RESEARCH ARTICLE

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Patterns of Plant Composition in Fragments of Globally Imperiled Pine Rockland Forest: Effects of Soil Type, Recent Fire Frequency, and Fragment Size

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ABSTRACT: Maintaining native plant diversity through fire management is challenging in the wildland-urban interface. In subtropical South Florida, fragments of fire-dependent, globally imperiled pine rockland forest are scattered throughout urban areas. To determine the effects of recent fire frequency, major soil type, and fragment size on species composition, we measured understory vascular plant presence and cover in 162 plots distributed among 16 publicly-owned pine rockland preserves in 1995 and 2003. Fragments received either 0, 1, or > 1 burn(s) between sampling periods. Native plant richness was very high overall. Major soil type, which varies regionally and is associated with latitude and elevation, strongly influenced the assemblage of species present at a given site. Native species cover was significantly different across different burn categories. Fragment size was positively associated with plant species richness, but small fragments had high variance in the total number of native plant species they supported, with some having nearly as many plant species as the largest fragment. Examining trends over time for rare native and invasive non-native plant species revealed the spread of the invasive grass *Rhynchelytrum repens* (Willd.) C.E. Hubb. and showed no major decreases in rare plant species. In general, this study provided encouraging results for managers of small urban forest fragments, showing that they can maintain high levels of native plant diversity, even when fire occurs infrequently.

Index terms: fire, forest fragments, pine rockland, species richness, wildland-urban interface

INTRODUCTION

As the world's forests continue to disappear, natural area managers must increasingly become experts in the "art and science" of maintaining urban forest fragments (Janzen 1988). Although the composition of such remnants differs from that of intact forests (Laurence and Bierregaard 1997), many of these scattered pieces play a vital role in conserving regional native plant richness. In fact, small fragments (< 40 ha) have been shown to contain species richness rivaling or even exceeding that of large preserves (Simberloff and Gotelli 1984; Shafer 1995; Gann et al. 2002; Pither and Kellman 2002).

Managing for native species richness in urban fragments is difficult, with a suite of unique issues spanning from social to ecological. Aside from direct destruction, societal impacts on urban forest fragments include increased influx of non-native plants (Noss and Csuti 1997) and animals (Castillo and Clarke 2003; Meshaka et al. 2004), as well as dumping of household trash (Chavez and Tynon 2000) and construction debris. Ecological issues include isolation and edge effects, which lead to an over-abundance of disturbance-adapted species and lower rates of pollination and propagule dispersal (Noss and Csuti 1997).

In pyrogenic forests, an additional effect of fragmentation is loss of the natural fire regime that is vital to maintain the system (Noss and Csuti 1997). As fire suppression becomes the norm, re-introducing fire to urban fragments poses a whole new suite of social issues (Davis 1990), while the major ecological issue becomes succession to a non-pyric community, threatening biodiversity in that system (Leach and Givnish 1996; Heuberger and Putz 2003; Varner et al. 2005). In fire-suppressed urban forest fragments, populations of rare species become extremely difficult to maintain. The "art and science" of management enters when managers must combine both species-based and process-based management (Hobbs 2007). Land managers face the conflicting goals of re-introducing fire to the landscape for the good of overall biodiversity while trying not to extirpate rare species that may be vulnerable to fire. Further complicating the issue, land management budgets are usually so woefully inadequate that money must be carefully allocated to only the most effective techniques (Laurence and Bierregaard 1997). It is, therefore, crucial that land managers adapt their restoration techniques to be as effective as possible. To this end, we present a management case study in remnants of fire-suppressed, globally critically imperiled pine rockland forest in Miami-Dade County (Florida).

It is the primary goal of Miami-Dade County land mangers to "maximize native biotic diversity" (Miami-Dade County Natural Areas Management Working Group 2004). Restoration strategies employed in the County's pine rockland preserves include controlling invasive plant species infestations and conducting regular burns. But given that prescribed fires are often unfeasible, the County's Natural Areas Management Division conducts manual hardwood reduction treatments as a surrogate for capturing some of the ecological benefits of frequent fires. This process also prepares a fragment for possible future prescribed fires by removing vegetation that is less likely to burn. Whether it is achieved through fire or through manual treatment, the target structure for pine rockland forests managed by Miami-Dade County is one in which hardwoods are reduced in stature and cover, palms occupy approximately 25% of the midstory cover, and shrub gaps contain a diverse mosaic of understory grasses and forbs (Maguire 1995). Reasons for this target vegetation structure include promoting diverse understory flora, increasing fine fuels (thereby reducing smoke output), and preventing hot burning fires that kill young pine trees (Maguire 1995).

In order to provide feedback to local land managers on the effectiveness of their restoration practices, we examined patterns in pine rockland plant diversity over an eight-year period. We looked at the effects of three environmental factors that we believed would influence plant species composition, and we examined changes in abundance and cover of both rare native plant species and non-native invasive plant species between sampling periods. Our goals in this endeavor were to: (1) elucidate some of the underlying factors that affect plant species composition, (2) determine whether fire management affects plant species richness and floristic composition within this time period, and (3) reveal any possible rare plant species losses or invasive plant species increases.

Environmental criteria we examined included major geographic region (based on edaphic factors), recent fire frequency, and fragment size. For major geographic region, we referred to the work of O'Brien (1998). In that study, he spatially defined three distinct geographic regions of Miami-Dade pine rockland forest that were previously suggested by Robertson (1955) and Snyder et al. (1990). For classification, he used major soil type, though he noted a north to south environmental gradient whereby elevation and soil characteristics were correlated with latitude. O'Brien (1998), as well as Robertson (1955) and Snyder et al. (1990) all suggested that plant community composition changes along this gradient (although this has never, to our knowledge, been quantified). We, therefore, predicted that floristic composition in this study would differ by geographic region, *sensu* O'Brien (1998). Second, because pine rocklands have been well-documented as a fire-dependent ecosystem (Robertson 1953; Wade et al. 1980; Snyder et al. 1990), we hypothesized that fragments receiving multiple fires from 1995 to 2003 would have greater native plant species richness and significantly different floristic composition than unburned or less frequently burned fragments. Third, we predicted that fragment size would be positively associated with plant species richness, per the theory of island biogeography (MacArthur and Wilson 1967). Though fragment size has been shown to be a reliable predictor of plant species richness in many different systems (e.g., Honnay et al. 1999; Gillespie 2005), it has not been supported in other studies (Robinson et al. 1992; Holt et al. 1995), and its over-use has been criticized as irrelevant for planning and managing preserves (Saunders et al. 1991).

In addition to the predictions described above, we also wanted to utilize this dataset to examine the changes in abundance and cover of both rare native plant species and non-native invasive plant species between sampling periods 1995 and 2003—something of great interest to local land managers. It has been shown that richness of native pineland understory plant species can be increased through fire management (Brockway and Lewis 1997; Sparks et al. 1998) and thinning of overstory vegetation (Maschinski et al. 2005). Additionally, it is generally accepted that biological invasions can reduce native biodiversity (Elton 2000; Simberloff 2005). Thus, if managing to maximize native biotic diversity on Miami-Dade County preserves has been successful, we expected to see decreased abundance and cover of non-native plant species, coupled with unchanged or increased abundance and cover of rare native plant species.

STUDY AREA

Pine rocklands were historically shaped by fires every two to 10 years that culled fire-intolerant trees and shrubs (Robertson 1953; Wade et al. 1980; Snyder et al. 1990). In the United States, pine rocklands are primarily located in subtropical southeast Florida, where they are distributed atop the Miami Rock Ridge. This limestone formation extends southwest from downtown Miami for approximately 60 km and then bends due West, extending 20 km into the Long Pine Key area of Everglades National Park (Figure 1). The ridge rarely exceeds 7 m in elevation. While most Florida pine rocklands are in Miami-Dade County, smaller parcels exist on geologically distinct limestone outcroppings in adjacent Collier and Monroe counties (Snyder et al. 1990). Pinelands sharing many of the same species, but dominated by *Pinus caribaea* Morelet, are found on the four northernmost islands of the Bahamas (Correll and Correll 1982; TNC 2003) and the Turks and Caicos Islands (TNC 2003). All Florida pine rocklands are characterized by an overstory of *Pinus elliottii* Engelm. var. *densa* Little & Dorman, a midstory dominated by palms and shrubs, and a diverse understory comprised of perennial grasses and herbs. The substrate is limestone with occasional shallow sand. Mean annual rainfall is 1400-1530 mm (Snyder et al. 1990). Outside Everglades National Park, Miami-Dade County pine rocklands occupy only about 920 ha, which is less than 2% of the original range (Bradley 2005). This substantial habitat loss has contributed to pine rocklands being listed as a globally critically imperiled natural community (FNAI 2006). Remaining pine rockland fragments of Miami-Dade County are extremely important for conserving the unique plant richness in South Florida. Florida pine rocklands contain 98 state listed and 16 federal listed vascular plant species (Gann et al. 2006). Furthermore, this plant community has a high degree of endemism, with 41 vascular plant taxa endemic to Florida and 25 species found only in pine rocklands of Florida (Gann et al. 2006). Most of these endemic plant

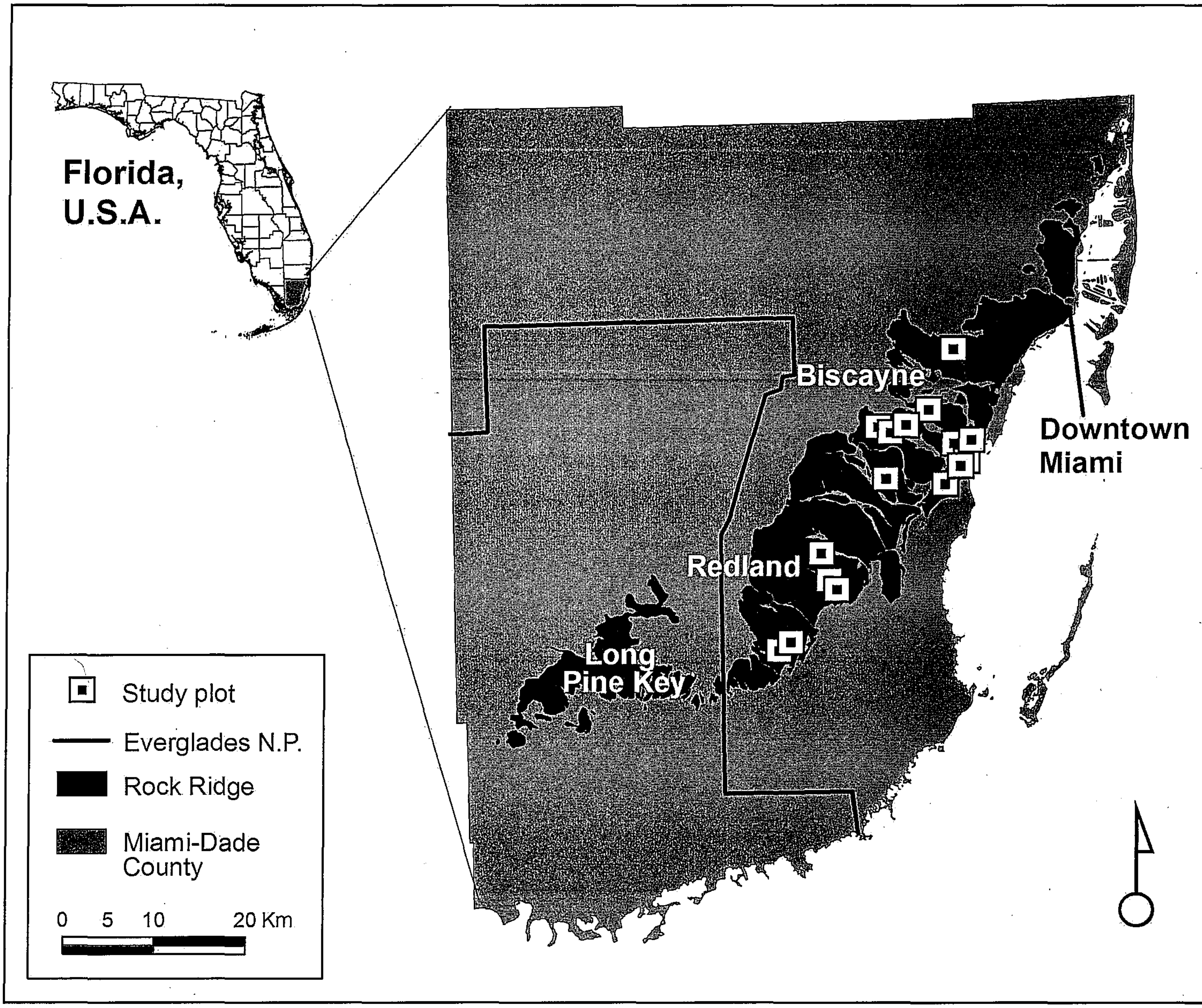

Figure 1. Location of 18 study plots in 16 of Miami-Dade County's managed pine rockland preserves. Geographic regions are labeled in white.

species require a fire return interval of less than five years to maintain their habitat (Robertson 1954).

Miami-Dade County is a matrix of roads, buildings, and agricultural fields with a human population of more than 2.4 million (U.S. Census Bureau 2004). Since its 1991 inception, the County's Natural Areas Management Division has maintained a prescribed fire program in its pine rocklands, yet weak public support has been a persistent barrier to its success. Residents of the Greater Miami area are the least educated in the state about the need for and benefits of prescribed fire (Anonymous 2004). Further management challenges are presented by the small size of pine rockland fragments, which poses acquisition, protection, and management issues. Of the 51 Miami-Dade pine rockland preserves, 45 are < 40 ha in size and 32 of those are < 10 ha.

METHODS

Sampling methods

We revisited historic plots and examined vegetation data held at Fairchild Tropical Botanic Garden to determine how major pine rockland region (*sensu* O'Brien 1998), recent fire frequency, and fragment size influence understory plant diversity. In 1994-1995, Fairchild staff installed 20-m x 40-m macroplots in each of the major pine rockland fragments of Miami-Dade County (Kernan 1994). Within each macroplot, they randomly selected three 5-m x 5-m subplots, and within each of these plots, they randomly selected three 1-m x 1-m subplots (Figure 2). They permanently marked all plots with subterranean rebars and mapped each rebar with a submeter accurate Trimble ProXR GPS unit. From March through October of 1995, Fairchild staff recorded all vascular vegetation

< 0.5 m tall in each 1-m x 1-m subplot, including trees, shrubs, vines, grasses, and herbs. They listed each species, estimating percent cover for each using an eight-class system: 0%, < 1%, 1-5%, 5-15%, 15-30%, 30-50%, 50-80%, and > 80%. They did not measure cover of non-photosynthetic vegetation, such as trunks of *Serenoa repens* (W.Bartram) Small.

From May through September 2003, we re-sampled 162 of the 1-m x 1-m subplots nested within 18 macroplots installed by Kernan (1994). While this sampling period was slightly truncated from that of 1995, it encompassed the growing season, ensuring that we were capturing all species present. Plots were distributed throughout 16 pine rockland fragments in a 42 km x 12 km area of the Miami Rock Ridge. All fragments are preserves owned and managed by Miami-Dade County. During the study period, the county thinned hardwoods and removed invasive plants from fragments regardless of plot placement.

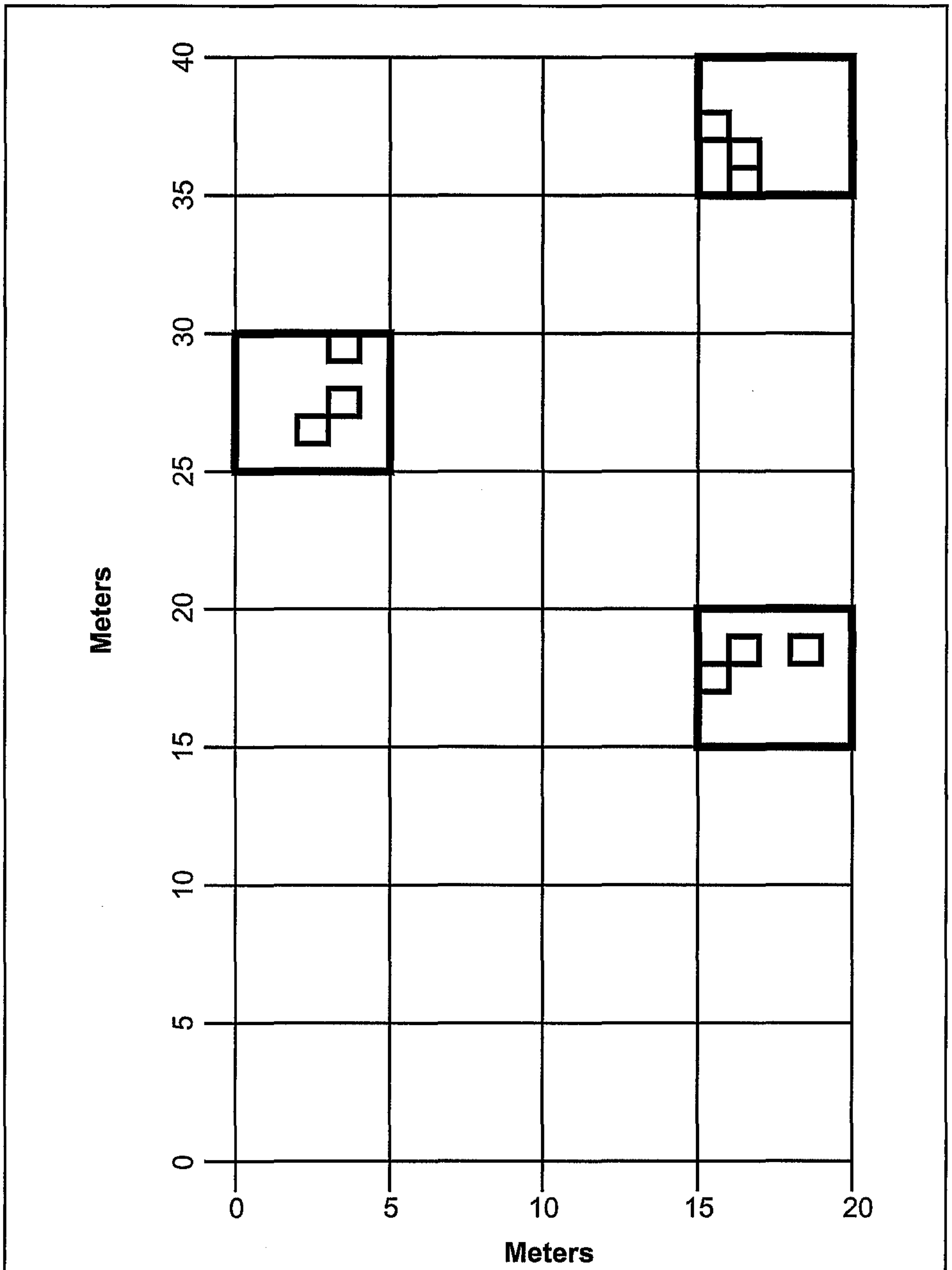

Figure 2. Schematic of original sampling design by Kernan (1994). Study plots (1 x 1 m) were nested within 5 x 5 meter plots, which were in turn nested within 20 x 40 m macroplots. This study only considered data from the 1 x 1 meter subplots.

To examine how environmental factors influenced assembly of native plant species in the pine rockland plant community, we subjected all presence/absence data for native species in 2003 to Principal Components Analysis (PCA) in PC-ORD (McCune and Mefford 2006). We assigned each study site to either the Biscayne or Redland pine rockland region, as circumscribed in O'Brien (1998). We did not collect data from the Long Pine Key region, which is located inside Everglades National Park (Figure 1). Using fire frequency data from Miami-Dade County records, we assigned macroplots to one of three categories depending on whether they received no fires, a single fire, or multiple fires between 1995 and 2003. Because all unburned plots occurred in the Biscayne region, we examined fire frequency in each region separately. In the Biscayne region, we assigned macroplots to three categories: five sites had no fires, three sites had a single fire, and five sites had multiple fires. In the Redland region, we compared three macroplots that received one fire to two macroplots that had multiple fires (Table 1). We sampled from two macroplots at Pineshore Pineland and Larry & Penny Thompson Park (in both cases, one unburned plot and one single-burn plot), because each represented a recent fire history that was underrepresented in the total dataset (Table 1). Burns included both controlled burns and wildfires.

We defined species richness as the number of species per sampling unit (McCune and Grace 2002). Taxonomy generally followed Wunderlin (1998). We conducted analyses of variance (ANOVA; SYSTAT Software 2002) to determine whether species richness was significantly different between sampling periods and whether major pine rockland region and recent fire history influenced species richness.

To determine whether community assemblage within the two regions predictably changed with fire frequency, we used both presence/absence and coverage data. First, we performed a factor analysis to reduce

Table 1. Eighteen sampling plots in Miami-Dade County preserves. Means are followed by standard errors.

Site Name	Fragment size (ha)	# burns 1995-2003	Months since last burn	Mean native plant richness 1995	Mean native plant richness 2003
Biscayne Region					
Larry & Penny ThompsonA	93	1	53	13.1 ± 0.7	18.9 ± 1.0
Larry & Penny ThompsonB	93	0	> 96	14.9 ± 0.8	19.7 ± 0.8
Nixon Smiley	48.5	> 1	52	8.8 ± 0.6	13.9 ± 1.1
Deering Estate South Addition	13.5	1	21	6.7 ± 0.9	9.2 ± 0.9
Tamiami Complex Addition	10.5	0	> 96	14.0 ± 0.9	18.8 ± 1.7
Bill Sadowski	8.5	0	> 96	13.2 ± 1.2	16.6 ± 1.1
Ludlam	4	> 1	8	13.3 ± 0.6	19.2 ± 0.9
Ned Glenn	4	> 1	40	15.6 ± 0.9	22.0 ± 0.7
Ron Ehman	3	0	> 96	12.3 ± 2.1	15.3 ± 1.5
Pineshore PinelandA	2.5	0	> 96	12.6 ± 0.9	10.7 ± 1.4
Pineshore PinelandB	2.5	1	27	15.8 ± 1.9	15.7 ± 0.8
Coral Reef	2	> 1	64	15.8 ± 0.8	17.8 ± 0.9
Tropical	2	> 1	70	5.0 ± 0.4	9.8 ± 0.7
Redland Region					
Navy Wells	143	1	27	18.1 ± 2.0	26.9 ± 1.9
Camp Owaissa Bauer	40	1	26	12.3 ± 0.7	14.8 ± 0.5
Sunny Palms	16.5	> 1	13	5.8 ± 0.3	13.4 ± 0.6
Seminole Wayside	6	> 1	1.5	16.6 ± 0.7	17.9 ± 1.1
Ingram	4	1	90 (est.)	8.8 ± 1.2	14.2 ± 1.3

the number of species present in the study plots and improve precision of classification analyses. Using species' coverages represented by medians calculated from percent cover class of each species present in a study plot, we selected variables within each region with component loadings > 0.3 in the first two axes to enter into the Stepwise Discriminant Analysis (SDA). We report the final reduced model that best defined the classification of plots by fire frequency for each region.

We used linear regression to examine the relationship between fragment size and native understory richness in 2003 (SYSTAT Software 2002). As suggested by Cook et al. (2002), we omitted non-native species from this analysis in favor of species native to South Florida pine rockland, so that species from the matrix would not obscure patterns in native species richness.

To examine trends over time in the presence of rare plant species and non-native, invasive plant species, we first needed to define the terms "rare" and "non-native invasive." In cases where we discuss rare species, we define these as native plant species listed as endangered by the state of Florida (Coile and Garland 2003). For non-native invasive plant species, we used those classified as "Category I" by the Florida Exotic Pest Plant Council. This classification indicates that the species is altering native plant communities (FLEPPC 2007). Significance tests for changes in most important non-native invasive plant species and rare plant species were generated using the paired t-test function in SYSTAT. All means we report include notation of standard error.

RESULTS

Native plant species

Study plots had a total of 182 native vascular plant species in 1995, with average species richness per 1-m x 1-m plot ranging from 5.0 $\pm$ 0.4 to 18.1 $\pm$ 2.0. In 2003, we recorded 187 native species, with average species richness ranging from 9.2 $\pm$ 0.1 to 26.9 $\pm$ 1.9 (Table 1). Comparing plant species lists from 1995 and 2003, there was a 68% overlap, as indicated in the Appendix. Per plot native plant richness changed significantly between sampling years, increasing by an average of 4.5 species in each plot (ANOVA, $F_{(1, 177)}$ = 100.10, p < 0.001).

Major pine rockland region, which was primarily differentiated by soil type, had a strong influence on the assemblage of native plant species present in study plots (PCA, Figure 3). The seven species that most distinguished major region along the first axis were all found primarily or exclusively in the Redland region: *Koanophyllon villosum* (Sw.) King & H.Rob., *Guettarda scabra* (L.) Vent., *Galium hispidulum* Michx., *Pteridium aquilinum* (L.) Kuhn var. *caudatum* (L.) Sadebeck, *Ardisia escallonioides* Schiede & Deppe ex Schltdl. & Cham., *Toxicodendron radicans* (L.) Kuntze, and *Forestiera segregata* (Jacq.) Krug & Urb. In the Biscayne region, *Euphorbia polyphylla* Engelm. ex Chapm. and *Dyschoriste angusta* (A. Gray) Small were most important for distinguishing region, but they were less important than the seven Redland species. Although region affected native plant species assemblage, it did not significantly influence overall native plant species richness (ANOVA, $F_{(1, 160)} = 2.56$, $p = 0.111$).

Recent fire frequency had less influence than region on the assemblage of native plant species present in study plots. Plots receiving zero, one, or multiple burn(s) did not form distinct clusters in plant species space when only presence/absence was considered (PCA, data not shown). In the Redland region, native plant species richness was not significantly different among recent fire frequencies (ANOVA, $F_{(1,43)} = 1.273$, $p = 0.266$). However, recent fire frequency significantly influenced native plant species richness in Biscayne plots (ANOVA, $F_{(2,114)} = 7.444$, $p = 0.001$). Contrary to expectations, a post-hoc analysis using Tukey's HSD multiple comparison test showed that plots experiencing a single burn over the study period had significantly lower native plant species richness than unburned ($p = 0.001$) and multi-burn ($p = 0.006$) plots.

While presence/absence data showed little effect of recent burn history, Stepwise Discriminant Analysis using coverage data revealed that native plant species cover was significantly different in study plots across different burn categories. For plots in the Biscayne region, native plant species presence and coverage in single burn

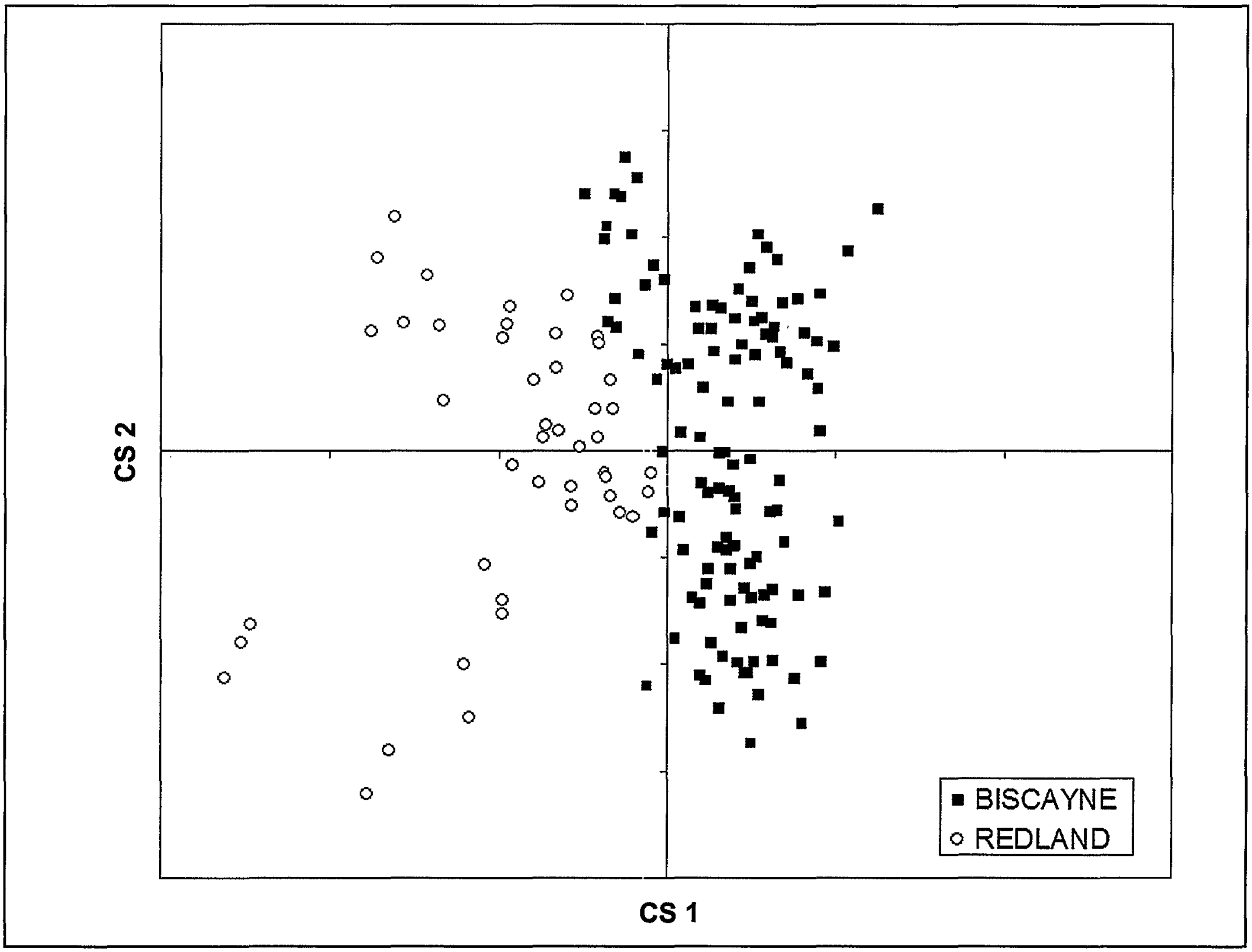

Figure 3. Principal Components Analysis of study plots in plant species space. Study plots (1 x 1 m) are separated by major pine rockland region as described in O'Brien (1998).

plots overlapped with that of the other two burn categories. Yet, differences were much more apparent when comparing unburned plots with those receiving multiple burns (SDA, $F_{(36, 194)} = 3.80$, $P < 0.001$, Figure 4). A total of 72% of Biscayne region plots were correctly grouped by the jackknife classification. Breaking this down by burn class, 60% of unburned plots were classified correctly, as were 78% of single burn plots and 80% of multiple burn plots. Plots in the Redland region also showed significantly different floristic composition between burn categories (SDA, $F_{(20, 24)} = 7.15$, $P < 0.001$). We could not generate a scatter plot of canonical scores for these plots because discriminant analysis yielded a single discriminant function axis. Overall, 82% of Redland plots were correctly grouped by the jackknife classification, with 78% of single burn plots and 89% of multiple burn plots correctly classified. In total, SDA used 36 species to classify plots by recent fire frequency, with 18 species used in the Biscayne region (of 30 total) and 20 in the Redland region (of 47 total) (Table 2). All species considered for inclusion in the models are indicated in the Appendix at the end of this manuscript.

Fragment size had a positive influence on native plant species richness in understory plots, explaining 32% of the variation ($r^2 = 0.32$, $p = 0.014$, Figure 5). However, there was a wide range in native plant species richness among the smallest preserves.

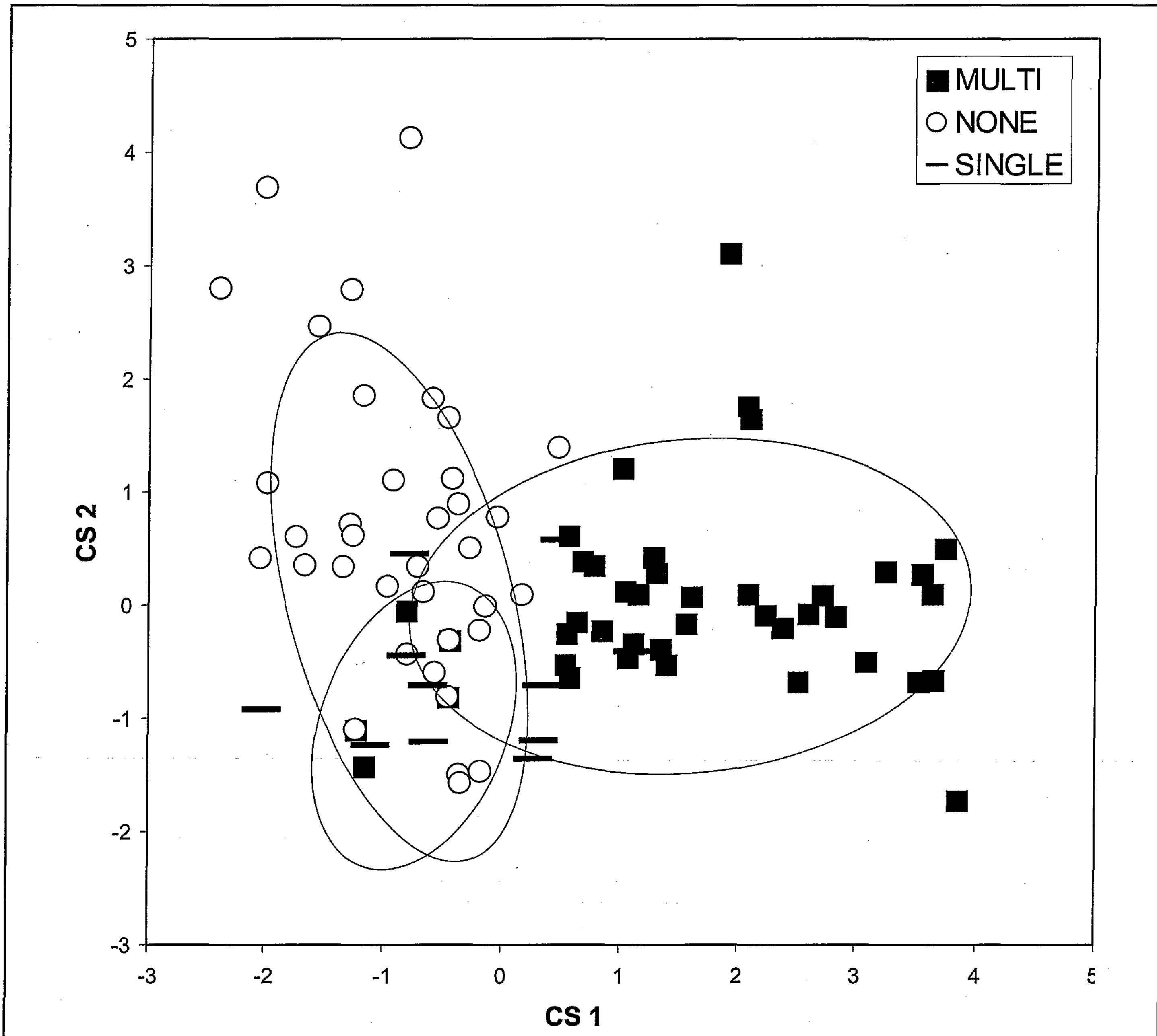

Figure 4. Canonical scores from Stepwise Discriminant Analysis of plant species' coverage in the Biscayne region, classified by recent fire frequency (zero, single or multiple burn(s) within the past eight years). Ellipses are centroids plus confidence intervals.

Table 2. Species used in model to classify study plots into one of three categories for recent fire frequency: no burns, one burn, or multiple burns. Species are sorted according to the burn category in which they were most prevalent. In the Biscayne region, there were no species with greatest mean coverage in single burn plots. Asterisks indicate non-native taxa.

Biscayne Region	Redland Region
Greatest mean coverage in unburned plots	**Greatest mean coverage in single burn plots**
Aeschynomene viscidula	*Ardisia escallanioides*
Angadenia berteroi	*Aster adnatus*
Croton glandulosus	*Aster concolor*
Paspalum monostachyum	*Ayenia euphrasiifolia*
Polygala grandiflora	*Chamaecrista deeringiana*
*Spermacoce verticillata**	*Chiococca parvifolia*
	Cnidoscolus stimulosus
Greatest mean coverage in multiple burn plots	*Galactia volubilis*
	Guettarda scabra
Chiococca parvifolia	*Schizachyrium sanguineum*
Cynanchum blodgettii	
Desmodium incanum	**Greatest mean coverage in multiple burn plots**
Dyschoriste angusta	
Elionurus tripsacoides	*Abildgaardia ovata*
Evolvulus sericeus	*Angadenia berteroi*
Nephrolepis biserrata	*Galactia smallii*
Parthenocissus quinquenervia	*Galium hispidulum*
Piriqueta caroliniana	*Koanophyllon villosum*
Ruellia succulenta	*Macroptilium lathyroides**
Schizachyrium rhizomatum	*Pityopsis graminifolia*
	Poinsettia pinetorum
Equal coverage in both plot types	*Pteridium aquilinum* var. *caudatum*
Acalypha chamaedrifolia	*Pteris bahamensis*

Presence and cover of rare native plant species in managed plots increased over the sampling period in many cases, but this change was significant for only one species, federally endangered *Galactia smallii* H.J. Rogers ex Herndon (Table 3). Study plots contained 14 Florida endangered plant species. From 1995 to 2003, only three of these 14 rare species decreased in number of plot occurrences. No plant species were lost from the study plots over this period; in fact, four previously undocumented rare species were recorded. Unfortunately, the dataset was not large enough to support analyses on the effects of fire frequency or fragment size on rare plant species abundance or cover.

Non-native invasive plant species

Non-native plant species were not a major component of vegetative cover in this study. Plots at Navy Wells had the highest mean non-native plant species cover at 3.1%. The majority of non-native cover at Navy Wells was comprised of *Schinus terebinthifolius* Raddi. For both sampling periods combined, all study plots contained a total of just 24 non-native plant species, many of which were not widely distributed. In fact, 70% (in 2003) to 72% (in 1995) of all plots did not contain any non-native plant species. In examining only those plots containing non-native plant species, average cover fell from 4.7% in 1995 to 1.8% in 2003, but this trend was not statistically significant (ANOVA, $F_{(1,93)} = 3.76$, $p = 0.06$). Of all non-native plant species, the most prevalent were *Schinus terebinthifolius*, *Neyraudia reynaudiana* (Kunth) Keng ex A.S. Hitchc., and *Rhynchelytrum repens* (Willd.) C.E. Hubb. (Table 4).

In comparing occurrences of most invasive non-native plant species between sampling periods, we found a general trend in which *Ardisia elliptica* Thunb., *Neyraudia reynaudiana*, and *Schinus terebinthifolius* were less abundant over time. This effect was not statistically significant for any of these species using paired t-tests (Table 4). The opposite was the case for *Rhynchelytrum repens*. This species was absent from all plots in 1995, but was present in 23 plots in 2003, with a significant increase in mean cover in those plots by 1.2% ($p = 0.01$). All but one of the 23 plots containing *R. repens* had at least one burn during the study period.

DISCUSSION

Native plant species

Native plant species richness is very high in Miami-Dade County's fragmented pine rockland preserves. The documentation of 182 and 187 native taxa in our 162 study plots (totaling 0.016 ha) is high compared to one study in Everglades pine rocklands, where DeCoster et al. (1999) found a maximum of 128 species in a 0.1-ha plot. While overall native plant richness in our plots did not change greatly between sampling periods, native plant richness on a per-plot basis significantly increased. Several factors may account for this. Natural Areas Management practices that commenced in 1991, such as removal of non-native invasive plant species and native hardwoods as well as prescribed burning, were likely to have favored the biologically rich pine rockland understory. In addition, observer influence could explain part or

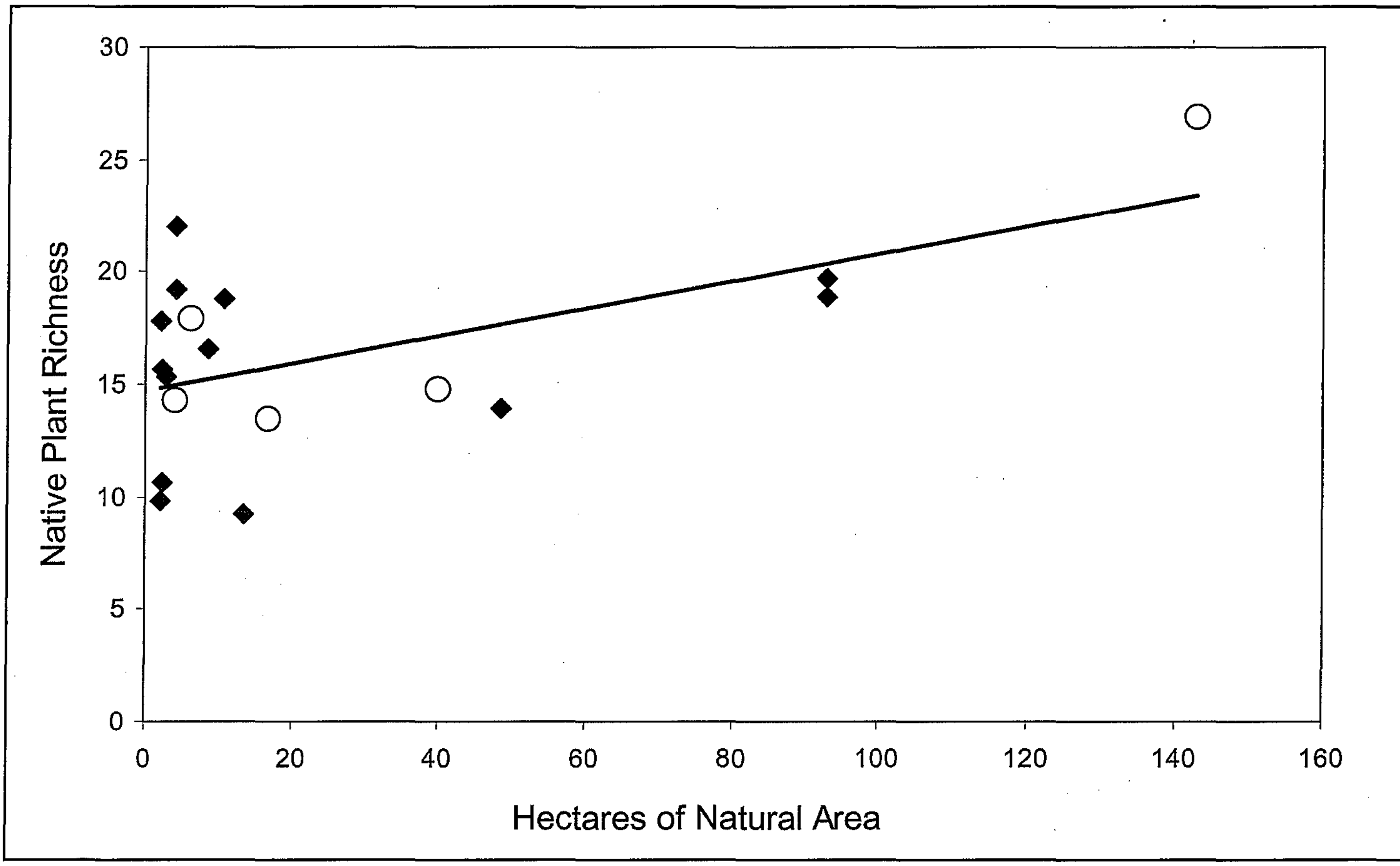

Figure 5. Regression of fragment size compared to native plant richness within 1 x 1 m study plots (r^2 = 0.32, P = 0.014). Mean native plant richness for each fragment (N = 18) is presented here. Solid diamonds represent sites in the Biscayne region and hollow circles represent sites in the Redland region.

all of the increase in native plant species richness. Assistant data collectors changed over time, and while the lead observer (Woodmansee) remained the same in both 1995 and 2003, he continued to build on his knowledge of plant taxonomy in the eight-year interim.

This work lends quantitative support to previous suggestions (Robertson 1955; Snyder et al. 1990; O'Brien 1998) that species composition is distinctly different between the Biscayne and Redland regions of the Miami Rock Ridge. In showing these differences, we underscore both the importance of considering edaphic factors in regional studies of species composition as well as the need to preserve fragments of different edaphic types in order to maximize native biodiversity.

Although our results suggest that a single burn will reduce native pine rockland plant richness on Biscayne soils, we assert that these results are most likely an artifact of the unusually low number of species at the Deering South Addition (Table 1), the limited time span of our study, and the low number of single burn plots we were able to sample from the Biscayne region (just 27, compared to 45 for both unburned and multi-burn plots). Deering South Addition plots are depauperate of both native and non-native plant species, with a mean of 10.1±0.9 total species (compared to 17.4±0.4 species in all other plots combined). In addition, when we removed Deering South Addition plots from the analysis, recent fire frequency no longer significantly affected native plant species richness (ANOVA, $F_{(2,105)}$ = 1.720, p = 0.184). Most likely, the low diversity at Deering South Addition is because the area was unmanaged for years and had begun to succeed to a closed-canopy hammock with few understory species. Repeated manual reduction of hardwoods by Miami-Dade County (in 1995, 1997, 1999, 2002, and most intensively in 2003) as well as a prescribed burn in 2001 has not yet promoted recovery of the diverse pine rockland understory. Overall, we believe that significant change in pine rockland plant species richness occurs over a longer time span than the length of this study, but we are not able to prove this with our existing dataset.

In contrast to the slow response time of plant species richness, it is interesting to note that even in the relatively short eight-year span of this study, the number of fires received by study plots affected floristic composition. Certain plant species appeared to be much more affected by recent fire frequency than others (Table 2). In both the Biscayne and the Redland regions, the majority of plant species used in the discriminant analysis function are found in pine rockland forests that have very sparse canopy and shrub layers permitting high herbaceous diversity. Those plant species that had the greatest mean coverage in unburned plots are mostly limited to small native herbs and grasses. Exceptions to this

Table 3. Presence and cover of fourteen Florida endangered plant species found in study plots. Asterisks indicate species that are also listed as endangered by the U.S. Endangered Species Act. Columns headed by "# plots" show the number of plots that contained each species.

	1995		2003		Difference		
	# plots	Avg. % cover	# plots	Avg. % cover	# plots	Avg. % cover	*p*- value in paired t-test
Alvaradoa amorphoides	0	0	1	0.5	1	0.5	N/A
Argythamnia blodgettii	3	1.33	5	1.25	2	-0.08	0.827
Bourreria cassinifolia	1	0.5	1	0.5	0	0	N/A
Chamaesyce deltoidea ssp. *adhaerens* *	0	0	1	3	1	3	N/A
Chamaesyce deltoidea *	7	1.57	11	1.41	4	-0.16	0.87
Chamaesyce deltoidea ssp. *pinetorum*	3	0.5	5	0.5	2	0	0.516
Chamaesyce porteriana	0	0	1	0.5	1	0.5	N/A
Galactia smallii *	5	0.5	7	0.5	2	0	0.001
Ipomoea tenuissima	6	0.92	2	0.5	-4	-0.42	0.185
Koanophyllon villosum	12	3.58	21	1.33	9	-2.25	0.615
Lantana depressa	4	4.13	1	3	-3	-1.13	0.239
Poinsettia pinetorum	3	0.5	10	0.75	7	0.25	0.067
Scutellaria havanensis	3	0.5	2	0.5	-1	0	0.638
Trema lamarckianum	0	0	1	3	1	3	N/A

included the sometimes aggressive native ferns *Nephrolepis biserrata* (Sw.) Schott and *Pteridium aquilinum* var. *caudatum*, native vine *Parthenocissus quinquefolia* (L.) Planch., native shrub *Koanophyllon villosum*, and non-native sub-shrub *Macroptilium lathyroides* (L.) Urb. With the exception of *K. villosum*, the authors have noted that each of these species can be quick to colonize disturbed areas.

Although there was a positive correlation between fragment size and native plant species richness, this relationship might have been stronger if we had data on mid-sized fragments. We lack these data because there are virtually no mid-sized pine rockland preserves in Miami-Dade County. Close to 95% of pine rockland preserves are < 40 ha in size. All remaining preserves are ≥ 80 ha, except for one newly acquired 54-ha unit that contains some pine rockland. As a whole, small fragments had wide variance in the total number of plant species they supported. It was striking that many of the smallest preserves in our study (< 15 ha) had levels of plant species richness that approached or exceeded those of plots in larger preserves. This highlights the importance of conserving even small frag-

Table 4. Presence and cover of the four non-native plant species found in study plots that are classified as "Category I" by the Florida Exotic Pest Plant Council (2007). Columns headed by "# plots" show the number of plots that contained that species in 1995 or 2003. Paired t-tests were conducted to test for significant differences in percent coverage of each taxon in 1995 versus 2003.

	1995		2003		Difference		
	# plots	Avg. % cover	# plots	Avg. % cover	# plots	Avg. % cover	*p*- value in paired t-test
Ardisia elliptica	3	14.2	1	3	-2	-11.2	0.053
Neyraudia reynaudiana	7	3.9	5	1	-2	-2.9	0.143
Rhynchelytrum repens	0	0	23	1.2	23	1.2	0.01
Schinus terebinthifolius	24	6.1	5	5.4	-19	-0.7	0.111

ments and indicates that preserve size is one of the factors influencing plant species richness, along with soil type, hydrology, fire history, and disturbance.

Over the study period, the significant increase in cover of federally endangered *Galactia smallii* as well as the increased occurrences of 11 other rare plant species suggests pine rockland preserves in Miami-Dade County are being managed in a positive way supporting floristic diversity. This study was not designed to detect rare plant species or track them over time; thus, we have insufficient data to explain directly why rare plant species presence and cover changed or did not change over time. Monitoring and research efforts that include focusing on specific taxa, tagging individual plants, and mapping with GPS and GIS technology would be more effective for detecting the response of rare plant species to land management activities. Nevertheless, data gathered during this larger study suggest many rare plant species are thriving in Miami-Dade County pine rockland preserves, and active management can prevent rare species losses.

Non-native invasive plant species

Ongoing invasive plant species programs in Miami-Dade County preserves most likely contributed to the fact that overall non-native plant species were not a significant component of plant cover in study plots. The decline in abundance of the invasive non-native species *Schinus terebinthifolius, Neyraudia reynaudiana,* and *Ardisia elliptica* Thunb. from 1995 to 2003 can be attributed to active invasive species management. These three species are all removed regularly when funds permit. An exception to the trend of non-native plant cover decreasing from 1995 to 2003 was the observed increase of the short-lived perennial non-native grass *Rhynchelytrum repens.* The sharp increase in *R. repens* occurrences since 1995 is a major management concern, especially considering that *R. repens* responds positively to fire. It is difficult to treat because it often grows interspersed with native grasses and herbs, and it has recently been shown to displace native grass species in pine rocklands (Possley and Maschinski 2006).

Conclusions

At the local scale, this study elucidates some of the factors influencing species assemblage and suggests directional trends for cover of both rare native species and non-native invasive species in managed preserves. Region and corresponding edaphic factors strongly influenced the assemblage of native species present in study plots. To a lesser degree, recent fire history also influenced native species assemblage. We showed that significant loss of native plant diversity did not occur during the eight-year time scale of this study. However, increase in occurrences of the invasive grass *Rhynchelytrum repens* should cause alarm for South Florida land managers. At the broader scale, this work demonstrates the ecological value that exists in urban fragments, even when they are small and fire-suppressed, emphasizing the importance of acquisition, preservation, and restoration of these parcels.

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Appendix. All vascular species recorded in study plots, 1995 and 2003. Column 1 indicates which species were used in the stepwise discriminant analysis (SDA) model, with species used from Biscayne region plots indicated with a 'B' and those used from Redland region plots indicated with an 'R'. Column 2 indicates which year(s) the taxon appeared in study plots. Non-native species are underlined.

R	BOTH	*Abildgaardia ovata* (Burm.f.) Kral
	1995	<u>*Acacia auriculiformis*</u> A. Cunn. ex Benth.
BR	BOTH	*Acalypha chamaedrifolia* (Lam.) Mull.Arg.
B	2003	*Aeschynomene viscidula* Michx.
	BOTH	*Agalinis fasciculata* (Elliott) Raf.
	BOTH	<u>*Albizia lebbeck*</u> (L.) Benth.
	BOTH	*Alvaradoa amorphoides* Liebm.
	2003	<u>*Alysicarpus vaginalis*</u> (L.) DC.
	BOTH	*Ambrosia artemisiifolia* L.
	1995	*Ampelopsis arborea* (L.) Koehne
B	2003	*Andropogon glomeratus* (Walt.) B.S.P. var. *hirsutior* (Hack.) C. Mohr
R	BOTH	*Andropogon glomeratus* (Walt.) B.S.P. var. *pumilus* Vasey ex Dewey
	1995	*Andropogon gyrans* Ashe
	BOTH	*Andropogon ternarius* Michx.
	2003	*Andropogon tracyi* Nash
	BOTH	*Andropogon virginicus* L.
	2005	*Andropogon virginicus* L. var. *decipiens* C. Campbell
	BOTH	*Anemia adiantifolia* (L.) Sw.
BR	BOTH	*Angadenia berteroi* (A.DC.) Miers
	BOTH	<u>*Ardisia elliptica*</u> Thunb.
R	BOTH	*Ardisia escallonioides* Schiede & Deppe ex Schltdl. & Cham.
	BOTH	*Argythamnia blodgettii* (Torr.) Chapm.
	2003	*Aristida beyrichiana* Trin. & Rupr.
B	BOTH	*Aristida purpurascens* Poir.
	2005	*Asclepias tuberosa* L.
R	BOTH	*Aster adnatus* Nutt.
	1995	*Aster bracei* Britton ex Small
R	BOTH	*Aster concolor* L.
B	2003	*Aster dumosus* L.
R	BOTH	*Ayenia euphrasiifolia* Griseb.
	2003	*Baccharis halimifolia* L.
	1995	*Berlandiera subacaulis* (Nutt.) Nutt.
	BOTH	*Bidens alba* (L.) DC. var. *radiata* (Schultz-Bip) Ballard ex T.E. Melchert
	1995	*Bletia purpurea* (Lam.) DC.
	BOTH	*Bourreria cassinifolia* (A.Rich.) Griseb.
	1995	*Brickellia mosieri* (Small) Shinners
	2003	*Buchnera americana* L.
B	BOTH	*Bulbostylis ciliatifolia* (Elliott) Fernald
	2003	*Bursera simaruba* (L.) Sarg.
	BOTH	*Byrsonima lucida* (P. Mill.) DC.
	BOTH	*Callicarpa americana* L.
R	BOTH	*Cassytha filiformis* L.
	1995	<u>*Casuarina equisetifolia*</u> L.
	1995	*Cenchrus gracillimus* Nash
R	BOTH	*Centrosema virginianum* (L.) Benth.
R	BOTH	*Chamaecrista deeringiana* Small & Pennell
	BOTH	*Chamaesyce deltoidea* (Engelm. ex Chapm.) Small subsp. *adhaerens* (Small) A. Herndon

(continued)

Appendix. Continued.

	BOTH	*Chamaesyce deltoidea* (Engelm. ex Chapm.) Small
R	BOTH	*Chamaesyce deltoidea* (Engelm. ex Chapm.) Small subsp. *pinetorum* (Small) A. Herndon
	BOTH	*Chamaecrista fasciculata* (Michx.) Greene
	BOTH	*Chamaesyce hirta* (L.) Millsp.
	2003	*Chamaesyce hypericifolia* (L.) Millsp.
	1995	*Chamaesyce hyssopifolia* (L.) Small
	BOTH	*Chamaecrista nictitans* L. var. *aspera* (Muhl. ex Elliott) H.S. Irwin & Barneby
	2003	*Chamaesyce porteriana* Small
	BOTH	*Chaptalia albicans* (Sw.) Vent. Ex Steud.
	BOTH	*Chiococca alba* (L.) A.S. Hitchc.
BR	BOTH	*Chiococca parvifolia* Wullschl. ex Griseb.
	2003	*Chromolaena odorata* (L.) King & H.Rob.
	2003	*Cirsium horridulum* Michx.
B	BOTH	*Clematis baldwinii* Torr. & Gray
R	BOTH	*Cnidoscolus stimulosus* (Michx.) Engelm. & Gray
	BOTH	*Coccothrinax argentata* (Jacq.) L.H. Bailey
	1995	*Coccoloba diversifolia* Jacq.
	BOTH	*Commelina diffusa* Burm.f.
	BOTH	*Commelina erecta* L.
	BOTH	*Conyza canadensis* (L.) Cronquist var. *pusilla* (Nutt.) Cronquist
	BOTH	*Crossopetalum ilicifolium* (Poir.) Kuntze
B	BOTH	*Croton glandulosus* L.
	BOTH	*Croton linearis* Jacq.
	BOTH	*Crotalaria pumila* Ortega
	BOTH	*Crotalaria rotundifolia* Walt. ex J.F. Gmel.
B	BOTH	*Cynanchum blodgettii* (A. Gray) Shinners
	1995	*Cyperus filiculmis* Vahl
	BOTH	*Dalea carnea (Michx.)* Poir.
B	BOTH	*Desmodium incanum* DC.
	BOTH	*Desmodium marilandicum* (L.) DC.
	1995	*Desmodium tortuosum* (Sw.) DC.
	BOTH	*Desmodium triflorum* (L.) DC.
R	BOTH	*Dichanthelium aciculare* (Desv. & Poir.) Gould & C.A. Clark
	2003	*Dichanthelium commutatum* (Schult.) Gould
	BOTH	*Dichanthelium ensifolium* (Baldwin ex Elliott) Gould var. *unciphyllum* (Trin.) B.F.Hansen & Wunderlin
	2003	*Dichanthelium erectifolium* (Nash) Gould & C.A.Clark
	BOTH	*Dichanthelium ovale* (Elliott) Gould & C.A. Clark
	2003	*Dichanthelium strigosum* (Muhl. ex Ell.) Freckmann var. *glabrescens* (Griseb.) Freckmann
	1995	*Digitaria filiformis* (L.) Koeler var. *dolichophylla* (Henrad) Wipff
B	BOTH	*Dyschoriste angusta* (A. Gray) Small
	BOTH	*Echites umbellata* Jacq.
B	BOTH	*Elionurus tripsacoides* Humb. & Bonpl. ex Willd.
	2003	*Emilia fosbergii* D.H. Nicols.
B	BOTH	*Eragrostis elliottii* S. Wats.
	1995	*Erechtites hieracifolia* (L.) Raf. ex DC.
	2003	*Eremochloa ophiuroides* (Munro) Hack.
	BOTH	*Ernodea cokeri* Britton ex Coker
	1995	*Eupatorium mikanioides* Chapm.
	2003	*Eupatorium mohrii* Greene
	1995	*Eupatorium serotinum* Michx.
	BOTH	*Euphorbia polyphylla* Engelm. ex Chapm.
	BOTH	*Eustachys petraea* (Sw.) Desv.
BR	BOTH	*Evolvulus sericeus* Sw.
	1995	*Exothea paniculata* (Juss.) Radlk. Ex T. Durand
	1995	*Ficus altissima* Blume
	BOTH	*Ficus aurea* Nutt.
	BOTH	*Ficus citrifolia* P. Mill.
	BOTH	*Forestiera segregata* (Jacq.) Krug & Urb.
	BOTH	*Galactia floridana* Torr. & Gray
	BOTH	*Galactia pinetorum* Small
R	BOTH	*Galactia smallii* H.J. Rogers ex Herndon
R	BOTH	*Galactia volubilis* (L.) Britton
R	BOTH	*Galium hispidulum* Michx.

(continued)

Appendix. Continued.

	1995	*Guapira discolor* (Spreng.) Little
	BOTH	*Guettarda elliptica* Sw.
R	BOTH	*Guettarda scabra* (L.) Vent.
R	BOTH	*Hedyotis nigricans* (Lam.) Fosberg var. *floridana* (Standl.) Wunderlin
	2003	*Hedyotis uniflora* (L.) Lam.
	BOTH	*Heliotropium polyphyllum* Lehm.
	BOTH	*Hieracium megacephalon* Nash
	1995	*Hypoxis sessilis* L.
R	BOTH	*Hyptis alata* (Raf.) Shinners
	BOTH	*Ilex krugiana* Loes.
R	BOTH	*Imperata brasiliensis* Trin.
	1995	*Indigofera spicata* Forsk.
	1995	*Ipomoea alba* L.
B	BOTH	*Ipomoea indica* (Burm.f.) Merr. var. *acuminata* (Vahl) Fosberg
	BOTH	*Ipomoea tenuissima* Choisy
	BOTH	*Jacquemontia curtisii* Peter ex Small
R	BOTH	*Koanophyllon villosum* (Sw.) King & H.Rob.
	1995	*Lantana camara* L.
R	BOTH	*Lantana depressa* Small
	BOTH	*Lantana involucrata* L.
	1995	*Lechea torreyi* (Chapm.) Legg. ex Britton
	BOTH	*Liatris chapmanii* Torr. & Gray
	2003	*Liatris gracilis* Pursh
R	BOTH	*Liatris tenuifolia* Nutt.
R	BOTH	*Licania michauxii* Prance
	BOTH	*Lyonia fruticosa* (Michx.) G.S. Torr.
	BOTH	*Lysiloma latisiliquum* (L.) Benth.
R	BOTH	*Macroptilium lathyroides* (L.) Urb.
	BOTH	*Melanthera parvifolia* Small
	BOTH	*Metopium toxiferum* (L.) Krug & Urb.
	BOTH	*Mimosa quadrivalvis* L. var. *angustata* (Torr. & Gray) Barneby
	1995	*Mitreola sessilifolia* (J.F. Gmel.) G. Don
	1995	*Momordica charantia* L.
	BOTH	*Morinda royoc* L.
	BOTH	*Muhlenbergia capillaris* (Lam.) Trin.
	BOTH	*Myrica cerifera* L.
B	BOTH	*Nephrolepis biserrata* (Sw.) Schott
	2003	*Nephrolepis exaltata* (L.) Schott
	2003	*Neptunia pubescens* Benth.
	BOTH	*Neyraudia reynaudiana* (Kunth) Keng ex A.S. Hitchc.
	BOTH	*Opuntia humifusa* (Raf.) Raf.
BR	BOTH	*Parthenocissus quinquefolia* (L.) Planch.
R	BOTH	*Paspalum blodgettii* Chapm.
	2003	*Paspalum caespitosum* Flugge
B	BOTH	*Paspalum monostachyum* Vasey
	BOTH	*Paspalum setaceum* Michx.
	1995	*Passiflora foetida* L.
	BOTH	*Passiflora suberosa* L.
	BOTH	*Pectis glaucescens* (Cass.) D.J. Keil
	2003	*Phlebodium aureum* (L.) J. Sm.
B	BOTH	*Phyllanthus pentaphyllus* C. Wright ex Griseb. var. *floridanus* G.L.Webster
R	BOTH	*Physalis walteri* Nutt.
	BOTH	*Piloblephis rigida* (W.Bartram ex Benth.) Raf.
	BOTH	*Pinus elliottii* Engelm. var. *densa* Little & Dorman
BR	BOTH	*Piriqueta caroliniana* (Walt.) Urb.
R	BOTH	*Pityopsis graminifolia* (Michx.) Nutt.
	BOTH	*Pluchea rosea* Godfrey
	BOTH	*Poinsettia cyathophora* (Murr.) Klotsch & Garcke
	1995	*Poinsettia heterophylla* (L.) Klotsch & Garcke ex Klotzsch
R	BOTH	*Poinsettia pinetorum* Small
	1995	*Polygala boykinii* Nutt.
BR	BOTH	*Polygala grandiflora* Walt.
	1995	*Polygala smallii* R.R. Sm. & D.B. Ward
	1995	*Psidium longipes* (O. Berg) McVaugh
B	BOTH	*Psilotum nudum* (L.) P.Beauv.
	BOTH	*Psychotria nervosa* Sw.
R	BOTH	*Pteridium aquilinum* (L.) Kuhn var. *caudatum* (L.) Sadebeck

(continued)

Appendix. Continued.

R	BOTH	*Pteris bahamensis* (J. Agardh) Fée
B	2003	*Pterocaulon pycnostachyum* (Michx.) Ell.
	1995	*Pteris vittata* L.
	BOTH	*Quercus pumila* Walt.
	BOTH	*Quercus virginiana* P. Mill.
	BOTH	*Randia aculeata* L.
	BOTH	*Rapanea punctata* (Lam.) Lundell
	BOTH	*Rhus copallinum* L.
	BOTH	*Rhynchospora colorata* (L.) H.Pfeiff.
	BOTH	*Rhynchospora floridensis* (Britton) H. Pfeiff.
	BOTH	*Rhynchospora grayi* Kunth
	1995	*Rhynchospora intermedia* (Chapm) Britton
	BOTH	*Rhynchosia michauxii* Vail
	BOTH	*Rhynchosia minima* (L.) DC.
R	BOTH	*Rhynchosia reniformis* DC.
	BOTH	*Rhynchelytrum repens* (Willd.) C.E. Hubbard
	BOTH	*Richardia grandiflora* (Cham. & Schltdl.) Scult. & J.H. Schult.
BR	BOTH	*Ruellia succulenta* Small
	BOTH	*Sabal palmetto* (Walt.) Lodd. ex J.A. & J.H. Schultes
	2003	*Sachsia polycephala* Griseb.
	2003	*Samolus ebracteatus* Kunth
	2003	*Schefflera actinophylla* (Endl.) Harms
	BOTH	*Schizachyrium gracile* (Spreng.) Nash
B	BOTH	*Schizachyrium rhizomatum* (Swallen) Gould
R	BOTH	*Schizachyrium sanguineum* (Retz.) Alston
R	BOTH	*Schinus terebinthifolius* Raddi
	BOTH	*Scleria ciliata* Michx.
R	BOTH	*Scutellaria havanensis* Jacq.
R	BOTH	*Senna mexicana* (Jacq.) H.S.Irwin & Barneby var. *chapmanii* (Isley) H.S.Irwin & Barneby
	1995	*Senna obtusifolia* (L.) H.S. Irwin & Barneby
	2003	*Senna pendula* (Humb. & Bonpl. ex Willd.) H.S. Irwin & Barneby var. *glabrata* (Vogel) H.S. Irwin & Barneby
	BOTH	*Serenoa repens* (W.Bartram) Small
	2003	*Setaria parviflora* (Poir.) Kerguelen
	2003	*Sida acuta* Burm.f.
	BOTH	*Sida elliottii* Torr. & Gray
	BOTH	*Sideroxylon salicifolium* (L.) Lam.
	BOTH	*Sisyrinchium nashii* Bickn.
	BOTH	*Smilax auriculata* Walt.
	BOTH	*Smilax havanensis* Jacq.
	BOTH	*Solidago odora* Aiton var. *chapmanii* (Gray) Cronquist
B	BOTH	*Solidago stricta* Aiton
	BOTH	*Sorghastrum secundum* (Elliott) Nash
	BOTH	*Spermacoce assurgens* Ruiz & Pavon
	BOTH	*Spermacoce terminalis* (Small) Kartesz & Gandhi
B	2003	*Spermacoce verticillata* L.
	1995	*Sporobolus junceus* (P. Beauv.) Kunth
	2003	*Stenotaphrum secundatum* (Walt.) Kuntze
BR	BOTH	*Stillingia sylvatica* L.
	1995	*Swietenia mahagoni* (L.) Jacq.
	BOTH	*Tephrosia florida* (F. Dietr.) C.E. Wood
	BOTH	*Tetrazygia bicolor* (P. Mill.) Cogn.
	BOTH	*Toxicodendron radicans* (L.) Kuntze
	BOTH	*Tragia saxicola* Small
	BOTH	*Tragia urens* L.
	2003	*Trema lamarckianum* (Schult.) Blume
	BOTH	*Trema micranthum* (L.) Blume
	BOTH	*Trichostema dichotomum* L.
R	BOTH	*Tripsacum floridanum* Porter ex Vasey
	BOTH	*Triumfetta semitriloba* Jacq.
	BOTH	*Vaccinium myrsinites* Lam.
R	BOTH	*Vernonia blodgettii* Small
	BOTH	*Vitis rotundifolia* Michx.
	1995	*Waltheria indica* L.
R	BOTH	*Zamia integrifolia* Aiton