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We are submitting herewith a dissertation written by Audrey Lorraine Mayer entitled "Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*) habitat and the Everglades: Ecology and conservation." We have examined the final copy of this dissertation for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Doctor of Philosophy, with a major in Ecology and Evolutionary Biology.

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CAPE SABLE SEASIDE SPARROW (*AMMODRAMUS MARITIMUS MIRABILIS*) HABITAT AND THE EVERGLADES: ECOLOGY AND CONSERVATION

A Dissertation
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Doctor of Philosophy
Degree
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Audrey Lorraine Mayer
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ABSTRACT

This dissertation examines conservation issues within the Everglades, a renowned wetland ecosystem in central and southern Florida, USA. Everglades National Park, which includes about one fifth of the wetland, is home to the federally endangered Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*). This sparrow is restricted to freshwater prairies within ENP, and is a useful indicator for the spatial extent of a habitat type called "short hydroperiod prairies." Water management protocols have increased hydroperiod and water depth in some areas that have historically supported sparrows. Taller vegetation with fewer grassy clumps, which may provide less habitat for nesting and foraging, now characterizes these sites. Combining satellite images with plant survey data can detect patches of vegetation in the freshwater prairies, with structural and floristic characteristics that are correlated with high sparrow density. These characteristics can be used to monitor the effects of water management changes on hydrologic conditions in the sparrow prairies. The current water management program has created water imbalances across the Everglades ecosystem, while agricultural and urban land use has decreased wetland area. The Comprehensive Review Study of the Central and Southern Florida Flood Control Project ("the Restudy") is an $8 billion restoration plan for the Everglades. Its initial assumptions and constraints prevent the consideration of alternatives that may be more ecologically and economically efficient, such as converting agricultural lands into above-ground storage reservoirs. A true restoration of the Everglades requires a water management system that provides appropriate hydrologic conditions for all native species throughout their range.

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PART I: EXECUTIVE SUMMARY

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INTRODUCTION

This dissertation examines conservation issues within the Everglades, a renowned wetland ecosystem in central and southern Florida, USA. Everglades National Park (ENP), which includes about one fifth of the incident watershed, is a Ramsar site, a World Heritage site, an International Biosphere Reserve, and home to over 2000 native plant and animal species (Figure 1-1) (Lodge 1994). The federally endangered Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*, American Ornithologists' Union 1999) is a useful indicator for the spatial extent of a habitat type called "short hydroperiod prairies." These prairies, in turn, are directly affected by water management decisions, and can therefore be used as indicators of the effect of water management decision-making. The expression of these prairies also illustrates the need for consideration of larger expanses of natural areas to maintain population resilience under many natural and anthropomorphic disturbance regimes. Policies (especially water storage) and restoration programs must be made with a broad understanding of Everglades communities and the system as a whole. The restoration of the Everglades system requires a reconnection across the entire watershed and enlargement of natural areas, and cannot be performed within the current administrative boundaries of the parks and preserves. Throughout this dissertation, I use the word "restoration" as defined by the Society of Ecological Restoration (1995):

> "Ecological restoration is the process of assisting the recovery and management of ecological integrity. Ecological integrity includes a critical range of variability in biodiversity, ecological processes and structures, regional and historical context, and sustainable cultural practices."

EVERGLADES PAST AND PRESENT

The current Florida landscape had its origins as a piece of the African continent, sheared off when Laurasia separated from Gondwanaland about 180 million years before present (YBP) (Klitgord *et al.* 1984, Opdyke *et al.* 1987, Dallmeyer 1987). As Northern Hemisphere glaciers advanced and retreated for almost 2 million years, the Florida land area expanded and shrank as sea levels fluctuated. Ecological communities oscillated between freshwater and marine communities along the coast, while inland "high elevation" (about 15 m (49 ft)) communities experienced dry, cool periods (glacial) and warm, wet periods (interglacial) (Brooks 1968, Perkins 1977). A meteoritic hit some 34 million YBP, along with successive reef formations, are most likely responsible for the formative topography, and hence hydrology, of the Everglades (Petuch 1987).

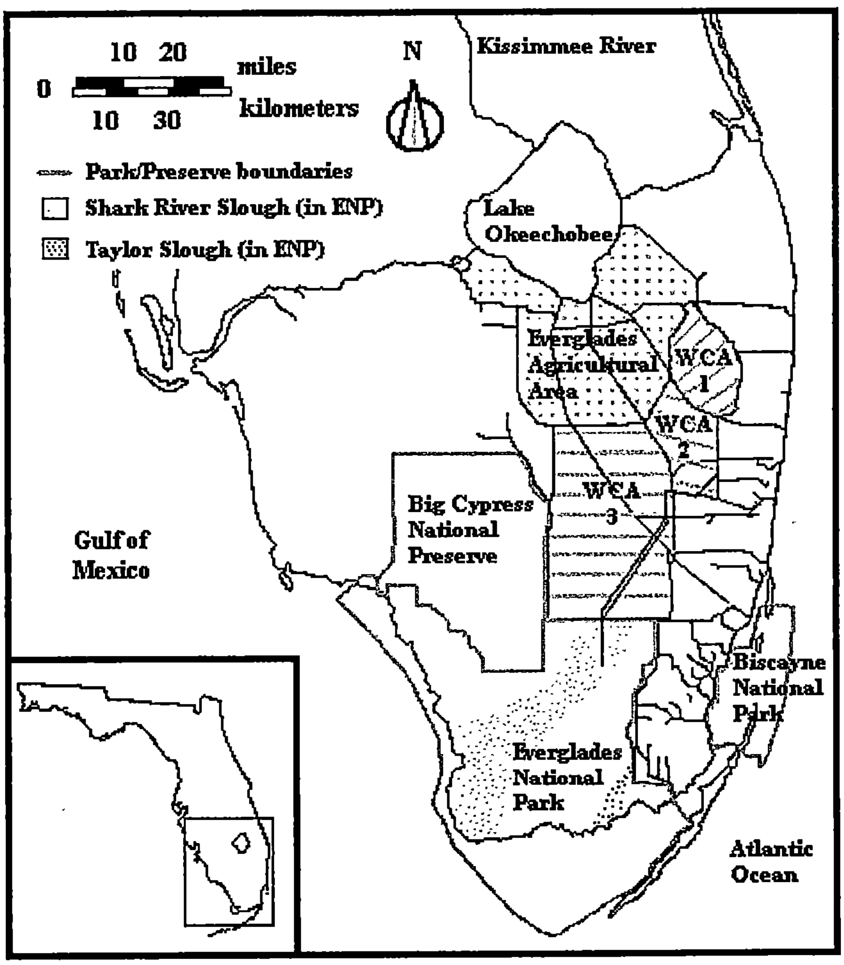

Figure 1-1. Map of the present Everglades watershed, including major canals, sloughs and political boundaries of parks and preserves.

The all-familiar "swamp and marsh" ecosystem characteristic of most of the extant Everglades probably developed about 5000 years ago, as evidenced by peat build-up along the interior (Gleason and Stone 1994). Such peat layers can increase by almost 10 cm (4 in) per century, and are formed primarily by wetland plant communities (Moore and Bellamy 1974, Gleason and Stone 1994). The earliest evidence of humans in the Everglades originates from about 10,000 YBP, when Florida had a much drier climate than today (Webb *et al.* 1984, Gleason and Stone 1994). Over 2400 YBP, native cultures practiced agricultural activity (primarily maize) in the expanding peatlands around Lake Okeechobee (Sears and Sears 1976). When European settlers arrived in the 1600's, about 20,000 Native Americans of the Calusa and Tequesta tribes inhabited the state (Tebeau 1968, Carr and Beriault 1984). The current Miccosukee and Seminole tribes settled in the Everglades region in the 1800's. With the arrival of European settlers, the human population of Florida quickly grew to 140,000 in 1860, and to over 14 million by 1998. It is expected to reach 20 million by 2020, with about 11 million people living within the Everglades watershed.

LEGISLATIVE HISTORY

Federal water and wetlands policy has profoundly affected Florida's environment throughout the state's history. Five years after Florida gained statehood, the 1850 Swamp and Overflowed Lands Act granted ownership of all wetlands to the state, with the expressed intent that the state should drain them for agriculture and development (Purdum *et al.* 1998). Six large canals drained water from Lake Okeechobee into the Atlantic and Gulf coasts by 1927, and a 136 km (85 mi) long levee along the southern shore of Lake Okeechobee was raised to 10 m (32 ft) (Light and Dineen 1994). In response to two 1947 hurricanes that overwhelmed the piecemeal drainage system, Congress passed the 1948 Flood Control Act, which authorized the Central and Southern Florida Flood Control Project. This project gave the U.S. Army Corps of Engineers (USACE) a mandate to drain and "reclaim" wetlands in Florida for the next 50 years. As one of its first undertakings, the USACE constructed a 160 km (99 mi) levee between the Everglades and the expanding coastal communities to the east. This levee lowered water tables in the east by 1.5 m (5 ft) (Light and Dineen 1994).

The 1970's brought a dramatic reversal in policy goals, as the Clean Water Act, the Water Pollution Control Act, and the Safe Drinking Water Act all targeted pollution of watersheds. While protection of wetlands lagged for decades, due to largely political disputes over the definition of a "wetland", wetland provisions in Congressional bills increased through the 1990's (Switzer 1998, Zinn and Copeland 1999). Along with the Coastal Wetlands Planning, Protection and Restoration Act of 1990 (Breaux Act), the Swampbuster and Wetlands Reserve Program provisions under the 1983 (and successive) Food Security Acts have preserved about 2833 km^2 (700,000 acres) of wetlands in the United States.

Florida's legislative record has paralleled the national pattern in most respects. The drought years in the 1930's inspired the State to establish water conservation commissions, eventually leading to the 1957 Florida Water Resources Act. This act created the first governmental agency (Department of Water Resources) that could regulate water withdrawals from surface and groundwater sources (Purdam *et al.* 1998). By the mid 1960's, most of the canals, levees, and spillgates were in place, but their hydrologic effects began to have serious negative consequences for Everglades National Park (ENP). The USACE initiated a "minimum water delivery" schedule in 1970 to prevent drought conditions in ENP. The 1970's brought as much environmental legislative activity in Florida as it did for the rest of the nation. The Florida legislature passed four laws that linked water control and conservation, natural areas protection, and population growth: the Environmental Land and Water Management Act, the Comprehensive Planning Act, the Land Conservation Act, and the Water Resources Act (Purdam *et al.* 1998).

By the early 1980's, attention turned to the quality and quantity of the state's groundwater supply. Pesticides contaminated numerous wells, and sinkholes (here, an indication of dropping water tables) began to appear throughout central Florida. The 1983 Water Quality Assurance Act attempted to address groundwater issues, and later the 1987 Surface Water Improvement and Management Act attempted to protect and restore the state's surface waters (Purdam *et al.* 1998). In 1990, the state legislature enacted Preservation 2000, a $3 billion program to acquire sensitive natural areas and incorporate them into the state's parks and preserves. However, all of the previous legislation did not prevent the federal government from suing the state over the deteriorating conditions in ENP. As a settlement, the state enacted the 1994 Everglades Forever Act (Florida Statutes 373.4592), which provided funding for a clean-up and restoration plan for the Kissimmee River and the Everglades. Florida has spent $2.8 billion over the past 25 years to purchase 8500 km^2 (2.1 million acres) for conservation purposes (Purdam *et al.* 1998).

The Restudy

The Comprehensive Review Study of the Central and Southern Florida Flood Control Project (referred to as "the Restudy") was authorized by Section 309(1) of the 1992 WRDA and Section 528 of the 1996 WRDA. The Restudy is a 30-year restoration program that attempts to restore a hydrologic condition that precludes further ecological decline in the Everglades. The plan recommends over sixty projects that will create almost 90 km^2 (35 mi^2) of reservoirs and water treatment areas, as well as 200 aquifer injection wells. These new facilities will attempt to end the damaging water level fluctuations in Lake Okeechobee, limit unseasonable freshwater discharges into the area's estuaries (which affect salinity conditions), and create more natural hydroperiods in the national parks and Water Conservation Areas. The USACE estimates the cost of the Restudy implementation at $7.8 billion, with $182 million in annual operation and maintenance costs (USACE and SFWMD 1999). The Federal Water Resources Development Act (WRDA) of 1996 obligated the Federal government to cover 50% of the costs of the Everglades restoration.

The USACE is leading the planning and organization of the Restudy. Other participating federal agencies include the U.S. Fish and Wildlife Service, the U.S. Environmental Protection Agency, the U.S. Department of Agriculture's Natural Resources Conservation Service, the National Park Service, and the U.S. Geological Survey. Participating state agencies include the South Florida Water Management District, the Florida Game and Fresh Water Fish Commission, and the Florida Department of Environmental Protection. Several universities, stakeholder groups, and Native American tribes also participated in the development of this plan.

So far, the reaction to the Restudy has been mostly adversarial. Urban residents complain that the agricultural industry should bear more of the financial burden, especially concerning water quality improvements. Agricultural and rural stakeholders feel unduly pressured to sell their land or have it be flooded, and criticize the burgeoning urban population and its increasing water demands. Scientists warn that the Restudy doesn't do enough to restore the ecosystem, and that sacrifices by both agricultural and urban sectors must be made. Despite the criticism, an important obstacle for the Restudy is Congress; it will decide whether to allocate the $4 billion in Federal funds needed to get the restoration project underway.

SUMMARY OF PART 2

Everglades National Park (ENP) is a major part of a network of natural reserves that includes Big Cypress National Preserve, Biscayne National Park, Loxahatchee National Wildlife Refuge (Water Conservation Area 1), and two other Water Conservation Areas. These areas incorporate over 10,000 km^2 (3861 mi^2), about half of the predrainage Everglades ecosystem (Figure 1-1). This natural preserve network joins 62 other parks and preserves in the Western Hemisphere that are this large. Established in 1947, ENP encompasses 4363 km^2 (1685 mi^2), 15% of the original extent of the Everglades system (Light and Dineen 1994). Like many globally recognized natural areas, neither the size nor the delimited boundaries of ENP isolate it from a growing human population and its attendant pressures. Because ENP protects only the bottom fifth of a watershed, activities that impact the flow and quality of water outside of ENP affect it as well.

The entire range of the Cape Sable seaside sparrow is within the boundaries of ENP. Between 1992 and 1995, the number of Cape Sable seaside sparrows declined by over half, mostly in the population on the west side of ENP (Curnutt *et al.* 1998). Among the possible causes of this population decrease (natural biological fluctuation, hurricane, fire, and inappropriate hydrology), excess water due to uninformed water management decisions seems to be the most likely. Shortened hydroperiods, leading to increased fire frequency, is the most parsimonious explanation for the disappearance of several of the smaller subpopulations. Although ENP is one of the largest parks in the U.S., it is still not "large enough" to protect the endemic species within it, since it does not include the entire interacting system. The decisions for the hydroperiod within ENP are not made solely with the needs of ENP as a priority, but rather include consideration of the surrounding urban and agricultural areas. As a result, the water regime within ENP is not adjusted to favor all the resident species.

SUMMARY OF PART 3

The Cape Sable seaside sparrow inhabits the freshwater marl prairies within ENP. All other subspecies of seaside sparrow, including the extinct Dusky seaside sparrow (*Ammodramus maritimus nigrescens*, American Ornithologists' Union 1999), prefer saltmarshes along the Atlantic and Gulf coasts (Post and Greenlaw 1994). When Howell first described the Cape Sable subspecies in 1919, he found it in freshwater *Spartina bakeri* marshes on the Cape Sable peninsula. Later, the Cape Sable seaside sparrow was rediscovered in freshwater prairies further inland (Stimson 1956). These prairies, classified as short-hydroperiod freshwater prairies, are typically dominated by graminoids such as *Cladium jamaicense*, *Muhlenbergia filipes*, and *Rhynchospora* sp.

Werner (1975) was the first to conduct detailed plant surveys in the inland Cape Sable seaside sparrow habitat, and included structural characteristics in his surveys as well as floristic composition. He found that a high "vertical obscurity" height (a measure of horizontal vegetation thickness) was correlated with sparrow occupancy, as well as a higher percent cover/abundance of *Muhlenbergia filipes* and lower percent cover/abundance of *Cladium jamaicensis*. A later sparrow survey by helicopter that recorded some vegetation characteristics also found more sparrows associated with *Muhlenbergia filipes* (Kushlan and Bass 1983). My results demonstrate that low maximum vegetation height, high effective height (similar to "vertical obscurity"), and the prevalence of *Rhynchospora* and *Schizachyrium rhizomatum* are also correlated with sparrow occupancy.

These marl prairies are subject to seasonal flooding, naturally brought on by heavy rains in the wet season, or unnaturally by spillgates and canals. The amount and timing of high water levels on these prairies not only affect the sparrow's breeding season, but also affect the vegetation that characterizes its habitat. On the western side of ENP and the southern portion of Taylor Slough (Figure 1-1), sparrow habitat has received more water and been inundated for a greater proportion of the year. The prairies in these areas are now dominated by *Cladium jamaicense*, which has increased the maximum height of the top of the vegetation, while decreasing its relative thickness. The increase in *Cladium jamaicense* cover/abundance between Werner's transects in 1975 and my transects in 1997 demonstrates an increase in average annual hydroperiod and average annual ponding depth on these prairies.

SUMMARY OF PART 4

Vegetation maps are critical to understanding the relationship between hydrology, elevation, disturbances, and the flora and fauna of the Everglades. Davis (1943) was the first to construct a habitat map of the entire Everglades ecosystem, combining previous vegetation surveys and aerial photography. Since then, mapping efforts have become more detailed and technical. Two concurrent efforts using satellite images, aerial photography, and wetlands and soil maps, are producing habitat maps for the Everglades with a high level of spatial and vegetational accuracy. The recent mapping schemes are dependent on the classification of the habitat into selective and detailed plant communities, an area where botanists have come to some (but not complete) agreement. Geographic Information Systems (GIS) have been instrumental in connecting all of these information sources. Researchers now can combine maps, point surveys, and hydrology reports, resulting in a better overall picture of the extent and distribution of the hundreds of plant communities in the Everglades.

The detail of habitat maps depends upon the plant survey data available. I combined my plant surveys with Landsat satellite images to add accuracy and detail to the prevailing freshwater prairie classifications. I focused my classifications on four study areas with known Cape Sable seaside sparrow occupancy, as well as one area (Werner's (1975) Taylor Slough site) which has been unoccupied since 1990 (Curnutt *et al.* 1998). I found seven different spectral classes in my study areas that are identifiable from satellite images. The cover of *Cladium jamaicense*, *Muhlenbergia filipes*, and the maximum vegetation height and effective height variables best differentiated these classes, as did an index value for the graminoids *Rhynchospora* sp., *Schizachyrium rhizomatum*, and *Schoenus nigricans*, and the herbaceous lily *Hymenocallis palmeri*. These classes may represent plant associations, expressed in areas with particular hydroperiod length and average depth of standing water above the soil surface. More accurately detailed maps of these prairies can better improve water management decisions, and promote an understanding of short-term and long-term consequences of water regimes on vegetation composition and structure.

SUMMARY OF PART 5

Although the Everglades receives over 1 m (3.3 ft) of rain per year on average, the distribution of rain is not consistent throughout the year or between years. Uninformed water management practices, such as the flushing of excess, wet season water to the estuaries instead of storing it above or below ground, have created subsequent periods of water shortages in south Florida. In addition, a burgeoning human population and continued loss of natural areas have further reduced the overall ecosystem's ability to store water. The Restudy proposes restoration without compromising the water demands of urban and agricultural areas, and recommends the use of Aquifer Storage and Recovery (ASR) technology for water storage. Engineering, real estate, and land acquisition data analyses from the USACE contend that ASR technology is the most cost-effective method for storage (per water storage unit). Reservoirs tend to cost more per unit as they require much more land than ASRs, and ASRs lose much less water to evaporation than reservoirs.

However, either changes in reactive policy or the inclusion of environmental costs and benefits can influence these cost data. The Food, Agriculture, Conservation and Trade Act of 1996 (1996 Farm Bill) as well as the Wetlands Reserve Program (under the Food Security Act of 1985) influence land value and land acquisition costs. Changes in the implementation of the Farm Bill can decrease the cost of land for above-ground reservoirs in the Everglades Agricultural Area (EAA). Uncertainty in the degree of water treatment needed can also increase the per storage unit cost of ASRs. If water price policy were influenced by water availability, water-conserving behavior in dry conditions could decrease the corresponding cost of water storage, since less land (or fewer ASR wells) would be needed. When the ecological benefits of reservoirs (such as flood protection, microclimatic influence and recreation) are accounted for, the cost per storage unit for reservoirs may be found equal to or less than that of ASRs. Finally, the use of above-ground water reservoirs may go further to achieve the primary restoration goal recommended by the Governor's Commission on a Sustainable South Florida and the U.S. Man and the Biosphere program. Well-managed reservoirs in the EAA can help increase the overall extent of wetlands in south Florida, and reconnect the fragmented Everglades system.

HOW THIS DISSERTATION CAN INFORM WATER AND WILDLIFE MANAGEMENT

The goal of this dissertation is to improve management and conservation of the Cape Sable seaside sparrow and the Everglades. Hydrology is the driving force that governs ecological patterns and processes in the Everglades ecosystem, and the hydrology that exists in most of the system does not resemble that of predrainage conditions. I focus on the influence of hydrology on the habitat conditions of one endemic subspecies, as an example of how changes in the Everglades hydrology can rapidly change communities. Maintaining the endemic species and ecological processes in the Everglades most likely requires the restoration of predrainage hydrologic conditions and a reconnection of the watershed. I hope this dissertation illustrates the importance of the following three themes:

1. *The primary influence of hydroperiod on the vegetation and the sparrow.*

In the freshwater prairies, as with most of the Everglades plant communities, hydroperiod directly influences species composition and structure. If the prairies remain too wet for too long, *Cladium jamaicense* can dominate, which significantly increases the expressed height of the habitat. Although sparrows use interspersed *Cladium* flower stalks for singing and perching, such stalks provide less of an advantage if the surrounding vegetation is similarly tall or taller. Increased hydroperiod and ponding depth may discourage the growth and persistence of bunchgrass species, such as *Muhlenbergia filipes* and *Rhynchospora* sp., that can provide thick clumps for sparrow nest sites. If the water rises too early in the spring, the sparrow's nests (which are only an average of 10 cm (4 in) above the substrate, Werner 1975) can be flooded. Taller, thicker grass clumps allow the sparrows to build their nests higher above the ground. Managers can survey these prairies for sparrow suitability by measuring the comparative cover-abundance of the dominant grasses and sedges, especially *Cladium*, *Muhlenbergia*, *Rhynchospora*, and *Schizachyrium*. Also useful are structural characteristics, such as the average maximum height of the vegetation, the effective height (density) of the grassy clumps, and possibly periphyton cover, which may indicate open areas on the ground for sparrow foraging and movement. These variables can also be surveyed over time to determine the effect of different hydrological regimes on the prairies.

2. *The dynamic nature of the Everglades system and disturbances.*

Plant communities also respond quickly to hydrologic alterations, as vegetation changes in Taylor Slough and changes observed by other researchers indicate (Alexander and Crook 1975). Longer hydroperiods and deeper average ponding depth were correlated with an increase in maximum vegetation height and *Cladium* predominance, and a decrease in effective height and the predominance of *Muhlenbergia* and *Spartina bakeri* between 1975 and 1997. These changes were also correlated with the disappearance of sparrows in that area. Vegetation characteristics in Shark Valley, especially the decrease in *Muhlenbergia* from survey reports in 1992 to my survey in 1997, parallel those that have occurred in Taylor Slough. Changes in plant species composition in these prairies may not be as important to sparrows as the changes in structural features that attend these floristic changes.

Populations of native species may decrease to "fit" within the available habitat, which leaves them more vulnerable to extinction through demographic or catastrophic disturbance events. The 80% decline in the western population of sparrows after 1992 (Curnutt *et al.* 1998) left the entire subspecies more vulnerable to extinction by hurricane or catastrophic fire. As water management protocols change, their effect on the availability of suitable sparrow nesting and foraging habitat should be monitored on an annual basis, or as often as possible. The changing spatial distribution of this habitat should be monitored as well, so as to prevent all available habitat from being concentrated in one small area (thus increasing the likelihood of a catastrophic event affecting all sparrow populations at once). Some of the vegetation characteristics correlated with high sparrow density can be detected from satellite images, as I demonstrate in Part IV. Using remote sensing and GIS technology can greatly increase the feasibility of tracking sparrow habitat in space and time.

3. *The benefits of a large, contiguous natural area.*

Disturbance regimes and species and community distribution patterns operate on several scales in the Everglades (DeAngelis and White 1994). The Cape Sable seaside sparrow's history of variability in abundance and distribution over the southern Everglades landscape is a reflection of the large, interacting and dynamic system across which it evolved. Restoration policies that increase the breadth and contiguity of the parks and preserves should be weighted more favorably than those that do not. Large areas are less likely to have their entire expanse impacted by a disturbance (even a catastrophic one). Once an ecosystem is reduced and fragmented, each disturbance will affect a greater proportion (if not all) of the total habitat available to a species. Both the scale of habitat patch size and the scale of disturbances should be considered in management decisions.

The Restudy does not provide a comprehensive analysis of the costs and benefits of all water supply options available to south Florida. Few environmental and social costs and benefits were included in the alternative presented by the Restudy. This alternative includes a water storage technology (Aquifer Storage and Recovery) that is untested at the proposed scale. Although the Restudy aspires to be a "restoration" plan, it gives no value in its water storage decisions to the ecological changes most needed to restore a more natural hydrologic regime; a connected, enlarged wetland system. Using above-ground reservoirs, optimally between Lake Okeechobee and the water conservation and natural areas to the south, is a water supply alternative that may increase the restorative capacity of the Restudy.

FUTURE RESEARCH NEEDED

The research in this dissertation does not appreciably diminish the amount of work that is still needed in the Everglades and on the Cape Sable seaside sparrow's habitat. Much of this work concerns the level of detail and accuracy of defining "suitable" or "preferred" sparrow habitat. It also draws attention to the dynamic ecology of the sparrow prairies, and how disturbances might affect the distribution, quantity, and quality of these prairies in the Everglades landscape. Finally, much more work is needed on the reliance of the growing south Florida human population on this fragmented, deteriorating ecosystem.

1. *Study the dynamic behavior of the prairies.*

This includes vegetative surveys done in areas under different hydrological conditions and regimes, and of different ages post-fire. The "quality" (as described below) and extent of these prairies should be delimited, and examined for patterns in abundance, patchiness, and range. Spectral signatures from satellite and aerial images can be used to track changes in the habitat, but these signatures must be very detailed and precise as indicators. Jenkins and Powell (1998) have used spectral characteristics at nest sites to classify habitat in Everglades National Park as suitable or unsuitable breeding habitat for sparrows. The amount of suitable habitat varies widely from year to year, depending on hydrologic conditions. Some structural vegetation variables that are important to sparrows may be visible from remote sensing images. These variables are perhaps more important than defining "vegetation communities" or alliances for sparrow conservation. Jenkins *et al.* (1999)

2. *Measure the "quality" of sparrow habitat and "habitat selection".*

Changes in size and character of sparrow territories should be followed throughout the breeding season and across years. Vegetation surveys should be conducted in patches of habitat with unusually high use (e.g. for foraging, nesting). These studies should focus on vegetation characteristics such as maximum vegetation height, effective height, dominance of *Cladium jamaicensis* versus *Muhlenbergia filipes*, and the role of other dominant graminoids, such as *Rhynchospora* sp., *Schizachyrium rhizomatum*, and *Schoenus nigricans*. This information may help determine whether habitat is "optimal" or "suboptimal" in terms of specific activities that influence sparrow persistence, such as breeding and foraging. Finally, research must explore the role of *Muhlenbergia* and *Spartina* in this "optimal" habitat and the adaptability of sparrows to new habitat. If *Muhlenbergia* disappears due to a new hydrologic regime imposed on the landscape, will another bunchgrass take its place?

3. *Determine the relationship between scale and habitat suitability.*

"Sparrow habitat" may be different at different scales. Research should examine the differences in apparent habitat choice for nest sites, territories, and subpopulations. Lockwood *et al.* (1999) surveyed the vegetation at over 100 nest sites, and compared those sites to areas with no nests. Several vegetation characteristics were distinctive at nest sites, including higher predominance of *Muhlenbergia filipes* and more prevalent grassy tussocks. The role of disturbance and its scale (e.g. fires, hurricanes, and floods) should be examined as a possible source for providing habitat patchiness and diversity.

4. *Conduct socioeconomic research on increasing the amount of land conserved in natural areas.*

The analysis I present here (ASR wells versus above-ground reservoirs for water storage) is one piece of a huge problem; how can a growing human population and its activities fit into a natural environment and not destroy it? Further research should determine (to the extent possible) the dependency of the human population on the numerous functions of the ecosystem. The sustainability limits for human population and demand for water (for agricultural and urban uses), and sustainable alternatives of land use patterns need to be determined. This research should also address the cost-effectiveness of replacing (where feasible) ecosystem functions with technology, such as water treatment facilities versus the use of wetlands. Of course, these issues depend on the definition of what exactly comprises an ecosystem function, how to decide which values (direct, indirect) should be used to determine the "worth" of that function, whose judgements should be included in the analysis, and how precise these adjudged values can even be. These investigations should be used to determine the effectiveness of policies designed to "protect" the Everglades system, especially those policies that depend on cost-benefit analyses to prioritize the application of restoration funds.

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PART II: INTEGRATING ENDANGERED SPECIES PROTECTION AND ECOSYSTEM MANAGEMENT: THE CAPE SABLE SEASIDE SPARROW AS A CASE STUDY

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ABSTRACT

National parks and other biological preserves come in all shapes and sizes, but globally protect about 5% of the land's surface. One goal for parks, large or small, is to maintain as large a set of native species as possible. Large parks, which we define here to be roughly 10,000 km^2 (3861 mi^2), or 1° latitude x 1° longitude block, may achieve this goal more successfully. Parks of this size are both rare and notable. We discuss Everglades National Park (ENP) and a subspecies found only within its boundaries — the Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*). Chance fluctuations, natural events and catastrophes threaten the species' survival. So, too, do managed activities inside and outside ENP that control fires and floods. The sparrow has declined precipitously in the past decade, due partially to environmental disturbance, and partially to uninformed management. Parks often resemble islands instead of smaller versions of the original ecosystem, due to an isolation produced by their highly modified surroundings. If endangered species are to persist in parks and preserves, we must be constantly vigilant in all parks, regardless of size.

INTRODUCTION

We should construct a global conservation system to encompass as many species as possible in protected areas. Such areas, "parks" for short, should require as little time, effort, and money to manage as is possible. How large must a park be to maintain its original species complement or for its ecosystem processes to be self-sustaining? This is an ecological question, but the answer comes as a statement of what is politically feasible. We will show that areas larger than 10,000 km^2 (roughly a square of one degree of latitude and longitude) are few. In our experience, such large parks are likely to be well known, flagships of their countries' efforts to protect the natural environment. It is on the lifestyles of one of these rich and famous parks that we concentrate.

Everglades National Park (ENP) and its adjacent resource conservation areas cover 10,000 km^2 (3861 mi^2) and encompass most of southern Florida. While the national parks include species persistence in their management priorities, not all of the 10,000 km^2 share the same focus. In the Western Hemisphere, few parks are larger or enjoy comparable prestige and associated financial resources. Nonetheless, we show that such areas will likely retain their species diversity only through constant vigilance and active, informed ecosystem management.

PARKS IN THE WESTERN HEMISPHERE

Currently, protected areas encompass ~5% of the earth's land surface, so perhaps the current rates of biological extinction should not surprise us (Ryan 1992). In industrial nations such as the United States, national parks are often islands surrounded by a sea of agriculture, commercial, and urban development. Even the largest of parks suffer invasions by exotic species and disturbance due to heavy tourism (Houston 1971). Roads and trails through parks increase the vulnerability of its interior to "edge effects", eliminating some of the advantages of large size. In less-industrialized countries, sometimes pristine habitats — but more often areas degraded by fires, clear-cutting, and agriculture — surround the park.

This "parks as islands" metaphor suggests an obvious calculation. In archipelagos of oceanic islands, the number of species scales as area raised to the 1/4 power. This is the familiar species (S) to area (A) relationship $S = cA^{1/4}$, where c is a constant (Rosenzweig 1995). Should the 1/4 estimate be accurate for continental areas, protect only 5% of the land area and one might lose ~50% of the planet's species.

At least two factors mitigate this bleak prospect. First, some species will survive in the human-dominated 95%, without dependence upon habitat in parks. The number of survivors will depend on a complex of factors. These include the quality of habitat patches in the park's surrounding matrix, and the adjustments of the particular species to this matrix. For many areas with burgeoning human populations, extensive parks may represent the best hope for protecting biodiversity.

Second, we might be judicious in our choice of the 5%, protecting those areas rich in otherwise vulnerable species. Extinctions concentrate in "hot spots" — areas of many species with small geographical ranges (Myers 1990, Pimm *et al.* 1995). For example, the great majority of the world's ~1,100 bird species deemed likely to soon become extinct have geographical ranges less than 50,000 km^2 (19,305 mi^2) (Collar *et al.* 1994). Protecting such range-restricted species in their special habitats may save more than 50% of the planet's biodiversity.

A third factor exacerbates the potential loss of species from the 5% of protected land. Unlike oceanic islands, parks, particularly small ones, are surrounded by areas whose land use practices may severely impact their flora and fauna. Examining these impacts is this paper's major theme.

We start by asking how many parks are there? Table 2-1 catalogues the area protected by parks, the total number of parks and the number of large parks in each country in the Western Hemisphere.

The amount of land set aside in parks does not increase significantly with the size of the country (World Resources Institute 1994). Wealth and aggressive conservation policies are likely key factors. Although Ecuador is only 2% of the size of North America, its proportion of land classified as "protected" (40%) is over twice that in the United States and Canada combined (16%). Large parks are few. Outside the United States and Canada, there are only 21 of them. Large parks are surely better than small ones for species persistence. By decreasing edge effects, large parks may also be better able to preserve its flora and fauna in the midst of encroaching environmental degradation and human activity. Our key question is: even given their size, are flagship parks up to the task?

Table 2-1. Parks and other protected areas in the Western Hemisphere (Source: World Resources Institute 1994). Countries are ordered by increasing size.

North and Central America	Total area in country (km^2)	Area protected (km^2)	Number of parks	Number of parks over 10,000 km^2
Trinidad, Tobago	5,130	180	9	0
Jamaica	10,830	20	1	0
El Salvador	20,720	190	5	0
Belize	22,800	2,910	10	0
Haiti	27,560	100	3	0
Dominican Republic	48,380	10,480	18	0
Costa Rica	51,060	6,210	25	0
Panama	75,990	13,280	15	0
Guatemala	108,430	8,330	17	0
Cuba	109,820	8,940	57	0
Honduras	111,890	5,430	38	0
Nicaragua	118,750	9,520	21	0
Mexico	1,908,690	98,970	60	2
United States	9,166,600	984,560	937	21
Canada	9,220,970	494,480	411	12
South America	**Total area in country (km^2)**	**Area protected (km^2)**	**Number of Parks**	**Number of parks over 10,000 km^2**
Surinam	156,000	7,360	13	0
Uruguay	174,810	320	8	0
Guyana	196,850	590	1	0
Ecuador	276,840	111,360	15	1
Paraguay	397,300	14,830	19	0
Chile	748,800	137,150	65	5
Venezuela	882,050	275,340	104	5
Columbia	1,038,700	93,910	79	2
Bolivia	1,084,380	92,500	26	4
Peru	1,280,000	41,760	22	2
Argentina	2,736,690	93,360	100	2
Brazil	8,456,510	277,420	214	4

We now turn to ENP. It is a "flagship" park surrounded by ocean on one front, the south Florida urban sprawl on the other, and water storage areas above it. In particular, we examine the ability of this park to maintain the Cape Sable seaside sparrow. This bird does not migrate, ENP includes its entire range, and its individuals hold territories typical of such a small passerine species. Can the park allow sparrows to prosper within it while humanity monopolizes its surround?

EVERGLADES NATIONAL PARK AND OTHER SOUTH FLORIDA PARKS AND PRESERVES

The comic referral of the two epochs in southern Florida, BC ("before canals") and AD ("after drainage"), betrays the seriousness of the hydrological transformation that has occurred there (Brown 1948). Before canals, the "River of Grass" was a 28,205 km^2 (10,890 mi^2) shallow river, extending from Lake Okeechobee in central Florida south to Florida Bay (Figure 2-1) (Light and Dineen 1994). At its peak flow in the wet season, it averaged 64 km (40 mi) wide and 0.5 m (1.6 ft) deep (U.S. Department of Interior 1994). The relatively flat terrain (0 to 6 m (20 ft) above sea level) allowed the water to flow in a vast sheet across the landscape, at up to 0.6 m (2 ft) per minute (Rosendahl and Rose 1982).

The natural hydrology proved disastrous to fledgling agricultural endeavors (Douglas 1988). The wet season rains flooded many fields and villages. Their inhabitants had constructed them on seemingly "high" ground in the dry season. To promote development in southern Florida, Congress passed the Swamplands Act of 1850. It paved the way for the construction of over 2,200 km (1366 mi) of canals and levees, and over 40 water structures, gates, and pumps over the next 140 years (Light and Dineen 1994). In the 1930's, the construction of massive levees that impound Lake Okeechobee delivered the most profound change to the natural hydrology of southern Florida. These projects have concentrated the flow of water into narrow, flooded tracts while leaving other areas unnaturally dry (U.S. Department of Interior 1994).

The original expanse of Everglades is now divided into three land use areas. Three park areas, ENP (4363 km^2; 1685 mi^2), Big Cypress National Preserve (BCNP; 2,306 km^2, 890 mi^2), and Biscayne National Park (BNP; 734 km^2; 283 mi^2), preserve about 26% of the original extent of the Everglades for recreational and educational use (Figure 1-1) (Light and Dineen 1994, U.S. Department of Interior 1994). Three "Water Conservation Areas" north of the national parks encompass 3,554 km^2 (1372 mi^2). These protect another 13% of the original expanse of the Everglades and the national parks from run-off from the "Everglades Agricultural Area" (3059 km^2; 1181 mi^2) downstream of Lake Okeechobee. They bear little resemblance to the original habitat (U.S. Department of the Interior 1994). In sum, these protected areas, though large, are embedded in a greatly modified landscape.

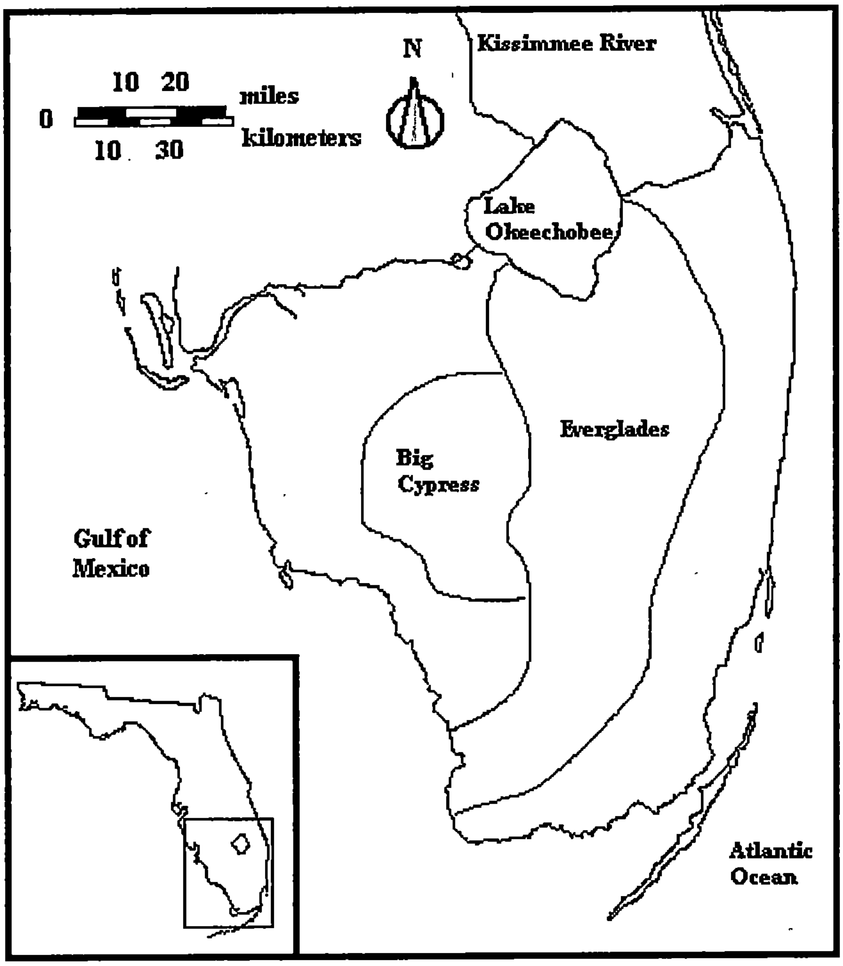

Figure 2-1. The historic drainage of the Everglades watershed, flowing uninterrupted from Lake Okeechobee to the Gulf of Mexico and Florida Bay.

Endangered and threatened species in south Florida

ENP, BCNP and BNP are known to support over 640 vertebrate species and 1650 plant species (National Park Service 1996). Forty-eight animal species are listed as federally endangered or threatened, as are 54 plant species (Florida Game and Freshwater Fish Commission 1997). These species depend on a mosaic of habitat types, including pinelands, mangrove forests, cypress swamps, and vast seasonally flooded prairies. Some of these habitats, such as the prairies, are highly dynamic, and can change rapidly in response to changes in hydrology (Nott *et al.* 1998).

Over 400 species of migratory birds depend on habitats in the region (U.S. Fish and Wildlife Service 1999), and are greatly affected by changes in hydrology (Kushlan 1979). Several resident breeding birds, such as the wood stork (*Mycteria americana*), snail kite (*Rostrhamus sociabilis*) and Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*), have experienced precipitous declines in the last decade. Critical to our discussion is whether our large parks offer enough protection from natural and human-caused disturbances. Alternatively, is active management essential or capable of saving these species from extinction?

As a case study, we describe our work on the ecology of the Cape Sable seaside sparrow and the factors inimical to its continued survival. As we discovered, proper management can negate some degrading effects. We can ameliorate other negatives only by providing a large and secure enough area for the sparrow to overcome adversity on its own.

THE CAPE SABLE SEASIDE SPARROW

The Cape Sable seaside sparrow's habitat preferences are unique among the seaside sparrows. They occupy seasonally inundated freshwater marshes and not the saltwater marshes used by other seaside sparrows. The closest relative of the Cape Sable race was the Dusky seaside sparrow. It became extinct in 1991 because of loss of habitat (Post and Greenlaw 1994).

The range of the Cape Sable seaside sparrow is presently fragmented and it occupies parts of that range episodically. Howell (1932) found the first population on Cape Sable, a peninsula in the southwest corner of ENP. In 1935, a hurricane flooded the prairies on the Cape with a 4 m (13 ft) saltwater storm surge. Eventually, this inundation caused major vegetational changes, destroying suitable sparrow habitat. In short, the sparrow is a freshwater sparrow, not a seaside sparrow, and it no longer occurs on Cape Sable.

Later populations, found in ENP and BCNP to the north, have disappeared and reappeared with regularity. Werner (1975) and Bass and Kushlan (1982) conducted extensive helicopter surveys of ENP and BCNP. Until then, the range of the Cape Sable seaside sparrow was unknown.

Current distribution (1981-1996)

Since 1992, we have continued extensive helicopter surveys first conducted by Bass and Kushlan (1982). The sparrow "metapopulation" is separated into two populations, one on each side of Shark River Slough (Figure 2-2). In a 1981 survey, Kushlan and Bass (1983) estimated about 2700 sparrows west of Shark Valley Slough, and over 3000 sparrows east of the slough (Curnutt *et al.* 1998). Sparrows on the eastern side of this slough are clustered into five fairly distinct subpopulations, which probably have a much higher genetic exchange with each other than with the western population. Before 1993, the western population (A) and eastern subpopulation (B) held the majority of the sparrows. Four other peripheral subpopulations (C-F) held small and variable numbers. In 1992, Hurricane Andrew passed directly over population A, and the 1993 survey revealed the near-extinction or displacement of that population. This population declined by 84% to about 400 sparrows, and had not recovered to 1981 levels by 1997 (Curnutt et al. 1998). Subpopulation B has remained approximately constant at about 2000 to 2500 sparrows each year. Two of the four peripheral subpopulations (C and F) were abandoned between 1992 and 1995, but were recolonized by small numbers in 1996. Peripheral subpopulations D and E have fluctuated in size since 1992.

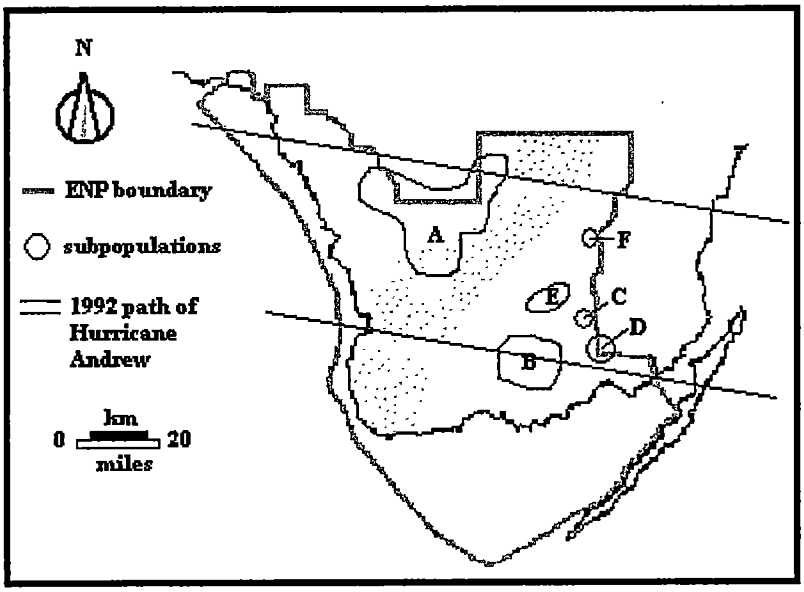

Figure 2-2. The path of Hurricane Andrew (parallel lines) and populations of Cape Sable seaside sparrows (outlined areas).

Hypotheses for the sparrow's decline

We have evaluated four phenomena that might explain these changes in sparrow abundance: chance population fluctuations, hurricanes, fire and hydrology (Curnutt *et al.* 1998, Nott *et al.* 1998). Each one could cause the extinction of the Cape Sable seaside sparrow. We cannot mitigate chance and hurricanes (apart from the costly process of transplanting individuals back into abandoned areas). We can manage fire and hydrology, but can we do so wisely?

Extinctions due to random fluctuations in populations have a higher probability of occurring in small populations (Pimm 1991). The four peripheral subpopulations have experienced several local extirpations and recolonizations. We first asked whether the duration of local extirpation in some of the smaller eastern subpopulations was longer than one should expect by chance alone.

We applied a population dynamics model developed by Curnutt *et al.* (1996) for grassland sparrows in North America. This model used population variability data for ten grassland sparrow species, obtained over twenty years from the North American Breeding Bird Survey. It estimates empirically the chance that a population of a given average size would be absent by chance alone for periods of 1, 2, 3... *n* years.

The model concluded that a three-year disappearance of birds from subpopulation C was unlikely to be due to chance alone. Three-year extirpations in similarly sized populations of grassland birds happened in less than 5% of the cases. In contrast, due to their smaller numbers, extirpations in subpopulations D and F were not significantly unusual. Finally, the large drop in subpopulation A from 1992 to 1993 had few precedents among the many small passerine species for which we have long term population counts.

We then explored whether hurricanes could better explain the unusual declines. Hurricane Andrew in 1992 swept over almost all of the subpopulations (Figure 2-2). It flooded little of the area, however. The storm was a relatively dry one, it passed east to west, and the storm surge affected only coastal areas. Our survey in 1993 found the greatest loss (84%) in population A, while other subpopulations were less affected. This western population had continued to decline into 1996. From the details of the local declines between 1992 and 1993 and the lack of recovery afterward, we conclude that Hurricane Andrew was probably not the principal cause of the lack of recovery of sparrow population A.

Hydrology has profound effects on sparrow habitat. During the breeding season (March through June), aboveground water levels higher than 10 cm (4 in) interrupt breeding activity. High water floods nests directly. It also leads to higher rates of nest predation (Lockwood *et al.* 1997). An increase in the length of the period of flooding — hydroperiod — also favors sawgrass (*Cladium jamaicense*) (Nott *et al.* 1998). Sparrows do not inhabit areas dominated by sawgrass. Although natural rainfall alters water levels and their duration, multi-targeted water management practices control most of the hydrological dynamics in ENP.

The decline of the western core population (A) coincided in time with large water releases in 1993, 1994 and 1995 from several floodgates to the north. These floodgates drain water from Water Conservation Area 3 into the park (Figure 1-1). Nott *et al.* (1998) obtained hydrological data on water levels, rainfall, and water releases from the floodgates at the north end of ENP. Statistical analyses demonstrated that water releases were crucial to determine the water levels in the sparrow's habitat (Figure 2-3). Rainfall was not as important. Simply, managed water flows were most likely responsible for the bird's precipitous decline in population A. Fire also affects the sparrows and their habitat. Sparrows abandon an area for several years after a fire, until suitable habitat has reestablished (Werner 1975). Curnutt *et al.* (1998) mapped fire frequency for areas that support relatively stable subpopulations, and compared these maps to changes in sparrow numbers found on our helicopter survey. Sparrow density was highest in sites that burned once or twice in a ten year period, and approached zero in sites that burned seven or more times in that period.

The extirpations of subpopulations C and F are likely due to the high frequency of anthropogenically set fires along ENP's border. The generally drier nature of these areas is due, in turn, to water management policies that move water to the west of Shark River Slough (Figure 1-1). Present management floods the western areas of the sparrow's range and overdrains the eastern ones.

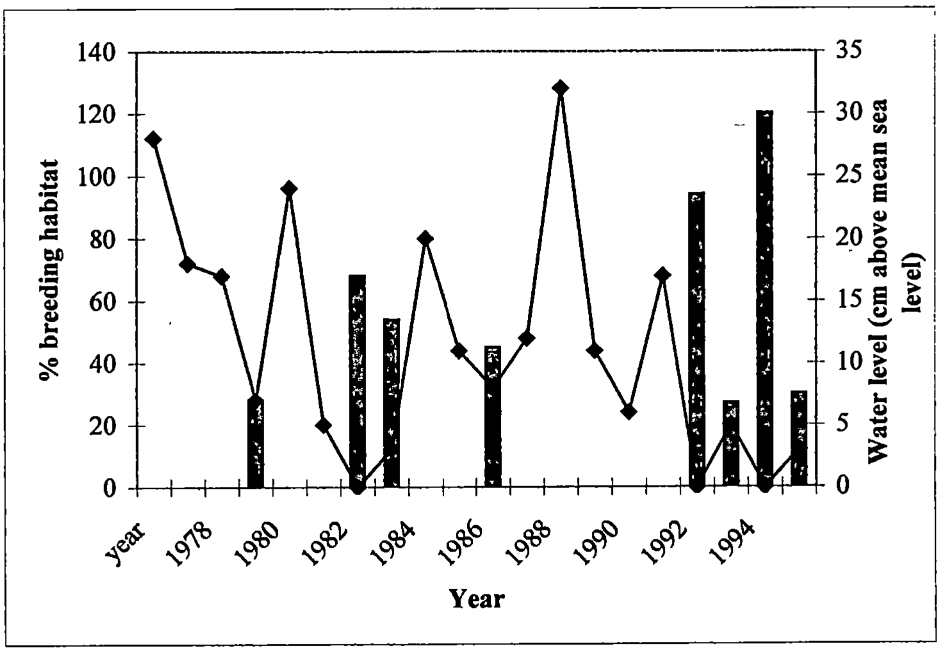

Figure 2-3. Water depth variation during the breeding season of Cape Sable seaside sparrows in population A. In wet years, there is almost no dry habitat in which the sparrows can breed (line with diamonds, left scale, percentage of maximum habitat possible).

We can only compensate for chance and hurricanes by protecting enough habitat to support a large sparrow population. Invariably, this would mean increasing the size of the park. For some parks, such as ENP, increasing the park into the degraded agricultural areas has proved expensive and politically difficult. In this way, ENP is much like a geographic island.

We cannot absorb all fire and hydrology impacts by simply providing an island that is large enough. Allowing only lightning-ignited fires to burn would approximate the natural fire regime and this is the current policy. The ability to completely prevent unplanned human ignitions, including arson, is limited. The hydrology of the Everglades cannot simply be left alone, the alterations intact, as it will never return to its pre-drainage flows. Water management must actively pursue water release programs that resemble the once natural flow. This hydrological influence also affects the frequency and intensity of fire in the park.

ECOSYSTEM MANAGEMENT AND CONSERVING ENDANGERED SPECIES

Parks as islands

The entire range of the Cape Sable seaside sparrow exists within the boundaries of ENP and BCNP — there is no occupied habitat elsewhere. In this regard, it is tempting to visualize the park as an island surrounded by unsuitable habitat. Many other parks are similar and this metaphor is certainly not a new one (Shafer 1990).

All populations vary in numbers over time. This variation may expose a small population to a low level from which it cannot recover, whereas a larger one may survive similar crises. Obviously, islands of habitat support populations of fewer individuals than did the habitat's original extent — thus island fragment populations are at greater risk of extinction. Moreover, when populations are on the verge of extirpation, immigrants will reconstitute nearby populations more often than isolated ones. These are two familiar mechanisms which explain why islands have fewer species than mainlands, and why park habitats, when isolated, might be expected to lose species.

The Cape Sable seaside sparrow offers an excellent illustration of these risks. Its metapopulation is fragmented into six isolated subpopulations of different sizes, whose numbers vary over time. The small, peripheral subpopulations have declined to zero for a few years, and were recolonized by sparrows dispersing from neighboring populations. As the total sparrow number declines within the park, the chances of extirpation for small subpopulations increase, and the likelihood of colonization from other subpopulations decreases.

Another threat to small, isolated populations is catastrophic disturbance. A hurricane may have a major impact on a sparrow population and so push it closer to extinction. If the disturbance has not drastically altered the entire vegetated landscape, sparrows from less-affected populations can recolonize the area. Large numbers of individuals spread over a large area is a species' natural defense against disturbance-caused extinction.

Such postulates are not unique to ENP. For large mammal species in western North America, Newmark (1987, 1995) found even 10,000 km^2 parks were too small to support a full complement of the historic assemblage. Only the Kootenay-Banff-Jasper-Yoho park assemblage, which covers almost 21,000 km^2 (8108 mi^2), showed no mammal extinctions since the delineation of park boundaries. Unfortunately, the success of this park may not be completely dependent on its large size, but may also rely on the relatively pristine condition of the surrounding area. Newmark (1995) found that survival of large carnivores in the western parks relied upon habitat conditions adjacent to the park.

ENP is not an island

Unlike true islands, where the oceanic surround poses little threat to the islands' species short of catastrophic disturbances such as storms, isolated parks have to contend with threats that penetrate the park's boundary. For ENP, these involve freshwater management and intrusive fire.

Water plays a key role for the sparrow. The South Florida Water Management District (SFWMD) controls the area's hydrology through a vast system of canals, gates and levees. As ecological knowledge accumulates, the SFWMD alters water distribution and delivery rates to ENP accordingly. From the sparrow's point of view, ENP is not an island: it is linked too tightly to the control of water outside the park and fires that originate elsewhere.

In general, the SFWMD must contend with several sources of adversity (U.S. Department of the Interior 1994). Drainage and flood control have dried and otherwise degraded historic wetlands, increasing fire frequency along the park's eastern boundaries. Freshwater diverted into estuaries on either side of ENP increases siltation, disrupting the saline balance of the waters and affecting fisheries and other marine systems in BNP. Human pressures, such as increased urban sprawl, agricultural demands for water, polluted runoff, and mining for limestone prevent the SFWMD from effecting optimal water delivery programs for habitat integrity.

Kushlan (1979) found a similar dependence on adjacent land for two wading bird species in ENP. Although overall, the community of wading birds in the park had dropped to 20% of its historic level, the decline was not equal among all species. In particular, the white ibis (*Eudocimus albus*), which has established breeding colonies outside park boundaries, did not experience as precipitous a decline as the wood stork (*Mycteria americana*). The wood stork remains dependent on breeding habitat in ENP. Water flow alteration and diversion by water management activity have severely altered its habitat. Even at 10,000 km^2, the "legal" delimitation of the Everglades-Big Cypress-Biscayne park trio may not be large enough to encompass its "biological" boundary (*sensu* Newmark 1985). This was once most certainly the entire watershed from Lake Okeechobee to Florida Bay.

Worldwide, flagship parks have a long, idiosyncratic list of external threats that jeopardize the survival of their unique species (Burton 1991). Oil spills constantly threaten the Northern Yukon National Park in Canada (10,169 km^2; 3926 mi^2). Flooding from hydropower projects and deforestation by wealthy ranchers and poverty-stricken squatters jeopardize the immeasurable biodiversity in Amazonia National Park (10,000 km^2) in Brazil. In Russia, the Baikal Region National Park (15,640 km^2; 6039 mi^2) must contend with oil spills and pollution in the air, soil, and water. And in the Serengeti National Park and Ngorongoro Conservation Area in Tanzania (31,339 km^2; 12,100 mi^2), cattle grazing and possible railroad construction compete with the impressive megafauna for water and other scarce resources.

CONCLUSIONS

Flagship parks are few. Even the largest areas are not large enough to be indifferent to the ecosystem processes that surround and impinge upon them. Endangered species — like the Cape Sable seaside sparrow — provide clues to how we must manage those processes. Roughly, the Everglades west is sometimes too wet and the east is probably always too dry. Redistribution of water from the west to the east would aid the sparrow's recovery. Such a redistribution would restore the water flows to something like their natural condition and perhaps begin to stabilize a favorable habitat.

Outside influences make it impossible to treat this park as an island unaffected by its surround. The Cape Sable seaside sparrow cannot survive natural disturbances in the face of profound hydrological alteration, even though the park boundaries protect the sparrow's entire range. The sparrow's response to appropriate changes in water management practices can be a good indicator of our ability to approximate the original water flow and timing. Good ecosystem management is essential for the sparrow's survival, and the sparrow's survival is a measure of the quality of that management.

More generally, we cannot assume that even if we design our parks to be technically large enough to support viable populations of the megafauna, the parks will no longer require our services. We must continually track the species within, paying close attention to dramatic declines that may correlate to human activity inside or outside park boundaries.

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PART III: CAPE SABLE SEASIDE SPARROW (*AMMODRAMUS MARITIMUS MIRABILIS*) HABITAT INDICATORS AND THE INFLUENCE OF HYDROLOGICAL CHANGES

PART III: CAPE SABLE SEASIDE SPARROW (*AMMODRAMUS MARITIMUS MIRABILIS*) HABITAT INDICATORS AND THE INFLUENCE OF HYDROLOGICAL CHANGES

ABSTRACT

The endangered Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*) is restricted to freshwater prairies within Everglades National Park (ENP). Although first discovered in *Spartina bakeri* (cordgrass) marshes, this subspecies now inhabits mostly *Muhlenbergia filipes* (muhly grass) prairies, as the distribution of *Spartina* marsh is greatly reduced. Water management protocols have altered the hydrologic patterns within ENP, and have affected the sparrow's habitat. I collected vegetation data from six sites that differed in sparrow density and hydrologic conditions. Out of the eleven vegetation variables I measured, maximum vegetation height, effective height, and percent cover/abundance of *Cladium jamaicense* (sawgrass), *Muhlenbergia filipes* and *Rhynchospora* sp. (beak-rush) proved most useful for characterizing habitat suitability. Low sparrow density areas and wetter areas had higher maximum vegetation height and percent cover/abundance of *Cladium*, and lower effective height and percent cover/abundance of *Muhlenbergia* and *Rhynchospora*. In two sites, changes in hydrologic patterns may have degraded suitable habitat.

INTRODUCTION

The Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*) is a federally endangered subspecies that inhabits the interior, freshwater marl prairies of Everglades National Park (ENP). These sparrows defend an "all purpose" territory during the breeding season, which includes nesting and foraging areas (Werner and Woolfenden 1983). Similar to other seaside sparrows, Cape Sable seaside sparrows are insectivorous during the breeding season and summer, but include seeds in their winter diets (Post and Greenlaw 1994). These sparrows forage primarily from the ground, gleaning insects from the surrounding vegetation (Post *et al.* 1983). Cape Sable seaside sparrows construct cupped or domed nests from 6 to 37 cm (2 to 15 in) above ground (depending upon the height of vegetation available to anchor nests), in tussocks of grasses of *Muhlenbergia filipes* (muhly grass) or *Spartina bakeri* (cordgrass) (Werner 1975, Post *et al.* 1983). The length of the breeding season is dependent on the duration of the dry season, from early February until the start of seasonal rainstorms in June or July (Werner 1975, Post and Greenlaw 1994). Nests flooded by rising water levels may be abandoned or at increased risk of predation (Lockwood *et al.* 1997).

This subspecies is restricted to several isolated prairies dominated by *Muhlenbergia filipes* or sparse *Cladium jamaicense* (called sawgrass, actually a sedge), although early research found the sparrow in *Spartina bakeri* marshes (Howell 1919, Semple 1936, Dietrich 1938, Nicholson 1938, Anderson 1942, Stimson 1944, 1948, 1956). All other seaside sparrows use coastal saltmarshes and brackish prairies along rivers, and the extinct Dusky seaside sparrow (*A. m. nigrescens*) used brackish and freshwater prairies dominated by *Spartina bakeri* (Post and Greenlaw 1994). Declines in all subspecies of seaside sparrow are correlated with increasing habitat loss and degradation as urban development along the Atlantic and Gulf coasts destroys saltmarshes and increases fragmentation between populations (Greenlaw 1992, U.S. Fish and Wildlife Service 1995).

For almost 25 years, researchers have been investigating what habitat cues Cape Sable seaside sparrows use in locating suitable breeding grounds (Werner 1975, Werner and Woolfenden 1983, Kushlan and Bass 1983, O'Meara and Marion 1985, Curnutt *et al.* 1998, Nott *et al.* 1998). Kushlan and Bass (1983) and Werner and Woolfenden (1983) identified several freshwater prairie communities which sparrows occupied. "Muhly prairie" consisted mostly of *Muhlenbergia, Schoenus nigricans* (black-top sedge) and *Cladium*. *Rhynchospora* sp. and *Panicum* sp. (spikerush) characterized "mixed prairie". Other prairie types included clumped *Spartina* prairie, unclumped *Spartina* prairie, and sparse *Cladium* prairie. Both studies found most sparrows in *Muhlenbergia* prairies. These past surveys of sparrow habitat may have been too general to discern the specific vegetation characteristics with which sparrows most likely occur. Kushlan and Bass (1983) surveyed the sparrow's entire distribution range by helicopter, but took limited vegetation data during the ten minute period at each survey point. Werner (1975) collected detailed vegetation data similar to the methods I employ, but surveyed only one site (Taylor Slough, Figure 3-1). Results reported here represent the first intensive habitat characterization across the entire distribution of the Cape Sable seaside sparrow.

During this century, water levels in south Florida have fluctuated beyond the extremes of predrainage conditions, especially in Shark Valley and Taylor Sloughs (Figure 1-1) (Light and Dineen 1994, Olmsted and Armentano 1997). Lower water levels in the middle of this century may have promoted the eventual disappearance of *Spartina* marshes and the spread of *Muhlenbergia* prairies (Olmsted *et al.* 1980, Werner and Woolfenden 1983). Although Davis (1943) reported *Muhlenbergia* in Big Cypress National Preserve, *Muhlenbergia* is not mentioned in any other early vegetation reports (Egler 1952, Craighead 1971). *Muhlenbergia* might have moved to predominance in freshwater prairies as late as 40 years ago.

In the mid to late 1960's, water management agencies increased water deliveries to ENP to aid declining species, such as the American alligator (*Alligator mississippiensis*, Mazzotti and Brandt 1994, Light and Dineen 1994). Curnutt *et al.* (1998) and Nott *et al.* (1998) suggest that these hydrological changes have increased the sparrow's probability of extinction, as high water levels have flooded the western population, and low water levels have encouraged shrub invasion and frequent fires in eastern subpopulations. These hydrology patterns can have direct effects on sparrow breeding activity (flooding nests), and indirect effects (altering breeding habitat). If water management activities maintain high water levels in sparrow breeding habitat, sparrows may not find sufficient dry areas for nesting. Indirectly, high water levels can alter the vegetative composition and structure of the prairies. Specific information on how these prairies may change as water management alters supply schedules to ENP is essential to this subspecies' chances for survival (O'Meara and Marion 1985).

This study expands previous knowledge of sparrow habitat with more detailed vegetation characteristics of areas with well-studied sparrow populations. I compare general habitat characteristics

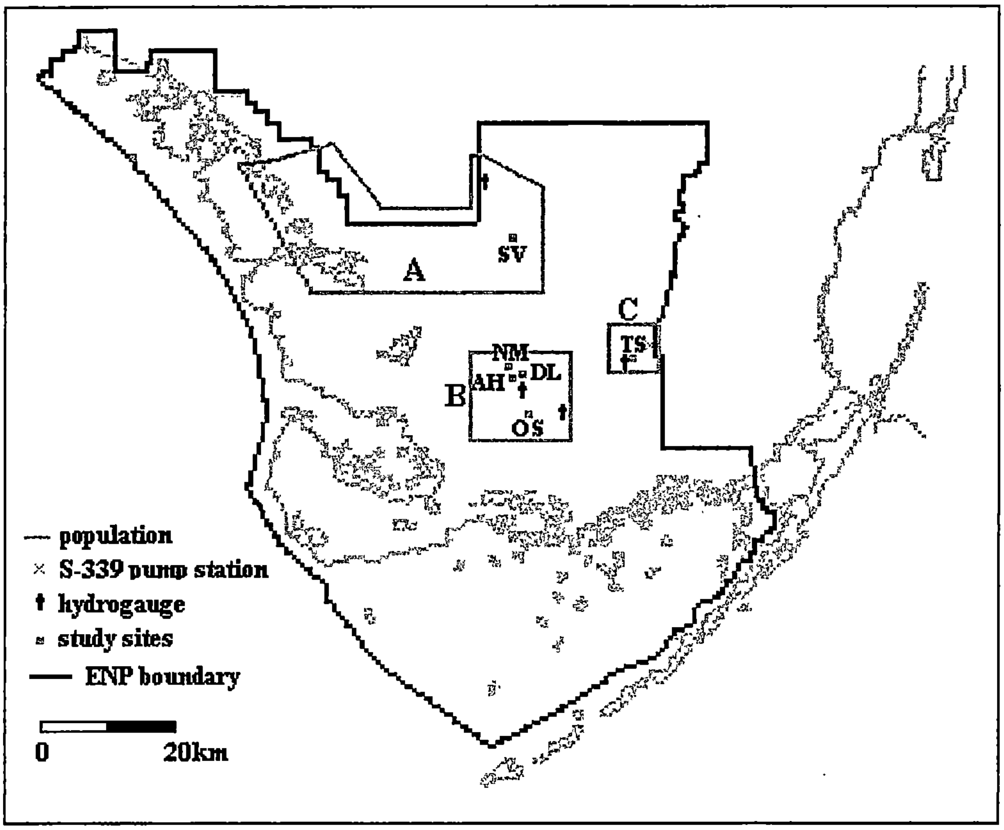


Figure 3-1. Map of Everglades National Park, Cape Sable seaside sparrow populations A, B, and C, location of six plant survey sites, and four water gauges. Site abbreviations: AH (Alligator Hammock), DL (Dogleg), NM (North Mahogany), OS (Old Ingraham Highway South), SV (Shark Valley), TS (Taylor Slough).

between sites that differ in sparrow density (during the breeding season) and hydrologic conditions. I also compare these results for Taylor Slough with those found by Werner (1975), to determine how hydrological changes alter the structure and species composition in these prairies over time. Vegetation characteristics that are correlated with high sparrow density can be used to determine the abundance and distribution of sparrow habitat throughout ENP. In areas where sparrows have once been common, vegetation characteristics may now indicate an inappropriate hydrologic regime. Although fire regime is important to these prairies, this study is not likely to demonstrate relationships between fire and vegetation characteristics. I did not include adequate sample sizes in all fire history age classes, but I did include years post-fire in my analyses to control for possible fire history effects.

STUDY SITES

Due to accessibility constraints, this analysis includes vegetation and hydrology data from only the sparrow population west of Shark Valley (A), and two of the five subpopulations east of Shark Valley (B and C). I conducted plant surveys from March to May 1997 on six sites within ENP (Figure 3-1). These sites differed in the number of sparrows supported (from 1995 through 1997) and hydrological conditions. Each study area had boundaries delineated either by permanent PVC poles, or by natural boundaries such as a pineland ecotone. To calculate sparrow density, I averaged the number of adult sparrows found on each site from 1995-1997, and divided this average by the site's area.

To compare hydrologic conditions between sites, I combined water gauge data (1985-1997; Figure 3-2) from the South Florida Research Station, ENP, with modeled hydrologic data from a 500 m^2 (5380 ft^2) resolution hydrology model produced by the Across Trophic Level System Simulation (ATLSS) project (Figures 3-3a and b) (S. Sylvester, pers. comm.). I used four water gauges: NP205 for Shark Valley; P67 and P46 for subpopulation B sites; and TSB for Taylor Slough. The gauges varied in distance to these sites, but no other gauges were more appropriate than these four. I averaged the modeled mean

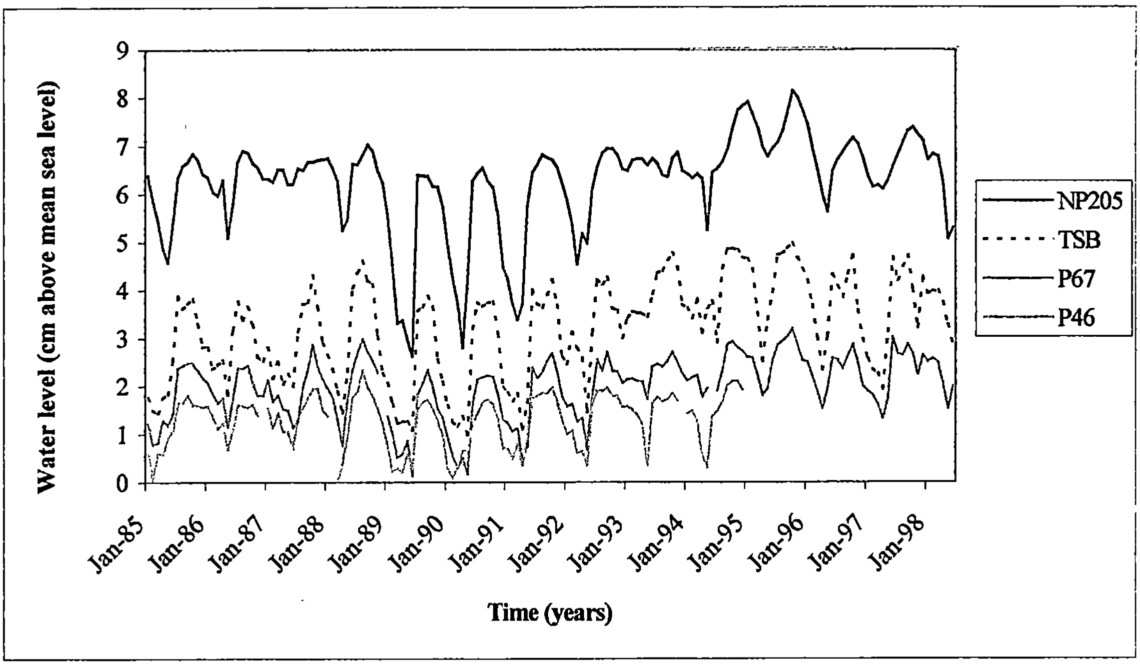
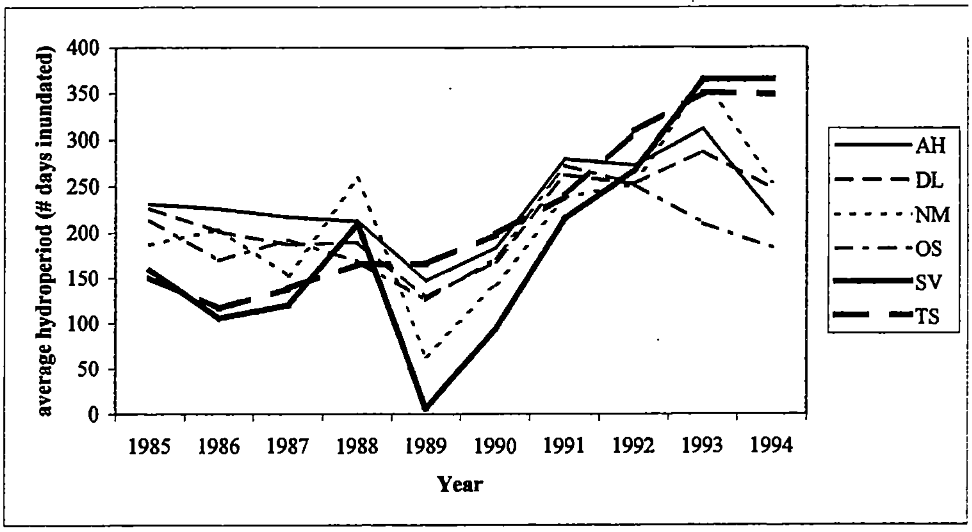


Figure 3-2. Hydrologic data from four water station gauges nearest my six study sites. Data do not account for topographic differences between gauge stations.

(a)



(b)

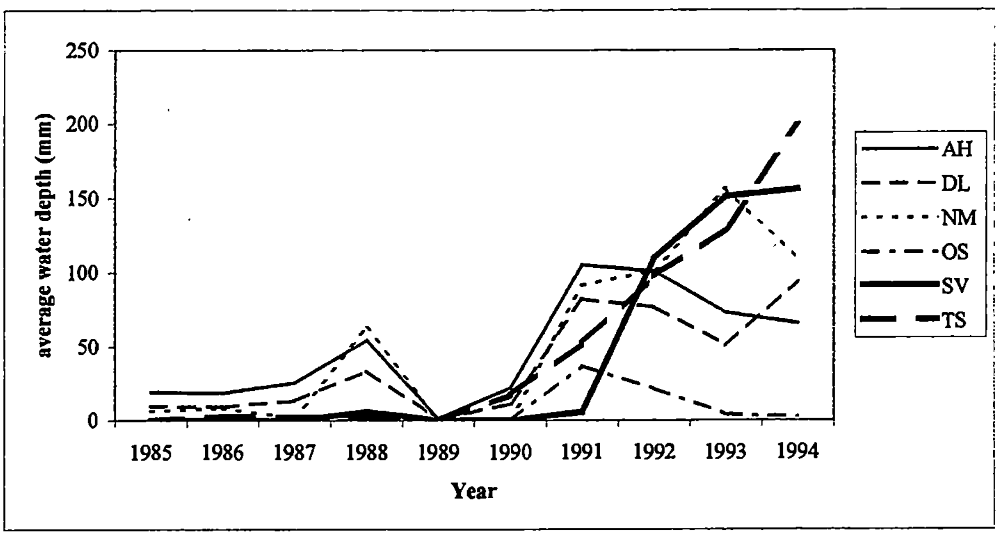


Figure 3-3. Average hydroperiod (a) and average ponding depth (b) for my six study areas, modeled for 1985 through 1994. After 1991, mean days inundated and water depth increase markedly on Shark Valley and Taylor Slough sites. (See Figure 3-1 for site abbreviations).

annual hydroperiod (number of days inundated, 1985 to 1994) and average water depth (in cm standing above the soil surface, 1985 to 1994) from the pixels in which I performed my transects. (A "pixel" is a square area 500 m^2 in size; the hydrology model uses a grid of pixels that cover the entire Everglades system). Each site corresponded to about six to nine 500 m^2 pixels from the modeled hydrology data. I interpreted the trends in hydroperiod, water depth, and water gauge data to subjectively rank my sites in order of "wetness" (Table 3-1). I obtained years post-fire data for my sites from fire boundary overlays (1981 through 1997) from the South Florida Research Center, ENP.

Population A

Shark Valley: This site was .41 km^2 (0.16 mi^2), located roughly 20 km (12 mi) south-southeast of the Shark Valley Tower at the north end of ENP. Vegetation surveys in the 1980's found moderate amounts of *Muhlenbergia* and *Cladium*, and a fairly high density of sparrows (Kushlan and Bass 1983). Population A was one of two "core" populations before 1992. This population declined by 80% after the 1992 breeding

Table 3-1. Rank of "wetness" of study sites based on modeled hydrologic data (1985-1994) and patterns from water gauge data (Figure 3-2). Modeled data do not reflect increases in hydroperiod and water depth in the Shark Valley and Taylor Slough sites from 1992 through 1997.

Site	Rank	Ave. annual hydroperiod	Ave. annual water depth
Old Ingraham Hwy. South (driest)	1	196.1 days/year	6.5 mm above soil level
Dogleg	2	215.1 days/year	38.2 mm above soil level
Alligator Hammock	3	230.0 days/year	48.6 mm above soil level
North Mahogany	4	212.0 days/year	53.8 mm above soil level
Shark Valley	5	190.8 days/year	42.9 mm above soil level
Taylor Slough (wettest)	6	218.2 days/year	50.3 mm above soil level

season, and has yet to recover. At the time of this study, the site held approximately 0.0044 sparrows per km^2, densities similar to those on North Mahogany (see below; J. Lockwood, pers. comm.). The modeled average annual hydroperiod around my transects was 190.8 days inundated, and the modeled annual water depths averaged 42.9 mm (1.7 in). Both of these modeled measures showed increases after 1991. At the time of my survey, 16 of the 20 transects were in areas 7 years post-fire, and the other four were 11 years post-fire.

Population B

Four of my sites were within the largest "core" population of sparrows in ENP. Extensive helicopter surveys over the sparrow's range have found relatively stable sparrow numbers in this population since 1981.

Alligator Hammock and Dogleg: Two of the sites, Alligator Hammock and Dogleg, were along Main Park Road, are each about 0.20 km^2 (0.08 mi^2), and supported approximately 0.01 sparrow per km^2 annually. The modeled average annual hydroperiods for the 20 transects across each of these sites were 230.0 and 215.1 days inundated, respectively. Modeled annual water depth averaged 48.6 mm (1.9 in) and 38.2 mm (1.5 in). In Alligator Hammock, one transect out of 20 total was in an area that was 6 years post-fire, and all other transects were more than 17 years post-fire. In Dogleg, 11 transects were 8 years post-fire, and the remaining 9 were 12 years post-fire.

North Mahogany: North Mahogany (0.29 km^2; 0.11 mi^2) was also along the Main Park Road, but supported only 0.0040 sparrows per km^2 from 1992 through 1997. The modeled average annual hydroperiod on this site was 212.0 days inundated, and water depth averaged 53.8 mm (2.1 in). Fourteen of the 20 transects were 8 years post-fire, 3 were 12 years post-fire, and the remaining 3 were more than 17 years post-fire.

Old Ingraham Highway South: Old Ingraham Highway South (0.3 km^2; 0.11 mi^2) was similar to Alligator Hammock and Dogleg (above) in sparrow density, but was at the end of Old Ingraham Highway, south and separate from the Main Park Road sites. The modeled average annual hydroperiod on this site was 196.1 days, and water depth averaged 6.5 mm (0.3 in). This entire site burned 8 years before the survey of the 20 transects.

Population C

Taylor Slough: The sixth plot, "Taylor Slough", near the eastern ENP boundary, supported about 400 sparrows in 1981, but surveys since 1990 have failed to locate any sparrows here (Curnutt *et al.* 1998). I concentrated 20 plant transects within a 0.41 km^2 (0.12 mi^2) area, divided by Main Park Road. My site is at the edge of Taylor Slough and directly downslope from a pumping station (S-332), activated in 1979. I collected data in the same area as Werner (1975). This presented a unique opportunity to look at the effects of hydrologic changes on vegetation in the past 22 years. The average annual hydroperiod on this site was 218.2 days inundated, and water depth averaged 50.3 mm (2 in). As at Shark Valley, both modeled hydrologic measures increased considerably after 1991. One transect burned less than a year before I conducted my survey, one was 16 years post-fire, and the remaining 18 transects were in areas more than 17 years post-fire.

METHODS

In the spring of 1997, I conducted plant surveys using a strip transect and 1 m^2 quadrat method. In each study site, I used a stratified random sampling technique to place twenty 50 m-strip transects. All transects were associated with UTM coordinates. I determined the direction of the transect by throwing a small plastic stake. I systematically placed ten quadrats per transect. This scheme yielded 200 1 m^2 quadrats of data per site, for a total of 1200 1 m^2 quadrats of data. This satisfies Mueller-Dombois and Ellenberg's

(1974) minimum sample area requirement for their "dry- grassland" and "hay meadow" habitat categories (the closest in structure to marl prairies) of 50-100 m^2.

In each 1 m^2 quadrat at every 5 m interval along the transect, I collected the following data:

Cover/abundance – I used the Braun- Blanquet method (Phillips 1959, Mueller-Dombois and Ellenberg 1974) to estimate cover/abundance from 1 (0 to 1.0%) to 6 (75-100%).

Dead matter cover/abundance – Using the same scale as above, I estimated the cover/abundance of dead plant matter in each quadrat. I counted dead leaves attached to living plants as "dead matter", along with debris on the ground.

Periphyton cover – Again using the same cover scale, I estimated the cover of the periphyton mat, which is a critical component of the marl prairies. Periphyton is a microalgal community that forms a thick mat above the soil. It is an extremely important primary producer in the Everglades ecosystem, and can profoundly affect water chemistry, sediment production, and hence plant communities (Browder *et al.* 1994). Hydrology and water quality directly affects periphyton diversity, cover and thickness (Browder *et al.* 1994).

Species composition – I took presence/absence data and estimated percent cover/abundance for every species in each quadrat. When identifying plants in the field, I referred to Hitchcock (1971), Godfrey and Wooten (1979), Cronquist (1980), and Taylor (1992), as well as specimens in the ENP museum (Appendix). Nomenclature follows that of Avery and Loope (1983). I also estimated the cover/abundance for each species, although here I only report the cover/abundance data for the four dominant graminoids: *Cladium*, *Muhlenbergia*, *Rhynchospora* sp. (*R. microcarpa* and *R. tracyi*), and *Schizachyrium rhizomatum.* Most beak-rush was *R. microcarpa*, so I combined measurements for the two species in this analysis.

Stem density/biomass – For each plant species, I counted the number of stems or tillers per quadrat. Pimm *et al.* (1995) conducted preliminary plant surveys, in which they counted stem densities of all species in a 1 m^2 area. They removed, oven-dried, and weighed all plant matter, and found a linear relationship between stem density and biomass. I used this relationship to indirectly estimate biomass in each quadrat based on my count data.

Maximum vegetation height – I measured the tallest plant in the quadrat in centimeters, which was especially critical for determining available perches.

Effective height (Wiens 1969) – Sighting on a field notebook, I recorded the maximum height (in centimeters) at which 90% of the side of the notebook was obscured by vegetation from 1-2 m away. I took this measurement while crouched, holding the notebook at eye level on the opposite side of the plot. Werner (1975) used a similar measurement, which he called "vertical obscurity profile". While this is a crude measurement, it allowed me to quantify an aspect of the habitat that may be important to the sparrow for predator evasion, thermal regulation and nest support sites.

I grouped the sites by sparrow density, with three sites (Alligator Hammock, Dogleg, and Old Ingraham Highway South) in a high density group, two sites (North Mahogany and Shark Valley) in a low density group, and one site (Taylor Slough) in an unoccupied group. I was unable to find more sites that were unoccupied by sparrows (but had a history of occupation), and that were accessible. I therefore have only one replicate of this category. I assumed that vegetation differences between sites of equal sparrow density were less than those between sites of different sparrow density, and combined all transects from the sites into the three sparrow density groups. I performed a Multivariate General Linear Model analysis (GLM) to determine overall differences between the density groups, and appropriate post-hoc tests to detect differences between the vegetation variable means of these groups. I used Bonferroni tests for significant differences between variable means when their error variances were equal across groups, and Tomhane's T^2 when they were not (SPSS Inc. 1997). An alpha value less than or equal to .05 determined significance.

My design was unlikely to pick up fire effects on vegetation. All sites were an average of at least 8 years post fire, and this is well within the recovery time for *Muhlenbergia* and *Cladium* prairies (Herndon and Taylor 1986, Loveless 1959, Steward and Ornes 1975). However, I included years post-fire and hydrology as fixed factors in a Multivariate GLM analysis to determine whether fire history could be an influence on vegetation differences between hydrologic groups. I established the following hydrologic order (from driest to wettest): Old Ingraham Highway South, Dogleg, Alligator Hammock, North Mahogany, Shark Valley, and Taylor Slough. The modeled hydrologic data demonstrated an increase in both hydroperiod and water depth from 1991 to 1994 in my Shark Valley and Taylor Slough sites (Figure 3-3a and b). Judging from the water gauge data, this trend most likely continued through 1997 (Figure 3-2). I used Multivariate GLM to detect vegetation differences between sites of differing hydrologic rank. Again, I used post-hoc Bonferroni tests of significant differences between variable means with equal error variances, and Tomhane's T^2 for those with unequal variances.

To determine differences between my plant survey results and those of Werner (1975) on Taylor Slough, I performed a t-test (testing first for equal variances and using the appropriate non-equal variance test when appropriate) between his 1975 transects (N=5) and my 1997 transects (N=20). I conducted all statistical tests using SPSS 8.0 for Windows (SPSS Inc. 1997).

RESULTS

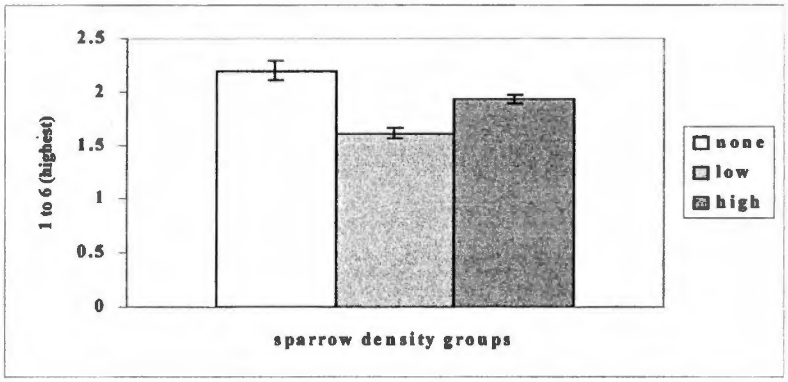
There were significant overall differences between the three density groups (Wilk's lambda=0.054, F=29.301, df=212, p=0.000). Several vegetation characteristics differed between sparrow density groups (Table 3-2, Figures 3-4a through k). Maximum vegetation height, dead matter cover/abundance, and percent cover/abundance of *Cladium* increased with decreasing sparrow density. Effective height, species richness, and biomass increased with increasing sparrow density. Live vegetative cover/abundance, percent cover/abundance of *Muhlenbergia* and *Schizachyrium* were lowest on the low density sparrow sites, while periphyton cover and *Rhynchospora* were highest on these sites.

I found an overall difference between vegetation characteristics in different hydrologic ranks (Wilks' lambda=0.153, F=5.434, df=44, p=0.000), but no difference based on fire history (Wilks' lambda=0.646, F=1.038, df=44, p=0.411) and no interaction between fire and hydrology (Wilks' lambda=0.755, F=1.346, df=22, p=0.142). Several vegetation characteristics differed with hydrologic conditions (Table 3-3, Figures 3-5a through k). Maximum vegetation height and percent cover/abundance *Cladium* increased with wetter conditions, as did periphyton cover (with the exception of Taylor Slough).

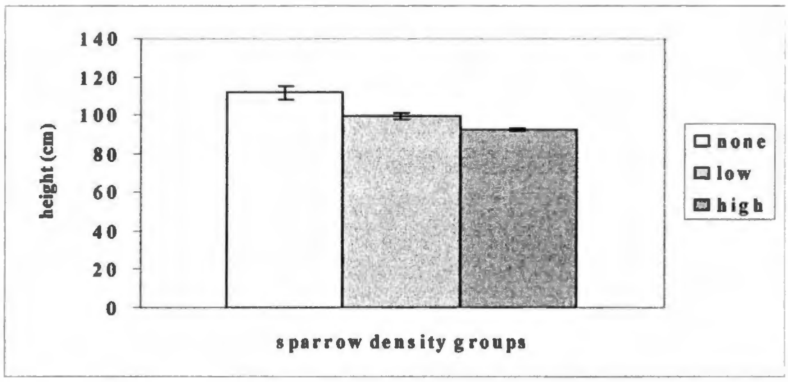
Table 3-2. Means and standard deviations of variables for density groups, and pairwise significant differences between sparrow density groups (none, low, and high). An asterisk indicates Tomhane's T^2 p values listed, otherwise Bonferroni p values are listed.

Variable	density group	mean	std. dev.	density group pair	p value
Cover/abundance	none	2.2	0.4	none and low	0.000
	low	1.6	0.3	none and high	0.014
	high	1.9	0.4	low and high	0.000
Maximum vegetation ht.	none	111.6	15.0	none and low	0.006*
	low	99.1	9.6	none and high	0.000*
	high	92.5	7.0	low and high	0.001*
Effective height	none	1.0	1.2	none and high	0.000*
	low	0.9	0.9	low and high	0.000*
	high	3.6	2.7		
Dead cover/abundance	none	2.3	0.6	none and low	0.000
	low	1.6	0.5	none and high	0.000
	high	1.7	0.6		
Periphyton cover	none	4.2	0.4	none and low	0.000*
	low	4.8	0.2	none and high	0.000*
	high	4.7	0.3	low and high	0.011*
Species richness	none	6.9	1.0	none and low	0.000*
	low	8.0	0.9	none and high	0.000*
	high	8.3	2.1		
Biomass	none	1707.2	339.7	none and high	0.004
	low	1839.1	447.0	low and high	0.030
	high	2063.7	408.8		
Percent cover/abundance	none	2.1	0.7	low and high	0.001*
Cladium	low	2.2	0.5		
	high	1.8	0.6		
Percent cover/abundance	none	1.0	0.9	none and low	0.034*
Muhlenbergia	low	0.4	0.5	low and high	0.000*
	high	1.5	0.6		
Percent cover/abundance	none	0.1	0.1	none and low	0.000*
Rhynchospora	low	1.7	0.7	none and high	0.000*
	high	1.2	0.5	low and high	0.002*
Percent cover/abundance	none	1.4	1.2	none and low	0.017*
Schizachyrium	low	0.5	0.4	low and high	0.000*
	high	1.3	0.3		

(a)



(b)



(c)

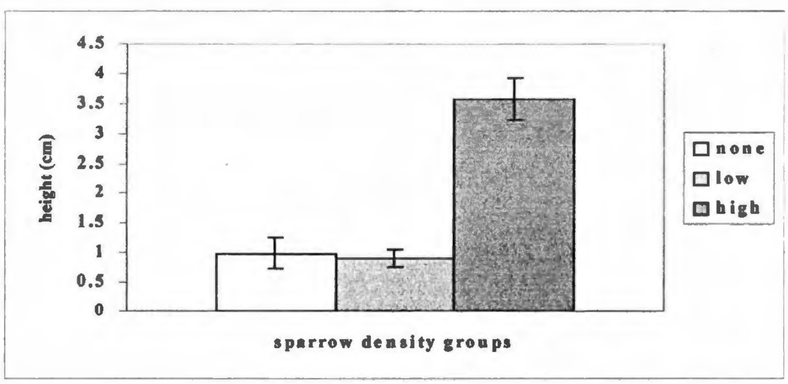


Figure 3-4. Three sparrow density groups and mean a) cover/abundance (all three groups differ significantly), b) maximum vegetation height (all three groups differ), and c) effective height (none and high, and low and high density groups differ from each other). Bars represent standard error.

(d)

(e)

(f)

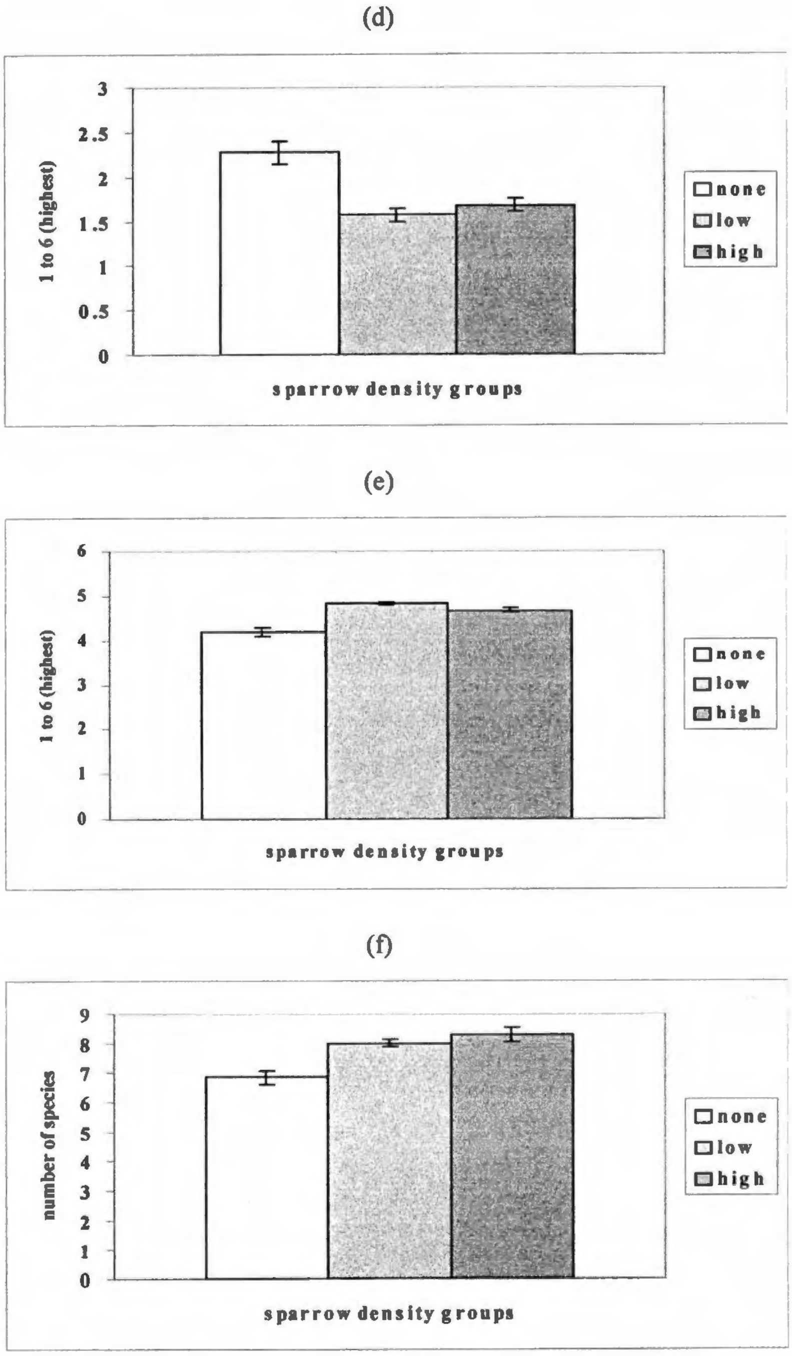
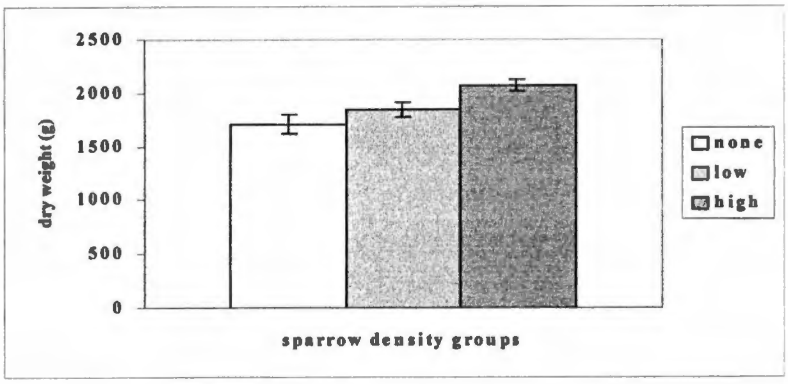
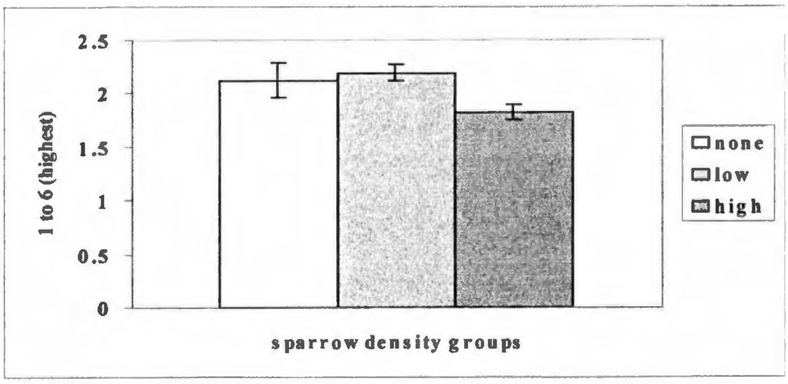


Figure 3-4 (continued). Three sparrow density groups and mean d) dead cover/abundance (none and low, none and high density groups differ from each other), e) periphyton cover (all three groups differ), and f) species richness (none and low, none and high density groups differ from each other). Bars represent standard error.

(g)



(h)



(i)

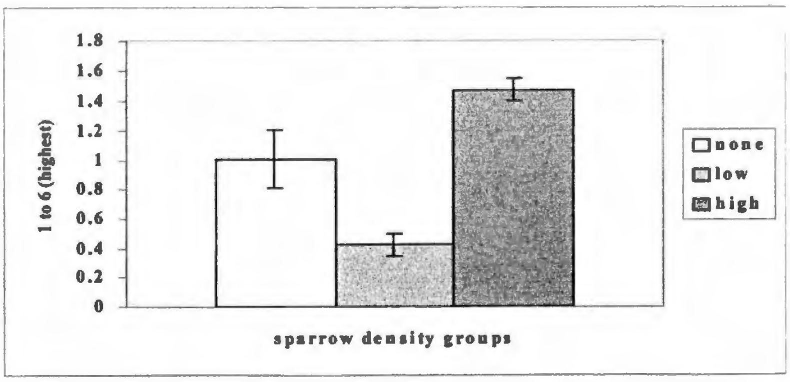
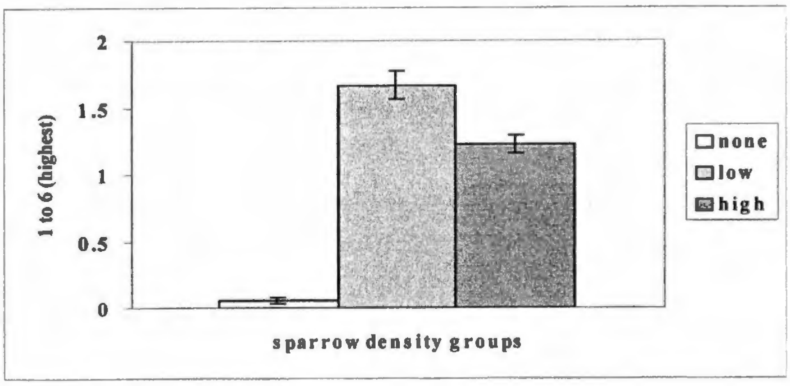


Figure 3-4 (continued). Three sparrow density groups and mean g) biomass (none and high, low and high density groups differ from each other), h) percent cover/abundance *Cladium* (low and high groups differ), and i) percent cover/abundance *Muhlenbergia* (none and low, low and high density groups differ from each other). Bars represent standard error.

(j)



(k)

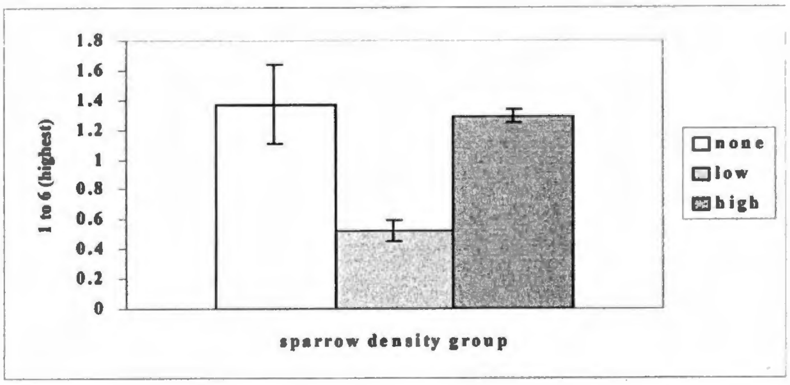


Figure 3-4 (continued). Three sparrow density groups and mean j) percent cover/abundance *Rhynchospora* sp. (all groups differ) and k) percent cover/abundance *Schizachyrium* (none and low, low and high density groups differ from each other). Bars represent standard error.

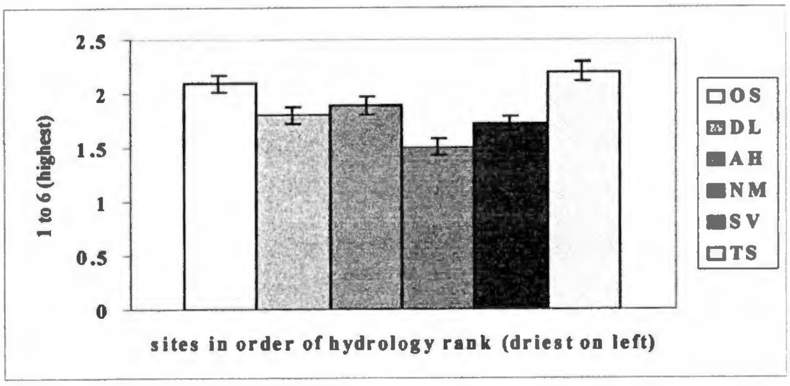
Table 3-3. Means and standard deviations of variables for hydrology groups, and pairwise significant differences between hydrology ranks (1 driest, 6 wettest). An asterisk indicates Tomhane's T2 p values listed, otherwise Bonferroni p values are listed.

Variable	hydrology rank	mean	std. dev.	hydrology rank pair	p value
Cover/abundance	1	2.1	0.3	1 and 4	0.000
	2	1.8	0.3	1 and 5	0.011
	3	1.9	0.4	2 and 6	0.005
	4	1.5	0.3	3 and 4	0.006
	5	1.7	0.3	4 and 6	0.000
	6	2.2	0.4	5 and 6	0.000
Maximum vegetation ht.	1	91.1	6.8	1 and 5	0.000*
	2	93.2	5.9	1 and 6	0.000*
	3	93.3	8.2	2 and 5	0.001*
	4	95.5	10.8	2 and 6	0.000*
	5	102.6	6.8	3 and 5	0.006*
	6	111.6	15.0	3 and 6	0.001*
				4 and 6	0.006*
Effective height	1	4.2	3.1	1 and 4	0.003*
	2	4.0	2.3	1 and 5	0.002*
	3	2.6	2.5	1 and 6	0.004*
	4	1.0	0.9	2 and 4	0.000*
	5	0.9	1.0	2 and 5	0.000*
	6	1.0	1.2	2 and 6	0.000*
Dead cover/abundance	1	1.7	0.3	1 and 6	0.012
	2	1.6	0.3	2 and 6	0.005
	3	1.8	1.0	5 and 6	0.000
	4	1.8	0.5		
	5	1.4	0.4		
	6	2.8	0.6		
Periphyton cover	1	4.6	0.3	1 and 5	0.003*
	2	4.6	0.3	1 and 6	0.034*
	3	4.8	0.3	2 and 5	0.012*
	4	4.7	0.3	2 and 6	0.011*
	5	4.9	0.1	3 and 6	0.000*
	6	4.2	0.4	4 and 6	0.001*
				5 and 6	0.000*
Species richness	1	8.9	1.8	1 and 3	0.011*
	2	8.7	0.6	1 and 6	0.002*
	3	7.3	2.8	3 and 5	0.011*
	4	8.0	1.0	4 and 6	0.021*
	5	8.0	0.7	5 and 6	0.003*
	6	6.9	1.0		
Biomass	1	1880.2	362.6	1 and 3	0.002
	2	1962.5	290.9	2 and 3	0.023
	3	2348.4	413.8	3 and 4	0.000
	4	1657.4	434.0	3 and 6	0.000
	5	2020.8	390.3	4 and 5	0.041
	6	1707.2	399.7		

Table 3-3 (continued).

Variable	hydrology rank	mean	std. dev.	hydrology rank pair	p value
Percent cover/abundance	1	1.4	0.5	1 and 3	0.000
Cladium	2	1.9	0.4	1 and 4	0.001
	3	2.2	0.5	1 and 5	0.000
	4	2.1	0.4	1 and 6	0.001
	5	2.3	0.5		
	6	2.1	0.7		
Percent cover/abundance	1	1.9	0.5	1 and 3	0.000*
Muhlenbergia	2	1.4	0.5	1 and 4	0.000*
	3	1.1	0.4	1 and 5	0.000*
	4	0.9	0.3	1 and 6	0.006*
	5	0	0	2 and 4	0.004*
	6	1.0	0.9	2 and 5	0.000*
				4 and 5	0.000*
				5 and 6	0.001*
Percent cover/abundance	1	1.4	0.4	1 and 3	0.019*
Rhynchospora	2	1.2	0.6	1 and 6	0.000*
	3	1.0	0.3	2 and 6	0.000*
	4	1.7	0.8	3 and 4	0.014*
	5	0.1	0.6	3 and 5	0.003*
	6	1.3	0.1	3 and 6	0.000*
				4 and 6	0.000*
				5 and 6	0.000*
Percent cover/abundance	1	1.3	0.5	1 and 4	0.000*
Schizachyrium	2	1.4	0.4	1 and 5	0.000*
	3	1.2	0.1	2 and 4	0.000*
	4	0.6	0.3	2 and 5	0.000*
	5	0.4	0.5	3 and 4	0.000*
	6	1.4	1.2	3 and 5	0.000*

(a)



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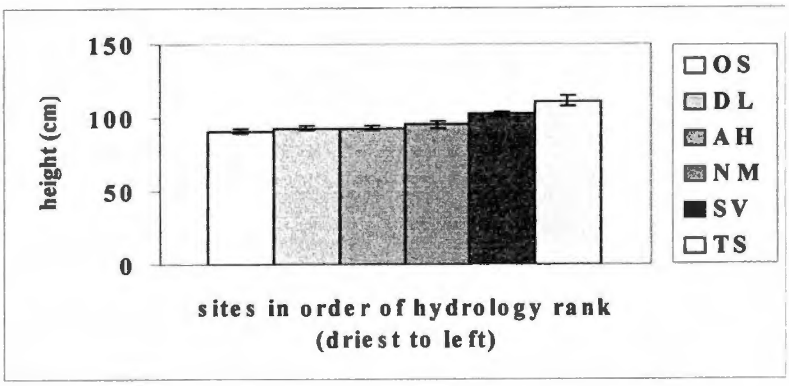

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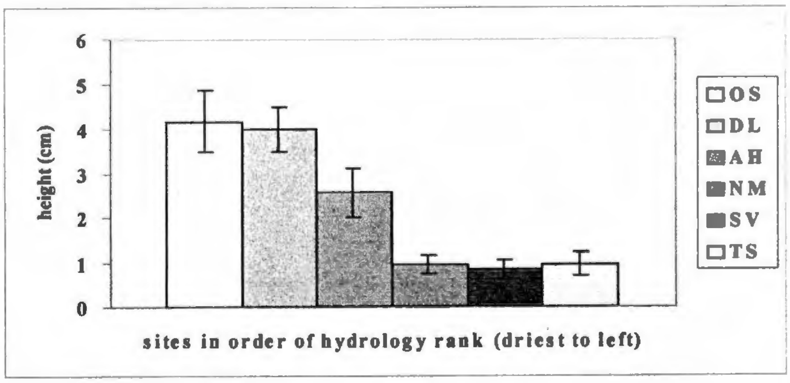
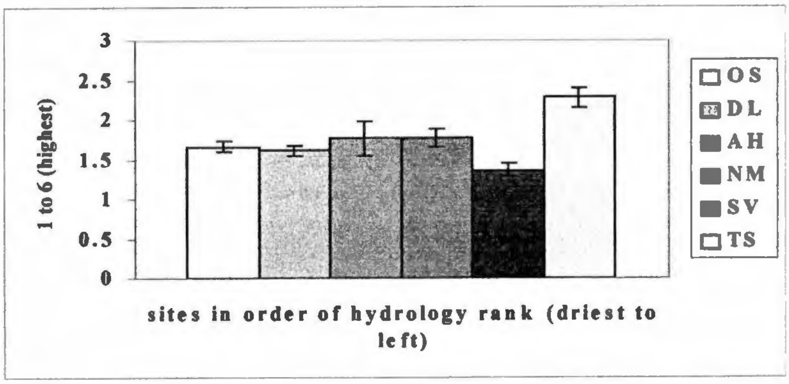
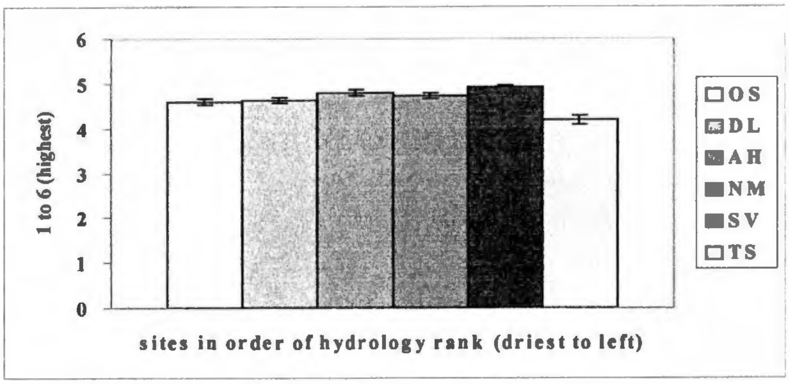


Figure 3-5. Ranked hydrology groups and mean a) cover/abundance, b) maximum vegetation height, and c) effective height (see Table 3-2 for significant differences between groups). Bars represent standard error.

(d)



(e)



(f)

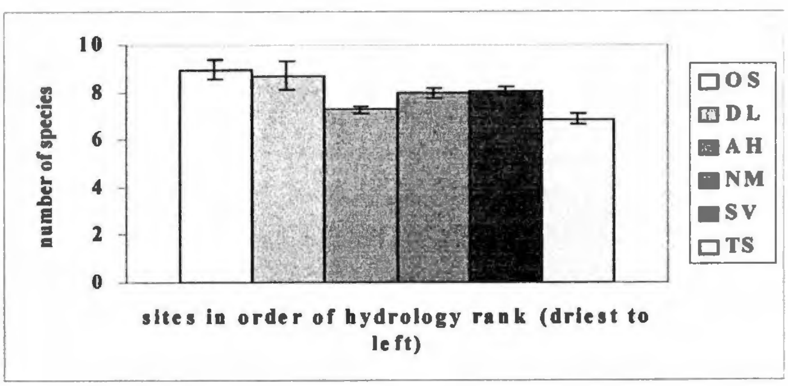
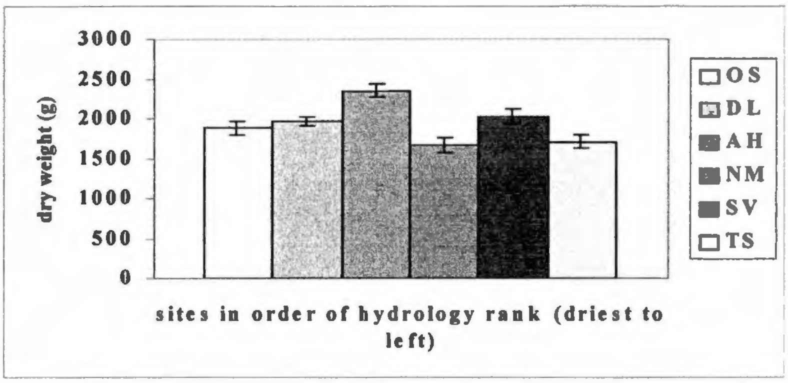
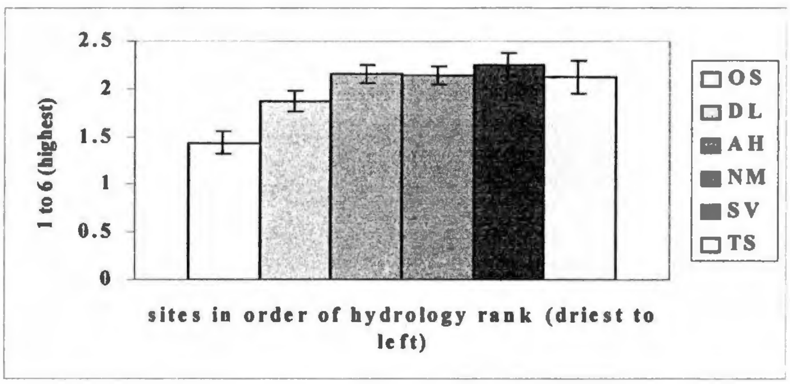


Figure 3-5 (continued). Ranked hydrology groups and mean d) dead cover/abundance, e) periphyton cover, and f) species richness (see Table 3-2 for significant differences between groups). Bars represent standard error.

(g)



(h)



(i)

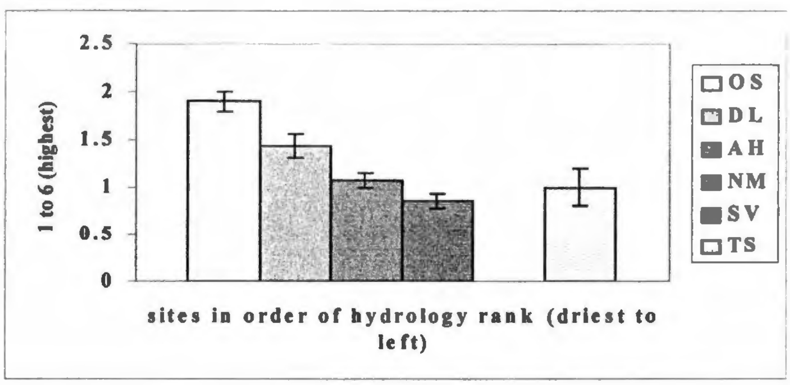
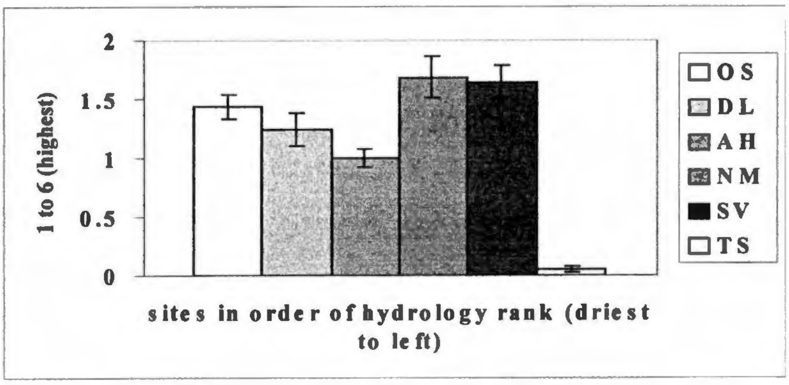


Figure 3-5 (continued). Ranked hydrology groups and mean g) biomass, h) percent cover/abundance *Cladium*, and i) percent cover/abundance *Muhlenbergia* (see Table 3-2 for significant differences between groups). Bars represent standard error.

(j)



(k)

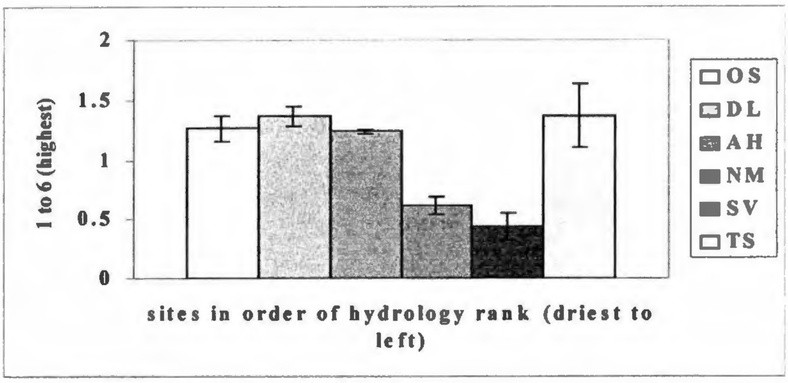


Figure 3-5 (continued). Ranked hydrology groups and mean j) percent cover/abundance *Rhynchospora* sp. and k) percent cover/abundance *Schizachyrium* (see Table 3-2 for significant differences between groups). Bars represent standard error.

Effective height, species richness and percent cover/abundance *Muhlenbergia* decreased with wetter conditions. Several variables had an ambiguous relationship with hydrology. Live plant cover/abundance was lowest on the intermediate hydrology sites. Percent cover/abundance *Rhynchospora* sp. was highest on North Mahogany and Shark Valley, and very low on Taylor Slough. In general, percent cover/abundance of *Schizachyrium* decreased with wetness, but was high on Taylor Slough. Dead cover/abundance and total biomass displayed no discernable trend.

Taylor Slough: Past and present.

I measured some of the same vegetation characteristics on five Taylor Slough plots as Werner (1975) did. He measured maximum vegetation height and effective height ("vertical obscurity profile"), dominance (number of quadrats a species dominated) of *Cladium*, *Muhlenbergia*, and *Schizachyrium*, and frequency (number of quadrats in which a species was found) for *Rhynchospora microcarpa*, (*R. tracyi* was not reported in 1975 or in 1997) and *Spartina bakeri*. Since 1975, mean maximum vegetation height has increased on Taylor Slough by more than 10 cm ($t=-3.844$, $p=0.001$), as has *Cladium* dominance ($t=-4.963$, $p<0.000$, Figure 3-6). Effective height ($t=3.360$, $p=0.028$), *Muhlenbergia* dominance ($t=7.079$, $p<0.000$), and *Rhynchospora* sp. frequency ($t=3.029$, $p=0.006$) decreased, and *Spartina bakeri* disappeared ($t=2.828$, $p=0.047$). *Schizachyrium* has remained relatively constant ($t=-1.494$, $p=0.149$).

DISCUSSION

Based on my results, several vegetation characteristics can help define sparrow habitat. Areas with higher percent cover/abundance of *Muhlenbergia* and *Rhynchospora* sp., high species richness, relatively higher biomass, and low percent cover/abundance of *Cladium*, were correlated with higher mean sparrow densities. Sparrows avoid areas dominated by large stands of tall sawgrass, which may include areas with vegetation of 100 cm (39 in) or taller. They appear to choose areas with high mean effective height (denser vegetation), and dominance of *Muhlenbergia* or *Rhynchospora* for breeding habitat and nest sites (Lockwood *et al.* in prep). This sparrow subspecies may need an overall mean effective height of at least 1-

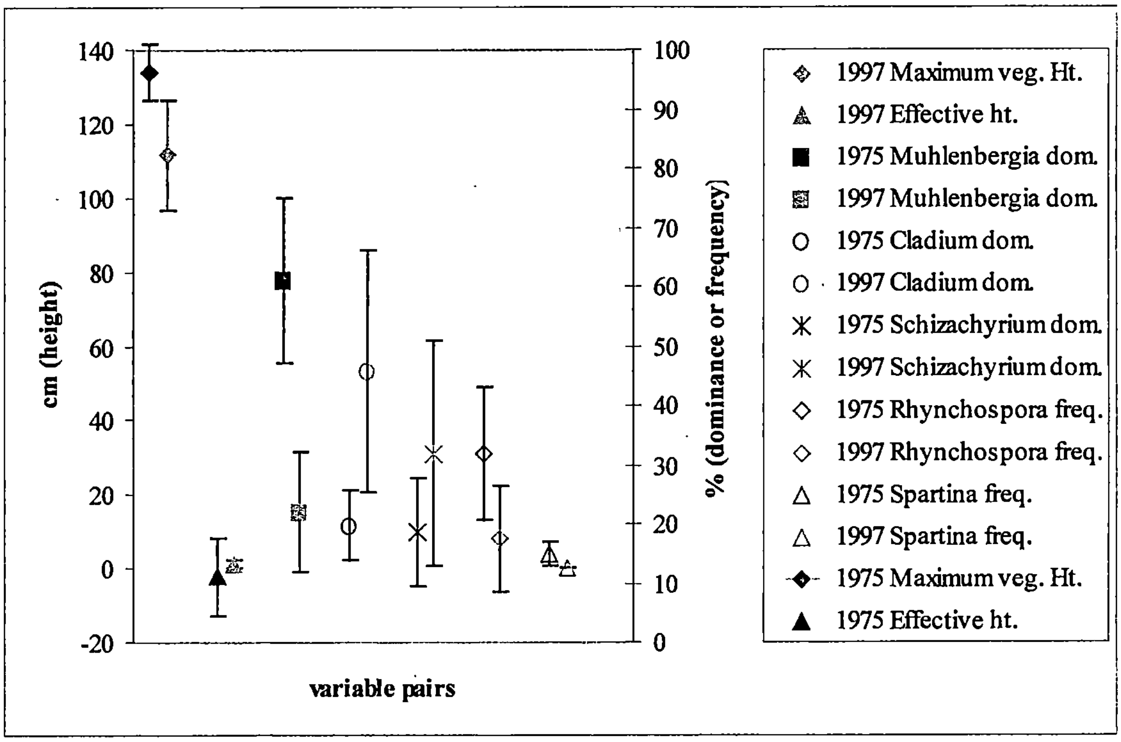


Figure 3-6. Change in Taylor Slough from Werner's 1975 survey to 1997. Points represent the mean, bars represent standard error.

2 cm (0.4-0.8 in) in their habitat, although Lockwood *et al.* (in prep) found mean effective height at nest sites of 3.3 cm (1.3 in). Predominant *Muhlenbergia* and *Rhynchospora*, along with less frequent forbs and grasses, may provide the bulk of this effective height characteristic. In the past, *Spartina bakeri* may have provided such low, dense clumps, but I did not find this grass on these sites. Higher biomass, along with higher effective height may offer more protection and resources. The combination of high periphyton cover, high biomass and taller effective height, may indicate a clumped, grassy habitat with some openness at the ground level. Since these sparrows spend a considerable amount of time travelling along the ground (Post and Greenlaw 1994), higher periphyton cover (which can restrict plant establishment, Van Meter-Kasanof 1973, Nott *et al.* 1998) may afford these sparrows passage through otherwise thick vegetation. Post *et al.* (1983) also found higher nest predation rates by rice rats (*Oryzomys palustris*) in denser vegetation.

Differences in extant habitat characteristics between areas of distinct hydrologic regimes echo those patterns seen over time in one area. In Shark Valley and Taylor Slough, where water levels have increased in the past 5 to 10 years, the percent cover/abundance of *Cladium* was higher, consistent with several other studies (Werner 1975, Olmsted *et al.* 1980, Alexander and Crook 1975). Olmsted and Armentano (1997) and Nott *et al.* (1998) have demonstrated that water levels in Shark Valley have been too high in wet years, too low in dry years, and have changed the plant communities in this area. Since Werner's survey, the regime controlled by pump station S-332 is primarily responsible for a general increase in water levels on this site, and the reduction in seasonal variation of these levels (Nott *et al.* 1998). In these wet areas, maximum vegetation height consistently exceeded 1 m (39 in). Percent cover/abundance of *Muhlenbergia* and *Rhynchospora* sp. was lower, decreasing vegetation thickness and the average height of clumps to less than 1 cm. On Taylor Slough, I did not observe an increase in *Spartina* or *Rhynchospora* sp. since Werner's (1975) survey to compensate for the loss in effective height. This study is not the first to indicate that recent hydrological changes have affected Everglades vegetation (Loveless 1959, McPherson 1973, Goodrick 1974, Volk *et al.* 1974, Alexander and Crook 1975, Davis *et al.* 1994), but it is one of few to quantify these changes. If high water persists in Shark Valley, *Cladium* dominance may increase, which would increase the average maximum vegetation height. If *Rhynchospora* prevalence also decreases as it has on Taylor Slough, and no other plant species that can provide effective height replaces it, this site may eventually become unsuitable for sparrows.

The disappearance of other suitable habitat types (sparse and clumped cordgrass marshes) has left only two habitat types now supporting Cape Sable seaside sparrows, *Muhlenbergia* prairies and the marginal sparse *Cladium* prairies. Higher water levels may be primarily responsible for the decrease in *Muhlenbergia* prairies, with no indication that other suitable habitat expressions are taking their place. As a consequence, the sparrow may be increasingly restricted to muhly prairies in areas, such as subpopulation B, where hydrologic patterns remain stable. These freshwater prairies exhibit highly dynamic patterns in structure and species composition in response to hydrological changes. Areas that are not now suitable for sparrows may become so in the future.

The sparrow may not have an ability to shift its range as fragmented suitable habitats become available through successional processes.As a result, the sparrow may require a much larger contiguous area for its "critical habitat" than just those areas they occupy now. Vegetation variables such as maximum vegetation height, effective height, and percent cover/abundance of *Cladium*, *Muhlenbergia* and *Rhynchospora* may be the most useful and efficient predictors of present and future suitable sparrow habitat. Future research should determine to what extent these variables and the continuity of suitable habitat are the most affected by dramatic hydrological changes.

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PART III APPENDIX: SCIENTIFIC NAMES OF PLANTS FOUND ACROSS STUDY SITES AND IDENTIFICATION SOURCE

Graminoids:

Grasses (Poaceae):

Andropogon glomeratus (Walt.) B.S.P.; Hitchcock 1971, Avery and Loope 1983.

Dichanthelium dichotomum (L.) Gould var. *dichotomum* (aka *Panicum albomarginatum, P. caerulescens, P. nitidum, P. roanokense*); Avery and Loope 1983.

Eragrostis elliottii S. Wats.; Hitchcock 1971, Godfrey and Wooten 1979, Avery and Loope 1983.

Muhlenbergia filipes M. A. Curtis; Hitchcock 1971, Avery and Loope 1983.

Panicum tenerum (Beyr.); Hitchcock 1971, Avery and Loope 1983.

Panicum virgatum L.; Hitchcock 1971, Avery and Loope 1983.

Schizachyrium rhizomatum (Swallen) Gould, (aka *Andropogon rhizomatus* Swallen); Hitchcock 1971, Godfrey and Wooten 1979, Avery and Loope 1983.

Setaria geniculata (Lam.) Beauv., (aka *Chaetochloa geniculata* (Lam.) Millsp. and Chase); Godfrey and Wooten 1979, Avery and Loope 1983.

Spartina bakeri Merr.; Hitchcock 1971, Avery and Loope 1983.

Sedges (Cyperaceae):

Cladium jamaicense Crantz (aka *Mariscus manaicensis* (Crantz) Britton); Godfrey and Wooten 1979, Avery and Loope 1983.

Cyperus haspan L.; Godfrey and Wooten 1979, Avery and Loope 1983.

Dichromena colorata (L.) Hitchc.; (possibly *Panicum coloratum*; Hitchcock 1971); Avery and Loope 1983

Eleocharis cellulosa Torr.; Godfrey and Wooten 1979, Avery and Loope 1983.

Rhynchospora microcarpa Baldw. ex Gray; Godfrey and Wooten 1979, Avery and Loope 1983.

Rhynchospora tracyi Britt.; Godfrey and Wooten 1979, Avery and Loope 1983.

Schoenus nigricans L.; Godfrey and Wooten 1979, Avery and Loope 1983.

Forbs:

Agalinis fasciculata (Elliott) Rafinesque (aka *Gerardia fasciculata*); Taylor 1992.

Asclepias lanceolata (Walt.); Avery and Loope 1983, Taylor 1992.

Asclepias longifolia (Michx.) (aka *Acerates floridana* and *A. delticola*); Avery and Loope 1983, Taylor 1992.

Aster dumosus L.; Cronquist 1980, Avery and Loope 1983.

Aster tenuifolius L.; Cronquist 1980, Avery and Loope 1983.

Bacopa caroliniana (Walt.) B. L. Robins. (aka *Hydrotrida caroliniana*); Avery and Loope 1983, Taylor 1992.

Centella asiatica L. Urb.; Avery and Loope 1983.

Crinum americanum L.; Godfrey and Wooten 1979, Avery and Loope 1983.

Cuscuta campestris Yuncker; Avery and Loope 1983.

Cyoctonum sessilifolium (Walt.) J. F. Gmel.; Avery and Loope 1983.

Erigeron quercifolium Lam.; Cronquist 1980, Avery and Loope 1983.

Erigeron strigosus (probably var. *beyrichii* (Fischer and Meyer) A. Gray.); Cronquist 1980.

Eupatorium leptophyllum DC.; Cronquist 1980, Avery and Loope 1983.

Eupatorium mikanioides Chapman; Cronquist 1980, Avery and Loope 1983.

Helenium pinnatifidum (Nutt.) Rydb.; Cronquist 1980.

Hymenocallis palmeri S. Wats., (aka *Hymenocallis tridentata*); Avery and Loope 1983, Taylor 1992.

Hyptis alata (Raf.) Shinners var. *stenophylla* Shinners; Avery and Loope 1983.

Hyptis verticillata Jacquin; Taylor 1992.

Ipomoea sagittata (Poir.); Avery and Loope 1983, Taylor 1992.

Linum floridanum (Planchon) Trelease, (aka *Cathartolinum floridanum* and *C. macrosepalum*); Taylor 1992.

Lobelia glandulosa Walt.; Avery and Loope 1983, Taylor 1992.

Ludwigia microcarpa (Michx.); Avery and Loope 1983.

Mikania scandens (L.) Willd.; Cronquist 1980, Avery and Loope 1983.

Oxypolis filiformis (Walt.) Britt.; Avery and Loope 1983, Taylor 1992.

Phyla nodiflora (L.) Greene, (aka *Lippia nodiflora*); Avery and Loope 1983, Taylor 1992.

Pityopsis graminifolia, (aka *Chrysopis graminifolia* (Michx.) Elliott); Cronquist 1980.

Pluchea rosea R. K. Godfrey; Cronquist 1980, Avery and Loope 1983.

Polygala grandiflora (Walt.) var *angustifolia* T. & G. (aka *Asemeia leiodes* and *A. cumulicola*); Avery and Loope 1983, Taylor 1992.

Prosperinaca palustris L. var. *palustris*, Avery and Loope 1983.

Sabatia stellaris Pursh; Avery and Loope 1983, Taylor 1992.

Sagittaria graminea Michx.; Godfrey and Wooten 1979.

Sagittaria lancifolia L.; Godfrey and Wooten 1979, Avery and Loope 1983.

Sisyrinchium atlanticum Bicknell; Taylor 1992.

Solidago stricta Ait.; Cronquist 1980, Avery and Loope 1983.

Teucrium canadense L. var *hypoleucum* Griseb., (aka *Teucrium nashii*); Avery and Loope 1983, Taylor 1992.

Utricularia cornuta Michx., (aka *Stomoisia cornuta*); Avery and Loope 1983, Taylor 1992.

Vernonia blodgettii Small; Avery and Loope 1983, Taylor 1992.

Woody perennials:

Conocarpus erectus L.; Avery and Loope 1983.

PART IV: USING REMOTE SENSING AND PLANT SURVEYS TO DIFFERENTIATE HABITAT SUBCLASSES WITHIN CAPE SABLE SEASIDE SPARROW PRAIRIES IN EVERGLADES NATIONAL PARK

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ABSTRACT

Combining satellite and other remote sensing images with plant survey data can increase the accuracy and detail of habitat maps. Such representations are especially helpful for the conservation and management of the endangered Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*). They can reveal changes in hydrologic conditions in its sole habitat: freshwater prairies in Everglades National Park. Within freshwater sparrow prairie areas, I delineated at least four distinct spectral classes (which I designated as "Green", "Yellow", "Orange", and "Purple") from Landsat TM images from 1996 and 1998. When I included vegetative information gathered in 1997, I found these four classes to be distinct in vegetation composition and structure as well. The classes most likely represent a cline in hydrologic conditions. The "Purple" class (dominated by *Cladium jamaicense*) represented wetter areas, the "Green" and "Yellow" classes intermediate hydrology, and the "Orange" class (dominated by *Muhlenbergia filipes*) drier areas. The classes I found were more detailed than the "alliances" of the Florida GAP project, and may represent "associations". The imagery coverage of these classes can be monitored to determine the effects of water management changes on hydrologic conditions on the sparrow prairies.

INTRODUCTION

Geographic Information Systems (GIS) and remote sensing imagery have been used increasingly as tools for population biologists and landscape ecologists (Quattrochi and Pelletier 1991). Remote sensing of plant communities, when combined with ground truth surveys of other taxa (such as birds), can indicate the amount and distribution of a specific habitat type available for a species or set of species (Palmeirim 1988, Sperduto and Congalton 1996). Elaborated habitat maps can also illustrate the degree and extent to which plant communities are fragmented, and help with species translocation programs (Franklin and

Steadman 1991). Remote sensing can also detect structural differences (such as canopy height in mangrove forests) between closely related community types (Ramsey and Jensen 1996). In some cases, habitat maps developed with GIS-based models can help determine policy choices that will not be detrimental to threatened species (Pereira and Itami 1991). GIS-based mapping updates can also allow wildlife managers to track habitat quantity and quality over time, and in most cases can be linked to discernable causal factors that can be influenced by policy changes (Breininger *et al.* 1991).

One of the first vegetation maps to use remote sensing in the Everglades was Davis' (1943) map. Davis used black-and-white aerial photographs from the U.S. Soil Conservation Service, reconnaissance flights and ground surveys, as well as field observations from Harshberger (Harshberger 1914, Davis *et al.* 1994). This map, at a scale of 1:400,000, grouped vegetation into 7 landscape categories and 13 vegetation classes, and extended from the Kissimmee River (north of Lake Okeechobee) to Florida Bay. More recently, the U.S. Fish and Wildlife Service's National Wetland Inventory map (1:58,000, 1985) and the State of Florida's Florida "Land Use and Cover" map (1:40,000, 1994-1995), employed more modern aerial photography to classify land covered by vegetation into 4 (with 14 subclasses) and 20 classes, respectively (Cowardin *et al.* 1979, FLUCCS 1985, Doren *et al.* 1999). A cooperative project between the University of Georgia's Center for Remote Sensing and Mapping Science, the National Park Service's South Florida Natural Resources Center, and the South Florida Water Management District is currently using SPOT (Le Système Pour l'Observation de la Terre, "the French Earth Observation System") panchromatic images with color-infrared aerial photographs and ground surveys. This system has a 10 m (33 ft) resolution and a hierarchical vegetation classification scheme, with 8 major classes that can be subdivided into subclasses of communities, and in some cases, sub-subclasses of individual species populations (Welch *et al.* 1995, Madden *et al.* 1999). However, Rutchey and Vilcheck (1994) have indicated that although SPOT images have better resolution, Landsat TM images may be better able to detect vegetation differences.

The Florida Gap Analysis Project (FGAP) uses Landsat TM (Thematic Mapper) images (with a 28.5 m (93.5 ft) resolution) and the classification system developed by The Nature Conservancy (Grossman *et al.* 1998). The purpose of the FGAP project is to develop one vegetation classification system for all of North America. The classification scheme uses physiognomic characteristics for the coarser levels (formation class, formation subclass, formation group, formation subgroup, and formation), and floristic characteristics for the finer levels (alliance and association). The Nature Conservancy is also in the process of giving each association a conservation (G) rank, to help identify "gaps" in habitat conservation programs. This program uses Landsat images from 1992 through 1994 plus several other mapping schemes (including the Florida Land Use and Cover Classification System (FLUCCS), National Wetlands Inventory maps, and Soil Conservation Service maps) to "crosscheck" their classifications. At this time, FGAP has produced two versions of this map (2.1 and 3.0) for Florida, using images from 1992 and 1994. Version 3.0 and later versions are reclassifications of the original 2.1 version, using updated knowledge from groundtruthing (Grossman *et al.* 1998).

The federally endangered Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*) is now limited to the short-hydroperiod freshwater prairie habitats within Everglades National Park (ENP) (Kushlan and Bass 1983). Kushlan and Bass (1983) conducted an extensive helicopter survey of all possible Cape Sable seaside sparrow habitat, and narrowed suitable habitat to three prairie types. Sixty-seven percent of their sparrow-occupied sites were "muhly prairie", with prevalent *Muhlenbergia filipes* (muhly grass), *Schoenus nigricans* (blacktop sedge), and *Cladium jamaicensis* (sawgrass, a sedge). Another twenty-nine percent of sparrow occupied habitat was "mixed prairie", characterized by *Rhynchospora* sp. (beak-rush) and *Panicum* sp. (spike-rush), but without *Muhlenbergia.* A small number of occupied sites were in "cordgrass marsh", predominated by *Spartina bakeri.* Werner and Woolfenden (1983) conducted plant surveys at some of the Kushlan and Bass (1983) helicopter survey defined sites. They found similar, suitable habitat types and found most sparrows in the *Muhlenbergia* prairies. These prairie types displayed differences in average biomass, stem density, stand height, and predominant graminoid species.

Botanists have classified the wet prairies of the Everglades in several different ways. Most classification systems delineate two general, short-hydroperiod prairie/marsh types, wet prairies and sawgrass marshes, which are further divided into subtypes by dominant plant species (typically graminoids). Porter (1967) conducted a plant survey in Big Cypress of "seasonally wet marl prairies", and found a hybrid between marl wet prairies and sawgrass marshes based on the dominant grasses and sedges (*Muhlenbergia capillaris* var. *filipes*, *Mariscus* [*Cladium*] *jamaicensis*, and *Andropogon rhizomatus*).

Gunderson and Loftus (1993), attempting to synthesize all the previous classification schemes, divided wet prairies into "peat" and "marl" prairies. "Peat" wet prairies are in lower elevation areas and have longer hydroperiods than drier, "marl" wet prairies. In peat prairies, dominant sedge and grass species include *Eleocharis cellulosa* (spike-rush), *Rhynchospora tracyi*, and *Panicum hemitomon* (Gunderson 1994). Dominant marl prairie grasses and sedges include *Muhlenbergia filipes*, *Schoenus nigricans* (black sedge), *Aristida purpurascens* (wiregrass), *Schizachyrium rhizomatum* (bluestem), and *Eragrostis elliottii* (lovegrass).

Sawgrass marsh, dominated by *Cladium jamaicense*, is one of the most common communities in the Everglades (Gunderson 1994). *Cladium* almost completely dominates the "tall" sawgrass marsh community, which is most prevalent in wet areas such as sloughs. The intermediate (or "sparse") sawgrass marsh community is shorter in stature, and intersperses with wet prairies in low areas that remain wetter throughout the year. Olmsted and Armentano's (1997) "sparse sawgrass" community is characterized by *Cladium jamaicense*, *Eleocharis cellulosa*, and the grasses *Panicum hemitomon* and *P. tenerum*. Both wet prairie types tend to support higher plant species diversity, and have a lower stature, than sawgrass marshes (Olmsted *et al.* 1980, Gunderson 1994). In the southeastern Everglades, both prairies and marshes have been invaded by non-native trees, including *Melaleuca quinquenervia* (melaleuca tree) and *Casuarina* spp. (Australian pine) (Bodle *et al.* 1994).

The extant Everglades ecosystem has been halved by regional hydrology projects that have altered historic flows and water levels. The altered hydrology has created some favorable conditions for some plant communities, but devastated others. Alexander and Crook (1975, 1984) suggested that the vegetation changes that occurred from the 1940's to the 1970's were probably caused by one or more of four factors: water management, increased fire frequency and size, rising sea levels, and displacement by invasive species. Plant community changes have occurred in Cape Sable seaside sparrow habitat as well. While originally found in *Spartina bakeri* prairies by Howell in 1919, surveys in the 1980's found these sparrows in prairies predominated by *Muhlenbergia filipes* (Howell 1919, Kushlan and Bass 1983, Curnutt *et al.* 1998). Atwater (1954) was one of the first to suggest that *Muhlenbergia* may be expanding because of the resulting drier conditions due to drainage. However, Nott *et al.* (1998) found that, in inland areas where

Cape Sable seaside sparrows were once prevalent, longer hydroperiods have corresponded to a decrease in *Muhlenbergia* and muhly prairies.

In this study, I used modeled hydrology data, 1996 and 1998 satellite images, and ground plant survey data across four areas occupied by Cape Sable seaside sparrows, as well as in one area where sparrows have not been found for about 10 years. First, I determined discernable spectral classes detectable from satellite images. I compare the spectral classification results between the 1996 and 1998 satellite images. A difference between images would indicate that either the atmospheric conditions were different between the two years, or that the vegetation had changed significantly. I then determined whether these spectral classes represent different vegetation, using data from plant surveys. I also examined the classes for hydrologic differences, which may explain vegetation patterns. Finally, I examined the degree to which my classification scheme coincides with the FGAP project. This analysis can help determine the capacity of satellite images to detect vegetative detail within known sparrow habitats.

METHODS

Image processing

I used ERDAS Imagine software (Version 8.3.1 for Windows NT 4.0) on a Dell Dimension 333 with a Pentium II processor, to perform all image processing and classification procedures. I used Landsat (5) TM images from 21 March 1996 (LT501504209608110) and from 14 May 1998 (LT5015042043098134). There were no images from 1997 that were suitably free from cloud cover, although images from this year would have been optimal (I conducted my plant surveys in 1997). The images used were rectified using a Trimble Pathfinder Pro Global Positioning System (GPS) receiver with real-time differential correction. The receiver was set to the North American Datum of 1983 (NAD 83), and used the Miami Coast Guard station beacon for corrections. I recorded all locations in UTM (Universal Transverse Mercator) coordinates. To determine the accuracy of the GPS unit, I gathered five ground control points (GCP's) along the Main Park Road at permanent structures or intersections with other roads, close to my study sites (Figure 4-1). I also used a set of 1:10,000 scale color-infrared aerial photos and overlaid my GCP's as another reference check (Kardoulas *et al.* 1996). I compared the accuracy of my

GCP's (and plant transect locations) with the aerial photos. For another estimate, I took repeated GPS readings each day at a fixed reference point over the course of the field season. With these two methods, I estimated the GPS unit's likely margin of error at ±5-15 m (±16-49 ft).

I delineated an Area of Interest (AOI) that encompassed my five study areas, which gave me a larger number of prairie pixels with which to run my analysis. This AOI was approximately 227 km^2 (88 mi^2). Within this AOI, I conducted plant transects on four study areas that supported intensively studied colonies of Cape Sable seaside sparrows during the previous six years (Figure 4-1). The site unoccupied by sparrows at the time of this study (TS, 75.13 ha) supported sparrows from before 1974 until around 1990 (Figure 4-1) (Curnutt *et al.* 1998). From 1992 to 1997, three sites had fairly high sparrow occupancy (AH,

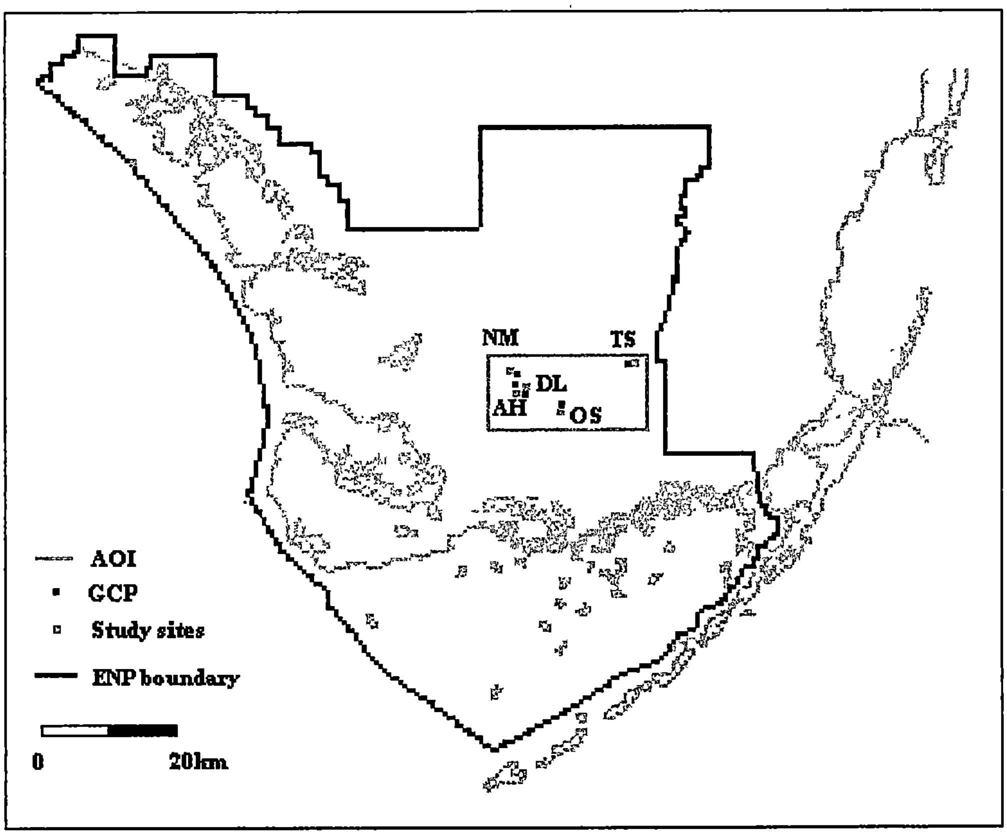

Figure 4-1. Map of Everglades National Park, Ground Control Points (GCP), my Area of Interest (AOI), and the 5 study areas it encompasses.

0.20 km^2, DL, 0.20 km^2, and OS, 0.3 km^2), while the NM site (0.29 km^2) supported low sparrow numbers (Lockwood *et al.* 1997). On each of these sites I collected structural and compositional vegetation characteristics in 200 1 m^2 quadrats (see Part 3 of this dissertation). I used modeled average annual hydroperiod (number of days inundated) and average annual ponding depth (height of water above the soil surface) from the Across Trophic Level System Simulations (ATLSS) project, to describe hydrologic conditions. These modeled averages include hydrologic data from 1985 to 1994.

Image classification

The Landsat 5 TM satellite detects seven wavelength bands: band 1 registers 0.45-0.52 µm (blue-green); band 2 registers 0.52-0.60 µm (green); band 3 registers 0.63-0.69 µm (red); band 4 registers 0.76-0.90 µm (near infrared); band 5 registers 1.55-1.75 µm (mid infrared); band 6 registers 10.4-12.5 µm (far infrared); and band 7 registers 2.08-2.35 µm (mid infrared) (Campbell 1996). Some of these bands are more advantageous than others, depending upon their use (e.g. band 1 can penetrate water bodies, Campbell 1996). I used spectral bands 1 through 7 in all analyses, as this did not alter the outcome of the separability between clusters, nor did it significantly complicate other procedures. Additionally, no radiometric preprocessing (e.g., stretching, destriping) of the images was necessary prior to classification.

For the 1996 and 1998 images, I performed an unsupervised classification, where patterns in the images are detected by the computer using statistical methods, and used the ISODATA algorithm to assign pixels to appropriate classes (Richards 1993, Campbell 1996, ERDAS 1997). This procedure finds classes (or groups) of pixels with similar wavelength values in the seven bands. In unsupervised methods, the user initially sets the number of classes the software is to delineate, and then uses statistical methods to refine the number of classes based on the degree to which the pixels in each class resemble each other (and not pixels in other classes).

I did not have any *a priori* knowledge of the number of distinct classes in these prairies, so I started with 50 classes. I used a separability measure to determine which classes were spectrally distinct from the others (Richards 1993, ERDAS 1997). Most of the 50 classes greatly overlapped one another (i.e. a pixel could easily be classified in several classes), so I used a "transformed divergence" (TD) measure to determine which classes should be merged (Swain and Davis 1978). A TD value of zero indicated that the spectral signature of two classes overlapped completely, while a TD value of 2000 indicated those classes had no overlap. I began by merging classes with a TD value below 1500, then reanalyzed the resulting signature file again. I then merged classes with a TD value below 1600, reanalyzed the signature file, and continued this process until I reached a signature file for which one more round of merging would result in a signature that left little differentiation within my five study areas. I performed this method for both the 1996 and 1998 images. I found that, for both years, this signature file contained classes that were separable from each other by at least 90% (TD=1800), with one exception (TD=1555, 78% for two classes in the 1996 image, TD=1696, 85% for two in the 1998 image). Once I found distinct classes, I designated each with a color.

Once I determined these classes, I used a paired t-test to determine an overall difference in number of pixels classified into each class between years. To detect differences in vegetation characteristics of each class between years, I used an independent sample t-test with a Bonferroni correction (Rencher 1995). I performed a General Linear Multivariate Model to detect vegetation differences between classes. For an overall test between classes, I utilized Wilk's lambda, and used post-hoc Bonferroni (for variables with equal variances) and Tamhane's T^2 (for variables with unequal variances) statistics to determine which vegetation variables differed between classes. I also computed cover/abundance-frequency indices for the most prominent plant species on the sites, to identify dominant species and indicate possible differences in plant communities between classes (Amundsen 1977). Alpha levels below 0.05 were considered significant, and all statistics were performed on SPSS Version 8.0.0.

The FGAP classification differed from my methods in several respects. Using maps from water management agencies, the FGAP project overlaid the satellite images and produced two classifications, one containing all pixels in "natural" areas and one with all "urban" areas (Florida Fish and Wildlife Cooperative Unit 1999). For each map, an unsupervised classification was then used to successively group the pixels in each map into classes, with each classification iteration "busting" classes into several, more specific classes. Ground truth data and other remote sensing images were used to identify the resultant classes. With each successive version of the FGAP map, the land cover classes (including plant alliances) were combined or split to create classes that are more accurate representations of land cover. In my analysis, I began with no prior classification, and instead of "busting" groups of pixels into smaller classes, I started with many small groups of pixels and combined them until the statistical similarity between classes was about 5%. My method assumes less prior knowledge than the FGAP method, and both methods rely on statistical boundaries, rather than subjective decisions, to group or split classes.

RESULTS

Number of prairie vegetation classes in the 1996 and 1998 satellite images

Unsupervised classification and subsequent merging led to seven classes in the 1996 and 1998 images. The "Green", "Yellow", and "Orange" classes were the most prevalent in the five study areas, and for the large AOI in general (Figures 4-2 through 4-6 a and b; Table 4-1). OS had the greatest proportion of the "Orange" class of the sites, while AH, DL, and TS were predominantly classified as "Green". The proportion of "Green" and "Yellow" classes switched in NM between 1996 and 1998. "Purple" was most abundant on TS, especially in 1998. Three other classes ("Magenta", "Cyan", and "Red") were not widespread in the five study areas, and were not represented by ground plant data.

Differences between 1996 and 1998

The number of pixels in each class in the study areas did not differ between image years (Table 4-1). There were differences in pixel distribution among the classes in the AOI between 1996 and 1998, mainly in pixels classified as "Magenta" or "Cyan" in 1996 that were classified as "Orange" in 1998. With the exception of the effective height characteristic, there were no significant differences in vegetation characteristics between years for the four most common ("Green", "Yellow", "Orange", and "Purple") spectral classes, and not enough plant transects in the other three classes to run statistical tests (Table 4-2). Although the distribution of pixels in classes did not differ much between years, 1112 (30%) of all pixels changed from the "Green" class to the "Yellow" class (or vice versa) between images (Figures 4-2 through

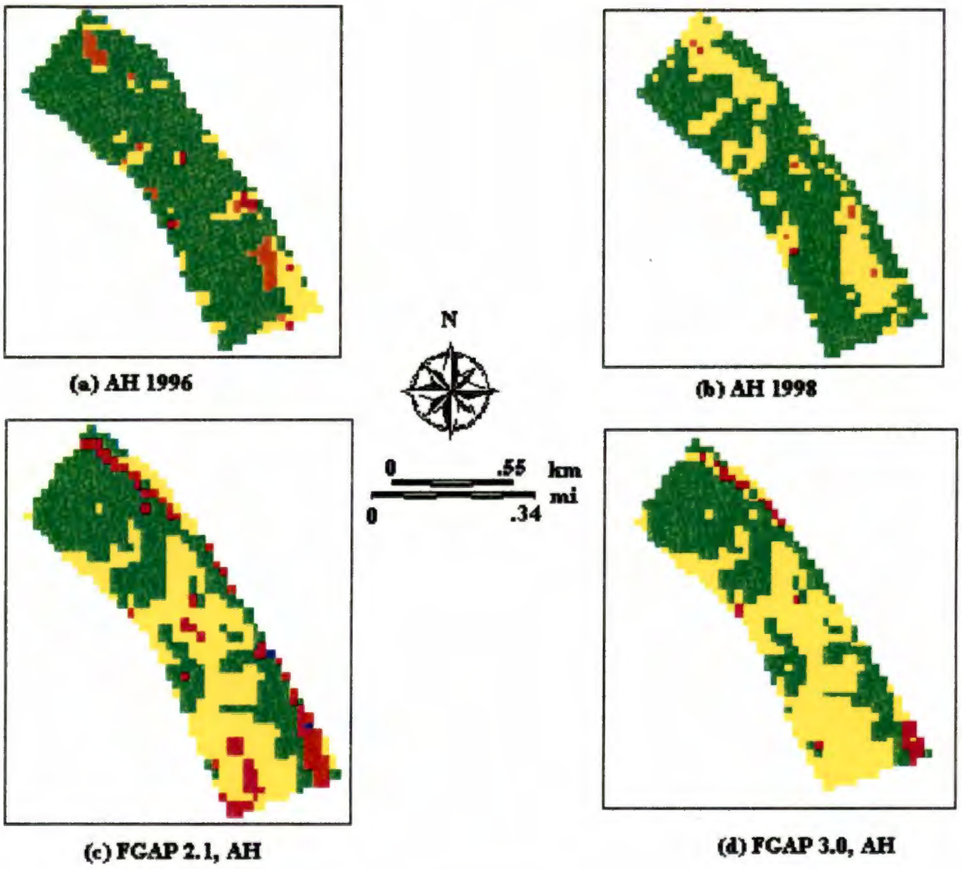

Figure 4-2. Classified images of the Alligator Hammock (AH) site from a (a) 1996 and (b) 1998 image, and FGAP version (c) 2.1 and (d) the updated 3.0 classifications of the site. In (c) and (d), the following colors represent FGAP plan alliances: green = *Muhlenbergia filipes* tropical herbaceous alliance, yellow = *Cladium mariscus* spp. *jamaicense* (sparse) freshwater herbaceous alliance, red = *Eleocharis cellulosa* (sparse) herbaceous freshwater alliance, and purple = *Cladium mariscus* spp. *jamaicense* freshwater herbaceous alliance.

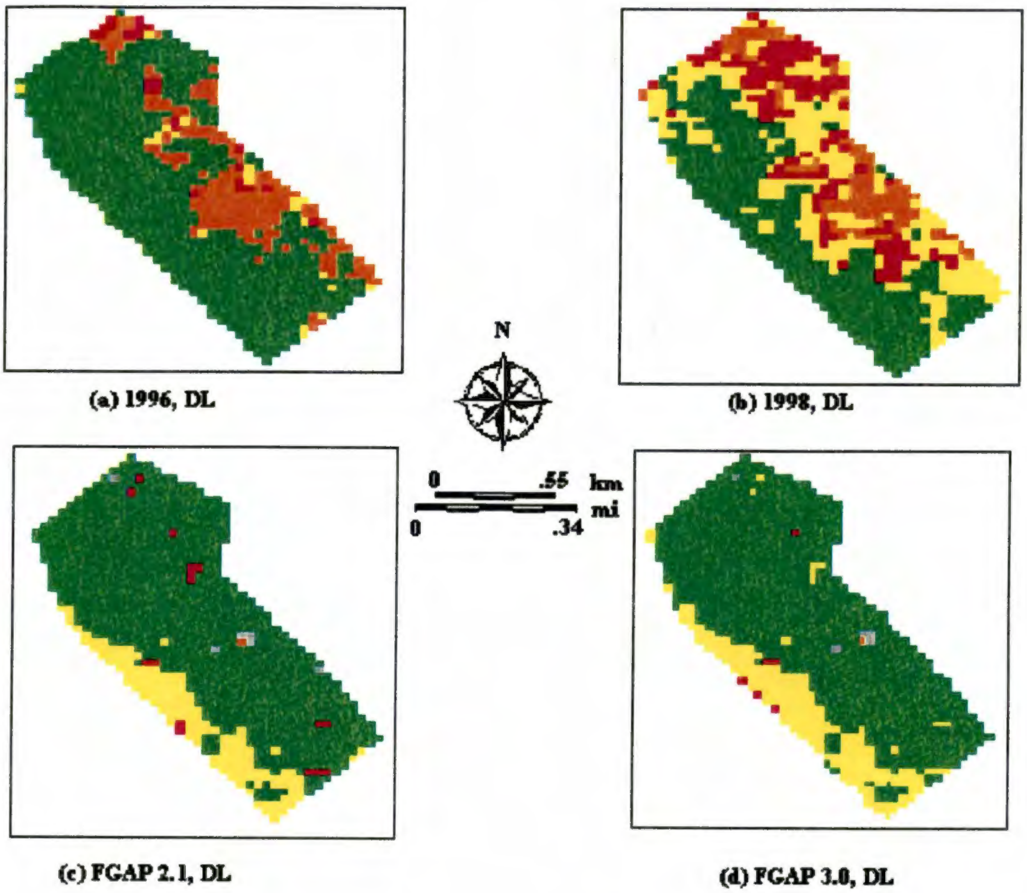

Figure 4-3. Classified images of the Dogleg (DL) site from a (a) 1996 and (b) 1998 image, and FGAP version (c) 2.1 and (d) the updated 3.0 classifications of the site. In (c) and (d), the following colors represent FGAP plan alliances: green = *Muhlenbergia filipes* tropical herbaceous alliance, yellow = *Cladium mariscus* spp. *jamaicense* (sparse) freshwater herbaceous alliance, red = *Eleocharis cellulosa* (sparse) herbaceous freshwater alliance, and purple = *Cladium mariscus* spp. *jamaicense* freshwater herbaceous alliance.

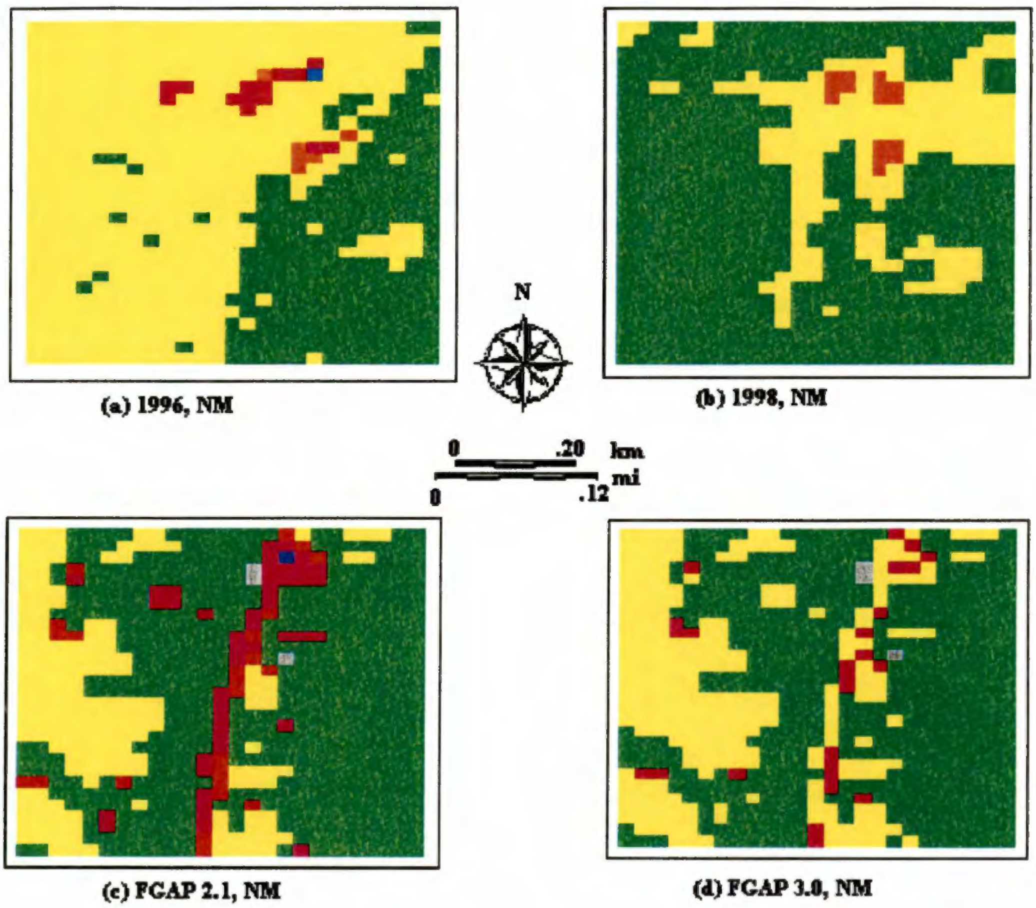

Figure 4-4. Classified images of the North Mahogany (NM) site from a (a) 1996 and (b) 1998 image, and FGAP version (c) 2.1 and (d) the updated 3.0 classifications of the site. In (c) and (d), the following colors represent FGAP plan alliances: green = *Muhlenbergia filipes* tropical herbaceous alliance, yellow = *Cladium mariscus* spp. *jamaicense* (sparse) freshwater herbaceous alliance, red = *Eleocharis cellulosa* (sparse) herbaceous freshwater alliance, and purple = *Cladium mariscus* spp. *jamaicense* freshwater herbaceous alliance.

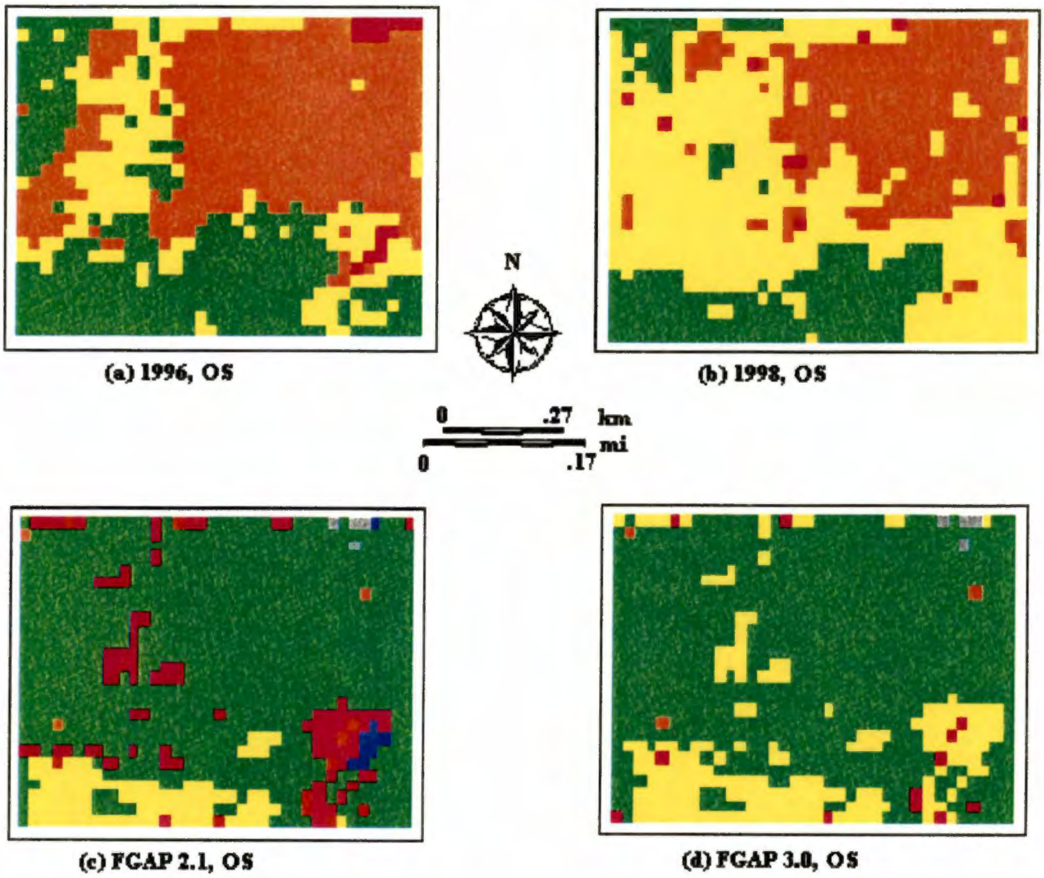

Figure 4-5. Classified images of the Old Ingraham Highway South (OS) site from a (a) 1996 and (b) 1998 image, and FGAP version (c) 2.1 and (d) the updated 3.0 classifications of the site. In (c) and (d), the following colors represent FGAP plan alliances: green = *Muhlenbergia filipes* tropical herbaceous alliance, yellow = *Cladium mariscus* spp. *jamaicense* (sparse) freshwater herbaceous alliance, red = *Eleocharis cellulosa* (sparse) herbaceous freshwater alliance, purple = *Cladium mariscus* spp. *jamaicense* freshwater herbaceous alliance, and blue = *Typha domingensis* tropical herbaceous alliance.

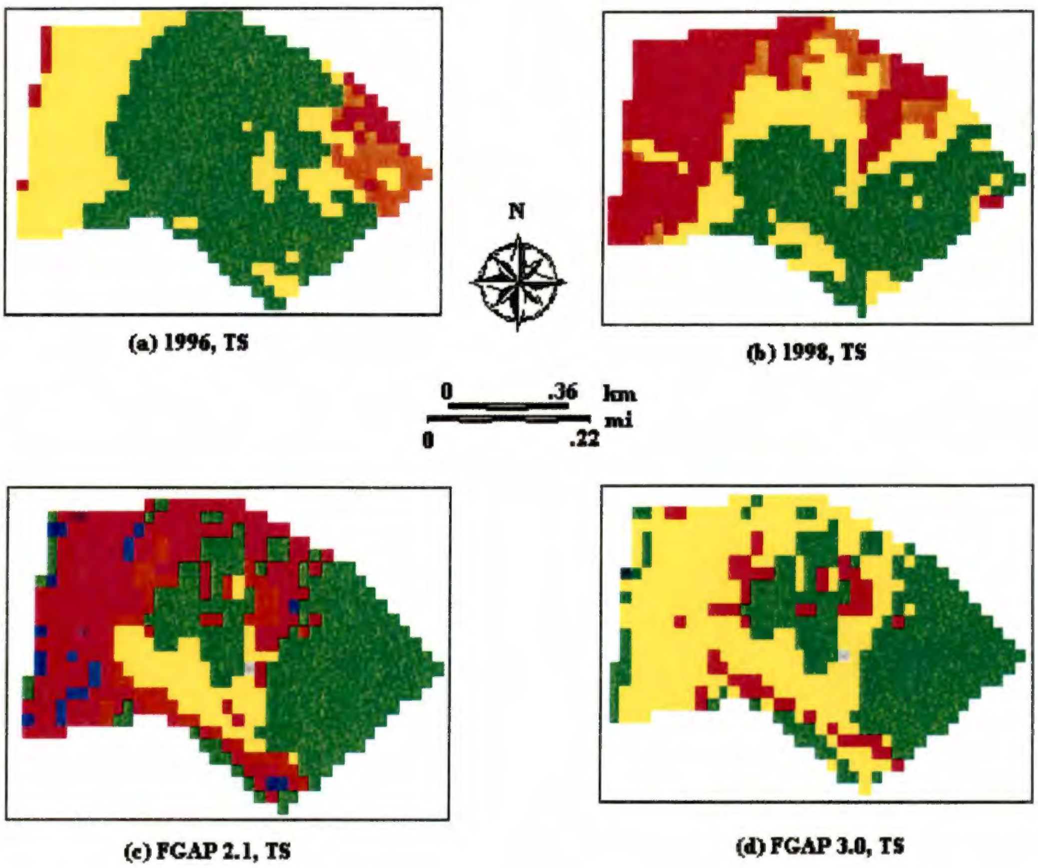

Figure 4-6. Classified images of the Taylor Slough (TS) site from a (a) 1996 and (b) 1998 image, and FGAP version (c) 2.1 and (d) the updated 3.0 classifications of the site. In (c) and (d), the following colors represent FGAP plan alliances: green = *Muhlenbergia filipes* tropical herbaceous alliance, yellow = *Cladium mariscus* spp. *jamaicense* (sparse) freshwater herbaceous alliance, red = *Eleocharis cellulosa* (sparse) herbaceous freshwater alliance, purple = *Cladium mariscus* spp. *jamaicense* freshwater herbaceous alliance, and blue = *Typha domingensis* tropical herbaceous alliance.

Table 4-1. Number of pixels in each class for Area of Interest and 5 study areas, and results from a paired t-test between years for each area.

Class	AOI		AH		DL		NM		OS		TS	
	1996	1998	1996	1998	1996	1998	1996	1998	1996	1998	1996	1998
Green	117,833	102,094	570	477	708	414	224	531	308	175	348	204
Yellow	55,072	65,404	82	218	22	283	480	179	172	280	190	149
Orange	19,714	51,524	38	6	197	120	6	15	389	419	35	44
Purple	37,326	18,869	8	0	21	0	5	0	9	0	2	191
Magenta	10,096	0	0	2	3	152	9	0	6	0	22	3
Cyan	29,245	25,793	0	0	0	0	1	0	0	0	0	0
Red	7092	0	0	0	0	0	0	0	0	0	0	0
other	2619	15,313										
t, df	-0.666, 6		-0.334, 6		-0.039, 6		-1.243, 6		0.053, 6		0.023, 6	
p (2-tailed)	0.530		0.749		0.970		0.260		0.959		0.982	

Table 4-2. Class means for vegetation and hydrologic variables, 1996 and 1998. Bolded numbers indicate a significant difference between years for effective height in the Yellow class (t=3.232, df=30.887, p=0.002). Hydrologic variables were not included in the analysis, and they are included here for descriptive purposes.

Variable	PURPLE		GREEN		YELLOW		ORANGE	
	1996	1998	1996	1998	1996	1998	1996	1998
Cover (1 to 6, highest)	1.96	2.15	1.94	1.84	1.79	1.90	1.97	2.00
Max. veg. ht (cm)	100.00	124.77	95.74	95.21	102.61	96.39	92.63	91.15
Effective ht. (cm)	0.70	0.58	2.79	2.03	**1.11**	**2.74**	4.01	4.07
Dead cover (1 to 6, highest)	1.78	2.50	1.72	1.68	1.96	1.87	1.87	1.75
Periphyton cover (1 to 6, highest)	4.66	4.29	4.56	4.63	4.60	4.58	4.59	4.63
Spp Richness (# species)	8.40	6.98	7.65	7.48	7.34	8.01	9.21	9.37
Biomass (g dry weight)	1846.1	1791.4	1996.9	1930.6	1791.8	1912.6	1914.6	1927.2
Cladium (1 to 6, highest)	2.53	2.27	2.01	2.04	2.08	1.94	1.61	1.55
Muhlenbergia (1 to 6, highest)	0.84	0.45	1.38	1.24	0.83	1.24	1.58	1.68
Rhynchospora (1 to 6, highest)	0.79	0.12	0.90	1.04	1.11	1.14	1.45	1.50
Schizachyrium (1 to 6, highest)	1.61	0.07	1.26	1.26	0.83	1.21	1.39	1.29
ave. annual hydroperiod (days)	187	213	211	209	211	214	198	197
ave. annual ponding depth (cm)	3.70	4.73	3.41	3.85	4.07	3.24	2.02	1.70

4-6). These two classes had the least spectral distance between them (78% in 1996, 85% in 1998), and were not as vegetatively distinct from each other as they were from the "Purple" and "Orange" classes.

Vegetative and hydrologic differences between classes

A multivariate general linear model test showed that, for both years, the four classes were vegetatively distinctive. For 1996, seven vegetation characteristics differentiated the classes: maximum vegetation height (F=3.335, df=3, p=0.023); effective height (F=7.143, df=3, p=0.000); species richness (F=6.076, df=3, p=0.001); percent cover of *Cladium* (F=4.091, df=3, p=0.009); percent cover of *Muhlenbergia* (F=7.726, df=3, p=0.000); percent cover of *Rhynchospora* (here, *Rhynchospora* includes both *R. microcarpa* and *R. tracyi*, although the former was much more common; F=3.090, df=3, p=0.031); and percent cover of *Schizachyrium* (F=4.247, df=3, p=0.007). Eight characteristics differentiated classes in the 1998 image: maximum vegetation height (F=16.714, df=3, p=0.000); effective height (F=3.423, df=3, p=0.020); dead cover (F=3.481, df=3, p=0.019); species richness (F=4.456, df=3, p=0.006); percent cover of *Cladium* (F=2.853, df=3, p=0.041); percent cover of *Muhlenbergia* (F=5.327, df=3, p=0.002); percent cover of *Rhynchospora* (F=5.162, df=3, p=0.002); and percent cover of *Schizachyrium* (F=6.721, df=3, p=0.000). However, not all of the above vegetation characteristics listed above were different between all classes (Tables 4-3 and 4-4).

The modeled hydrologic data was not included in the multivariate analysis. Their use in this study was primarily descriptive, and these data do not include hydrologic changes at the sites from 1994 (the last year of data the hydrologic model incorporates) to 1996 and 1998, when the images were taken. The "Purple" class had the deepest average annual ponding depth but a somewhat shorted average annual hydroperiod than the "Yellow" or "Green" classes (Table 4-2). The "Yellow" class had the longest hydroperiod of the four classes. "Orange" was the driest in both hydroperiod and ponding depth.

Comparison between my classes and FGAP classes

The majority of the plant transects in the "Green" class (63 to 76%) or "Orange" class (93 to100%) in my images were classified as "*Muhlenbergia filipes* Tropical herbaceous alliance" in the FGAP 2.1 and 3.0 versions (Table 4-5, Figures 4-2 through 4-6, c and d). About 50% of the transects classified as

Table 4-3. 1996 vegetation differences between classes (numbers represent standard error and p value. Normal font are Bonferroni values, italicized are Tamhane's T^2 values).

Variable	Green and Yellow	Green and Orange	Yellow and Orange	Orange and Purple
Max. veg. ht.			*0.344, 0.049*	*8.759, 0.013*
Effective height	*0.558, 0.003*		*0.655, 0.000*	*1.717, 0.044*
Spp. richness		*0.429, 0.032*	*0.473, 0.007*	
Cladium		0.146, 0.041	0.161, 0.025	
Muhlenbergia	0.145, 0.002		0.170, 0.000	
Rhynchospora		*0.187, 0.001*		
Schizachyrium	*0.153, 0.108*		*0.180, 0.018*	

Table 4-4. 1998 vegetation differences between classes (numbers represent standard error and p value. Normal font are Bonferroni values, italicized are Tamhane's T^2 values).

Variable	Green and Yellow	Green and Orange	Green and Purple	Yellow and Purple	Orange and Purple
Max. veg. ht.			4.462, 0.000	4.433, 0.000	5.079, 0.000
Effective ht.		*0.815, 0.027*	*1.081, 0.009*	*1.074, 0.000*	*1.231, 0.000*
Dead cover			0.263, 0.014		
Spp. richness		0.558, 0.006			0.842, 0.034
Muhlenbergia	0.272, 0.025			0.270, 0.025	0.309, 0.001
Rhynchospora			0.313, 0.025	0.311, 0.009	0.356, 0.001
Schizachyrium	0.273, 0.000			0.271, 0.000	0.311, 0.001

Table 4-5. Distribution of transects among my classes from 1996 and 1998 images and their classifications in FGAP maps (versions 2.1 and 3.0). One transect in the 1998 image was classified as "Magenta", so the number of transects in the 1998 section will not add to 100.

	Muhlenbergia alliance		*Cladium* alliance		*Cladium* (sparse) alliance		*Eleocharis* alliance	
1996	FGAP 2.1	FGAP 3.0	FGAP 2.1	FGAP 3.0	FGAP 2.1	FGAP 3.0	FGAP 2.1	FGAP 3.0
Purple	0	2	0	0	2	0	0	0
Green	39	41	3	0	12	12	0	1
Yellow	12	11	6	0	4	13	2	0
Orange	20	19	0	0	0	1	0	0
1998								
Purple	3	3	7	0	2	7	0	0
Green	25	28	0	0	15	14	1	1
Yellow	29	27	1	0	1	5	1	0
Orange	13	14	1	0	0	0	0	0

"Yellow" in my scheme were classified as "*Muhlenbergia filipes* Tropical herbaceous alliance" in 1996, but about 90% of the "Yellow" transects in the 1998 fell in this FGAP class. Some transects (22% to 35%) in the "Green" class were also classified as "*Cladium mariscus* var. *jamaicense* Freshwater herbaceous alliance" and "*Cladium mariscus* var. *jamaicense* (sparse) Freshwater herbaceous alliance". ("*Cladium mariscus* var. *jamaicense*" and *Cladium jamaicense* are the same species). A few transects were in pixels classified under the FGAP scheme as "*Eleocharis cellulosa* (sparse) herbaceous alliance (with *Typha* *domingensis*)". I attempted to match my classes (based on vegetative characteristics) with those of the FGAP classification, and recolored the FGAP maps to reflect which FGAP alliances were most similar to my classes, labeled by color (Figures 4-2 through 4-6).

DISCUSSION

This analysis found seven distinct plant classes within the 5 study sites, four of which were common and had associated vegetation data from plant surveys. More vegetation surveys might verify whether the three less common classes ("Magenta", "Cyan" and "Red") were distinct, less frequent, plant communities. There was some difference in the distribution of pixels in each class the AOI between years, but less difference for each of the five study areas between years. Many pixels, classified as either "Yellow" or "Green", switched between these two classes in the 1996 and 1998 images. The vegetative characteristics of each class did not differ significantly between years (i.e. the "Green" class in 1996 was not different from the "Green" class in 1998). The differences between these classes were mostly floristic in terms of presence, although maximum vegetation height and effective height did differ somewhat between classes. A description of these classes proceeds as follows:

- The "Purple" class: This class had very high cover/abundance of *Cladium jamaicense*, as it dominated the community index (Appendix) and had the highest mean percent cover/abundance of *Cladium* for both years. This high predominance of *Cladium* may also be responsible for the high maximum vegetation height, as *Cladium* can grow taller than any other graminoids (Steward and Ornes 1975). Species richness was lowest of the four classes, as were effective height and percent cover/abundance of *Muhlenbergia filipes* and of *Rhynchospora* sp. Two water-tolerant herbaceous species, *Hymenocallis palmeria* and *Oxypolis filiformis*, also had high index values, suggesting that this class may be indicative of wetter areas. These pixels had the deepest average ponding depth of the classes and supported hydrophilic species.
- The "Yellow" class: This class, along with the "Green" class, represents an "average" in most of the vegetation characteristics. Relative to the "Green" class, the "Yellow" class was characterized by higher maximum vegetation height, and lower percent cover/abundance of *Muhlenbergia filipes* and

Schizachyrium rhizomatum. There were high community index values for *Rhynchospora* sp., *Panicum virgatum* and *Cuscuta campestris* (a parasitic vine). This class had the longest average annual hydroperiod, indicating that this class may be slightly wetter than the "Green" class.

- The "Green" class: In addition to the differences mentioned above between "Green" and "Yellow" classes, the "Green" class had high community index values for *Cladium jamaicense*, *Schizachyrium*, *Muhlenbergia* and *Rhynchospora*, *Cuscuta campestris* and *Schoenus nigricans*. This class was similar to the "marl prairie" class of Werner (1975) and Olmsted and Loope (1984). "Green" was drier than "Yellow", but wetter than "Orange".
- The "Orange" class: This class had very low maximum vegetation height and high effective height, high species richness, lower percent cover/abundance of *Cladium*, and high percent cover/abundance of *Muhlenbergia*, *Rhynchospora* and *Schizachyrium*. The prominence of the latter three graminoids may explain the high effective height, as these three species tend to grow in thick clumps (Herndon and Taylor 1986). All four graminoid species had the highest four community index values, and were all almost equally ranked. This class was the only one that had *Muhlenbergia* (instead of *Cladium*) ranked first. This class had the shortest hydroperiod and average ponding depth of the classes, and is probably indicative of drier patches across the prairie.

None of my four classes seem to match the peat prairies of Gunderson and Loftus (1993), but three of the classes ("Green", "Yellow", and especially "Orange") did fit their description of marl prairies. Although the "Yellow" class had a slightly longer average annual hydroperiod than the "Purple" class, the deeper standing water in the "Purple" areas may encourage species more common in slough or marsh areas. These hydroperiod differences suggested by the modeled data may result from only a few centimeters' difference in elevation (Olmsted and Armentano 1997). This relationship is most obvious in the TS site, where the western half of the site is on the edge of Taylor Slough (Figures 4-6a through d). Prior to 1979 (when pumping station (S-332) was activated, Figure 4-1), the TS site was fairly dry and dominated by *Muhlenbergia* (Werner 1975, Herndon and Taylor 1986). Now, the area is much wetter and dominated by *Cladium* (see Part 3 of this dissertation) (Nott *et al.* 1998). As hydroperiod and ponding depth decrease, grasses and sedges that form tussocks may increase and dominate the structure of the habitat. As the community index value of *Cladium* decreased, the combined values of *Muhlenbergia, Rhynchospora*, and *Schizachyrium* outweighed that of *Cladium* (Figure 4-7). In this way, hydrology can indirectly affect sparrow breeding by influencing the availability of structural characteristics for nest sites. Of the four classes found here, the vegetation characteristics of the "Orange" class are most similar to vegetation conditions found in areas that support high densities of sparrows (Figure 3-4 a through k).

The FGAP classification scheme failed to detect the degree of vegetative differences between my classes, as this scheme classified most of my transects (and the majority of my sites) in the "*Muhlenbergia filipes*" alliance. The 3.0 FGAP map seems to homogenize most classes from the 2.1 map into the "*Muhlenbergia filipes*" and "*Cladium mariscus* var. *jamaicense*" alliances on my sites. The FGAP classification alliances are fairly broad, and do not include the detail I found in my study. Perhaps what I found should be considered "associations" (Grossman *et al.* 1998). There are also important physiognomic differences at this finer scale (such as maximum vegetation height and effective height) that the FGAP scheme does not use, but which may be important to the Cape Sable seaside sparrow's life history strategy. Using purely floristic characteristics (as the FGAP scheme does) to define alliances and associations may not be sufficient for these prairie and marsh community classifications.

There are several problems inherent to any classification system that uses advanced remote sensing, and these apply to this research as well. The resolution of the images may not correspond to the patch size of habitat on the ground (Obeysekera and Rutchey 1997). The Landsat images I used had a 28.5 m resolution. If the wet prairies and marshes interlace in patches smaller than this, I may not have detected them. Indeed, several studies have suggested that standard aerial photographs give much more precision to vegetation mapping in the Everglades (Rutchey and Vilcheck 1994). The degree of heterogeneity in the habitats can exacerbate this problem as well. An accuracy assessment, or sampling specific pixels in the field to determine whether the classification matches the plant community on

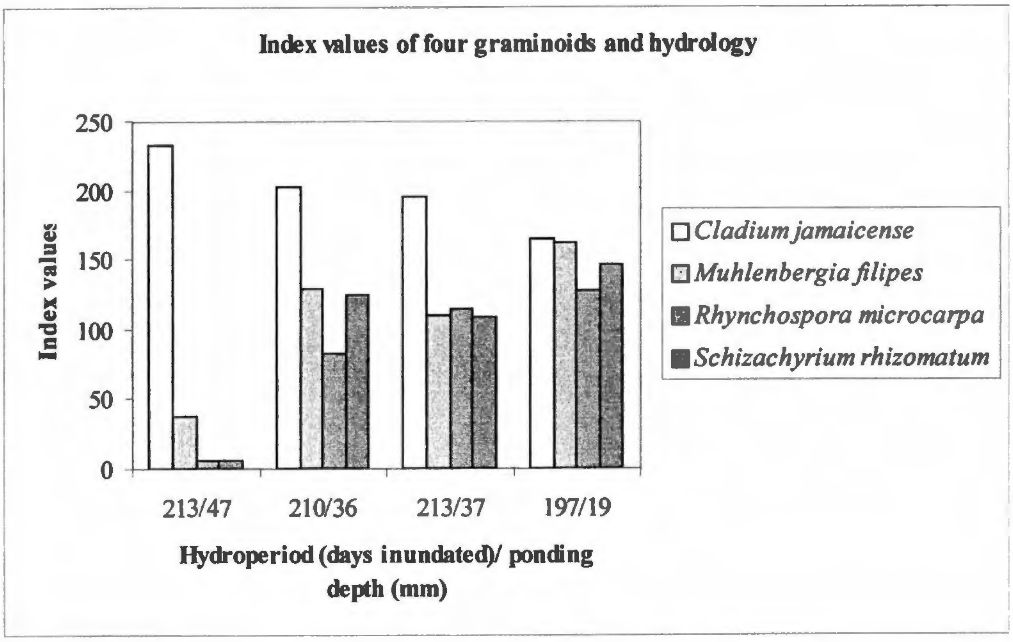

Figure 4-7. Predominance of four graminoid species in areas of decreasing average annual hydroperiod and average annual ponding depth (see Appendix).

the ground, would partially alleviate this problem. Most critical is more ground data for the classes that I have few transects for. Qualitatively, I can judge the accuracy of my classifications based on my time spent in the study areas, but only more transects would give me any quantifiable estimate of the classification accuracy. More detailed studies are necessary to determine if the sparrow prefers a subset of these four classes for particular life history events (such as nesting or molting). In addition, studies should focus on the effect of scale on habitat use and on patch dynamics. Creating a series of maps that span many years can increase our understanding of the dynamic relationship of these prairie expressions to hydrology and fire.

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PART IV APPENDIX: COMMUNITY INDEX VALUES FOR PLANT SPECIES IN EACH CLASS

Cover, frequency, and Index values for plant species in each class (see Amundsen 1977 for methods). Height characteristics (approximate): 5=75-150 cm, 4=50-75 cm, 3=25-50 cm, 2=10-25 cm, 1=10 cm or less. Bolded numbers are the ten highest species indices in each class.

Species	Ht.	Purple			Yellow			Green			Orange		
		cover	freq.	index	cover	freq.	index	cover	freq.	index	cover	freq.	index
Muhlenbergia filipes	5	1.14	0.10	**23.54**	1.63	0.59	**105.35**	1.87	0.66	**127.56**	2.19	0.74	**163.25**
Cladium jamaicense	5	2.62	0.83	**217.92**	2.46	0.79	**197.18**	2.40	0.84	**201.52**	2.20	0.70	**157.98**
Rhynchospora microcarpa	5	0.42	0.24	**21.25**	1.42	0.57	**102.77**	1.26	0.46	**87.09**	1.66	0.81	**139.02**
Schizachyrium rhizomatum	4	1.02	0.37	**75.42**	1.56	0.60	**109.54**	1.80	0.69	**134.03**	1.81	0.76	**134.80**
"Schoenus"	4				0.35	0.05	8.13				1.21	0.31	**60.64**
Cuscuta campestris	3	0.78	0.09	14.38	1.40	0.22	**44.39**	1.38	0.20	**41.60**	1.61	0.23	**55.72**
Ipomoea sagittata	1	0.86	0.11	13.54	0.80	0.11	11.20	0.87	0.07	8.56	0.93	0.29	**30.16**
Hymenocallis palmeri	4	1.62	0.24	**35.83**	1.49	0.21	**32.17**	1.47	0.22	**32.75**	1.15	0.23	**27.57**
Pluchea rosea	3	1.34	0.15	**32.92**	1.41	0.15	**25.48**	1.34	0.09	**17.45**	1.05	0.22	**26.40**
Panicum tenerum	5	0.71	0.07	11.25	1.14	0.19	23.24	1.00	0.11	12.18	1.13	0.21	**25.60**
Panicum virgatum	5	0.76	0.19	16.67	1.24	0.20	**26.15**	1.07	0.14	**18.06**	1.06	0.19	22.15
Centella asiatica	2	1.18	0.32	**51.46**	0.85	0.20	**27.54**	0.93	0.18	**26.09**	0.99	0.14	16.32
Eleocharis cellulosa	5	0.26	0.09	13.96	0.40	0.05	7.96				0.28	0.06	11.73
Aster tenuifolius	3	0.83	0.11	8.13	0.85	0.10	9.84	0.93	0.10	10.07	0.79	0.09	9.53
Crinum americanum	5	0.86	0.09	13.96	1.13	0.08	15.54	0.82	0.06	12.36	0.52	0.06	9.29
Rhynchospora tracyi	5				0.72	0.06	10.33	0.53	0.09	15.22	0.76	0.05	8.54
Proserpina palustris	2										0.63	0.07	8.26
Aster dumosus	3	1.02	0.13	12.08	0.70	0.06	6.10	0.70	0.07	7.90	0.59	0.06	6.14
Agalinis fasiculata	4										0.76	0.06	5.74
Schoenus nigricans	4	0.94	0.22	**21.04**	1.07	0.12	**24.04**	1.2	0.15	**36.77**			
Phyla nodiflora	1	1.02	0.21	**21.88**	0.37	0.06	9.26	0.49	0.07	9.16			
Eupatorium mikanioides	4	0.71	0.08	14.58	0.39	0.04	7.17	0.44	0.04	7.71			
Solidago stricta	5	0.51	0.04	5.63				0.82	0.06	6.65			
Eragrostis elliotti	5	0.63	0.09	16.25	0.25	0.03	5.15						
Oxypolis filiformis	5	1.01	0.31	**41.67**									
Bacopa caroliniana	1	0.42	0.07	13.13									
Hyptis alata	4	0.45	0.07	6.67									
Polygala grandiflora	3	0.21	0.08	6.25									

PART V: THE EFFECT OF LIMITED OPTIONS AND POLICY INTERACTIONS ON WATER STORAGE POLICY IN SOUTH FLORIDA

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ABSTRACT

Due to environmental constraints and reactive water management practices, water shortages exist across the Everglades ecosystem. A growing human population and continued wetlands damage and loss decrease the system's ability to provide water for sustained natural areas and for human uses. A plan called "The Restudy" proposes that $8 billion dollars be spent to restore the Everglades while also continuing to provide water storage for urban and agricultural areas. The Restudy proposes a mix of water storage systems to provide for the predicted future growth in water demand. This mix is purported to be the most cost-efficient at providing water supplies, within the constraints of unchanged agricultural and urban land use. However, several land use and agricultural policies affect cost factors, and can change the relative cost-efficiency between storage systems. The recommended storage plan does not include consideration of ecological benefits, nor does it examine options that alleviate increased water demands. Because of the Restudy's initial assumptions and constraints, it may not advocate the most economically and ecologically sound remediation plan and will not result in the maximization of net social benefits.

INTRODUCTION

In 1992, Congress passed a bill requiring the U.S. Army Corps of Engineers (USACE) to create a plan to rectify environmental damage done by the 1948 Central and South Florida Project for Flood Control and Other Purposes (C&SF Project). The overriding goal of the "Comprehensive Review Study" (otherwise known as the "Restudy"), is to restore the Everglades ecosystem over the next 30 years, while maintaining adequate flood control and water supply for human uses (USACE and SFWMD 1999). The final draft of the Restudy recommends $8 billion worth of hydrologic projects to restore, protect, and preserve the water resources of the Everglades ecosystem (USACE and SFWMD 1999). The Restudy uses a combination of water storage systems to develop enough storage capacity to meet the projected future urban and agricultural water demand. These storage systems include above-ground and in-ground reservoirs, and Aquifer and Storage Recovery (ASR) wells, which inject water into aquifers during periods of high rainfall, and pump water out of the aquifer during periods of water shortage.

The Restudy does not begin with a water budget for the watershed, which would determine water needs for the environment and available "excess" water for urban and agricultural demands. Instead, the Restudy begins with the assumptions of constant agricultural production and urban activity, and future projections of human population growth and human water demands. The options the Restudy considers are constrained by these assumptions. The Restudy endeavors to increase water storage capacity across the Everglades system, without converting a significant amount of land from agricultural or urban uses to provide storage. It also aspires to improve or completely restore ecosystem function and improve habitat conditions for native species. By beginning its proposal with limited options, the Restudy presents a restoration plan that may not be socially or ecologically optimal, and is not ecologically "restorative" as defined by the Society of Ecological Restoration (1995).

Within the framework of the above assumptions and constraints, I determine relative differences in cost-effectiveness (measured in $\$/m^3$ of water stored) between reservoirs and ASR systems, using the Restudy's cost data. I then use sensitivity analysis to demonstrate how changes in cost equation variables can have a measurable effect on total cost estimates for storage. Changes in initial assumptions and agricultural policies can produce changes in these cost variables, which can affect the relative cost-effectiveness of one storage system over another. A storage option consisting of all above-ground reservoirs would require flexibility in urban and agricultural land use and changes in some agricultural policies. But this option may provide gains in cost-effectiveness and bring the project closer to a true ecological restoration of the watershed. Finally, I briefly discuss some other water supply solutions the Restudy does not consider. None of these analyses represent true costs and values of storage scenarios, as they rely on the accuracy of the Restudy's estimated costs. Rather, they can illustrate the sensitivity of the Restudy's results to its methods of analysis and its ability to meet its narrow objectives, and highlight relative differences between storage choices. Including a widened range of initial assumptions and less constrained management options on Everglades restoration plans can have a considerable effect on the economic efficiency of these plans.

DRAINAGE AND WATER DEMANDS IN THE EVERGLADES

Prior to wetlands drainage activity in the late 1800's, the Everglades ecosystem was a 10,000 km^2 (3861 mi^2) watershed, flowing from the Kissimmee River in central Florida, through Lake Okeechobee, and down into Florida Bay (U.S. Department of Interior 1994, Davis and Ogden 1994). This large ecosystem supported a diversity of species and communities, from pinelands and hardwood hammocks on higher elevations, to sloughs, freshwater prairies, and mangrove swamps in lower and coastal areas. The vast expanse of the predrainage Everglades provided a great deal of diversity in hydrologic conditions and habitats (Harwell 1998). Aperiodic disturbance regimes (fire, flooding, drought, and hurricanes) helped maintain a patchy network of habitat types (DeAngelis and White 1994).

Draining the Everglades became a priority when the U.S. Congress passed the Swamplands Act in 1850 (Light and Dineen 1994, Purdum *et al.* 1998). By the early 1900's, over 380 km (236 mi) of canals created direct flowways draining Lake Okeechobee into the Atlantic Ocean. Floods resulting from hurricanes in 1926 and 1928, 1947, and 1948 prompted the U.S. Congress to authorize the C&SF Project in 1948. This plan created one massive, coordinated drainage project to supercede the more limited efforts of the past.

By 1990, over 2830 km^2 (1093 mi^2) of drained and filled agricultural land on the south end of Lake Okeechobee split the ecosystem in half (Light and Dineen 1994). Urbanization from the eastern and western seashores has combined with agricultural activity to reduce the overall extent of the Everglades by 50%. This is a loss of the expanses that allow the persistence of many habitats with different hydrological requirements. Conservation efforts for one species often conflict with others. As an example, water management plans to aid the declining wood stork (*Mycteria americana*) population often conflict with plans to aid the endangered snail kite (*Rostrhamus sociabilis*) population (Weller 1999). Drought periods that help concentrate the wood stork's food (fish) decimate the only food source of the snail kite (apple snails (*Pomacea paludosa*) (Bancroft *et al.* 1994, Bennetts *et al.* 1994). Optimally, restoration should lead to the continued persistence of all native species.

Everglades National Park, Biscayne National Park, Big Cypress National Preserve, and three water conservation areas constitute the extant natural areas (Figure 1-1). Agricultural and urban runoff pollutes these areas and enhances the spread of cattails and exotic species (Walters and Gunderson 1994). Excessive drainage and tilling in the agricultural areas have exposed the peat soils and led to the sinking and loss of the solum. Since 1924, estimates of soil loss due to oxidation, erosion, and rock mining, range from 2 to 2.5 m (7-8 ft) (Stephens 1974, Snyder 1994). This may have a substantial impact on regional water flows and agricultural productivity (U.S. Department of Interior 1997, Gunderson and Loftus 1993).

As the extent of drained land increased, population growth followed. In the Lower East Coast area (Miami, Ft. Lauderdale, etc.), the urban population increased from 22,961 in 1900 to over 4 million in 1990 (Figure 5-1) (U.S. Census Bureau 1999). The population of south Florida is forecasted to reach 12 million by 2050, an increase of 300% in 60 years (Maltby and Dugan 1994, Governor's Commission for a Sustainable South Florida 1995, U.S. Census Bureau 1999). The South Florida Water Management District (SFWMD) predicts an increase in water demand of 38% by 2010, with the greatest increases in urban and industrial users (USACE and SFWMD 1999). Such growth will place an increased pressure on a diminished Everglades ecosystem to provide both fresh water and flood control, as well as economic and recreational resources.

WATER AVAILABILITY AND CURRENT STORAGE CAPACITY

With over 1 m (3.3 ft) of rainfall annually, the Everglades may seem unlikely to suffer from water shortages. Although average rainfall is about 1.2 m (3.9 ft) per year, dry years can receive as little as 0.9 m (3 ft) per year, while hurricanes in wet years can increase rainfall to 1.3 m (4.3 ft) in a year (USACE and SFWMD 1999). Rainfall is highly variable throughout the year as well, as about 60-70% falls in the wet season from June to October (Abtew and Khanal 1994, Henry 1998). Very little of the rain that falls is available for human use. In urban areas where impermeable surfaces cover over 75% of the land, more than 55% of the rainfall can be lost to runoff, instead of being retained in surficial wetlands or in underground

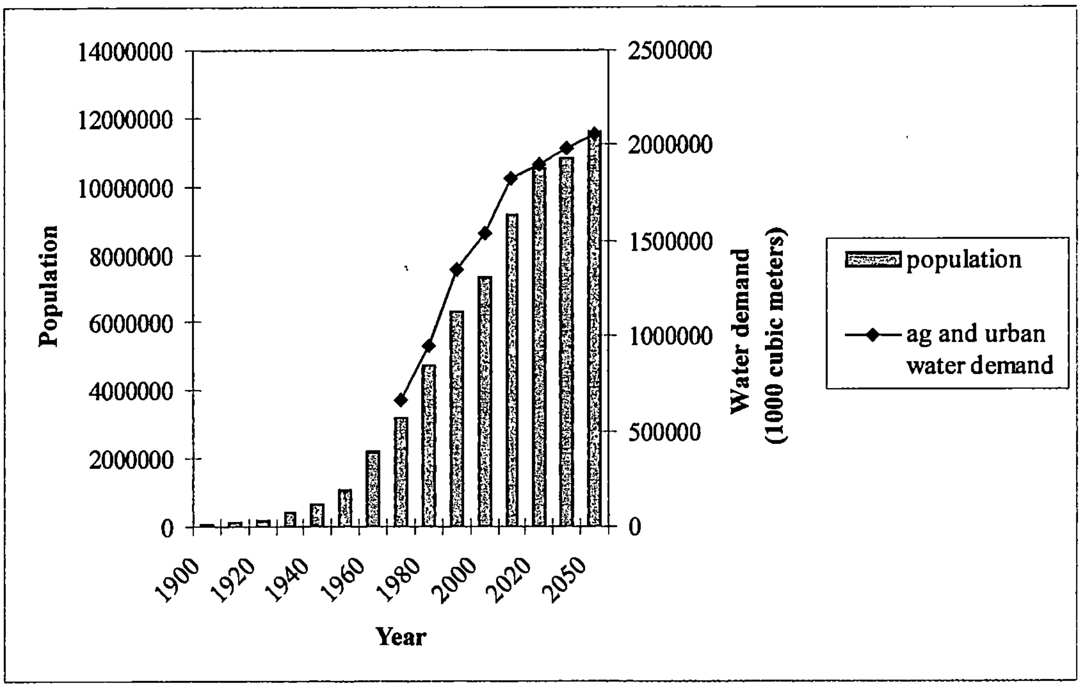

Figure 5-1.Population growth in south Florida and water demand, 1900-2050 (U.S. Census Bureau 1999, USACE and SFWMD 1999).

aquifers (Livingston *et al.* 1998). In addition, evapotranspiration rates from 70% to 100+% in the Everglades also limit the actual amount of water available for human use (Betz 1984, Burns and McDonnell 1994, U.S. Geological Survey 1999). In fact, with the exception of June and August through October, most of south Florida experiences a moisture shortage or "water deficit" (Visher and Hughes 1975, Henry 1998). In dry years and seasons, human water use can further stress the ecosystem by creating even drier conditions in natural areas.

There are three major aquifers in south Florida. The Biscayne aquifer lies close to the surface, while the Upper and Lower Floridan aquifers lie below it. At present, water demand is met primarily from the Biscayne aquifer or from surface water bodies (such as Lake Okeechobee and the Water Conservation Areas) (Berndt *et al.* 1998). The Biscayne aquifer is the primary source of potable water for Dade, Broward, and parts of Palm Beach and Monroe counties. The Biscayne aquifer is a "surficial aquifer" and recharged primarily by rainfall. It lies just below the soil layer, and can be a few meters to over 60 m (200 ft) thick (Light and Dineen 1994). It experiences evaporation-to-rainfall ratios of 90-98% (i.e. it loses over 90% of water up through the surface substrate), depending upon temperature and solar radiation at the soil surface (Merritt 1996). An "intermediate confining unit", composed of sand, clay, and limestone separates the Biscayne aquifer from the Upper and Lower Floridan aquifers that lie below (Yobbi 1996). Another confining unit from 380 m (1247 ft) to about 610 m (2001 ft) below sea level separates the Upper and Lower Floridan aquifers. The salinity of the water in all three aquifers increases with depth. Water in the Biscayne aquifer is fresh to slightly brackish, while water in the Lower Floridan aquifer has salinity equal to seawater, especially near the coasts (Yobbi 1996, Merritt 1996). Aquifers can be spoiled for human use by pollution and saltwater intrusion through overpumping. This has already occurred along the eastern edge of the Biscayne aquifer (VanArman et al. 1998).

Since the 1960's, three large, above-ground "Water Conservation Areas" (3554 km^2; 1372 mi^2) have been used for water storage, as well as water treatment, aquifer recharge, and wildlife habitat (Figure 1-1) (Light and Dineen 1994). These are completely enclosed by dikes and canals, and located between the Everglades Agricultural Area (EAA) on the southeastern shore of Lake Okeechobee and the Fort Lauderdale/Miami urban area. "Stormwater Treatment Areas" are also above-ground facilities through which water is regulated by gates and canals. Currently, there are two facilities in the southwest corner of the EAA which cover about 21 km^2 (8 mi^2), and are primarily used to filter phosphorus and other agro-chemicals from agricultural runoff. At present there are several small-scale ASR systems, including those at Cape Coral (southwest Florida), St. Petersburg, Florida, and Hialeah (Dade County, Florida) (Quiñones-Aponte and Wexler 1995, Yobbi 1996, Merritt 1996).

Although the Restudy does not state the current storage capacity of the system in south Florida, it does suggest that current capacity is insufficient to meet the needs of agricultural, urban, and environmental demands. The Restudy estimates municipal and industrial (a.k.a. "urban") water demand in the 1990's at about 1.3 billion m^3/yr (1,094,500 acre-feet (acre-ft)/yr) (USACE and SFWMD 1999), and Abtew and Khanal (1994) estimate an average annual agricultural demand (in excess of water provided by rainfall) of 50,736,000 m^3/yr (41,132 acre-ft/yr). Water demands of natural areas would be above and beyond those of urban and agricultural demands. However, the Restudy does not offer a water budget estimate for the Everglades, and hence no drought and flood tolerances of natural areas.

THE RESTUDY'S SCENARIO FOR WATER SUPPLY

The Restudy uses several assumptions as constraints for its proposal. It assumes human population will continue to grow, and water demand will increase in proportion to population growth. It also assumes that land use patterns, the intensity of agricultural and urban activities, and the physical environment will remain relatively unchanged from present conditions. The Restudy correctly places responsibility for land use choices on the state and local governments (USACE and SFWMD 1999). However, the Restudy implies a set of policies that, in effect, allows continued agricultural and urban activity and increased human population growth to impinge on the Everglades by assuming these increases as constraints to available storage alternatives. The Restudy's solution to predicted water shortage due to population growth is to expand water storage capacity. The Restudy presents three water storage system options: above-ground reservoirs, in-ground reservoirs, and Aquifer Storage and Recovery systems.

Above-ground storage areas, such as reservoirs and wetlands that retain a shallow sheet of water over a very large area, lose a large quantity of water to evaporation and evapotranspiration. Real estate costs associated with creating these reservoirs can be very high, dependent on perceived land values, but construction costs are relatively low (in Appendix F, USACE and SFWMD 1999). Levees, canals, and pump stations regulate the flow of water into and out of the areas. Depending upon the permeability of the underlying substrate, some areas (such as those proposed in the Restudy) must be lined with a semi-permeable rock or rock and sand mix to prevent bedrock seepage. Some above-ground reservoirs, such as the Water Conservation Areas and "Stormwater Treatment Areas", are already in use in the region.

In-ground water storage areas are also reservoirs, formed by either natural (such as Lake Okeechobee) or man-made depressions. Although evaporation losses are still high, they are not as high as those in heavily vegetated above-ground areas, since there is less surface area of water exposed to the air per volume and less evapotranspiration (Yingling *et al.* 1998). They require less land per unit of water stored, although their construction costs are higher. They also require canals, levees, and pumping stations, as well as some sort of "seepage barrier" if the rock beneath and surrounding the reservoir is porous. The in-ground reservoirs proposed for the Restudy will need seepage barriers, and in some cases 30.5 m (100 ft) "curtain walls", to keep the water from seeping into the surrounding rock and aquifers (in Appendix F, USACE and SFWMD 1999).

The proposed Aquifer Storage and Recovery (ASR) systems will use deep injection wells to pump surface water into the Upper Floridan aquifer. By storing water underground, all evaporation losses are greatly diminished, although not all of the water injected into the aquifer is recoverable. Injected water can mix with "native" groundwater (water that has seeped into the aquifer by natural means), which may be more saline. There is a fair amount of uncertainty with using ASR technology at the scale proposed by the Restudy. While ASR systems have been used in Florida for several decades, none have been built to this scale (in Appendix O, USACE and SFWMD 1999). Land requirements for ASR systems are fairly small, although construction and engineering costs are substantial. These costs are especially high if the water must be extensively treated before injection into the aquifer (to prevent groundwater pollution), or after the water is recovered (due to pollution from the native ground water) (National Academy of Sciences 1994, USACE and SFWMD 1999, p. N-133).

The Restudy recommends a mixture of water storage systems to meet present and future water demands. For all of south Florida, the Restudy plans to construct 10 reservoirs (above-ground and in-ground) with a capacity of 1,842,782,116 m^3 (1,493,970 acre-ft). In addition, the Restudy proposes 200 ASR wells with a capacity of about 1,323,401,925 m^3 (1,072,901 acre-ft, assuming an injection period of 7 months of the year, with a recovery rate of injected water of 70%; USACE and SFWMD 1999, p. 4-31). The total additional storage capacity provided by the Restudy's plan would be between 2 and 3 billion m^3 of water per year (1.7 - 2.5 million acre-feet per year) (USACE and SFWMD 1999, p. 7-62). This capacity would meet the total projected 2050 urban and agricultural demand of about 2 billion m^3 per year (1,664816 acre-ft/yr) (Figure 5-1), and hopefully the presupposed water needs of natural areas as well.

Cost per m^3 of water stored comparisons between storage systems

I organized the construction and real estate cost estimates provided in the Restudy document, to determine relative differences in costs per cubic meter of water storage between reservoirs and ASRs (Table 5-1) (USACE and SFWMD 1999). All data are in 1999 US dollars, unless otherwise noted. The data provided by the Restudy document are preliminary estimates, and my comparisons between storage capacity of reservoirs with that of ASR capacities are dependable only within the constraints of the accuracy of the cost estimates (USACE and SFWMD 1999, p. 9-57 through 9-58, and F-101 through F-123). I do not debate the accuracy of the cost estimates here, but rather illustrate how total project costs may be sensitive to variations in some of the cost components. The Restudy's cost equation for water storage projects is the following:

Total Cost = C + L + A + (L+A)c + S + F

where:

C = construction costs

L = land costs (per km^2 cost of land times number of km^2)

A = Federal and non-federal land acquisition costs

c = contingency percentage

S = subterranean seepage barrier costs (in-ground reservoirs only)

F = filtration costs (ASR systems only)

The Restudy estimated construction using data from previous, similar projects and MCACES software (Micro Computer Aided Cost Engineering System) (USACE and SFWMD 1999). Land cost was based on either market value (based on sales data of nearby land of similar use), or property appraisals by county assessors (which use income from the land for properties classified as agriculture) (A. Zech, pers. comm. 8 July 1999). The Restudy did not indicate from which source each land cost estimate originated. Federal and state land acquisition costs were based on prior purchases of nearby land (or an average of $72,000 per tract if prior acquisition costs were unobtainable; here, "tract" indicates an area of land owned by one party). I gathered construction costs for each project from Appendix C, Attachment A, and all other costs from Tables 9-2 and F-2 (USACE and SFWMD 1999). I did not include annual operation and maintenance costs.

Table 5-1. Storage capacity and cost data (in \$1000, October 1999 \$'s) for reservoir and ASR projects in the Restudy plan. (Source: USACE and SFWMD 1999, Table F-2, Table 9-2). Cost abbreviations follow those in cost equation. Some projects have different land costs for some tracts.

Project name	Type	Size (km^2)	Capacity (1,000 m^3)	C costs	Land cost per km^2	A costs	c costs	c %	S costs	F costs	Total costs	Total cost/m^3
North of Lake Okeechobee Storage Reservoir (A)	above ground	80.94	246696	95134	1483	6480	36240	50	0	0	284854	\$1.15
Taylor Creek/Nubbin Slough Storage and Treatment Area (W)	above ground	40.47	61674	74326	445	1800	9900	50	0	0	104026	\$1.69
C-43 Basin Storage Reservoir and Aquifer Storage and Recovery (D)	above ground	80.94	197357	68614	40.47 @ \$618, 40.47 @ 1557	414	44207	50	0	0	193973	\$0.98
C-44 Basin Storage Reservoir (B)	above ground	40.47	49339	21888	1483	450	30225	50	0	0	112563	\$2.28
C-23/C-24/C-25/Northfork and Southfork Storage Reservoirs (UU)	above ground	195.67	430978	281175	124 @ \$1433, 72 @ \$1483	2052	143016	50	0	0	710223	\$1.65
Everglades Agricultural Area Storage Reservoirs (G)	above ground	70.82	148015	350112	840	180	26856	45	0	0	436648	\$2.95
Palm Beach County Agricultural Reserve Reservoir (VV)	above ground	6.72	24670	21777	4942	324	19219	50	0	0	76958	\$3.12
Site 1 Impoundment (M)	above ground	9.95	18502	14947	3 @ \$2965, 7 @ \$1237	611	5076	≈50	0	0	33583	\$1.82
Bird Drive Recharge (U)	above ground	11.64	14185	52459	2471	18980	23875	50	0	0	124084	\$8.75
North Lake Belt Storage Area (XX)	in-ground	23.72	111013	54314	5 @ \$247, 19 @ \$4942	16200	44248	40	326879	0	536061	\$4.83
Central Lake Belt Storage Area (S)	in-ground	23.35	234361	89258	2.5 @ \$13591, 15 @ \$1853, 5.7 @ \$865	2088	31146	45	313244	0	502861	\$2.15
Water Preserve Areas/L-8 Basin (GGG)	in-ground	7.28	59207	327815	2471	234	9117	50	246067	0	601233	\$10.15
Lake Okeechobee Aquifer Storage and Recovery (GG)	ASR	1.21	556395	609741	865	3960	2505	50	0	499056	1116312	\$2.01
C-43 Basin Storage Reservoir and Aquifer Storage and Recovery (D)	ASR	0.27	122407	134502	618	1.37	83.18	50	0	110458	252222	\$2.06
Water Preserve Areas/L-8 Basin (K)	ASR	1.54	139099	55725	.2 @ \$2471, 1.4 @ \$741	1440	1430	50	0	25081	85096	\$0.61
C-51 Regional Groundwater Aquifer Storage and Recovery (LL)	ASR	0.14	94587	122391	37065	1530	3315	50	0	0	132336	\$1.40
Palm Beach County Aquifer Storage and Recovery VV	ASR	0.06	41730	44666	4942	4.21	250	50	0	0	47142	\$1.13
Site 1 Aquifer Storage and Recovery (M)	ASR	0.12	83459	101844	1799	7.3	61	50	0	0	106795	\$1.28

Contingency costs represent the degree of uncertainty in the cost estimates for land value and acquisition. The Restudy used Monte Carlo simulations to calculate the range of costs within 90% confidence limits of the estimated cost (in Appendix C, USACE and SFWMD 1999). Although contingency percentage estimates ranged from 5% to 50%, the average percentage estimated was close to 50% for all of the reservoir and ASR projects. In other words, the Restudy simulations found that land value and Federal and non-federal acquisition costs could be one and a half times the cost initially estimated.

For components that included both reservoir and ASR systems (Components D, M, and VV), I used the Restudy's construction costs specific to each system from Appendix C, Attachment A (USACE and SFWMD 1999). Since specific land requirements for ASR systems were only given for ASR Components LL (1 acre per well/well cluster) and GG (1.5 acres per well/well cluster), I took the average (1.25 acres per well/well clusters per acre) of those two components to estimate land requirements (and hence land, acquisition, and contingency costs) for the other ASR systems. Throughout, I note the conversion from cubic meters and square kilometers to acre-feet and acres in parentheses, to maintain consistency between my analysis and that of the Restudy.

Construction costs composed the highest proportion of total costs for ASR systems, and a significant percentage of total costs for above-ground reservoirs (Figure 5-2). Land and contingency costs were important components of total costs for above-ground reservoirs, while the cost of subterranean seepage barrier costs dominated total costs of in-ground reservoirs. Filtration costs were a notable portion of total costs for ASR systems. The initial average cost per cubic meter of water storage for above-ground reservoirs, in-ground reservoirs, and ASR systems was \$2.71, \$5.71, and \$1.41, respectively (\$3343, \$7043, and \$1739 per acre-foot) (USACE and SFWMD 1999). Hazen and Sawyer *et al.* (1994) found a greater cost disparity between reservoirs and ASR systems, with average costs for reservoirs between \$2 to \$8 per cubic meter (\$2467 to \$9868 per acre-foot), but ASR costs from \$0.4 to \$0.7 per cubic meter (\$493 to \$863 per acre-foot). Above-ground reservoirs cost on average twice as much per cubic meter stored as ASR facilities.

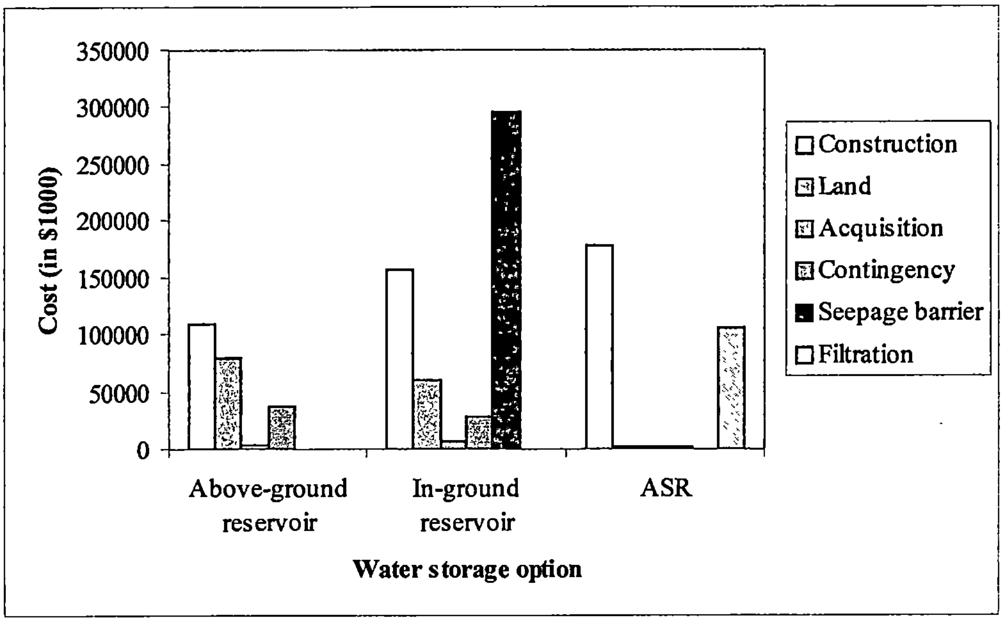

Figure 5-2. Restudy estimates of construction, land, acquisition, contingency, seepage barrier, and filtration costs as proportion of total costs (Source: USACE and SFWMD 1999).

In-ground reservoirs are the least cost-efficient, since they require substantial construction costs (due to the seepage barriers) and real estate costs. All of the in-ground reservoirs proposed in the Restudy document are in sites with past or present rock mining activities. Since excavation costs do not compose the majority of the considered engineering costs, use of rock mines as water storage facilities should not be used as an indirect cost justification for further rock mining activity. This benefit of rock-mining activity should be compared with other costs and benefits.

SENSITIVITY OF STORAGE UNIT COSTS TO CHANGES IN COST COMPONENTS

Sensitivity analyses can help determine which variables most affect the cost efficiency of the water storage systems (Winpenny 1991, Gupta 1994). Changes in the implementation of policies that influence these variables can affect the total cost of the project, and may alter the relative cost-efficiency of the different water storage systems.

I conducted a sensitivity analysis of land, acquisition and contingency costs for above-ground reservoirs versus ASR systems, and water treatment costs for ASRs. I calculated the total cost of storing 2 billion m^3 of water per year (the estimated increase in storage capacity in the system) using only ASRs versus only above-ground reservoirs (USACE and SFWMD 1999, p. 7-62). I used the Restudy's descriptions of above-ground reservoir capacities, and estimated the capacity of ASRs by multiplying the million gallons per day pumping capacity by 210 days (7 months). I took 70% of that value, since the USACE assumed a recovery efficiency of 70% for their wells (USACE and SFWMD 1999, p. 4-31).

First I determined how many square kilometers would be required to store 2 billion m^3 of water annually for each system. I estimated capacity at 2,133,574 m^3/km^2 (7 acre-ft/acre) for above-ground reservoirs and 530,345,499 m^3/km^2 (1740 acre-ft/acre) for ASRs (USACE and SFWMD 1999, p. vii). An all above-ground reservoir system would require about 934 km^2 (230,794 acres), while an ASR system would only require 4 km^2 (988 acres).

As a starting point for the analysis, I used average cost estimates for land, acquisition, contingency, and filtration costs from the Restudy data. As a base cost, I used the land cost the USACE estimated for an above-ground reservoir project in the EAA (Component G, \$840,000/km^2; \$3400/acre). This land cost was the most relevant to the alternative water storage option I will present in the next section. The average per km^2 acquisition costs were \$210,000/km^2 (\$850/acre) for above-ground reservoirs, and \$2,575,000/km^2 (\$10421/acre) for ASR projects. I used a base contingency rate of 50%. For ASR projects, the average filtration cost per km^2 was \$140,155,000/km^2 (\$567,195/acre). When multiplied by 4 km^2 (the area needed for storing 2 billion m^3), average filtration cost was \$560,620,000. To test the sensitivity of the total cost to changes in equation variables, I calculated total cost using 25%, 50%, 125%, and 150% of the base estimate for each variable (Table 5-2).

1. Sensitivity of total cost to per km^2 value of land: I calculated the total cost for 2 billion m^3 capacity storage using land costs from 25% of the EAA estimate (\$420,000/km^2, \$1700/acre) to 150% of the estimate (\$1260000/km^2, \$5100/acre). The range of land cost estimates over above-ground reservoir and ASR projects listed in the Restudy was \$247,100/km^2 to \$13,591,000/km^2 (\$1000/acre to \$55,000/acre) (Table 5-1).

2. Sensitivity of total cost to changes in Federal and non-federal acquisition costs: I calculated the change in total project cost as acquisition cost per km^2 changed from 25% (\$53,000/km^2, \$215/acre for reservoirs; \$644,000/km^2, \$2606/acre for ASRs) to 150% (\$316,000/ km^2, \$1279/acre for reservoirs; \$3,863,000/km^2, \$15,633/acre for ASRs) of the base estimate.
3. Sensitivity of total cost to contingency percentage: I calculated total cost using contingency percentages of 13% to 75% of real estate costs (land costs plus acquisition costs).
4. Sensitivity analysis of water treatment requirements: This analysis affected only ASR costs, as above-ground reservoirs have much lower pre-treatment requirements (especially if some natural water purification is expected to occur in the reservoir). I calculated total costs for a range of treatment costs, from \$186,513,000/km^2 (\$754,799/acre, 25% of average) to \$1,119,048,000/km^2 (\$4,528,677/acre, 150% of average). This is a range of about \$0.07/m^3 to \$0.41/m^3, with an average of \$0.27/m^3, which is in accordance with estimates in the literature. Postel and Carpenter (1997) estimated about \$0.08/m^3 for primary and secondary treatment, to \$0.42/m^3 for advanced treatment.

SENSITIVITY ANALYSIS RESULTS

Water storage in above-ground reservoirs required about 250 times as much land as ASR systems, and reservoir storage for 2 billion m^3 per year was at least twice as expensive in all cases examined. In a base scenario using the average of USACE's land values, acquisition, contingency, construction and treatment costs, an all ASR technology project would cost about \$1 billion, while an all above-ground reservoirs project would cost over \$2 billion (Table 5-2). However, when filtration costs were 125% to 150% of the base scenario, above-ground reservoirs were less expensive than ASR systems when land costs were lower than the base scenario costs. Changes in land cost affected total cost for reservoirs the most, decreasing costs by 52% with a 25% drop in land cost, and increasing total cost by 35% at 150% of the base land cost (Table 5-2). Filtration costs had a large effect on total costs for ASRs, with a 56% decrease with a reduction to just 25% of average treatment costs. Changes in land acquisition costs and contingency rates affected total cost of above-ground reservoirs somewhat, but did not affect total costs of ASRs.

Table 5-2. Results of sensitivity analysis, effect on total cost of changing variables in cost equation by 25% to 150% of average (base). All costs in $1000's, October 1999 $'s.

	Construction	Land/km^2	Land cost	Acquisition cost	Cont. %	Contingency cost	Filtration cost	Total Cost	% change in total cost
Reservoir base:	108937	840	784560	270900	0.5	527730	0	1692127	
ASR base:	178145	840	3360	43775	0.5	23568	746024	994872	
Change in per km^2 land cost									
reservoir	base	25% (210)	196140	base	base	233520	0	809497	-52%
	base	50% (420)	392280	base	base	331590	0	1103707	-35%
	base	125% (1050)	980700	base	base	625800	0	1986337	17%
	base	150% (1260)	1176840	base	base	723870	0	2280547	35%
ASR	base	25% (210)	840	base	base	22308	base	991092	<1%
	base	50% (420)	1680	base	base	22728	base	992352	<1%
	base	125% (1050)	4200	base	base	23988	base	996132	<1%
	base	150% (1260)	5040	base	base	24408	base	997392	<1%
Change in Federal/non-federal Acquisition cost									
reservoir	base	base	base	25% (68370)	base	426465	0	1388332	-18%
	base	base	base	50% (136740)	base	460650	0	1490887	-12%
	base	base	base	125% (339270)	base	561915	0	1794682	6%
	base	base	base	150% (407640)	base	596100	0	1897237	12%
ASR	base	base	base	25% (10948)	base	7154	base	945631	-5%
	base	base	base	50% (21896)	base	12628	base	962053	-3%
	base	base	base	125% (54723)	base	29041.5	base	1011294	2%
	base	base	base	150% (65671)	base	34515.5	base	1027716	3%
Change in contingency percentage cost									
reservoir	base	base	base	base	0.13	137209.8	0	1301607	-23%
	base	base	base	base	0.3	316638	0	1481035	-12%
	base	base	base	base	0.6	633276	0	1797673	6%
	base	base	base	base	0.75	791595	0	1955992	16%
ASR	base	base	base	base	0.13	6127.55	base	977432	-2%
	base	base	base	base	0.3	14140.5	base	985445	-1%
	base	base	base	base	0.6	28281	base	999585	<1%
	base	base	base	base	0.75	35351.25	base	1006655	1%
Change in filtration costs for ASR systems									
ASR	base	base	base	base	base	base	25% (186506)	435354	-56%
	base	base	base	base	base	base	50% (373012)	621860	-37%
	base	base	base	base	base	base	125% (932530)	1181378	19%
	base	base	base	base	base	base	150% (1119036)	1367884	37%

Policies that decrease the cost of land could increase the cost-efficiency of above-ground reservoirs relative to ASR systems. Federal and state water quality regulations could affect filtration costs, as could hydrogeological conditions surrounding the ASR well sites. By ignoring the influence of exogenous policies, such as agricultural subsidies, the Restudy may not present the best management option for providing water storage. As the restoration plan progresses, any policy applications could substantially affect the economic and ecological efficiency of the restoration. The next section addresses this issue in detail.

POLICY "SIDE EFFECTS" AND MISSING COSTS AND BENEFITS

Public policy is never made in a vacuum, nor does it behave as in one when implemented. Policies can and often do influence the effectiveness of other policies that may seem only marginally related. This "centrality of side effects" (Hirschman 1967) is often ignored by policymakers, even when these side effects can modify the efficacy of a policy choice. Policies that influence land costs and filtration costs could have substantial effects on water storage cost components presented in the Restudy. In addition, uncertainty surrounding ASR technology could also increase storage costs in ASR systems relative to above-ground reservoir systems.

Real estate costs

Agricultural production in the EAA and the Farm Bill

Sugarcane is grown on about 80% of the land in EAA, vegetables on about 15%, and rice on about 5% (Snyder and Davidson 1994). Florida provides half of the U.S. domestic sugar supply, although the U.S. imports about 15% of its supply from abroad (Johnson and Ortega 1999). Sugar and high fructose corn syrup (HFCS) comprise all but 1% (by weight) of the sweeteners consumed in the U.S. Corn growers have benefited from high sugar prices, as these prices make HFCS inexpensive relative to sugar. The 1996 Farm Bill provides sugarcane farmers with a substantial amount of crop subsidies through price supports and conditional non-recourse loans. Payments to sugar growers are based on the difference between a "guaranteed price" (which was $0.18 per pound of raw sugar from 1991-1998) and the world market price.

This world price has fluctuated between an average of $0.12 in 1994 per pound to $0.08 per pound in 1999 (U.S. Department of Interior 1994, Coffee, Sugar, and Cocoa Exchange 1999).

According to the U.S. Department of Agriculture's Economic Research Service (ERS) report on costs and returns of sugarcane in Florida, despite the subsidies, farmers have realized profits in only two of the past 16 years (ERS 1999a). Subsidies, such as those received by sugar growers, may encourage more agricultural activity in the EAA than is economically optimal, and may not be justifiable on economic or environmental grounds (Baumol and Oates 1988, Faeth *et al.* 1991, Faeth 1995, Runge 1996, Legg 1996). If these subsidies were eliminated, sugar growers that ceased to make a profit might discontinue farming, and that land could be purchased at lower cost (Knutson 1997). Lifting all protectionist policies for sugar worldwide would likely lead to an increase in the world price of sugar, which would level out at the opportunity cost of sugar production. However, most economists believe this world price would be lower than the current U.S. $0.18 per pound domestic price (Neff and Josling 1991).

The value of land in the EAA depends on the value of the activity or product for which it is used. If the USACE based their land value estimates in their cost analyses on county property assessments that use agricultural income from the land, the land estimates would include government subsidies as income (FL Statute Section 193.461 §6(a)). When government payments are included in income flows used to determine land value, it can greatly increase the present value of the land (Lamb 1997). If ERS residual returns are instead used to calculate land value, and these values are used in a cost-effectiveness calculation of reservoirs, real estate costs drop dramatically. In this way, Farm Bill price supports greatly influence the choice of alternatives of a seemingly unrelated policy (water storage in south Florida).

Wetlands Reserve Program, Farm Bill provisions

Another factor which lessens the apparent gap between above-ground reservoir and ASR costs concerns the provisions in the Food Security Act of 1985 (7 CFR Part 1467, 61 FR 42137), and the Food, Agriculture, Conservation and Trade Act of 1990 (1990 Farm Bill). These provisions are already in use to some extent; a portion of the land for an above-ground reservoir in the EAA will be purchased by the Department of the Interior using Farm Bill funds (USACE and SFWMD 1999, p. 10-56).

The Wetlands Reserve Program was reauthorized in the 1996 Farm Bill until 2002 (U.S. Department of Agriculture 1999). In this program, farmers are paid an easement for each acre they take out of agricultural use and restore to wetlands (Parks and Kramer 1995). If the easement payment (minus restoration costs) is greater than agricultural returns (for sugarcane or rice), it is cost effective for the farmer to convert farmland to wetlands (Parks and Kramer 1995). The more marginal the farmland (in terms of monetary returns to the farmer), the greater the benefit of conversion.

Since the ERS data demonstrate that much of the sugarcane grown in the EAA yields negative returns to the farmers (when government subsidies are excluded), it might make economic sense to convert that land back into wetlands. These wetlands could provide some water storage services to the region. The farmer can lease enrolled land for fishing, hunting, and other recreational uses that are compatible with water storage. Land does not have to be permanently enrolled, but can be enrolled for 30 years (farmers receive 75% of the permanent easement payment, 61 FR 42137). After 30 years (when the Restudy plan is scheduled to be completed), if an excess of water storage capacity exists in the system, some enrolled land could revert back to agricultural use. As of 1995, 133 km^2 (32,864 acres) were enrolled in Florida, representing 22 contracts (Natural Resource Conservation Service 1999). If the USACE were to use the Wetlands Reserve Program to include agricultural land in the restoration projects (instead of requiring that land be bought outright), real estate acquisition and contingency costs could be greatly reduced.

Filtration costs

The degree to which water requires treatment depends on what it will be used for (i.e. drinking water or irrigation), the degree of contamination of the source water, the cleanliness demanded by the injection mechanisms, and the potential for injected water to contaminate (or be contaminated by) "native" ground water. Water pumped directly from the surface into the aquifer will not percolate though wetlands and substrate, which act as a filter (Ewel 1997). Over 80% to 90% of total organic carbons can be filtered out by 25-35 m (82-115 ft) of sediment, and most organic matter is removed within the first meter of soil (Idelovitch and Michail 1984, Wilson *et al.* 1995).

The more demanding the water purity requirements, the more expensive it is to treat. The U.S. Environmental Protection Agency considers ASR wells as Class V injection wells, subject to the underground injection control (UIC) regulations (40 CFR Parts 144 and 146). As such, the water pumped into them must meet minimum pollutant standards, especially if there is a possibility of mixing with other underground water sources that are used for drinking water (and thus regulated by the Safe Water Drinking Act). Municipal wastewater may be much more consistent in its quality and quantity, and much easier to treat, than agricultural run-off (National Academy of Sciences 1994).

Sediments and particulates must be removed, as these can clog the wells on their way into or out of the aquifer (Pyne 1995). Suspended materials also cause water flow changes in the aquifer, which may reduce the water extraction capacity of the well. Water must be treated for bacteria such as *Giardia lamblia* and *Legionella*, since these can not only grow and obstruct the well, but cause serious health problems if released into the drinking water supply (Environmental Protection Agency 1989). Although chlorine or other chemicals can be used to disinfect the water, any excesses can create byproducts with organic compounds known as trihalomethanes, which are carcinogenic and must be reduced before the water can be considered potable (Environmental Protection Agency 1998, USACE and SFWMD 1999, p. N-133). Chlorine is usually removed by dechlorination processes (commonly using sulfur dioxide or thiosulfate), but trace amounts of chlorine are frequently present in water considered potable (National Academy of Sciences 1994). Ozone (O_3), ultraviolet radiation, and reverse osmosis are alternatives to chlorine for microorganism and organic material removal.

The degree to which polluted injection water spreads and pollutes drinking water wells depends on many factors, and therefore can be hard to predict. A steeper hydraulic gradient will spread contaminants faster and throughout a larger area. Large numbers of wells (which can be sources of pollutants) with high pumping capacities will alter the flow of groundwater towards them (Nyer 1992). These wells can hydraulically connect the aquifer layers through which they are drilled, and may increase mixing between previously separated water bodies. The chemical properties of a contaminant, the amount of any contaminant, and the interactions with other chemicals (such as chlorine) also affect pollution probabilities for wells. Although aquifers do have some water cleansing capacity, this can take from decades to thousands of years, depending upon the contaminant and its concentration (Nyer 1992). Often, contaminant plume models are performed with little quantitative data for the above factors, and rarely include likely heterogeneities in hydraulic conductivity within the same area of the aquifer (Hamed *et al.* 1996, Tesoriero and Voss 1997).

ASR technology uncertainties

The recovery efficiency (the proportion of water injected that can be recovered) of the wells, and the effect of the injected volume on the hydrogeology of the aquifer system, are sources of uncertainty. The "buoyancy stratification" of the aquifer (freshwater rises, saline water sinks), the degree of mixing between injected and "native" groundwater, and the dispersal rate and direction of the injected water all affect recovery rates (Merritt 1985). In a pilot project in Cape Coral, Florida, increasing the amount of water injected (from 12,335 m^3 (10 acre-ft) in 7.5 days to 196,123 m^3 (159 acre-ft) in 120 days) decreased water recovered from 100% to 43% (Quiñones-Aponte and Wexler 1995). In addition, the longer the injected water remains in the aquifer, the higher the probability it will mix with "native" aquifer water. Finally, large volumes of water pumped into the aquifer could create enough pressure to fracture the rock system, which might alter the underground water flow or cause sinkholes or subsidence (in Appendix O, USACE and SFWMD 1999). In some areas (e.g. Colorado, Ohio), injected water has increased pressure on the existing fault system and triggered earthquakes (Healy *et al.* 1968, Ahmad and Smith 1988, Nicholson and Wesson 1992).

The USACE will conduct several pilot projects in which ASR technology will be tested on the regional geology. These projects will determine the recovery efficiency possible (under different injection and recovery regimes), and the effects of injected water on the underground aquifer system. If these tests demonstrate considerable negative consequences, the ASR components may be eliminated from the Restudy's storage options, and other options (most likely continuing to use Lake Okeechobee as a reservoir) will be substituted (USACE and SFWMD 1999, p. N-73). Fluctuating water levels in the past have caused considerable damage to the lake's ecological communities, and complete ecological restoration of the lake is unlikely if these fluctuations continue (U.S. Department of Interior 1994, VanArman *et al.* 1998).

ALTERNATIVE WATER MANAGEMENT OPTIONS

One alternative to the above mix of storage systems is to use only above-ground reservoirs to provide for present and future water demand. Reservoirs located in the EAA could increase the connectivity between the headwaters of the Everglades (Kissimmee-Lake Okeechobee) and the Water Conservation Areas and natural areas to the south. This alternative does not maintain the assumption that agricultural land use remain constant, but does hold the same assumptions of unchanged urban land use, and increasing human population and water demands. Keeping the framework of the cost equation used in the Restudy (total costs equal to construction and real estate costs), changes in implementation of the Farm Bill could increase the economic feasibility of this option. Converting residential land into above-ground reservoirs is another option, although this option would require similar changes in policies that decreased the cost of land zoned as residential. This option may not be as ecologically desirable either. Given the current land use patterns, there is far less residential land that could serve as a connection between Lake Okeechobee and the water conservation and natural areas to the south.

There has been no cost-benefit analysis to decide between water storage technologies (USACE and SFWMD 1999, p. 6-15). Economic justifications for the storage system plan in the final Restudy document were based on the estimated construction and real estate cost data presented above. Although the Restudy suggested that treatment requirements may be less than expected for ASR systems, the possibility that seepage barriers may not be needed for reservoirs was not discussed. If reservoirs were intended for aquifer recharge as well as water storage, water loss to the underlying substrate may be a benefit if the reservoir water were not polluted (so as not to pollute the aquifer). Also not included were any of the potential environmental costs and benefits from the various storage technologies. Benefits provided by reservoirs (but not by ASR systems) include flood protection, treatment of agricultural and urban run-off, and the provision of recreational opportunities and wildlife habitat. If water storage in the EAA provides greater net social and economic benefits to south Florida than agriculture does, then this alternative option may be socially optimal. This would be achieved if the opportunity cost of using EAA land for agricultural production were higher than its use for water storage.

Opportunity cost of water storage in the EAA: agricultural productivity

Sugar production in south Florida is rarely profitable without government programs. An agricultural alternative to sugar production in the EAA is rice production (Aillery *et al.* 1997). Sugarcane, grown in the dry season (November through June), requires drained land and relatively deep soil (Alvarez 1993). Rice is grown as a rotation crop in the wet season. It can persist on relatively shallow soil, and can filter phosphorus from agricultural run-off. In areas where soil subsidence has been acute, the maintenance of longer hydroperiods and higher standing water, in combination with rice production, may help stabilize the remaining soil layer. Alvarez (1993) calculated the agricultural returns of rice production when grown in rotation with sugarcane. He found that two rice crops per season resulted in a return of about \$33,853/km^2 (\$137/acre). When only one rice crop was grown, there was a small, negative return to the grower. Alvarez credits rice's profitability to low production costs for rice (\$85,003/km^2; \$344/acre), compared to \$218,189/km^2 (\$883/acre) for sugarcane, \$478,638/km^2 (\$1937/acre) for sweet corn, and \$1,072,179/km^2 (\$4339/acre) for celery.

However, ERS (1999b) data, which did not include direct government payments, indicated that rice production in the United States was almost never profitable (no data were available for south Florida). Residual returns to management and risk from rice crops ranged from a profit of \$714/km^2 (in the Mississippi River Delta in 1997) to a loss of \$84,913/km^2 (in California in 1998). The positive residual returns, realized in the Mississippi River Delta in 1997, were the only positive returns for the entire U.S. in the 1997 and 1998 growing seasons. There may be no crop that can be grown profitably in the EAA without government subsidies. While there may be social or political benefits from these agricultural programs, these benefits should be weighed against the environmental and economic costs to society of the subsidies.

Opportunity cost of agricultural production in the EAA: water storage

The Restudy analysis does not include the functional benefits derived from reservoirs, particularly those above ground, which have nonzero (albeit difficult to measure) value. Reservoirs can provide flood protection, water filtration, groundwater/aquifer recharge and soil regeneration (if no seepage barrier is installed), wildlife habitat, and recreation (Owen 1995). These benefits can also be thought of as "natural capital" (*sensu* Repetto *et al.* 1989), with stock (km^2 of wetland able to provide these services) and flows of resources (e.g. amount of water cleaned and stored, wildlife produced) (Kahn 1995, Wackernagel and Rees 1997). In fact, with an increasing population or an increase in material consumption, stock of natural capital must increase accordingly, to provide the same amount of resource flow (Barbier 1994, Wackernagel and Rees 1997).

Many studies have calculated a monetary value per land unit of a wetland or freshwater resource (Heimlich *et al.* 1998, Wilson and Carpenter 1999). Assigning a price to land based on its worth as an ecosystem that provides functions, services, and goods usually involves a combination of direct market value (e.g. commercial fish harvests) with indirect market values based on heuristic valuation, travel cost, and contingent valuation techniques (Costanza *et al.* 1997). Other estimates use the cost of replacing the service (water treatment, transportation) as a value for the ecosystem (Postel and Carpenter 1997, Ewel 1997, Chichilnisky and Heal 1998). Here, I assume that while above-ground reservoirs do not provide all of the ecosystem services that wetlands do, the services they do provide (water storage, flood protection, water treatment) have the same value as those provided by wetlands.

Costanza *et al.* (1997) and Heimlich *et al.* (1998) surveyed many studies that evaluated the ecosystem benefits provided by wetlands. I converted Heimlich *et al.*'s (1998) annual value per acre or hectare estimates into 1999$ per km^2, then calculated the annual value 934 km^2 of the EAA might provide as above-ground reservoirs. I then calculated the annual value of sugarcane or rice production in 934 km^2 of the EAA. I included the most profitable residual rate of return to management provided by the ERS (1999a and 1999b), the Restudy's \$840,000/km^2 land value estimate in the EAA (assuming this is based on agricultural income and not market value of the land), and Alvarez' (1993) profit estimate for rice in the EAA. Once I calculated the annual value of each service or agricultural productivity, I estimated the present value of these services using a 6% interest rate and an infinite time horizon. With the exception of the estimates for sugarcane production and Alvarez' rice estimate, none of the service estimates are specific to Florida.

Assuming that the only service value the above-ground reservoirs could provide is water supply, this value probably eclipses that of agricultural productivity in the EAA (Table 5-3). Even the lower estimates for the value of water supply alone are higher than productivity estimates for either rice or sugarcane production. These data also demonstrate that other services, such as flood protection and water treatment, have large, positive value. The opportunity cost of agricultural production on EAA lands is probably much higher than the cost of using the land to provide water supply and flood protection for south Florida. Converting agricultural land in the EAA into above-ground water storage reservoirs is economically efficient when ecological services are included in cost-benefit considerations.

These "ecosystem services" values are more useful as conceptual comparisons, rather than empirical. Measurement of these values has become increasingly popular but contentious (Dasgupta 1991, Costanza *et al.* 1997, Masood and Garwin 1998a, 1998b, 1998c). Few studies have attempted to measure

Table 5-3. Estimated present value (interest rate 6%, infinite time period, US$1999) of opportunity costs of using 934 km^2 of EAA land for water storage versus agricultural production. Services converted from Heimlich *et al.* (1998), adjusted to 1999 dollars (using Gross Domestic Product implicit price deflator), and converted from $/acre into $/934$km^2$. Values represent lowest and highest values, and source of estimate.

	Service provided	Present value	Source (State study occurred):
Reservoirs:			
	Wastewater treatment:	$4,908,164	Farber 1996 (LA)
		$254,606,087,578	Thibodeau and Ostro 1981 (MA)
	Water quality improvement	$83,438,784	Farber 1987 (LA)
		$363,204,119	Farber 1996 (LA)
	Storm protection	$11,917,021,642	Lant and Roberts 1990 (midwestern US)
		$26,999,808,917	Lant and Roberts 1990 (midwestern US)
	Flood control	$19,220,369,336	Anderson and Rockel 1991 (MA)
		$325,082,411,199	Thibodeau and Ostro 1981 (MA)
	Water supply	$673,630,753,399	Gupta and Foster 1975 (MA)
		$986,511,469,457	Thibodeau and Ostro 1981 (MA)
Agriculture:			
	Sugarcane production	$280,457,784	ERS 1999a (FL)
		$13,967,495,528	USACE and SFWMD 1999 (FL)
	Rice production	$11,116,468	ERS 1999b (LA)
		$653,974,358	Alvarez 1993 (FL)

the value of ecosystem services in monetary amounts, and those that do often yield results that are limited to the location in which the data were taken (Masood and Garwin 1998a). Aside from the inconsistencies and inadequacies of available valuation methods, per square kilometer value estimates of wetlands often consist of several local examples that rarely account for all direct and indirect use benefits (Pearce 1993, Bartelmus 1996, Barbier *et al.* 1998). These estimates have a limited, local applicability, and may not account for price increases expected as the scarcity of wetland resources increases (Masood and Garwin 1998c, Wilson and Carpenter 1999).

OTHER MANAGEMENT OPTIONS NOT CONSIDERED

One water management option not considered by the Restudy is to decrease water demand. Human water demands could be reduced to fit within environmental constraints through policies that encourage water-conserving behavior, such as water pricing that responds to scarcity (Lynne *et al.* 1978). Currently, there is no market tool in Florida (such as water prices per unit) that is responsive to environmental conditions (Yingling *et al.* 1998). Urban and agricultural users are not encouraged to conserve water in drier seasons and years (until faced with water restrictions). Several studies have demonstrated that residential (Howe and Linaweaver 1967, Danielson 1979), agricultural (Lynne 1977, Herrington 1987), and industrial and commercial (Renzetti 1992, Yingling *et al.* 1998) water demands are responsive to price (especially high prices), and will adjust their water conservation activities accordingly. Decreased water demand could lower the total cost of water storage, regardless of whether ASR or reservoir systems are used. The Restudy does mention water conservation through the use of water flow restriction devices (such as low-flow toilets and shower heads). If these devices are used to the maximum extent possible, urban water demand could drop by almost 20% (USACE and SFWMD 1999, p. E-37).

Another way to deter increased water demand is to prevent increases in the human population in the region. There are several policies at the state and local level that encourage "smart growth" plans, that discourage sprawl of urban areas into rural or natural areas by creating "urban development boundaries" (National Audubon Society 1999). Florida's 1975 Comprehensive Planning Act spurred the Governor's Commission for a Sustainable South Florida (1995) to forge the "Eastward Ho!" development initiative for

Monroe, Dade, and Palm Beach counties. However, none of these policies focus on the absolute number of people in an area, but rather change zoning laws to allow for multifamily and high density housing units. Although potentially politically difficult to implement, zoning laws could be used to limit the number of people living in south Florida. As vacant property became scarcer, property values would increase, decreasing the demand to live in south Florida. This scenario could affect current residents unfavorably; as property values increase, property taxes would increase. The socioeconomic composition of south Florida might be affected, favoring those in the higher income brackets that could afford the higher cost of living.

Another possibility (but probably the least desirable) is to satisfy water demand through desalination of seawater, without increasing storage capacity in the system. Postel and Carpenter (1997) estimate that desalination costs on average $\$1.50/m^3$, (range is $\$1$-$\$2/m^3$, 1997 $US), so the cost of meeting an annual demand of 2,763,050,000 m^3 (2BGD) with desalination technology would be about $4,144,575,000 per year (1997$). Although the Restudy does not consider desalination, it does propose several water reclamation and reuse facilities, including two in Dade County with a combined treatment capacity of 231 million gallons per day (USACE and SFWMD 1999). Water from these facilities would be used primarily to supply Everglades National Park, Biscayne National Park, and several aquifer recharge areas (to reduce saltwater intrusion). None of the reclaimed water is targeted for human water supply.

CONCLUSION

The Restudy does not represent a thorough evaluation of all water storage alternatives under all possible land use and environmental conditions. Rather, it assumes growing human water demand can be met simply through increased storage capacity, without further damage to the Everglades ecosystem. A true ecological restoration plan would begin with the needs of the ecosystem, and then determine to what degree human needs could be met by the services that ecosystem provided. This plan would require a water budget for the Everglades, outlining rainfall, evapotranspiration, runoff to the ocean, and water flow through wetland areas, as well as seasonal and annual variations of these parameters. The decisions made in the Restudy did not arise from a complete cost-benefit analysis of all water storage options, nor does the Restudy meet the above criteria for an ecologically restorative plan.

Above-ground reservoirs may be a more efficient way of storing water in the Everglades, if all costs due to policy changes, ecosystem benefits, and treatment costs are included in water storage project analyses. Using land values based on the most economical use of the land, and reducing acquisition costs by using the Wetlands Reserve Program (or other easement programs, Aillery *et al.* 1997) would probably be the most effective way to reduce above-ground reservoir costs. Uncertainty in water treatment requirements should not be disregarded, as it could amount to a substantial cost for underground storage facilities.

Above-ground reservoirs alone cannot restore the system to its predrainage condition. Soil subsidence in the EAA has affected the hydraulic head that would have pushed water south through the agricultural fields into the natural areas (Synder and Davidson 1994, USACE and SFWMD 1999, p. N-70). Wherever these reservoirs are placed, they will likely be vulnerable to invasion and dominance by cattail (*Typha domingensis*), or exotic species such as water hyacinth (*Eichhornia crassipes*). By definition, any facility with water storage as a primary function will not “restore” the Everglades. Water storage primacy will reduce other ecosystem services that the original extent of wetland could provide.

Above-ground reservoirs in the EAA would increase the overall extent and connectivity of wetlands in south Florida. They could aid in the transition from an Everglades that is disconnected to one that is connected. Indeed, increasing "the total spatial extent of natural areas" is the primary ecological goal of the Restudy (USACE and SFWMD 1999, p. 5-21), and was a suggested fundamental goal by the Governor's Commission for a Sustainable South Florida (1995) and the U.S. Man and the Biosphere Program (1994, Harwell 1998). With an all-reservoir system, located in the EAA, south Florida would join over 140 urban areas in the United States, by considering the restoration of services provided by a watershed to protect and enhance its water supply (Chichilnisky and Heal 1998). Decreasing sugarcane subsidies, converting marginal farmland into reservoirs, and encouraging urban and agricultural water demand to respond to rainfall patterns would all help the human population in south Florida harmonize its activities with the ecosystem upon which it depends. A policy decision that can affect an entire ecosystem, such as the choice between water storage technologies, should not be made without any consideration of its environmental effects and the gains and losses of biological capital.

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VITA

Audrey Lorraine Mayer was born in Park Ridge, Illinois on November 22, 1972. She attended public school in Glenview, Illinois, and graduated from Glenbrook South High School, Glenview, Illinois in June of 1990. She began college that August at Pomona College in Claremont, California. After graduating with a Bachelor of Arts in Biology and Public Policy Analysis in June of 1994, she continued as a research associate for Dr. Bill Wirtz at Pomona College for another year. She studied the effects of fire on the abundance of California gnatcatcher (*Polioptila californica*), and other vertebrate species of the coastal sage scrub. She was also interested in policies, such as the Natural Communities Conservation Plan, designed to help reconcile endangered species management and economic development. In September of 1995, she began graduate school at the University of Tennessee, Knoxville in the Department of Ecology and Evolutionary Biology. There, she studied the ecology of the Cape Sable seaside sparrow (Ammodramus maritimus mirabilis) in Everglades National Park, focusing on the sparrow's habitat. She also examined policy issues and conservation of the Everglades ecosystem. She completed her Ph.D. in Ecology and Evolutionary Biology, with a minor in Environmental Policy, in January 2000, and was awarded her Ph.D. in May 2000. She began a post-doctoral postition at the University of Cincinnati in February 2000, studying the effects of surrounding land use on bird species in riparian habitat along the Little Miami River in Ohio.