDC, it is somewhat easier to apply on larger public conservation lands.

Prescribed fire is actively being used at ENP and now appears to be effective in maintaining populations of Florida pineland crabgrass at this location (Sadle 2010). In 1998, Herndon (1998, p. 91) had reported a sharp decline in the number of plants in one ENP location, which he attributed to prescribed fire followed by flooding caused by Tropical Storm Dennis in 1981. At BCNP, the degree to which fire is currently a factor is not known, as the extent of the species occurrence and habitat has not yet been determined. FNAI (2007 p. 190) had suggested applying regular prescribed fire for this element occurrence in 2007. This implied, at least, that Florida pineland crabgrass within BCNP may need prescribed fire on a more regular basis than is currently occurring. The frequency at which the prairies supporting this species within BCNP burn should be further investigated (J. Sadle 2010). At this time, fire suppression and lack of prescribed fire is a threat, though it may not be as much of a threat as previously believed at some sites.

Florida Prairie Clover (proposed endangered)

Legal Status: Florida prairie-clover (*Dalea carthagenensis* var. *floridana*) first became a candidate on October 25, 1999, and proposed to be listed as endangerd by the Service on October 11, 2016 (81 FR 70282 70308) under the Act. No CH has been designated for Florida prairie-clover.

Life History and Population Dynamics: Florida prairie-clover is a suffrutescent (having a stem that is woody only at the base; somewhat shrubby) shrub 3 to 6 ft (0.5 to 2 m) tall (Bradley and Gann 1999; Chafin 2000). Bradley and Gann (1999) describe it as follows, “Leaflets 15 to 23, ovate to elliptic, 5 to 14 mm long, glandular punctuate beneath; spikes subcaptitate to shortly oblong, 0.5 to 1.5 (-2) cm long, pubescent; peduncles opposite the leaves, terminal or appearing axillary, 1 to 3.5 cm long; bracts shorter than calyx; calyx 5 to 7 mm long, subequal and exceeding the tube, plumose; corolla subpapilionaceous, initially greenish white, turning maroon or dull purple, 4 to 5 mm long; stamens 9 to 10 (Adapted from Isely 1990).”

Although the reproductive biology and ecology of this taxon has not been studied, reproduction is sexual (Bradley and Gann 1999). Research by FTBG has shown that scarification has a positive effect on the germination of this plant’s seeds (Carroll 2005). Both concentrated sulfuric acid and boiling water function equally well as scarifying agents; this information can lead to greater success in propagation and reintroduction efforts (Carroll 2005).

This shrub is found in pine rocklands, edges of rockland hammocks, coastal uplands, and marl prairie (Chafin 2000). Bradley and Gann (1999) suggested fire is probably very important to the livelihood of this taxon. Plants probably do not tolerate shading by hardwoods in the absence of periodic fires. Two of the extirpated occurrences were reported from rockland hammocks (Castellow and Cox). Historically, this species likely occurred at the edges of rockland hammocks and was also known to occur in coastal uplands, at least within Palm Beach County.

In 1999, each of the five occurrences known at that time were located in slightly different habitat types: disturbed pine rockland, pine rockland and rockland hammock ecotone, pine rockland and rockland hammock ecotone along road edges, edge of roadside in marl prairie, and ecotone between rockland hammock and marl prairie and flatwoods (Bradley and Gann 1999). In 2007, Jimi Sadle (National Park Service [NPS], 2007) characterized one occurrence in BCNP at an ecotone between pineland and hammock habitats. Florida prairie-clover occurs in association with South Florida slash pine, live oak (*Quercus virginiana*), gumbo-limbo (*Bursera simaruba*), poisonwood, willow bustic, white stopper (*Eugenia axillaris*), bluestem grasses, and paspalum grasses (*Paspalum* spp.) (Bradley and Gann 1999).

Although Bradley and Gann (1999) estimated the total population (based on a $\log_{10}$ scale) to be 101 to 1,000 plants, they indicated that the total population size is probably closer to 200 to 300 individuals and that the population is probably declining since it has been extirpated on many sites where it once occurred. Updated information for the occurrences at MDC preserves was provided by Joyce Maschinski (2007) for 2007. Maschinski (2007) indicated that 10 woody plants and 4 seedlings occurred at the R. Hardy Matheson Preserve in 2007. Since 2003, the number of woody plants had declined dramatically at this preserve - from 31 to 1 (Possley and Maschinski 2009). Eleven seedlings were found in September 2008 (Possley and Maschinski 2009). Overall, the population at this site performed poorly, likely due to fire suppression for decades (Possley and Maschinski 2009). By 2008, only four plants remained, and only one was large enough to reproduce (Possley 2008). Plants are failing to thrive for unknown reasons, and the population at this preserve is essentially extirpated leading some to speculate that the population would soon be extirpated (Possley 2008). However, the population rebounded to 50 to 200 plants in 2010, apparently as a result of managers raking away pine straw and using a string trimmer (weed-eater) on competing plants in the immediate area (Possley 2011).

<u>Status and Distribution</u>: Florida prairie-clover was historically known from Miami-Dade, Collier, Monroe, and Palm Beach counties (Bradley and Gann 1999). Collections were made in Palm Beach County at an unknown location near Palm Beach by Curtiss in 1895 and south of Palm Beach by Small in 1918. In Monroe County it has been known historically from the Pinecrest region in the BCNP. It was discovered in Collier County portion of the BCNP in 1999 (Bradley and Gann 1999).

In MDC, this plant was reported from many locations, including Key Biscayne, Castellow Hammock, the Charles Deering Estate, R. Hardy Matheson Preserve, the edge of ENP, the Coral Gables area, pinelands south of the Miami River, and Cox Hammock (Bradley and Gann 1999). There have been no reports of this plant from Palm Beach County since 1918 (Bradley and Gann 1999). Gann *et al.* (2002) accounted for essentially every herbarium specimen and reliable sighting. Gann *et al.* (2006) did not find Florida prairie-clover in ENP and it is presumed to be extirpated at this location. Previous records (2) at this location may have represented waif populations established on road fill or disturbed soil (Gann *et al.* 2006).

Based upon Bradley and Gann (1999) and data from the IRC (Bradley 2007), Florida prairie-clover has been extirpated from the sites in Table 8.

Table 8. Extirpated occurrences of Florida prairie-clover.

Site	Owner	County	Last Observation	Cause
Castellow Hammock Environmental Education Center	MDC	Miami-Dade	1975	fire suppression, exotic pest plants
Coral Gables area	Private	Miami-Dade	1967	Development
Cox Hammock	Private	Miami-Dade	1930	development, fire suppression, exotic pest plants
ENP	NPS	Miami-Dade	1964	Unknown
Palm Beach area	Private	Palm Beach	1918	Development

This plant is extant at the sites in Table 9 (Bradley and Gann 1999; data from IRC [Bradley 2007]; data from FNAI (2007, 2011) [Jenkins 2007]; data from NPS [Sadle 2007, 2011]; and data from FTBG [Maschinski 2007; Possley 2008,2009, 2011; Possley and Maschinski 2009; Maschinski *et al.* 2010]).

Table 9. Extant occurrences of Florida prairie-clover.

Site	Owner	County	Occurrence Size	Threats
BCNP, Florida Trail	NPS	Collier	11-100	off-road vehicles, fire suppression, exotic plants
BCNP, 11-Mile Road	NPS	Collier	2-10	fire suppression, exotic plants, Brazilian pepper (*Schinus terebinthifolius*); off road vehicle activity at this location is minimal
BCNP, Pinecrest	NPS	Monroe	11-100	off-road vehicles, fire suppression, exotic plants, changes in mowing practices
Charles Deering Estate, north of Addison Hammock	MDC	Miami-Dade	500 (46 woody plants; 453 seedlings)	fire suppression, exotic plants
Charles Deering Estate, south of Addison Hammock	MDC	Miami-Dade	4 woody plants, 7 seedlings	fire suppression, exotic plants
Virginia Key Beach	City of	Miami-	4	dune erosion,

Site	Owner	County	Occurrence Size	Threats
Park (reintroduction)	Miami	Dade		competition from early successional dune species
Crandon Park	MDC	Miami-Dade	1,000-1,500	fire suppression, encroachment of sea grape (*Coccoloba uvifera*)
R. Hardy Matheson Preserve	MDC	Miami-Dade	50-200	fire suppression, mountain biking, exotic plants, lobate lac scale (*Paratachardina pseudolobata*)
Strawberry Fields Hammock (next to Natural Forest Community)	private	Miami-Dade	2-10	not yet assessed
Florida Power and Light property	Florida Power and Light	Miami-Dade	2-10	not yet assessed

Only nine occurrences of Florida prairie clover remain, seven of which are on conservation lands. There is one additional reintroduced occurrence, consisting of four plants, at Virginia Key Beach Park (Maschinski *et al.* 2010). The species' range is restricted and there are a small number of plants at most sites. Although no population viability analysis has been conducted for this plant, indications are that most existing occurrences are not viable, at least in MDC. As a result, threats associated with small population size are present. These include potential vulnerabilities from environmental (catastrophic hurricanes), demographic (potential episodes of poor reproduction), and genetic (potential inbreeding depression) threats.

Florida prairie-clover is currently known from two occurrences in Collier County, one occurrence in Monroe County, and seven occurrences in MDC. The species is present at the Deering Estate and R. Hardy Matheson Preserve within the action area (Bradley and Gann 1999; data from IRC [Bradley 2007]; data from FNAI [Jenkins 2007]; and data from FTBG [Maschinski 2007]). These two occurrences represent approximately 550 to 700 plants. Overall, the population at the R. Hardy Matheson Preserve was previously declining, likely due to fire suppression for decades (Possley and Maschinski 2009). However, the population rebounded to 50 - 200 plants in 2010, apparently as a result of managers raking away pine straw and using a string trimmer (weed-eater) on competing plants in the immediate area (Possley 2011).

Threats: The MDC pine rocklands have largely been destroyed by residential, commercial, and urban development and agriculture. Pine rocklands in the county (including patches of marl prairie) have been reduced to about 11 percent of their former extent (Kernan and Bradley 1996, p. 2). Of the estimated historical extent of 182,780 ac (74,000 ha), only 20,106 ac (8,140 ha) of pine rocklands remained in 1996. Outside of ENP, only about one percent of the Miami Pine

Rock Ridge pinelands remain and much of what is left is in small remaining blocks isolated from other natural areas (Herndon 1998, p. 1).

Habitat loss continues to occur in the species range and most remaining suitable habitat has been negatively altered by human activity. MDC has developed a network of small public conservation lands and has encouraged conservation of natural vegetation on private land. The County's actions may have averted extirpation of this and other pineland plants. As a result, some opportunities exist to conserve this plant on private land in MDC, but there is little opportunity to acquire more conservation lands. Conservation of privately owned pine rocklands in MDC is largely a matter of County government cooperation with private landowners and the County offers incentives for landowners to maintain their natural forest communities.
Exotic plant taxa have significantly affected pine rocklands. As a result of human activities, at least 277 taxa of exotic plants have invaded pine rocklands throughout South Florida (Service 1999, p. 3-175). Brazilian pepper is a serious threat to Florida prairie-clover (Bradley and Gann 1999, pp. 42-43). Exotic plants threaten nearly all extant occurrences (Table 9) (Bradley and Gann 1999, pp. 43-45; Bradley 2007). Bradley and Gann (1999, pp. 42-43) indicated that the control of exotic plants is an important part of habitat maintenance of pine rocklands.

In a recent study to better understand the location and extent of invasive exotic plants and rare native plants along roadways in Miami-Dade and Monroe Counties, 88 (of 121) total targeted exotic plant species were found (Gordon *et al.* 2007, p. 10). Of the road segments surveyed (16,412), 38 percent (6,264) contained at least one exotic plant; some segments contained more than one species of invasive exotic plant (and as many as 15) (Gordon *et al.* 2007, pp. 10-11). In MDC, the most frequent naturalized invasive exotic plants recorded were Brazilian-pepper, *Tribulus cistoides* (punctureweed), and *Pennisetum purpureum*(napier grass) (Gordon *et al.* 2007, p. 11).

Fire is required to maintain the pine rockland community, and fire suppression threatens Florida prairie-clover at the majority of sites where it is known to exist (Table 9) (Bradley and Gann 1999, p. 45; Bradley 2007). Under natural conditions, lightning fires typically occurred at 3 to 7-year intervals. With fire suppression, hardwoods eventually invade pine rocklands and shade out understory species like Florida prairie-clover. Fire suppression has reduced the size of the areas that burn, and habitat fragmentation has prevented fire from moving across the landscape in a natural way. Thus, many pine rocklands are gradually becoming tropical hardwood hammocks. Natural fires are unlikely to occur or will likely be suppressed in the remaining highly fragmented pine rockland habitat. Establishment of a natural fire regime at all sites where this species occurs is recommended (Bradley and Gann 1999, p. 43; Chafin 2000, NA).

Garber's Spurge (threatened)

Legal Status: The Service listed the Garber's spurge (*Chamaesyce garberi*) as threatened on July 18, 1985 (50 FR 29345 29349) under the Act. No CH has been designated for Garber's spurge.

Life history and Population Dynamics: Garber's spurge is a prostrate to erect herb with pubescent stems. The leaves are ovate in shape and 4 to 9 mm long, with entire or obscurely serrate leaf margins. The cyathia are about 1.5 mm long and borne singly at the leaf axils. The appendages are minute or completely absent. The fruit is a pubescent capsule 1.5 mm wide. The seeds either are smooth or have transverse ridges, but are not wrinkled; this is not, however, a distinctive character for this species.

Reproductive ecology in *Chamaesyce* has been poorly studied, but is known to be highly variable (Ehrenfeld 1976, 1979; Webster 1967). Some species are completely reliant on insects for pollination and seed production while others are self-pollinating. Pollinators may include bees, flies, ants, and wasps (Ehrenfeld 1979). The seed capsules of many *Euphorbiaceae* are explosively dehiscent (spontaneous), ejecting seeds a short distance from the parent plant. Some seeds are dispersed by ants (Pemberton 1988).

Garber's spurge is still found nearly throughout its historical range. It has been extirpated from Collier County and part of MDC. Within its historical range, many stations where it once occurred have been lost. On mainland Florida, Garber's spurge occurs in conservation lands like ENP. It probably occurs on less than half of the islands where it once occurred in the Florida Keys. Some populations are very small and are thus threatened with extirpation due to their small sizes. Examples include Cudjoe Key with 1 plant, Lower Matecumbe Key with 10 to 20 plants, Crocodile Lake National Wildlife Refuge on Key Largo with 10 to 20 plants and Crawl Key with fewer than 10 plants. Two populations are large, with probably over 1 million plants on Cape Sable and over 100,000 plants on Long Pine Key in ENP. There have been insufficient studies to determine long-term population trends on any site. At many sites where Garber's spurge does occur, management is insufficient to ensure long-term persistence of the species.

<u>Status and Distribution</u>: Garber's spurge is endemic to South Florida. It is abundant on Cape Sable, Long Pine Key, and throughout the Keys in small numbers. Historically, it occurred from Perrine, MDC, west to Cape Sable, Monroe County, and to the Sand Keys west of Key West, Monroe County (Small 1933; Long and Lakela 1971).

Garber's spurge is currently known from about 17 populations, including two in MDC, and one at Cape Sable (on two Capes) (ENP) and on 14 islands in the Keys in Monroe County (Bahia Honda Key, Big Torch Key, Boca Grande Key, Crawl Key, Key Largo, Cudjoe Key, Fat Deer Key, Grassy Key, Long Key, Long Point Key, Lower Matecumbe Key, Marquesas Keys, Sugarloaf Key, Summerland Key) (FNAI 2006). Some islands contain more than one colony.

Most (96 percent) known extant populations of Garber's spurge are on publicly owned conservation lands and are protected from further habitat loss. On private property, two particularly significant populations occur in privately owned coastal rock barrens, one on Long Key and another on Crawl Key. Other populations probably exist on private lands but have not been seen due to lack of access and surveys. Several populations occur on public lands that are not considered protected, for example, along the road shoulders on Grassy Key. Because of the species' tendency to grow on disturbed substrates, it is often found in places that are not typically managed for their natural resources.

Pine rocklands in the lower Florida Keys (Keys), now mostly protected in the NKDR, historically contained populations of Garber's spurge, although this does not seem to be its primary habitat in the Keys. It has been collected in pine rockland on Big Pine and No Name Keys, although no populations are currently known from pine rockland habitat in the Keys. This may be due to the lack of a proper fire regime, compounded with an increase in Key deer

(*Odocoileus virginianus clavium*) population sizes and subsequent increases in herbivory. Implementation of prescribed fire in the lower Keys, especially in NKDR, has been a highly contentious issue, with much public opposition. Lack of a proper fire cycle has probably contributed to the dense hardwood and palm understory on islands with pine rockland, and a subsequent reduction in diversity and density of the herb layer, limiting habitat suitability for Garber's spurge.

Garber's spurge is currently known from about 17 populations, including 2 within MDC: Long Pine Key and the Deering Estate at Cutler. Fire suppression historically has been a problem at the Deering Estate at Cutler. With a long interval between fires, populations of Garber's spurge will probably decline due at least in part to increasing hardwood and palm densities and accumulations of leaf litter. At the Deering Estate, DERM (1993) reported a population size of 250-500 plants based on 4 days of searches specifically for this species.Herndon (2002) estimated a population size of 600-6,000 plants. In contrast, J. Possley (2007) estimated that only 100-200 plants were present in 2004. However, neither the Herndon nor Possley estimates were based on thorough surveys. The total rangewide population size has not yet been determined. Numbers of individuals in populations vary widely and some have fewer than 20 plants (*e.g.*, Crawl Key rock barren, Cudjoe Key, Key Largo, Lower Matecumbe Key).Two populations are extremely large. On Northwest Cape Sable (ENP), there may be over 1 million plants (Green *et al.* 2007b). On Long Pine Key (ENP), there may be over 100,000 plants (Green *et al.* 2007a).

Threats: All populations are threatened to a degree by exotic plant invasion. Populations on Long Pine Key are probably the least threatened by exotic plants, because of their isolation and continued management by prescribed fire. Populations in coastal habitats are threatened by invasive plants which constantly colonize via ocean dispersed seeds and can rapidly invade, especially following coastal disturbances such as tropical cyclones.

Fire suppression is a problem at the Deering Estate at Cutler population in MDC. The pine rockland area with Garber's spurge has not burned since 1993. Like all pine rockland fragments in MDC, it has been impossible to maintain a proper fire cycle at this site. This situation is not likely to change in the near future.

Sand Flax (endangered)

Legal Status: The service listed Sand flax (*Linum arenicola*) as endangered on September 29, 2016 (81 FR 66842 66865) under the Act. No CH has been designated for sand flax.

Life history and Population Dynamics: Sand flax is a wiry, yellow-flowered herb (Bradley and Gann 1999; Bradley 2006). Bradley and Gann (1999) state sand flax "is a glabrous perennial herb; stems 1-several from the base, wiry, 35 to 53 cm tall; leaves mostly alternate, linear, 7 to 10 mm long, 0.6 to 1 mm wide, entire or with scattered marginal glands; stipules glandular, reddish; inflorescence a cyme of a few slender, spreading or ascending branches; pedicels 2 mm long or less; sepals lanceolate to ovate with a prominent midrib, 2.4 to 3.2 mm long; petals yellow, obovate, 4.5 to 5.5 mm long; fruit 2.1 to 2.5 mm long, 2 to 2.3 mm diameter, pyriform, dehiscing into ten segments; seeds ovate, 1.2 to 1.4 mm long, 0.7 to 0.8 mm wide. (Adapted from Rogers 1963)". The reproductive ecology and biology of this taxon has not been studied

(Bradley and Gann 1999). No studies have been conducted on the ecology of the species (Bradley 2006).

Sand flax is found in pine rockland, disturbed pine rockland, marl prairie, roadsides on rocky soils, and disturbed areas (Bradley and Gann 1999; Hodges and Bradley 2006). The pine rockland and marl prairie where this species occurs requires periodic wildfires in order to maintain an open, shrub free subcanopy and reduce litter levels (Bradley and Gann 1999). This taxon is currently rare in relatively undisturbed natural areas, with the exception of plants on Big Pine Key and the grounds of an office building on Old Cutler Road in Coral Gables (Bradley and Gann 1999; Hodges and Bradley 2006). Several occurrences are in scarified pine rockland fragments that are dominated by native pine rockland species, but have little or no canopy or subcanopy. One population in MDC occurs entirely on a levee composed of crushed oolitic limestone in the middle of a sawgrass marsh (Bradley and Gann 1999; Hodges and Bradley 2006).

More recently, Hodges and Bradley (2006) found in the Keys sand flax seems to only rarely occur within intact pine rockland, but more frequently adjacent to it. Its persistence on roadsides is not fully understood, but it is possible this species has evolved to occur in this habitat as fire regimes and natural areas were altered and destroyed over the last several hundred years (Hodges and Bradley 2006).

In MDC, Kernan and Bradley (1996) reported six mainland occurrences for sand flax. They estimated that approximately 1,000 plants occurred in MDC, with about 600 at Homestead Air Reserve Base. In 2008, Bradley (2008) estimated that hundreds of plants, possibly thousands, remained at this site, now owned by the MDC Homeless Trust. In 2009, Bradley (2009) estimated that approximately 74,000 sand flax plants occur on the site, with densities ranging as high as 4.5 plants per 10.8 sq ft (per 1.0 sq m). This is the largest known population in Miami-Dade, but a portion of it is threatened by development; the U.S. Army Special Operations Command Center South (SOCSOUTH) seeks to locate permanent headquarters at this site (Department of Defense 2009). Project plans include avoidance of the majority of the population with accompanying protection and management of approximately 60,000 individuals (Service 2011b). However, this project will need to be carefully monitored because impacts would affect the largest known occurrence of the species.

An occurrence called Old Cutler contained 26 percent of the known individuals in MDC, prior to being cleared (Bradley and Gann 1999). As of 1996, there were fewer than 200 plants in the remaining sites on the mainland (Kernan and Bradley 1996). According to Bradley (2006), the population size in 2006 in MDC was unknown. A new occurrence has been confirmed recently in MDC on a tract of land enrolled in the Environmentally Endangered Lands (EEL) program, which is an addition to Camp Owaissa Bauer Pineland (Possley 2011).

In the Florida Keys, neither Dickson (1955) nor Alexander and Dickson (1972) reported the species in their studies. Carlson *et al.* (1993) recorded it at a frequency of 1.3 percent in study plots (0.5 sq m) on Big Pine Key. Ross and Ruiz (1996) found sand flax on only 16 plots across 5 Big Pine Key transects. According to their analysis, sites most likely to support sand flax had a high relative representation of graminoids in the understory, abundant pine regeneration, and high cover of exposed rock (Ross and Ruiz 1996).

More recently, in the first comprehensive study of distribution and abundance in the Keys, Hodges and Bradley (2006) estimated that there were between 101 and 1,000 plants in the Keys outside of Big Pine Key. In a follow-up study, examining the distribution and population size of three pine rockland endemics on Big Pine Key, sand flax was found to be extremely rare, located at only five sample locations throughout the island and at three places not associated with sample locations (Bradley 2006). Bradley (2006) found a total of 33 plants, mostly in the interior of the island away from the coast. In the northern pinelands it was found in 6 of 427 plots (1.4 percent) at a density of 0.07 ± 0.09 plants/plot (Bradley 2006). In the southern pinelands, it was found in 1 of 114 plots (0.9 percent) at a density of .009 ± 0.91 plants/plot (Bradley 2006). The difference in density was significant ($U = 32{,}978.5$, $P = 0.033$). Since sand flax was found at such low densities in so few plots, the mean density had an extremely broad range; 95 percent confidence intervals showed a range from -3,353 to 56,404 individuals (Bradley 2006). All plants were found prior to Hurricane Wilma; sand flax was not found at all in surveys 8 to 9 weeks after the hurricane (Bradley 2006). In 2007, Bradley and Saha (2009) found sand flax in northern plots, but did not find it in any of the southern plots. Additional surveys have not been conducted, so it is not possible to determine if sand flax has recovered.

<u>Status and Distribution</u>: Sand flax historically was distributed in Monroe County in the lower Keys and in central and southern MDC (Bradley and Gann 1999). In Miami-Dade, the plant was widespread from the Coconut Grove area to southern MDC, close to what is now the main entrance to ENP and Turkey Point (Bradley and Gann 1999). In Monroe County, the plant was recorded from Big Pine Key, Ramrod Key, Sugarloaf Key, Park Key, Boca Chica Key, and Middle Torch Key (Bradley and Gann 1999). Based upon Bradley and Gann (1999), Hodges and Bradley (2006), and data from IRC (K. Bradley 2007), sand flax has been extirpated from the sites in Table 10.

Table 10. Extirpated occurrences of sand flax.

Site	Owner	County	Last Observation	Cause
Boca Chica Key	Department of Defense	Monroe	1912	unknown, probably development
Middle Torch Key	unknown	Monroe	1979	unknown
Park Key	unknown	Monroe	1961	unknown, probably development
Ramrod Key	unknown	Monroe	1979	unknown
Allapatah Linum Site	private	Miami-Dade	1996	land clearing
Camp Jackson Area	unknown	Miami-Dade	1907	unknown
Camp Owaissa Bauer	MDC	Miami-Dade	1983	fire suppression
Cemetery Pineland	private	Miami-Dade	1996	property scarified, may regenerate

East of Naranja	unknown	Miami-Dade	1907	unknown
Homestead to Camp Jackson	unknown	Miami-Dade	1907	unknown
Homestead to Big Hammock Prairie	unknown	Miami-Dade	1911	unknown

Sand flax is currently known from four occurrences in the Keys and eight occurrences in MDC (Bradley 2006; Bradley 2007, 2011; Maschinski 2007, 2011; Possley 2011). Based upon Bradley and Gann (1999), Hodges and Bradley (2006), Bradley (2009), data from IRC (Bradley 2007; Gann *et al.* 2001-2010), data from FTBG (Maschinski *et al.* 2002; J. Maschinski 2007; Possley 2011; J. Maschinski 2011) and Bradley and Saha (2009), sand flax is extant at the sites in Table 11. On Big Pine Key, sand flax occurs at the Terrestris Preserve, which is owned by The Nature Conservancy (TNC); this occurrence is included within the Big Pine Key site in Table 11.

Table 11. Extant occurrences of sand flax.

Site	Owner	County	Population Size	Threats (site specific only)
Big Pine Key (primarily conservation lands)	NKDR, TNC, other public and private entities	Monroe	2,676	development, fire suppression, exotic plants
Lower Sugarloaf Key	Florida Department of Transportation (FDOT)	Monroe	101-1,000	road clearing or other maintenance, illegal dumping, exotic plants
Big Torch Key	Monroe County Department of Transportation	Monroe	11-100	road clearing or other maintenance, exotic plants
Middle Torch Key	Monroe County Department of Transportation	Monroe	2-10	road clearing or other maintenance, exotic plants
Village of Palmetto Bay	private	Miami-Dade	11-100	development, fire suppression, exotic plants
Cocoplum Development	private	Miami-Dade	11-100	development
Country Ridge Estates/ Camp Owaissa Bauer (partial conservation lands)	private / MDC	Miami-Dade	11-100	development, herbicide
Homestead Air Reserve Base and adjacent land	MDC Homeless Trust	Miami-Dade	74,000	development; proposed military facilities and operations
Homestead Bayfront Park	MDC	Miami-Dade	101-1,000	road clearing or other maintenance

(conservation lands)				
IRC Preserve and adjacent canal bank (primarily conservation lands)	IRC and South Florida Water Management District	Miami-Dade	2-10	herbicide application on canal bank
Luis B. Martinez U.S. Army Reserve Station, Richmond Pine Rocklands	U.S. Army	Miami-Dade	30-50	not assessed
Camp Owaissa Bauer Pineland Addition #1 (conservation lands)	MDC	Miami-Dade	1-10	not assessed

Hodges and Bradley (2006) initiated population surveys for sand flax in the Keys on Big Pine Key and other keys with potential habitat. The survey included extant occurrences, historic sites, and exploratory surveys of potential habitat. This project provided the first comprehensive survey of distribution and abundance for the area. Negative survey results (*i.e.*, location surveyed, but sand flax absent) included: Boca Chica Key (southern edge), No Name Key (roadside edges and NKDR), Ramrod Key (Dan Austin Site), roadsides from Little Torch Key to Lower Sugarloaf Key, and Upper Sugarloaf Key (NKDR) (Hodges and Bradley 2006).

In 2009, an assessment of rare plants and pine rockland habitat was conducted for the proposed SOCSOUTH headquarters at the site adjacent to the Homestead Air Reserve Base (Bradley 2009). During a survey of the 90-ac (36.4-ha) tract, Small's milkpea and sand flax were found in 27 different locations covering 13.2 ac (5.3 ha) in disturbed pine rocklands (Bradley 2009).

Sand flax is currently known from four occurrences in the Keys and six occurrences in MDC (Bradley 2006; Bradley 2007, 2011).In 1996, the species' mainland range was from just north of SW 184 Street south to SW 288 Street and west to SW 264 Street and 177 Avenue; a distance of approximately 11.5 mi (18.5 km) northeast to southwest (Kernan and Bradley 1996). The geographic range on the mainland has contracted approximately 61 percent (Kernan and Bradley 1996). This species is present at several sites proximal to the action area (Bradley and Gann 1999; Hodges and Bradley 2006; Bradley 2007), including the nearby Richmond Pine Rocklands (Luis B. Martinez U.S. Army Reserve Station), Camp Owaissa Bauer, and Owaissa Bauer Addition. These are all populations of 100 or fewer plants, compared to the population at the Homestead Air Reserve Base (HARB) and adjacent land in MDC which is estimated to be approximately 74,000 plants.

<u>Threats</u>: Less than 2 percent of the original acreage of pine rockland habitat remains (Bradley and Possley 2002). Most of that habitat occurs in small, isolated stands in an urban landscape that are difficult to protect and manage. Habitat fragmentation reduces the size of plant populations and increases spatial isolation of remnants. Many of the fragments are overgrown and in need of restoration. The known sites where sand flax occurs on public lands are protected from development, but these sites must be managed to prevent habitat degradation and potential loss of plants. Approximately 25 percent of extant *Linum arenicola* occurrences are located on private land where increased pressure from development threaten to extirpate those plants (Service 2012g).

One of the primary threats to *Linum arenicola* is habitat modification and degradation through inadequate fire management, which includes both the lack of prescribed fire and suppression of natural fires. Where the term ‘‘firesuppressed’’ is used below, it describes degraded pine rocklands conditions resulting from a lack of adequate fire (natural or prescribed) in the landscape. Historically, frequent (approximately twice per decade), lightning-induced fires were a vital component in maintaining native vegetation and ecosystem functioning within South Florida pine rocklands. A period of just 10 years without fire may result in a marked decrease in the number of herbaceous species due to the effects of shading and litter accumulation (FNAI 2010). Exclusion of fire for approximately 25 years will likely result in gradual hammock development over that time period, leaving a system that is very fire-resistant if additional pre-fire management (*e.g.*, mechanical hardwood removal) is not undertaken. Today, natural fires are unlikely to occur or are likely to be suppressed in the remaining, highly fragmented pine rocklands habitat. The suppression of natural fires has reduced the size of the areas that burn, and habitat fragmentation has prevented fire from moving across the landscape in a natural way. Without fire, successional climax from pine rocklands to rockland hammock is rapid, and displacement of native species by invasive, nonnative plants often occurs.

Small’s Milkpea (endangered)

Legal Status: The Service listed the Small’s milkpea (*Galactia smallii*) as endangered on July 18, 1985 (50 FR 29345) under the Act. No CH has been designated for Small’s milkpea.

Life history and Population Dynamics: Small’s milkpea is a perennial herb with numerous trailing stems radiating from large woody taproots and with relatively large flowers (calyx 6 to 8 mm [0.2 to 0.3 in] long, standard and keel 1 to 1.5 cm [0.4 to 0.6 in] long) (Herndon 1981). This species has compound leaves, usually with 3 elliptic leaflets 1.5 to 3 cm (0.6 to 1.2 in) long. The stem pubescence is ascending or spreading-sericeous, and upper leaf surface is puberulent (hairs 0.1 to 0.2 mm [0.004 to 0.008 inch] long; hairs on stem less than 0.5 mm (0.02 in) long) (Herndon 1981).

There is limited knowledge about the demographic features and trends of this plant. Small’s milkpea is a perennial legume and, therefore, probably experiences little annual variation in population size (Fisher 2000; Bradley and Possley 2002). This species does not experience seasonal dieback and is thought to be long-lived, as most of the plants used in a pollination study survived over a period of 5 years (Bradley and Possley 2002). Flowering occurs throughout the year but most abundantly during the dry season. Because most flowers do not produce fruit, it may be self-incompatible (Bradley and Possley 2002). Once pollinated, seeds take several months to mature and often germinate in response to fire. Annual variability in flowering, seed production, seed viability, and establishment requirements are unknown (Bradley and Possley 2002). FTBG is conducting propagation trials in order to expand the ex situ collection of this species. Because of the small size of seeds, seed storage has been difficult (Maschinski 2005).

Small’s milkpea prefers open sun and little shade and can be threatened by shading from hardwoods and displacement by invasive exotic species in the absence of periodic fires. Disturbance, such as prescribed fire, is a necessary management tool to maintain suitable habitat for the species. Habitat degradation on these sites continues to be a moderate threat because vegetation restoration and management programs are costly and depend upon availability of funding.

O'Brien (1998) located the species on 10 sites. In 2002, FTBG reported this species occurred on fewer than 12 sites located in a 6.5-mi (10.5-km) area (Bradley and Possley 2002). The total population at that time was estimated to be less than 10,000 plants and ranged from 3 to over 1,000 individuals per site, with only two sites that contained over 1,000 plants (Bradley and Possley 2002). The most recent comprehensive survey of pine rocklands documented the presence of Small's milkpea on five public sites but did not determine population sizes (IRC 2006). These sites have been purchased by MDC for conservation purposes.
The County is working to restore and manage these lands.

Status and Distribution: When this species was listed, it was known from two sites near Homestead in MDC. In a study of distribution and habitat preference of two plant genera native to South Florida pine rocklands, Small's milkpea was found in the Redland region and a few sites at the southern end of the Biscayne region (O'Brien 1998). The distribution of this species is correlated with soil depth and color in Redland pine rocklands. Small's milkpea appears to prefer calcareous soils with less quartz sands, but not at low elevations, and does not occur in pine forests off of the limestone rock ridge (O'Brien 1998). As elevation decreases southward along the Miami Rock Ridge, so does quartz sand (Bradley and Possley 2002). Preferred soils are mapped as Cardsound Rock outcrop complex and are porous and well-drained (Bradley and Possley 2002). The elevation where the plants occur generally ranges from 7 to 10 ft (2 to 3 m) with a smooth slope from 0 to 2 percent (Bradley and Possley 2002).

The distribution of this plant is fragmented. One study noted several sites had large numbers of plants distributed throughout each site with no well-defined population clusters (Fisher 2000). In 2002, this species occurred in less than 12 fragmented sites located along a 6.5-mi (10.5-km) portion of the ridge (Bradley and Possley 2002). The total population at that time was estimated to be less than 10,000 plants and ranged from 3 to 1,000 individuals per site, with only 2 sites that contained over 1,000 plants (Bradley and Possley 2002). Results of a project to map extant pine rockland habitat indicated that the plants remained on 7 public and 15 private sites (IRC 2006; Bradley 2010a). MDC owns six of the public sites, purchased for conservation purposes, and is working to restore and manage these lands through their EEL program. The remaining public site is owned by the County's Board of Education (Bradley 2010b) and is, therefore, subject to future development. However, the EEL program is currently attempting to acquire this site (Guerra 2010).

In 2009, a large population containing as many 100,000 individuals was documented on an additional public property (County owned) adjacent to the HARB (Bradley 2009). Although HARB is seeking to develop these lands, they are also coordinating with the Service and IRC to retain and manage the plant at this site. Therefore, the most current assessment of NFCs in MDC recorded the species on eight public sites (IRC 2006; Bradley 2009, 2010a). Also in 2009, an additional small population was discovered on the private Palms Woodlawn Cemetery along Old Dixie Highway in Homestead (Bradley 2010b). Because this species has no apparent mechanism for long-distance dispersal of seeds, it is presumed that these fragmented populations are relicts of larger populations prior to fragmentation (O'Brien 1998). Not much is known about how fragmentation has impacted the population dynamics of the species, but most likely populations have become isolated and more imperiled (O'Brien 2006 in litt.).

Small's milkpea is found in the Redland region and a few sites at the southern end of the Biscayne region in MDC (O'Brien 1998). The most current assessment of NFCs in MDC recorded the species on 8 public sites and 15 private sites (IRC 2006; Bradley 2009, 2010a). Within MDC, proximal to the project area, it is found on Navy Wells (data unavailable), Ingram Pineland (11-100 plants), SWP (data unavailable), Palm Drive Pineland (11-100 plants), Sunny Palms Pineland (100-1,000 plants), and Rock Pit #39 (11-100 plants). FNAI reported an occurrence of this species on August 14, 1991 within the Larry and Penny Thompson Park, also within the Richmond Area.

Threats: Less than 2 percent of the original acreage of pine rockland habitat remains (Bradley and Possley 2002). Most of that habitat occurs in small, isolated stands in an urban landscape that are difficult to protect and manage. Many of the fragments are overgrown and in need of restoration. The known sites where Small's milkpea occurs on public lands are protected from development, but these sites must be managed to prevent habitat degradation and potential loss of plants. Privately-owned sites remain at risk of being developed and management remains a concern.

Limited distribution renders the species vulnerable to random natural or human induced events, such as hurricanes and encroachment of invasive exotic species. All of the populations require active management, including exotic plant control, thinning of overgrown vegetation, and/or prescribed fire. The current number of individuals in wild populations is not known, therefore, trend analysis is not available. Although some demographic information is available, additional long-term research will be necessary to develop accurate population models.

There is an ongoing effort to conduct prescribed burns at the publicly-owned sites. Management of these small preserves is difficult because exotic plants are present within and near the properties. Habitat degradation on these sites continues to be a moderate threat because vegetation restoration and management programs are costly and depend upon availability of funding. Continued habitat loss and fragmentation, fire suppression, and invasion by exotic plant species threaten the existence of Small's milkpea (Service 2007a).

Tiny Polygala (endangered)

Legal Status: The Service listed the tiny polygala (*Polygala smallii*) as endangered on July 18, 1985 (50 FR 29345) under the Act. No CH has been designated for tiny polygala.

Life History and Population Dynamics: Tiny polygala is 1 of 9 species of *Polygala* native to MDC and 1 of 11 from Palm Beach County (Wunderlin and Hansen 2004). The most similar species is candyroot (*Polygala nana*) (Bradley and Gann 1995), which is distributed through much of Florida. Bradley and Gann (1995) found existing identification keys were inadequate, but the two species could be distinguished by seed size. The seed body length (not including the rostrum) of tiny polygala is between 1.2 and 1.4 mm; the length for candyroot is between 0.6 and

0.8 mm (Bradley and Gann 1995). Bradley and Gann (1995) found both species occur at the Jupiter Ridge Natural Area in Palm Beach County, and the distribution maps in Wunderlin and Hanson (2004) show the distribution of candyroot extending south to Broward County.

The life span of tiny polygala is short, averaging only 180 days, with only 9 percent of wild plants living beyond 1 year (Koptur *et al.* 1998). Plants typically appear, flower, and then disappear until the next fire or other suitable disturbance. Tiny polygala produces a seed bank that persists within the soil for at least 2 years (Kennedy 1998). Seedling emergence peaks from September-November, but a few seedlings emerge from May-June. Seed germination experiments have been conducted in the field, but few demographic studies have been initiated (Wendelberger and Frances 2004). Kennedy (1998) found *ex situ* seeds germinated within 3 weeks, and 80-100 percent of older, buried seeds germinated regardless of seasonal photoperiod (Koptur *et al.* 1998). Seeds buried to a depth of 1 cm for over 2 years had a high viability rate, suggesting seeds may persist for 10 years or more when slightly buried (Kennedy 2006 in litt.). It is, therefore, important to manage not only for above-ground plants, but for the conservation of the seed bank.

Because seeds may remain dormant in the soil until fire disturbs the site, abundance and population trends for this species are difficult to assess. Koptur *et al.* (1998) suggested that fire is a requirement for seed germination, because fresh seeds collected from the wild exhibited a 50 percent greater germination rate following soaking in a smoke extract. Fellows (2002) repeated the experiment and found that initial germination rates of seeds treated with smoke extract averaged a rate that was 4.3 days faster than non-smoke treated seeds. Total percent germination was similar. Due to fragmentation of populations and the short generation time of tiny polygala, Wendelberger and Frances (2004) believe that the species may experience low genetic diversity. Current knowledge of this species' life history is presented in the Conservation Action Plan (Wendelberger and Frances 2004).

<u>Status and Distribution</u>: When tiny polygala was listed, it was known from sandy pine rockland and Florida scrub vegetation in Miami-Dade and Broward Counties (the Miami and Fort Lauderdale metro areas, respectively). A survey of 56 sites between Broward and Indian River Counties extended its known range into northern Palm Beach and south-central Martin Counties, but only at a few sites (Bradley and Gann 1995). Later, Bradley *et al.* (1999) conducted an endangered plant survey in Florida scrub vegetation in Martin, St. Lucie, and Indian River Counties, covering 25 properties. They found no new populations. Surveys for rare plants in Brevard County did not find tiny polygala (Kennedy 2003a, 2003b, 2004), although this was not a target species and may have been missed. In 2004, thirteen sites contained approximately 22 populations in Miami-Dade, Broward, Palm Beach, and Martin Counties grouped into four population clusters, with the highest density of populations located in southern MDC (Wendelberger and Frances 2004). Clusters of populations are separated by an average of 38 mi, and the distribution of this plant remains fragmented. The overall number of plants is estimated at approximately 11,000, with the majority of these occurring on a single site in MDC (Maschinski 2010).

There have been no new finds of tiny polygala since 1995, despite surveys of possible scrub sites (Bradley and Gann 1995; Bradley *et al.* 1999; Woodmansee *et al.* 2007; Maschinski *et al.* 2008; FNAI 2010), as well as a project to map the pinelands of MDC (IRC 2006). The species is currently known from four sites in MDC (Maschinski *et al.* 2008; Maschinski 2010), two sites in Palm Beach County, and single occurrences in Martin and St. Lucie Counties (Bradley and Gann 1995; Walesky 2005; Woodmansee *et al.* 2007; FNAI 2010). Seven of eight known occurrences are on publicly owned lands, and all these sites are currently being managed for conservation of tiny polygala.

During 2008, FTBG conducted surveys for the species at all known sites within MDC (Maschinski *et al.* 2008; Maschinski 2010). The four known sites where it remains include the publicly owned Miami Metrozoo and adjacent U.S. Coast Guard property, both located within the 2,100-ac Richmond pinelands (Maschinski *et al.* 2008; Maschinski 2010). The Coast Guard site contains the largest population of plants, which was estimated at over 10,000 plants during a 2008 survey (Maschinski *et al.* 2008; Maschinski 2010). The species was also reported from the Deering Estate at Cutler (441 ac) and the Pine Shore Pineland Preserve (Pine Shore Park) (8 ac) (Maschinski 2005 in litt; Maschinski *et al.* 2008; Maschinski 2010; FNAI 2010). This survey failed to locate the plant at two previously documented sites, the County owned Ludlam pineland and the adjoining Florida Power and Light Company easement (Maschinski 2005 in litt.; IRC 2006; Maschinski *et al.* 2008; Maschinski 2010; FNAI 2010), suggesting the species may be extirpated from these sites. The survey also did not report finding the species at former sites on University of Miami and Air Force lands, both occurring within the Richmond pinelands (Maschinski *et al.* 2008; Maschinski 2010). However, Woodmansee *et al.* (2007) indicate tiny polygala occurrences appear to be cyclic, suggesting historical occurrences, if given appropriate management, may reappear.

In Broward County, tiny polygala was known to occur only at one site, the 16.5-ac Gopher Tortoise Preserve at Fort Lauderdale Executive Airport, managed by the City of Fort Lauderdale (FNAI 2010; Maschinski 2010). This site was surveyed in 2002 and no plants were found (Possley 2006 in litt.), but it is presumed that seeds remain in dormancy. However, Woodmansee *et al.* (2007) also failed to locate the plant at this site during 2006 surveys and suggested that drought conditions, exotic plants, and lack of fire may have hindered this population. The nearly adjoining Cypress Creek Scrub Preserve (8 ac), also managed by the City (FNAI 2010), has not been surveyed for tiny polygala (Possley 2006 in litt; Maschinski 2006 in litt.), but plants may occur there.

Palm Beach County's Department of Environmental Resources Management (Walesky 2005 in litt.) reports that tiny polygala is found in two locations in the County. Walesky (2005 in litt.) indicates all of the locations are characterized by open patches of white sand with a ground water table that is relatively near the surface. At Jupiter Ridge Natural Area (269 ac), which had 100 plants when discovered by Gann in 1994, there were 12 plants in 2004 and 86 in August 2005. County biologists attribute the increased population in 2005 to the opening up of the site's dry hammock (hardwood forest) from hurricane activity and above-normal spring and summer rainfall (Walesky 2005 in litt.; Woodmansee *et al.* 2007). Further surveys by Woodmansee *et al.* (2007) found smaller densities in 2006 and noted the species abundance at the site fluctuates dramatically from year to year. Tiny polygala was also discovered at Limestone Creek Natural Area in 2002. A survey conducted in July 2003 recorded 13 plants (Walesky 2005 in litt.).Since

2006, the number of plants recorded at this site has ranged from 3 to 60, with 26 encountered during April 2010 (Woodmansee *et al.* 2007; Shearer 2010). Walesky (2005 in litt.) indicated the County's oceanfront Diamondhead/Radnor Future Park Site (154 ac), discovered in 2001, maintained a population of about 50 plants. However, further surveys at this site determined that the plants reported from this site were candyroot, the closest congener of tiny polygala (Woodmansee *et al.* 2007; Bradley 2010).

In southern Martin County, tiny polygala is known to occur in Jonathan Dickinson State Park (JDSP) (17,314 ac). Surveys of the site conducted from 2000 to 2008 have recorded varying numbers of plants (Woodmansee *et al.* 2007; FNAI 2010). Woodmansee *et al.* (2007) indicated that while the species appears to be in decline at JDSP, it is expected plant numbers will increase in the long run, provided fires are administered. In St. Lucie County, the species was determined to occur at the privately owned Lynn University, based on a specimen collected in 1984 (Bradley and Gann 1995). Woodmansee *et al.* (2007) located 14 plants at this site in 2006, further noting the site had recently been burned and that exotics were being managed. Bradley and Gann (1995) documented the species at the Lynngate portion of Savanna Preserve State Park, also in St. Lucie County. However, Woodmansee *et al.* (2007) reported no plants during a 2006 survey and indicated fire suppression over time was the most likely cause for the plants' disappearance from this site.

The species current range encompasses areas in four counties, including MDC (Maschinski *et al.* 2008; Maschinski 2010).Within MDC tiny polygala occurs entirely on protected sites, several of which are managed by MDC and are sites proposed for habitat management associated with the FWC project. The largest site (not the largest population) is the publicly-owned pineland at The Deering Estate at Cutler, followed by the Miami Metrozoo portion of the County-owned pineland at Richmond and the adjacent U.S. Coast Guard property at Richmond. The Coast Guard property, adjacent to CRC, contains the largest population of these plants (Maschinski *et al.* 2008; Maschinski 2010).

<u>Threats</u>: Fire suppression and invasion by exotic plant species continue to threaten tiny polygala. Management of pine rocklands in MDC is problematic because most of the remaining habitat occurs in small, fragmented areas surrounded by residential or disturbed areas. These environments are often a source of exotic plants. The small size of the pine rockland fragments, in particular the high perimeter to area ratio, make it easier for exotics to invade (Service 1999). Exotic plants have detrimental impacts on pine rocklands. At least 277 taxa of exotic plants are now known from pine rocklands in South Florida (Service 1999).

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