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THE EFFECTS OF URBANIZATION ON THE STRUCTURE, QUALITY, AND DIVERSITY OF CYPRESS
PLANT COMMUNITIES IN CENTRAL FLORIDA

by

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B.A. University of Central Florida, 2002

A thesis submitted in partial fulfillment of the requirements
for the degree of Master of Science
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ABSTRACT

The integrity of wetland ecosystems is largely determined by hydrological functionality, degree of connectivity to like ecosystems, and permeability to external influence. Land use changes in upland areas adjacent to wetland ecosystems may influence hydrology and connectivity while introducing novel biotic and abiotic materials. There is an increasing trend toward the use of remote assessment techniques to determine the degree of impact of external influences on adjacent wetlands. Remote assessment and predictive capabilities are provided by indices such as the Landscape Development Intensity Index (LDI) (Brown and Vivas 2005) which may be beneficial in determining site condition, and which have the added benefit of providing a quantitative gradient of human impact. This study assessed the predictive ability of the LDI in cypress ecosystems, by testing its correlations with plant community metrics including an index of floral quality calculated using coefficients of conservatism (CC)(Cohen et al. 2004), plant species diversity, and fluctuation in community composition assessed by changes in the wetland status and native status of component plant species. LDI was also compared against an independent measure of disturbance which was used to construct an *a priori* disturbance gradient. Overall, diversity measures showed little correlation with any of the disturbance indices, while CC scores were significantly correlated. Models were constructed in an attempt to explain each of the variables of plant community response to development in the surrounding landscape. The length of time since the development of the land adjacent to the cypress domes was a predictor of plant community response only when included in models with other variables. LDI was the strongest predictor in all models except where increases in

land use associated with hydrological changes helped predict or better predicted proportions of exotic and upland species.

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INTRODUCTION

Land Use and Urbanization

The loss of species diversity is a problem frequently attributed to disturbance caused by human development. However, the rate of loss worldwide remains relatively unknown as the growth of developing nations and the transfer of agricultural technologies to these nations makes predictions tenuous at best (Van Vuuren et al. 2006). The spread of agriculture and urbanization initiates a complex and potentially unpredictable feedback mechanism caused by changes in landscape scale processes, such as drainage and soil retention (Claessens et al. 2009). Some studies suggest that over the next several decades, land use change and urbanization will contribute more to species loss than will climate change (Sala et al. 2000, Van Vuuren et al. 2006). While the problems of declining biodiversity and biotic homogenization are occurring on a global scale, Florida is at the frontline of urbanization in industrialized nations. Over the last 14 to 18 years, more than 703,000 ha of Florida's natural lands have been converted to agricultural uses and more than 611, 000 ha have been converted to urban uses. Likewise, more than 355,000 ha of agricultural lands have been converted to urban uses. Specifically, more than 243,000 ha of pinelands have been converted (uplands are being converted at the greatest rate), and 25% of the remaining dry prairie was converted and lost during this time (Kautz et al. 2007).

While the origins of human urbanization reach far into the past, the science of measuring the urban to rural gradient is relatively new. There are several issues which must be carefully

addressed in these measurements. McDonnell and Pickett (1990) emphasize the importance of assessing the urban to rural gradient by looking at each link in the chain between urbanization as a primary cause and species change as an ultimate effect. They assert that in a complete study of the urban-rural interface, urbanization should itself be quantified, along with its indirect environmental impacts, and the direct effects it has on a natural system. Most studies include one or two of these metrics only which may be likened to studying just one part of an equation and inferring knowledge of the other parts. Urbanization impact studies must include measurement or assessment of the urban and the natural parts of the equation. Another potential pitfall is the failure to measure a continuous quantitative gradient of disturbance. In a literature review of ecological studies on urban gradients, McDonnell and Hahs (2008) found that of 300 papers reviewed, only five fully quantified an urbanization gradient, further demonstrating the paucity of work on this point.

There is also frequently confusion about whether direct or indirect measures of human impact are being used (McDonnell and Hahs 2008). Land use is an indirect measure of human disturbance, though it has been associated with detrimental changes in natural systems in a host of studies employing a variety of study organisms. Land use may also be a good proxy or dummy variable (McDonnell and Pickett 1990) for more direct measurements of human impact. It has, for example, been found to be a strong predictor of the biological and structural integrity of adjacent natural lands (Allan et al. 1997). Land use differences also coincide with differences in plant community composition (Galatowitsch et al. 2000). For example, non-native species

abundance is often correlated with road density. Wetlands receiving surface water from agricultural and urban watersheds tend to have weedy and invasive plant species, low species richness, and high numbers of exotic species (Zedler and Kerscher, 2004). Native plant species richness has also been found to be negatively correlated with increases in the amount of impervious surface and urbanization, while edge effects and fragmentation lead to increases in the abundance of non-native species in forests (Burton and Samuelson 2008). Species composition also shifts with changes in land use, as pioneer species begin to dominate increasingly urbanized landscapes (Burton and Samuelson 2008). Tasser and Tappeiner (2002) found highly specific associations between vegetation types and land use patterns and were able to ascribe vegetative communities to specific land use types. They also found that at landscape scales, the distribution of species was more heterogeneous on grazed lands than on lands managed for crop production. They conclude that land use may be the most important predictor of plant community dynamics. Vallet et al. (2008) found significant differences between the species composition of urban and rural forests. They theorize that these differences are due to variability in soil chemistry and the differential ability of species to disperse through the landscapes. Intensive land use has also been associated with changes in water quality including increases in the concentrations of nutrients, heavy metals, sediment, and chemicals such as sodium and sulfates (Tong and Chen 2002, Lenat and Crawford 1994). Likewise, enrichment of runoff from agricultural lands leads to eutrophication and a decrease in levels of dissolved oxygen in nearby wetlands (Mitsch and Gosselink 2000). Finally, intensive land use and urbanization, when viewed at the largest spatial scales, lead to homogenization of

the landscape and the species it contains. This may be a problem of global significance as diversity drops worldwide.

Wetland Ecology

Hydrology and hydroperiod may be the most important factors determining wetland integrity.

Hydrology is largely a determinant of the biotic and abiotic processes occurring within wetlands and the biota in turn frequently shape and impact other wetland processes (Mitch and Gosselink 2000). Wetlands play an important role in the filtering of nutrients and sediments from adjacent areas due to their hydrological linkages with these areas. Their ability to perform these services is severely diminished once they are drained and even minor changes in hydrology may lead to increases in local flooding as wetlands lose their ability to mitigate the impacts of severe storm events (Hunter et al. 2008, Young et al. 1995). Studies have also shown that shifts in the ability of wetlands to act as a landscape sink for nutrients and sediments can cause the quality of water in adjacent aquatic systems to decline (Detenbeck et al. 1999, Mitch and Gosselink 2000).

Wetlands are highly variable systems with regard to their biogeochemical cycling. Some wetlands may have mineral soils while others have primarily organic soils; some may be nutrient poor while others are nutrient rich; some may act as a sink for nutrients and sediments from the surrounding landscape while others may actually be a source for downstream systems. Wetlands are also variably influenced by their hydrological sources. Some wetlands

receive most of their water input from precipitation, while others receive inputs from surface flow or groundwater. Likewise, wetlands may have a greater or lesser ability to act as an area of groundwater recharge. Most wetlands can receive nutrient and chemical influxes from the surrounding landscape; though as Mitch and Gosselink (2000) point out, wetlands can act as a sink for nutrients and chemicals for only so long, before they become a source of these materials for other connected systems. They may act as stabilizers of local climate and chemical cycling, but anthropogenic disturbances can cause these systems, in turn, to become unstable.

Wetland ecosystems are also particularly susceptible to invasion by non-native and low quality plant species (Zedler and Kerscher 2004). This is due to a synergistic effect that results from certain characteristics of wetlands which increase their vulnerability to invasion and characteristics of wetland plant species which make them strong invaders. As wetlands are often connected to the surrounding landscape by surface water runoff in a way that terrestrial systems are not, they may be landscape sinks; collecting materials from surrounding areas.

Wetlands also are often subject to higher levels of natural disturbance, with water flow creating canopy gaps, disturbed soils, and opportunities for species colonization. Anthropogenic disturbances only compound these issues leading to nutrient enrichment of water and soils and altered hydrological regimes. Likewise, fire suppression and the alteration of hydrology can lead to a synergistic effect promoting invasion of species atypical of the base community composition (Knickerbocker et al. 2009). Wetland plant species must adapt to a number of

natural stresses including reduced levels of light, potential saline conditions, the mechanical forces of waves and water, oxygen poor soils, low levels of available nutrients in the soil and, if submerged, the inability to photosynthesize. In spite of this adversity, wetland plant species are often widely distributed, showing low levels of endemism, and a great dispersal ability. Santamaria (2002) gives several explanations for this phenomenon. While it has long been thought that wetlands were relatively homogenous, they actually retain a degree of small scale heterogeneity; accounting, in part, for the distribution of similar species across larger spatial scales. Likewise, plants found in wetlands have the ability to disperse widely and abundantly using clonal propagation and great phenotypic plasticity allowing them to colonize more readily. This plasticity also leads to a lower degree of genetic differentiation, leading to fewer genera per family in strictly aquatic families than in terrestrial families. Wetland plants also use a number of adaptive strategies, such as specialized cells, to cope with environmental stress. All of these features which allow wetland plant species to adapt to high-stress environments also permit them to become super invaders. Some of the world's most invasive species are aquatic plants (Zedler and Kerscher, 2004) such as *Caulerpa taxifolia*, *Eichornia crassipies*, and *Hydrilla verticilata*.

Cypress Ecosystems

Cypress swamps cover a large area across the southeastern United States, as far west as Texas and into parts of the north including Illinois and Missouri, following the Mississippi floodplain. Their hydrology is highly variable from swamp to swamp, though the wet seasons seem to be

predominantly summer and winter, and the driest seasons are spring and fall (Mitch and Gosselink 2000, Ewel and Odum 1984). The primary water inputs into cypress domes are through surface flow from the surrounding landscape, and precipitation. Water is often held in domes for long periods, not percolating into the groundwater due to the fact that they are frequently perched on top of a clay layer or hardpan. Water is lost most significantly by lateral movement into the soils of the surrounding landscape and through evapotranspiration (Mitch and Gosselink 2000, Riekerk and Korhnek 2000).

There is a great diversity of species that thrives in the understory of cypress swamps, varying according to soil, light, climate and hydrological conditions. The dominant tree species is the characteristic *Taxodium*. There is much debate as to whether the two types of *Taxodium* which predominate in the southeast are distinct species or are varieties of the species *T. distichum*. Some would call Pond cypress, found predominantly in still, acidic wetlands, *T. ascendens*, while others would call it *T. distichum* var. *nutans*. All refer to bald cypress, which thrives in flowing water systems such as riparian swamps, *T. distichum*. For the sake of this study, all species nomenclature follows Wunderlin and Hansen (2003), which distinguishes these types as two distinct species, *Taxodium distichum* and *Taxodium ascendens*. The only exception to this is the observation and cataloging of a possible hybridization between the species, or what would appear to be hybridization, as individual trees exhibit the leaf characteristics of both species. Some believe that the difference in leaf structure is simply the construction by the tree of sun vs. shade leaves which can exhibit a high amount of morphological distinctiveness in other

species. One researcher working in cypress domes (McCauley, pers. comm.), observed many specimens whose leaves exhibited a sharp contrast in morphology exactly at the line where the trees fall from sunshine into shade. While no genetic differentiation has yet been made, the question is still open to debate.

In central Florida *T. ascendens* is accompanied in cypress ponds by *Nyssa sylvatica* var. *biflora*, *Myrica cerifera*, *Magnolia virginiana*, *Persea palustris*, *Acer rubrum*, *Liquidambar styraciflua*, and occasional pine and oak species. Many domes have an understory of *Lyonia lucida*, *Cephalanthus occidentalis* and *Hypericum fasciculatum*. One excellent source distinguishes between cypress ponds and cypress savannahs (Kirkman et al. 2000). While *Taxodium ascendens* was found in both cypress ponds and savannahs within my study domes, the shrub and herbaceous understory varied dramatically between the two. *Hypericum fasciculatum* dominated open savannah-like settings while *Lyonia lucida*, or even small shrubby *Myrica* was found most abundantly in ponds. Cypress savannahs were also characterized by a much larger proportion of graminoid species while herbaceous forbs and ferns were in greater abundance in ponds. Most common in these Seminole and Orange county ponds were the fern species *Woodwardia virginica* and *Blechnum serrulatum*. Forbs included species of *Ludwigia*, *Polygonum*, *Sagittaria*, *Eriocaulon*, and more. Submerged or floating aquatic species included, most commonly, *Utricularia* spp., *Proserpinaca* spp., *Lemna minor* and *Azolla*. Species of *Xyris*, *Juncus*, and *Panicum* were also exceptionally common in all ponds studied, along with a number of bryophytes. Cypress savannahs were found along the edges of domes having somewhat

drier soils and sparser tree cover. They occasionally contained carnivorous plants such as *Drosera* and *Sarracenia*, though each time these fell outside of study plots. Also highly characteristic in savannahs were the forbs *Oxypolis filiformis* and *Bigloweia nudata*. Non-native species were much more common in cypress ponds than savannahs, and even in ponds, these species remained on the outside edges usually not penetrating the deeper wetter areas, that are indicative of still cypress basins. However, occasionally domes long surrounded by heavily developed landscapes, with a severely altered hydrology, seemed to have non-native species throughout the dome.

Assessing Community Level Integrity

Diversity

There is some debate as to the effects of diversity on ecosystem processes such as whether diversity influences community or population stability, how it is connected with the invasibility of communities by non-native species, whether community composition or diversity is more directly responsible for biogeochemical cycling, and what the current issues of relevance are in biodiversity studies from a policy and management perspective. As described in Maguuran (1988), the study of diversity can be roughly divided into richness: the number of species present in a given area; and evenness: the “equitability” of those species. There is a grand array of indices for calculating these two aspects of diversity. These indices attempt to measure or model simple species richness, species abundance, or a combination of richness and abundance.

Other diversity measures include those occurring on larger scales such as spatial and temporal landscape heterogeneity. Spatial heterogeneity may be sampled by examining the rate of increase in the number of species per area. For example, cypress domes with a flatter curve should have a greater heterogeneity of microhabitats than those with a steeper curve, where most of the community wide diversity will be encountered in a short period of sampling effort. Temporal heterogeneity may be captured in sites with repeated visits made through time. This heterogeneity can be examined at any spatial scale, and fluctuations may vary across spatial scales, but heterogeneity may be dampened at larger spatial scales. Finally, community diversity may be measured according to life history characteristics including assessments of the diversity of wetland status (i.e., facultative vs. obligate species), and longevity, by recording which species are annuals / biennials / perennials, or by growth form. Other measures may include guild information as a metric of diversity

Tilman (1999) neatly explores several questions regarding diversity and provides a foundation for the understanding of biodiversity using a combination of modeling and field studies. It can be demonstrated through a series of equations that diversity is directly responsible for the stability of a community through time. This is in line with what Tilman (1999) calls the “Portfolio Effect”, which is the well known principle in economic investing that stability or security is attained through diversification of the investment portfolio. Two additional theories describe the relationship between diversity and system stability. These are the “Rivet

Hypothesis”, which states that all species are equally responsible for the maintenance of system stability; and the “Drivers and Passengers Hypothesis”, which states that some species are more responsible for the maintenance of community integrity than others (Thompson and Starzomski 2007). Diversity may also be responsible for the stability of levels of biomass through time and the maintenance of biomass and community integrity in the face of stress. While the diversity-stability principle applies at the community level, and positive correlations can be demonstrated between diversity and stability, the opposite is observed in the relationship between population level dynamics and diversity. Population dynamics, or the turn-over of individual species through time, is inversely related to diversity in most cases (Tilman 1999).

More diverse systems sometimes show higher levels of productivity and biomass. This is because an area with a greater diversity of species is more likely to have some species that use available resources more efficiently thus producing more biomass per area than less capable species (Tilman 1999). A greater diversity of species will also use a greater range of resources, such as soil nutrients. This means, in general, that fewer resources remain unused, which may be one of the factors that lead to the theory that more diverse systems are less susceptible to invasive species due to the limitation of available resources (Elton 1958). The productivity-diversity relationship may be scale dependent. One study (Chase and Leibold 2002) indicates that diversity peaked at median levels of productivity at smaller scales, but that it increased

linearly at larger scales. This pattern may be due to spatial and temporal heterogeneity, or to the order of colonization of the habitat in question (Chase and Leibold 2002).

The above mentioned principles apply simply enough on small scales with relatively uniform conditions. Over larger areas with greater heterogeneity, a greater diversity of species is required to maintain the same levels of productivity and production of biomass (Tilman 1999). This may, in part, be responsible for the differential degrees of invasibility seen in, or the different relationships between, native and non-native richness that seem to occur in studies conducted over different spatial scales.

While theories abound with relation to the effects of diversity at the community level, some generalities may be drawn about the current state of diversity as a whole. Thompson and Starzomski (2007) describe the global fluctuations occurring at different scales of biodiversity due to the influence of human activities. Beta diversity, or the diversity of species found occurring across sites, is declining. This leads to the homogenization discussed by many biologists (McKinney and Lockwood 1999, McKinney 2006, Noss 1990). Gamma diversity, the diversity found within the total species pool, is also declining. This is due, in part, to the high rate of extinctions occurring worldwide. Finally, Alpha diversity, or the diversity of individual sites, is increasing. This increase is likely due to the introductions of non-native species to new areas. The increase in Alpha diversity due to additions of non-native species could complicate the use of biodiversity as a metric of system functionality.

The usefulness of the diversity metric has been under debate for some time. It is possible that the lack of correlation between diversity metrics and measures of system integrity is found due to the inappropriate application and interpretation of these diversity measures. Entropies such as the Shannon-Weiner Index (H') and diversities such as the Simpson Index (D) are frequently used as measures of diversity and are compared without first being converted to the effective number of species. This can lead to great difficulty with their interpretation (Jost 2006). The Shannon-Weiner Index is a measure of evenness which assumes that the full population has been measured rather than a sample of the population. It is therefore most sensitive to the rarest species in the community but also exhibits a bias in accordance with this assumption (Gurevitch et al. 2006). Simpson's Index, is a diversity index that is sensitive to the most common species in the community and can thus be thought of as measuring dominance concentration (Hill 1973). Richness, which is a simple count of the species present in sampling units, is, like the Shannon-Weiner Index, sensitive to the rarest species. These diversity metrics are best reported together as they may give a more complete picture of the dominance of species within the community. If all three measures are equal (after having a transformation applied that yields the effective number of species) then there is perfect equitability among the species within the community. If there is a great degree of spread among the measures it indicates that there are some species in the community that are much more dominant than others (Jost 2006).

Quality

Floral quality has recently been measured using indices based on qualitatively assigned coefficients of quality called the Coefficients of Conservatism (CC's). These coefficients are assigned by expert botanists to individual plant species and are based on their determinations of the "quality" of the plant species. Quality is assigned based on the sensitivity of the species to disturbance and its fidelity to a specific habitat. Several indices have subsequently been created incorporating CC's, though it is thought that the CC scores may be a more accurate measures of disturbance and ensuing changes in plant community quality than are the indices they comprise (Miller et al. 2006). This is most likely due to the fact that some of these indices such as the Floral Quality Assessment Index (FQAI) utilize species richness as a part of the index, dividing the CC scores by richness. Richness, however, is often tweaked by taking its square root or by including / excluding data on non-native species. This is done to correct for the fact that richness is often poorly correlated with measurements of disturbance. It is thought that the manipulation of the FQAI in this way may render it less predictive than the raw CC scores.

CC Scores also are frequently manipulated to improve their predictive ability (Miller and Wardrop, 2006). One method is to weight them using species abundances, and another simply averages the scores of the CC's of all the species found within the sampling unit. One excellent part of the CC metric is that subsequent studies can use the values of previously determined CC's. These values can be employed from study to study provided the studies cover the same area and plant communities (Reiss, 2006). Correlations have been found between these CC's

and wetland area, and distance to nearest wetland (Matthews et al. 2005), implying that CC's predict changes in the plant community caused by fragmentation. Likewise, they have been correlated with disturbance caused by changes in land use and development intensity (Cohen et al. 2004)

Assessing Ecosystem Level Integrity

Rapid Assessments

From a review of the literature on studies of wetland rapid assessment indices, several relevant patterns emerge. There are both benefits and drawbacks to the use of these indices in the assessment of impacts on ecosystems due to disturbance. One major difficulty is with indices that require a reference habitat against which to measure disturbance, because locating a habitat that is pristine and untouched by disturbance is challenging. In some cases, as in the case of studies of the shoreline wetlands of Lake Huron, no reference habitat was available (Wilcox et al. 2002). In a landscape increasingly structured and engineered by the human hand, it has become nearly impossible to find a true reference, and poor substitutes may become the norm.

Another problem inherent in the use of disturbance indices is that natural and seasonal fluctuations in wetland hydrology can make it difficult to determine which effects are anthropogenic in origin, especially in riparian systems (Chipps et al. 2006). In these systems the effects of disturbance may be confounded with the effects of seasonal flooding or drought

events or even longer term natural fluctuations. Some authors indicate that wetland assessment index results can be confounded by the effect of growing season on measurements of parameters like richness (Chipps et al. 2006, Matthews et al. 2005). This might be less evident in indices like those measuring floristic quality. Here the prior knowledge of expert botanists makes up the bulk of the ranking system. Indices such as the Index of Biological Integrity (IBI), that directly measure parameters like vegetation cover and proportions, may be severely affected by seasonal variation. It may be possible to account for this at least qualitatively by taking measurements of proportions of annual / perennial / biennial life history status.

There are several challenges encountered in the development of wetland assessment indices. First, there are often areas of overlap among metrics used in an index. This makes its use potentially inefficient, requiring increased sampling effort that produces lesser results. Additionally, with the use of some metrics, it is possible that the measurements exhibit collinearity, making their independent consideration problematic. This is mitigated if the index metrics are taken as a whole, though the temptation to dissect an index into other meaningful information may be strong. Metrics that are not meaningful may potentially dampen the correlative strength of an index. Individual metric scores are also often more accurate measures of disturbance than the indices of which they are a part (Miller et al. 2006).

A final concern with rapid assessment indices is interpretative in nature. It is easy to forget that small suburban and urban wetlands, while diminished in terms of biological integrity, still play key roles in the storage and processing of nutrients and pollutants, in flood abatement and in storm water treatment (Reiss 2006). For this reason, the findings of indices of wetland integrity should be weighed against the pragmatic value that highly disturbed wetlands are still able to play in the context of heavily developed areas. Unfortunately, these interpretative elements are not always immediately obvious in the face of low index rankings.

While it is easy to point to the difficulties encountered with the use of wetland disturbance indices, it is worth mentioning their value and use as well. An important point is the replicability of their results. Good indices can be calibrated to a certain area and subsequently used by many researchers, given that their study covers that same area. Some indices, perhaps most importantly, provide a framework that can be used quickly and efficiently by individuals in regulatory and management positions. This helps managers to make increasingly informed decisions about the fate of small isolated wetlands. Finally, these indices clearly demonstrate the impact of anthropogenic forces and development on adjacent wetland ecosystems.

Landscape Development Intensity Index

The Landscape Development Intensity Index (LDI) is an index that measures the intensity of human activities that take place in a given unit of area based on the quantities of nonrenewable energy used in these activities. Using this index to calculate the intensity of land use in areas

adjacent to natural lands may give an estimate of the degree of impact that human activities have on those lands. A clear picture of the local disturbance gradient can be established for an ecosystem patch by calculating the intensity of human land use activities in a buffer around the patch.

To construct the LDI index, past studies first established the average amounts of nonrenewable energies that are used in human activities. These activities and subsequent energy use vary over different types of land use and the nonrenewable energy calculations can be applied to a given area of that land use type. For example, a commercial downtown business district will consume a different quantity of energy per unit of area than an agricultural operation. However, the types of energy used in these variable human activities are usually not directly comparable. To account for this, LDI uses the “emergy metric” in its calculations, which converts all nonrenewable energy types into a single energy measure, rendering even the most varied energy types comparable.

Emergy is a calculation used in the practice of energy accounting. The emergy concept was developed by H.T. Odum in the late 1960’s because it was observed that different types of energies were being compared and evaluated side by side and that researchers were erroneously assigning a single unit to energies that ought to be denoted by different units (HT Odum and EP Odum 2000). Emergy calculations attempt to solve the problem of erroneous comparison by standardizing units. Thus products that are not directly comparable such as

sunlight, fuels, and services become comparable by calculating their emergy. To do this, their total energy is calculated as the amount of solar energy used to produce them. Emergy has “memory”, meaning that it accounts for all of the energy of the processes that go into the production of a final result, or, in other words, the final energy of anything is the sum total of the energies that went into its formation. The more work done to produce something, i.e. the more energy transformed, the higher its emergy value (Brown and Vivas, 2005). This is quite different from standard energy calculations of embodied energy, or exergy, as it considers not just the current energy content of a product, but the amount of energy used in its creation. The biological example given in HT Odum and EP Odum (2000), is the energetic comparison of trophic levels. A very different amount of energy and work is used in the formation of a joule of whale than of a joule of phytoplankton, as is clear from an understanding of how food chains work. Thus, it becomes apparent that standard energetic comparisons do not give a full picture of the actual energetic value of an end product and that emergy calculations may prove crucial in the development of indices like LDI that consider the impacts of the interactions between humans and natural communities

Study Description

Many studies indicate that there is little relationship between disturbance and species richness; and some even report increases in richness with disturbance. This may be due to the introduction of novel species to the ecosystem. Edge effects and small scale disturbance frequently lead to the colonization of areas by new species, increasing the total pool of species

present. However, these disturbances are highly localized and are may appear in measures of large total ecosystem diversity. Quality of the plant species present will also diminish as structural and functional integrity is diminished. The following hypotheses were formulated to guide this study and to help discern the utility of several disturbance metrics in the prediction of plant community quality, diversity and structure:

Diversity

Hypothesis 1: Little or no correlations will be found between the diversity of cypress dome vegetation and the various disturbance metrics including LDI, Class, TSD_LU, TSD_LDI, and Retention.

Hypothesis 2: Disturbance will cause the collapse of microhabitat diversity, leading to lower heterogeneity within the domes, which can be measured as variance of plant species cover among plots within each dome. Measures of total dome variance of plant species cover will be negatively correlated with measures of development and that variability will decrease with increased disturbance. Smaller scale, more localized disturbances, will often temporarily increase spatial heterogeneity. However, total ecosystem impacts sustained through time should actually decrease heterogeneity due to homogenization caused by the successful colonization by novel species.

Quality

Hypothesis 1 The quality of the component species within the domes will decrease with increases in disturbance. Weighted and mean quality scores (CC's) will decrease with increases in disturbance and increased non-native richness will be observed based on the metric called the Coefficients of Conservatism (CC's) (Cohen et al. 2004).

To test this, attributes of the component species were assessed by looking at the average wetland status-ranking of all species within the dome, and the richness and cover of non-native species within each dome.

Hypothesis 2: The average wetland status of the plant species, per NWI rankings (Reed 1988), will shift from OBL / FACW to FAC / FACU as increases in some types of human activities lead to shifts in the hydrology of the nearby wetlands.

To further test this point, I directly measured the percentage of land use in the cypress dome buffers that involved the drainage of the wetlands. This was used as an additional disturbance metric against which to study changes in the plant communities.

Disturbance indices

Hypothesis 1: The predictive power of the disturbance measures will decline with an increase in the total area of the cypress dome. To assess this I tested the disturbance indices using a partial correlation, controlling for dome area.

Finally, I compared several disturbance indices to determine their ability to predict changes in the vegetation of cypress ecosystems with increases in disturbance. To do this, I first used a categorical measure of disturbance based on land use occurring within a buffer around cypress domes to establish an *a priori* disturbance gradient. I then tested the Landscape Development Intensity Index that relies on measures of nonrenewable energy use to determine the intensity of human development occurring in a buffer around cypress domes. Next, I tested the impacts of drainage on the plant community by measuring the area of land use types that may be associated with shifts in the hydrology of nearby wetlands. This included canals, culverts, retention, and detention ponds. I then assessed changes in the plant community associated with the duration of development by constructing and testing two closely related indices measuring the time since the development of parcels occurring within a buffer area around cypress domes. Finally I attempted to construct a model combining all of these disturbance indices which best explained each of the dependent variables of plant community response.

METHODS

Site Selection

Sites were selected using Digital Orthophoto Quarter Quads and GIS based Land Use / Land Cover layers (2004) available through the St. Johns River Water Management District (SJRWMD) and the South Florida Water Management District (SFWMD). Potential cypress domes were selected using their Florida Land Use Cover Classification (FLUCCS) code, and the Feature Analyst Tool (VLS 2006) in ArcGIS (ESRI v.9.2), and riparian and lakeside cypress strands were eliminated from the pool by hand using aerial photos from SJRWMD leaving 2,376 cypress domes in Orange and Seminole Counties, FL. Several hundred domes were randomly selected from the pool, many of which were also eliminated as they were either too large, not accessible, or because permission to access the property was denied. The remaining 150 domes were sorted into two size classes (small = less than ½ ha, large = greater than ½ ha) and placed into one of five categorical disturbance classes (1= natural, 2 = agricultural, 3 = low urban, 4 = medium urban, 5 = high urban). To develop the classes, a 234.42 m buffer was placed around each dome. This buffer size was selected based on the average distance between all adjacent cypress domes found in Orange and Seminole counties. All land use types found within the buffer were placed in one of the five disturbance classes and the proportions of the total area each class occupied within the cypress dome buffers were calculated. SPSS (v.16.0) was then used to analyze the class data and the domes were separated into one of the five classes based on their position in ordination analysis. Canonical Discriminant Analysis was then used to verify the validity of the five classes. Domes were selected and placed into

urbanization categories by McCauley (unpublished data). Once domes were sized and classed, a random number generator tool was used to select the final 30 domes yielding three from each size / disturbance class combination.

Calculating Disturbance Indices

Once the sites were selected, an LDI score was calculated for each dome. This was done first by redrawing a 100 m buffer around the dome using ArcGIS. Coefficients were applied to each parcel found within the new buffer, based on its FLUCCS code and on values from Brown and Vivas (2005). The coefficients are calculated as the normalized natural log of the empower density. These coefficients are normalized on a scale from one to ten. The empower density is the calculation of energy use per unit area per unit of time. Using the established coefficients, the LDI equation (1), is used to derive a single LDI score for each ecosystem patch.

$$LDI_{total} = \sum \%LU_i \cdot LDI_i \quad (1)$$

Where:

LDI_{total} = LDI score for each cypress dome

$\%LU_i$ = the percent of the buffer area occupied by land use i

LDI_i = the LDI coefficient assigned to each land use i

The time since the development of the different land use types occurring in a 100 m buffer around each cypress dome was then calculated. The Orange County and Seminole County Property Appraiser's Websites were used to determine the dates of development of each parcel. Both websites have interactive GIS maps which allow individual parcels to be selected and the property data and deeds pertaining to these parcels to be accessed remotely.

The development dates of subdivisions were used to estimate approximate time since development of improvements such as roads and drainage structures contained within them. All natural lands were considered developed zero years ago and cattle grazing and agricultural operations were considered “developed” at the time of purchase by the rancher / grower. Parcels that were classified as under construction (for example, FLUCCS code 1190: low density under construction) in LULC maps were considered “developed” at the date of sale that coincided with a shift in categorization from vacant to improved on the Property Appraisers’ websites. When this data was unavailable, sale price was used to infer a change in the vacant / improved status. If the selling price of a property jumped significantly in a short period, the date of the previous purchase was used as the development date. Information pertaining to the development of major public roadways and structures was readily available through a variety of web sources. Orange County Public Works also kindly provided information on the major, long-developed roadways in Orange County. Two separate Time Since Development Indices were then constructed, each weighted by a different variable. The first index, Time Since Development weighted by land use (TSD_{LU}), was calculated as the sum of the products of the percentages that each type of land use occupied within a 100 meter buffer around the cypress dome and the time since each land use type was developed for that specific land use (equation 2).

$$TSD_LU = \sum (\%LU_i \cdot TSD) \quad (2)$$

Where:

TSD_LU = Land Use weighted Time Since Development Index

$\%LU_i$ = the percentage of land use i

TSD = the time since development of each parcel within the buffer of a given cypress dome.

The second index, Time Since Development weighted by LDI values (TSD_LDI), was calculated by taking the sums of the product of the percent of area each land use type occupied within a 100 m buffer, the LDI coefficient of each land use type, and the time since each land use type was developed for that specific land use (equation 3).

$$TSD_LDI = \sum (\%LU_i \cdot LDI_i \cdot TSD) \quad (3)$$

Where:

TSD_LDI = LDI weighted Time Since Development Index

$\%LU_i$ = the percentage of land use i

TSD = the time since development of each parcel within the buffer of a given cypress dome.

To attempt to directly measure the impacts of hydrological shifts on the plant communities, the retention independent variable was calculated. This was done using ArcGIS to calculate the area of each parcel within the LULC layer that fell within a 100 m buffer around the cypress domes. Each land use type associated with human induced changes to local hydrology was included in the calculation. The total areas of land use types such as reservoirs, canals, pits, and retention ponds, were summed. These values were transformed using the $\ln(x+1)$ transformation to account for extreme outliers before analysis.

Sampling

Using nested plots, 2.5% of each cypress dome was sampled. This percentage was selected through test sampling of cypress domes which reached diversity curve asymptotes at roughly 3% coverage. Using Hawth's tool (v.3.0 2004) in ArcGIS, the appropriate number of plots, totaling up to 2.5% coverage, were randomly placed in each dome. While this method implies that a differential sampling effort is employed for each dome, the effort is proportionally equal from one dome to the next. Circular nested plots were placed at each point. The percent cover and identity of the herbaceous plants, vines, and low growing shrubs was determined in small plots (1 meter diameter) at each point. Percent cover was estimated visually using arcsine square root cover classes of values: 0, 1, 5, 25, 50, 75, 95, and 99%. The use of these classes eliminates the need for the transformation of data after collection (McCune and Grace 2002). The herbaceous plots were centered and nested within larger tree plots. Tree identity and diameter at breast height (DBH) was determined in these large tree plots (5 meter diameter). Trees with diameters less than 5 cm were not counted, though basally branching trees whose branches added up to more than 5 cm in diameter were included. Within each dome, two specimens of each plant species were collected as vouchers for placement within the University of Central Florida Herbarium (FTU).

For each tree species, DBH was transformed into basal area (m²) using equation 4.

$$\left(\left(\frac{DBH}{2} \right)^2 \cdot \pi \right) \quad (4)$$

The basal area was then divided by the large tree plot area to determine the percent cover of trees within the larger tree plots.

Plant Collection and Analysis

Two specimens of all species identified within each cypress dome were collected. These were transported from the field and put into plant presses. These specimens were then dried in the University of Central Florida Herbarium dryer for several days at 140° F. These specimens were tentatively identified and stored in herbarium cabinets. Once all the domes were sampled and all specimens were pressed and dried, the specimens were re-sorted so that all tentatively identified specimens were placed together. This allowed for comparisons to be made during formal identification between individual plants of the same species occurring in different wetlands. This also aided in the identification of individuals that were collected outside of their flowering period. Formal identifications were then made using Wunderlin and Hansen (2003), and its supporting online database Wunderlin and Hansen (2008). The United States Department of Agriculture plant database (USDA NRCS 2009) was also used in identifications. A Microscope was used to help in study of small floral parts especially in the graminoids, and the herbarium collection was used to verify identifications. All identifications were entered into a database (appendix A) and the Department of Environmental Protection (DEP) and NWI

wetland status of each species was listed, along with its native status (native or non-native).

Wetland status info was taken from DEP and NWI rankings. Any species were listed as FAC for which an NWI / DEP ranking was not assigned and which were not clearly an aquatic (OBL).

Each species was then assigned a coefficient of conservatism (CC) quality score. CC values were assigned to each species based on values reported in Cohen et al. (2004). These CC's are quality rankings assigned by expert botanists based on a plant's affinity to a particular habitat and its tolerance to disturbance. High scores indicate a species is high in quality, sensitive to disturbance, and has a high fidelity to a very specific habitat. Lower scores are indicative of weedier species that grow abundantly across a wide array of habitats and are less sensitive to, or are even colonizers after, disturbance (Andreas et al. 2004). For plant species with no CC ranking, the values of the all species within the particular genus were averaged and that value used. If the genus of interest had no species ranked, a value of zero was used if the plant was an Exotic Plant Pest Council (EPPC) listed species, 0.6 if it was an unlisted exotic, 1 if it was a low quality native, and 5 if it was a medium quality native. Total plot and total dome quality were also calculated using both the mean CC score of all species occurring within the dome, calculated using equation 5, and using a weighted CC score, calculated using equation 6, which compiles all the CC scores across the entire plot or dome. This calculation weights a given CC score by the abundance of that species across all of the plots within the dome.

$$\text{mean } CC_j = (\sum CC_{ij})/N_j \quad (5)$$

$$\text{frequency weighted } CC_j \text{ score} = \sum (CC_{ij} \cdot \text{relative frequency}_{ij})/N_j \quad (6)$$

Where:

CC_{ij} = is the CC score for species i at site j , and N is the number of species at site j .

Finally, both the proportion of exotic species occurring within each dome and the exotic species richness were calculated to give a further indication of total plant community quality. The numbers and proportions of exotic species were calculated using only individuals that were able to be identified to the level of the species. Native status could not usually be determined for specimens identified to the level of genus or family; thus, these individuals were eliminated from the pool of data before proportions were calculated.

The diversity of species occurring within the domes was also calculated. A simple species count, or richness, was determined for each dome and these values were then used in the calculation of diversity entropies. First the Shannon-Weiner index (H') was calculated using formula 7. Then the Gini-Simpson index (D) was calculated using equation 8.

$$H' = - \sum [p_i \ln (p_i)] \quad (7)$$

$$D = 1 - \sum p_i^2 \quad (8)$$

Where: p_i = the proportion of individuals in the i th species

These values were then transformed to the effective number of species using methods described in Jost (2006). See equations 9 and 10 for methods used to transform the indices.

$$H' \text{ eff} = \exp\left(-\sum [p_i \ln(p_i)]\right) \quad (9)$$

$$D \text{ eff} = 1/\left(1 - \sum p_i^2\right) \quad (10)$$

Species data were initially recorded as the percent cover of herbaceous species and tree species. It was later found that the tree values for percent cover were quite low in comparison with the herbaceous values of percent cover. The tree data had disproportionately small values due to the use of basal area rather than canopy area measurements. To correct for this discrepancy, percent cover was used to calculate species Importance Values (IV's) (Gurevitch et al. 2006, McCune and Grace 2002). These are measurements of the relative value of an array of metrics summed and divided by the total number of metrics used. The metrics utilized in calculations often include density, frequency, and cover. However modifications of these may be used depending upon the data available and the nature of the study (Gurevitch et al. 2006). Calculations of density could not be used in the importance value calculations as individuals of each species were not distinguished; rather, the coverage of all individuals together was recorded. The importance values were thus calculated for each species j using equation 11.

$$IV\% = (Relative\ frequency\ \% + Relative\ density\ \%)/2 \quad (11)$$

Where:

$$Relative\ frequency_j\ \% = \frac{100 \cdot frequency_j}{\sum frequency_j}$$

$$Relative\ density_j\ \% = \frac{100 \cdot density_j}{\sum density_j}$$

Data Analysis

All data was checked for normality, errors, and outliers using SPSS and the variables TSD_LU, TSD_LDI, and retention were subsequently transformed using a $\ln(x+1)$ transformation to adjust for large outliers. PC-ORD (v.5.0., McCune and Grace 2002) was used to build the species area curve and perform Sorensen (Bray-Curtis) distance measurements and Jackknife analyses which were used to assess expected diversity. Pearson's correlations and curve fitting were then conducted using SPSS to explore relationships between variables. Partial correlations were analyzed also using SPSS to control and test for the impact of variable dome size on the disturbance indices. NMS Ordinations were constructed using PC-ORD to assess relationships between domes based on vegetation and suite of dependent variables. Regressions and scatter plots, constructed using SPSS, were then used to further assess relationships between variables and to complete hypothesis testing. Finally R (v.2.8.1) was used to calculate AIC values of possible models to determine the best combination of independent variables explaining the axes developed during ordinations and the other plant community variables.

RESULTS

Diversity and structure

In total, 1,266 individual trees were measured across 257 plots and 30 domes. Jackknife estimates, calculated using PC-ORD, give an estimate of the expected number of species. In total 188 different plant species were identified across all of the cypress domes. First order jackknife estimates indicate that 246.8 species were expected and second order jackknife estimates indicate that 277.6 total species were expected. The most diverse dome contained 46 species, while the least diverse dome had only 6. The mean diversity across all of the plots was 21.6 species. Exotic species diversity was lower than anticipated with exotics showing up in only a third of the sampled domes and making up, at the most, no more than a quarter of the total species observed in any given dome (see Appendix B for data pertaining to these results). A species area curve was constructed in PC-ORD using the percent cover data, to determine whether the plant community was adequately sampled. Sorenson distance measures were used to conduct this analysis. See figure 1 for this species area curve and the confidence intervals associated with it. The bottom part of this graph represents the differences between each sample and the rest of the samples. The upper confidence interval nearly reaches an asymptote indicating that a sufficient sample was taken to adequately represent the plant species diversity encountered within the domes.

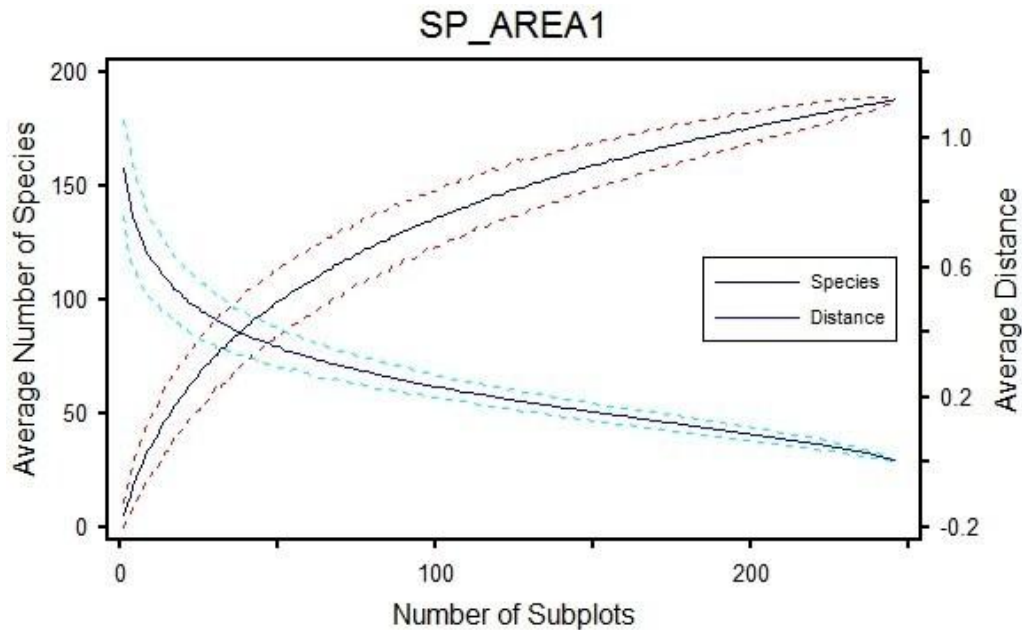


Figure 1. Species area curve

Two indices of species diversity were calculated in addition to simple richness measures. The relationship between these indices and richness yields information about the dominance of species within the community. When richness, the Simpson effective number of species, and the Shannon-Weiner effective number of species are compared, the degree of spread among them indicates dominance or degree of evenness; the greater the spread, the lower the evenness, and the greater the dominance of a few species within the community. This is due to the fact that richness and the Shannon-Weiner Index are sensitive to rare species while Simpson's Index is sensitive to the most common species. If the Simpson effective number of species is less than the Shannon-Weiner effective number of species which is in turn less than the species richness, this indicates a degree of dominance among the species as well. In the cypress dome communities, all domes exhibited very low levels of evenness. This is most likely

because woody species and trees recurred frequently across most plots along with several fern species. Many of the herbaceous species were rarer, occurring in only one or two plots per dome.

Table 1. Diversity indices and richness by dome

Dome	D_eff	H'_eff	richness
174	2.72	4.40	19.00
179	6.19	7.72	17.00
193	1.12	1.33	6.00
565	3.95	7.01	33.00
612	6.00	8.63	24.00
679	7.40	8.85	18.00
913	7.83	10.83	33.00
986	3.31	4.74	12.00
1035	3.51	4.95	15.00
1038	2.61	3.36	14.00
1086	5.76	7.35	16.00
1091	11.83	16.39	42.00
1181	6.53	10.55	38.00
1192	2.38	4.00	20.00
1252	7.28	9.68	29.00
1398	4.07	5.01	12.00
1459	2.33	2.77	10.00
1462	4.69	5.80	12.00
1852	11.40	17.02	46.00
1854	8.05	12.07	33.00
1928	4.47	6.39	19.00
2092	3.21	3.99	7.00
2169	3.35	5.08	19.00
2248	10.18	14.69	32.00
2265	4.47	6.54	16.00
2283	3.00	5.47	29.00
2285	5.84	8.18	17.00
2292	4.57	6.64	15.00
2315	4.76	5.97	12.00
2373	5.25	6.57	16.00

Those domes with the highest levels of species richness, such as 1091 and 1181, also seem to be those that exhibit the highest degrees of dominance.

Pearson's Correlations were used in SPSS (Pallant 2007) to explore the relationships between all measures of disturbance stemming from human impact and the three measures of diversity (table 2). No significant correlations were found between any of the variables tested, implying that disturbance played very little role in the diversity of the plant species found within the domes.

Table 2. Correlations between disturbance indices and plant diversity

	Richness		D effective		H' effective	
	Pearson correlation	sig. (2-tailed)	Pearson correlation	sig. (2-tailed)	Pearson correlation	sig. (2-tailed)
LDI	0.12	0.527	0.124	0.512	0.114	0.549
Class	0.018	0.923	-0.016	0.935	-0.048	0.8
Retention	0.089	0.642	0.006	0.975	0.016	0.933
TSD_LDI	-0.109	0.565	-0.182	0.336	-0.197	0.296
TSD_LU	-0.151	0.426	-0.235	0.211	-0.247	0.188

The variance of the vegetative cover among plots was calculated and analyzed using Pearson's correlations. One extreme outlier, dome 1852, was removed before analysis, and all data was transformed using the natural log transformation. The variance among plots was not

significantly correlated with any disturbance metric, implying that a different mechanism is responsible for the heterogeneity in the degree of vegetative cover within the domes.

Structure among the domes was further analyzed with ordinations generated using the non-metric multidimensional scaling (NMS) technique. Importance values (IV) of each species within the domes were used to construct the ordinations. Mean values of plant species cover and percent cover were also initially tested using ordinations, though these measures were found to be less informative.

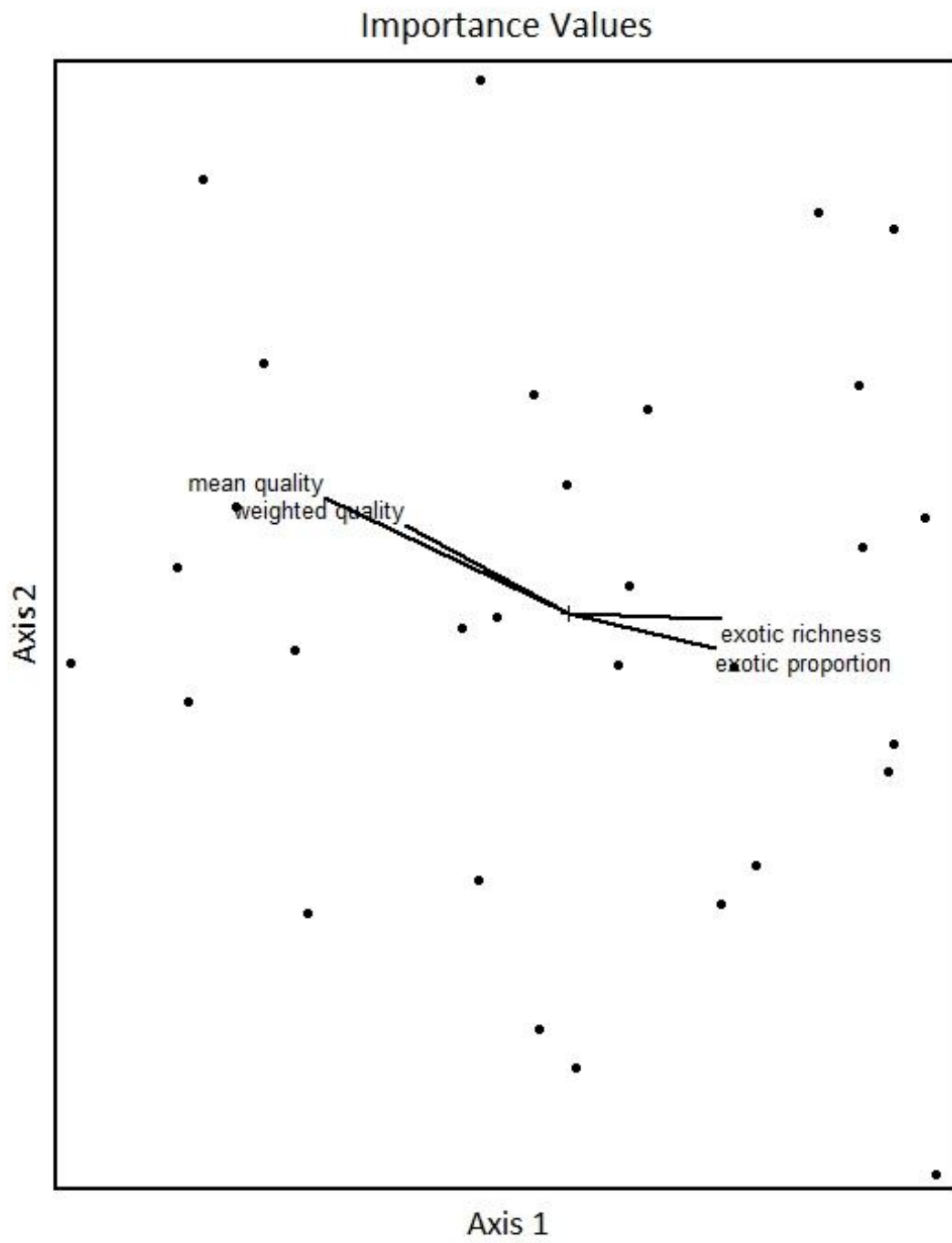


Figure 2. NMS Ordinations of importance values of species

In ordinations (figure 2), there was a negative correlation between axis 1 and mean species quality ($r^2 = 0.418$) and weighted species quality ($r^2 = 0.277$). There was a positive correlation between axis 1 and exotic richness ($r^2 = 0.275$), and the percentage of exotic species within plots ($r^2 = 0.258$). No strong correlations were observed for axis 2. Plots of individual species along the axes elucidate several interesting points. First, domes that were drained by retention features (figure 3) had higher occurrences of exotic species, lower quality species, and species usually found in upland settings. For example, *Toxicodendron radicans* (L.)Kuntze (poison ivy), a low quality species with a CC of 1.3, was found largely in domes drained by retention features (figure 4). Higher quality species and those that were less drought-tolerant occurred together in domes that were not drained by retention features, for example, *Xyris fimbriata* Elliott, a high quality species with a CC of 5.7, occurs primarily with wetland species (figure 5).

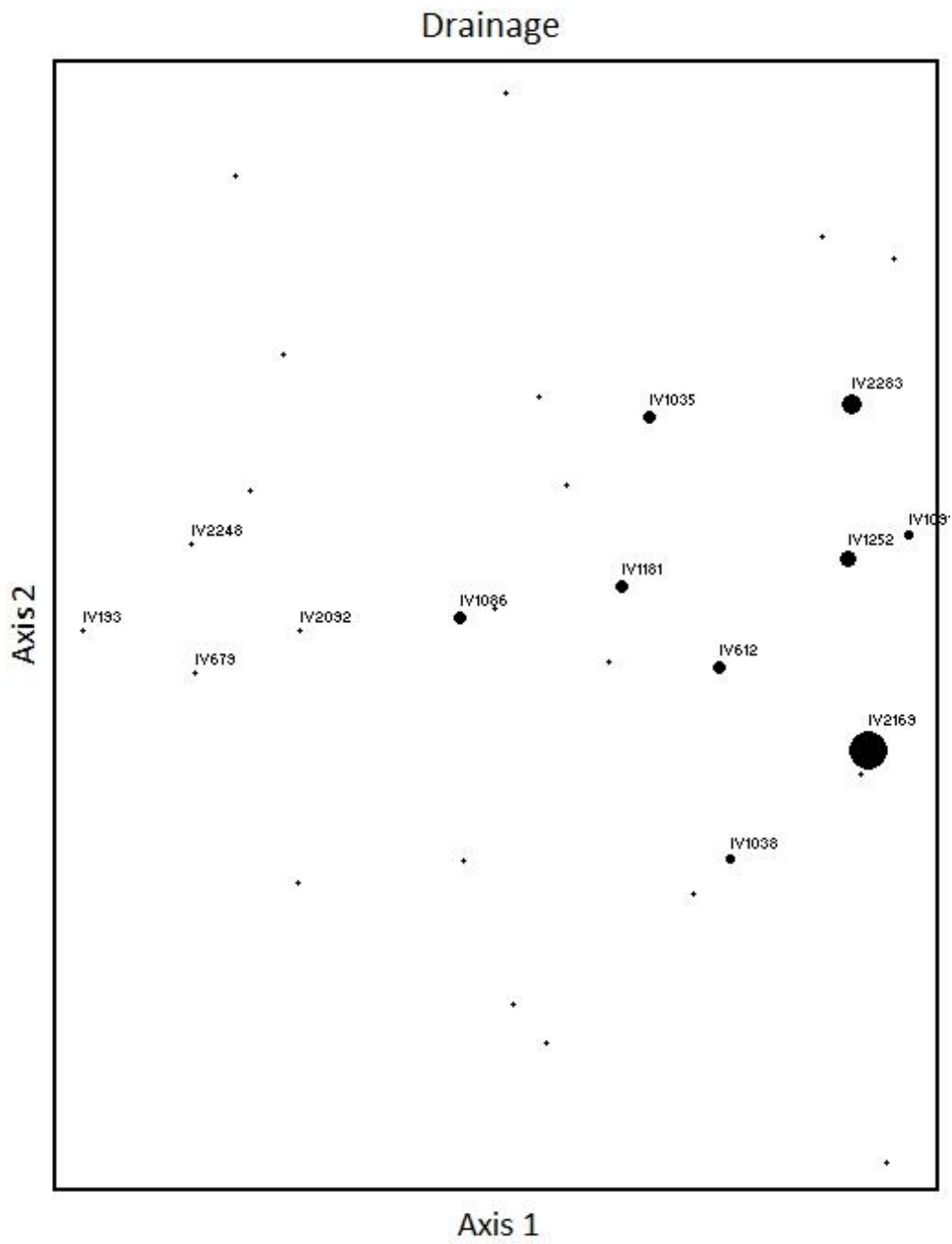


Figure 3. NMS Ordinations of domes with adjacent land use associated with drainage

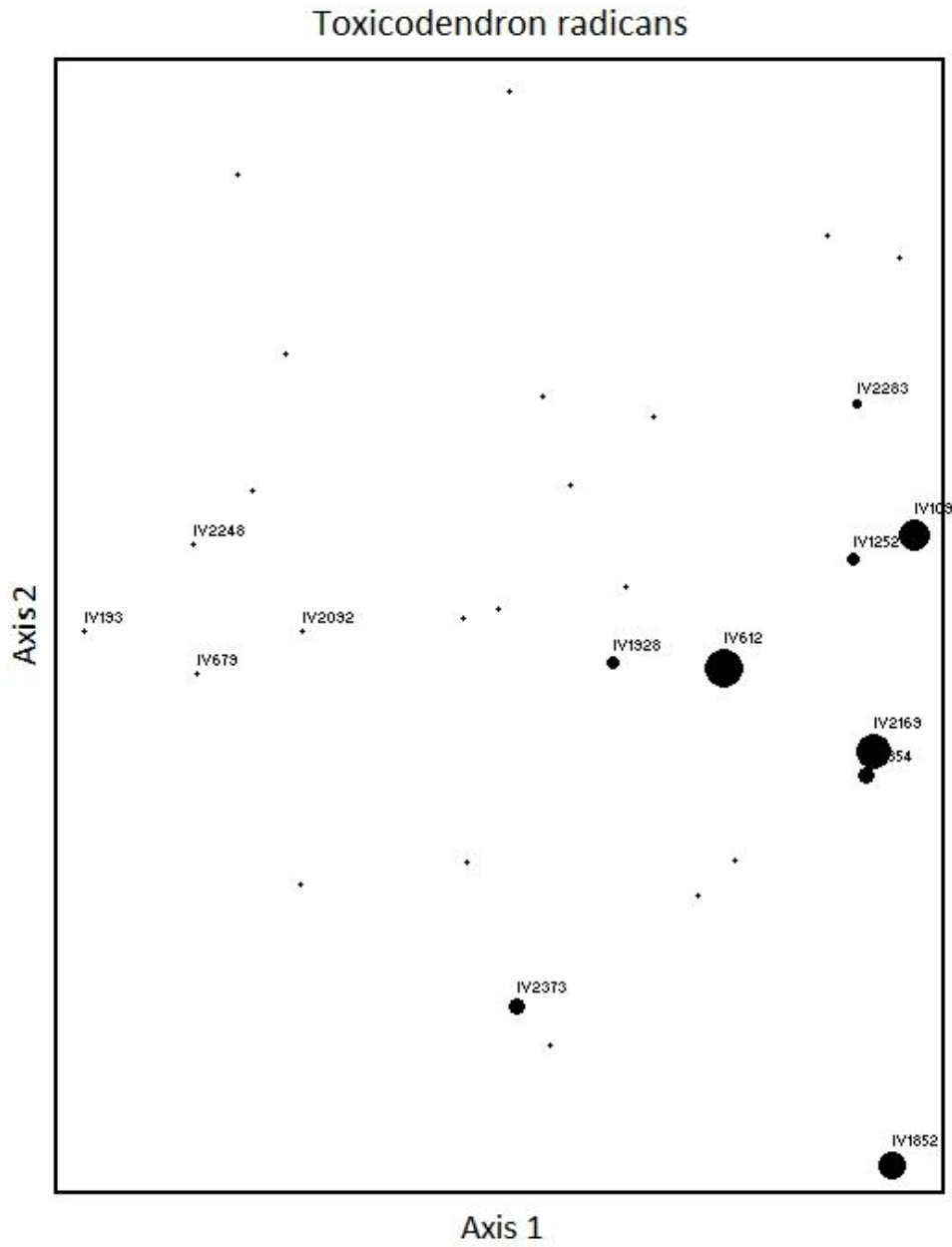


Figure 4. NMS Ordination of domes with *Toxicodendron radicans*

Xyris fimbriata

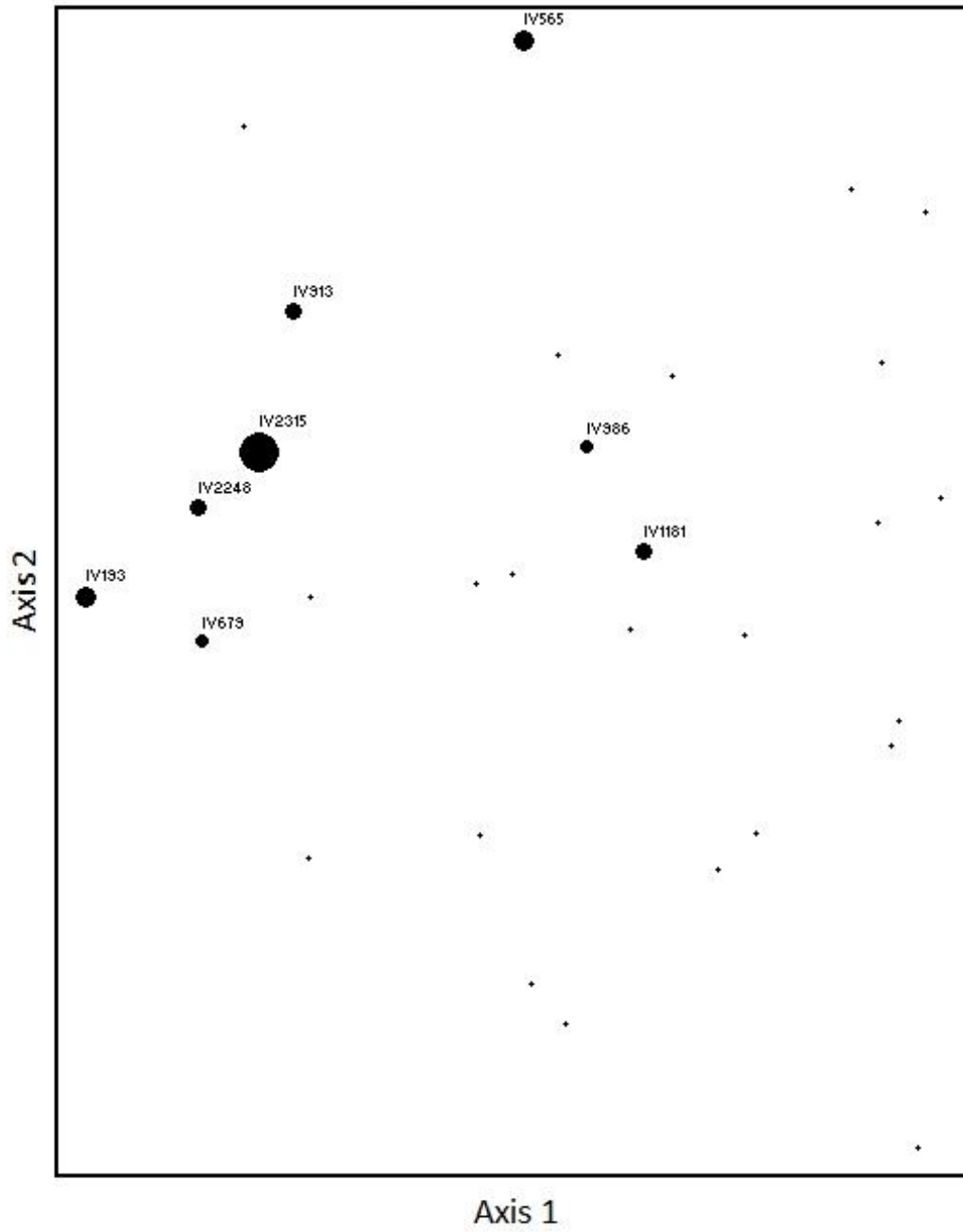


Figure 5. NMS Ordination of domes with *Xyris fimbriata*

Quality

The relationship between disturbance and quality was explored using Pearson product-moment correlation coefficients (table 3). There was a strong negative correlation between LDI and mean species quality ($r = -0.521, p < 0.01$) and between LDI and weighted species quality ($r = -0.522, p < 0.05$). While the percentage of exotic species was positively correlated with LDI ($r = 0.455, p < 0.05$), exotic species richness showed a weaker, nearly significant, positive correlation ($r = 0.341, p = 0.065$). Class, Retention, TSD_LU, and TSD_LDI were not significantly correlated with any measurements of plant species quality though retention showed nearly significant positive correlations with percentage of exotics ($r = 0.341, p = 0.065$) and nearly significant negative correlations with mean quality ($r = -0.336, p = 0.069$). Linear regressions of CC scores were created to further assess the nature of their relationship with LDI (figures 6 and 7).

Table 3. Correlations between disturbance indices and quality measures

	Percentage of exotics		Exotic species richness		Mean CC scores		Weighted CC scores	
	Pearson correlation	sig. (2-tailed)	Pearson correlation	sig. (2-tailed)	Pearson correlation	sig. (2-tailed)	Pearson correlation	sig. (2-tailed)
LDI	0.455	0.011*	0.341	0.065	-0.521	0.003**	-0.522	0.033*
Class	0.22	0.243	0.187	0.322	-0.296	0.112	-0.333	0.072
Retention	0.341	0.065	0.251	0.181	-0.336	0.069	-0.277	0.138
TSD_LDI	0.055	0.771	0.026	0.893	0.018	0.925	-0.104	0.583
TSD_LU	0.07	0.714	0.025	0.895	0.008	0.968	-0.111	0.56

* = significant at the 0.05 level

** = significant at the 0.01 level

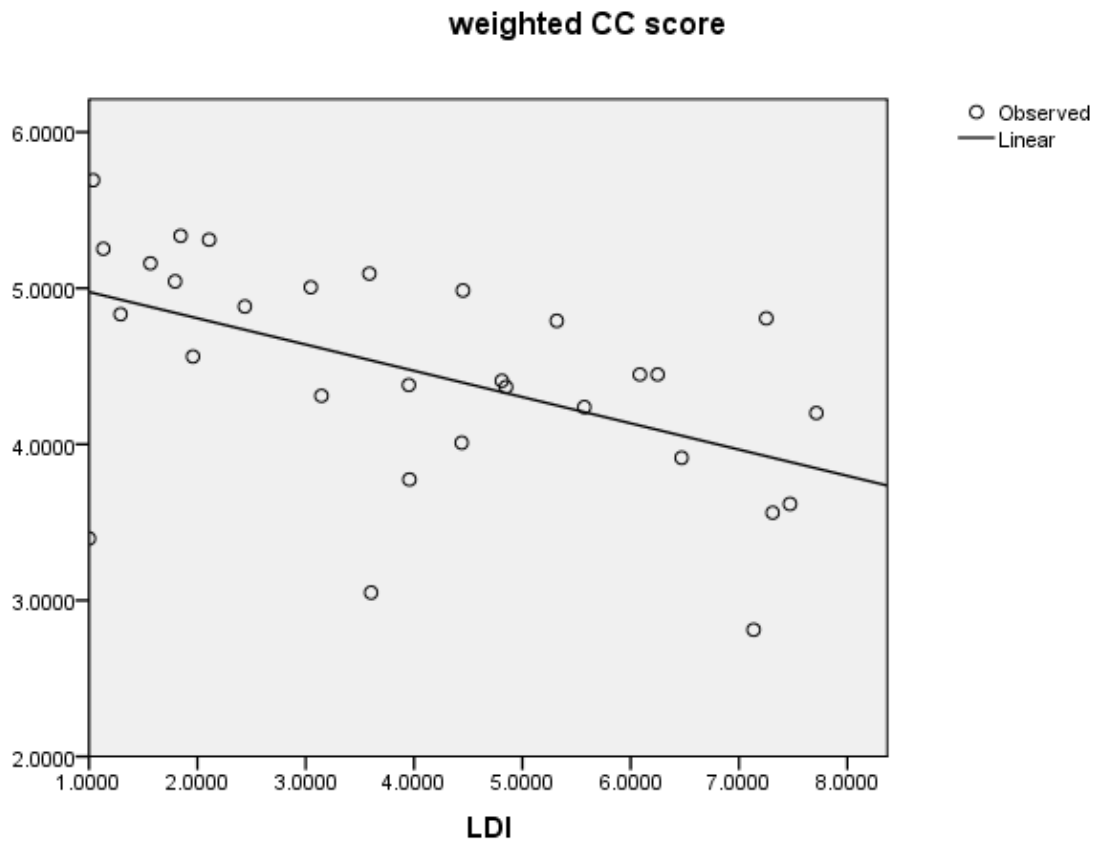


Figure 6. Linear regression of LDI and Weighted CC scores

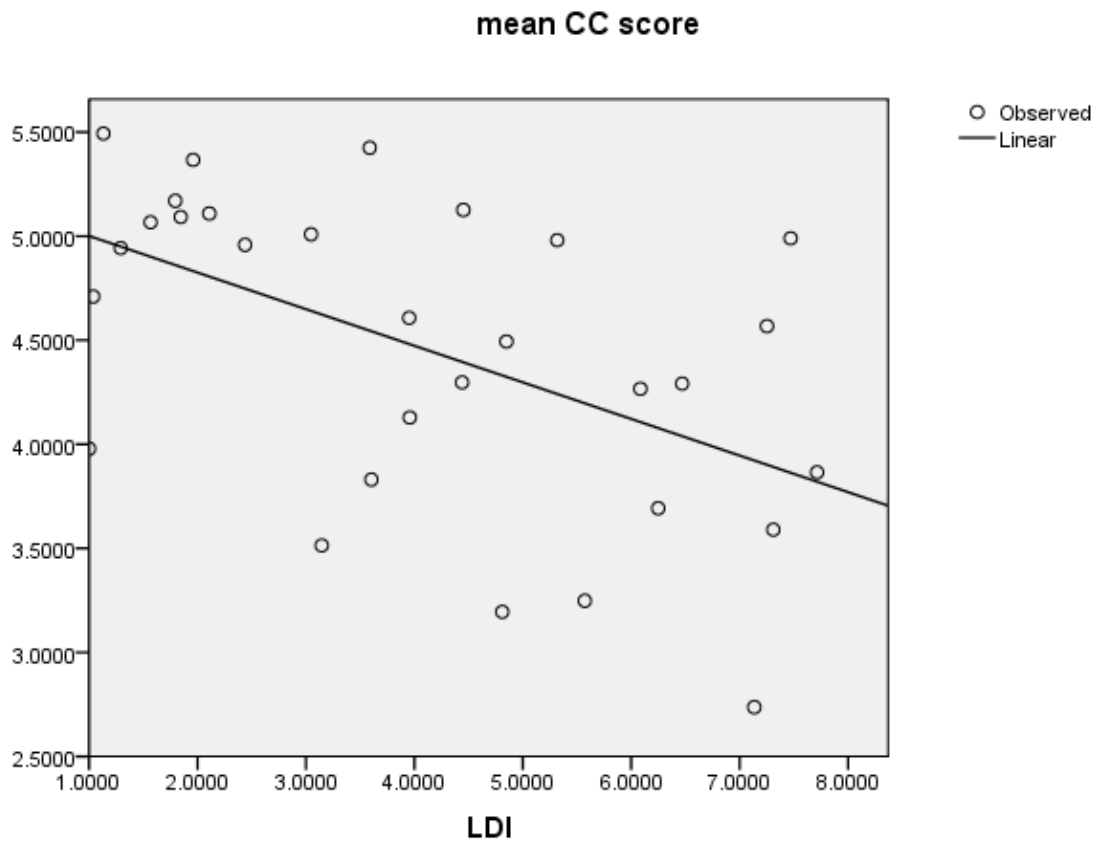


Figure 7. Linear regression of LDI and Mean CC scores

Analyses of disturbance and plant community composition based on the wetland status of the plant species showed interesting results as well. NWI wetland rankings were coded so that obligate (OBL) wetland species received a 1, high fidelity facultative wetland (FACW+) species a 1.5, up through upland (UPL) species which received a 6. Pearson's correlations show a significant positive relationship between the area of adjacent land use associated with drainage and the average wetland status of plant species within the dome ($r = 0.460$, $p < 0.05$). This implies that more plant species with upland type rankings occurred in domes that were

drained. The percentage of exotics within the dome showed a negative correlation with the average wetland status of the species ($r = -0.516$, $p < 0.01$), showing that a greater percentage of exotic species occurred in domes with greater numbers of upland species. Finally, the mean quality of the plant species was strongly negatively correlated with the average wetland ranking of species ($r = -0.676$, $p < 0.01$), which shows that low quality species occurred with upland type species. This reinforces the findings in ordination analyses that drier / drained sites had a plant community composed of greater numbers of upland, low quality, and exotic species.

Disturbance Indices

Tests were conducted to analyze the effects of dome size on the predictive ability of the disturbance metrics using partial correlation analyses in SPSS. Comparisons were made between correlations of variables before and after controlling for the area of the cypress domes. There were strong negative correlations between LDI and plant quality metrics while controlling for cypress dome area, for example LDI and weighted quality scores ($r = -0.522$, $p = 0.003$) However inspection of the zero order correlation ($r = -0.512$, $p = 0.004$) suggests that controlling for cypress dome area had very little effect on the strength of the relationship between these variables. One study found that richness and quality were heavily correlated with size of ecosystem patch, increasing significantly with increases in wetland size (Matthews et al. 2005); however, the small dome sizes used in this study may have prevented the size of domes from significantly affecting the strength of disturbance indices.

As a final test of the disturbance indices, Aikake Information Criterion Analysis (AIC) (Anderson et al.2000, Anderson et al. 2001, Johnson and Omland 2004), was used in an attempt to select the most parsimonious model explaining each of the plant metric response variables (figure 8). A chart of previously calculated Pearson’s correlation values for each pair of variables was used to select the most viable models for inclusion in AIC model selection. Those correlations with p values > 0.2 and r values < 0.1 were omitted during the selection process. Class and LDI were not used simultaneously in model selection processes as they were each constructed using the same land use coding system, FLUCCS, and were based on similar assumptions. Also, bivariate correlations between them yielded very high r values and significance ($r = 0.813$, $p < 0.001$) implying possible collinearity. TSD_LU was used over TSD_LDI in tests of the time since development variable, as it could be included in analyses with LDI and it yielded slightly higher correlations with variables. No models were constructed to explain richness as none of the potential model parameters significantly explained richness.

Table 4. AIC model selection results

Model	R²	F	Sig.	Predictors	Response	AIC	ΔAIC
1	0.272	10.460	0.003	LDI	Mean_CC	62.481	0.000
2	0.301	5.806	0.008	LDI, Retention		63.269	0.788
3	0.314	3.971	0.019	LDI, Retention, TSD_LU		64.684	2.203

Model	R²	F	Sig.	Predictors	Response	AIC	ΔAIC
1	0.273	10.490	0.003	LDI	W_CC	59.745	0.000
2	0.276	5.149	0.013	LDI, Retention		61.598	1.853
3	0.276	3.308	0.036	LDI, Retention, TSD_LU		63.590	3.845

Model	R ²	F	Sig.	Predictors	Response	AIC	ΔAIC
1	0.354	7.401	0.003	LDI, Retention	Exotic_p	-74.029	0.000
2	0.295	11.700	0.002	Retention		-73.395	0.634
3	0.355	4.759	0.009	LDI, Retention, TSD_LU		-72.046	1.983

Model	R ²	F	Sig.	Predictors	Response	AIC	ΔAIC
1	0.207	7.315	0.012	LDI	Exotic_n	-69.879	0.000

Model	R ²	F	Sig.	Predictors	Response	AIC	ΔAIC
1	0.212	7.516	0.011	Retention	wetland	19.979	0.000
2	0.214	3.670	0.039	LDI, Retention		21.898	1.919
3	0.213	3.661	0.039	Class, Retention		21.914	1.935

Model	R ²	F	Sig.	Predictors	Response	AIC	ΔAIC
1	0.296	3.640	0.026	LDI, Retention, TSD_LU	axis 1	52.514	0.000
2	0.167	5.630	0.025	Retention		53.537	1.023
3	0.249	2.872	0.055	Class, Retention, TSD_LU		54.446	1.932

Models were constructed for each dependent variable measured in the study. Several variables were not significantly correlated with any of the predictor variables and thus were excluded from further analysis. Plant species mean and weighted quality were best predicted by LDI alone though LDI*Retention was the next best model for each. LDI*Retention best predicted

the percentage of exotic species occurring in the domes, though the number of exotics was only significantly predicted by LDI. Retention alone best predicted the wetland status of plant species, though LDI*Retention followed closely behind.

Finally, all three ordination axes were tested to attempt to discern the model best describing the distribution of species from the previously constructed ordinations. Axis 1 was the only one with significant correlations with any of the predictor variables. The model combining LDI, retention, and the time since development weighted by area best fit this axis (table 4).

DISCUSSION

Diversity and Structure

No correlations were found between the diversity of cypress dome vegetation and the metrics of disturbance. No models predicting ecosystem diversity could be developed that significantly attributed levels of diversity to disturbance, as none of the predictors to be included in the models were correlated with diversity. This may seem to be in contrast with the expectations of the intermediate disturbance hypothesis (IDH) (Grime 1973, Connell 1978); however, the IDH may apply most readily to natural perturbations. One study showed that intermediate disturbance was 371% more likely to increase diversity in systems affected by natural disturbance than those affected by human disturbances (Mackey and Currie 2001).

Other studies of biodiversity have had little success correlating anthropogenic disturbance and diversity as well. Mackey and Currie (2001) found that 35% of richness studies, 28% of diversity studies, and 50% of evenness studies failed to find correlations between diversity and disturbance. Some possible causes of low correlations are that richness, as a metric is challenging to assess as it is so heavily affected by sampling methodology and estimation methods (Fleishman et al. 2006, Mackey and Currie 2001). Also, disturbance may actually lead to increases in diversity while component species quality and the diversity of species interactions are lost (Gurevitch et al. 2006). Ehrenfeld (2005) found that increases in development around wetlands led to a shift in the plant community from native herbs and

shrubs to non-native herbs and vines and that impacted sites showed an increase in total richness. Other studies of native species richness in relation to non-native species richness have even found that the two were *positively* correlated, or that increases in the diversity of native species implied an increase in the diversity of non-native species (Stohlgren et al. 2002, 2003, 2006; Bruno et al. 2004; Davies et al. 2005, 2007; Knight and Reich 2005; Lu and Ma 2005; Fridley et al 2007; Belote et al. 2008). The idea that ecosystems exhibiting the highest levels of biodiversity are simultaneously the most invaded may have enormous implications for the integrity of some of the world's great centers of biodiversity (Stohlgren et al. 2003), and may change the way we think of and use the diversity metric.

While diversity has classically been used as a measure of ecosystem functionality and integrity and has been theorized to dictate the degree of invasibility by non-native species (Crawley et al. 1999, Tilman 1999, Byers and Noonberg 2003, Lu and Ma 2005, Casey et al. 2006, Stachowicz and Byrnes 2006, Capers et al. 2007), it may be best used in systems where an understanding of natural disturbance processes is being sought. A better assessment of changes in plant communities caused by human activities may be through the use of functional diversity rather than standard species diversity (Diaz and Cabido 2001). Structural and functional diversity studies that examine processes operating at the ecosystem and landscape scales along with the diversity of interactions among species may yield more reliable information (Noss 1990), especially about the consequences of urbanization. Further studies are needed to understand the connections between functional diversity, species diversity, and ecosystem functioning

(Hooper et al. 2005) especially as ecological literature has historically been heavily focused on biodiversity as an indicator rather than as a goal (Noss 1990). This may mean that the consequences of biodiversity loss continue to be misunderstood, and that the consequences of functional diversity loss remain undervalued.

This study further hypothesized that disturbance would cause the collapse of microhabitat diversity, leading to lower heterogeneity within the cypress domes. To test this question, total dome variance was studied against measures of development intensity. No relationship was found between total dome heterogeneity and disturbance though several factors may account for this. First, by their very nature, wetlands are highly heterogeneous (Santamaria 2002). Loss of microhabitats, or increases in the homogenization of the plant community, may be difficult to assess due to high levels of background heterogeneity. The structure of cypress domes wetlands is affected by hydrology, cypress dome basin structure, parent soils, ecological processes such as fire regime, and the composition of the surrounding matrix (Casey and Ewel 2006, Ewel and Odum 1984, Kirkman et al. 2000, Riekerk and Korhnak 2000). Thus, the impacts of development intensity on cypress dome spatial heterogeneity may be difficult to extract from this background network of influences. Finally, the largest dome used in this study was 1.8 ha. This most likely reduced the impact of disturbance on heterogeneity as the influences of development around the edges of the dome were more likely to penetrate throughout the studied domes.

Quality and Disturbance Indices

While plant species quality as an indicator has only been in use in its current form since 1979 (Swink and Wilhelm), it is a powerful indicator of ecosystem integrity, widely employed in wetland monitoring programs. Within the study cypress domes, urbanization heavily affected the quality of the plant species as measured by Coefficients of Conservatism (CC's) and the proportion and richness of exotic species. LDI was significantly correlated with almost every measure of quality except for exotic species richness, which did not correlate significantly with any of the measures of disturbance. Class was not significantly correlated with quality or exotics; nor was retention or the time since development indices, though these made significant contributions to models explaining each of the quality metrics. This may be because the retention and time since development metrics only looked at a cross section of development, using land use in a 100 m buffer, and thus failing to capture impacts stemming from alterations at the watershed / landscape scale. Results may also have been stronger if other alterations such as paving of adjacent lands and elevation/grading changes had also been measured, though theoretically LDI should account for these types of development as well.

The retention metric will likely never be as predictive as the direct measurement of water levels in the domes (soil water retention, depth, etc.), though these types of direct abiotic measurements only indicate that hydrology affects the composition of the plant communities (an observation made long ago), and does not indicate what the cause or degree of impact may actually be. It is important for future studies to begin to elucidate the intensity of impact

stemming from different types of hydrological manipulation; ditching, retention / detention pond construction, grading, etc. To do this, a study measuring more than land use may be necessary. As long as we measure only the direct impacts of human alteration of the landscape (i.e., measuring the dryness of soils rather than changes to the surrounding landscape that led to the drying soils) we will be unable to directly predict the consequences of landscape alteration.

The time since development metrics (TSD_LU and TSD_LDI) should probably be restructured before being utilized in future studies as several issues might have occurred with regard to their calculation. It seemed, through qualitative observation of the cypress dome plant communities, that wetlands embedded in a matrix of older suburban developments had severely diminished species quality and much higher incidences of non-native species. While correlations between the time since development of the wetland buffer areas and wetland plant community quality may have been strong if the metric had been used in an entirely suburban landscape, the inclusion of agricultural lands may have confounded the metric. For example, subdivisions found in the buffer area around wetlands, were usually developed fairly recently, as most cypress domes were located in recently developed parts of Orange and Seminole county (because you do not find cypress domes in subdivisions developed more than 20 to 30 years ago). Agricultural lands, by contrast, have been used for grazing and farming for 50 years or more in many parts of Orange and Seminole counties. Thus, these agricultural areas counted more heavily toward the calculation of the TSD metric than the more intensively

developed suburban areas. In an attempt to account for this, LDI coefficients were used in the TSD_LDI.

It can be definitively stated that the Landscape Development Intensity Index(LDI) was correlated with decreases in the quality of the plant community, though this metric too, was accompanied by a large degree of noise, stemming from the generalized nature of the index. Thus correlations, while significant, were only able to predict up to 27% of the variance occurring in the quality of the plant species, and at most 21% of that occurring in the percentage of exotic species. This is somewhat weaker than correlations found in other studies where LDI was significantly correlated ($r^2 = 0.73$, $p < 0.001$) with measures of floral quality (wetland mean CC scores) (Cohen et al. 2004). This may be due to the fact that the Cohen et al. (2004) study utilized CC scores normalized on a 10 point scale, whereas, this study utilized the assigned CC values of which 7.3 was the highest ranking. LDI has been successfully used in a number of other wetland impact studies since its development. For example, Lane and Brown (2006) and Lane et al. (2007) successfully used LDI to assess fluctuations of diatom diversities with land use changes and Mack (2006) found strong correlations between LDI and several wetland indicators such as bryophyte richness. LDI has also subsequently been used in the development of wetland rapid assessment procedures such as WRAP (Wetland Rapid Assessment Procedure - Miller and Gunsalus 1999) and FWCI (Florida Wetland Condition Index - Lane 2003 and Reiss 2006). It has also been found to be correlated with abiotic measures of

system integrity such as total pollutant load (total nitrogen load: $r^2 = 0.75$ and total phosphorous load: $r^2 = 0.74$; $p = 0.05$).

Possible confounding factors.

Field work was started early in the season in May, before all plant specimens, namely grasses and sedges, were blooming. The flowering parts are essential for an accurate identification and thus, there may be a greater number of grasses and sedges from the earlier part of the study that were not identified or which were only identified to the level of the genus. The cross-referencing of flowering and non-flowering specimens of the same species was utilized in an attempt to overcome this issue.

The LULC maps available through the SJRWMD (and really any digitized map of landscape vegetation and features) occasionally had errors that may have affected the calculations of LDI. When it was noticed, these features were hand digitized and reclassified using ArcGIS, though the use of four year old LULC maps (2004) could further increase the misclassification of land use features due to the fact that development in Central Florida is occurring at such a rapid pace. In most studies ground-truthing is utilized to overcome these difficulties, though in this study, no discrepancies were observed between the landscape and the maps.

Another potential concern was that the coverage data used in this study was not equally weighted, as tree data was collected using a different method than was used for herbaceous

data. The use of Basal Area in the calculations of tree coverage meant that the influence of trees was underweighted compared to the cover class estimation of herbaceous plants. A potential solution would be to use estimates of tree canopy coverage rather than basal area, though this method can be significantly less accurate than basal area calculations. With a greater sample size, tree and herbaceous data could also be analyzed separately as was done in Mack (2006). To attempt to correct for the problem, importance values (IV) were calculated and utilized for most analyses. The inclusion of frequency data in the IV metric helped to overcome the underweighting of tree data.

CONCLUSIONS

Plant communities are complex and are highly sensitive to perturbations in the surrounding landscape, which makes them simultaneously difficult to model and perfect as indicators of change. Any change in the biogeochemical cycle shows up quickly, if one only knows what to look for. Cypress ecosystems are an excellent study model as their vegetation is moderately consistent from site to site. Unfortunately, though, community changes may not always be obvious, as they most likely involve the loss of the highest quality, most sensitive, and rarest species. Changes in the composition of the hardiest and most abundant species are more readily observed and most likely occur much later in the disturbance cycle.

Discussions of ecosystem disturbance must also be intimately connected with questions of scale. While multiple scales were not assessed in this study, the scale of ecosystem disturbances may affect such important processes as the dispersal and establishment of non-native species. Disturbance occurring over larger areas may cause greater resource availability, slower recolonization of the disturbed area, and the establishment of weedy colonizer species. Smaller scale disturbances may in turn facilitate non-native species that are typically associated with later successional stages (Pauchard and Shea 2006). The assessment of a variety of temporal scales may be the only way to understand the degrees of change occurring in plant communities impacted by localized disturbances.

Several other key points arose during the course of this study regarding the use of the disturbance indices. A thorough study of the impacts of human disturbance might attempt to quantify human impact, resulting abiotic change, and biotic / community change for each ecosystem. These studies can lead to a great increase in the depth of understanding in this field. The utility and scope of metrics of disturbance can also be expanded. For example, LDI may give quantitative information on the disturbance gradient, however, there is enough understanding on the ways that specific human activities change ecosystems, that teasing apart indices like LDI may be worthwhile. The retention metric in this study better explained shifts in some plant community features than a total metric like LDI could. This would be particularly beneficial in assessing the ability of wetlands to continue to perform the host of services for which we depend on them.

In conclusion, LDI is a useful predictor of quality, but its role and usefulness has yet to be fully determined. So far, the true utility of LDI is in the quick remote assessment capabilities it provides. While comprehensive metrics like LDI cannot tell us about the specifics of the causes of impacts to natural communities, in conjunction with the development of more specific metrics, the use of different scales of observation, and the use of powerful measurements of biotic community integrity, it has the potential to be a powerful tool. An expansion to this study may involve the modeling of trajectories of wetland plant communities under different types and intensities of human development and at variable temporal and spatial scales. It would seem that very different floral communities would result from different types and

intensities of development and the mapping of likely outcomes may be of great benefit to restoration efforts as well. More work is certainly needed on the long term effects of urbanization, largely to better understand the abiotic and resulting biotic changes caused by urbanization through time

APPENDIX A: LIST OF SPECIES IDENTIFIED

Family	Species
Acanthaceae	<i>Hygrophila polysperma</i> (Roxb.)T.Anderson
Adoxaceae	<i>Sambucus nigra</i> L. subsp. <i>canadensis</i> (L.)Bolli
Alismataceae	<i>Sagittaria graminea</i> Michx.
Alismataceae	<i>Sagittaria lancifolia</i> L.
Alismataceae	<i>Sagittaria</i> sp.
Anacardiaceae	<i>Schinus terebinthifolius</i> Raddi
Anacardiaceae	<i>Toxicodendron radicans</i> (L.)Kuntze
Apiaceae	<i>Oxypolis filiformis</i> (Walter)Britton
Apiaceae	<i>Ptilmnum capillaceum</i> (Michx.)Raf.
Apocynaceae	<i>Nerium oleander</i> L.
Aquifoliaceae	<i>Ilex cassine</i> L.
Araceae	<i>Allocasia</i> sp.
Araceae	<i>Epipremnum pinnatum</i> (L.)Engl.
Araceae	<i>Xanthosoma sagittifolium</i> (L.)Schott
Arecaceae	<i>Serenoa repens</i> (W.Bartram)Small
Araliaceae	<i>Centella asiatica</i> (L.)Urb.
Araliaceae	<i>Hydrocotyle</i> sp.
Arecaceae	<i>Sabal palmetto</i> (Walter)Lodd. ex Schult. & Schult.f.
Asteraceae	<i>Baccharis halimifolia</i> L.
Asteraceae	<i>Baccharis</i> sp.
Asteraceae	<i>Bidens mitis</i> (Michx.)Sherff
Asteraceae	<i>Bigelowia nudata</i> (Michx.)DC. subsp. <i>australis</i> L.C. Anderson
Asteraceae	<i>Erechtites hieracifolius</i> (L.)Raf. ex DC
Asteraceae	<i>Eupatorium capillifolium</i> (Lam.)Small ex Porter & Britton
Asteraceae	<i>Mikania scandens</i> (L.)Willd.
Asteraceae	<i>Pluchea foetida</i> (L.)DC.
Asteraceae	<i>Verbesina virginica</i> L.
Begoniaceae	<i>Begonia cucullata</i> Willd.
Blechnaceae	<i>Blechnum serrulatum</i> Rich.
Blechnaceae	<i>Woodwardia aerolata</i> (L.)T.Moore
Blechnaceae	<i>Woodwardia virginica</i> (L.)Sm.
Bromeliaceae	<i>Tillandsia</i> sp.
Bromeliaceae	<i>Tillandsia usneoides</i> (L.)L.
Caryophyllaceae	<i>Drymaria cordata</i> (L.)Willd. ex Schult
Clusiaceae	<i>Hypericum fasciculatum</i> Lam.
Commelinaceae	<i>Commelina diffusa</i> Burm.f
Commelinaceae	<i>Commelina</i> sp.

Cornaceae	<i>Nyssa sylvatica</i> Marshall var. <i>biflora</i> (Walter)Sarg.
Cornaceae	<i>Persea palustris</i> (Raf.)Sarg.
Cucurbitaceae	<i>Momordica charantia</i> L.
Cupressaceae	<i>Taxodium ascendens</i> Brongn.
Cupressaceae	<i>Taxodium distichum</i> (L.)Rich.
Cupressaceae	<i>Taxodium</i> hybrid
Cyperaceae	<i>Carex glaucescens</i> Elliott
Cyperaceae	<i>Carex</i> sp.
Cyperaceae	<i>Cyperus croceus</i> Vahl.
Cyperaceae	<i>Cyperus enterianus</i> Boeck
Cyperaceae	<i>Cyperus haspan</i> L.
Cyperaceae	<i>Cyperus lecontei</i> Torr. ex Steud.
Cyperaceae	<i>Eleocharis cellulosa</i> Torr.
Cyperaceae	<i>Eleocharis equisetoides</i> (Elliott)Torr.
Cyperaceae	<i>Eleocharis flavescens</i> (Poir.)Urb.
Cyperaceae	<i>Rhynchospora glomerata</i> (L.)Vahl
Cyperaceae	<i>Rhynchospora inundata</i> (Oakes)Fernald
Cyperaceae	<i>Rhynchospora leptocarpa</i> (Chapm. ex Britton)Small
Cyperaceae	<i>Rhynchospora microcephala</i> (Britton)Britton ex Small
Cyperaceae	<i>Rhynchospora rariflora</i> (Michx.)Elliott
Cyperaceae	<i>Rhynchospora</i> sp.
Cyperaceae	<i>Rhynchospora wrightiana</i> Boeck.
Cyperaceae	<i>Scleria reticularis</i> Michx.
Cyperaceae	unknown
Dioscoreaceae	<i>Dioscorea bulbifera</i> L.
Ericaceae	<i>Lyonia lucida</i> (Lam.)K.Koch
Ericaceae	unknown
Eriocaulaceae	<i>Eriocaulon decangulare</i> L.
Eriocaulaceae	<i>Eriocaulon</i> sp. 1
Eriocaulaceae	<i>Eriocaulon</i> sp. 2
Euphorbiaceae	<i>Phyllanthus urinaria</i> L.
Euphorbiaceae	<i>Sapium sebiferum</i> (L.)Roxb.
Fabaceae	<i>Desmodium incanum</i> D.C.
Fagaceae	<i>Quercus laurifolia</i> Michx.
Fagaceae	<i>Quercus nigra</i> L.
Fagaceae	<i>Quercus</i> sp.
Haemodoraceae	<i>Lachnanthes carolina</i> (Lam.)Dandy
Haloragaceae	<i>Proserpinaca palustris</i> L.
Haloragaceae	<i>Proserpinaca pectinata</i> Lam.
Juncaceae	<i>Juncus canadensis</i> J.Gay ex. Laharpe

Juncaceae	<i>Juncus effusus</i> L. subsp. <i>solutus</i> (Fernald & Weigand) Hämet-Ahti
Juncaceae	<i>Juncus</i> sp.
Lamiaceae	<i>Callicarpa americana</i> L.
Lamiaceae	<i>Lycopus rubellus</i> Moench
Lauraceae	<i>Cinnamomum camphora</i> (L.)J.Presl
Lentibulariaceae	<i>Utricularia foliosa</i> L.
Lentibulariaceae	<i>Utricularia purpurea</i> Walter
Lentibulariaceae	<i>Utricularia</i> sp.
Lycopodiaceae	<i>Lycopodiella alopecuroides</i> (L.)Cranfill
Magnoliaceae	<i>Magnolia virginiana</i> L.
Malvaceae	<i>Urena lobata</i> L.
Melastomataceae	<i>Rhexia mariana</i> L.
Menyanthaceae	<i>Nymphoides aquatica</i> (J.F.Gmel.)Kuntze
Myricaceae	<i>Myrica cerifera</i> L.
Nephrolepidaceae	<i>Nephrolepis exaltata</i> (L.)Schott
Nymphaeaceae	<i>Nuphar advena</i> (Aiton)Aiton f.
Nymphaeaceae	<i>Nymphaea odorata</i> Aiton
Onagraceae	<i>Ludwigia linearis</i> Walter
Onagraceae	<i>Ludwigia octovalvis</i> (Jacq.)P.H.Raven
Onagraceae	<i>Ludwigia peruviana</i> (L.)H.Hara
Onagraceae	<i>Ludwigia pilosa</i> Walter
Onagraceae	<i>Ludwigia repens</i> J.R.Forst.
Onagraceae	<i>Ludwigia</i> sp.
Osmundaceae	<i>Osmunda cinnamomum</i> L.
Osmundaceae	<i>Osmunda regalis</i> L. var. <i>spectabilis</i> (Willd.)A.Gray
Oxalidaceae	<i>Oxalis</i> sp.
Pinaceae	<i>Pinus elliotii</i> Engelm.
Pinaceae	<i>Pinus serotina</i> Michx.
Pinaceae	<i>Pinus</i> sp.
Poaceae	<i>Amphicarpum muhlenbergianum</i> (Schult.)Hitchc.
Poaceae	<i>Andropogon brachystachyus</i> Chapm.
Poaceae	<i>Andropogon glomeratus</i> (Walter)Britton et al.
Poaceae	<i>Andropogon</i> sp.
Poaceae	<i>Aristida palustris</i> (Chapm.)Vasey
Poaceae	<i>Coelorachis tuberculosa</i> (Nash)Nash
Poaceae	<i>Dicanthelium</i> sp.
Poaceae	<i>Dichantherium laxiflorum</i> (Lam.)Gould
Poaceae	<i>Dichantherium strigosum</i> (Muhl. ex Elliott)Freckmann
Poaceae	<i>Oplismenus hirtellus</i> (L.)P.Beauv.
Poaceae	<i>Panicum dichomotiflorum</i> Michx.

Poaceae	<i>Panicum hemitomom</i> Schult.
Poaceae	<i>Panicum sp. 1</i>
Poaceae	<i>Panicum sp. 2</i>
Poaceae	<i>Panicum sp. 3</i>
Poaceae	<i>Panicum verrucosum</i> Muhl.
Poaceae	<i>Paspalum conjugatum</i> P.J.Bergius
Poaceae	<i>Paspalum repens</i> P.J. Bergius
Poaceae	<i>Sacciolepis indica</i> (L.)Chase
Poaceae	<i>Tripsacum dactyloides</i> (L.)L.
Poaceae	unknown
Poaceae	<i>Urochloa mutica</i> (Forsk.)T.Q.Nguyen
Polygonaceae	<i>Polygonum hydropiperoides</i> Michx.
Polygonaceae	<i>Polygonum punctatum</i> Elliott
Polygonaceae	<i>Polygonum setaceum</i> Baldwin
Polygonaceae	<i>Polygonum sp.</i>
Polypodiaceae	<i>Phlebodium aureum</i> (L.)J.Sm.
Pontederiaceae	<i>Pontederia cordata</i> L.
Portulacaceae	<i>Portulaca oleracea</i> L.
Rosaceae	<i>Rubus argutus</i> Link.
Rosaceae	<i>Rubus sp.</i>
Rubiaceae	<i>Cephalanthus occidentalis</i> L.
Rubiaceae	<i>Diodia virginiana</i> L.
Rubiaceae	<i>Galium tinctorium</i> L.
Salicaceae	<i>Salix caroliniana</i> Michx.
Salviniaceae	<i>Salvinia minima</i> Baker
Sapindaceae	<i>Acer rubrum</i> L.
Sapindaceae	<i>Koelreuteria elegans</i> (Seem.)A.C.Sm. <i>subsp. Formosana</i> (Hayata)F.G.Mey.
Saururaceae	<i>Saururus cernuus</i> L.
Smilacaceae	<i>Smilax auriculata</i> Walter
Smilacaceae	<i>Smilax laurifolia</i> L.
Smilacaceae	<i>Smilax sp.</i>
Smilacaceae	<i>Smilax walteri</i> Pursh
Solanaceae	<i>Solanum viarum</i> Dunal
Sphagnaceae	<i>Sphagnum sp.</i>
Theaceae	<i>Gordonia lasianthus</i> (L.)J.Ellis
Thelypteridaceae	<i>Thelypteris dentata</i> (Forssk.)E.P.St.John
Thelypteridaceae	<i>Thelypteris hispidula</i> (Decne.)C.F.Reed var. <i>versicolor</i> (R.P.St.John)Lellinger
Typhaceae	<i>Typha domingensis</i> Pers.
unknown	unknown
Urticaceae	<i>Parietaria floridana</i> Nutt.

Urticaceae	<i>Parietaria praetermissa</i> Hinton
Veronicaceae	<i>Bacopa caroliniana</i> (Walter)B.L.Rob.
Veronicaceae	<i>Gratiola ramosa</i> Walter
Veronicaceae	<i>Micranthemum glomeratum</i> (Chapm.)Shinners
Vitaceae	<i>Parthenocissus quinquefolia</i> (L.)Planch.
Vitaceae	<i>Vitis rotundifolia</i> Michx.
Xyridaceae	<i>Xyris ambigua</i> Beyr. ex Kunth
Xyridaceae	<i>Xyris elliotii</i> Chapm.
Xyridaceae	<i>Xyris fimbriata</i> Elliott
Xyridaceae	<i>Xyris sp.</i>

APPENDIX B: DATA

Table 5. Cypress dome data

Dome	# of plots	size	Area	Samp_Date	Location
174	10	L	7548.61	10/28/2008	TM Ranch
179	2	S	1851.51	10/15/2008	TM Ranch
193	2	S	1579.80	10/15/2008	TM Ranch
565	20	L	16118.55	5/8/2008	Beeline
612	13	L	10515.07	5/23/2008	Hal Scott N, Bassett/Seaview
679	4	S	2792.70	10/26/2008	Hal Scott N, Hampshire/Reynolds
913	14	L	10701.74	11/1/2008	Hal Scott N, Archer/Peabody
986	3	S	2065.09	10/26/2008	Hal Scott N, Bancroft/Oberly
1035	14	L	11195.93	7/22/2008	Andover Lakes, Fairhaven
1038	6	S	4703.84	5/9/2008	Hal Scott N, Wembly
1086	8	L	7395.09	6/2/2008	Andover Lakes, Curry Ford
1091	23	L	18076.40	9/20, 10/11	Hidden Hollow, Curry Ford
1181	10	L	8976.80	10/30/2008	Dean/Curry Ford - Branchwater
1192	10	L	7468.69	7/26/2008	Hal Scott Preserve
1252	12	L	9038.21	5/21 - 5/22	Eastwood Golf Course
1398	4	S	1576.93	10/28/2008	Avalon Park
1459	5	S	3837.14	10/15/2008	Bithlo, Hollister
1462	4	S	2784.72	10/19/2008	S 419 and Colonial
1852	12	L	9377.21	10/7/2008	Ranch/Econ Forest
1854	10	L	7496.55	10/21/2008	Yarborough Ranch
1928	19	L	14887.57	10/18/2008	Sutton St., Seminole
2092	3	S	2202.63	11/1/2008	I-Drive, Orange
2169	6	S	4334.51	9/20/2008	Off Central Florida Pkwy
2248	11	L	8436.57	10/26/2008	Hal Scott N, Williston/Coronet
2265	4	S	2761.93	7/25/2008	Hal Scott Preserve
2283	15	L	11504.29	10/23/2008	John Young Parkway S
2285	4	S	3266.28	7/16/2008	Andover Lakes, Curry Ford
2292	2	S	1877.37	5/9/2008	Hal Scott N, Moorgate
2315	3	S	2433.59	10/28/2008	Eastwood, neighborhood
2373	3	S	2692.17	10/19/2008	Osceola and Riverfront

Table 6. Disturbance Index data by dome

Dome	Class	LDI	exotic_per	exotic_num	retention	TSD_LU	TSD_LDI
174	2	1.5667	0.00	0	0.00	58.00	90.87
179	2	1.8451	0.00	0	0.00	58.00	107.02
193	2	1.9589	0.00	0	0.00	58.00	113.62
565	1	2.4384	0.00	0	0.00	8.30	68.71
612	3	4.4399	0.00	0	5136.23	4.43	30.58
679	3	3.0472	0.00	0	0.00	1.32	9.10
913	3	2.1085	0.00	0	0.00	3.11	21.45
986	1	1.1320	0.00	0	1057.35	0.67	4.61
1035	4	3.9520	0.09	4	5947.73	4.95	26.25
1038	5	3.9576	0.00	0	4037.36	6.04	28.82
1086	4	4.4522	0.00	0	5067.15	3.13	24.38
1091	4	7.1346	0.25	27	2470.74	18.59	139.78
1181	5	7.2519	0.00	0	5156.34	15.56	115.59
1192	1	1.0378	0.00	0	0.00	0.64	1.17
1252	5	6.2486	0.08	6	8151.33	13.95	97.94
1398	3	3.5880	0.00	0	0.00	1.17	8.51
1459	5	7.4700	0.00	0	0.00	31.00	231.57
1462	4	6.4703	0.08	2	1063.02	223.56	1661.29
1852	2	3.1455	0.10	9	0.00	0.00	0.00
1854	2	3.6039	0.16	12	0.00	0.00	0.00
1928	3	4.8493	0.01	1	0.00	0.00	0.00
2092	1	1.2915	0.00	0	0.00	0.45	3.36
2169	4	7.3105	0.25	5	26131.34	25.95	190.02
2248	1	1.7944	0.00	0	3.09	0.81	5.58
2265	1	1.0000	0.00	0	0.00	0.00	0.00
2283	5	5.5717	0.15	9	11481.64	14.95	113.43
2285	4	7.7141	0.04	1	453.57	19.89	153.50
2292	3	6.0846	0.00	0	0.00	10.34	71.36
2315	5	5.3177	0.00	0	771.10	7.15	52.50
2373	2	4.8107	0.19	4	0.00	0.00	0.00

Table 7. Plant Data by dome

Dome	exotic %	exotic #	mean_CC	weighted_CC	D_eff	H_eff	richness	wetland rank
174	0.00	0	5.0668	5.1594	2.72	4.40	19.00	1.210
179	0.00	0	5.0924	5.3355	6.19	7.72	17.00	1.423
193	0.00	0	5.3667	4.5621	1.12	1.33	6.00	1.071
565	0.00	0	4.9585	4.8831	3.95	7.01	33.00	1.528
612	0.00	0	4.2971	4.0099	6.00	8.63	24.00	1.747
679	0.00	0	5.0089	5.0064	7.40	8.85	18.00	1.196
913	0.00	0	5.1091	5.3106	7.83	10.83	33.00	1.344
986	0.00	0	5.4933	5.2526	3.31	4.74	12.00	1.281
1035	0.09	4	4.6073	4.3801	3.51	4.95	15.00	1.291
1038	0.00	0	4.1286	3.7743	2.61	3.36	14.00	2.030
1086	0.00	0	5.1263	4.9857	5.76	7.35	16.00	1.077
1091	0.25	27	2.7371	2.8119	11.83	16.39	42.00	1.601
1181	0.00	0	4.5679	4.8065	6.53	10.55	38.00	1.569
1192	0.00	0	4.7100	5.6926	2.38	4.00	20.00	1.458
1252	0.08	6	3.6928	4.4467	7.28	9.68	29.00	1.851
1398	0.00	0	5.4250	5.0936	4.07	5.01	12.00	1.118
1459	0.00	0	4.9900	3.6177	2.33	2.77	10.00	1.500
1462	0.08	2	4.2917	3.9136	4.69	5.80	12.00	1.500
1852	0.10	9	3.5137	4.3107	11.40	17.02	46.00	2.239
1854	0.16	12	3.8303	3.0497	8.05	12.07	33.00	2.049
1928	0.01	1	4.4942	4.3670	4.47	6.39	19.00	1.779
2092	0.00	0	4.9429	4.8333	3.21	3.99	7.00	1.000
2169	0.25	5	3.5895	3.5617	3.35	5.08	19.00	2.206
2248	0.00	0	5.1700	5.0437	10.18	14.69	32.00	1.290
2265	0.00	0	3.9788	3.3963	4.47	6.54	16.00	1.868
2283	0.15	9	3.2483	4.2379	3.00	5.47	29.00	1.879
2285	0.04	1	3.8653	4.2002	5.84	8.18	17.00	1.130
2292	0.00	0	4.2667	4.4466	4.57	6.64	15.00	1.367
2315	0.00	0	4.9808	4.7908	4.76	5.97	12.00	1.167
2373	0.19	4	3.1950	4.4069	5.25	6.57	16.00	1.525

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