(1999), Nott *et al.* (1998), Curnutt *et al.* (1998), Pimm (1997) and others, and are not predicated in any way on the Van Lent *et al.* (1999) hydrologic modeling.

The Service has determined the following species may occur within the action area:

1. Cape Sable seaside sparrow - (E)(CH)	(*Ammodramus maritimus mirabilis*)
2. Snail kite - (E)(CH)	(*Rostrhamus sociabilis plumbeus*)
3. Wood stork - (E)	(*Mycteria americana*)
4. American crocodile - (E)(CH)	(*Crocodylus acutus*)
5. West Indian manatee - (E)(CH)	(*Trichechus manatus*)
6. Florida panther - (E)	(*Felis concolor coryi*)
7. Red-cockaded woodpecker - (E)	(*Picoides borealis*)
8. Bald eagle - (T)	(*Haliaeetus leucocephalus*)
9. Eastern indigo snake - (T)	(*Drymarchon corais couperi*)
10. Garber's spurge - (T)	(*Euphorbia garberi*)

(E) = federally listed as endangered
(T) = federally listed as threatened
(CH) = federally designated critical habitat

A summary of the status of the species listed above across their entire range, as well as the biological and ecological information relevant to our analysis of effects is provided below. The analysis of the species and critical habitat likely to be affected is also provided. This analysis will be presented in more detail later in the *Effects of the Action* section of this biological opinion. Detailed information regarding the status of the above-mentioned species along with the biological and ecological information utilized by the Service in evaluating potential adverse effects can be found in the Technical Agency Draft of Volume I of the Multi-Species Recovery Plan for the Threatened and Endangered Species of South Florida (U.S. Fish and Wildlife Service 1998). That document is incorporated here by reference.

Cape Sable Seaside Sparrow

The Cape Sable seaside sparrow was listed as an endangered species on March 11, 1967, pursuant to the Endangered Species Preservation Act of 1966 (32 FR 4001). That protection was continued under the Endangered Species Conservation Act of 1969 and the Endangered Species Act of 1973, as amended. The Cape Sable seaside sparrow was listed because of its limited distribution and threats to its habitat posed by large-scale conversion of land in southern Florida to agricultural uses. Critical habitat for the Cape Sable seaside sparrow was designated on August 11, 1977 (42 FR 40685).

A. Distribution

The eight surviving subspecies of seaside sparrow are distributed along the east coast of the United States, from Massachusetts to southern Florida, and along the Gulf coast, from southeast Texas to the west coast of Florida. The distribution of the Cape Sable seaside sparrow is now limited to the Everglades region of Dade and Monroe Counties in South Florida (Figure 3). They are non-migratory and are isolated from other breeding populations of seaside sparrows.

When first discovered on Cape Sable in Monroe County, the sparrows were utilizing freshwater and brackish water marshes across the area east of the mangrove zone in Carnestown to Shark Valley and Taylor Sloughs. The original range most likely included all suitable habitat in south and southwestern Florida (Werner 1978), and extended from Cape Sable (south) to Ochopee (northwest), and east to Taylor Slough and the east Everglades.

The historical distribution of the Cape Sable seaside sparrow included areas that periodically experienced extensive flooding, fires and hurricanes; the sparrow probably adapted to these natural disturbances by varying their distribution within their range as habitat suitability changed. Since its discovery in the early 1900's, the Cape Sable seaside sparrow has been episodically extirpated from portions of its total range. For example, Howell (1919) found the Cape Sable seaside sparrow to be "moderately numerous" on Cape Sable when he first discovered them in 1918. The Great Labor Day Hurricane of 1935 initiated vegetative changes in the Cape Sable area that were later responsible for extirpating the sparrow population. In 1970, Werner discovered Cape Sable seaside sparrows in three cordgrass marshes on Cape Sable. In 1979, fires on Cape Sable appear to have extirpated the sparrow population again; as no sparrows were noted in surveys conducted on Cape Sable in 1979, 1980, or 1981 (U.S. Fish and Wildlife Service 1983). By 1983, the stands of *Spartina*-dominated vegetation that once covered extensive areas of Cape Sable were gone (Werner and Woolfenden 1983) along with the Cape Sable seaside sparrow.

Cape Sable seaside sparrows were first documented in the Big Cypress basin in 1928 by Nicholson. They appeared to flourish there in the 1950s (Stimson 1956), but had been extirpated as a result of widespread frequent fires by the time surveys were conducted in the early 1960s (Stimson 1968). In the early 1970s, they were rediscovered in the Big Cypress area (Kushlan and Bass 1983, Werner and Woolfenden 1983), but were considered rare.

Similar changes in distribution and abundance were noted in other subpopulations. Cape Sable seaside sparrows were initially located in Ochopee by Anderson (1942), but few birds were found in the Ochopee area between the mid-1970's and the 1990's. Pimm *et al.* (1994) located a small number of seaside sparrows there in 1993, but they were not found in 1994 (although the area may not have been adequately surveyed due to logistical problems). The decline of the Cape Sable seaside sparrow subpopulation in this area has been attributed to fires and salinity changes associated with altered hydrology (U.S. Fish and Wildlife Service 1983).

Cape Sable seaside sparrow surveys in 1974, 1975, and 1978-81 indicate the Taylor Slough area supported one of the largest and most viable populations of the sparrow that were known at the time (Bass and Kushlan 1982, Werner 1978). Both studies reported that the extent of suitable habitat in the Taylor Slough area was decreasing because of invasion by exotic trees and shrubby vegetation. These habitat changes may explain why later studies reported the Taylor Slough population as a peripheral population (Curnutt and Pimm 1993, Pimm *et al.* 1994).

Curnutt and Pimm (1993) identified six subpopulations (subpopulations A-F) of the Cape Sable seaside sparrow across their range (Figure 3), with their distribution fluctuating through time, noting that four of the six subpopulations show signs of wide population variation and probably underwent periodic, local extirpations.

B. Habitat

The preferred nesting habitat of Cape Sable seaside sparrows appears to be short-hydroperiod mixed marl prairie communities that often include muhly grass (*Muhlenbergia filipes*) (Stevenson and Anderson 1994). These short-hydroperiod prairies contain moderately-dense, clumped grasses, with open space permitting ground movements by the sparrows. Sparrows tend to avoid tall, dense, sawgrass-dominated communities, coastal spike-rush (*Eleocharis*) marshes, extensive cattail (*Typha*) monocultures, long-hydroperiod wetlands with tall, dense vegetative cover, and sites supporting woody vegetation (Werner 1975, Bass and Kushlan 1982). Cape Sable seaside sparrows also avoid sites with permanent water cover (Curnutt and Pimm 1993).

Studies completed since the 1970s document that *Muhlenbergia* prairies support the highest densities of sparrows (Kushlan and Bass 1983, Werner and Woolfenden 1983, Pimm *et al.* 1994). The suitability of this vegetative community for the sparrow is driven by a combination of hydroperiod and periodic fires (Kushlan and Bass 1983). Fires prevent hardwood species from invading these communities and prevent the accretion of dead plant material, both of which decrease the suitability of these habitats for Cape Sable seaside sparrows. Werner (as cited by Pimm *et al.* 1994) found that sparrow numbers increased annually in areas that had been burned up to 3 years previously. Four years after a fire, he expected the suitability of these habitats to decline sharply. Curnütt (personal communication 1998) suggested that could be a localized response, but typically sparrow numbers increase up to 10 years post-fire.

Little is known about the wintering habitat of Cape Sable seaside sparrows. Birds have been observed near tree islands in December but may additionally be wintering within the salt marshes of the southern Everglades (Pimm *et al.* 1996).

C. Critical Habitat

Critical habitat for the Cape Sable seaside sparrow was designated on August 11, 1977 (50 CFR 17.95). The critical habitat, as designated, does not adequately account for the distribution of the present-day core subpopulations, or the areas necessary for the birds to maintain a stable population (Figure 3a). An important area west of Shark River Slough, which until 1993 supported one of two critical subpopulations (nearly half of the entire population), is not included within the designation, and has been undergoing detrimental changes in habitat structure as a result of water management practices (Curnutt *et al.* 1998, Nott *et al.* 1998).

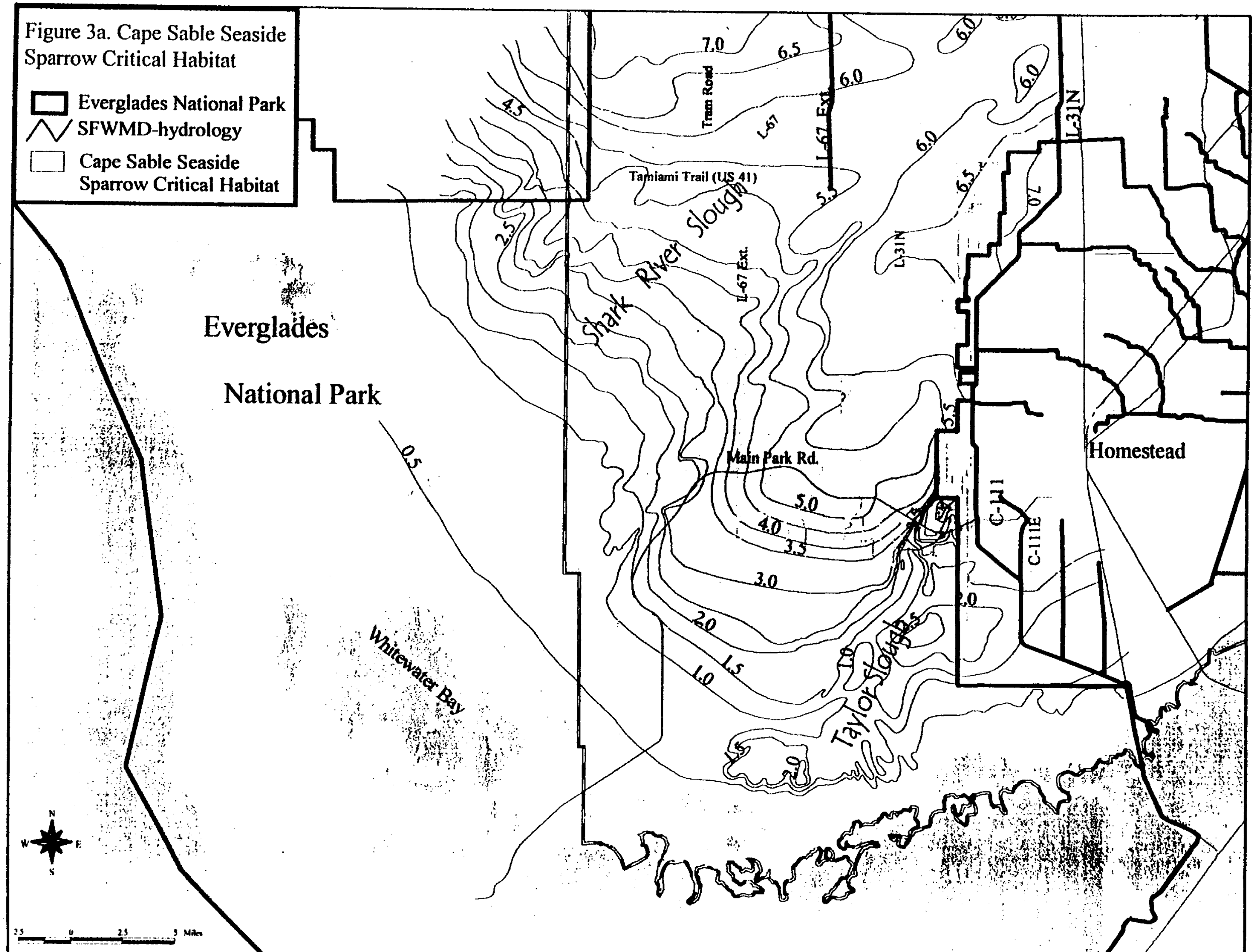
Figure 3a. Cape Sable Seaside Sparrow Critical Habitat
Everglades National Park
SFWMD-hydrology
Cape Sable Seaside Sparrow Critical Habitat
Everglades National Park
Whitewater Bay
Shark River Slough
Taylor Sloughs
Tamiami Trail (US 41)
Tram Road
L-67
L-67 Ext.
L-31N
Main Park Rd.
C-111
C-111E
Homestead
7.0
6.5
6.0
5.5
5.0
4.5
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
2.5 0 2.5 5 Miles

Additionally, other parts of the designated critical habitat have been converted to agriculture and are no longer occupied by sparrows.

Although the critical habitat that was designated for the Cape Sable seaside sparrow did not include constituent elements, hydroperiods that sustain the short-hydroperiod prairies are considered essential for the sparrows to successfully breed and are considered necessary to ensure the species' survival.

D. Reproduction

In most years, nesting typically occurs from mid-March until the onset of the rainy season (mid-June)(Nott et al. 1998). In dry years, nesting can begin as early as mid-February and continue through early August. The majority of nesting occurs in the spring when marl prairies are dry. Cape Sable seaside sparrows usually raise two broods in a season, although they may raise a third brood if weather conditions allow (Kushlan *et al.* 1982, U.S. Fish and Wildlife Service 1983). Sparrows need at least 40 days to complete one nesting cycle, a 60 day period is required for the initiation of second clutches, and an 80 day period allows for the fledglings of the second clutch to leave the nest (Nott et al. 1998). Nest cups are placed approximately 14 cm above the ground and are constructed with grasses (Werner 1975, Lockwood *et al.* 1997). Pimm (personal communication, 1997) suggests that nesting will not be initiated if water levels are at a depth greater than 10 cm during the breeding season. The end of the breeding season appears to be triggered by the onset of the summer rains. When water levels rise above the mean height of the nests off the ground, sparrows cease breeding.

Werner (1975) documented a 62 percent nest success rate in the Taylor Slough area, demonstrating a high reproductive potential for this subspecies. However, Pimm *et al.* (1996) report a significantly lower success rate (42 percent) during the 1995 and 1996 breeding seasons. Lockwood *et al.* (1997) report an 88 percent hatching rate, but only 40 percent of the eggs laid contribute to the total population each year. Kushlan *et al.* (1982) contend that the population has the ability to maintain or expand due to the high survival rate of adult males and the potential to produce two clutches of four eggs each breeding season.

Cape Sable seaside sparrows lay three to four eggs in each clutch (Werner 1978). Incubation has been estimated to take 12 to 13 days (Sprunt 1968, Trost 1968). The young spend 9 to 11 days at the nest. Both parents rear and feed the young birds and may do so for an additional 10 to 20 days after the young fledge (Woolfenden 1956, 1968, Trost 1968). Fledglings are incapable of flight until they are approximately 17 days of age.

E. Foraging

Cape Sable seaside sparrows typically forage by gleaning items from low vegetation or from the substrate (Ehrlich *et al.* 1992). They commonly feed on soft-bodied insects such as grasshoppers, spiders, moths, caterpillars, beetles, dragonflies, wasps, marine worms, shrimp, grass and sedge seeds (Stevenson and Anderson 1994). Significant differences were detected in nestling diet between years and sites (Lockwood *et al.* 1997), which reflects the patchy distribution of insects and opportunistic nature of the sparrow. The sparrow appears to shift the importance of prey items in its diet in response to their availability (Pimm *et al.* 1996).

F. Movements

The Cape Sable seaside sparrow is non-migratory. The fidelity of breeding male sparrows to their territories is high; many male seaside sparrows will defend the same area for two to three years (Werner 1975). During the non-breeding season they appear to congregate, and fly short distances within their range (Pimm, personal communication 1996). Cape Sable seaside sparrows have never been observed outside of nesting habitat areas during the wet-season. Results of a wintering ecology study (Dean and Morrison 1998) report that birds equipped with radio transmitters during the 1997-1998 wet season were generally sedentary or moved short distances (less than 1 km), and sometimes moved longer distances (5-7 km) within marl prairie habitat areas.

G. Status and Trends

The results of several studies suggest that Cape Sable seaside sparrows exist as several subpopulations whose distribution, size, and importance to the persistence of the species change with time. Bass and Kushlan (1982) described two core subpopulations of the sparrow, one northwest of Shark River Slough in the southeast portion of the Big Cypress National Preserve, and a second one in the Taylor Slough area southeast of Shark River Slough. Curnutt and Pimm (1993) recognized six subpopulations (subpopulations A-F) of the Cape Sable seaside sparrow that roughly correspond to the groupings recognized by Bass and Kushlan in 1982 (Figure 3). Pimm (1998) suggested that three breeding subpopulations are critical to the long-term survival of the Cape Sable seaside sparrow.

In 1981, Bass and Kushlan (1982) estimated a total of 6,656 birds in the six subpopulations; two core subpopulations that held most of the sparrows, and four peripheral subpopulations. Core subpopulation A inhabited the marl prairies west of Shark River Slough extending into Big Cypress National Preserve and held an estimated 2,688 individuals. Core subpopulation B held 2,352 birds inhabiting the marl prairies southeast of Shark River Slough near the center of ENP. Peripheral subpopulation E, north of subpopulation B, held about 672 sparrows, while subpopulation C, located along the eastern boundary of ENP, and subpopulation D, just to the southeast of subpopulation C, held about 400 birds each. Peripheral subpopulation F, the northern most peripheral subpopulation located on the western edge of the Atlantic Coastal Ridge, was the smallest subpopulation with an estimated 112 birds. Bass repeated the survey in 1992, with population estimates similar to those in 1981.

In 1981 and 1992, the area west of Shark River Slough (subpopulation A) supported nearly half of the total Cape Sable seaside sparrow population. Starting in 1993, the number of individuals declined precipitously in this area. By 1994 and 1995, the birds were absent from this area except for a few locations (Pimm *et al.* 1994, Pimm *et al.* 1995), and the number of individuals had dropped to less than ten percent of 1992 numbers. Population estimates improved slightly during the 1996 breeding season as the numbers of sparrows found west of Shark River Slough increased from approximately 240 in 1995 to 272 birds in 1996 and 1997 (Pimm *et al.* 1996). However, in 1998, the total number of birds west of Shark River Slough declined again to 192 birds (Sonny Bass, ENP, personal communication 1998).

Core subpopulation B increased by more than 800 birds from 1981 to 1992, declined slightly from 1992-1995, remained stable from 1995-1997, and decreased by approximately 1,000 individuals in 1998 (Sonny Bass, ENP, personal communication 1998). Since 1992, subpopulation B has held the majority of sparrows (Curnutt *et al.* 1998).

Curnutt *et al.* (1998) noted the following regarding the peripheral subpopulations: subpopulation C declined to 11 percent of its 1981 value by 1992. After three years of no birds, 48 birds were estimated in this area in 1996 and 1997 and 80 birds were estimated in 1998. Subpopulation D declined from 1981 to 1993 , was not counted in 1994, no birds were found in 1995, but 80 birds were estimated in this area in 1996, and 48 in 1997 and 1998. Subpopulation E decreased little between 1981 and 1992, 320 birds were estimated in 1993, 112 in 1994, 352 in 1995, 208 in 1996, 835 in 1997 and 912 in 1998. No sparrows were observed in subpopulation F in 1993, and only 16 birds were estimated in 1996 - 1998.

The most recent data indicate that Cape Sable seaside sparrows have declined by as much as 60 percent rangewide since 1981 (Curnutt *et al.* 1998, Nott *et al.* 1998). Biologists studying the sparrow have documented that high water levels in western Shark River Slough have caused the decline of the western subpopulation and continue to contribute to the absence of a population rebound (Nott *et al.* 1998). These declines cannot be attributed to the effects of Hurricane Andrew, which traversed this area in 1992 (Curnutt *et al.* 1998, Nott *et al.* 1998). Declines in sparrow population numbers were detected following Hurricane Andrew, however; a leveling off of declines, or rebound in population numbers, would be expected if populations were recovering from a single adverse event such as Hurricane Andrew. Instead, declines continued steadily as would be expected under continuing adverse hydrological conditions. Between 1992 and 1998, the size of the western breeding subpopulation of the Cape Sable seaside sparrow, which had represented 50 percent of the total population in 1992, had declined to about 10 percent of its previous size.

The smaller subpopulations composing the eastern breeding subpopulations C and F appeared to have been extirpated by 1993; however, the 1996, 1997 and 1998 surveys located a small number of birds at each of these sites. Frequent fires and shrubs invading these areas are thought to preclude the use of this habitat by the birds.

H. Recovery Plan Objective

According to the Technical Agency Draft of Volume I of the Multi-Species Recovery Plan for the Threatened and Endangered Species of South Florida (U.S. Fish and Wildlife Service 1998), criteria are provided below for reclassifying the Cape Sable seaside sparrow from endangered to threatened:

1. The loss of functional Cape Sable seaside sparrow habitat, as a result of current and past water management practices, and the invasion of woody and exotic species, is eliminated;
2. If the habitat west of Shark River Slough and in Taylor Slough is restored so that it supports breeding subpopulations larger than those measured in those areas for 1981 for 10 years;
3. When demographic information on the Cape Sable seaside sparrow supports, for a minimum of five years, a probability of persistence that is equal to or greater than 80 percent, for a minimum of 100 years;
4. When the rate of increase for the total population is equal to or greater than 0.0 as a 3-year running average for at least 10 years;
5. When a minimum of two stable, self-sustaining core breeding areas are secured;
6. If three auxiliary sites, in addition to the two core areas, are maintained and are used for breeding by a minimum of 10 percent of the total population for a minimum of two out of every five years during a 10 year period;
7. When a stable age structure is achieved in the core subpopulations and two other subpopulations;
8. When a minimum population of 6,600 birds is sustained for an average of 5 years, with all fluctuations occurring above this level; and,
9. When the ecosystem of the Cape Sable seaside sparrow is protected from the effects of water management practices throughout the species' historic range.

West Indian Manatee

The West Indian manatee was listed as an endangered on March 11, 1967 due to impacts to the population from harvesting for flesh, oil, and skins as well as for "sport"; coastal feeding grounds modification by siltation; and the volume of injuries and deaths resulting from collisions with the keels and propellers of powerboats (32 FR 4001). Manatees are also protected under the provisions of the Marine Mammal Protection Act of 1972, as amended (16 U.S.C. 1361 et seq.) and have been protected by Florida law since 1893. Critical habitat was designated for the manatee in 1976 and is described at 50 CFR § 17.95.

A. Distribution

The present distribution of the West Indian manatee includes the coasts and rivers of Florida, the Greater Antilles, eastern Mexico and Central America and northern and eastern South America

(Husar 1977, Lefebvre *et al.* 1989). *T. manatus latirostris* is found in and around Florida and *T. manatus manatus* occurs in the remaining areas of its range. The cooler winters along the U.S. coast of Gulf of Mexico, in combination with the deep water and strong currents of the Straits of Florida, create a barrier between the Antillean and Florida manatee; the resulting isolation contributes to their status as a subspecies.

The Florida manatee occurs primarily in the southeastern United States. The only year-round populations of manatees occur throughout the coastal and inland waterways of peninsular Florida and Georgia (Hartman 1974). During the summer months, manatees may range as far north along the East Coast of the U.S. as Rhode Island, west to Texas, and, rarely, east to the Bahamas (U.S. Fish and Wildlife Service 1996a, Lefebvre *et al.* 1989).

In Florida, manatees are found from the Georgia/Florida border south to Biscayne Bay on the east coast and from Wakulla River south to Cape Sable on the west coast (Hartman 1974, Powell and Rathbun 1984). Manatees are also found throughout the waterways in the Everglades and occasionally in the Florida Keys. Although temperatures are suitable for manatees in the Florida Keys, the low number of manatees has been attributed to the lack of fresh water (Hartman 1974). Manatees also occur in Lake Okeechobee.

Manatees frequently migrate throughout the waterways in South Florida. In South Florida, manatees are most prominent year round in the Indian River, Biscayne Bay, Everglades and Ten Thousand Island area, Estero Bay and Caloosahatchee River area, and Charlotte Harbor area.

B. Habitat

Manatees occur in both fresh and salt water habitats within tropical and subtropical regions and show preferences to waters with salinity levels of < 25 parts per thousand (ppt) (Hartman 1979). Manatees depend on areas with access to natural springs or man-made warm water refugia and access to areas with vascular plants and freshwater sources. Several factors contribute to the distribution of manatees in Florida. These factors are habitat-related and include proximity to warm water during cold weather, aquatic vegetation availability, proximity to channels of at least 6.5 feet in depth, and location of fresh water sources (Hartman 1979).

Manatees often seek out quiet areas in canals, creeks, lagoons or rivers. Deeper channels are often used as migratory routes. The combination of suitable seagrass beds, nearby deeper water access, and minimal boat traffic may be indicative of important mating, calving, and nursery grounds for manatees (MMC 1988, Smith 1993). Due to light limitations, most seagrass beds, are limited to shallow, nearshore waters.

C. Critical Habitat

Critical habitat was designated for the manatee in the early 1970s, although no specific primary or secondary constituent elements were included in the designation. Critical habitat for the manatee identifies specific areas occupied by the manatee, which have those physical and biological features essential to the conservation of the manatee and/or may require special management considerations.

D. Reproduction

The manatee population sex ratio is considered to be 1:1 for both adults and calves (Rathbun *et al.* 1992). Females reach sexual maturity at 3-5 years of age (Marmontel 1993) and males may reach sexual maturity at 3-4 years of age. Manatee longevity has been estimated at 50 years or more and they appear to be able to reproduce their entire adult life (Marmontel *et al.* 1992). The minimum interval between manatee birth is 2 years, but not all female manatees are this fecund. Gestation of the single calf takes 12-14 months (Reid *et al.* 1992). Age to weaning varies from 1-2 years. Per capita reproductive rates in Florida manatees have been estimated from a low of 0.15 in the Blue Spring population to a high of 0.19 in the Atlantic coast population. The maximum potential rate of population increase has been estimated at 2.0-7.0 percent, this rate is most sensitive to changes in adult survival and, secondarily, subadult survival (Packard 1985, Marmontel 1993).

E. Foraging

Normally, manatees feed on a variety of submergent, emergent, and floating vegetation. In northeast Florida, where submerged vegetation is scarce, especially in the St. Johns River, manatees feed on floating vegetation. In South Florida, submerged vegetation is more plentiful. Seagrasses comprise the largest component of the manatee's diet, especially in South Florida (Hartman 1979). Some manatees have been observed to return to the same seagrass beds to feed year after year and may show preferences for certain areas (Sirenia 1993). Preference was also shown for areas with healthy seagrass beds adjacent to relatively deeper waters with little boat traffic (Sirenia 1993).

F. Movements

Typically manatees migrate northward in the springtime and southward in the fall and winter. Manatees do not range far offshore, but may travel along the coast (Beeler and O'Shea 1988).

The increase in the number of man-made warm water sources over the years has influenced manatee behavior, specifically migratory patterns and social behavior. Although man-made, warm-water sites provide a type of refugia for manatees in winter, the effects of the associated behavioral changes (e.g., mating, cavorting, calving) as a result of the availability of artificial refugia has not been closely examined to determine if there are any negative effects to the manatee. During the 1996 die-off, a large number of manatees aggregating in warm waters near the Ft. Myers powerplant died from exposure to a red tide brevetoxin outbreak. Their vulnerability was increased because they were congregated near this artificial refuge when the red tide occurred (Lefebvre *et al.* 1989).

G. Status and Trends

Exact estimates of the historic manatee population is uncertain, but overhunting between the 1700s and 1900s is believed to be responsible for reducing manatee population to only a few relict groups (Hartman 1979). A slow increase in manatee numbers in the late 1800s is attributed to their protection by the 1893 State law prohibiting their killing.

Synoptic aerial surveys were conducted twice in 1997, resulting in total counts of 2,229 and 1,709 animals, respectively. During the January 19-20, 1997, survey, 900 manatees were counted on the east coast and 1,329 on the west coast. During the February 13, 1997, survey, 791 manatees were counted on the east coast and 918 on the west coast. This estimate represents the minimum number of manatees in Florida waters. Although this has been the highest estimate of manatees since the synoptic surveys were started, the results of these surveys may vary because of such factors as sampling methodology, manatee behavior, and weather conditions. Because of this variation and the high degree of uncertainty in surveying, it is difficult to correlate these manatee population estimates with overall manatee population trends (Ackerman *et al.* 1995).

H. Recovery Plan Objective

The West Indian manatee can be considered for reclassification to threatened when data and population models are available to assess population size and trends; when analyses indicate that the population is growing or stable; when mortality factors are controlled at acceptable levels or are decreasing; and when critical habitats are secure and threats to them are controlled or decreasing.

The long-range recovery goal is to contribute toward restoring Florida manatees to optimum sustainable population levels which is a range between the largest number supportable by the ecosystem and the population size that results in maximum net productivity.

Florida Panther

The Florida panther was listed as endangered on March 11, 1967 due to heavy hunting and trapping pressures, the inability of the species to adapt to changes in the environment, and developmental pressures (32 FR 4001). No critical habitat has been designated for this species.

A. Distribution

The only known, remaining panther population is centered in and around the Big Cypress Swamp/Everglades physiographic region of South Florida. Data on radio-instrumented members of this population indicate that it is centered in Collier and Hendry counties of southwest Florida. Instrumented panthers have also been documented in Broward, Dade, Glades, Hardee, Highlands, Lee, Monroe, and Palm Beach counties. There are still large areas of private land in

Charlotte, Collier, Hendry, Lee, and Glades counties where uncollared individuals may reside (Maehr 1990).

B. Habitat

The greatest concentration of unprotected, occupied panther habitat is found on private land in eastern Collier County and southern Hendry County. In general, these private lands are located north of important panther habitat on key publicly owned lands (Maehr 1988). For the most part, privately owned lands are higher in elevation, better drained, have a higher percentage of hardwood hammocks and pine flatwoods, and are higher in natural fertility/productivity than public lands south of Interstate 75. A difference in soils and drainage patterns is reflected in more upland vegetation and more abundant prey in lands north of Interstate 75. These factors, in combination with some management practices (e.g. prescribed fire) tend to make the area more attractive to, and increase carrying capacities for, white-tailed deer (*Odocoileus virginianus*) and feral hogs (*Sus scrofa*).

Native, upland forests are preferred by panthers in southwest Florida (Maehr 1990). Understory thickets of tall, almost impenetrable, saw palmetto have been identified as the most important resting and denning cover for panthers (Maehr 1990). Early radio-telemetry investigations indicated that panther use of mixed swamp forests and hammock forests was greater than expected in relation to their availability within the panthers' home range area (Belden *et al.* 1988). Hardwood hammocks were consistently preferred by panthers, followed by pine flatwoods. This may be related to the fact that, among major vegetation types in south Florida, hammocks have the greatest potential for producing white-tailed deer, an important panther prey species (Harlow 1959, Belden *et al.* 1988, Maehr 1990).

Dispersing males may wander widely through unforested and disturbed areas. Agricultural and other disturbed habitats, freshwater marsh, thicket swamp, and mixed swamp are not preferred, and are either used in proportion to their availability or are avoided (Maehr 1990). Habitats avoided by panthers include agricultural, barren land, shrub and brush, and dry prairie. Panthers have not been found in pastures during daytime radio-telemetry flights but may travel through them at night (Maehr *et al.* 1991).

C. Reproduction

Male Florida panthers are polygynous. Breeding activity peaks in fall and winter. Parturition is distributed throughout the year with 81 percent of births occurring between March and July. Litter sizes range from one to four kittens, with a mean of 2.2 kittens per successful litter. Intervals between litters range from 16 to 37 months (Land 1994). Age at first reproduction has been documented at 18 months for females (Maehr *et al.* 1990). The dispersal of young typically occurs around 1.5 to 2 years of age.

D. Foraging

Food habit studies of panthers in southwest Florida indicated that the feral hog was the most commonly taken prey followed by white-tailed deer, raccoon (*Procyon lotor*), and 9-banded armadillo (*Dasypus novemcinctus*). Deer and hogs accounted for 85.7 percent of consumed biomass north of Interstate 75, and 66.1 percent south of Interstate 75 (Maehr 1990). No seasonal variation in diet was detected; however, panthers inhabiting an area of better soils north of Interstate 75 consumed more large prey. In addition, deer abundance was 8-fold greater north of Interstate 75. Fewer large prey may, in part, explain the poorer physical condition, larger home ranges, and lower reproductive output of panthers in the south. Hogs dominated the diet of panthers in the north in terms of both estimated biomass and numbers. In the south, deer accounted for the greatest estimated biomass consumed, whereas raccoons were the highest estimated number of consumed prey.

E. Movements

Adult panthers space themselves throughout available habitat where home range overlap is extensive among resident females and limited among resident males (Maehr *et al.* 1991). Dispersal distances average 36.5 miles for subadult males and 10 miles for a subadult female. Dispersing males wander widely. Although some travel occurs during the day, panthers are mostly crepuscular (Maehr *et al.* 1990).

F. Status and Trends

The population of the Florida panther may have numbered as many as 500 at the turn of the century (Seal and Lacy 1989). Hunting, habitat loss through residential and agricultural development, loss of the panther's prey base, and other forms of persecution have led to the decline of this species since that time (Belden *et al.* 1988). In 1950, the Florida panther was declared a game species in the State of Florida. This action resulted in the first regulation of panther harvest. By 1958 it was listed under state law as an endangered species. The population was estimated at 100 to 300 statewide in 1966 (Schemnitz 1972). The federal government followed suit and listed the species as endangered in 1967.

The Florida panther's existence is severely threatened by both rapid and gradual extinction processes. Population viability analysis projections indicate that under existing demographic and genetic conditions the Florida panther will likely be extinct in only a few decades (24-63 years) (Seal *et al.* 1989). Environmental factors affecting the Florida panther include: habitat loss and fragmentation, environmental contaminants, prey availability, human-related disturbance and mortality, disease, and genetic erosion (U.S. Fish and Wildlife Service 1998).

G. Recovery Plan Objective

The present recovery objective for the Florida panther is to achieve three viable, self-sustaining populations within the historic range of the animal. First priority will be to secure the population in south Florida. A viable population level will be determined when enough data are available to develop a panther population model. An essential criteria for recovery of the panther needs to ensure 95 percent probability of persistence of the south Florida population over 100 years. Re-established populations may require separate population goals. Population objectives will generally be based on the size of the respective areas, and other ecological factors important to panthers.

Snail Kite

The species was federally listed as endangered in 1967 and critical habitat was determined in 1977 (see FR 42(155): 40685-40688). That protection was continued under the Endangered Species Conservation Act of 1969 and the Endangered Species Act of 1973, as amended. The snail kite was listed because of its limited distribution and threats to its habitat posed by large-scale conversion of land in southern Florida to agricultural uses.

A. Distribution

The current distribution of the snail kite in Florida is limited to central and southern portions of the State (Figure 4). Six large freshwater systems generally encompasses the current range of the species, although radio tracking of snail kites has revealed that the network of habitats used by the species also includes many other smaller widely dispersed wetlands within this overall range (Bennetts and Kitchens 1997).

B. Habitat

Snail kite habitat consists of fresh-water marshes and the shallow vegetated edges of lakes (natural and man-made) where apple snails can be found. Suitable foraging habitat for the snail kite is typically a combination of low profile (≤10 feet) marsh with an interdigitated matrix of shallow (0.65 - 4.25 feet deep) open water, which is relatively clear and calm. Low trees and shrubs are also often interspersed with the marsh and open water. Snail kites require foraging areas to be relatively clear and open in order to visually search for apple snails, therefore, dense growth of herbaceous or woody vegetation is not conducive to efficient foraging. Nearly continuous flooding of wetlands for ≥1 year is needed to support apple snail populations that in turn provide forage for the snail kite (Beissinger 1988).

Nesting and roosting sites almost always occurs over water, which deters predation. Nesting substrates include small trees (usually < 32.8 feet in height), but can also occur in herbaceous vegetation, such as sawgrass, cattail, bulrush, and reed (U.S. Fish and Wildlife Service 1998). It is important to note that suitable nesting substrate must be close to suitable foraging habitat, so extensive areas of contiguous woody vegetation are generally unsuitable for nesting.

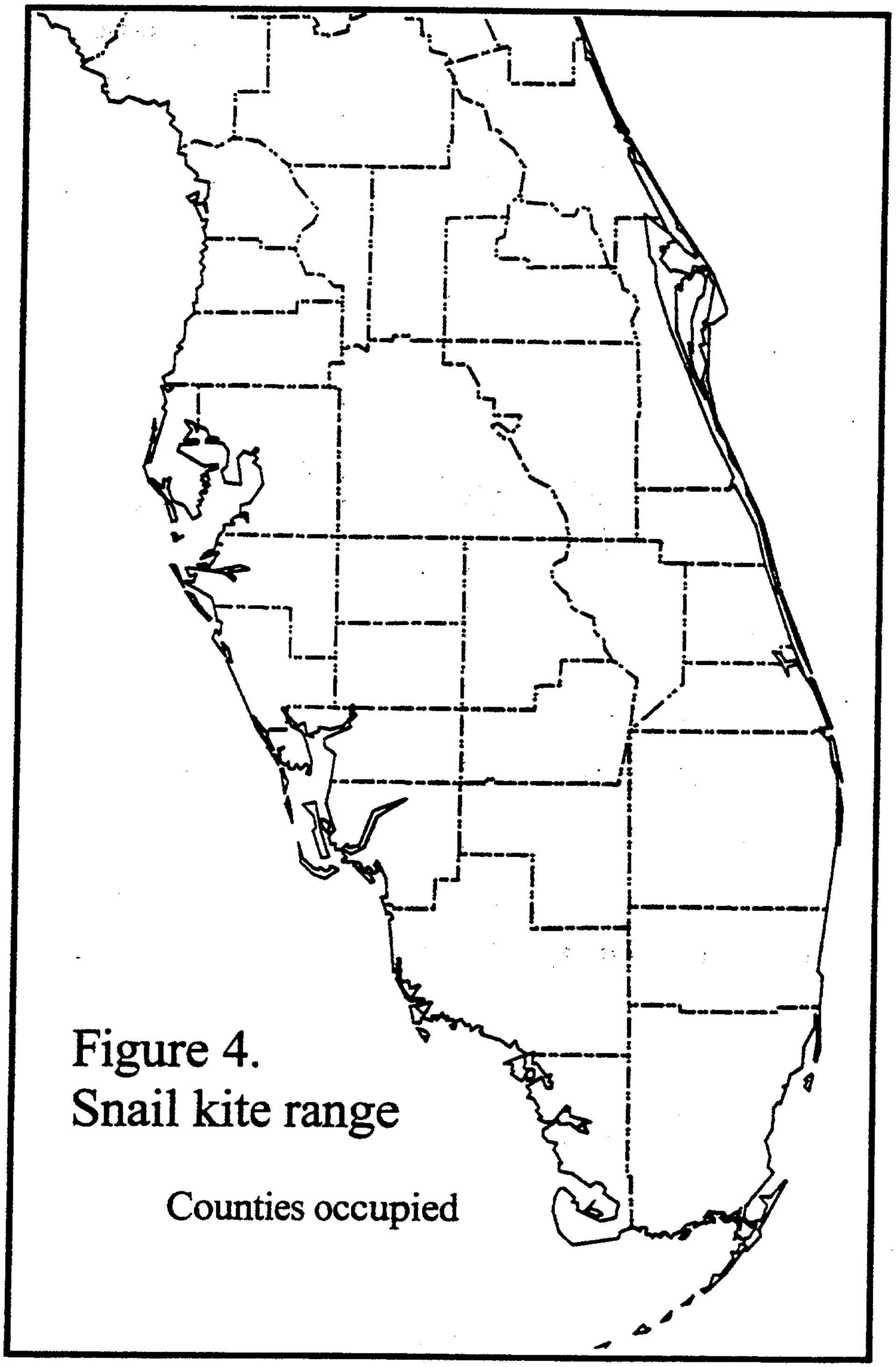

Figure 4.
Snail kite range

F. Movements

Snail kites in Florida are not migratory in the strict sense; they are restricted to South and Central Florida. Snail kites are nomadic in response to water depths, hydroperiod, food availability, nutrient loads, and other habitat changes (Bennetts *et al.* 1994). Radio-tracking and sighting of marked individuals have revealed that nonbreeding individuals disperse widely on a frequent basis (Bennetts *et al.* 1994). Shifts in distribution can be short-term, seasonal, or long-term, and can take place between areas among years (Rodgers *et al.* 1988), between areas within a given nesting season (Beissinger 1986), within areas in a given nesting season, and within or between areas for several days to a few weeks (Bennetts *et al.* 1986). Sykes (1983) noted that during colder winters, snail kites will shift their distribution more to the southern part of their range.

G. Status and Trends

Several authors (Nicholson 1926, Howell, 1932, Bent 1937) indicated that the snail kite was numerous in central and South Florida marshes during the early 1900s, with groups of up to 100 birds. Sprunt (1945) estimated the population to be 50 to 100 individuals. The snail kite apparently plummeted to its lowest population between 1950 and 1965. By 1954, the population was estimated at no more than 50 to 75 birds (Sprunt 1954). Stieglitz and Thompson (1967) reported eight birds in 1963 at the Loxahatchee National Wildlife Refuge, 17 on the refuge and two at Lake Okeechobee in 1964, eight in WCA 2A and two at Lake Okeechobee in 1965, and 21 in WCA 2A in 1966.

While acknowledging the problems associated with making year-to-year comparisons in the count data, some general conclusions are apparent. Lake Okeechobee apparently retains some suitable snail kite habitat throughout both wet and dry years. In contrast, kite use of WCA 3A fluctuates greatly, with low use during drought years, such as 1991, and high use in wet years, such as 1994. Although sharp declines have occurred in the counts since 1969 (for example, 1981, 1985, 1987), it is unknown to what extent this reflects actual changes in the population. Rodgers *et al.* (1988) point out that it is unknown whether decreases in snail kite numbers in the annual count are due to mortality, dispersal (into areas not counted), decreased productivity, or a combination of these factors. Despite these problems in interpreting the annual counts, the data since 1969 have indicated a generally increasing trend (Rodgers *et al.* 1988, Bennetts *et al.* 1994).

The snail kite has apparently experienced population fluctuations associated with hydrologic influences, both man-induced and natural (Sykes 1983, Beissinger 1986), but the amount of fluctuation is debated. The abundance of its prey, apple snails, is closely linked to water regime (Sykes 1979, 1983). Drainage of Florida's interior wetlands has reduced the extent and quality of habitat for both the apple snail and the kite (Sykes 1983). The kite nests over water, and nests become accessible to predators in the event of unseasonal drying (Beissinger 1986, Sykes 1987). In dry years, the kite depends on water bodies which normally are suboptimal for feeding, such as canals, impoundments, or small marsh areas, remote from regularly used sites (Bennetts *et al.* 1988). These secondary or refuge habitats are vital to the continued survival of this species in Florida.

H. Recovery Plan Objective

Pursuant to the Florida snail kite Recovery Plan (U.S. Fish and Wildlife Service 1986), to achieve downlisting the following criteria must be met:

> The population goal is the number of kites that have a 95 percent probability of surviving 3-4 successive drought years of no reproduction and high mortality, while remaining reproductively viable. In the absence of adequate data, upon which to establish a specific population goal, the interim goal is an annual average of 650 birds for a ten-year period with annual population declines of less than 10 percent of the average.

Wood Stork

The United States population of the wood stork was listed as endangered in 1984 because it had declined by more than 75 percent since the 1930s (49 FR 7335). At the time, the Service believed that the United States breeding population would be extirpated by the turn of the century if it continued to decline at the same rate. The original listing recognized the relationship between the declining wood stork population, the loss of suitable foraging habitat and colony nesting failures, particularly in the breeding colonies in South Florida where human actions have reduced wetland areas by about 35 percent (Ogden and Nesbitt 1979), influenced the hydrologic regime, and altered the prey base. No critical habitat has been designated for this species.

A. Distribution

In the United States, the wood stork has been reported both as a casual and regular visitor, ranging from southern California and southern Arizona, the Gulf of Mexico north to Canada; from Maine, southern New Brunswick, Canada, and New York, south to its breeding range in Florida, Georgia, and South Carolina. It is suspected that most wood storks sighted in Arkansas, Louisiana, Texas, and points farther west are birds that have dispersed from colonies in Mexico. Wood storks have nested in every county in South Florida. During the non-breeding season (July to October), wood storks are much less common in South Florida. The range of the wood stork in Florida is identified in Figure 6.

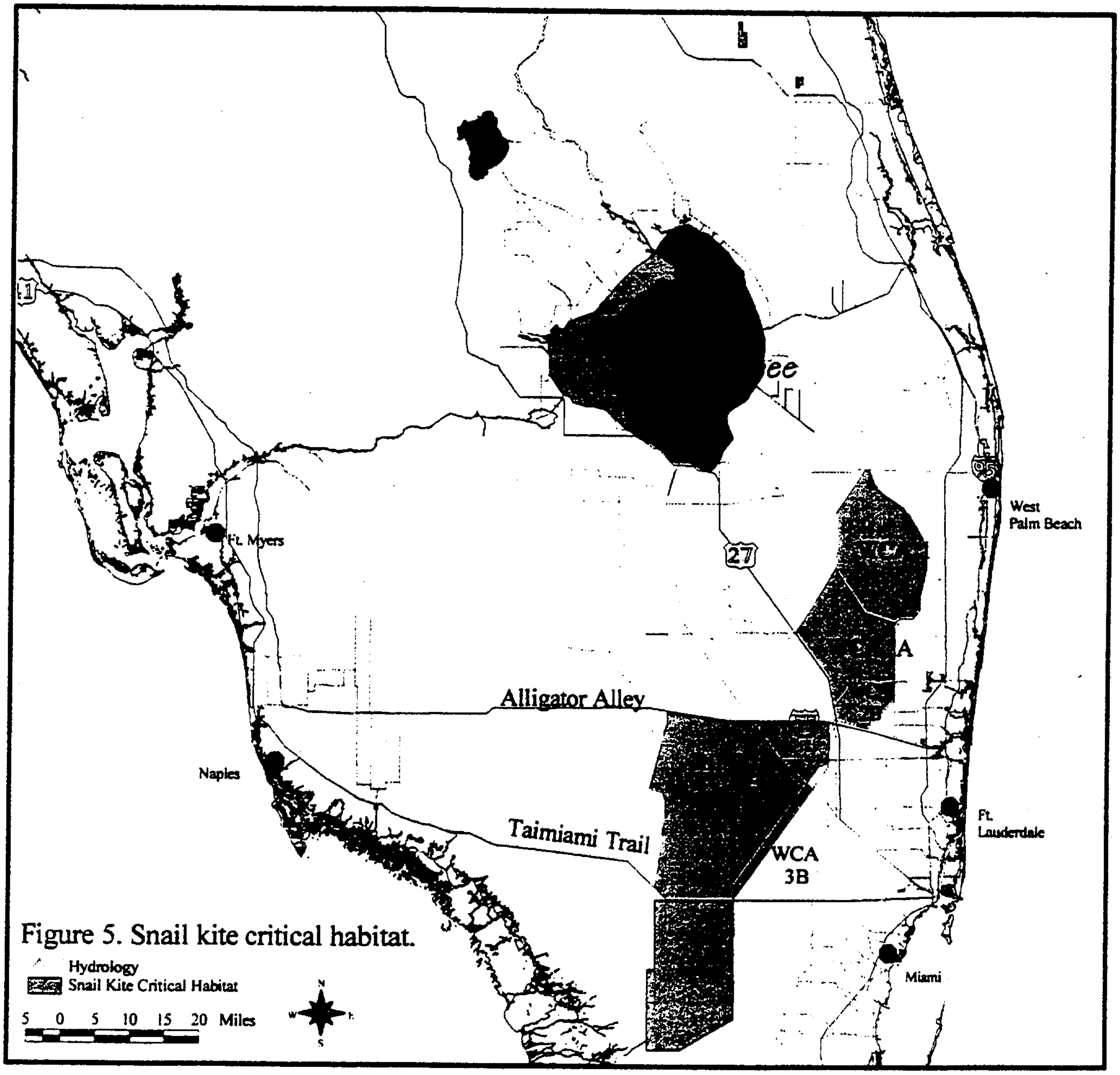

Figure 5. Snail kite critical habitat.

B. Habitat

The wood stork is primarily associated with freshwater habitats for nesting, roosting, foraging, and rearing. Wood storks typically construct their nests in medium to tall trees that occur in stands located either in swamps or on islands surrounded by relatively broad expanses of open water (Ogden 1991). During the non-breeding season or while foraging, wood storks occur in a wide variety of wetland and other aquatic habitats. Typical foraging sites for the wood stork include freshwater marshes and stock ponds, shallow, seasonally flooded roadside or agricultural ditches, narrow tidal creeks or shallow tidal pools, managed impoundments, and depressions in cypress heads and swamp sloughs. Because of their specialized feeding behavior, wood storks forage most effectively in shallow-water areas with highly concentrated prey (Ogden *et al.* 1978, Browder 1984, Coulter 1987). In South Florida, low, dry-season water levels are often necessary to concentrate fish to densities suitable for effective foraging by wood storks (Kushlan *et al.* 1975). As a result, wood storks will forage in many different shallow wetland depressions where fish become concentrated, either due to local reproduction by fishes, or as a consequence of seasonal drying.

C. Reproduction

Wood storks tend to use the same colony sites over many years, as long as the sites remain undisturbed and sufficient feeding habitat remains in the surrounding wetlands. Traditional wetland sites may be abandoned by storks once local or regional drainage schemes remove surface water from beneath the colony trees. As a result of such drainage, many storks have shifted colony sites from natural to managed or impounded wetlands. Ogden (1991) has suggested that recent increases in the number of colonies in north and central Florida have occurred as a result of an increase in the availability of altered or artificial wetlands.

The date on which wood storks begin nesting varies geographically. In Florida, wood storks lay eggs as early as October and as late as June (Rodgers 1990). In general, earlier nesting occurs in the southern portion of the state (below 27°N). Storks nesting in the Everglades and Big Cypress basins, under pre-drainage conditions, formed colonies between November and January (December in most years) regardless of annual rainfall and water level conditions (Ogden 1994). In response to deteriorating habitat conditions in South Florida, wood storks in these two regions have delayed the initiation of nesting to February or March in most years since the 1970s. This shift in the timing of nesting is believed to be responsible for the increased frequencies of nest failures and colony abandonment in these regions over the last 20 years; colonies that start after January in South Florida risk having young in the nests when May-June rains flood marshes and disperse fish.

Female wood storks lay a single clutch of eggs per breeding season. However, they will lay a second clutch if their nests fail early in the breeding season. Wood storks lay two to five (usually three) eggs depending on environmental conditions; the average clutch size may increase during years with favorable water levels and food resources. Once an egg has been laid in a nest, the breeding pair never leave the nest unguarded. Both parents are responsible for incubation and foraging (Palmer 1962). Incubation takes approximately 30 days, and begins after the first one or two eggs are laid.

The productivity of wood stork colonies varies considerably between years and locations, apparently in response to differences in food availability; colonies that are limited by food resources may fledge an average of 0.5-1.0 young per active nest; colonies that are not limited by food resources may fledge between 2.0 and 3.0 young per active nest (Ogden 1996).

D. Foraging

The natural hydrologic regime in South Florida involves seasonal flooding of extensive areas of the flat, low-lying peninsula, followed by drying events which confine water to ponds and sloughs. Fish populations reach high numbers during the wet season, but become concentrated into smaller areas as drying occurs. Consumers, such as the wood stork, are able to exploit high concentrations of fish in drying pools and sloughs. In the pre-drainage Everglades, the dry season of South Florida provided wood storks with ideal foraging conditions over a wide area.

Storks forage in a wide variety of shallow wetlands, wherever prey reach high enough densities, and in water that is shallow and open enough for the birds to be successful in their hunting efforts (Ogden *et al.* 1978, Browder 1984, Coulter 1987). Good feeding conditions usually occur in relatively calm water, where depths are between 4 - 10 inches, and where the water column is uncluttered by dense patches of aquatic vegetation (Coulter and Bryan 1993). In South Florida, dropping water levels are often necessary to concentrate fish to suitable densities (Kushlan *et al.* 1975). Typical foraging sites throughout the wood stork's range include freshwater marshes and stock ponds;, shallow, seasonally flooded roadside or agricultural ditches; narrow tidal creeks or shallow tidal pools; managed impoundments; and depressions in cypress heads and swamp sloughs. Almost any shallow wetland depression that concentrates fish, either through local reproduction or the consequences of area drying, may be used as feeding habitat.

E. Movements

During the non-breeding season (summer-fall), juvenile wood storks from South Florida colonies have been located throughout the Florida Peninsula, southern Georgia, coastal South Carolina, central Alabama, and east-central Mississippi (Ogden 1996). Additionally, marked individuals from a colony in east-central Georgia were found in the central Everglades during the winter. This information suggests the notion of a single population of wood storks in the southeast responding to changing environmental conditions through temporal relocation. Although the majority of nesting by the southeastern wood stork population no longer occurs in South Florida, the wetlands of the Everglades remain as important feeding areas for large numbers of storks during the dry season (winter-spring) (Bancroft *et al.* 1992).

F. Status and Trends

Although we cannot accurately estimate the size of the United States breeding population prior to the late-1950s, we have reliable estimates of the size of some breeding colonies. Historically, larger breeding colonies at areas like Corkscrew Swamp, Okaloacoochee Slough, and the southern Everglades contained 5,000 to 15,000 pairs of wood storks (Palmer 1962, Ogden 1996). Most authors also agree that, since the late 1930s, the number of wood storks in the United States declined by more than 90 percent; additionally, the rate of decline in South Florida accelerated in the 1960s and 1970s (Palmer 1962, Ogden 1994).

Between 1957 and 1960, the Florida and National Audubon Societies conducted a series of statewide aerial wood stork surveys of all known or suspected stork nesting colonies. In 1974, statewide aerial surveys were initiated and repeated, annually, until 1986. In 1959, 14 breeding colonies supported an estimated 7,657 pairs of wood storks in Florida; in 1960, 15 breeding colonies supported an estimated 10,060 breeding pairs. By 1975, 15 breeding colonies supported an estimated 5,382 breeding pairs; in 1976, 17 breeding colonies supported an estimated 5,110 breeding pairs. Since 1983, the United States breeding population of wood storks has fluctuated between 5,500 and 6,500 pairs.

While the number of wood storks breeding in South Florida has substantially decreased in the 1970's; in north Florida, Georgia, and South Carolina the number of breeding wood storks has significantly increased (Ogden *et al.* 1987). From 1958-1960, 80-88 percent of wood stork nesting pairs were located at six sites in South Florida. Surveys from 1976 showed a decline to 68 percent, with a further decline to 13 percent in 1986. Since the late 1970s, a majority of wood storks have nested in central and north Florida, and an increasing number have nested in coastal colonies in Georgia and South Carolina. Between 1965 and 1995, the number of wood storks nesting in Georgia increased from four pairs to 1,501 pairs; between 1981 and 1995, the number of wood storks nesting in South Carolina increased from 11 pairs to 829 pairs. Since the 1970s, associated with this shift to the north, the southeast wood stork population appears to be gradually increasing, from a low of 3,000-4,000 pairs in the late 1970s, to over 6,000 pairs in the mid-1990s.

H. Recovery Plan Objective

Pursuant to the wood stork Recovery Plan (U.S. Fish and Wildlife Service 1996b), to achieve recovery the following criteria must be met:

To downlist:
An average of 6,000 nesting pairs and annual regional productivity greater than 1.5 chicks per nest year, calculated over 3 years.

To delist:

An average of 10, 000 nesting pairs calculated over 5 years beginning at time of reclassification, annual regional productivity greater than 1.5 chicks per nest per year (also calculated over a 5-year average). As a subset of the 10,000 pairs, a minimum of 2,500 successful nesting pairs must occur in the Everglades and Big Cypress systems.

American Crocodile

The American crocodile was listed as endangered throughout its range in 1975 and critical habitat was established for this species in 1979 (40 FR 44151 and 44 FR 75076, respectively). The listing of the species and protection of habitat was required because of documented population declines most likely associated with habitat alterations and direct human disturbances to crocodiles and their nests (U.S. Fish and Wildlife Service 1984). Critical habitat was designated for the crocodile in 1979 and is described at 50 CFR § 17.95.

A. Distribution

Historically, American crocodiles occurred at least as far north on the Florida east coast as Lake Worth, Palm Beach County (DeSola 1935, U.S. Fish and Wildlife Service 1984), to Tampa Bay on the west coast (Kushlan and Mazzotti 1989), and as far south as Key West (Neill 1971). The distribution of crocodiles during the non-nesting season may vary considerably among years since adult crocodiles can disperse great distances (Kushlan and Mazzotti 1989). Today, the majority of crocodiles are present in the vicinity of their core nesting areas (Figure 7), located in Biscayne and Florida Bays (Kushlan and Mazzotti 1989).

The current distribution of the American crocodile is limited to extreme South Florida including coastal areas of Dade, Monroe, Collier, and Lee counties (U.S. Fish and Wildlife Service 1984). Occasional sightings are still reported farther north on the east coast, and a few isolated crocodiles may still survive in remnant mangrove habitats in Broward County.

B. Habitat

The American crocodile is found primarily in mangrove swamps and along low energy mangrove-lined bays, creeks, and inland swamps (Kushlan and Mazzotti 1989). In Florida, patterns of crocodile habitat use shifts seasonally. During the breeding and nesting seasons, adults use the more open waters of Florida Bay, whereas during the fall and winter, they are found primarily in the fresh and brackish-water inland swamps, creeks, and bays (Kushlan and Mazzotti 1989). Along northeastern Florida Bay, crocodiles were found in inland ponds and creeks (50 percent of observations), protected coves (25 percent of observations), exposed shorelines (6 percent of observations) and a small number were observed on mud flats (Kushlan and Mazzotti 1989). The high use of inland waters suggests crocodiles prefer less saline waters; using sheltered areas such as undercut banks and mangrove snags and roots that are protected from wind and wave action. Access to deep water (>3 feet) is also an important component of preferred habitats (Mazzoti 1983).

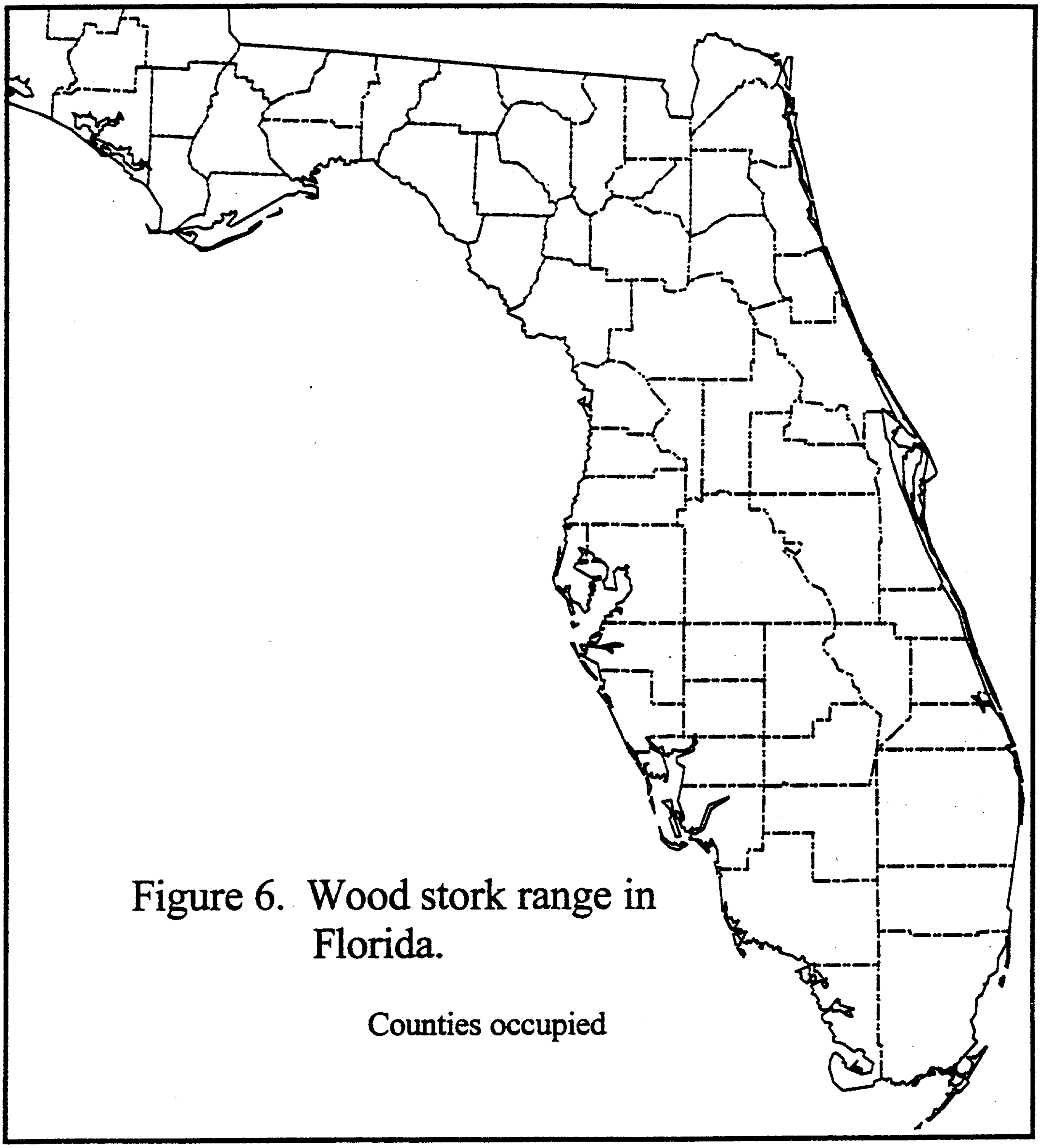

Figure 6. Wood stork range in Florida.

Natural nesting habitat includes sites with raised marl creek banks or sandy shorelines adjacent to deep water. Crocodiles also nest on elevated man-made structures such as canal berms and other places where fill has been introduced. In natural nesting situations, creek bank nests are generally considered optimal since these sites provide a good incubation medium and are generally protected from wind and wave action. These nest sites also provide deep water refuge for adult females. Nests adjacent to open water provide little protection of the nest or adults. Shore nests are typically not located near good nursery habitat, and mortality of hatchlings is generally higher than in inland nests (Kushlan and Mazzotti 1989).

C. Critical Habitat

Critical habitat was designated for the American crocodile in 1979 (44 FR 75076) and, since then, has not been revised (Figure 8). Although the critical habitat that was designated for the American crocodile did not include constituent elements, water level management is required to maintain favorable habitat conditions that are considered necessary to ensure the species survival.

D. Reproduction

As with most crocodilians, courtship and mating are stimulated by increasing ambient water and air temperatures. In South Florida, temperatures sufficient to allow initiation of courtship behavior are reached by late February through March. Like all other crocodilians, the mating system of the American crocodile is polygynous; each breeding male mates with a number of females (Magnusson *et al.* 1989).

Nest sites are typically selected where a sandy substrate exists above the normal high water level. Nesting sites include areas of well drained sands, marl, peat, and rocky spoil and may include areas such as sand/shell beaches, stream banks and canal spoil banks that are adjacent to relatively deep water (Ogden 1978, Kushlan and Mazzotti 1989).

The success of American crocodile nesting in South Florida is dependent primarily on the maintenance of suitable egg cavity moisture throughout incubation and on nest predation. On Key Largo, and other island nests, failure of crocodile nests is typically attributed to dessication due to low rainfall (Moler 1991). On Key Largo, about 52 percent of nests were successful in hatching at least one young (Moler 1991). Nest failures on the mainland may be associated with flooding, desiccation, or predation (Mazzotti *et al.* 1988, Mazzotti 1989). On the mainland, about 13 percent of nests monitored were affected by flooding or desiccation, while 13 percent of nests were partially or entirely depredated (Mazzotti *et al.* 1988, Mazzotti 1989). More recently, Mazzotti (1994) found that predation rates on the mainland increased to 27 percent, while only nine percent of nests failed because of infertility or embryonic mortality. Most examined eggs have been fertile (90 percent, range 84-100 percent)(Kushlan and Mazzotti 1989, Mazzotti 1989). Incubation of the clutch takes about 86 days (Lang 1975), during which time the female periodically visits the nest (Neill 1971, Ogden 1978).

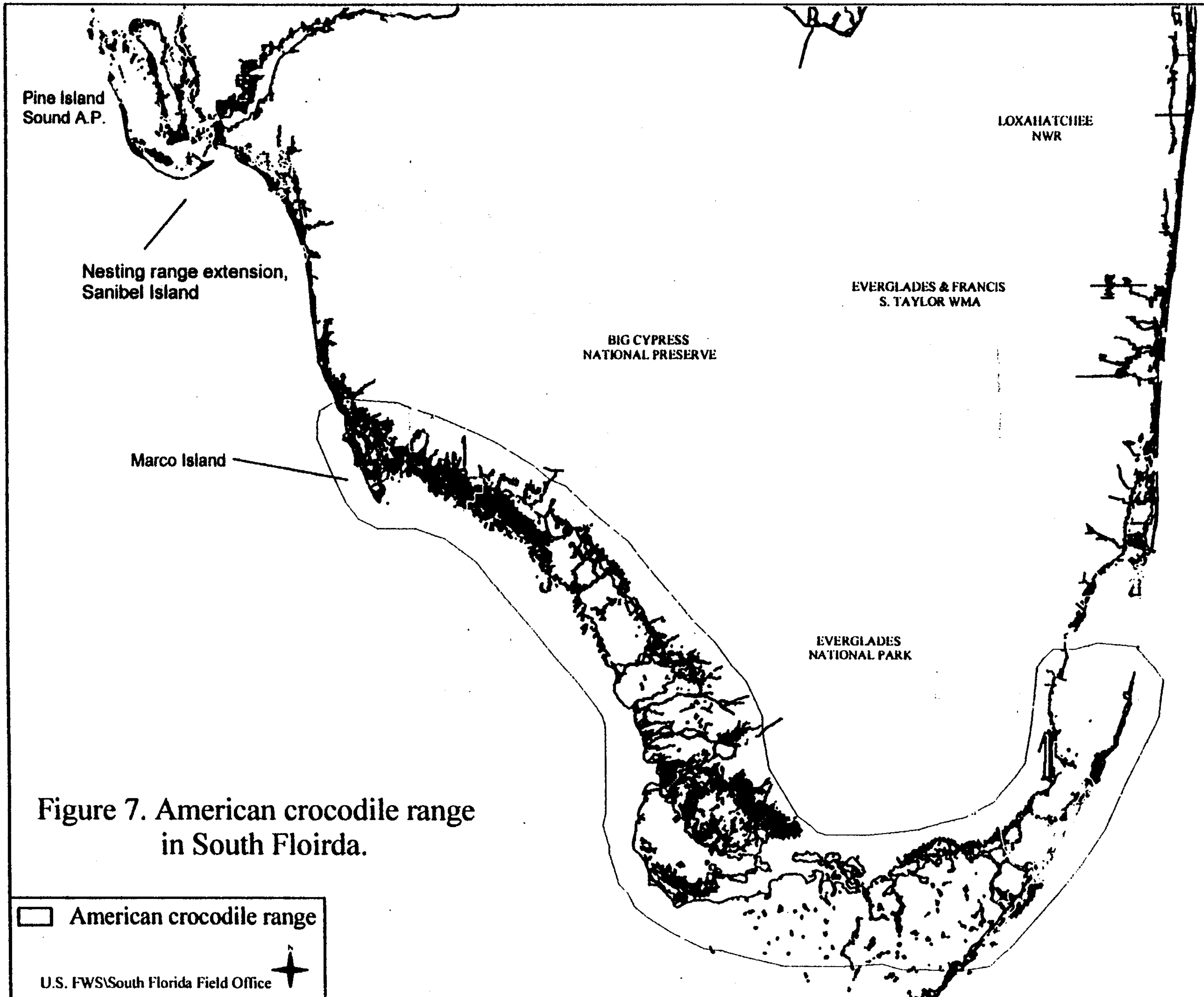

Figure 7. American crocodile range in South Floirda.