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Effects of Prescribed Fire on Upland Plant Biodiversity and Abundance in Northeast Florida

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Effects of prescribed fire on upland plant biodiversity and abundance in Northeast Florida

by

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CERTIFICATE OF APPROVAL

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Abstract

Terrestrial ecosystems in the southeastern United States have evolved with fire as a common disturbance and as a result many natural communities require the presence of fire to persist over time. Human development precludes natural fires from occurring within these communities; however, prescribed fire is considered to be a critical tool in the effort to restore fire-dependent ecosystems after decades of fire exclusion. Direct effects of fire on individual floral and faunal species as well as benefits to biodiversity at the landscape (gamma diversity) level have largely been supported in previous research. However, information on the effects of natural and prescribed fire on plant diversity at the local level (alpha diversity) is limited, particularly for southeastern forests. The applicability of the Intermediate Disturbance Hypothesis (IDH), which suggests that the highest levels of biodiversity are found at intermediate levels of disturbance, is also untested for North Florida upland plant communities.

This study compared the effects of fire on local scale mean plant species diversity by examining burned and unburned portions of three fire-dependent communities to determine if there is an effect of prescribed fire on in alpha biodiversity. Alpha biodiversity was not significantly different ($p=0.433$) between burned and unburned fire-dependent plant communities in northern Florida, suggesting that prescribed fire does not affect plant species diversity in these communities and/or the IDH for plant communities is not supported at the time scale tested. However, the application of prescribed fire did result in changes in abundance of species, particularly with species such as *Dicanthelium acuminatum*, *Quercus myrtifolia*, and *Vaccinium myrsinites*, that respond positively to fire, which may have implications for associated faunal diversity.

1. Introduction

1.1 Pyrohistory

The majority of terrestrial ecosystems found in the southeastern United States have evolved with fire and coexisted with fire over the past 10,000 to 12,000 years, resulting in many natural communities becoming fire-dependent; that is, these communities require the presence of fire in order to persist over time (Goldammer, 1993; Frost, 1998; Nowacki and Abrams, 2008). Plant species found within these communities often exhibit structural or reproductive adaptations, such as thick bark, serotiny, or winged seeds that promote survival in a fire-dominated environment (Keeley and Fotheringham, 2000; Bond and Keeley, 2005). The frequency of fire recurrence varies, from as often as 1 - 2 years in some communities and as long as 20 - 30 years in others (Frost, 1998; Kirkman et al., 2001). In natural ecosystems, lightning is the primary ignition source (Ruth et al., 2007; Waldrop et al., 2012).

In addition to naturally occurring fire, anthropogenic sources of fire have influenced the development of plant communities. Native Americans frequently conducted broadcast burning activities across the continent for hunting and agricultural purposes (Fowler and Konopik, 2007), likely resulting in increased beta (diversity between habitats) and gamma (diversity of habitats within a landscape) diversity in some areas (Delcourt and Delcourt, 1997; Frost, 1998; Bowman, 2009; USGS, 2010). Upon their arrival, Europeans likewise adopted the practices of the aboriginals to improve game forage and management of vegetative growth near settlements (Nowacki and Abrams, 2008; Waldrop et al., 2012). During the twentieth century, however, a nationwide policy of aggressive fire suppression became prominent and effectively removed fire

from the landscape, negatively impacting dependent ecosystems (Keane et al., 2002; Stephens and Ruth, 2005; Nowacki and Abrams, 2008).

By the 1950s and 1960s, it was recognized that fire exclusion was detrimental to many fire-dependent communities and associated species, and the active use of fire as part of land management programs began to emerge. These planned burns are termed “prescribed fire” (Pyne 2004; Stephens and Ruth, 2005; Waldrop et al., 2012). It is worth noting that the use of fire has historically been more culturally acceptable in the southeastern United States, with the first prescribed fire allowed on federal lands occurring in the Osceola National Forest in Florida in 1943 (Stephens and Ruth, 2005). By the 1980s, prescribed fire had become an important tool in restoring fire-adapted ecosystems for land management agencies nationwide and an important consideration for the protection and maintenance of biodiversity (Mutch et al., 1993; Beckage et al., 2005; Van Lear et al., 2005).

While the direct effects of fire on individual floral and faunal species and plant communities have been well studied and its benefits to biodiversity at the landscape (gamma diversity) level largely supported (Romme 1982; Baker, 1992; Keane et al., 2002), information on the effects of natural and prescribed fire on plant diversity at the local level are (alpha diversity) is limited, particularly for southeastern forests. Definitions of biodiversity are wide ranging, but in its simplest form it is the number and/or abundance of species in a given area or region. High biodiversity is considered to be important for the stability of ecosystems and communities because having greater numbers of species tends to result in greater resiliency to environmental and anthropogenic perturbations (Loreau, 2000; Stiling, 2002). To date, the

results of prescribed burns on biodiversity have been mixed, with some studies showing decreased species richness due to fire exclusion while others indicate no difference in species richness between burned and unburned communities (Pastro et al., 2011; Penman et al., 2011) or even positive effects from reduced fire frequency for certain functional groups (e.g., woody plants) (Peterson and Reich, 2008). Fire impacts on biodiversity are generally considered at the landscape level and are based upon creating heterogeneity between habitats (gamma biodiversity) that result from variation in temporal, spatial, intensity and frequency of fire patterns across the landscape (Romme, 1982; Sieg, 1995; Keane et al., 2002). Martin and Sapsis (1992) contend it is the temporal and spatial variability of fire regimes across the landscape that creates the most diverse complexes of species.

1.2 Intermediate Disturbance Hypothesis

In 1978, Joseph H. Connell proposed an alternative view as to why high levels of species diversity are locally observed in tropical coral reefs and rain forests. The previously held assumption was that high levels of diversity are maintained at equilibrium; that is, species assemblages are where they should be in the absence of disturbance. When disturbance does occur, species assemblages move to return to that point of equilibrium. Connell, in an effort to address the question of the mechanism of this process, observed that the frequency of natural disturbance is often much faster than the rate of species recovery. If this is so, then high species diversity is not a result of maintaining equilibrium, but instead another mechanism must be operating. Connell examined a number of hypotheses that had been proposed to explain the

diversity of species observed in tropical systems, some in support of equilibrium theory and some in opposition.

One of the non-equilibrium hypotheses he explored was the “intermediate disturbance” hypothesis (IDH). This hypothesis proposes that the greatest diversity of species in a community does not occur in the absence of disturbance or with constant disturbance, but instead at intermediate levels of disturbance (frequency and/or intensity) (Figure 1). At the heart of the hypothesis is successional theory. At high levels of disturbance, diversity is low because the time for colonization of early seral stages is short, so only species that can colonize rapidly will be able to occupy the newly disturbed areas. As the frequency between disturbance episodes decreases, additional species are able to occupy the community and diversity increases. If the interval between disturbances becomes too infrequent, however, the community is able to move towards the climax successional stage where the most fit competitor species occupy the available niches and diversity decreases. As a result, the highest species diversity is found at some intermediate point between the two extremes.

Whether or not the IDH has general uniform applicability for all communities has been debated. Connell’s original work cited several examples from research in tropical forests in Uganda and Nigeria and coral reefs off the coast of Queensland, Australia, all of which exhibit the highest species diversity at intermediate levels of disturbance. More recent research in forested landscapes and with fire has produced mixed results. One large-scale test of the hypothesis was initiated in Guiana by Molino and Sabatier (2001), which involved a tropical forest that had previously been disturbed as a result of forestry operations, including thinning for

fuels reduction, girdling and harvesting. Species diversity was measured in control plots (undisturbed) and plots exhibiting varying levels of disturbance ten years post treatment. Highest levels of diversity were found within plots associated with intermediate levels of disturbance, consistent with the IDH and Connell's findings for tropical forests. Bongers et al. (2009), utilizing a large dataset of 2,504 one-acre plots, examined disturbance and functional groups in wet and dry tropical forests. As might be expected, colonizing species responded positively to disturbance and shade-tolerant species that are more common in climax communities decreased with increased disturbance. Overall patterns indicated that maximum species diversity occurred at intermediate disturbance levels. However, patterns supporting the IDH were more strongly associated with dry tropical forests and much less so in wetter forests.

Peterson and Reich (2008) looked at the relationship between disturbance and vegetative species richness in several types of communities, including grasslands, savannas, woodlands and forests, using fire frequency as a measure of disturbance. Their research site was located within the Cedar Creek Natural History Area (CCNHA), a 2,300-ha reserve located on the Anoka Sand Plain in east-central Minnesota, which represents a transition zone between tallgrass prairie and temperate deciduous forest ecosystems. The use of prescribed fire in the reserve began in the 1960s for the purposes of ecosystem restoration and to test the effects of fire frequency on habitat structure and species composition. Fire management units are excluded from livestock grazing; however, natural herbivores influence plant community composition and nutrient flux within habitat patches, a potentially compounding factor. The authors measured vegetation parameters such as tree canopy cover and functional class species richness on 26 plots, with each

plot consisting of four parallel 50 m long sampling transects placed 25 m apart in savanna, woodland, and forest stands. At 10 m intervals, understory vegetation sampling points were established along each transect. The authors identified a total of 190 vascular plant species, including 38 grass and sedge species, 39 shrubs and trees, and 113 forbs. Approximately 50% of the species recorded were rare (<10 observations). They found that overall, plants species richness reached its peak at biennial fire return intervals. In the communities studied, biennial fire corresponds to an intermediate disturbance regime and thus supports the IDH. Hiura (1995) evaluated species diversity in relation to gap formation in Japanese beech forests. The author found that locations experiencing an intermediate frequency of disturbance (measured as windstorm interval) exhibited the highest species diversity. At a larger geographic scale, a similar pattern was observed and related to both windstorm interval and temperature.

Though a substantial body of research supports the general applicability of the IDH, other studies do not. Beckage and Stout (2000) examined the effects of the frequency of burning in Florida sandhill communities to examine the relationship between plant species richness and diversity, stem density and burn frequency. Sandhill plant communities require frequent fire (1 – 3 year fire return interval) to maintain the composition of the community which represents an intermediate level of disturbance. To test for relationships among the dependent variables, the authors collected weekly measurements for over a year on six sites with varied burn histories (burned from one to six times) and compared the results to models of community response (linear, quadratic, saturating and null). Although the authors did not find a relationship between

species richness, diversity or flowering stem density and fire frequency, thus this study did not find support for the IDH, their statistical power was low due to small sample size.

As previously mentioned, Peterson and Reich (2008) found that the response of vegetation to fire as a mechanism of disturbance in a number of communities they studied supported the predictions of the IDH. This result was only part of the picture, though. Overall, understory plant species richness was greatest at intermediate fire frequency (approximately five fires per decade), which was consistent with the IDH as mentioned earlier, but these results varied between the different functional vegetation groups. The authors found that overstory tree species richness was highest on plots without fire and decreased as fire frequency increased. For woody plants, species richness was highest in plots protected with low fire frequency, but forb species richness increased with fire frequency to an intermediate point and then decreased again. Species richness for grasses was greatest with high fire frequency and decreased with decreasing fire frequency. The authors suggested that although in general the results of the study corresponded with the predictions of the IDH, they should also be consistent with assumptions related to competitive interaction theory. When disturbance (frequency of fire) is low, overstory trees and woody understory plants exhibit competitive dominance and suppress grasses and forbs. However, when disturbance is high, grasses and forbs can rapidly colonize and dominate. Peterson and Reich (2008) observed intermediate levels of disturbance created spatial heterogeneity in plant resource availability and therefore supported high species richness; this suggests that fire frequency effects varies with environmental factors, such as soil fertility, topography, and species pools, which influence vegetation productivity and competitive

interactions. Their research also illustrates that whether or not a community conforms to the predictions of the IDH depends upon the scale of study (in this case, habitat patch versus landscape level).

In another study, Schwilk et al. (1997) measured species richness in highly diverse African shrublands termed "fynbos", where fire regulates community structure. They compared diversity to three different fire regimes over varied spatial scales ranging from 1 m^2 to 0.1 ha and found that species diversity was highest in sites where fire was infrequent (approximately 40 years between fires) and lowest in intermediate and high fire frequency sites. Species diversity was also highest in the largest study plots, which indicates that scale is important and should be taken into account as part of the methodology. Regardless, Schwilk et al. (1997) did not find IDH to be applicable in their study system.

1.3 Purpose and Objectives

Research regarding the specific, direct effects of fire on individual floral and faunal species and plant communities is well documented. However, analysis of the effects of fire, including prescribed fire, on plant diversity is surprisingly small, particularly for southeastern forests of the United States. The goal of this study was to determine if alpha biodiversity differs between burned and unburned plant communities that have evolved with fire (fire-dependent communities) in northern Florida and thereby help land managers in developing restoration plans to help meet biodiversity goals. To this end, the study compared burned and unburned portions of three fire-dependent plant communities to determine if alpha biodiversity (local scale mean

species diversity) differs as a result of the application of prescribed fire. A second goal was to determine if fire, as a mechanism of disturbance, follows the predictions of the IDH.

2. Environmental Setting

The study area was located within Pumpkin Hill Creek Preserve State Park (PHCPSP), a 1577-ha state park located in Duval County, Florida (Figure 2). The property was originally acquired in 1994 as a State of Florida buffer preserve and managed by the Florida Department of Environmental Protection, Office of Coastal and Aquatic Managed Areas. Management of the preserve was transferred to the Division of Recreation and Parks in 2003. The Division emphasizes restoring and maintaining, to the degree practicable, the natural processes that shape the structure, function and species composition of Florida's diverse natural communities as they occurred in the original domain (DRP, 2006). PHCPSP contains twenty-one distinct natural communities that are managed for the purposes of natural community restoration, utilizing both timber management and placing a high priority on the use of prescribed fire to restore fire-adapted natural communities.

2.1 Climate

Northeast Florida is described as a subtropical climate zone with humid hot summers and mild winters. The average daily minimum temperature in January ranges from approximately 5.6° – 7.2°C, while the average daily maximum temperature in July is greater than 32.8°C. Annual precipitation ranges from 122 – 132 cm, arriving primarily during the summer months from May through September. During this period, weather is typically warm and moist with frequent thundershower activity in the afternoon and early evening. This pattern is driven by a

semi-permanent high pressure system located off the Atlantic coast called the Bermuda High which draws moisture northward or westward from the Atlantic and Gulf of Mexico. In addition to these generalized weather patterns, the region occasionally experiences major tropical storms and associated storm surges (Ingram et al., 2013).

2.2 Topography

PHCPSP is located west of Pumpkin Hill Creek, a tidal creek within the Timucuan Ecological and Historic Preserve. Elevation within the park ranges from approximately 3 m above mean sea level in depression marshes and basin swamps in the flatwoods to 9 m above mean sea level near the western edge of the preserve. Topography has been modified in some areas by raised sections of natural surface roads, Cedar Point Road (a paved public roadway that passes through the southern part of the preserve), and fire plow lines from previous wildfire containment efforts. These anthropogenic features impound drainages and channel runoff during storm events, altering natural hydrology (DRP, 2006).

2.3 Geology and Soils

Pumpkin Hill Creek Preserve is located in the St. Marys Meander Plain physiographic region, which consists of marine limestone overlain by Pleistocene marine terrace materials consisting of sand and clay of Appalachian origin. Within the park boundary, Holocene and Pleistocene deposits of unconsolidated sand, clay, shell and limestone range from 3 to 30 meters thick and overlay the Hawthorn Group, Ocala Limestone and the Avon Park Formation (Hoenstine, 1984).

Within the Pumpkin Hill Creek Preserve there are fourteen different soils that influence the development of its plant communities (Figure 3). The majority of soils in the study area are Spodosols, which include a typically subsurface illuvial horizon containing amorphous organic material and aluminum. Spodic soils are the most common soil order found in Florida and are characteristic of flatwoods. Within this order, Leon fine sand 0-2% slope is the dominant (62%) soil type, which is poorly drained and has a depth to water surface of approximately 15 – 46 cm (Soil Survey Staff, 2014).

2.4 Natural Communities

Twenty-one different natural communities, as defined by the Florida Natural Areas Inventory (FNAI) are found within the park boundary (Figure 4), three of which were evaluated during this study: sandhill, mesic flatwoods, and scrubby flatwoods (FNAI, 2010). A summary description of these three communities follows (FNAI, 2010):

- Sandhill is comprised of elevated areas with mesic or xeric woodlands or shrublands. The overstory canopy is open and is dominated by longleaf pine (*Pinus palustris*) or a mixture of pine and deciduous hardwoods. Turkey oak (*Quercus laevis*) and wiregrass (*Aristida berychiana*) comprise the understory. Other common species may include bluejack oak (*Q. incana*), scrub oak (*Q. inopina*) and ground lichens (*Cladonia* spp.), often in areas where fire is absent or infrequent. Natural fire frequency is estimated to be frequent, with a 1 - 3 year return interval (FNAI, 2010). Patches of sandhills are located

in several areas of Pumpkin Hill Creek Preserve bordered by mesic flatwoods and frequently blend with adjacent scrubby flatwoods (DRP, 2006).

- Mesic Flatwoods are the dominant community at PHCPSP. It is characterized by a sandy substrate and an overstory of widely spaced pines (longleaf pine or slash pine) and an understory of shrubs and herbaceous plants, including saw palmetto (*Serona repens*), gallberry (*Ilex glabra*), dwarf live oak (*Q. minima*) and wiregrass (*A. berychiana*). The fire return interval for mesic flatwoods is 2 - 4 years (FNAI, 2010) and as a result, fuel loading is very high in the unburned portions of this community.
- Scrubby Flatwoods consist of flatlands with sand substrate and a widely scattered longleaf pine overstory and scrubby understory that includes saw palmetto (*S. repens*), myrtle oak (*Q. myrtifolia*), sand live oak (*Q. geminata*), Chapman's oak (*Q. chapmanii*) and coastal plain staggerbush (*Lyonia fruticosa*). The average fire return interval for a scrubby flatwoods community is 5 - 15 years (FNAI, 2010).

2.5 Hydrology

Most of the park is located within the St. Johns River watershed, though the northeastern portion of the park is part of the Nassau River basin, and lies adjacent to the salt marshes and tidal creeks of the Nassau River-St. Johns River Marshes Aquatic Preserve. Surface waters that influence the development of plant communities within the park consist of tidal creeks and marshes (including Clapboard Creek, which drains to the St. Johns River), blackwater streams, and freshwater marshes and swamps. Surrounding urban development is of low- to moderate-density and the hydrologic resources of the park tend to exhibit good water quality. However,

increasing development within the watersheds will likely lead to increased nutrient and bacteria levels in surface and groundwater, which could influence the development of plant communities (DRP, 2006).

2.6 Land Management

Prior to state ownership, the land was used for several resource purposes that influence the current community composition. During the late 1800s and early 1900s, the site was used for the extraction of turpentine from the pines. As the turpentine industry dwindled, the site was used to produce timber, which removed many of the longleaf pines from the property. Restoring the park to its natural domain became a priority management objective upon acquisition by the State of Florida (DRP, 2006).

To meet that objective, the Division of Recreation and Parks placed a high priority on burning fire-dependent natural communities within the natural fire return interval for those communities, with the explicit intent of maintaining ecological diversity. Fire management units have been established based upon fuel type, plant community type and control line locations (Figure 5). Most management units consist of a dominant habitat type based upon Florida Natural Area Inventory (FNAI) designation, but often include other classified communities as well (DRP, 2006).

3. Data Collection and Analysis Methods

3.1 Treatment Area Determination

The three FNAI upland habitats evaluated during this study (sandhill, mesic flatwoods and scrubby flatwoods) were selected because they are fire-dependent, relatively common

throughout Florida, and the preserve includes both burned and unburned portions of each community located within the study area. The most recent FNAI community designations were used to define each community because it provides a consistent standard for community typing that is applied across the state by many different land management agencies. Additionally, it classifies communities based not only vegetation present, but on other common site characteristics as well, including soil type, hydrologic regimes, and fire frequency which will allow for a meaningful comparison between similar areas.

3.2 Plot Stratification and Transect Location

Determination of fire disturbance was based upon whether or not the unit has experienced fire (either wildfire or prescribed fire) at least once within the natural fire return interval as defined for the respective FNAI community type (Figure 6). The starting points of vegetation transects (see below) were randomly selected using ESRI's ArcMap Spatial Analyst extension to generate random points within appropriate combinations of GIS layers of FNAI vegetation, management units and burn history obtained from the Florida Park Service (Figure 7). A minimum of three transects per treatment type were established, totaling a minimum of 18 transects (three habitat types each in burned and unburned management zones). In a few instances, transect locations that were identified during the GIS analysis as one community type were determined to be a different community type in the field. Rather than discard data collected at those transects, they were reclassified with the appropriate community type and an additional transect was randomly generated to maintain a minimum of three transects per treatment type.

3.3 Field Data Collection

Vegetation species composition data were collected using randomly established 25-m line transects (Sutherland 1996). The starting point of each transect was identified in the field with a handheld GPS receiver. All transects were oriented along a north-south axis, with the start of the transect beginning at the south end. If a significant obstacle was encountered (e.g., management road) that would not appropriately represent the treatment plot, the transect was run in a due south direction, which only occurred in a single instance with one transect (PHC-17). Transects were installed by extending a measuring tape reel from the starting point northward to the 25-m mark. Each plant that intersected a transect was recorded to the species level, if possible, and to the genus level at a minimum (Figure 8). Voucher specimens of plants that were unidentifiable in the field were collected within the same treatment area, but outside of the transect, and identified using dichotomous and polyclave keys (Nelson, 1996; USDA, 2008). If more than one branch of a shrub or tree intersected the transect, they were traced back to the main stem and the individual was scored as a single observation. Plants identified to the species level were designated with a code consisting of the first two letters of the genus and the first two letters of the species name following the USDA PLANTS database (USDA, 2014). Plants that could only be identified to the genus level were designated with a code consisting of the first two letters of the genus and two digits (e.g., *Hypericum* species would be HY01, another would be HY02, etc.).

3.4 Ethical and Bias Considerations

A research permit was required from the Florida Park Service, which outlined the purpose of the research, proposed methodology and data to be collected. The permit was reviewed by park and district biologists to ensure that resources were not significantly impacted, that the methodology is sound and the research would benefit park management. Removal or destruction of natural elements of the park is generally prohibited. As part of the permit, some minor collection of voucher samples were allowed in order to properly identify plants to the species level that were in question.

The timing of controlled burns can influence plant communities, since heat, wind and humidity have a direct effect on fire behavior and intensity and these abiotic factors vary seasonally. This is a source of potential bias in this study as season of burn was not considered. However, prescribed fire implementation occurs within specific prescriptive parameters (prescriptions), which does provide some control on bias because the fire behavior parameters and lighting techniques will be similar regardless of time of year.

Two indices of alpha diversity were used to compare treatment plots, Simpson's and Shannon-Weiner. The first, Simpson's (D), represents the probability that any two individuals selected at random from an infinitely large community will belong to different species. It is a dominance index and as such, it is weighted toward the abundance of the most common species; that is, the most common species influence index values more than rare species do. In contrast, the Shannon-Wiener index (H´) is an information-statistic index that incorporates measures of species richness and eveness (a measure of the relative abundance of the different species

making up the richness of an area), biasing it more towards the presence of rare species (Stiling, 2002). Analyzing both indices therefore provides a means to assess the broader species diversity within the study area while also potentially detecting the occurrence of less common species that may occupy communities as a result of the use or exclusion of fire (Rossi et al. 2010).

Values for Simpson's D were calculated for each transect using the reciprocal of the following formula:

$D_S = \sum_{i=1}^{n} \frac{n_i(n_i - 1)}{N(N - 1)}$, where S is the number of species present, n is the total number of individuals in each species and N is the total number of individuals in all species.

Values for computing the Shannon-Wiener index for each transect were obtained using the formula below:

$H' = -\sum_{i=1}^{n} p_i \ln p_i$, where p_i is the proportion of total sample represented by species i.

3.5 Data Analysis

The IDH predicts that species richness is highest at intermediate levels of disturbance within communities (Connell, 1978). Since natural fire regimes represent intermediate levels of disturbance in fire-dependent ecosystems, it was expected that portions of the communities in this study that have been burned through the use of prescribed fire would exhibit a higher level of biodiversity versus similar areas that have not been burned within the natural fire return interval. To test this hypothesis, a multivariate analysis of variance (MANOVA) compared Shannon-Weiner index and the Simpson's D index diversity values between burned and

unburned portions of similar plant communities to test for global significances (Gotelli and Ellison, 2004). Using species and species abundance as dependent variables, a two-way MANOVA was used to test for a burn, habitat, and burn x habitat effects on both diversity index values. The same analysis was conducted for vegetation species observed at each transect by creating a table grid of transects with associated habitat and burn status, and counts of each individual species observed. When data violated assumptions of normality or homogeneity, logarithmic, square root, etc. transformations were applied. Univariate analyses (ANOVA) were then conducted to determine which species were causing the significant effect.

A post hoc power analysis was also included in the analysis of diversity and species abundance data to determine if the amount of data collected was statistically adequate (generally, power value > 0.8) (Gotelli and Ellison 2004). Power (β) is the probability of not making a Type II error, or failure to reject a false null hypothesis, with power values less than 0.8 generally indicate that the null hypothesis should potentially be rejected if no significant interaction effect is found during statistical analysis. The power analysis was not used to definitively determine whether or not the null hypotheses should be accepted or rejected, but rather to help guide the analysis of the data in this study.

For all analyses, assumptions of normality and sphericity were assessed and verified by observing the shape of histogram distributions for each covariate (diversity index value, vegetation species), Kolmogorov-Smirnov statistic values, Mauchly's sphericity test, and examination of skewness and kurtosis values. Characteristic bell-shaped histogram distributions and non-significant Kolmogorov-Smirnov statistic values indicated normality. Assumptions of

homogeneity were confirmed by inspection of plotted residuals versus fitted values and a Levene's test of equal variances. Overall significance tests utilized the most commonly used Wilks Lambda multivariate test statistic. All statistical analyses were made using the statistical software package SPSS (version 20). Non-metric multidimensional scaling (NMDS) is a common ordination technique used in ecological applications to visualize the relationships between variables based on a rank order distance matrix. NMDS plots were created with species abundance data for each plot generated from the output from a Bray-Curtis dissimilarity index using Past version 3.04.

4. Results

4.1 Multivariate Analysis of Diversity Indices

A total of 23 vegetation transects were measured in 2013 (Table 1). Shannon-Weiner (H') diversity values from community type and treatment data ranged from a low of 1.366 in unburned mesic flatwoods to a high of 2.458 in unburned sandhill. Simpson's (D) diversity values from community type and treatment data ranged from a high of 0.378 in unburned mesic flatwoods to a low of 0.096 in unburned sandhill. For both indices, there was no significant difference ($p<0.05$) in measures of alpha diversity between habitat types, burn treatment or the interaction of habitat and burn treatment (Tables 2 and 3, Figures 9 and 10).

4.2 Multivariate Analysis of Vegetation Species Interactions

Numbers of species documented in each community ranged from 20 species in mesic flatwoods to 27 in unburned sandhill, based on pooled data for each treatment type. Scrubby flatwood, mesic flatwood, and burned sandhill community types were dominated by a few

species, while plant abundance in unburned sandhill was more evenly distributed. The most abundant vegetation in unburned scrubby flatwoods were *Gaylussacia dumosa* (28%), *Quercus myrtifolia* (13%) and *Serenoa repens* (9%). The most abundant plant species in burned scrubby flatwoods included *Q. myrtifolia* (26%), *Vaccinium myrsinites* (12%) and *S. repens* (10%). The most abundant plants in unburned mesic flatwoods were *Lyonia lucida* (37%), *G. dumosa* (12%), *Ilex glabra* (11%) and *S. repens* (11%). For burned mesic flatwood communities, abundant vegetation included *L. lucida* (20%), *I. glabra* (17%) and *Dichanthelium acuminatum* (10%). Abundant plants in burned sandhill included *Q. myrtifolia* (35%), *V. myrsinites* (11%) and *Aristida beyrichiana* (10%). The most abundant species found in unburned sandhill were *V. myrsinites* (11%), *A. beyrichiana* (9%) and *Q. myrtifolia* (9%).

Important changes were observed in the distribution and composition of the most abundant species found within each treatment type (Figures 11-13). Abundance of *Q. myrtifolia* in scrubby flatwoods approximately doubled as a result of burning, as did *V. myrsinites* (from 7% to 12%). Within sandhill communities, where species richness changed only slightly between unburned and burned plots (27 to 26 species), *Q. myrtifolia* again increased in relative abundance in burned areas (from 9% to 35%). In mesic flatwoods, *D. acuminatum*, a widespread perennial grass species, was one of the most abundant species in burned treatments in mesic flatwoods, but was not present in unburned transects. Conversely, *L. lucida*, one of the more abundant species, decreased significantly in abundance in burned mesic flatwood treatment sites from 37% to 20%.

Results from multivariate analysis of the effects of community type and treatment on individual species indicated that there were significant ($p<0.05$) responses for some species due to vegetation community type and the interaction between burning and community type, but not specifically for burn treatment alone (Tables 4 and 5). Additional ANOVA analysis of transformed species abundances identified differences in species abundances that can be explained by community habitat alone (Table 6). However, many of these same species and several others can be explained by the interaction between community type and burn treatment (Table 7). Similarly, some species exhibited changes in abundance due to fire, but this was not always consistent across communities (Table 11). Differences in the number of positive changes in species abundance due to burning or not burning were not significantly different across communities ($p=0.185$). Significant differences between the numbers of burns in a treatment were not observed (Table 8).

To better understand the progression of diversity relative to disturbance frequency, it is worth examining changes in the number of species present within a community since the time of last fire. For study plots that were in burn treatments, numbers of species are plotted by community type versus years since the unit was last burned (Figure 19). The figure suggests that sandhill communities exhibit a decrease in the number of species as the time since last fire increases, though this is not significant ($p=0.850$) for the first eight years, but species numbers significantly increase when changes are viewed over sixteen years ($p=0.033$). Species numbers slightly increase over the first eight years for scrubby flatwoods and mesic flatwoods ($p=0.654$

and p=0.345, respectively). Over a sixteen year span, species numbers increase in scrubby flatwoods, but decrease in mesic flatwoods (p=0.66 and p=0.96, respectively).

NMDS plots were generated from species abundance data for each plot (Figures 14 – 18). Often, relationships between variables can be realized better through visualization techniques rather than using computational methods. Plotting the principle components indicates there is some similarity in the distribution in the abundance of many of the herbaceous species and shrub species, suggesting these species may prefer similar habitats or respond in a similar manner to fire (Figures 14 and 15). It also indicates that there is similarity between many of the species shown in that exhibited a significant interaction effect between habitat and burn on individual plant species. Additionally, NMDS plots provide a visualization of the similarity between study transects (Figures 16-18). For instance, transects 8 and 26 plot close together and they are located in scrubby flatwood and mesic flatwood communities, respective, though both were burned several years prior. Transects 13, 19, 24 and 28 also plot closely, but are all located in different communities and treatments, making their proximities unusual. Transects 3, 9 and 16 also plot near each other, potentially due to their all having experienced fire recently. NMDS plots therefore suggest that there is some similarity between sites that have experienced fire and species with similar growth habits. Extrapolation of similarity should be viewed with some caution, given the stress values for Shepard diagrams in Figures 15 and 19. Shepard diagrams plot obtained versus observed ranks and indicate the quality of the result by how well the points fit a line that extends from the bottom left to the top right of the diagram. Stress measures how well the dissimilarities fit the line. Clarke (1993) suggests that a stress greater than 0.20 is

basically random, a value of 0.15 is good and less than 0.10 is close to actual dissimilarities. Since stress values for species and transects Shepard diagrams are 0.3148 and 0.1519, respectively, the NMDS plots likely provide a sense of the relationship between the variables, but may not represent those relationships well.

5. Discussion

For this study, the null hypotheses were no differences in plant species diversity between burned and unburned areas, and no differences in individual species abundances due to community type or burn treatment. Based on my results, whether or not a community was burned did not result in significant differences between Shannon-Wiener or Simpsons diversity values, nor did it result in significant differences between species abundances across communities. However, observed power values were low (less than 0.8), primarily for effects on diversity and the effects of burn treatment on species abundance. The power analysis therefore suggests there may be an increased chance of committing a Type II error occurring when drawing conclusions from the data analysis, but the null hypotheses should potentially be rejected. Other studies support this. Using the Shannon-Weiner species diversity index, Anguyi (2010) collected vegetation data on thirty 2.5m x 50m plots and found that fire frequency explained nearly all of the variation in species richness of woody species in the savannas of the Queen Elizabeth National Park in Uganda. DiTomasso et al. (1999) described management efforts in Sugarloaf State Park in northern California to reduce the invasive species, Yellow starthistle (*Centaurea solstitialis*), through the use of prescribed fire in a grassland community. Vegetation cover and diversity, species richness, and seedling density were measured using

point-intercept transects, random quadrats and soil samples, respectively. In addition to effectively controlling the invasive species through the reduction of the seedbank, the researchers found that native species diversity and richness increased at the study site.

However, Ruth et al. (2007) analyzed burned and unburned areas within three communities in Northwest Florida (longleaf pine, sand pine scrub and ecotonal ridges) in the 558-ha Naval Live Oaks area. They utilized 15 m x 25 m plots (20 plots and 10 plots in unburned and burned communities, respectively) to sample herbaceous and woody vegetation and did not find a significant difference in species richness between treatments. Abrahamson (1984) investigated fire-dependent communities in Lake Wales Ridge in Florida, including scrubby flatwoods and sandhills communities such as were evaluated in this study, utilizing permanent line transects at nine sites over five years. Shannon-Wiener diversity values were calculated, as were Horn's Index values for each site based on measures of species composition and abundance that were taken pre- and post-fire (both prescribed and natural fire). Though Abrahamson (1984) found that diversity increased significantly post-fire on two poorly drained sites, this was a short-lived change (two years) in gallberry-dominated flatwoods. At other sites, significant changes in diversity were not observed. Further, the authors suggested that the changes in diversity were not a result of an increase in species numbers, but rather changes in the evenness of the species at those sites. The current study supports this assertion, since diversity values were not significantly different between burned and unburned communities, but changes in the abundance of existing species were observed between treatments.

As mentioned above, differences in the abundance of specific species can be explained by the interaction between community type and burn treatment. That is, the abundance of some species is influenced by both the community they inhabit and the impact of fire within that community. Freeman and Kobziar (2011) alluded to this through their examination of wildfire sand pine scrub habitat in the Juniper Prairie Wilderness Area of north-central Florida. Analyzing post-burn vegetation response, they found that fire severity (high, low, none) varied with stand class and time since last fire. Severity, in turn, influenced the establishment of species and the successional trajectory of the system. Abundance of *Q. myrtifolia* increased in scrubby flatwoods and increased significantly in sandhill communities in response to fire. This is not unexpected, as *Q. myrtifolia* has been identified as a highly aggressive colonizer of postfire areas in southeastern forests (Freeman and Kobziar 2011). Abrahamson (1984), examining plant response to fire in Florida using permanent line transects over a five-year period, corroborates this assertion through the observation that *Q. myrtifolia* and other oak species responded quickly following fire (both prescribed and lightning-caused). Therefore, the results of my study are consistent with what others have found, particularly for Florida upland vegetation communities.

Dicanthium. acuminatum, a widespread perennial grass species, was not recorded in unburned transects, but was one of the most abundant mesic flatwood species in burned transects. This observation is consistent with research by DeSelm et al. (1974) and Ruth et al. (2007), who observed that frequency of *D. acuminatum* was positively correlated with burn frequency such that the most frequently burned areas had the greatest relative abundance of the species. *D. acuminatum* is an early successional species that reproduces by seed, has basal buds

that can sprout following fire and can produce thick tufts form at the base to protect it from damage. These characteristics allow *D. acuminatum* to proliferate well in fire-dependent communities. DeSelm et al. (1974) note that the species exhibits its highest abundances when fire frequency is greatest.

Lyonia lucida was observed to have decreased in abundance in mesic flatwoods as a result of fire application. Though research on the effects of fire on this species is limited, its basic biology is well understood. *Lyonia lucida* grows slowly and sprouts from rhizomes, characteristics that are consistent with persistence in fire-dependent communities and the reduced abundance observed, since it would take time for the species to return to pre-fire levels of abundance following burning (Van Deelen, 1991). Changes to *L. lucida* and other species abundances indicates that although diversity values for plant communities may not have significantly changed, burning positively affected the relative abundance of fire-dependent species. Additionally, these species serve as forage for native wildlife species and may provide an increase in faunal diversity, though this was outside of the scope of this study. Of the 12 species that exhibited a significant variation in abundance based on the interaction between vegetation community and fire, the majority were herbaceous (7) and the remainder consisted of trees (2), shrubs (2) and a vine (1). This is consistent with what would be expected given ecological succession theory which suggests that following disturbance, the early stages of a community area dominated by pioneer colonizers consisting of annual and perennial herbaceous species with life history characteristics such as longer seed viability and dispersal distances, higher photosynthetic productivity and rapid growth rates (Bazzaz 1979). Thus, whether or not

fire significantly influenced diversity within one or more of the communities in this study, it clearly had a significant effect on the development of successional stages and thereby likely influenced broader beta diversity within the park.

Burned treatment zones have generally either been burned one or two times within their natural fire return interval. This is a source of potential difference in diversity values between treatments. However, a significant difference was not observed between treatments that experienced one or two burns (Table 8). Treatment zones did have similar species compositions between communities; that is, burned and unburned scrubby flatwoods shared many of the same species even though abundances differed, and the same was true for mesic flatwoods and sandhill communities. Since there was no significant difference in diversity index values between burned and unburned treatments, this suggests that initial species composition within communities may determine the trajectory of species composition following disturbance. Keely et al. (2003) found that in higher elevation coniferous forests, time since fire played an important role in determining levels of diversity within those communities by creating patches of early successional habitat within the forest structure. However, the seed banks that provide the input for species to colonize newly available habitat patches only persist for finite periods of time and therefore if the time since the last fire is greater than the persistence time of the seed bank, diversity may be significantly less when patches are created (Keely et al., 2003). For PHCPSP, areas within the park have not experienced fire beyond recent prescribed fire application in the last forty years, well outside the natural fire return interval, supporting the idea that a lower available initial species composition may have influenced the results of this study. Ruth et al.

(2003) in studying fire interactions in longleaf pine, ecotonal and scrub habitats in Northwest Florida also did not find a difference in herbaceous layer diversity between burned and unburned areas longleaf communities, citing limited dispersal from “a depauperate seed bank” as a potential causal factor. They suggest that restoration of the unburned fire-dependent communities within their study area will require repeated fire application, along with supplemental seeding.

Although overall the number of species responding positively to burning was not significantly different across communities (p=0.185), responses for some species varied depended upon which community they were in (Table 11). In fact, only one species (*Andropogon virginicus*) positively responded to fire when present in all three communities, and two others (*Gaylussacia dumosa* and *Smilax auriculata*) responded negatively in all three. Responses varied across community type for the remaining species, suggesting that how fire affects a species may be dependent on the broader community composition surrounding it. This may potentially be a result of how surrounding vegetation affects fire behavior due to changes fuel levels, continuity and other factors, as has been observed in other research (Whisenant, 1990; Sampaio et al., 1993). This result may also have confounded the determination of whether or not diversity changes significantly due to fire, since species abundances may go up or down depending on the community type.

Information on the effects of natural and prescribed fire on plant diversity at the local level (alpha diversity) is limited, particularly for southeastern forests. This study found that alpha biodiversity is not significantly different between burned and unburned fire-dependent

plant communities in northern Florida and does not support the IDH, at least at the time scale measured. This suggests that the use of prescribed fire may not significantly affect biodiversity in the three vegetation communities studied. The study also did not include data from plots that had very recently experienced disturbance, i.e., immediately following the application of fire. Other research suggests that diversity is low immediately following disturbance from fire because, as would be expected, since much of the biomass has been removed and the site is re-set to its earliest successional stage (Trabaud and Lepart, 1980; Safford and Harrison, 2004). Taking into account that the time period immediately following fire would have limited species richness, the burned areas would likely fit the assumptions of the IDH, at least for the left portion of the graph in Figure 1.

Given that sandhill communities have a short fire return intervals of 1 – 3 years, results from this study fit well with the IDH if it is assumed that numbers of species are low immediately following burning (Figure 19). Likewise, scrubby flatwoods have a much longer fire return interval (5 – 15 years) and the change in the number of species suggests a similar pattern to that of sandhills, but stretched over a longer period of time. Mesic flatwoods do not show a clear trend, with a nearly flat regression line. However, the coefficient of determination (R^2), which is a measure of the variation in the data explained by the regression line, is very low for both mesic flatwoods and scrubby flatwoods, indicating the regression line provides a poor fit to the data (R^2=0.001, p=0.345 and R^2=0.073, p=0.654, respectively). Another view of how species numbers vary over time since fire depicted in Figure 20, which includes all treatments. Unburned areas do not have a burn history; therefore, a minimum of 16 years is used for

unburned areas (the earliest documented burn within the park). Despite R^2 values being low again (nearly 0), regression lines now show increasing species numbers over time for sandhill and scrubby flatwoods, and decreasing species for mesic flatwoods (R^2=0.0356, p=0.033; R^2=0.0064, p=0.66; and R^2=0.222, p=0.96, respectively). Although this figure does not provide clear support for or against IDH, it does show the importance of including abundance when analyzing the diversity of a community. Further, the IDH may operate on different temporal or spatial scales dependent upon the habitat in question.

The power analysis suggests that there is an increased chance of a Type II error occurring and the null hypothesis should be rejected rather than supported. If a Type II error did indeed occur, there would be a difference in diversity values between burned and unburned treatments, as well as support for the IDH. Interestingly, this is consistent with Beckage and Stout (2003) who similarly did not find a relationship between species richness, diversity or flowering stem density and fire frequency in their analysis of a Florida sandhill community, but there study also had low statistical power. Despite the application of prescribed fire as a surrogate to natural fire in three fire-dependent communities, there was no significant difference in alpha (local) diversity values in any of the communities within the study area.

The prediction of the IDH is that within communities, maximum species diversity will occur at intermediate levels of disturbance, which may be measured in terms of frequency or intensity. As with many ecological hypotheses, theoretical and empirical data can be found to support or refute the IDH. Those who find fault with the IDH reasonably suggest that the hypothesis is too simple and does not account for the wide variation in systems and species

interactions found within communities (Garstecki and Wickham, 2003; Fox 2012). Supporters do not disagree, but suggest that IDH is not a mechanism in and of itself, but rather a manifestation of the multitude of interactions that play out in a community (Roxburgh et al., 2004; Hughes, 2010). For example, Hall et al. (2012) present an alternative to the IDH, which is that diversity patterns observed in disturbed systems occur due to the evolution of species adapting to disturbances. Whether or not this is the direct mechanism, IDH can still serve as a prediction tool for the same observable patterns.

Biswas and Mallik (2010) found that in their study their system followed the prediction of IDH, but make the important note that species richness and patterns of diversity are dependent upon the community and therefore their findings are best applied to similar communities, not broadly applied to all systems. Likewise, Peterson and Reich (2008) found that when fire frequency is broadly analyzed across several different types of assemblages, it generally agreed with the predictions of IDH. However, when the interaction between fire and specific vegetation guilds was analyzed, they found something different. Overstory tree and other woody species richness was highest in unburned units and declined with increasing fire frequency, while species richness was greatest with biennial fires for forbs and with annual fires for grasses. When results are combined, IDH predictions correspond with the study, but they do not when more defined groups of vegetative species are analyzed. This suggests that the IDH describes community interactions well as a whole, but may not be appropriate for individual groups.

The results of this study should be considered in the context of the Florida Park Service fire program (a long-term vision) and the unique challenges of implementing prescribed fire at

PHCPSP, which include high fuel concentrations, limited burn history and difficult coastal weather patterns. As a result, sites that have been burned within the normal fire return interval have only been so one or two times. This frequency may not be enough to begin to see changes in vegetation abundance and species compositional changes in burned areas, as has been observed in areas with older burn programs within the park system. Simulation work by Baker (1994) and Keane et al. (1996) support this observation, indicating that where fire has been excluded, two to seven applications of fire and up to 75 years may be needed to return for fire-dependent ecosystems to natural diversity levels. Further, supply may limit recruitment of additional species in burned areas in the short term; that is, habitat (e.g., exposed soil) may become available due to the application of fire, but if seed stock is not available then the area will simply be re-colonized by the same existing vegetation and changes in diversity will not be observed.

Continuing to collect data at the transects established during this study is recommended, since monitoring changes in vegetation over time at permanently established study plots would likely yield the best information to determine the effects of fire (and other management) on vegetation species and help to inform the Florida Park Service fire program. Greatest benefit would be derived from gathering data within thirty days of fire application and one year following the burn in order to determine immediate and short term post-fire effects, as well as long-term effects over time. It will also allow researchers and managers to track changes resulting from repeated fire application to determine if there is a requisite threshold number of applications within the natural fire return interval for each community that is necessary before

changes in diversity will be seen. In addition to collecting abundance information, information on cover may be beneficial as well, as fire not only affects species abundance, but can reduce or enhance the cover of different species, which in turn influences invertebrate and vertebrate species habitat conditions. Beyond informing the fire program, these suggested studies would also be beneficial for exploring the applicability of the Intermediate Disturbance Hypothesis to these communities, particularly if other taxonomic groups are included (e.g., birds, insects, microbes).

Tables

Table 1. Vegetation transect treatment types and associated diversity index values.

Plot #	Community Type	Burn Treatment	Shannon-Weiner (H’)	Simpson’s (D)
23	Mesic Flatwoods	Burned	2.046	0.149
26	Mesic Flatwoods	Burned	2.362	0.105
29	Mesic Flatwoods	Burned	1.750	0.221
6	Mesic Flatwoods	Unburned	1.366	0.378
13	Mesic Flatwoods	Unburned	1.517	0.281
15	Mesic Flatwoods	Unburned	2.010	0.161
3	Sandhill	Burned	2.168	0.177
10	Sandhill	Burned	1.758	0.250
28	Sandhill	Burned	1.951	0.178
1	Sandhill	Unburned	1.836	0.169
2	Sandhill	Unburned	1.620	0.242
12	Sandhill	Unburned	2.262	0.144
20	Sandhill	Unburned	1.982	0.137
24	Sandhill	Unburned	1.837	0.173
25	Sandhill	Unburned	2.458	0.096
7	Scrubby Flatwoods	Burned	2.106	0.178
8	Scrubby Flatwoods	Burned	2.381	0.126
9	Scrubby Flatwoods	Burned	1.950	0.202
16	Scrubby Flatwoods	Burned	1.629	0.235
19	Scrubby Flatwoods	Burned	2.064	0.156
17	Scrubby Flatwoods	Unburned	2.196	0.126
18	Scrubby Flatwoods	Unburned	1.745	0.284
27	Scrubby Flatwoods	Unburned	2.314	0.119

Table 2. Results of MANOVA analysis of effects of habitat and burn treatment on Shannon-Wiener and Simpson's diversity index values.

Effect	F	Significance	Observed Power (α=0.05)
Habitat	1.087	0.380	0.302
Burn	2.064	0.159	0.362
Habitat x Burn	1.639	0.189	0.466

Table 3. Overall significance tests from multivariate tests of habitat and burn treatment on Shannon-Wiener and Simpson's diversity index values.

Effect		Value	F	Hypothesis df	Error df	Significance
Habitat	Pillai's Trace	.483	.689	12	26	.747
	Wilks Lambda	.562	.668	12	24	.764
	Hotelling's Trace	.701	.643	12	22	.784
	Roy' Largest Root	.561	1.215	6	13	.359
Burn	Pillai's Trace	.348	1.068	6	12	.433
	Wilks Lambda	.652	1.068	6	12	.433
	Hotelling's Trace	.534	1.068	6	12	.433
	Roy' Largest Root	.534	1.068	6	12	.433
Habitat x Burn	Pillai's Trace	.754	1.311	12	26	.271
	Wilks Lambda	.382	1.235	12	24	.317
	Hotelling's Trace	1.261	1.155	12	22	.370
	Roy' Largest Root	.832	1.803	6	13	.175

Table 4. Change in relative abundance in most abundant species by treatment type and community.

Community	Species	Abundance (%) (Unburned)	Abundance (%) (Burned)
Sandhill	*Quercus myrtifolia*	9	35
	Vaccinium myrsinites	11	11
	Aristida beyrichiana	9	10
Mesic Flatwoods	*Dicanthelium acuminatum*	0	10
	Lyonia lucida	37	20
	Ilex glabra	11	17
	Serenoa repens	11	10
Scrubby Flatwoods	*Quercus myrtifolia*	13	26
	Vaccinium myrsinites	7	12
	Gaylussacia dumosa	29	3
	Serenoa repens	9	10

Table 5. Results of MANOVA analysis of habitat and burn treatment on plant species.

Effect	F	Significance	Observed Power (α=0.05)
Habitat	34.276	0.029	0.836
Burn	36.947	0.129	0.305
Habitat x Burn	43.162	0.023	0.896

Table 6. Overall significance tests from multivariate tests of habitat and burn treatment on plant species abundance.

Effect		Value	F	Hypothesis df	Error df	Significance
Habitat	Pillai's Trace	1.948	4.433	34	4	.078
	Wilks Lambda	.000	34.276	34	2	.029
	Hotelling's Trace	17611.987	.000	34	0	.
	Roy' Largest Root	17593.624	2069.838	17	2	0
Burn	Pillai's Trace	.998	36.947	17	1	.129
	Wilks Lambda	.002		17	1	.129
	Hotelling's Trace	628.105		17	1	.129
	Roy' Largest Root	628.105		17	1	.129
Habitat x Burn	Pillai's Trace	1.884	1.905	34	4	.282
	Wilks Lambda	.000	43.162	34	2	.023
	Hotelling's Trace	62796.254	.000	34	0	.
	Roy' Largest Root	62788.656	7386.901	17	2	0

Table 7. Species with significant abundance due to habitat.

Species Code	F	Significance	Observed Power (α=0.05)
ANVI	7.443	0.005	0.894
AREB5	5.469	0.015	0.777
CACO37	4.941	0.020	0.732
COFL	6.128	0.010	0.824
HYCI	3.198	0.066	0.533
PISE	5.226	0.017	0.757
PTAQ	7.045	0.006	0.876
QUMY	5.416	0.015	0.772
SERE	4.401	0.029	0.678
VIRO	6.108	0.010	0.823

Table 8. Significant effects of interaction between habitat and burn on individual plant species.

Species Code	F	Significance	Observed Power (α=0.05)
AREB5	3.904	0.040	0.623
CACO37	4.941	0.020	0.732
CLEV	5.762	0.012	0.799
COFL	6.128	0.010	0.824
DICHA2	3.653	0.048	0.592
GAEL	5.977	0.011	0.814
HYCI	3.198	0.066	0.533
PISE	5.226	0.017	0.757
PTAQ	4.029	0.037	0.637
QUCH	10.717	0.001	0.973
QUMY	4.001	0.038	0.634
VIRO	6.108	0.010	0.823

Table 9. Overall significance tests from multivariate tests of habitat and burn treatment on plant species abundance.

Effect		Value	F	Hypothesis df	Error df	Significance
Burn	Pillai's Trace	0.493	0.873	12	32	0.581
Freq	Wilks Lambda	0.537	0.913	12	30	0.546
	Hotelling's Trace	0.808	0.942	12	28	0.522
	Roy's Largest Root	0.731	1.950	6	16	0.134

Table 10. Results of chi-square test of change in abundance in response to burning by community type.

	Mesic Flatwoods	Sandhill	Scrubby Flatwoods
Positive response to burn	15	16	15
Positive response to lack of burning	14	20	10

p= 0.185

Table 11. Change in abundance in response to burning.

	Mesic Flatwoods	Sandhill	Scrubby Flatwoods
Andropogon virginicus	+	+	+
Aristida beyrichiana	+	+	-
Baccharis halimifolia	0	0	+
Befaria racemosa	+	+	-
Carphephorus corymbosus	0	+	+
Clandina evansii	0	+	-
Cornus florida	-	0	0
Dichanthelium acuminatum	+	-	+
Galactia elliottii	+	-	+
Gaylussacia dumosa	-	-	-
Gelsimium sempervirens	-	-	0
Gordonia lasianthus	-	0	0
Hypericum cistifolium	+	+	0
Hypericum spp.	+	-	-
Ilex glabra	+	-	+
Ilex opaca	-	-	+
Ilex vomitoria	0	-	-
Kalmia hirsuta	0	0	+
Lechea torreyi	0	+	0
Liatris tenufolia	0	+	0
Lyonia ferruginea	+	-	+
Lyonia lucida	-	+	-
Myrica cerifera	-	-	+
Pinus elliottii	+	0	0
Pinus palustris	0	+	0
Pinus serotina	-	0	0
Pteridium aquilinum	0	-	-
Pterocaulon Elliott	0	+	0
Quercus chapmanii	+	-	-
Quercus geminata	+	-	+
Qurecus laurifolia	-	-	0
Quercus laevis	0	-	0
Quercus minima	0	+	0
Quercus myrtifolia	-	+	+
Quercus pumila	0	-	0
Sabatia brevifolia	+	0	0
Serenoa repens	-	-	+
Seymeria cassioides	0	+	0
Solidago fistulosa	+	0	0
Smilax auriculata	-	-	-
Tillandsia usneoides	-	+	+
Urtica dioica	0	+	0
Vaccinium myrsinites	+	-	+
Vaccinium staminium	0	-	0
Vitis rotundifolia	-	0	0

Figures

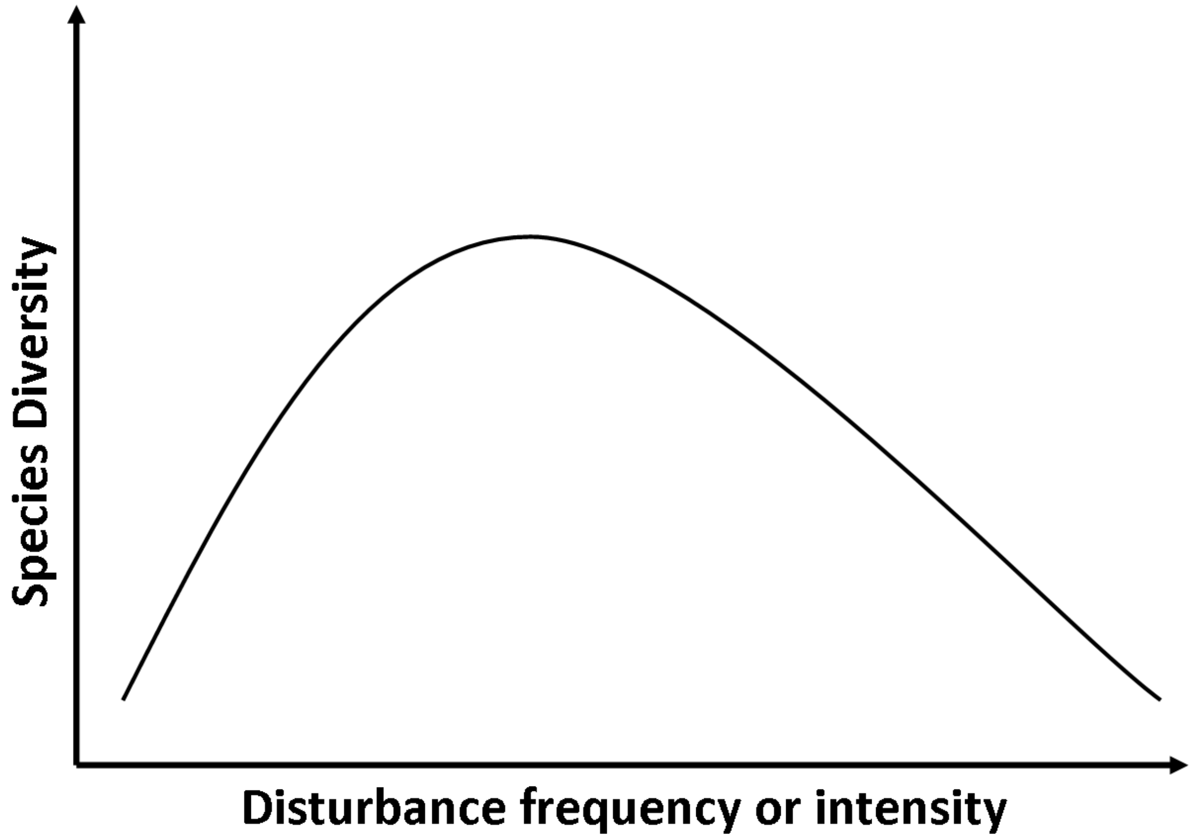

Figure 1. Graph of the Intermediate Disturbance Hypothesis showing the relationship between diversity and disturbance. Highest species diversity is found at intermediate disturbance levels. (After Connell, 1978).

Map redacted, paper copy available upon request to home institution.

Figure 2. Location of Pumpkin Hill Creek Preserve State Park.

PUMPKIN HILL CREEK STATE PARK

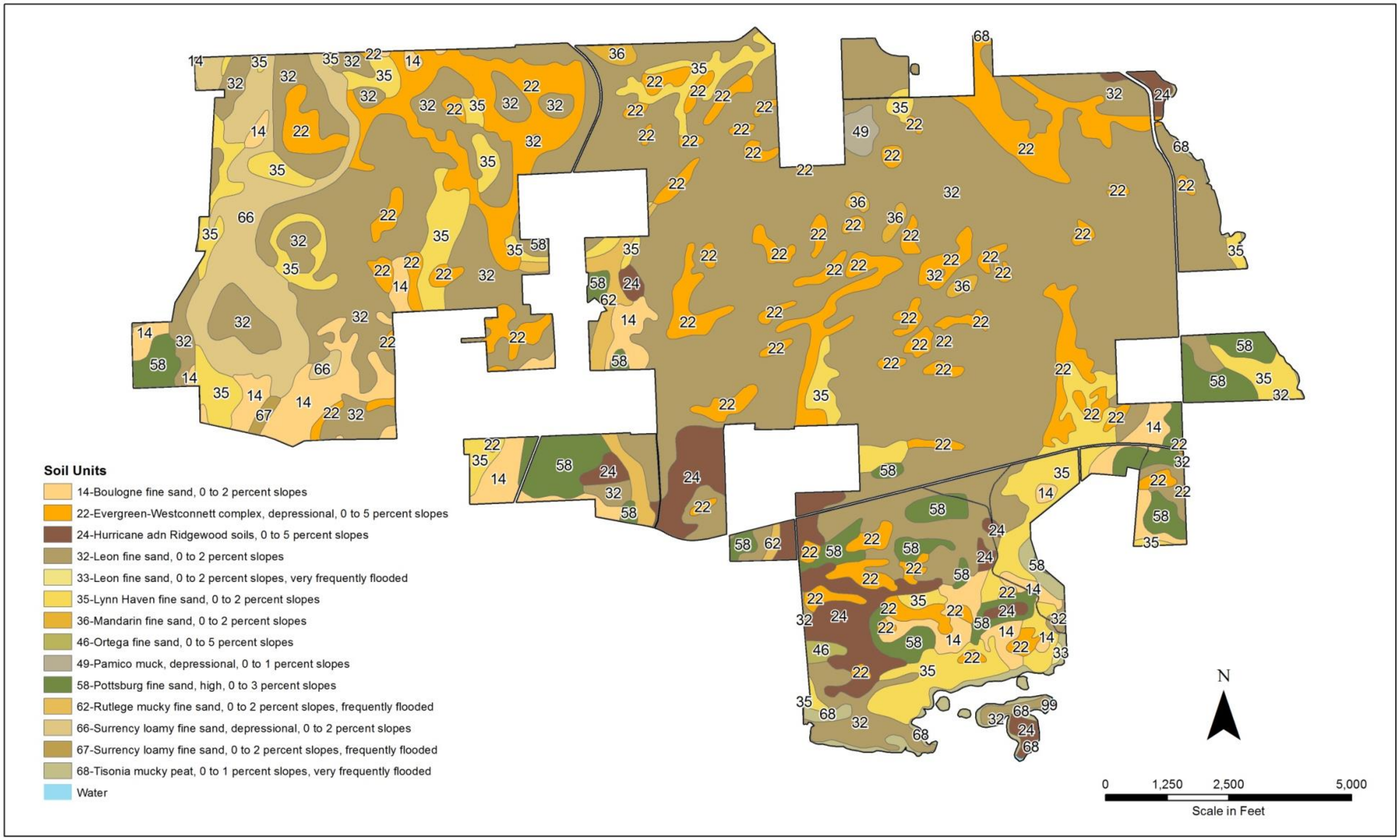

Figure 3. General soils map for Pumpkin Hill Creek Preserve State Park (data source Soil Survey Staff, 2014).

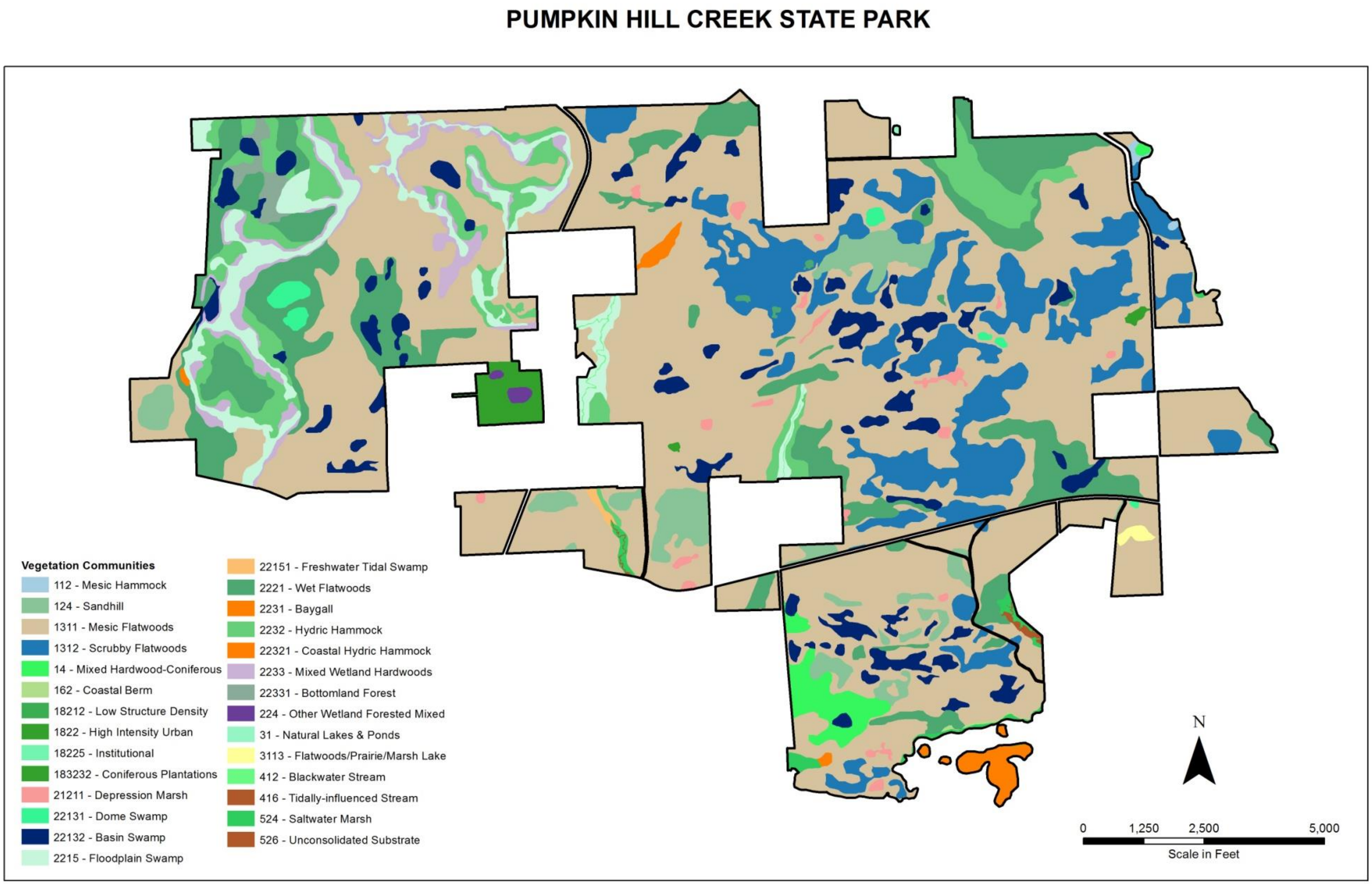

Figure 4. Vegetation communities of Pumpkin Hill Creek Preserve State Park based on Florida Natural Area Inventory (2010).

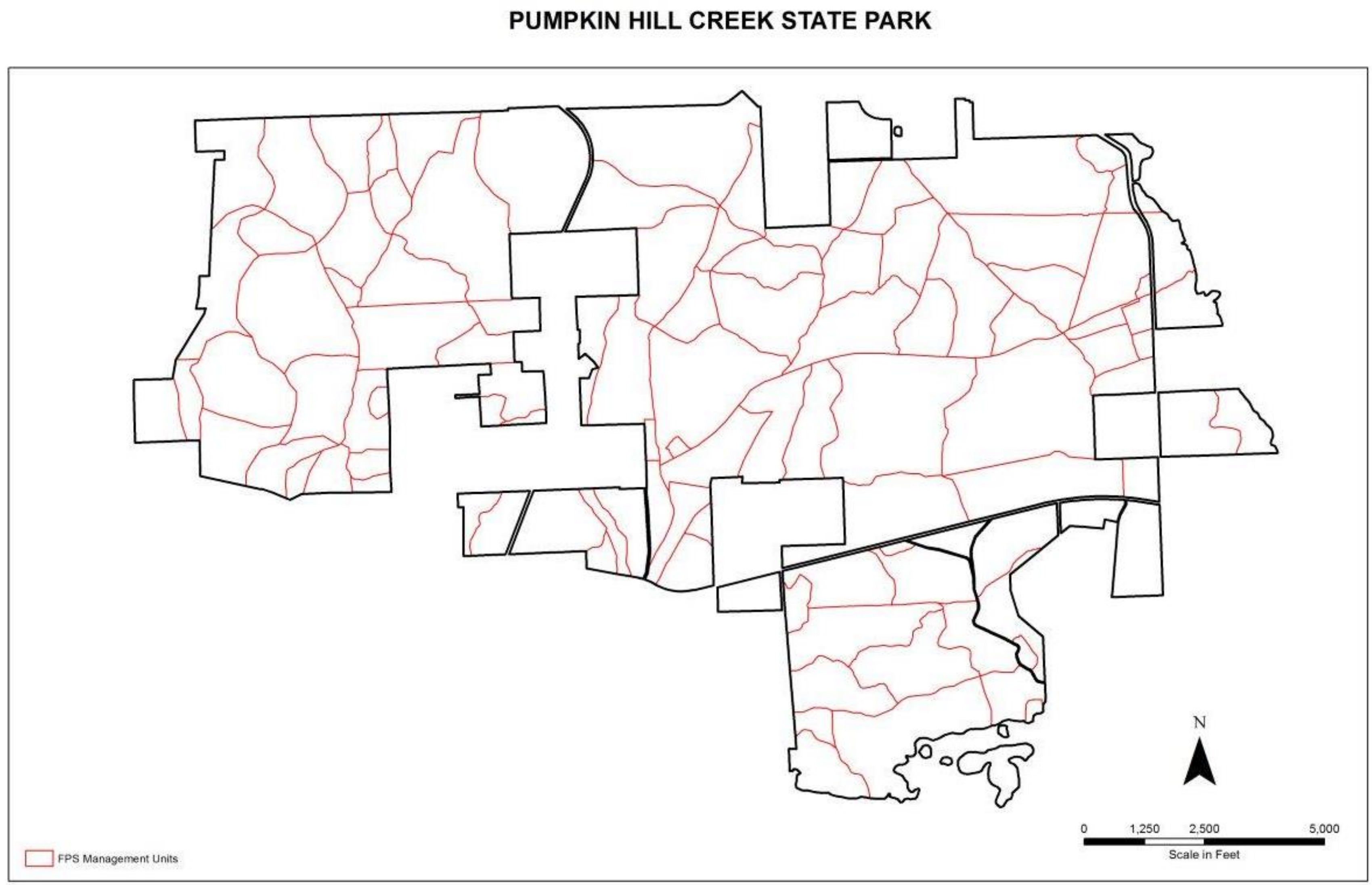

Figure 5. Fire management units within Pumpkin Hill Creek Preserve State Park.

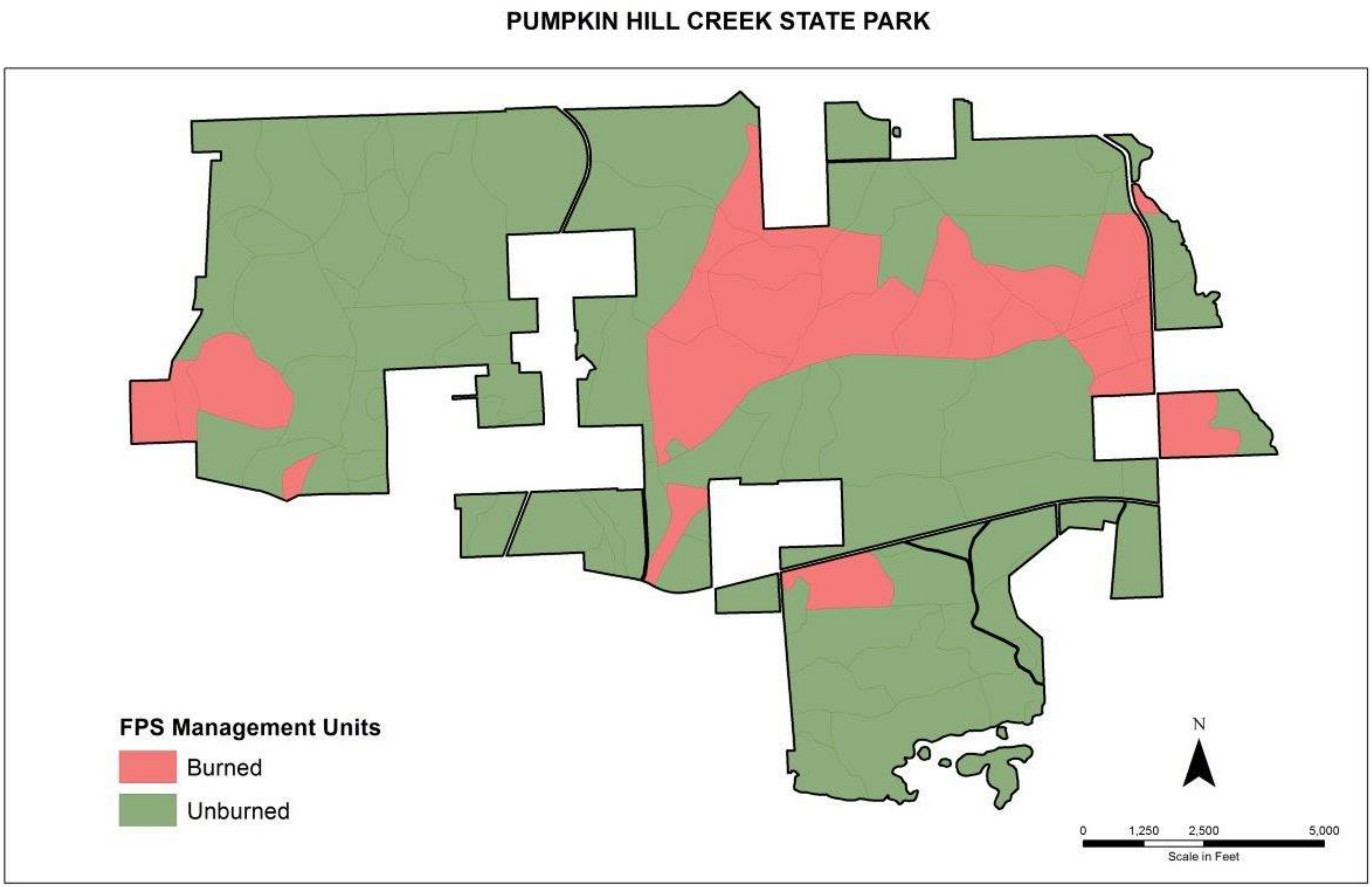

Figure 6. Pumpkin Hill Creek Preserve State Park management units showing burned and unburned units.

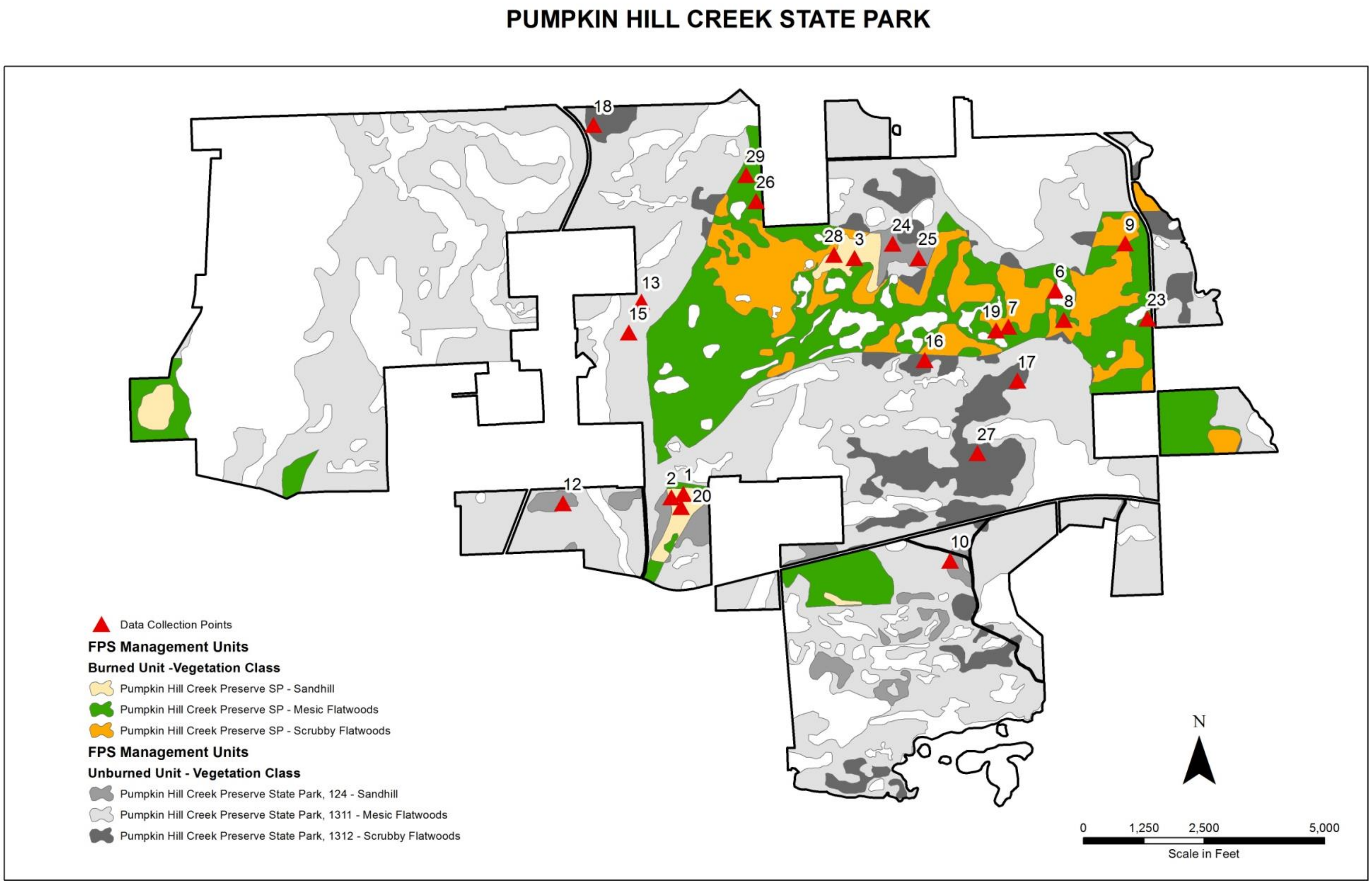

Figure 7. Study transect locations based on vegetation community and burn status.

Figure 8. Example of vegetation species intersecting transect showing how species are counted as part of a study plot.

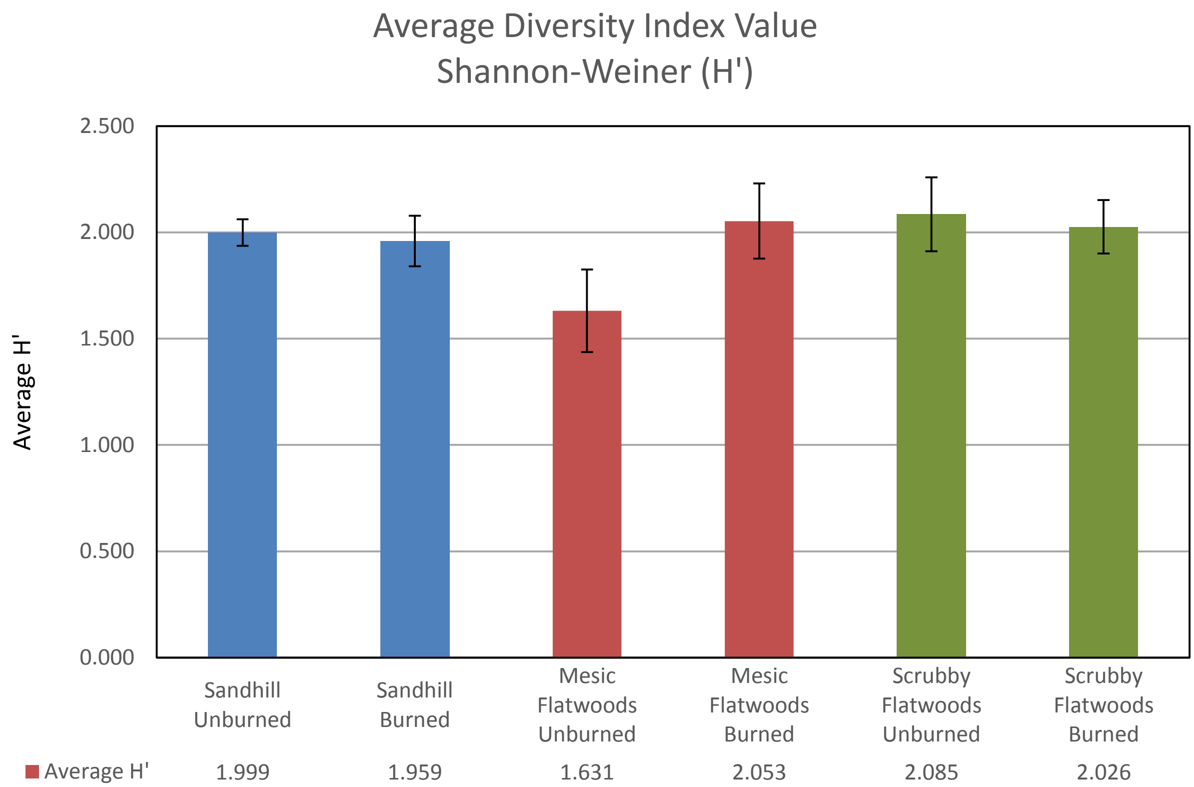

Figure 9. Average Shannon-Weiner diversity index values for burned and unburned areas by community type.

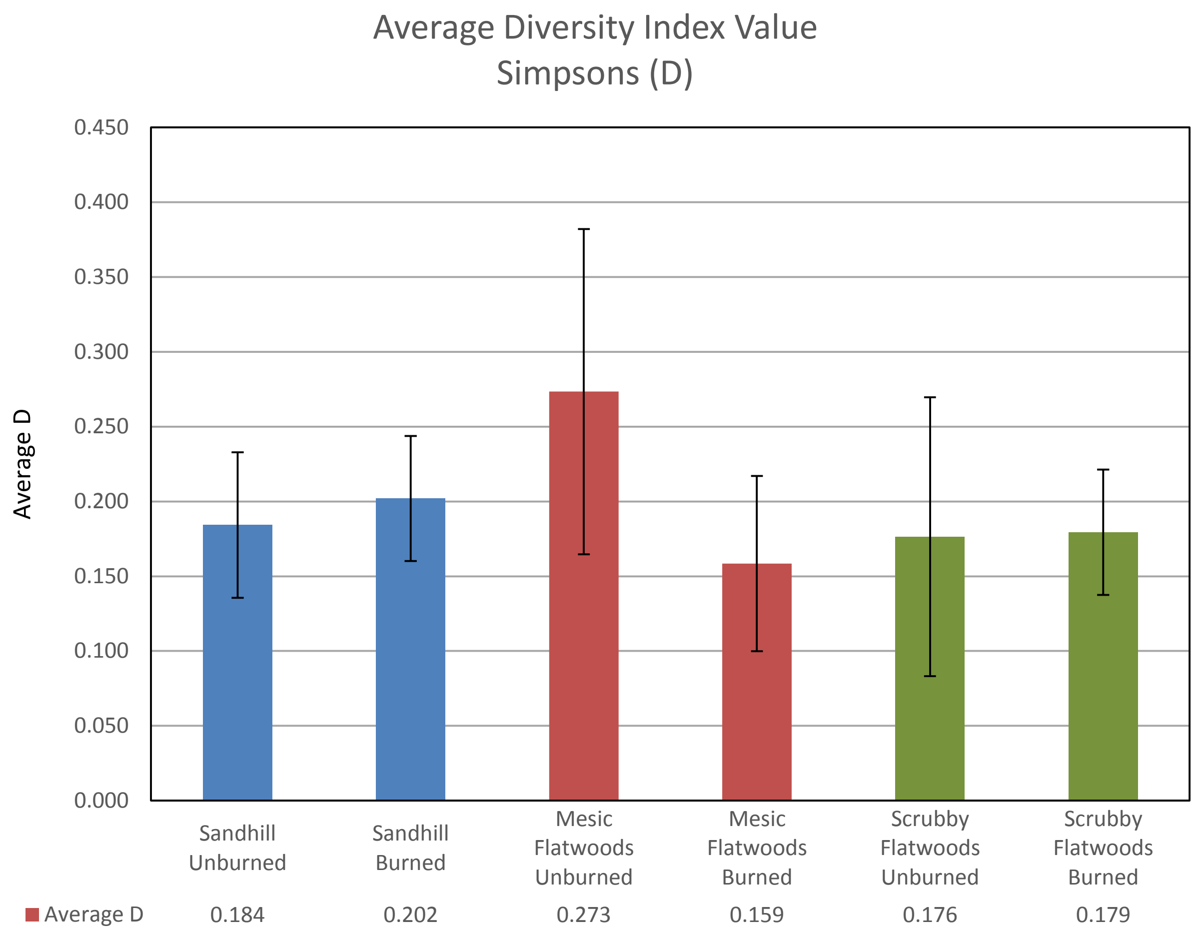

Figure 10. Average Simpson's diversity index values for burned and unburned areas by community type.

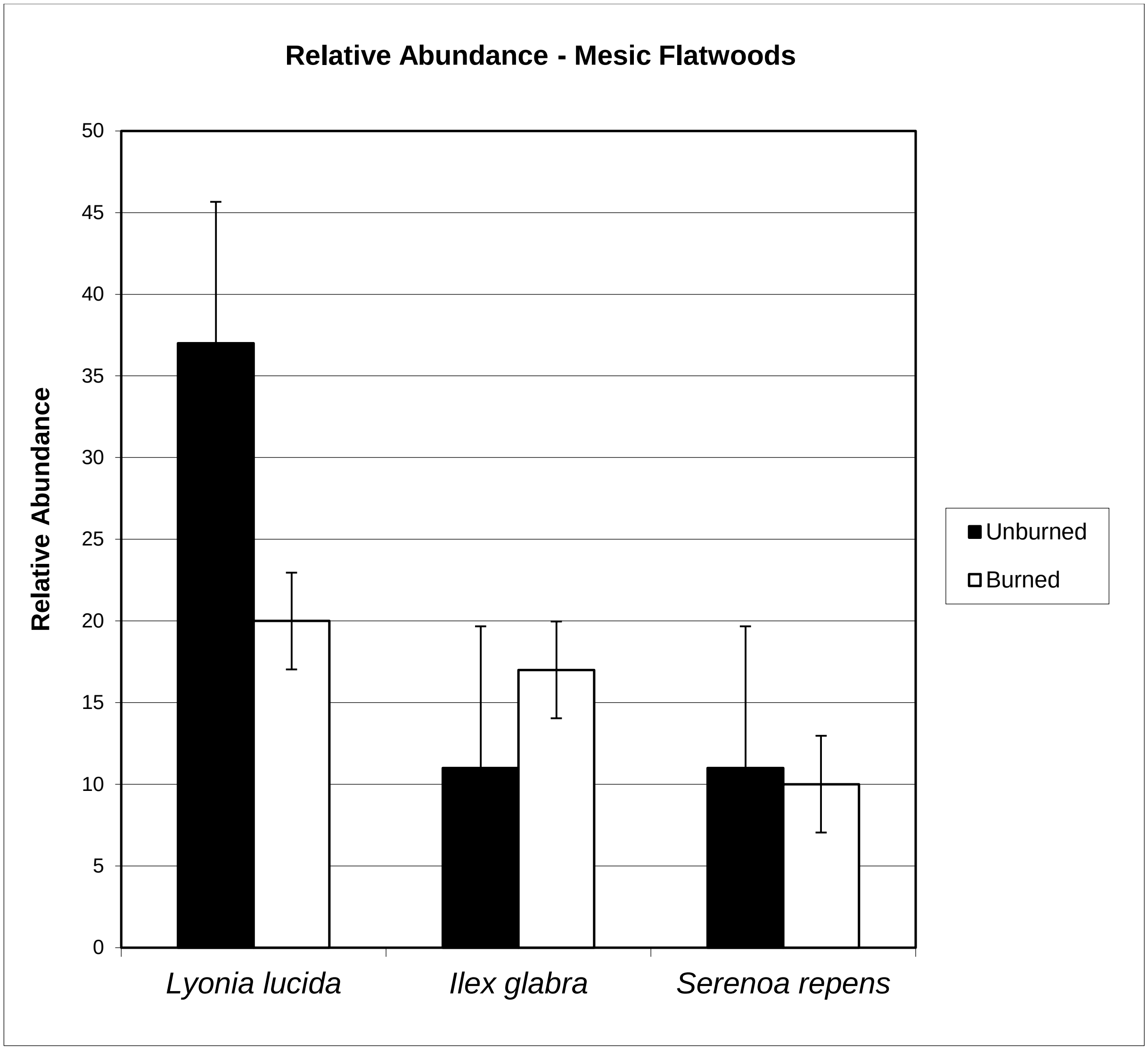

Figure 11. Comparison of relative abundance of most abundant species within unburned and burned mesic flatwoods community type. *L. lucida* abundance is reduced in burned areas of mesic flatwoods.

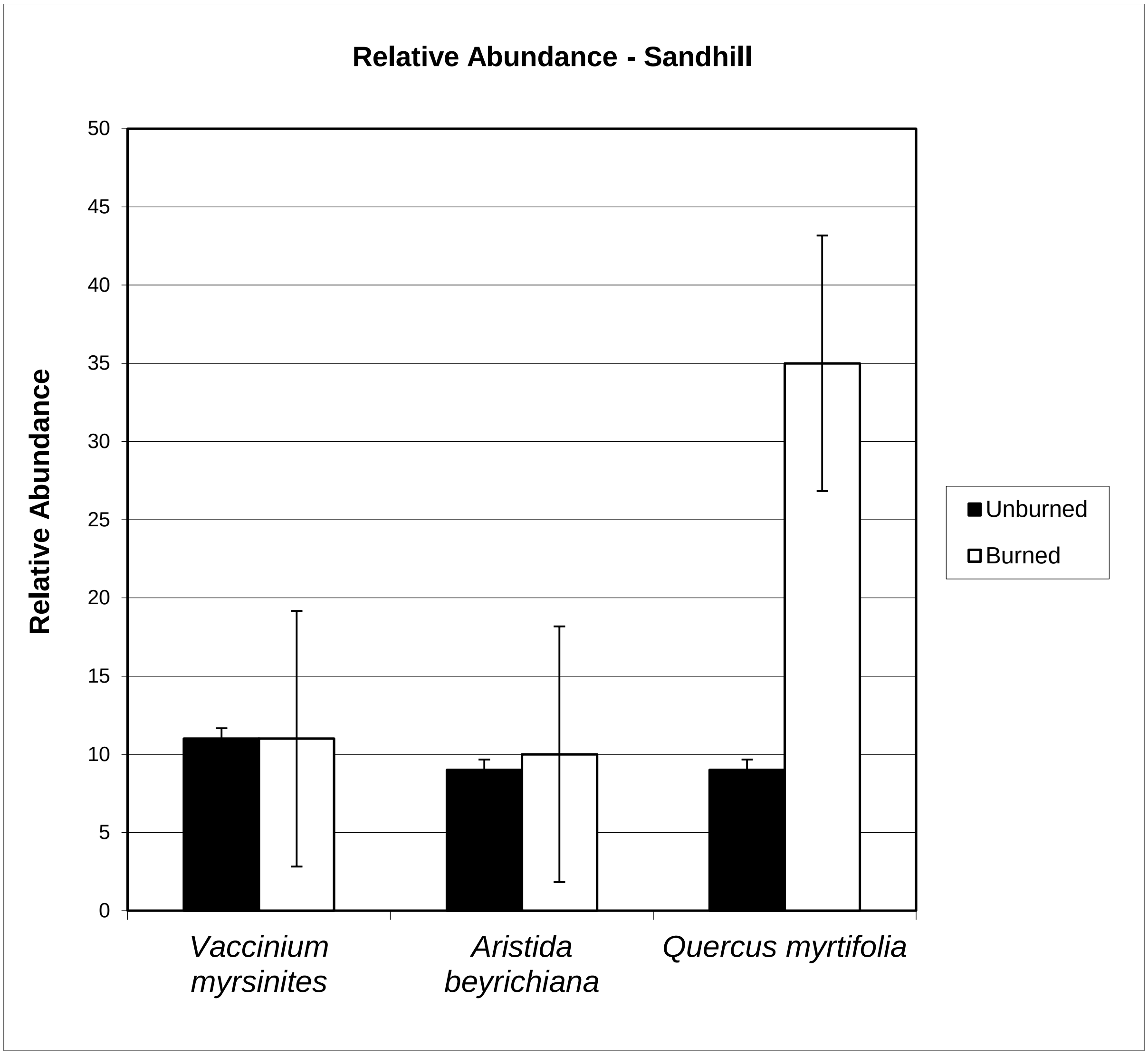

Figure 12. Comparison of relative abundance of most abundant species within unburned and burned sandhill community type.

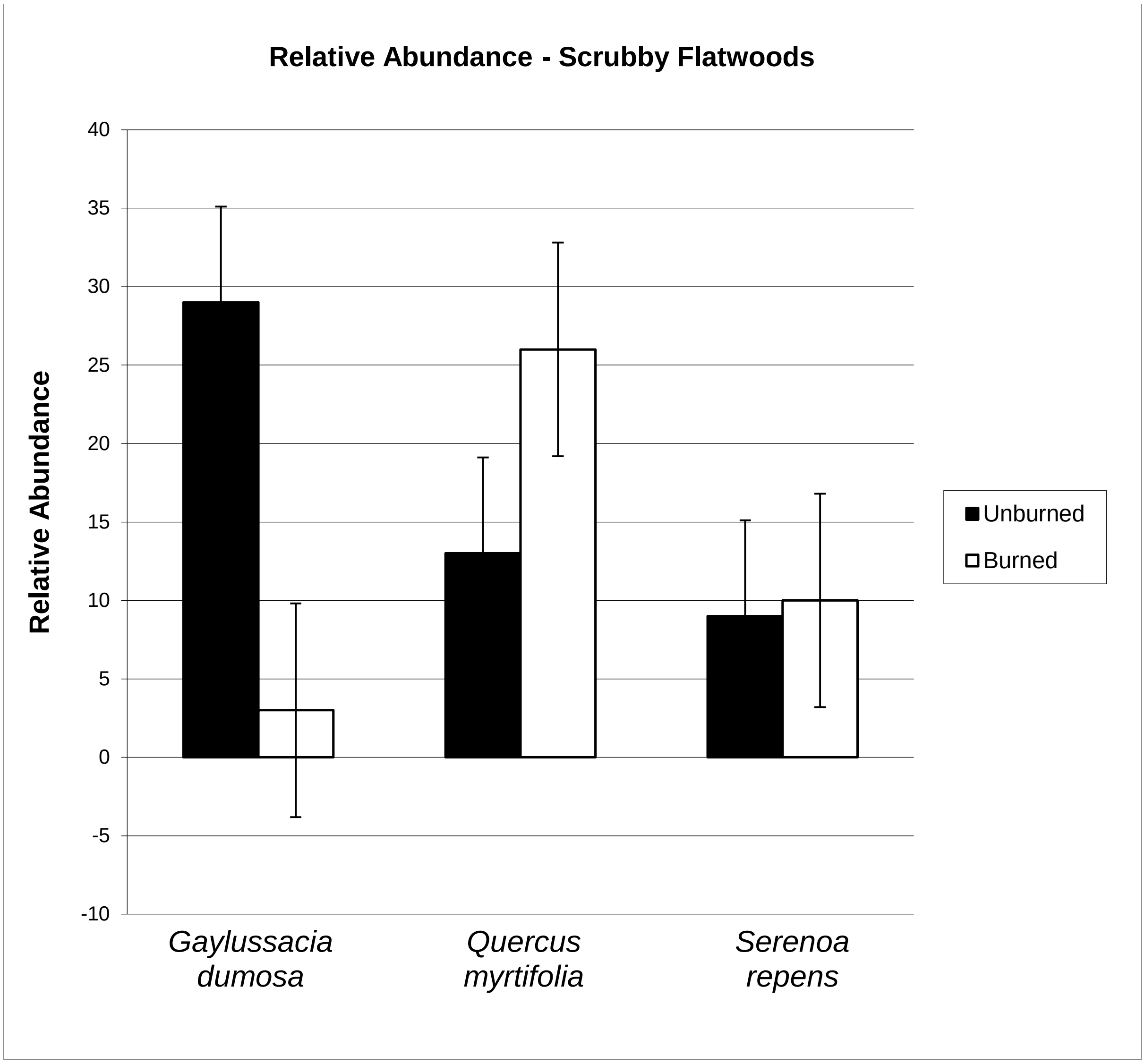

Figure 13. Comparison of relative abundance of most abundant species within unburned and burned scrubby flatwoods community type.

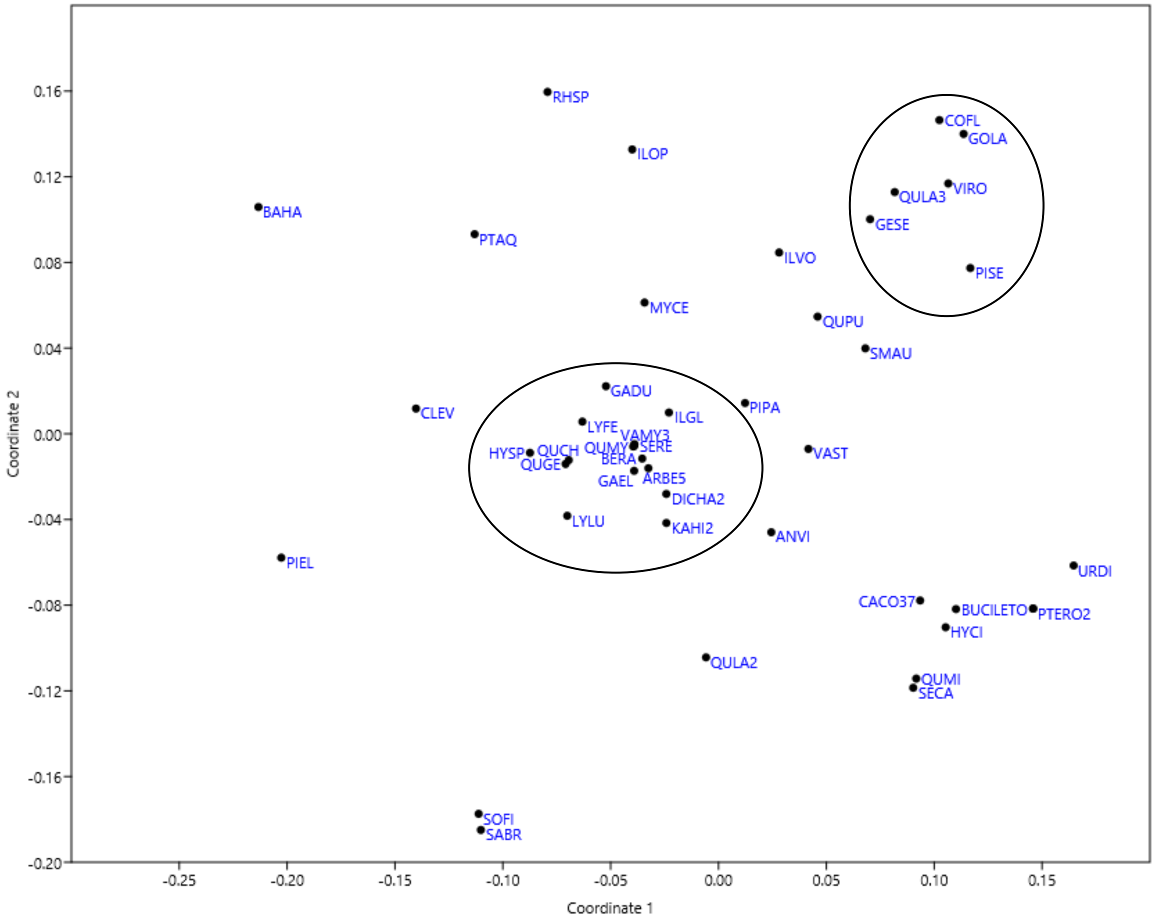

Figure 14. NMDS plot of species similarity based on abundance. Many of the herbaceous species are found in similar treatments.

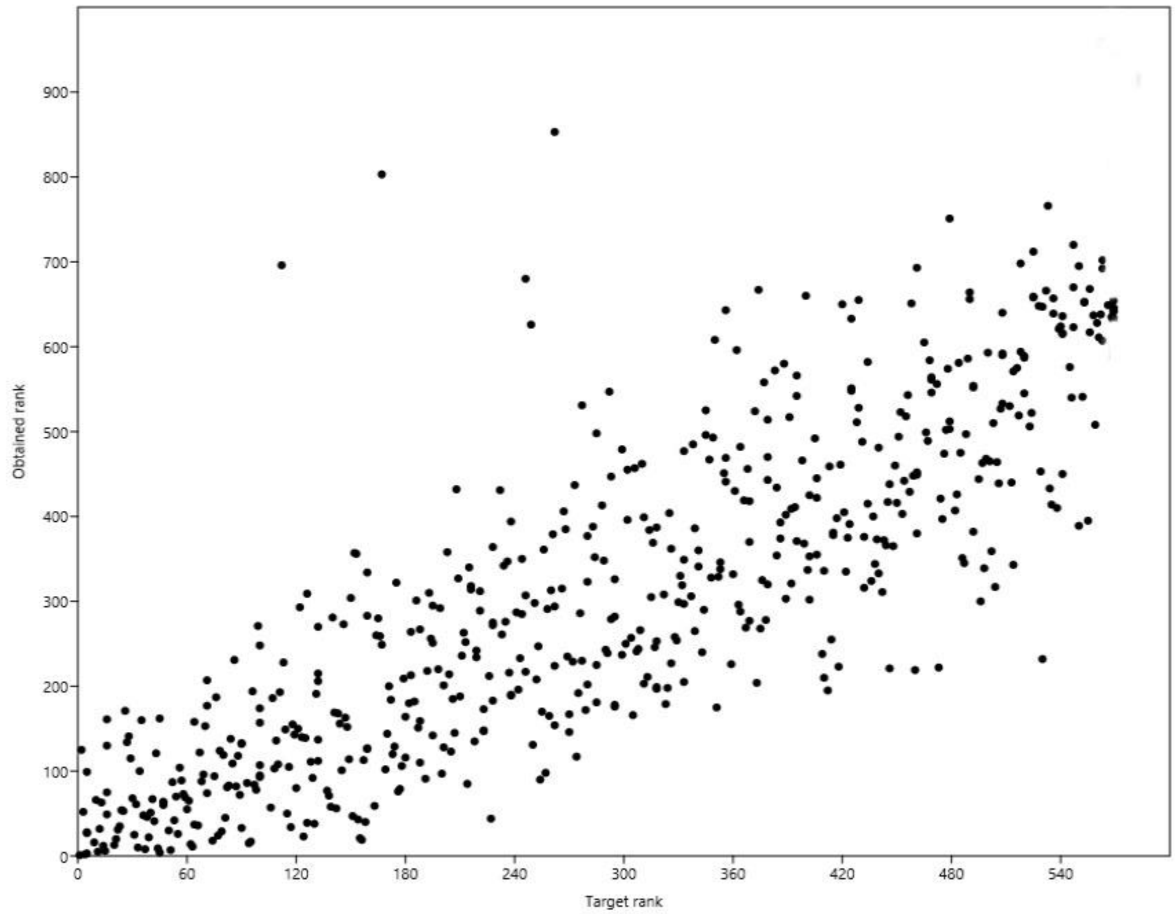

Figure 15. Shepard diagram of species similarity based on abundance. Stress value = 0.3148.

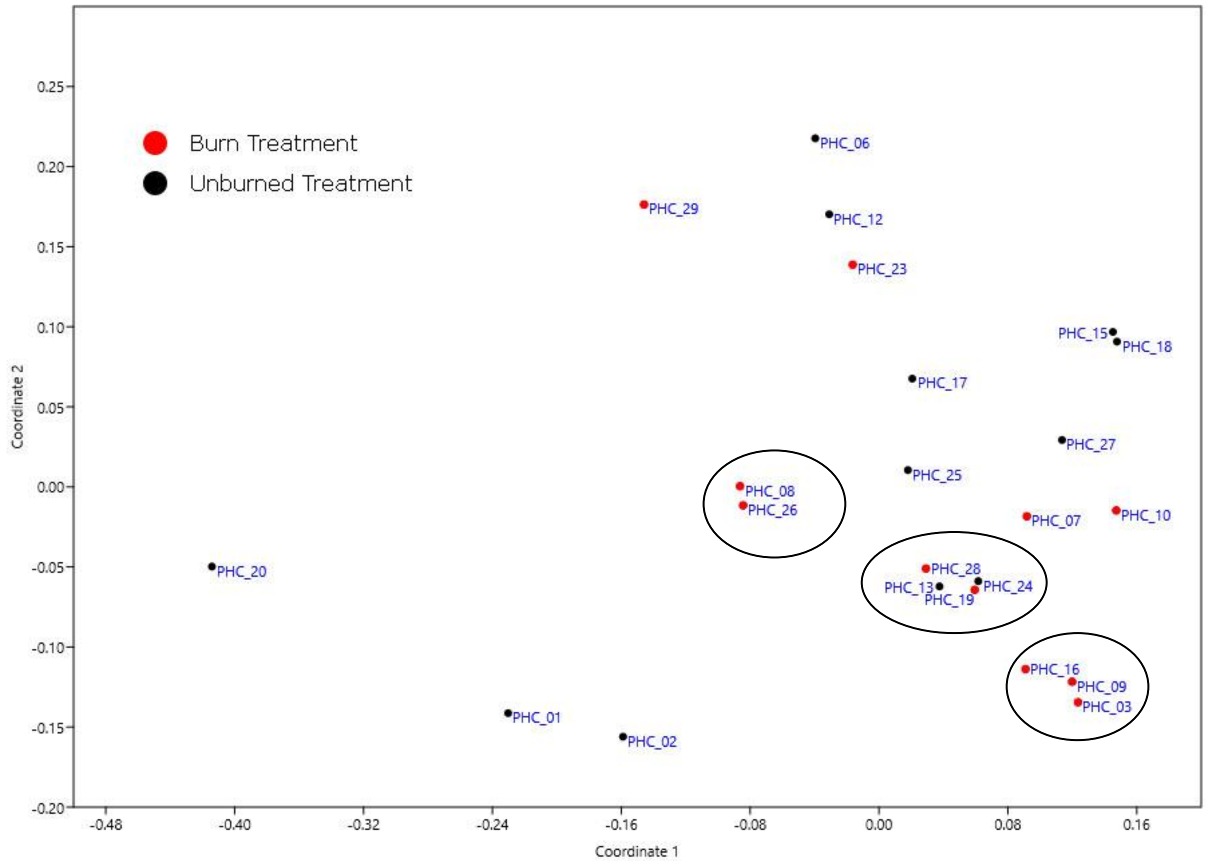

Figure 16. NMDS plot of transect similarity based on abundance for burned versus unburned treatments. Similarity is found between a few treatment sites.

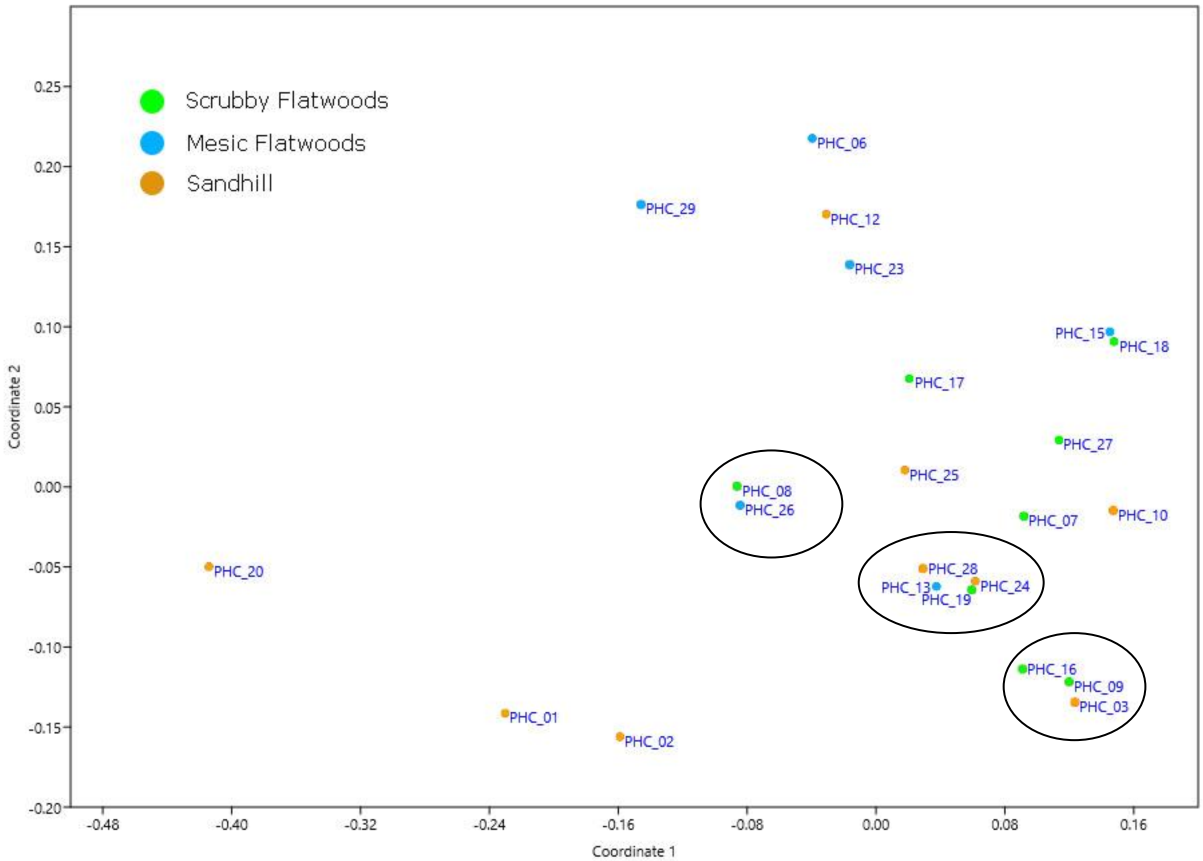

Figure 17. NMDS plot of transect similarity based on abundance between different communities. Similarity is found between a few treatment sites.

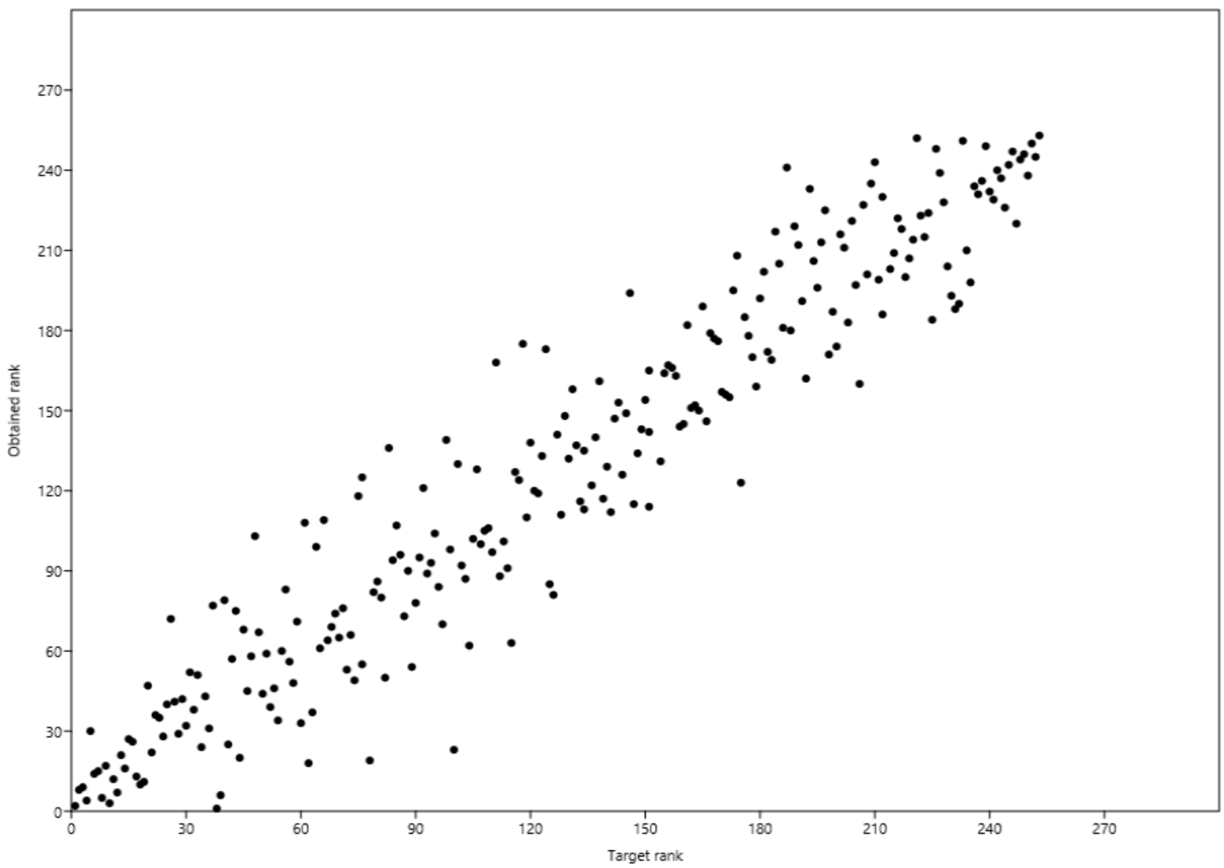

Figure 18. Shepard diagram of species similarity based on abundance. Stress value = 0.1519.

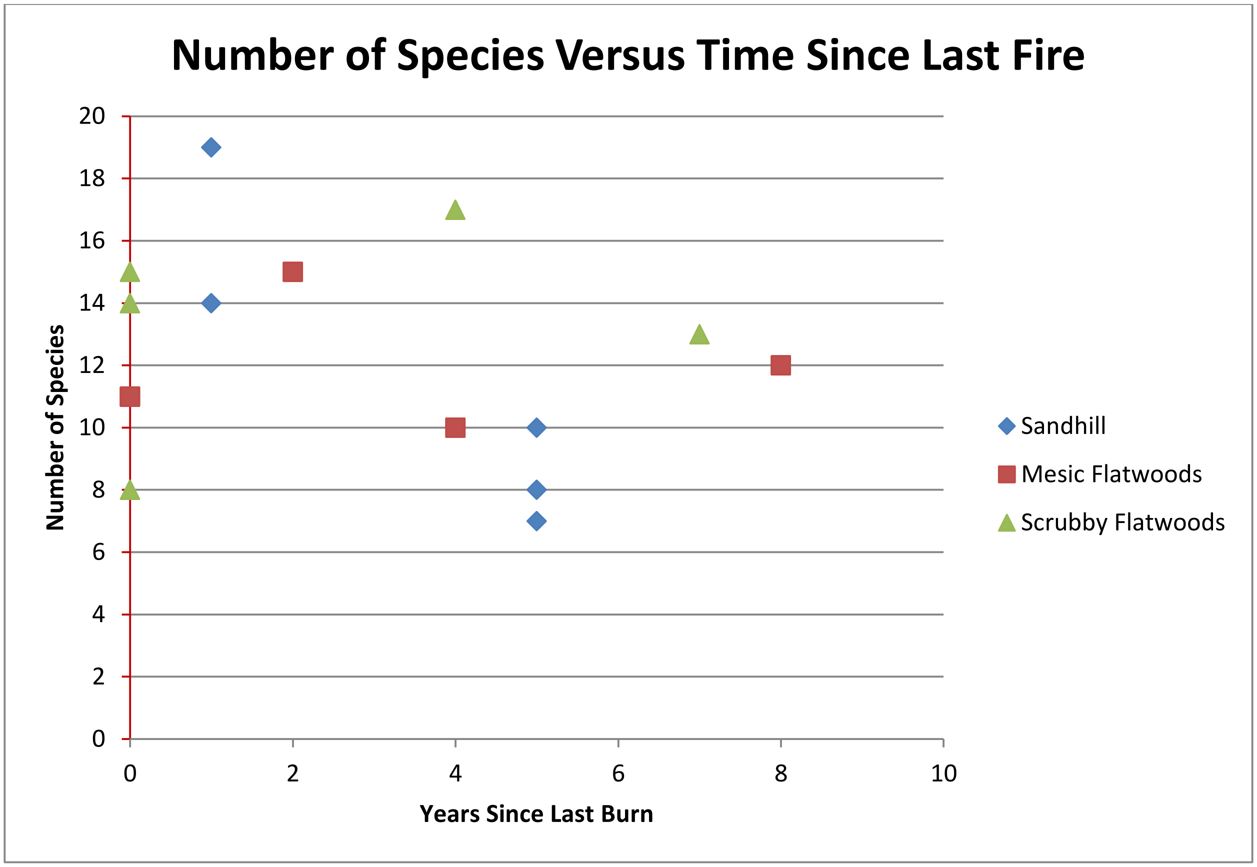

Figure 19. Number of species as a function of time since last fire (in years) for burned treatments by community.

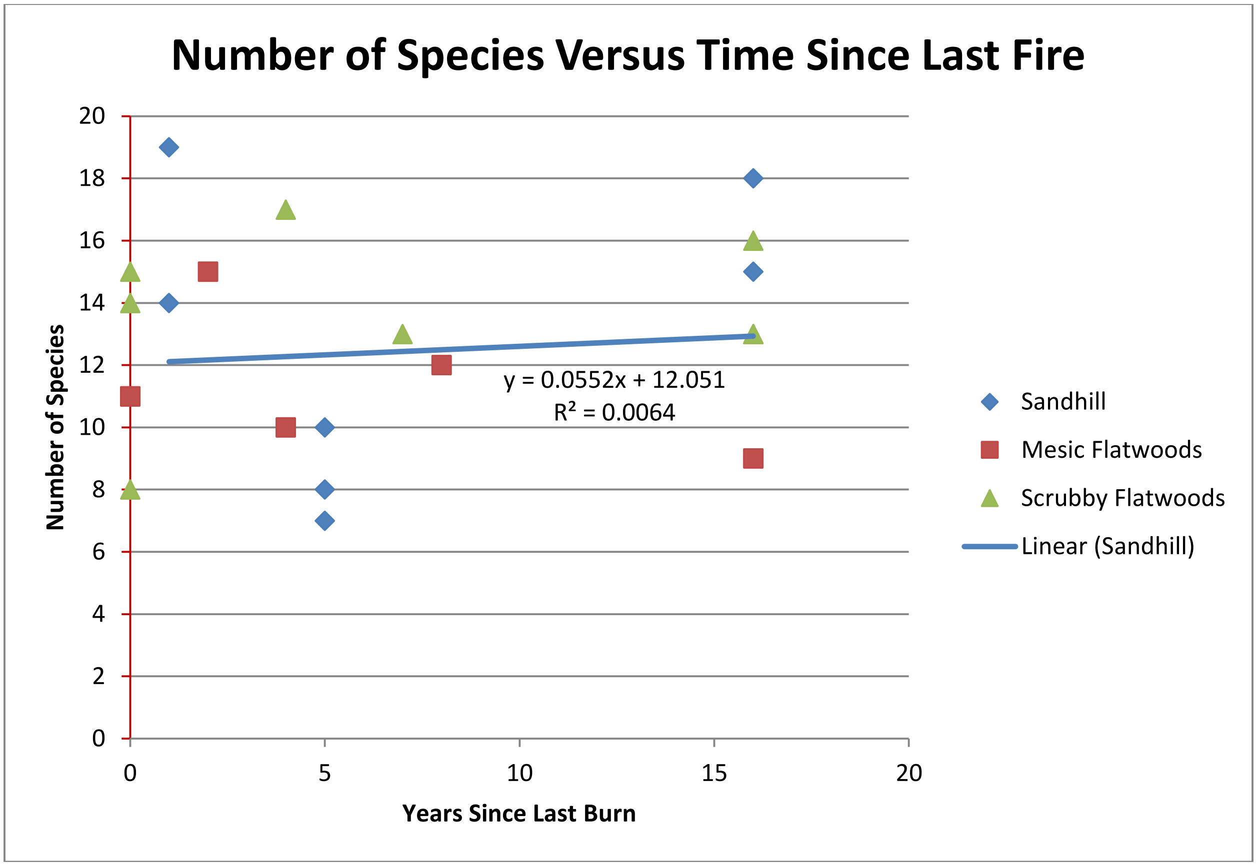

Figure 20. Number of species as a function of time since last fire (in years) for all treatments by community.

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Appendices

Appendix A: List of Plants Observed

Code	Scientific Name	Common Name
ANVI	*Andropogon virginicus*	Broomsedge Bluestem
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn
BAHA	*Baccharis halimifolia*	Eastern baccharis
BERA	*Befaria racemosa*	Tarflower
BUCI	*Bulbostylis ciliatifolia*	Capillary Hairsedge
CACO37	*Carphephorus corymbosus*	Coastal Plain Chaffhead
CLEV	*Clandina evansii*	Reindeer lichen
COFL	*Cornus florida*	Flowering Dogwood
DICHA2	*Dichanthelium acuminatum*	Rosette Grass
GADU	*Gaylussacia dumosa*	Huckleberry
GAEL	*Galactia elliottii*	Elliott's Milkpea
GESE	*Gelsimium sempervirens*	Yellow Jessamine
GOLA	*Gordonia lasianthus*	Loblolly Bay
HYCI	*Hypericum cistifolium*	Hypericum cistifolium
HYSP	*Hypericum spp.*	St Johns Wort spp
ILGL	*Ilex glabra*	Gallberry
ILOP	*Ilex opaca*	American Holly
ILVO	*Ilex vomitoria*	Yaupon Holly
KAHI2	*Kalmia hirsuta*	Hairy Wicky
LETO	*Lechea torreyi*	Piedmont Pinweed
LYFE	*Lyonia ferruginea*	Rusty Lyonia
LYLU	*Lyonia lucida*	Fetterbush
MYCE	*Myrica cerifera*	Wax Myrtle
PIEL	*Pinus elliottii*	Slash Pine
PIPA	*Pinus palustris*	Longleaf Pine
PISE	*Pinus serotina*	Pond Pine
PTAQ	*Pteridium aquilinum*	Bracken Fern
PTERO2	*Pterocaulon Elliott*	Blackroot
QUCH	*Quercus chapmanii*	Chapman's Oak
QUGE	*Quercus geminata*	Sand Live Oak
QULA2	*Quercus laevis*	Turkey Oak
QULA3	*Qurecus laurifolia*	Laurel Oak
QUMI	*Quercus minima*	Dwarf Live Oak
QUMY	*Quercus myrtifolia*	Myrtle Oak
QUPU	*Quercus pumila*	Runner Oak
RHSP	*Rhyncospora species*	Rhyncospora species
SABR	*Sabatia brevifolia*	Narrow Leaved Sabatia
SECA	*Seymeria cassioides*	Black-senna
SERE	*Serenoa repens*	Saw Palmetto
SMAU	*Smilax auriculata*	Earleaf Greenbriar
SOFI	*Solidago fistulosa*	Pine Barren Goldenrod
URDI	*Urtica dioica*	Stinging Nettle
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry
VAST	*Vaccinium staminium*	Deerberry
VIRO	*Vitis rotundifolia*	Muscadine

Appendix B: Transect Data Sheets

Transect:	**PHC-01**	Community	Sandhill
Date:	11-Aug-13	Burn Year	2008

Code	**Scientific Name**	**Common Name**	N_i **Count**
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	12
GAEL	*Galactia elliottii*	Elliott's Milkpea	20
LYLU	*Lyonia lucida*	Fetterbush	15
QUGE	*Quercus geminata*	Sand Live Oak	8
SERE	*Serenoa repens*	Saw Palmetto	6
SMAU	*Smilax auriculata*	Earleaf Greenbriar	2
ANVI	*Andropogon virginicus*	Broomsedge Bluestem	1
QULA2	*Quercus laevis*	Turkey Oak	8

Total Count:	72
Species Count:	8
Shannon-Weiner Diversity Index (H'):	1.8355
Simpson's Diversity Index (D):	5.9030

Transect:	**PHC-02**	Community	Sandhill
Date:	11-Aug-13	Burn Year	2008

Code	Scientific Name	Common Name	N_i Count
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	17
GAEL	*Galactia elliottii*	Elliott's Milkpea	4
LYLU	*Lyonia lucida*	Fetterbush	2
ANVI	*Andropogon virginicus*	Broomsedge Bluestem	6
SERE	*Serenoa repens*	Saw Palmetto	5
SMAU	*Smilax auriculata*	Earleaf Greenbriar	2
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	2

Total Count:	38
Species Count:	7
Shannon-Weiner Diversity Index (H'):	1.6200
Simpson's Diversity Index (D):	4.1353

Transect:	**PHC-03**	Community	Sandhill
Date:	7-Aug-13	Burn Year	2012
			N_i
Code	**Scientific Name**	**Common Name**	**Count**
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	26
BERA	*Befaria racemosa*	Tarflower	59
CACO37	*Carphephorus corymbosus*	Coastal Plain Chaffhead	5
DICHA2	*Dichanthelium acuminatum*	Rosette Grass	8
GAEL	*Galactia elliottii*	Elliott's Milkpea	4
ILGL	*Ilex glabra*	Gallberry	9
QUMY	*Quercus myrtifolia*	Myrtle Oak	123
SERE	*Serenoa repens*	Saw Palmetto	10
SMAU	*Smilax auriculata*	Earleaf Greenbriar	1
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	26
QUMI	*Quercus minima*	Dwarf Live Oak	33
LITE	*Liatris tenufolia*	Shortleaf Gayfeather	6
ANVI	*Andropogon virginicus*	Broomsedge Bluestem	7
UNKN11	*Plot 3, Rhyncospora or Fimbristylis*	Unknown	20
LETO	*Lechea torreyi*	Piedmont Pinweed	2
SECA	*Seymeria cassioides*	Black-senna	2
HYCI	*Hypericum cistifolium*	Hypericum cistifolium	5
PTERO2	*Pterocaulon Elliott*	Blackroot	2
URDI	*Urtica dioica*	Stinging Nettle	1

Total Count:	349
Species Count:	19
Shannon-Weiner Diversity Index (H'):	2.1680
Simpson's Diversity Index (D):	5.6458

Transect:	**PHC-06**	Community	Mesic Flatwoods
Date:	7-Aug-13	Burn Year	2009

Code	**Scientific Name**	**Common Name**	N_i **Count**
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	4
GAEL	*Galactia elliottii*	Elliott's Milkpea	2
ILGL	*Ilex glabra*	Gallberry	31
LYFE	*Lyonia ferruginea*	Rusty Lyonia	1
SERE	*Serenoa repens*	Saw Palmetto	23
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	36
GADU	*Gaylussacia dumosa*	Huckleberry	38
LYLU	*Lyonia lucida*	Fetterbush	196
PISE	*Pinus serotina*	Pond Pine	2
UNKN09	*woody, alt leaves, heavy tooth, cherry like*	Unknown	2

Total Count:	335
Species Count:	10
Shannon-Weiner Diversity Index (H'):	1.3663
Simpson's Diversity Index (D):	2.6427

Transect:	**PHC-07**	Community	Scrubby Flatwoods
Date:	26-Jul-13	Burn Year	2013

Code	**Scientific Name**	**Common Name**	**N_i Count**
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	1
BERA	*Befaria racemosa*	Tarflower	4
DICHA2	*Dichanthelium acuminatum*	Rosette Grass	4
GAEL	*Galactia elliottii*	Elliott's Milkpea	16
ILGL	*Ilex glabra*	Gallberry	17
ILOP	*Ilex opaca*	American Holly	1
LYFE	*Lyonia ferruginea*	Rusty Lyonia	20
MYCE	*Myrica cerifera*	Wax Myrtle	7
QUCH	*Quercus chapmanii*	Chapman's Oak	4
QUMY	*Quercus myrtifolia*	Myrtle Oak	83
SERE	*Serenoa repens*	Saw Palmetto	23
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	25
GADU	*Gaylussacia dumosa*	Huckleberry	15
HYSP	*Hypericum spp.*	St Johns Wort spp	6
BAHA	*Baccharis halimifolia*	Eastern baccharis	1

Total Count:	227
Species Count:	15
Shannon-Weiner Diversity Index (H'):	2.1059
Simpson's Diversity Index (D):	5.6240

Transect:	**PHC-08**	Community	Scrubby Flatwoods
Date:	26-Jul-13	Burn Year	2009

Code	**Scientific Name**	**Common Name**	**N_i Count**
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	19
BERA	*Befaria racemosa*	Tarflower	1
DICHA2	*Dichanthelium acuminatum*	Rosette Grass	14
GAEL	*Galactia elliottii*	Elliott's Milkpea	9
ILGL	*Ilex glabra*	Gallberry	7
LYFE	*Lyonia ferruginea*	Rusty Lyonia	16
LYLU	*Lyonia lucida*	Fetterbush	25
MYCE	*Myrica cerifera*	Wax Myrtle	3
QUCH	*Quercus chapmanii*	Chapman's Oak	1
QUMY	*Quercus myrtifolia*	Myrtle Oak	10
SERE	*Serenoa repens*	Saw Palmetto	19
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	26
QUGE	*Quercus geminata*	Sand Live Oak	67
KAHI2	*Kalmia hirsuta*	Hairy Wicky	8
GADU	*Gaylussacia dumosa*	Huckleberry	5
ANVI	*Andropogon virginicus*	Broomsedge Bluestem	5
HYSP	*Hypericum spp.*	St Johns Wort spp	3

Total Count:	238
Species Count:	17
Shannon-Weiner Diversity Index (H'):	2.3805
Simpson's Diversity Index (D):	7.9557

Transect:	**PHC-09**	Community	Scrubby Flatwoods
Date:	11-Aug-13	Burn Year	2006

Code	**Scientific Name**	**Common Name**	**N_i Count**
BERA	*Befaria racemosa*	Tarflower	7
GAEL	*Galactia elliottii*	Elliott's Milkpea	8
CACO37	*Carphephorus corymbosus*	Coastal Plain Chaffhead	3
ILOP	*Ilex opaca*	American Holly	16
QUMY	*Quercus myrtifolia*	Myrtle Oak	48
SERE	*Serenoa repens*	Saw Palmetto	7
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	19
QUGE	*Quercus geminata*	Sand Live Oak	1
DICHA2	*Dichanthelium acuminatum*	Rosette Grass	6
LYFE	*Lyonia ferruginea*	Rusty Lyonia	5
MYCE	*Myrica cerifera*	Wax Myrtle	1
PTAQ	*Pteridium aquilinum*	Bracken Fern	1
SMAU	*Smilax auriculata*	Earleaf Greenbriar	1

Total Count:	123
Species Count:	13
Shannon-Weiner Diversity Index (H'):	1.9496
Simpson's Diversity Index (D):	4.9459

Transect:	**PHC-10**	Community	Sandhill
Date:	13-Aug-13	Burn Year	2011

Code	Scientific Name	Common Name	N_i Count
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	10
GAEL	*Galactia elliottii*	Elliott's Milkpea	1
LYLU	*Lyonia lucida*	Fetterbush	1
GADU	*Gaylussacia dumosa*	Huckleberry	27
QUMY	*Quercus myrtifolia*	Myrtle Oak	77
SERE	*Serenoa repens*	Saw Palmetto	10
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	17
HYSP	*Hypericum spp.*	St Johns Wort spp	3
PIPA	*Pinus palustris*	Longleaf Pine	3
CLEV	*Clandina evansii*	Reindeer lichen	19
QUPU	*Quercus pumila*	Runner Oak	2
ANVI	*Andropogon virginicus*	Broomsedge Bluestem	2

Total Count:	172
Species Count:	12
Shannon-Weiner Diversity Index (H'):	1.7580
Simpson's Diversity Index (D):	3.9940

Transect:	**PHC-12**	Community	Sandhill
Date:	13-Aug-13	Burn Year	

Code	Scientific Name	Common Name	N_i Count
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	13
GAEL	*Galactia elliottii*	Elliott's Milkpea	1
ILGL	*Ilex glabra*	Gallberry	51
ILOP	*Ilex opaca*	American Holly	1
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	16
QUGE	*Quercus geminata*	Sand Live Oak	4
DICHA2	*Dichanthelium acuminatum*	Rosette Grass	7
GADU	*Gaylussacia dumosa*	Huckleberry	11
GESE	*Gelsimium sempervirens*	Yellow Jessamine	25
PIPA	*Pinus palustris*	Longleaf Pine	1
PTAQ	*Pteridium aquilinum*	Bracken Fern	2
QUCH	*Quercus chapmanii*	Chapman's Oak	1
QULA3	*Qurecus laurifolia*	Laurel Oak	3
SMAU	*Smilax auriculata*	Earleaf Greenbriar	6
QUPU	*Quercus pumila*	Runner Oak	21
ANVI	*Andropogon virginicus*	Broomsedge Bluestem	4
VAST	*Vaccinium staminium*	Deerberry	1
ILVO	*Ilex vomitoria*	Yaupon Holly	3

Total Count:	171
Species Count:	18
Shannon-Weiner Diversity Index (H'):	2.2618
Simpson's Diversity Index (D):	6.9446

Transect:	**PHC-13**	Community	Mesic Flatwoods
Date:	13-Aug-13	Burn Year	

Code	**Scientific Name**	**Common Name**	N_i **Count**
BERA	*Befaria racemosa*	Tarflower	20
GADU	*Gaylussacia dumosa*	Huckleberry	2
QUMY	*Quercus myrtifolia*	Myrtle Oak	1
SERE	*Serenoa repens*	Saw Palmetto	27
PISE	*Pinus serotina*	Pond Pine	5
QULA3	*Qurecus laurifolia*	Laurel Oak	1
VIRO	*Vitis rotundifolia*	Muscadine	6
GOLA	*Gordonia lasianthus*	Loblolly Bay	1
COFL	*Cornus florida*	Flowering Dogwood	1

Total Count:	64
Species Count:	9
Shannon-Weiner Diversity Index (H'):	1.5169
Simpson's Diversity Index (D):	3.5556

Transect:	**PHC-15**	Community	Mesic Flatwoods
Date:	11-Aug-13	Burn Year	2005

Code	Scientific Name	Common Name	N_i Count
GADU	*Gaylussacia dumosa*	Huckleberry	24
ILGL	*Ilex glabra*	Gallberry	26
ILOP	*Ilex opaca*	American Holly	1
MYCE	*Myrica cerifera*	Wax Myrtle	3
QULA3	*Qurecus laurifolia*	Laurel Oak	3
QUMY	*Quercus myrtifolia*	Myrtle Oak	31
SERE	*Serenoa repens*	Saw Palmetto	6
SMAU	*Smilax auriculata*	Earleaf Greenbriar	2
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	15
GESE	*Gelsimium sempervirens*	Yellow Jessamine	5
VIRO	*Vitis rotundifolia*	Muscadine	7
COFL	*Cornus florida*	Flowering Dogwood	1

Total Count:	124
Species Count:	12
Shannon-Weiner Diversity Index (H'):	2.0102
Simpson's Diversity Index (D):	6.2304

Transect:	**PHC-16**	Community	Scrubby Flatwoods
Date:	2-Aug-13	Burn Year	2013

Code	**Scientific Name**	**Common Name**	**N_i Count**
BERA	*Befaria racemosa*	Tarflower	18
GAEL	*Galactia elliottii*	Elliott's Milkpea	27
ILGL	*Ilex glabra*	Gallberry	8
ILOP	*Ilex opaca*	American Holly	1
QUMY	*Quercus myrtifolia*	Myrtle Oak	55
SERE	*Serenoa repens*	Saw Palmetto	23
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	7
QUGE	*Quercus geminata*	Sand Live Oak	1

Total Count:	140
Species Count:	8
Shannon-Weiner Diversity Index (H'):	1.6289
Simpson's Diversity Index (D):	4.2471

Transect:	**PHC-17**	Community	Scrubby Flatwoods
Date:	2-Aug-13	Burn Year	

*road located north of plot, so plot direction is south from beginning point

Code	Scientific Name	Common Name	N_i Count
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	2
BERA	*Befaria racemosa*	Tarflower	12
GAEL	*Galactia elliottii*	Elliott's Milkpea	1
ILGL	*Ilex glabra*	Gallberry	17
LYFE	*Lyonia ferruginea*	Rusty Lyonia	3
QUCH	*Quercus chapmanii*	Chapman's Oak	9
QUMY	*Quercus myrtifolia*	Myrtle Oak	11
SERE	*Serenoa repens*	Saw Palmetto	30
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	22
GADU	*Gaylussacia dumosa*	Huckleberry	35
LYLU	*Lyonia lucida*	Fetterbush	20
ILVO	*Ilex vomitoria*	Yaupon Holly	2
PTAQ	*Pteridium aquilinum*	Bracken Fern	3

Total Count:	167
Species Count:	13
Shannon-Weiner Diversity Index (H'):	2.1965
Simpson's Diversity Index (D):	7.9115

Transect:	**PHC-18**	Community	Scrubby Flatwoods
Date:	1-Oct-13	Burn Year	

Code	**Scientific Name**	**Common Name**	**N_i Count**
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	11
BERA	*Befaria racemosa*	Tarflower	14
CLEV	*Clandina evansii*	Reindeer lichen	16
DICHA2	*Dichanthelium acuminatum*	Rosette Grass	1
GAEL	*Galactia elliottii*	Elliott's Milkpea	2
HYSP	*Hypericum spp.*	St Johns Wort spp	1
ILOP	*Ilex opaca*	American Holly	1
QUCH	*Quercus chapmanii*	Chapman's Oak	11
GADU	*Gaylussacia dumosa*	Huckleberry	101
QUMY	*Quercus myrtifolia*	Myrtle Oak	17
SERE	*Serenoa repens*	Saw Palmetto	16
SMAU	*Smilax auriculata*	Earleaf Greenbriar	1
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	8

Total Count:	200
Species Count:	13
Shannon-Weiner Diversity Index (H'):	1.7446
Simpson's Diversity Index (D):	3.5184

Transect:	**PHC-19**	Community	Scrubby Flatwoods
Date:	26-Jul-13	Burn Year	2013

Code	**Scientific Name**	**Common Name**	**N_i Count**
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	3
BERA	*Befaria racemosa*	Tarflower	3
GAEL	*Galactia elliottii*	Elliott's Milkpea	24
ILGL	*Ilex glabra*	Gallberry	8
ILOP	*Ilex opaca*	American Holly	1
LYFE	*Lyonia ferruginea*	Rusty Lyonia	40
MYCE	*Myrica cerifera*	Wax Myrtle	1
QUCH	*Quercus chapmanii*	Chapman's Oak	4
QUMY	*Quercus myrtifolia*	Myrtle Oak	56
SERE	*Serenoa repens*	Saw Palmetto	25
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	44
GADU	*Gaylussacia dumosa*	Huckleberry	9
HYSP	*Hypericum spp.*	St Johns Wort spp	5
UNKN02	*Rhyncospora spp*	need flower	2

Total Count:	225
Species Count:	14
Shannon-Weiner Diversity Index (H'):	2.0641
Simpson's Diversity Index (D):	6.4138

Transect:	**PHC-20**	Community	Sandhill
Date:	11-Aug-13	Burn Year	2008

Code	**Scientific Name**	**Common Name**	**N_i Count**
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	4
GAEL	*Galactia elliottii*	Elliott's Milkpea	1
PIPA	*Pinus palustris*	Longleaf Pine	1
QUCH	*Quercus chapmanii*	Chapman's Oak	7
SERE	*Serenoa repens*	Saw Palmetto	1
SMAU	*Smilax auriculata*	Earleaf Greenbriar	1
QUGE	*Quercus geminata*	Sand Live Oak	6
QULA2	*Quercus laevis*	Turkey Oak	4
VAST	*Vaccinium staminium*	Deerberry	1
QUPU	*Quercus pumila*	Runner Oak	1

Total Count:	27
Species Count:	10
Shannon-Weiner Diversity Index (H'):	1.9824
Simpson's Diversity Index (D):	7.3125

Transect:	**PHC-23**	Community	Mesic Flatwoods
Date:	25-Aug-13	Burn Year	2013

Code	**Scientific Name**	**Common Name**	N_i **Count**
BERA	*Befaria racemosa*	Tarflower	5
GADU	*Gaylussacia dumosa*	Huckleberry	48
GAEL	*Galactia elliottii*	Elliott's Milkpea	3
LYFE	*Lyonia ferruginea*	Rusty Lyonia	14
QUCH	*Quercus chapmanii*	Chapman's Oak	22
SERE	*Serenoa repens*	Saw Palmetto	29
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	43
ILGL	*Ilex glabra*	Gallberry	63
HYSP	*Hypericum spp.*	St Johns Wort spp	5
LYLU	*Lyonia lucida*	Fetterbush	27
PIEL	*Pinus elliottii*	Slash Pine	3

Total Count:	262
Species Count:	11
Shannon-Weiner Diversity Index (H'):	2.0460
Simpson's Diversity Index (D):	6.7186

Transect:	**PHC-24**	Community	Sandhill
Date:	25-Aug-13	Burn Year	

Code	**Scientific Name**	**Common Name**	**N_i Count**
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	3
BERA	*Befaria racemosa*	Tarflower	25
GAEL	*Galactia elliottii*	Elliott's Milkpea	5
LYLU	*Lyonia lucida*	Fetterbush	1
MYCE	*Myrica cerifera*	Wax Myrtle	2
QUCH	*Quercus chapmanii*	Chapman's Oak	30
QUMY	*Quercus myrtifolia*	Myrtle Oak	33
SERE	*Serenoa repens*	Saw Palmetto	19
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	20

Total Count:	138
Species Count:	9
Shannon-Weiner Diversity Index (H'):	1.8368
Simpson's Diversity Index (D):	5.7711

Transect:	**PHC-25**	Community	Sandhill
Date:	6-Sep-13	Burn Year	

Code	Scientific Name	Common Name	N_i Count
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	12
BERA	*Befaria racemosa*	Tarflower	9
DICHA2	*Dichanthelium acuminatum*	Rosette Grass	7
GAEL	*Galactia elliottii*	Elliott's Milkpea	5
LYFE	*Lyonia ferruginea*	Rusty Lyonia	3
MYCE	*Myrica cerifera*	Wax Myrtle	12
QUCH	*Quercus chapmanii*	Chapman's Oak	7
QUMY	*Quercus myrtifolia*	Myrtle Oak	27
SERE	*Serenoa repens*	Saw Palmetto	21
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	34
GADU	*Gaylussacia dumosa*	Huckleberry	25
HYSP	*Hypericum spp.*	St Johns Wort spp	8
LYLU	*Lyonia lucida*	Fetterbush	6
ANVI	*Andropogon virginicus*	Broomsedge Bluestem	2
QUGE	*Quercus geminata*	Sand Live Oak	22

Total Count:	200
Species Count:	15
Shannon-Weiner Diversity Index (H'):	2.4580
Simpson's Diversity Index (D):	10.3646

Transect:	**PHC-26**	Community	Mesic Flatwood s
Date:	6-Sep-13	Burn Year	2011

Code	**Scientific Name**	**Common Name**	**N_i Count**
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	4
BERA	*Befaria racemosa*	Tarflower	36
GAEL	*Galactia elliottii*	Elliott's Milkpea	16
LYLU	*Lyonia lucida*	Fetterbush	39
QUCH	*Quercus chapmanii*	Chapman's Oak	15
QUMY	*Quercus myrtifolia*	Myrtle Oak	7
SERE	*Serenoa repens*	Saw Palmetto	31
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	22
DICHA2	*Dichanthelium acuminatum*	Rosette Grass	31
HYSP	*Hypericum spp.*	St Johns Wort spp	29
LYFE	*Lyonia ferruginea*	Rusty Lyonia	2
ANVI	*Andropogon virginicus*	Broomsedge Bluestem	1
QUGE	*Quercus geminata*	Sand Live Oak	3
SABR	*Sabatia brevifolia*	Narrow Leaved Sabatia	4
SOFI	*Solidago fistulosa*	Pine Barren Goldenrod	8

Total Count:	248
Species Count:	15
Shannon-Weiner Diversity Index (H'):	2.3624
Simpson's Diversity Index (D):	9.4882

Transect:	**PHC-27**	Community	Scrubby Flatwood s
Date:	25-Aug-13	Burn Year	

Code	Scientific Name	Common Name	N_i Count
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	34
BERA	*Befaria racemosa*	Tarflower	11
GAEL	*Galactia elliottii*	Elliott's Milkpea	1
LYLU	*Lyonia lucida*	Fetterbush	4
GADU	*Gaylussacia dumosa*	Huckleberry	58
QUCH	*Quercus chapmanii*	Chapman's Oak	32
QUMY	*Quercus myrtifolia*	Myrtle Oak	61
SERE	*Serenoa repens*	Saw Palmetto	17
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	17
HYSP	*Hypericum spp.*	St Johns Wort spp	20
LYFE	*Lyonia ferruginea*	Rusty Lyonia	2
ANVI	*Andropogon virginicus*	Broomsedge Bluestem	1
QUGE	*Quercus geminata*	Sand Live Oak	26
ILOP	*Ilex opaca*	American Holly	2
PTAQ	*Pteridium aquilinum*	Bracken Fern	4
CLEV	*Clandina evansii*	Reindeer lichen	15

Total Count:	305
Species Count:	16
Shannon-Weiner Diversity Index (H'):	2.3141
Simpson's Diversity Index (D):	8.4276

Transect:	**PHC-28**	Community	Sandhill
Date:	2-Nov-13	Burn Year	2012

Code	Scientific Name	Common Name	N_i Count
ANVI	*Andropogon virginicus*	Broomsedge Bluestem	7
ARBE5	*Aristida beyrichiana*	Beyrich Threeawn	37
CACO37	*Carphephorus corymbosus*	Coastal Plain Chaffhead	6
DICHA2	*Dichanthelium acuminatum*	Rosette Grass	3
GADU	*Gaylussacia dumosa*	Huckleberry	1
HYCI	*Hypericum cistifolium*	Hypericum cistifolium	2
LYLU	*Lyonia lucida*	Fetterbush	47
QUCH	*Quercus chapmanii*	Chapman's Oak	3
QULA2	*Quercus laevis*	Turkey Oak	1
QUMY	*Quercus myrtifolia*	Myrtle Oak	62
SERE	*Serenoa repens*	Saw Palmetto	5
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	36
VAST	*Vaccinium staminium*	Deerberry	1
SECA	*Seymeria cassioides*	Black-senna	11

Total Count:	222
Species Count:	14
Shannon-Weiner Diversity Index (H'):	1.9512
Simpson's Diversity Index (D):	5.6058

Transect:	**PHC-29**	Community	Mesic Flatwoods
Date:	2-Nov-13	Burn Year	2011

Code	Scientific Name	Common Name	N_i Count
BERA	*Befaria racemosa*	Tarflower	4
DICHA2	*Dichanthelium acuminatum*	Rosette Grass	46
HYCI	*Hypericum cistifolium*	Hypericum cistifolium	2
ILGL	*Ilex glabra*	Gallberry	66
LYLU	*Lyonia lucida*	Fetterbush	87
MYCE	*Myrica cerifera*	Wax Myrtle	2
QUCH	*Quercus chapmanii*	Chapman's Oak	6
QUGE	*Quercus geminata*	Sand Live Oak	15
QUMY	*Quercus myrtifolia*	Myrtle Oak	1
SERE	*Serenoa repens*	Saw Palmetto	16
VAMY3	*Vaccinium myrsinites*	Shiny Blueberry	11

Total Count:	256
Species Count:	11
Shannon-Weiner Diversity Index (H'):	1.7499
Simpson's Diversity Index (D):	4.5183

Appendix C: Transect Images

Transect PHC_01

Transect PHC_02

Transect PHC_06

Transect PHC_07

Transect PHC_08

Transect PHC_09

Transect PHC_10

Transect PHC_12

Transect PHC_16

Transect PHC_19

Transect PHC_20

Transect PHC_21

Transect PHC_22

Transect PHC_23

Transect PHC_24

Transect PHC_25

Transect PHC_26

Transect PHC_27

Vita

Peter Maholland is currently an ecologist with Atkins Global and has been involved with fire and restoration ecology for nearly twenty years. Prior to his current position, he was a Park Services Specialist with Talbot Islands State Parks, the Nevada State Parks resource specialist on the Nevada Tahoe Resource Team, managing natural resources in the 14,000-acre Lake Tahoe Nevada State Park. Before that, he was the coordinator of both the Forest Habitat Enhancement and Wildlife Habitat Enhancement programs for the California Tahoe Conservancy, overseeing the implementation of a variety of forest and river restoration projects in the Lake Tahoe basin. He received his B.S. degree in Conservation Biology from the University of Nevada, Reno.