- Pilot Study -

The Florida Wetland Condition Index (FWCI): Preliminary Development of Biological Indicators for Forested Strand and Floodplain Wetlands

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Kelly Chinners Reiss
and
Mark T. Brown

Howard T. Odum Center for Wetlands
University of Florida
Gainesville, Florida 32611-6350

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EXECUTIVE SUMMARY

PILOT STUDY
THE FLORIDA WETLAND CONDITION INDEX (FWCI): PRELIMINARY DEVELOPMENT OF BIOLOGICAL INDICATORS FOR FORESTED STRAND AND FLOODPLAIN WETLANDS

Over 30 years ago, the federal Water Pollution and Control Act obliged states to protect and restore the chemical, physical, and biological integrity of waters, and charged states with establishing water quality standards for all waters within state boundaries including wetlands. Criteria for defining water quality could be narrative or numeric, and it could be addressed through chemical, physical, or biological standards. Initially, states used chemical and physical criteria (testing waters for chemical concentrations or physical conditions that exceeded criteria), assuming losses in ecosystem integrity if the criteria were exceeded (Danielson 1998). The United States Environmental Protection Agency (USEPA) recognized the potential of biological criteria to assess water quality standards and in the late 1980s required states to use biological indicators to accomplish the goals of the Clean Water Act (USEPA 1990). In effect, biological assessment has evolved into one of the standard monitoring tools of water resource-protection agencies over the past 2 decades (Gerristen et al. 2000). Such biological assessment programs have been created for lakes and streams throughout the United States (Barbour et al. 1996a; Karr and Chu 1999; Gerristen et al. 2000), and more recently efforts to assess wetland condition have been initiated (Mack 2001; USEPA 2002). Within Florida, biological indices have been created based on macroinvertebrate community composition for streams (Barbour et al. 1996a; Fore 2004), lakes (Gerristen and White 1997), and isolated depressional freshwater herbaceous (Lane et al. 2003) and forested (Reiss and Brown 2005) wetlands. Biological indices have also been created based on the community composition of the diatom and macrophyte assemblages for Florida freshwater herbaceous and forested wetlands (Lane et al. 2003; Reiss and Brown 2005).

The primary objective of this research was to develop a preliminary Florida Wetland Condition Index (FWCI) for forested strand and floodplain wetlands. Wetland study sites were sought in various *a priori* designated land use categories that included natural, agricultural, and urban land uses. An independent measure of anthropogenic activity in the landscape was calculated for each wetland using the Landscape Development Intensity index (LDI) (Brown and Vivas 2005). The contribution of this research to our understanding of changes in the macrophyte community composition of forested strand and floodplain wetlands in relation to different anthropogenic activities in the surrounding landscape can be summarized in five main points:

1. Five macrophyte based metrics including proportion tolerant indicator species, proportion sensitive indicator species, Floristic Quality Assessment Index (FQAI) score, proportion exotic species, and proportion native perennial species, were useful biological indicators for defining biological integrity for forested strand and floodplain wetland vegetation;

2. Vegetation richness, evenness, and diversity were not sensitive to *a priori* land use categories or development intensities in the surrounding landscape for forested strand and floodplain wetlands;
3. The Landscape Development Intensity (LDI) index was a useful tool correlating with the measured biological condition of vegetation for forested strand and floodplain wetlands;
4. Regional species lists for metrics would enhance the forested strand and floodplain Florida Wetland Condition Index (FWCI);
5. An FWCI with a set of core metrics could be developed for Florida freshwater wetlands, which includes separate species lists for indicator species by wetland type and ecoregions and separate Floristic Quality Assessment Index (FQAI) scores for species by wetland type.

The FWCI provided a quantitative measure of the biological integrity of forested strand and floodplain wetlands in Florida. Comprised of five metrics, the FWCI was developed based on the community composition of the macrophyte species assemblages. Metrics were selected for inclusion in the FWCI based on the correlation (nonparametric Spearmans correlation coefficient) of each metric with a quantitative gradient of Landscape Development Intensity (LDI); based on a metrics visually distinguishable correlation with LDI in a scatter plot; and based on a statistical difference of metric values between low and high LDI groups (Mann-Whitney U-test). The FWCI was composed of individual metrics, which were scaled and added together, creating the preliminary forested strand and floodplain wetland FWCI (0-50 scale), with the highest score of 50 reflecting the highest biological integrity and the lowest score of zero reflecting a lack of biological integrity or no similarity to the reference wetland condition.

The five macrophyte metrics that met the three selection criteria (Spearmans correlation coefficient ($|r| \geq 0.50$, $p < 0.05$), visually distinguishable scatter plots, and Mann-Whitney U-test between LDI groups ($p<0.10$)) were proportion tolerant indicator species; proportion sensitive indicator species; Floristic Quality Assessment Index (FQAI) score; proportion exotic species; and proportion native perennial species.

Forested Strand and Floodplain FWCI Metrics for Wetland Vegetation
1. Proportion Tolerant Indicator Species
2. Proportion Sensitive Indicator Species
3. Floristic Quality Assessment Index (FQAI) Score
4. Proportion Exotic Species
5. Proportion Native Perennial Species

The variable sensitivities of three different independently derived indices compared to the forested strand and floodplain FWCI, including the Landscape Development Intensity index (LDI; Lane et al. 2003; Brown and Vivas 2005), the Human Disturbance Gradient (HDG; Fore 2004), and the Stream Condition Index (SCI; Fore 2004), suggest that multiple measures of biological integrity may be more effective at describing ecosystem wide biological integrity than any single measure based on an individual species assemblage or surrounding land use activity. However, the strong

correlations among FWCI, LDI, and HDG (Spearmans correlation coefficient $|r| \geq 0.58$, $p < 0.05$), and lack of correlations of SCI with both FWCI and LDI, suggest that in-stream macroinvertebrate based measures of biological condition and surrounding forested wetland macrophyte based measures of biological condition did not respond in a consistent manner to changes in anthropogenic activity. Using both the in-stream macroinvertebrate SCI biological assessment and the surrounding wetland macrophyte FWCI biological assessment methods may provide a more complete picture of the overall condition of a wetland and associated stream at a particular spatial location. While agreement in the ranking of the biological condition of study wetlands using the FWCI and SCI was anticipated, discrepancies among the ranking from the different assemblages may provide great insight into biological condition as different species assemblages respond to changes in anthropogenic activities and the associated changes in inflows (e.g. nutrient enrichment) over different time scales. Additionally, use of the forested strand and floodplain FWCI may lead to specific conclusions as to the biological condition of local or nearby anthropogenic activity, while use of the SCI may enhance understanding of larger watershed scale influences from anthropogenic activity (i.e. due to the convergence of water within the watershed associated with stream flow).

The quantitative score of biological integrity established through the FWCI can be used as an objective, quantitative means of comparing changes in macrophyte community composition for wetlands, including those impacted by varying degrees of anthropogenic influence. While the forested strand and floodplain FWCI for flowing water systems can not be used to predict changes in the physical and chemical parameters of a wetland, its strength lies in providing an overview of biological integrity through the integration of changes in macrophyte community composition from cumulative effects.

CHAPTER 1
INTRODUCTION AND OVERVIEW

Assessment techniques for categorizing ecosystem condition have been established for many Florida ecosystems, including freshwater lakes (Lake Condition Index, LCI: Gerristen and White 1997), streams (BioRecon and Stream Condition Index, SCI: Fore 2004), and depressional freshwater wetlands (Florida Wetland Condition Index, Lane et al. 2003; Reiss and Brown 2005). This research furthers the development of the Florida Wetland Condition Index (FWCI), establishing preliminary metrics with species lists specific to forested strand and floodplain wetlands, referred to as flowing water systems.

The overall goal of the FWCI is to use changes in community composition to characterize the biological condition of wetland ecosystems. Wetland condition is scored on a numeric scale developed from the addition of scores from individual metrics. A metric is defined as a biological attribute having a consistent and predictable response to anthropogenic activities (Karr and Chu 1997). Each metric represents an indication of biological integrity, or a signal of ecosystem condition, based on a change in community composition from the reference standard condition. The reference standard condition is defined as the condition of wetlands surrounded by undeveloped landscapes and without apparent human induced alterations. By designating a measure of ecosystem condition we refer to what others have described as ecosystem integrity, defined by Karr and Dudley (1981) as "the ability of an aquatic ecosystem to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitats of the region."

Forested Strand and Floodplain Wetlands

Wetlands have been categorized in many different ways based on any number of community attributes including, but not limited to, dominant vegetation, hydrology, soil type, and location in the landscape (Mitsch and Gosselink 1993; Keddy 2000; Kent 2000). One of the most widely recognized classification systems in North America is that by Cowardin et al. (1979). Our study focused on what Cowardin et al. (1979) categorize palustrine wetlands, commonly described as freshwater marshes and swamps. More specifically, palustrine wetlands are defined as nontidal wetland ecosystems with trees, shrubs, persistent emergents, or emergent mosses or lichens as the dominant vegetation type, and tidal wetlands with these vegetation types with ocean-derived salinity levels below 0.5‰ (Cowardin et al. 1979). Palustrine wetlands occur throughout the landscape as small, shallow, permanent or intermittent water bodies; shoreward of lakes, river channels, or estuaries; on river floodplains; in isolated catchments; on slopes; or as islands in lakes or rivers (Cowardin et al. 1979). The category of palustrine wetlands includes eight classes: aquatic bed, emergent, forested, moss-lichen, rock bottom, scrub-shrub, unconsolidated bottom, and unconsolidated shore.

Wetlands for this study were primarily in the forested wetland class, which includes wetlands in all water regimes, except subtidal wetlands, that are characterized by

woody vegetation 6m tall or greater. The structure of palustrine forested wetlands typically includes an overstory of trees with an understory of young trees and shrubs and an understory of herbaceous species (Cowardin et al. 1979). The class of palustrine forested wetlands includes six subclasses and dominance types, including broad-leaved deciduous, needle-leaved deciduous, broad-leaved evergreen, needle-leaved evergreen, dead, and indeterminate deciduous. Four categories of water modifiers are used to describe the palustrine forested wetlands in this study (as categorized on the National Wetlands Inventory GIS coverage available from the Florida Geographic Data Library at http://www.fgdl.org) including temporarily flooded, seasonally flooded, semipermanently flooded, and permanently flooded. During the growing season, temporarily flooded wetlands have surface water present for brief periods with a water table typically well below the soil surface. Vegetation in temporarily flooded wetlands consists of facultative species, including those that grow in both uplands and wetlands. Early in the growing season seasonally flooded wetlands have surface water standing for extended periods most often without surface water late in the season but with a water table near the soil surface. Similarly, semipermanently flooded forested wetlands generally have standing surface water throughout the growing season and a water table at or near the soil surface when not flooded. At the flood extreme, permanently flooded wetlands have standing surface water throughout the year with a vegetation community of obligate wetland species (Cowardin et al. 1979).

Within the text *Ecosystems of Florida* (Myers and Ewel, eds. 1990), Ewel (1990) describes approximately 10 distinctive types of swamps. The freshwater forested strands in this study most closely resemble wetlands in the cypress pond and strand category. Wharton et al. (1976) described cypress strands as "a diffuse freshwater stream flowing through a shallow depression on a greatly sloping plain." While Mitsch and Gosselink (1993) suggest that cypress strands are found primarily in south Florida, Ewel (1990) notes that cypress strands are common throughout Florida and are found where water flow is sufficient to create a depression channel in areas with little slope but where actual flow is seldom observed. The definition of forested strands for this study broadly encompasses all of these definitions with the primary distinction of forested strands including evidence of channelized flow (Figure 1-1), though actual flow was rarely observed at any of the strands during the 2003 growing season during the period of sampling.

The forested floodplain wetlands in this study most closely resemble the river swamp category in *Ecosystems of Florida* (Myers and Ewel, eds. 1990); however the floodplain forests in this study were associated with smaller river systems than those typically characterized by Ewel (1990). Forested floodplain wetlands in this study were associated with low order streams and rivers and were not associated with the main channels of the largest river systems in Florida (ex. Apalachicola, Suwannee, etc.). The forested floodplain wetlands in this study can also be categorized as riparian wetlands such named for the influence on the wetland environment by the adjacent stream or river system. Mitsch and Gosselink (1993) note vast differences among riparian wetlands, with the common link being the interconnection between the riparian zone, the river or stream, and the adjacent upland environment. Riparian wetlands in the southeastern United States are characterized by low-lying, low slope, and broad floodplain areas with seasonally pulsing hydrologic influences on well developed soils (Mitsch and Gosselink



Figure 1-1. Photograph showing the interior of a freshwater forested strand wetland in Osceola County, Florida. The dark organic layer in the center of the photo shows evidence of flowing water during times of high water.

1993). The floodplain wetlands in this study are similar to the strands, however standing water was always observed in the channelized stream (Figure 1-2).

The average hydroperiod (the seasonal pattern and length of saturated soils or standing water level during a year (Ewel 1990; Mitsch and Gosselink 1993)) varies among strand and floodplain wetlands, with strands having a moderate length hydroperiod with saturated soils or standing water for six to nine months a year and floodplain forests having a short length hydroperiod with generally less than six months of saturated soils or standing water during a year (Ewel 1990). Fire frequency in forested strand and floodplain wetlands ranges from moderate frequency (approximately one per 20 years) for strands to low frequency (approximately one per 100 years) for floodplain wetlands (Ewel 1990). Strands and floodplains also differ in their organic matter accumulation depths, with strands having high organic matter accumulation with an organic soil layer greater than 1 m deep and floodplain wetlands having low organic matter accumulation with an organic soil layer less than 1 m deep (Ewel 1990). Additionally, strand and floodplain wetlands have different primary water sources with strands receiving most water from shallow groundwater sources and the main source of water for floodplain wetlands from surface water originating from the associated stream or river (Ewel 1990).

Despite these differences in fire frequency, organic matter accumulation, and water source, the species composition is similar among forested strand and floodplain wetlands. Common shared tree species in strand and floodplain wetlands include *Acer*



Figure 1-2. Photograph showing the interior of a freshwater forested floodplain wetland in Polk County, Florida. The channelized stream is visible in the center of the photo showing the presence of a permanently flooded and flowing stream adjacent to the floodplain wetland.

rubrum (red maple), *Fraxinus caroliniana* (water ash), *Gordonia lasianthus* (loblolly bay), *Liquidambar styraciflua* (sweetgum), *Magnolia virginiana* (sweet bay), *Nyssa sylvatica* (black gum), *Persea palustris* (swamp bay), *Quercus laurifolia* (swamp laurel oak), *Sabal palmetto* (cabbage palm), *Salix caroliniana* (coastal plain willow), and *Taxodium distichum* (baldcypress). Additional tree species common to strands include *Annona glabra* (pond apple), *Pinus elliottii* (slash pine), and *Pinus palustris* (longleaf pine). The tree stratum of floodplain wetlands has greater species richness with additional common tree species including *Alnus serrulata* (hazel alder), *Betula nigra* (river birch), *Carpinus caroliniana* (American hornbeam), *Carya aquatica* (water hickory), *Carya glabra* (pignut hickory), *Celtis laevigata* (hackberry), *Chamaecyparis thyoides* (Atlantic white cedar), *Diospyros virginiana* (persimmon), *Fraxinus pennsylvanica* (green ash, red ash), *Fraxinus profunda* (pumpkin ash), *Gleditsia aquatica* (water locust), *Magnolia grandiflora* (southern magnolia), *Nyssa aquatica* (water tupelo), *Pinus glabra* (spruce pine), *Pinus taeda* (loblolly pine), *Planera aquatica* (planer tree), *Platanus occidentalis* (American sycamore), *Quercus lyrata* (overcup oak), *Quercus michauxii* (basket oak, swamp chestnut oak), *Quercus nigra* (water oak), *Quercus virginiana* (live oak), *Rhapidophyllum hystrix* (needle palm), *Sabal minor* (bluestem, dwarf palmetto), *Salix nigra* (black willow), and *Ulmus americana* (American elm).

Forested strand and floodplain wetlands also share a number of species in the shrub stratum including *Cephalantus occidentalis* (buttonbush), *Clethra alnifolia* (sweet pepperbush), *Cliftonia monophylla* (black titi), *Cyrilla racemiflora* (titi), *Ilex cassine* (dahoon holly), *Itea virginica* (Virginia willow), *Lyonia lucida* (fetterbush), *Myrica cerifera* (wax myrtle), and *Rubus argutus* (blackberry). Additional common shrub species in forested strands include *Chrysobalanus icaco* (coco plum), *Ilex glabra* (gallberry), *Leucothoe racemosa* (fetterbush), *Myrica heterophylla* (northern bayberry), *Myrsine guianensis* (myrsine), *Psychotria sulzneri* (wild coffee), *Psychotria undata* (wild coffee), and *Vaccinium arboretum* (sparkleberry). Other common shrub species in river swamps include *Aronia arbutifolia* (red chokeberry), *Crataegus marshallii* (parsley haw), *Ilex decidua* (possum haw), *Ilex vomitoria* (yaupon), *Leucothoe axillaries* (dog-hobble), *Rhododendron viscosum* (swamp honeysuckle), *Rubus betulifolius* (blackberry), *Sambucus canadensis* (elderberry), *Sebastiana fruticosa* (Sebastian bush), *Viburnum nudum* (swamp haw), and *Viburnum obovatum* (small viburnum, black haw).

The species composition of woody vines in strand and floodplain wetlands vary a great deal according to Ewel (1990), as there is only one shared species, *Smilax laurifolia* (bamboo-vine, catbrier). Common woody vine species in strands include *Ampelopsis arborea* (pepper vine), *Ficus aurea* (strangler fig), *Ficus citrifolia* (wild banyan tree), *Vitis aestivalis* (summer grape), and *Vitis shuttleworthii* (calusa grape); and common woody vine species in floodplain wetlands include *Ampelopsis arborea* (pepper vine), *Aster carolinianus* (climbing aster), *Smilax walteri* (coral greenbrier), *Toxicodendron radicans* (poison ivy), and *Vitis rotundifolia* (muscadine grape). Species lists for common tree, shrub, and woody vine species were adopted from Ewel (1990).

Forested strand and floodplain wetlands provide important habitat for wildlife such as invertebrates, amphibians, reptiles, birds, and mammals. Benthic invertebrates form the base of the forested wetland food chain, and water quality is strongly related to the diversity of the benthic macroinvertebrate community (Ewel 1990). Strands and floodplain wetlands provide valuable habitat for bird and mammal species characterized by low vegetation density and high cavity density. Though these wetlands differ somewhat in their relative contributions to bird and mammal habitat as strands have low canopy insect production, low production of edible fruits and seeds, and high presence of water, whereas forested floodplain wetlands have high canopy insect production, high production of edible fruits and seeds, and low presence of water (Ewel 1990).

Historical Perspective

Over 30 years ago, the Water Pollution and Control Act (later referred to as the Clean Water Act, 1972) required states to "restore and maintain the chemical, physical, and biological integrity of the Nation's waters" (USEPA 1990). This legislation included establishing water quality standards for all waters within state boundaries, including wetlands. Such water quality criteria could be qualitative or quantitative, and it could be addressed through chemical, physical, or biological standards. Initially, states used chemical and physical criteria (testing waters for chemical concentrations or physical conditions that exceeded known standards), assuming losses in ecosystem integrity if these standards were exceeded (Danielson 1998).

Several shortcomings have been noted when deriving ecosystem integrity based on exceeding established limits for chemical and physical parameters. Such criteria have been considered incomplete in their ability to reflect more than the temporal concentration of substances within a water body (Karr 1993). For instance, the use of toxicity parameters for determining ecosystem integrity may falsely indicate high ecosystem integrity when a single toxicity parameter went overlooked. This same water body could have elevated levels of other toxins or metals that went untested, or be physically altered so that it has lost functions typically associated with a fully functioning water body (Karr and Chu 1997). Furthermore, chemical and physical sampling may not occur during specific loading events and may therefore incompletely describe the ecological condition of the system. Adams (2002) points out that other environmental factors such as sedimentation, alterations to habitat, varying temperature and oxygen levels, and changes in ecological aspects like food availability and predator-prey relationships are not reflected with chemical criteria alone. James and Kleinow (1994) note that different organisms respond in different ways to the amount, persistence, and exposure of chemical compounds otherwise foreign to an organism; and single-valued chemical and physical criteria of water quality may overlook important biological implications.

Alternatively, biological indicators integrate the spatial and temporal effects of the environment on resident organisms, and are suitable for assessing the possible effects of multifaceted changes in ecosystems (Adams 2002). Karr and Chu (1997) and Adams (2002) note that biological indicators signal changes in the environment that might otherwise be overlooked or underestimated by methods that depend on chemical criteria alone. Organisms have an intricate relationship with their environment, reflecting current and cumulative ecosystem condition (Karr 1981). The presence of biological organisms reveals chemical exposure, expressing changes in the physical, chemical, and biological components of the ecosystem through changes in community composition (Adams 2002).

The United States Environmental Protection Agency (USEPA) recognized the potential of biological criteria to assess water quality standards and in the late 1980s required states to use biological indicators to accomplish the goals of the Clean Water Act (USEPA 1990). In effect, biological assessment has evolved into one of the standard monitoring tools of water resource protection agencies over the last two decades (Gerristen et al. 2000). Biological criteria and monitoring programs through the USEPA have been created for lakes and streams throughout the United States (Barbour et al. 1996a; Karr and Chu 1999; Gerristen et al. 2000), and more recently efforts to assess wetland condition have been initiated (USEPA 2002).

Indicators of Biological Integrity

Biological monitoring to assess ecosystem condition has been applied widely in ecological research. The primary aim of biological monitoring is to detect changes in abundance, structure, and diversity of target species assemblages. One trend in biological monitoring has led to the development of indices of biotic integrity (referred to as IBIs), for different species assemblages including diatoms (Fore and Grafe 2002; Fore 2004); macrophytes (Galatowitsch et al. 1999a; Gernes and Helgen 1999; Mack 2001; Lane

2003); macroinvertebrates (Kerans and Karr 1994; Barbour et al. 1996b); amphibians (Micacchion 2004); fish (Schulz et al. 1999); and birds (O'Connell et al. 1998).

Perhaps the most common species assemblage chosen for use in the development of IBIs is the macroinvertebrate assemblage, because many of the macroinvertebrate species rely entirely on the conditions of their aquatic environment for habitat, food, and reproductive activities. In Florida there are currently three biological indices that use the community composition of the macroinvertebrate assemblage to detect changes in biological integrity including the Lake Condition Index (LCI, Gerristen and White 1997), Stream Condition Index (SCI, Fore 2004), and the Florida Wetland Condition Index for depressional freshwater wetlands (FWCI, Lane et al. 2003; Reiss and Brown 2005). However, the forested strand and floodplain systems targeted in this study have varying hydrologic regimes from temporarily to permanently flooded, complicating macroinvertebrate collection due to variable hydrologic conditions. As such, we have chosen the macrophyte species assemblage for use in the preliminary forested strand and floodplain FWCI.

Wetland macrophytes are defined as aquatic emergent, submergent, or floating plants growing in or near water (USEPA 1998); and are described as distinguishing landscape features. The spatial distribution of macrophytes in the landscape occurs according to a multitude of factors, including hydroperiod, water chemistry, and substrate type, as well as other factors such as available seed source and climate. Fennessy et al. (2001) state that the community composition of wetland macrophytes typifies the physical, chemical, and biological wetland dynamic in time and space.

Macrophytes play a vital role in supporting the structure and function of wetlands by providing food and habitat for other assemblages including algae, macroinvertebrates, fish, amphibians, reptiles, birds, and mammals; and macrophyte populations can be used as a diagnostic tool to assess other aspects of the wetland environment. Crowder and Painter (1991) state that a lack of macrophytes where they are otherwise expected to grow suggests reduced wildlife populations from lack of food or cover and/or water quality concerns such as toxic chemical constituents, increased turbidity, or increased salinity. In contrast, an overgrowth of particular macrophytes may signify increased nutrient loading (USEPA 1998).

Many advantages of studying macrophytes as indicators of wetland condition have been noted, including their large, obvious size; ease of identification, to at least some useful taxonomic level; known response to toxicity tests; and general lack of ability to move to avoid unfavorable conditions (Danielson 1998; Cronk and Fennessy 2001). Additionally, macrophytes readily respond to changes in nutrient, light, toxic contaminant, metal, herbicide, turbidity, water, and salt levels. They can also be sampled in the field with transects, or remotely from aerial photography; and well-established field methods of sampling macrophytes exist (USEPA 2003). Furthermore, the USEPA (2003) states additional advantages of using the macrophyte assemblage, including that they do not require laboratory analysis, can easily be used for calculating simple abundance metrics, and are superb integrators of environmental condition. In general, macrophytes represent a useful assemblage for describing wetland condition (Mack 2001). Schindler (1987) alleges that macrophytes can provide a more integrated picture of wetland function than measures such as nutrient cycling, productivity, decomposition, or chemical and physical composition.

There are however some noted shortcomings of using macrophytes as biological indicators. These include the potential delay in response time for perennial shrub and tree species, difficulty identifying taxa to the species level in certain seasons and for some genera, uneven herbivory patterns, and varied pest-management practices (Cronk and Fennessy 2001). Despite these limitations, macrophytes have provided strong signals of anthropogenic influence (USEPA 2003). In fact, many states have begun using macrophytes in their wetland biological assessment programs, including, Minnesota (Galatowitsch et al. 1999a; Gernes and Helgen 1999), Montana (Apfelbeck 2000), North Dakota (Mushet et al. 2002), Ohio (Mack 2001), and Florida (Lane et al. 2003; Reiss and Brown 2005).

Quantifying Anthropogenic Influence

Wetlands occupy a large portion of the Florida landscape. An estimate from the 1780s reported 8,225,000 ha of wetlands in Florida (Dahl 2000). By the mid-1980s, the National Wetlands Inventory estimated Florida had 4,467,000 ha of wetlands remaining, translating into a loss in Florida of roughly 45% of the pre-1780s wetland area (Mitsch and Gosselink 1993; Dahl 2000). Throughout the continental United States, similar trends were apparent, with a drastic decline in the surface area of wetlands. More specifically, Dahl (2000) reported that 98% of all wetland losses throughout the continental United States from 1986 to 1997 were losses to freshwater wetlands. Of the remaining freshwater wetlands, 40% were adjacent to agricultural lands and therefore potentially affected by land use practices such as herbicide and pesticide application, irrigation, livestock watering and wastes, soil erosion, and deposition. An additional 17% were adjacent to urban or rural development. Freshwater non-tidal wetlands experienced the greatest development pressure just inland from coastlines as the demand for housing, transportation infrastructure, and commercial and recreational facilities increased (Dahl 2000). These changes in land use were proportionally more widespread in Florida than much of the continental United States due to the remarkable length of coastline along both the Atlantic Ocean and Gulf of Mexico coasts of Florida.

Agricultural and urban development activities influence an array of changes to the physical, chemical, and biological characteristics in nearby ecosystems. There have been numerous attempts at quantifying anthropogenic influence based on quantitative indices, for example, the Wetland Rapid Assessment Procedure (WRAP; Miller and Boyd 1999), the Minnesota disturbance index (Gernes and Helgen 1999), the Human Disturbance Gradient (HDG; Fore 2004), and the Landscape Development Intensity (LDI) index (Brown and Vivas 2005). Our study incorporates the LDI index as a measure of anthropogenic influence. Additionally, for thirteen floodplain wetlands, HDG scores have been obtained from previous studies (Fore 2004; Florida Department of Environmental Protection Geographic Information Systems map layers).

Landscape Development Intensity Index

The LDI index has been used as a gauge of human activity based on a development intensity measure derived from nonrenewable energy use in the surrounding landscape. The underlying concept behind calculating the LDI (quantifying the nonrenewable energy use per unit area in the surrounding landscape) stems from earlier

works by Odum (1995), who pioneered emergy analysis for environmental accounting. [Emergy is an environmental accounting term referring to expressing energy use in solar equivalents (Odum 1995).] Brown and Ulgiati (2005) suggest that landscape condition, or ecosystem health, is strongly related to the surrounding intensity of human activity, and that ecological communities are affected by the direct, secondary, and cumulative impacts of activities in the surrounding landscape. Healthy ecosystems are defined as those with integrity and sustainability, which correlate to limited development in the surrounding landscape and the maintenance of ecosystem structure and function, even when stressors (e.g. flooding, drought, etc.) are present (Brown and Ulgiati 2005).

The LDI scale encompasses a gradient from completely natural to highly developed land use intensity, and is calculated based on the percent of the area in a particular land use within the designated area surrounding the wetland (ex. 100 meters, 200 meters, etc.) multiplied times the LDI coefficient (Table 1-1), which is defined by the amount of nonrenewable energy use for a given land use (Brown and Vivas 2005). The LDI coefficient does not account for any individual causal agent directly, but instead, may represent the combined effects of air and water pollutants, physical damage, changes in the suite of environmental conditions (ex. groundwater levels, increased flooding), or a combination of such factors, all of which enter the natural ecological system from the surrounding developed landscape. Wetlands surrounded by more intense activities such as highways and multi-family residential land uses receive higher LDI index values, as the highest LDI coefficient of 10.0 is assigned to the urban land use category of Central Business District. Undeveloped land uses such as wetlands, lakes, and upland forests are assigned an LDI coefficient of 1.0, the lowest possible value, based on no use of nonrenewable energy in these ecosystems.

Human Disturbance Gradient

The Human Disturbance Gradient (HDG) is a quantitative measure used to assess the level of human disturbance to an ecosystem based on the cumulative score of four independent measures of the environment including ammonia concentration in the water (as mg N/L), hydrologic index, habitat assessment, and LDI for the buffer which included an area of 100 m on each side of the stream and 10 km upstream of the sampling point (Fore 2004). The first measure included in the HDG, ammonia concentration, is included as a summary variable defining water quality because of its consistent correlation with other water quality parameters (i.e. total phosphate) and its availability in the dataset (Fore 2004). Additionally, increases in ammonia concentration are thought to be evidenced from both agricultural (e.g., fertilizer and farming practices) and urban activities, whereas changes in total phosphorus concentrations are thought to be solely associated with agricultural operations (Fore 2004), so using ammonia concentration accounts for changes in both agricultural and urban land use activities.

The second measure included in the HDG is an estimate of hydrologic condition of the stream site, which is scored on-site at the discretion of the biologist conducting the sampling effort. Scores for the hydrologic condition range from 1-10, with a score of 1-2 Excellent representing a natural, undisturbed system with few impervious surfaces, high connectivity with ground water and surface features, and a natural flow regime (Fore 2004). Intermediate categories include 3-4 Good, 5-6 Moderate, and 7-8 Poor. The final category, 9-10 Very Poor, is reserved for those systems with a flow regime entirely

Table 1-1. Non-renewable energy use and LDI coefficients per land use used in the calculation of the LDI index.

Land Use	Nonrenewable Energy Use (E14 solar equivalent joules/ha/yr)	LDI Coefficient
Natural Land / Open Water	0.0	1.00
Pine Plantation	5.1	1.58
Low Intensity Open Space / Recreational	6.7	1.85
Unimproved Pastureland (with livestock)	8.3	2.06
Improved pasture (no livestock)	19.5	2.89
Low Intensity Pasture (with livestock)	36.9	3.51
High Intensity Pasture (with livestock)	51.5	3.83
Citrus	65.4	4.06
Medium Intensity Open Space / Recreational	67.3	4.09
Row crops	117.1	4.63
High Intensity Agriculture (dairy farm)	201.0	5.15
Single Family Residential (Low-density)	1077.0	6.79
Recreational / Open Space (High-intensity)	1230.0	6.92
Single Family Residential (Med-density)	2461.5	7.59
Low Intensity Transportation	3080.0	7.81
Single Family Residential (High-density)	3729.5	7.99
Low Intensity commercial (Comm Strip)	3758.0	8.00
Institutional	4042.2	8.07
Highway (4 lane)	5020.0	8.28
Industrial	5210.6	8.32
Multi-family residential (Low rise)	7391.5	8.66
High intensity commercial (Mall)	12661.0	9.18
Multi-family residential (High rise)	1285.0	9.19
Central Business District (Avg 2 stories)	16150.3	9.42
Electric Power Facility	29401.3	10.00

controlled by human modification, with a flashy hydrograph and extreme alteration of the natural ecosystem (Fore 2004).

Habitat assessment (or habitat condition index as termed in Fore 2004) is the third measure included in the HDG. The Florida Department of Environmental Protection (FDEP) has developed Standard Operating Procedures (SOPs) for river and stream habitat assessments (FDEP-SOP-001/01: Form FD 9000-5 June 1, 2001 available from the Bureau of Laboratories at http://www.dep.state.fl.us/labs/library/forms.htm). The habitat assessment is separated into four primary habitat components: substrate diversity (number of diverse, productive habitats), substrate availability (percent of stream reach composed of productive habitat), water velocity (based on the maximum observed

velocity, where higher velocities receive higher scores), and habitat smothering (as the percent of the stream reach covered by sand or silt accumulation). The four secondary habitat parameters include artificial channelization (as an assessment of anthropogenic modification of the stream reach), bank stability (evidence of bank stability/instability from erosion or bank failure), riparian buffer zone width (estimate of the width of the vegetation on the least buffered side), and riparian zone vegetation quality (estimate of community composition and structure). These eight habitat parameters are scored by the field biologist on a scale of 1-20, with 20 representing the highest habitat quality. Three of the secondary habitat components (bank stability, riparian buffer zone width, and riparian zone vegetation quality) are scored separately on a scale of 1-10 (with 10 representing the highest habitat quality) for both the right and left banks (when added together these three categories are still given equal weighting, with half of the score reflecting the condition of each bank). Scores for each of the eight parameters are summed, and a stream is then assigned a categorical score of optimal (134-160), suboptimal (91-123), marginal (54-80), or poor (11-43) based on the total score (Fore 2004).

The fourth measure of the HDG is an LDI score calculated within a 100 m buffer on each side of the stream for 10 km upstream from the sampling point (LDI_BF) (Fore 2004). Each measure included in the HDG is categorically assigned a score based on the score for each measure (Table 1-2). Ammonia concentration in the water (as mg N/L), habitat assessment, and LDI for the buffer are assigned scores of 0, 1, or 2; whereas the hydrologic index is assigned scores of 0, 1, 2, or 3. Overall HDG values potentially range from 0 (no detectable human induced disturbance) to 9 (extreme anthropogenic disturbance) (Fore 2004). Original categorical scoring criteria for the HDG were established from correlations of the HDG categories (ammonia, hydrologic condition, habitat index, and LDI_BF) with the macroinvertebrate metric EPT (Ephemeroptera, Plecoptera, and Trichoptera) taxa richness. All HDG scores used in this analysis were taken directly from HDG scores provided by FDEP.

Project Overview

The community composition of the macrophyte assemblage was sampled in flowing wetland systems including forested strand (n = 10) and forested floodplain (n = 14) wetlands throughout Florida. Our primary goal was to develop a preliminary Florida Wetland Condition Index (FWCI) for freshwater flowing water wetlands, building upon the FWCI for isolated depressional herbaceous (Lane et al. 2003) and forested (Reiss and Brown 2005) freshwater wetlands. Our secondary goal was to determine the appropriate scale for LDI buffers for flowing water wetlands that would best capture the current wetland condition, based on correlations with the FWCI. Our third goal was to correlate measures of the HDG (Fore 2002) and macroinvertebrate community data from the Florida SCI with our macrophyte FWCI and landscape LDI measures of ecosystem condition.

Wetland study sites were sought in various landscape settings that included natural, agricultural, and urban land uses. The elongated forested wetlands associated with flowing waters were in many areas continuous for extended geographic distances, and the forested edges bordered a great variety of land use activities. Therefore, it was

Table 1-2. Categorical scoring criteria used to calculate the Human Disturbance Gradient (HDG). The HDG is the sum of the categorical scores for the four individual measures ammonia concentration in the water (as mg N/L), hydrologic index, habitat assessment, and LDI for the buffer. The HDG range is from 0 (no detectable human induced disturbance) to 9 (extreme anthropogenic disturbance).

	HDG Categorical Score			
HDG Measure	0	1	2	3
Ammonia concentration (mg N/L)	< 0.1	0.1 – 2.0	> 2.0	
Hydrologic Condition	< 6	6 - 7	8 - 9	10
Habitat Assessment	> 65	50 - 65	< 50	
LDI for the buffer	< 2.0	2.0 – 3.5	> 3.5	

difficult to ascertain the specific land use activity contributing most heavily or frequently to perturbations in community composition. While *a priori* land use categories were assigned (reference, agricultural, urban), wetlands were also divided into two groups representing low and high landscape development intensities.

CHAPTER 2
METHODS

Twenty-four freshwater forested wetlands, including 10 strand and 14 floodplain wetlands, were sampled during 2003. This chapter describes site selection, Landscape Development Intensity (LDI) index calculations, field-data collection, and methods of statistical analyses.

Study Area

Florida has been dissected into smaller geographic zones using a number of different methods, including bioregions (Griffith et al. 1994) with 13 subecoregions (grouped into four ecoregions for small wadeable streams of panhandle, northeast, peninsula, and Everglades); lake ecoregions of Florida (Griffith et al. 1997) with 47 subecoregions; and Florida wetland regions determined with a hydrologic model by Lane (2000), which included physical (surficial geology, soils, digital elevation model, slope) and climatic (precipitation, potential evapotranspiration, runoff, annual days of freezing) variables with four ecoregions (panhandle, north, central, and south). When boundaries for the bioregions of Griffith et al. (1994) and wetland regions of Lane (2000) were compared, there were similarities among the panhandle and south/Everglades boundaries; however, the north/northeast and central/peninsula ecoregions differed in extent (Figure 2-1). We adopted the four wetland regions developed by Lane (2000) to partition the state during site selection. However, Human Disturbance Gradient (HDG) and Stream Condition Index (SCI) scores available for the floodplain forests were designated according to the Florida bioregions (Griffith et al. 1994); as such we use both wetland region and bioregion categories in our analyses.

The four wetland regions defined by Lane (2000) are categorized as panhandle, north, central, and south. The panhandle wetland region is characterized by less human development than the other ecoregions. Streams in the panhandle wetland region typically run from north to south, discharging into the Gulf of Mexico. The primary ecosystems are scrub and high pine, temperate hardwood forests, pine flatwoods and dry prairies, and swamps (Fernald and Purdum 1992). The major population centers of Tallahassee, Panama City, and Pensacola are included within the panhandle wetland region (Davis 1967). The north wetland region has similar ecosystem types, primarily with pine flatwoods and dry prairies and less scrub and high pine, temperate hardwood forests, and swamps (Fernald and Purdum 1992). Two major drainage features in the north wetland region include the Santa Fe and Suwannee River system and the lower St. John's River, with Jacksonville being the major city situated along the northeastern Atlantic coast. The central wetland region has a distinguishing central ridge feature with a higher maximum elevation than the north and south wetland regions (Fernald and Purdum 1992). The central wetland region is characterized by pine flatwoods and dry prairies with scrub and high pine along the ridge. There is less area in temperate hardwood forests, less area in swamps, and an increase in the area of inland lakes and freshwater marshes (Fernald and Purdum 1992). The largest population centers include

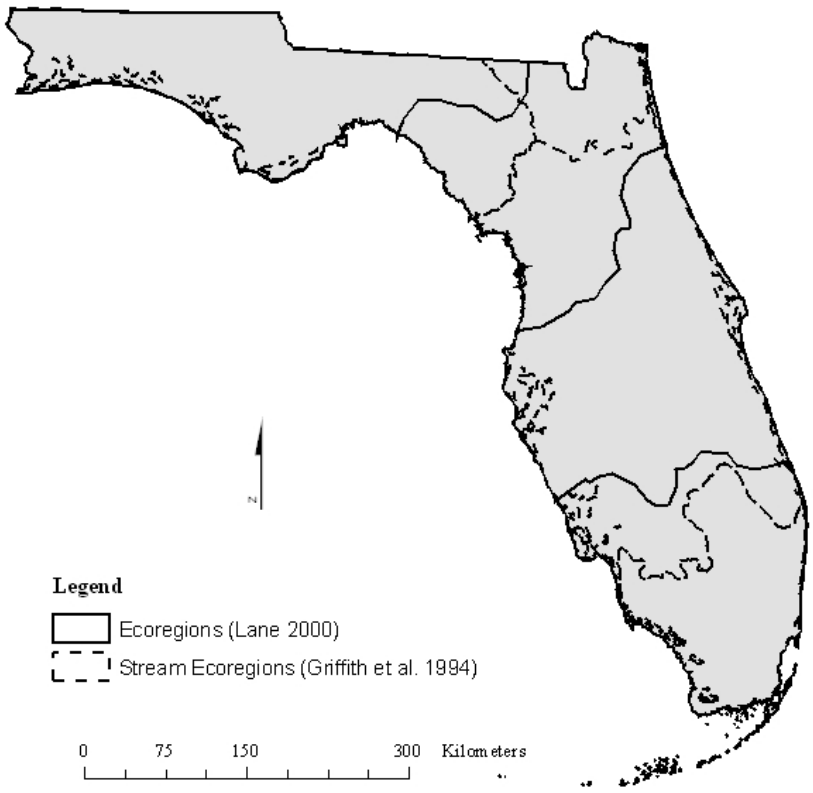


Figure 2-1. Florida wetland regions defined by climatic and physical variables (solid line; Lane 2000) and Florida bioregions (dashed line; Griffith et al. 1994).

Orlando and Tampa. The south wetland region is unique with the nearly flat terrain of the Everglades (less than 2 m relief; Fernald and Purdum 1992). Much of the south is swamps and freshwater marsh, and few streams remain unaltered. Lake Okeechobee is in this wetland region as are the urban centers of Miami and Ft. Myers located on the densely populated east and west coasts, respectively.

Site Selection

During the 2003 growing season (May-July), 24 forested wetlands were sampled throughout the state of Florida (Figure 2-2). Random site selection was not feasible given the necessity of obtaining permission to access private lands and the non-random pattern of land development in Florida. The forested strand wetlands (n=10) were arranged spatially throughout Florida, so that the distribution was spread among three of the four Florida wetland regions (Lane 2000), including north (n=2), central (n=5), and south (n=3). No strands were sampled in the panhandle wetland region. The forested floodplain wetlands (n=14) were selected as a subset of systems that have been sampled using the SCI (Fore 2004). Many of the streams that had received a score of poor according to the SCI no longer had floodplain forest vegetation along the banks, but rather had mowed grass or paved banks. As such, these streams did not fit our criteria for

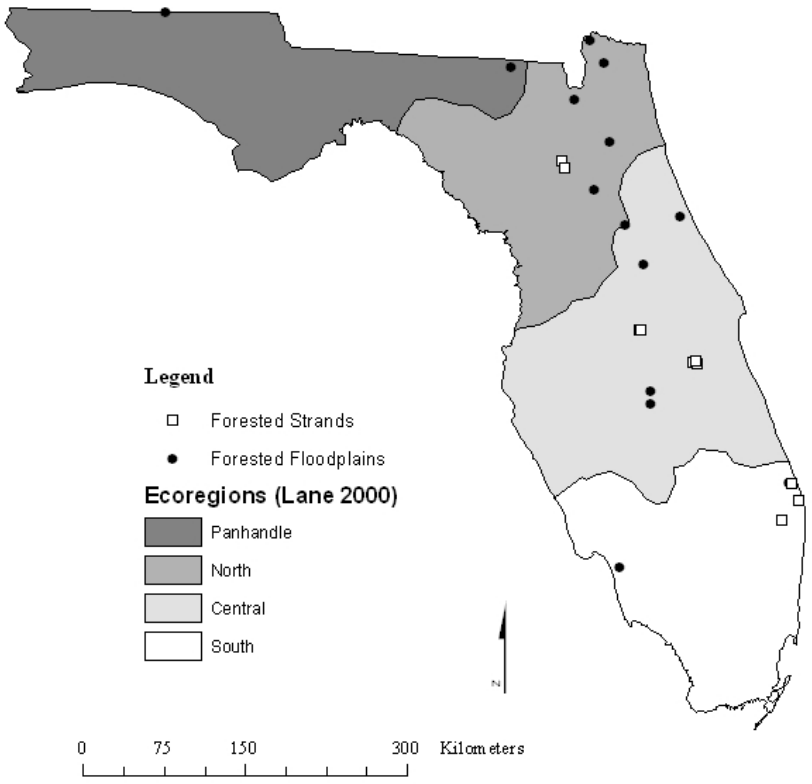


Figure 2-2. Study site location of 24 forested wetlands in Florida (◆strands n=10; ■ floodplains n=14). Wetland region boundaries follow Lane (2000).

site selection, and where therefore excluded from sampling. Forested floodplain wetlands sampled were located in the panhandle (n=2), north (n=8), central (n=3), and south (n=1) wetland regions.

A priori categories were used to define sample wetlands as reference, agricultural, and urban. The suite of reference wetlands included wetlands surrounded by intact native ecosystems. These wetlands were thought to represent the best possible wetland condition currently in Florida. These wetlands could be categorized as reference standard wetlands by the USEPA (1998a), and have been described as wetlands considered to be the least altered that are reflective of characteristic levels of wetland function. Agricultural wetlands were defined as those currently surrounded by cattle pasture, rangeland, row crops, citrus, and silvicultural land uses. Urban wetlands included those embedded within commercial, industrial, and residential land uses. Many of the urban wetlands sampled were suspected to previously have been embedded in agricultural land uses due to the development patterns throughout Florida. Hereafter wetlands embedded in primarily undeveloped landscapes are called reference; wetlands embedded in primarily agricultural land uses are called agricultural; and wetlands embedded in primarily urban land uses are called urban.

Table 2-1 provides general information about each sample wetland, including site code, sample date, wetland type, wetland region (Lane 2000), county, bioregion (Griffith

Table 2-1. Site characterization for 24 freshwater forested wetlands.

Site Code*	Sample Date	Wetland Type	Wetland Region^	County	Bioregion^	*A Priori* Category	Associated Stream✧
FF1	May 19 2003	Floodplain	North	Putnam	Peninsula	Urban	Orange Creek
FF2	May 20 2003	Floodplain	North	Marion	Peninsula	Reference	Juniper Creek
FF3	May 22 2003	Floodplain	North	Volusia	Peninsula	Urban	Groover Branch
FF4	May 23 2003	Floodplain	Central	Lake	Peninsula	Agricultural	Blackwater Creek
FF5	May 25 2003	Floodplain	North	Nassau	Northeast	Urban	Pigeon Creek
FF6	May 25 2003	Floodplain	Central	Polk	Peninsula	Reference	Livingston Creek
FF7	May 25 2003	Floodplain	Central	Polk	Peninsula	Reference	Tiger Creek
FF8	May 26 2003	Floodplain	North	Nassau	Northeast	Urban	Alligator Creek
FF9	May 29 2003	Floodplain	North	Clay	Northeast	Urban	Green Creek
FF10	June 8 2003	Floodplain	South	Lee	Everglades	Urban	Leitner Creek
FF11	June 13 2003	Floodplain	Panhandle	Hamilton	Northeast	Agricultural	Rock Creek
FF12	June 14 2003	Floodplain	North	Baker	Northeast	Agricultural/Urban	Turkey Creek
FF13	June 24 2003	Floodplain	North	Martin	Peninsula	Reference	Kitchen Creek
FF14	July 1 2003	Floodplain	Panhandle	Walton	Panhandle	Agricultural	Limestone Creek
FS1	May 24 2003	Strand	Central	Osceola	Peninsula	Reference	NA
FS2	May 26 2003	Strand	Central	Osceola	Peninsula	Agricultural	NA
FS3	May 27 2003	Strand	Central	Osceola	Peninsula	Agricultural	NA
FS4	May 31 2003	Strand	Central	Osceola	Peninsula	Urban	NA
FS5	June 1 2003	Strand	Central	Osceola	Peninsula	Urban	NA
FS6	June 16 2003	Strand	South	Palm Beach	Peninsula	Urban	NA
FS7	June 24 2003	Strand	South	Palm Beach	Peninsula	Reference	NA
FS8	July 7 2003	Strand	North	Alachua	Peninsula	Agricultural	NA
FS9	July 8 2003	Strand	North	Alachua	Peninsula	Reference	NA
FS10	July 14 2003	Strand	South	Martin	Peninsula	Reference	NA

*Site Codes correspond to the wetland type (FF = floodplain forest; FS = forested strand) and were numbered in the order sampling occurred.

^ Wetland region from Lane (2000); Bioregion from Griffith et al. (1994).

✧ Associated Stream = Stream Condition Index (SCI) stream data; NA = Not Applicable, forested strands were not associated with SCI streams.

et al. 1994), *a priori* category of surrounding land use, and associated stream. Site codes were assigned to preserve the anonymity of individual land owners.

Gradients of Landscape Development Intensity

Digital orthophoto imagery, available from Labins, The Land Boundary Information System from the Florida Department of Environmental Protection (FDEP) (available at http://www.labins.org/2003/index.cfm), were used to georectify transect locations from GPS coordinates using ArcGIS 8.3 from Environmental Systems Research Institute, Inc. Polygons were established for wetlands delineated from digital orthophoto imagery. LDI scores were calculated using 1995 and 2000 land use coverages available for download through the Florida Geographic Data Library (available at http://www.fgdl.org/). Coverages for 1995 land use were available separately for each Florida Water Management District (Northwest Florida, NWFWMD; Suwannee River, SRWMD; St. Johns River, SJRWMD; Southwest Florida, SWFWMD; and South Florida, SFWMD). More recent coverages (2000 land use) were available for a limited number of sample wetlands including only those within the boundaries of the SJRWMD.

LDI index scores were calculated at the transect (LDI_T), wetland feature (LDI_F), and watershed (LDI_WS) scale. At the transect scale, a 100 m buffer was created around the downstream sample transect. Two LDI_T scores were then calculated based on the area of the land uses within the 100 m buffer (Figure 2-3), one including the wetland area and the other excluding the wetland area. The LDI_T score including the wetland area will generally be lower than the LDI_T score excluding the wetland area, based on the increase in the percent of land assigned a 1.0 Natural Land/Open Space LDI coefficient (because of the inclusion of the wetland area, which is assigned an LDI coefficient of 1.0). Including the wetland area in the calculation was thought to weight the LDI_T based on the size of the wetland sampled, which could be an important factor influencing wetland condition. LDI_T calculations were repeated using a 200 m buffer (Figure 2-3). LDI_T 1995 identification of land uses within the 100 m buffer were taken from land use/land cover coverages and checked with 1995 digital orthophoto imagery. LDI_T 2000 identification of land uses were taken from 2000 land use/land cover coverages and checked with 1999 digital orthophoto imagery. Land uses that had changed since the photos were taken were updated based on land use maps drawn during field site visits. This step was not possible for larger buffer areas where visibility of surrounding land uses on site was hindered due to distance and barriers, and so discrepancies may be apparent in 100 m buffer calculations (hand corrected for each buffer) and larger 200 m buffers (not corrected, land use/land cover straight from available GIS coverages).

LDI_F and LDI_WS were also calculated once including and once excluding the wetland area. The boundary for the LDI_F was established as the area making up the 200 m buffer upstream of the downstream transect (Figure 2-4). The boundary of the feature was established when a distance of at least 30 m showed a break in wetland vegetation established from photo-interpretation of the digital orthophoto imagery. The upstream boundary for the LDI_WS was established using the Better Assessment Science Integrating Point and Nonpoint Sources 3.0 (BASINS 3.0) environmental analysis system (USEPA 2001) (Figure 2-5). BASINS is a multi-purpose environmental assessment tool

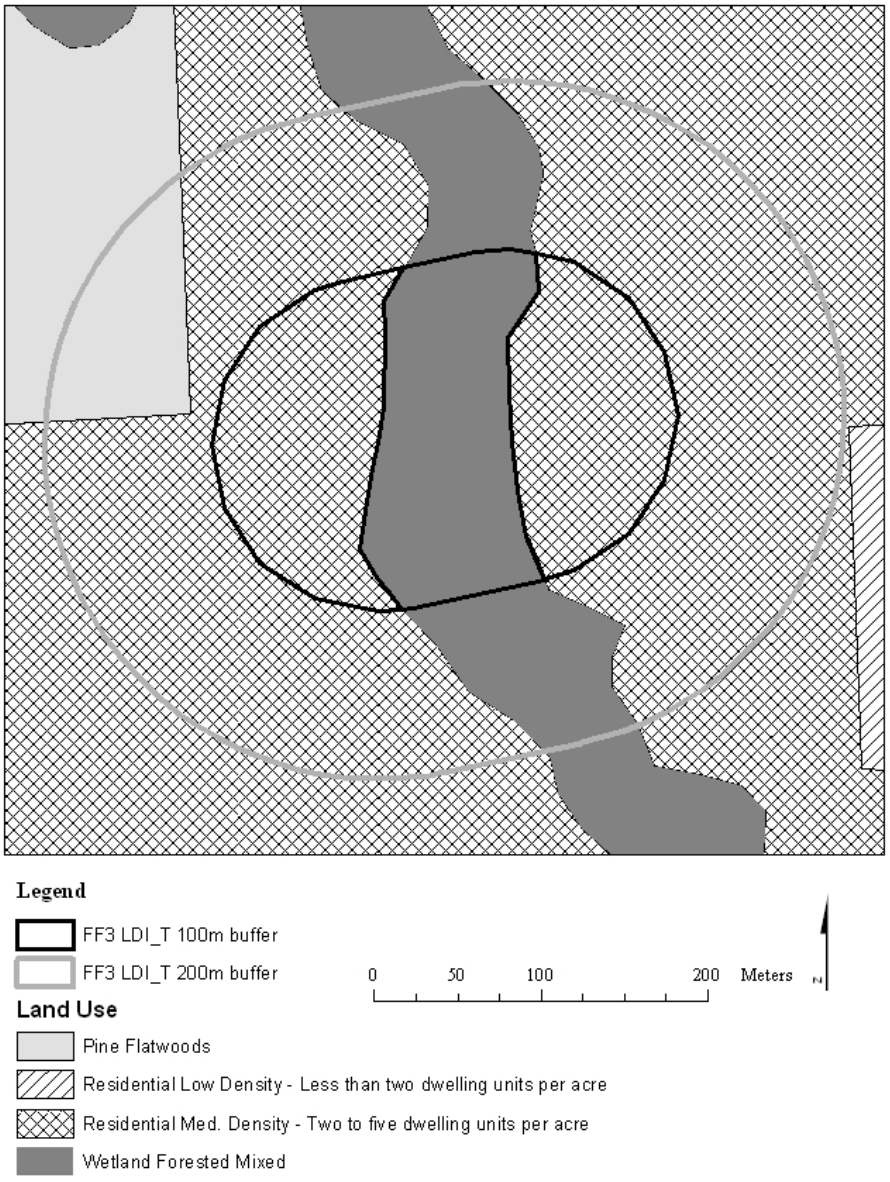


Figure 2-3. Boundaries of the 100 and 200 m buffers drawn around the downstream transect for the LDI_T calculation at Forested Floodplain 3 (FF3). Two LDI_T values were calculated for each sample wetland, one including the wetland area and the other excluding the wetland area.

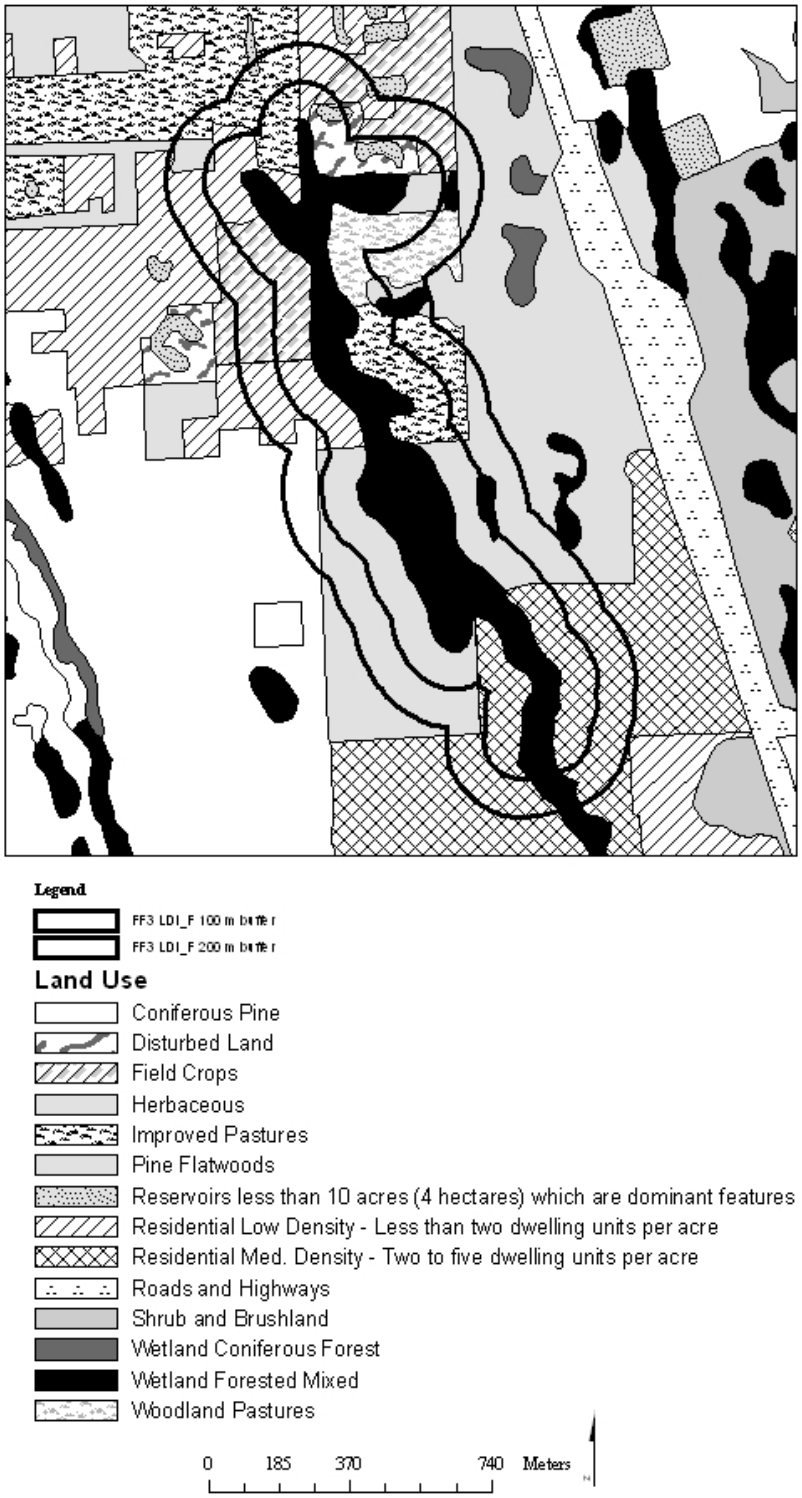


Figure 2-4. Boundaries of 100 and 200 m buffers around a wetland feature and designated land use. The 200 m buffer was used for the LDI_F calculation at Forested Floodplain 3 (FF3).

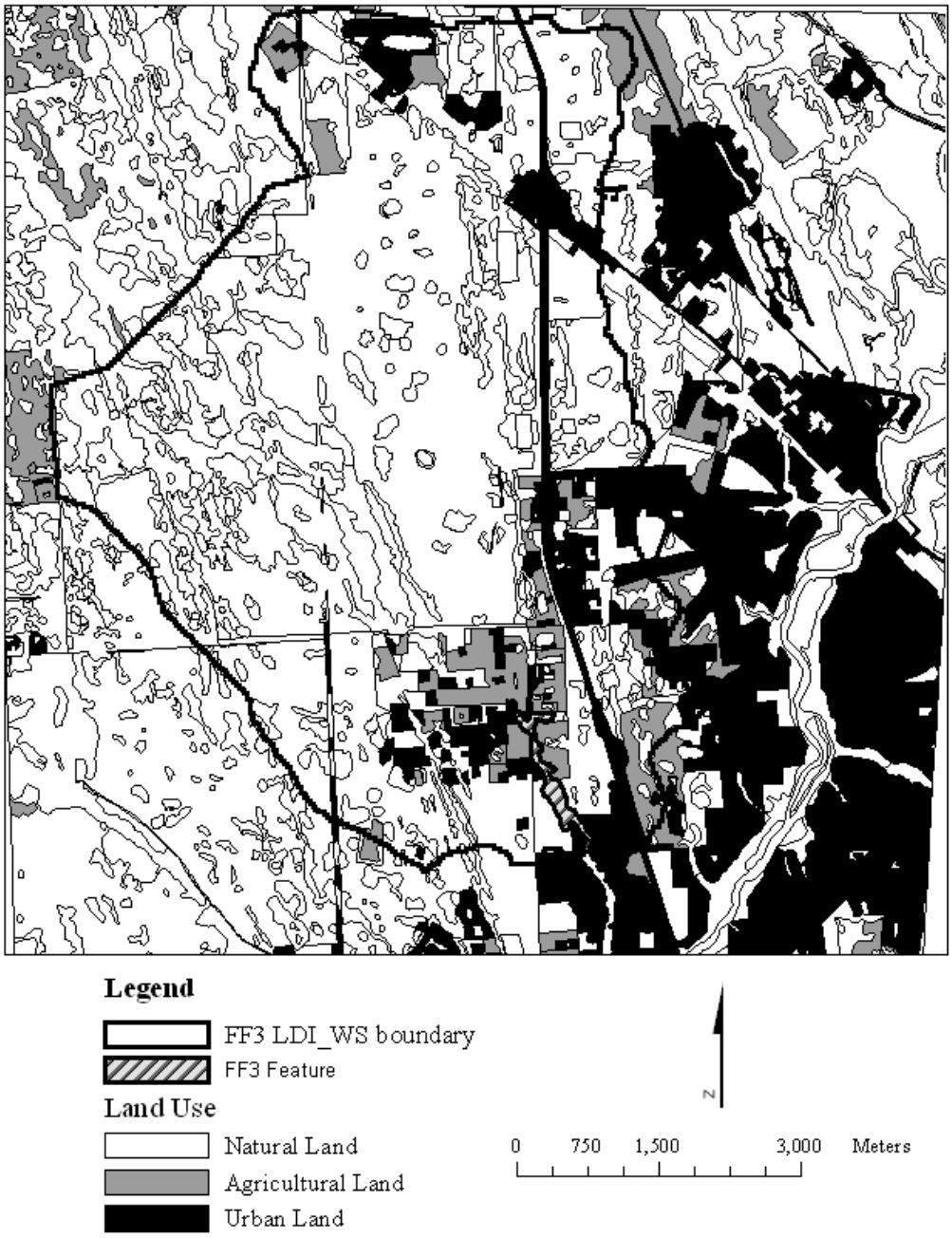


Figure 2-5. Boundary of the watershed buffer around Forested Floodplain 3 (FF3) for the LDI_WS calculation. Four LDI_WS values were calculated for each sample wetland: two LDI_WSs were calculated based on an equal weighting by area of all land uses within the upstream watershed including a calculation inclusive of the wetland area (LDI_WS_ED/w) and exclusive of the wetland area (LDI_WS_ED/wo); and two distance weighted LDIs were calculated in which land uses nearest to the sample wetland were more highly weighted, including linear weighting (LDI_WS_DW_lin) and weighting based on an exponential decay function (LDI_WS_DW_exp).

designed for watershed and water quality studies that integrates a GIS (ArcView 3.2), national watershed database, and environmental assessment and modeling tools (USEPA 2001). The digital terrain model from the National Elevation Dataset, which is a 30 m raster-based dataset produced by the United States Geological Survey (USGS), was used for watershed delineation.

Two separate types of LDI_WSs were calculated based on an equal weighting by area of all land uses within the upstream watershed (LDI_WS_ED) and a distance weighted calculation, where land uses occurring closer to the sampling point were weighted more than those occurring farther away (WS_LDI_DW). LDI_WS_ED scores were calculated both including (LDI_WS_ED/w) and excluding (LDI_WS_ED/wo) the wetland area. The distance weighted calculations were done in two ways. First, a calculation was made using a linear weighting of land uses (LDI_WS_DW_lin); second, a calculation was made using an exponential decay function, where land uses nearest the sampling point were weighted most significantly (LDI_WS_DW_exp).

GIS analyses were performed in ArcGIS 8.3 (Environmental Systems Research Institute, Inc. 2002). The following equation was used to calculate LDI:

$$LDI_{Total} = \sum \% LU_i * LDI_i \quad (2\text{-}1)$$

where $\%LU_i$ is the percent of a land use within the buffer of interest and LDI_i is the LDI coefficient for a particular land use based on the amount of nonrenewable energy use per unit area in the surrounding landscape (Table 1-1). The LDI coefficient values and LDI equation were based on work by Brown and Vivas (2005). Potential LDI coefficients ranged from a minimum of 1.0 (Natural Land/Open Space) to a maximum of 10.0 (Central Business District).

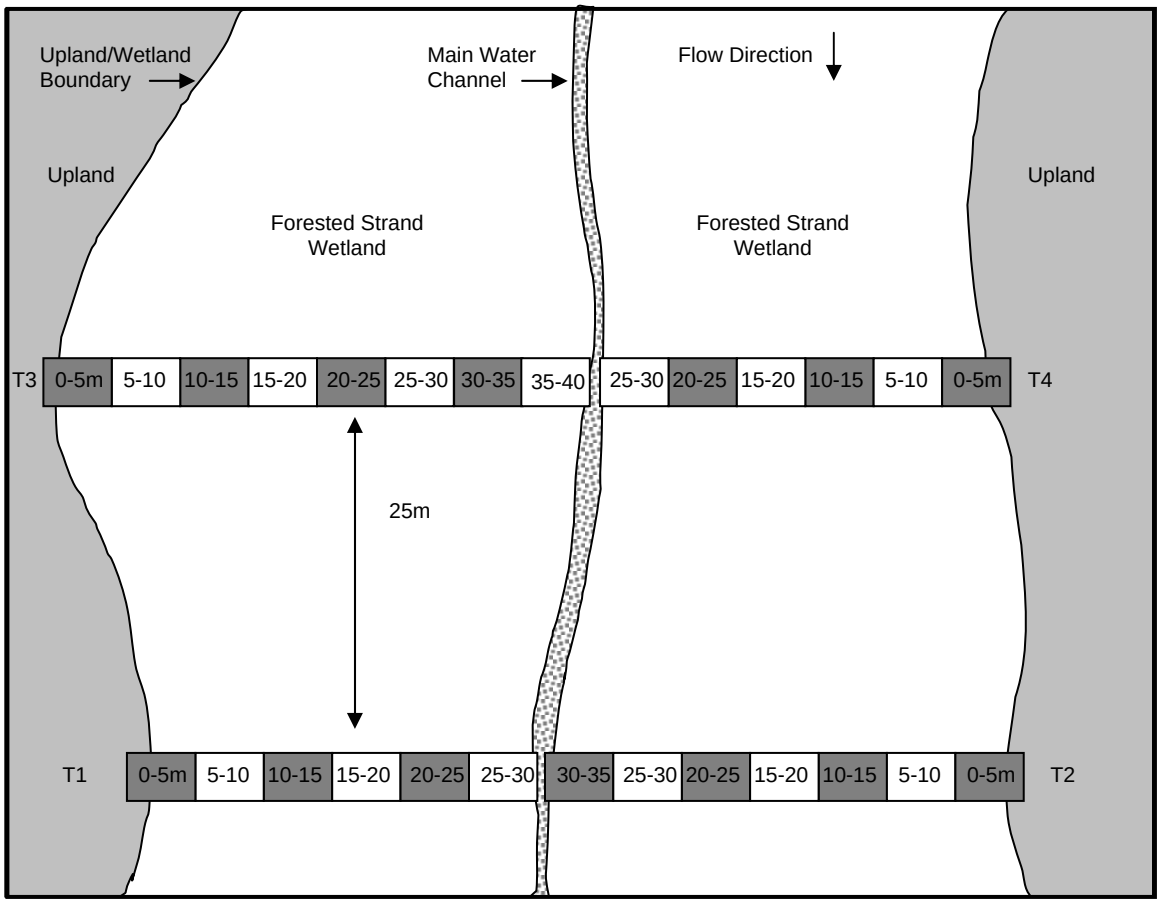
Field-data Collection

A concise summary of field-data collection procedures follows. Appendix A provides more detailed descriptions of field-data collection techniques in the format of Standard Operating Procedures (SOPs) for field use.

Transect Sampling Design

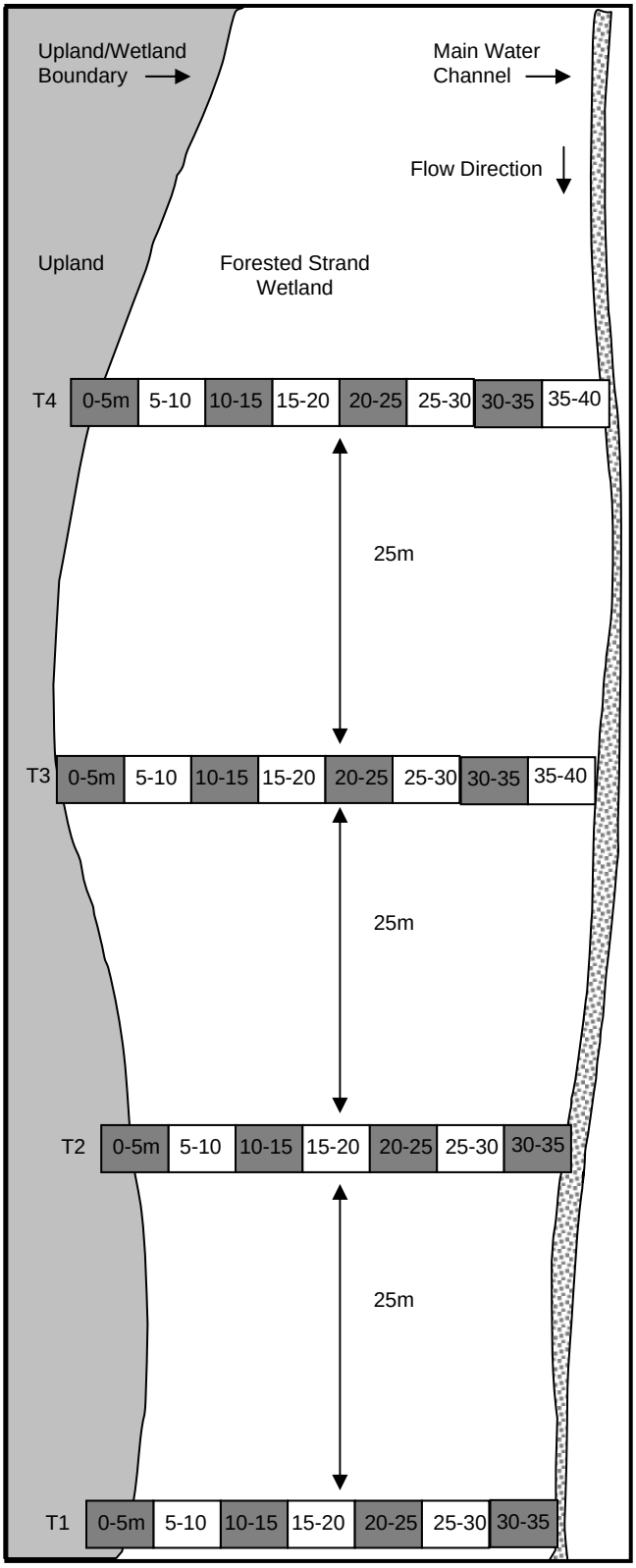
Transects were established perpendicular to the hydrologic flow of the system. The wetland/upland boundary was determined based on the Florida Unified Wetland Delineation Methodology (Chapter 62-340, F.A.C.), using a combination of wetland plant presence according to wetland plant status (e.g. obligate, facultative wetland, facultative, or upland) and wetland hydrologic indicators (e.g. lichen lines, moss collars). Modifications of transect establishment were implemented for strand and floodplain wetlands types, described below.

Sampling in forested strands consisted of establishing four transects that extended from the wetland/upland boundary to the middle of the channelized flow in the center of the strand (Figure 2-6a). Transects were established at 25 m intervals. While it is recognized that strands were not perfectly symmetrical in nature, an effort was made to establish transects that represented a cross-section of the strand along the gradient of



(A)

Figure 2-6. Idealized transect layout for forested strand wetlands. (A) Standard transect layout included four transects that represented a cross-section of the strand along the gradient of wetland/upland boundary to the water channel, with transect establishment along both sides of the water course. (B) Alternative transect placement included four transects established on the same side of the water channel due to limited access or permission restrictions.


Upland/Wetland Boundary
Main Water Channel
Flow Direction
Upland
Forested Strand Wetland
T4
0-5m
5-10
10-15
15-20
20-25
25-30
30-35
35-40
25m
T3
0-5m
5-10
10-15
15-20
20-25
25-30
30-35
35-40
25m
T2
0-5m
5-10
10-15
15-20
20-25
25-30
30-35
25m
T1
0-5m
5-10
10-15
15-20
20-25
25-30
30-35

(B)

upland/wetland boundary to the water channel. In some sampling efforts wetland forests were sampled along both sides of the water course. However, on some occasions, limited access or lack of permission restricted sampling to the same side of the water course. In these instances, four transects were established on the same side of the water channel (Figure 2-6b).

Forested floodplain wetlands varied in size, with many wetlands spanning an area broader than 100 m across. To encourage comparable sampling efforts among the different width strand and floodplain systems, a maximum 50 m transect length was established for floodplain systems. For forested floodplains narrower than 50 m wide, four transects were established that spanned the area from the upland/wetland boundary, as determined by wetland plant presence and hydrologic indicators, to the edge of the stream channel. The initial transect was established downstream at a point closest to the SCI sampling location. Consecutive transects were established upstream at 25 m intervals. In forested floodplain wetlands wider than 50 m, the initial transect was established at the upland/wetland boundary and extending 50 m inward perpendicular to the stream. The second transect was established 25 m upstream, starting at the waters edge and extending a full 50 m towards the upland/wetland boundary, perpendicular to the water course. The pattern was repeated for the third and fourth transects (Figure 2-7). While four transects of 50 m length were considered the optimal sampling effort, this was not always achieved due to limited access and development pressures resulting in limited remaining areas of forested floodplain.

Along each transect, a series of 1 m wide by 5 m long quadrats was established back to back. Living macrophytes rooted within each quadrat were identified to the lowest taxonomic level possible. When field identification was limited, a sample specimen was collected, pressed, and identified in the laboratory. An expert Florida botanist (Dr. David Hall) was consulted for identification of unknown specimens. Taxonomic information including species, genus, and family were compiled for all of the macrophytes identified. Additional characteristics were collected for use in metric development, including category (annual or perennial, evergreen or deciduous, indigenous or exotic) and growth form (aquatic, fern, grass, herb, sedge, shrub, tree, or vine). References specific to Florida were consulted first (Tobe et al. 1998; Wunderlin and Hansen 2003), and additional information was supplemented from other sources (in the following order: Godfrey and Wooten 1981, Wunderlin 1998, and USDA NRCS 2002). Wetland indicator status (i.e. obligate, facultative wetland, etc.) for each species was obtained from FAC Ch. 62-340. Wetland indicator statuses for species not listed in FAC Ch. 62-340 were obtained from Tobe et al. (1998), Wunderlin and Hansen (2003), and USDA NRCS (2002), in that order. When species were not assigned a Florida specific indicator status, the National Wetlands Inventory indicator status was used (Wunderlin and Hansen 2003; USDA NRCS 2002, in that order). When information was still unavailable for plant characteristics in published literature, Florida botanists (who also participated in the Floristic Quality Assessment Index) were consulted.

Floristic Quality Assessment Index

A Floristic Quality Assessment Index (FQAI) has been included in many of the multimetric biotic indices created for the macrophyte assemblage. The concept of FQAI was developed by Wilhelm and Ladd (1988) for vegetation around Chicago, Illinois.

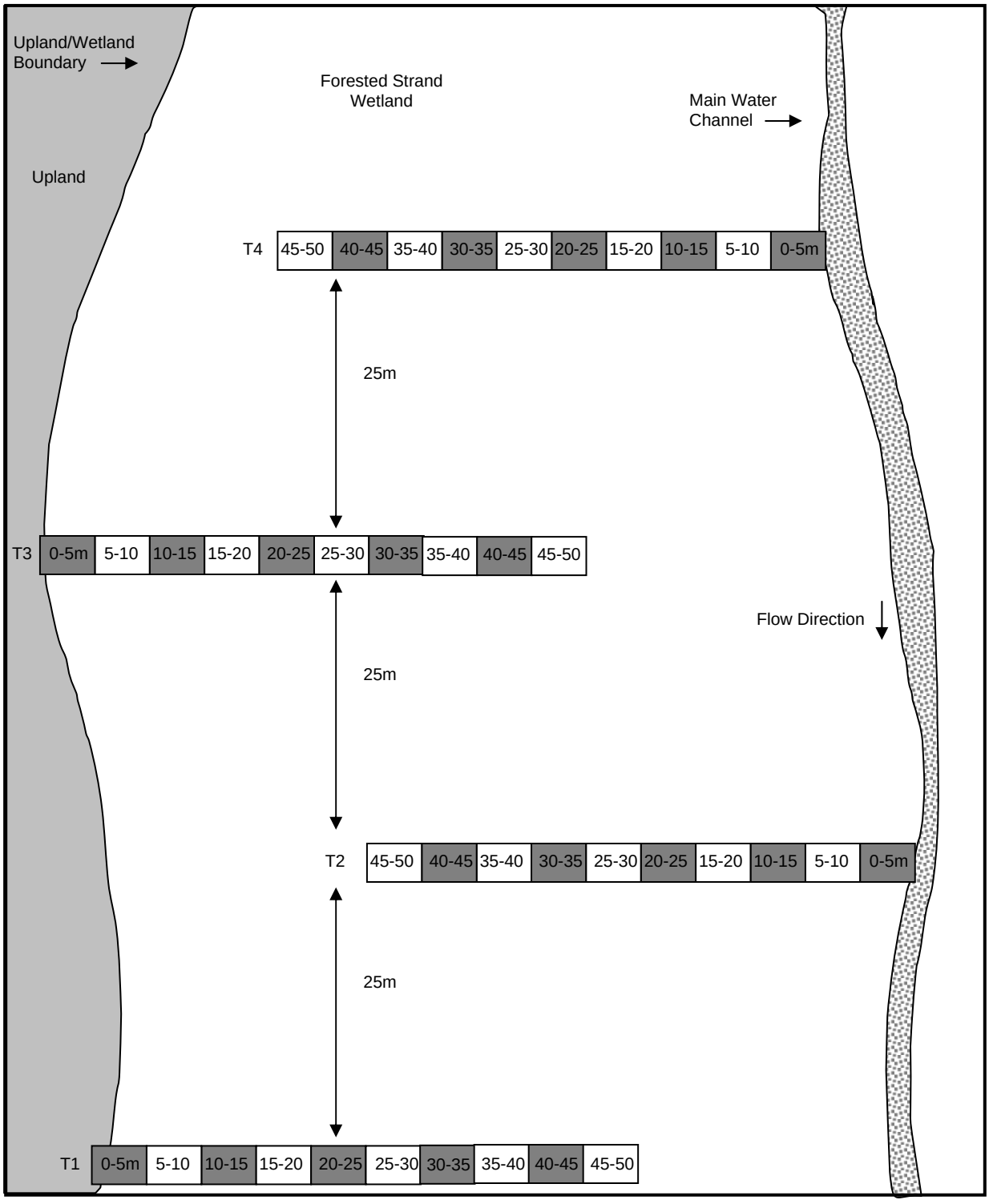


Figure 2-7. Idealized transect layout for forested floodplain wetlands. Transect 1 began at the upland/wetland boundary and extended 50 m inward, perpendicular to the channelized flow. Transect 2 began at the edge of the channel and extended 50 m towards the wetland/upland boundary. This pattern repeated for transects 3 and 4.

This method of scoring plant species based on expert botanist opinion has been used in Michigan (Herman et al. 1997), Ohio (Andreas and Lichvar 1995; Fennessy et al. 1998; Mack 2001), Ontario (Francis et al. 2000), North Dakota (Mushet et al. 2002), and Florida (Lane et al. 2003; Cohen et al. 2004; Reiss and Brown 2005). The FQAI provides a quantitative means of assessing the fidelity of a plant to a particular environment through the Delphi technique (Kent 2000), where individual botanists assigned coefficients (termed coefficients of conservatism) to individual species. This technique assumes that the collective decision by a group of expert botanists is more accurate than the professional judgment of one individual (Kent 2000).

The Florida FQAI enlisted regional expert botanists to provide quantitative scores for vegetation in the form of a coefficient of conservatism assigned individually to each species. The FQAI score for an individual wetland was calculated as:

$$FQAI = [\sum (C_1 + C_2 + \ldots C_n)] / N \qquad (2\text{-}2)$$

where C = species specific coefficient of conservatism score (CC), and N = species richness. This equation was considered a modified FQAI because previous studies used the square root of native species richness as the denominator. Theories on the importance of species richness suggest that higher species richness signifies a more valuable ecosystem, which can be quantified by using the square root function (Fennessy et al 1998). However, a recent study by Cohen et al. (2004) found a stronger relationship along a disturbance gradient for the mean CC (Eq. 2-2) than with the traditional FQAI equation (using the square root of native species richness in the denominator). They also reported that using total species richness (i.e. including exotic species in the calculation of species richness) improved the relationship of mean CC with the human disturbance gradient (measured with LDI). The sum of the species CC scores was divided by total species richness (Eq. 2-2) in this study to account for potential differences in species richness due to differences in wetland regions (Lane 2000), bioregions (Griffith et al. 1994), *a priori* land use categories, or other unspecified differences.

Coefficient of conservatism scores were obtained from Florida botanist surveys to compliment previous FQAI efforts for Florida. FQAI scores were adopted first from depressional forested wetlands (Reiss and Brown 2005) and second from depressional herbaceous wetlands (Lane et al. 2003; Cohen et al. 2004). Correlations between CC scores for species from the depressional forested and herbaceous wetlands were strong (unpublished data). In an effort to streamline the acquisition of species specific CC scores for the FQAI for strand and floodplain wetlands, CC scores were only obtained for those species not included in previous Florida FQAI surveys. As such, each botanist was sent a list of species identified in the forested strand and floodplain wetlands that did not already have a CC score for the previous Florida FQAI efforts (n = 86). Botanists participating in the 2003 FQAI effort for forested strand and floodplain wetlands included Tony Arcuri, Dan Austin, David Hall, Nina Raymond, and Bruce Tatje. Botanists

scored each species based on its faithfulness to Florida forested strand and floodplain wetlands. Potential CC scores ranged from 0 to 10:

- 0 - Exotic and native species that act as opportunistic invaders, included species that commonly occur in disturbed ecosystems
- 1-3 - Species that were widely distributed and occurred in disturbed ecosystems
- 4-6 - Species with a faithfulness to a particular ecosystem, but also tolerant of moderate levels of disturbance
- 7-8 - Species typical of well-established ecosystems that sustain only minor disturbances
- 9-10 - Species occurring within a narrow set of ecological conditions

Species with low CC scores were considered tolerant of many disturbances, whereas species with high CC scores were considered to occur within a narrow set of stable ecological condition. Appendix B lists the CC scores for the macrophyte species identified in this study.

Data Analysis

Summary Statistics

Summary statistics for the macrophyte assemblage were calculated at the species level, including richness (R), evenness (E), Shannon diversity (H'), Simpson diversity (D), and Whittaker's beta diversity (β_W). Richness (R) was defined as the total number of distinct taxa identified within the sample wetland. Evenness (E), described as the fraction of maximum possible diversity in a wetland (McCune and Grace 2002), was calculated as the Shannon diversity (H') value divided by the natural log of richness:

$$E = H' / \ln (R) \quad (2\text{-}3)$$

The Shannon diversity index has been described as measuring the "information content" of a sample unit where maximum diversity yields maximum uncertainty (McCune and Grace 2002). For Shannon diversity calculations (H'), the sample unit was an individual forested wetland:

$$H' = -\Sigma\, p_i * \log(p_i) \quad (2\text{-}4)$$

$$p_i = n_i / N \quad (2\text{-}5)$$

where n_i was the number of occurrences of taxon i, and N was the total number of occurrences of all taxa at a wetland. The number of occurrences represented the number of quadrats a species occurred in, and the total number of occurrences of all taxa at a wetland represented the sum of the total number of quadrats of all of the species identified.

Simpson diversity (the compliment of Simpson's original index, which was a measure of dominance) was a measure of the likelihood species that are randomly chosen in sampling will be different. The equation used to calculate Simpson diversity was:

$$D = 1 - \sum p_i^2 \quad (2\text{-}6)$$

Whittaker's beta diversity (β_W) was computed as a calculation of overall beta diversity, or the compositional change represented in a sample. Whittaker's beta diversity was calculated as the number of species at a particular forested wetland (S_c) divided by the average species richness per quadrat (S), minus one:

$$\beta w = [S_c / S] - 1 \quad (2\text{-}7)$$

The resulting value for Whitaker's beta diversity was described as the "number of distinct communities" (McCune and Grace 2002). When βw equaled zero, all of the sample units contained all of the species. Some multivariate methods strongly depend on beta diversity, and as a general rule beta diversity greater than 5 was considered high (McCune and Grace 2002).

Summary statistic means within *a priori* land use categories were compared with Fisher's Least Significant Difference (LSD) pair wise comparison test using Minitab (Version 13.1, Minitab Statistical Software). The strength of using Fisher's LSD was in the comparison of unequal group sizes (Ott and Longnecker 2001; Minitab 2000). Sample wetlands were divided into two groups based on 1995 LDI_F/wo scores including low (LDI < 2.0) and high (LDI ≥ 2.0) LDI groups. Comparisons were made for summary statistics between low and high LDI groups using the non-parametric Mann-Whitney U-Test in Minitab (Ott and Longnecker 2001).

Overall calculations of beta and gamma diversity were calculated for sample wetlands in the three *a priori* land use categories. Gamma diversity was calculated as the overall number of taxa encountered at all sample wetlands per *a priori* land use category. Higher gamma diversity for an *a priori* land use category suggested a greater difference among the species composition of wetlands within that *a priori* land use category, assuming a similar number of wetlands were sampled within each *a priori* land use category. Beta diversity was calculated as *a priori* category gamma diversity divided by the mean site taxa richness.

Regional Compositional Analysis

The Multi-Response Permutation Procedure (MRPP) was used to test the similarity of macrophyte community composition among Lane's (2000) four Florida wetland regions (see further application of MRPP in Zimmerman et al. 1985; McCune et al. 2000; McCune and Grace 2002). MRPP is a nonparametric technique which tests for no difference between groups (the null hypothesis) and is available in PCORD (Version 4.1 from MJM Software, Gleneden Beach, Oregon). It is an appropriate procedure for ecological community data as it does not require distributional assumptions of normality and homogeneity of variances. The Sørensen distance measure was used to calculate the average weighted within-group distance.

MRPP analysis provided a test statistic (T), p-value, and chance-corrected within-group agreement (A), which described within-group similarity. When A equaled one, all items were identical within groups, and when A equaled zero, differences within-groups equaled that expected by chance. McCune and Grace (2002) found that most values of A were less than 0.1 in community ecology. MRPP was calculated across all wetland region groups (panhandle versus north versus central versus south; wetland regions according to Lane 2000) as well as for multiple pair wise comparisons (panhandle versus north, panhandle versus central, panhandle versus south, north versus central, north versus south, and central versus south) when sample size was appropriate. An appropriate

sample size was defined by a minimum of two wetlands within each category for comparison. For example, if there were seven wetlands being compared with six in one region and only one in a different region, the MRPP comparison analysis could not be executed. Additional MRPP tests were calculated across all bioregions (panhandle versus northeast versus peninsula versus Everglades; bioregions according to Griffith et al. 1994) as well as for multiple pair wise comparisons (panhandle versus northeast, panhandle versus peninsula, panhandle versus Everglades, northeast versus peninsula, northeast versus Everglades, and peninsula versus Everglades) when sample size was appropriate.

Community Composition

Community composition of the macrophyte assemblage was summarized in a non-metric multidimensional scaling (NMDS) ordination. NMDS, an ordination technique designed to compress multi-dimensional space, is particularly agreeable with ecological data because it does not rely on linear relationships among variables, which generally do not accurately characterize ecological trends. This has been described as a compensation for the "zero-truncation problem" through the use of ranked distances (a common characteristic of non-parametric analyses) and the use of an appropriate distance measure (McCune and Grace 2002). The "zero-truncation problem" refers to the extraordinary number of zeros (denoting a species is absent from a sampling location) typical in community ecology data sets. For example, in the species by site matrix used in our data analysis (283 species by 24 sites), 85% of the cells within the matrix were assigned a zero (i.e. the species does not occur at the site). Other ordination techniques depend upon a value for each measured variable for each sample unit. However NMDS is useful for the presence/absence data sets, where many species are not present, or receive a zero in the species by site matrix. By ranking the variables, NMDS caters to the non-parametric community ecology data set.

NMDS was used to explore the dissimilarities of the community composition of the macrophyte assemblage among sample wetlands. The Sørensen (Bray-Curtis) distance measure was used for ordination. The dimensionality was chosen based on an initial 6-dimensional run in autopilot mode, which suggested an optimal 2-dimensional solution. To find the optimal 2 dimensional solution, 50 runs with real data and 50 randomized runs were performed with the instability criterion set at 0.00001 and the maximum number of iterations to reach a stable solution set at 500. This procedure was repeated 20 times to insure stability and reproducibility in results. The final run was completed with the starting point set as the results from the best experimental 2-dimensional run, with the lowest stress and best overall fit. LDI_F/wo, species richness, Shannon diversity, Simpson diversity, Whittaker's beta diversity, and sampling date (as Julian date) were tested for correlation with the NMDS ordination axes with Pearson correlation coefficients.

Metric Development

In the context of this study, metrics are defined as biological attributes which have a consistent and predictable response to anthropogenic activities (Karr and Chu 1997). Metrics were summarized in three main categories, including tolerance indicators,

community structure, and community balance. Metrics were as the proportion (P) by dividing the number of species (n) fitting a particular metric category (ex. exotic species) divided by the total species richness (N) for each sample wetland:

$$P_i = n_i / N_i \quad (2\text{-}8)$$

where i represents an individual wetland.

Scatter plots were constructed for each candidate metric versus 1995 LDI_F/wo_200m to ensure correlations were visually distinguishable. Candidate metrics were accepted if they successfully distinguished between low (LDI < 2.0) and high LDI (LDI ≥ 2.0) groups using the Mann-Whitney U-test ($p < 0.10$) calculated with Version 13.1, Minitab Statistical Software, and showed consistent and predictable trends along the gradient of anthropogenic land use activity according to the strength and significance of the Spearman's correlation coefficient ($p < 0.05$) calculated with Analyze-It software v. 1.67 for Microsoft Excel. The Spearman rank correlation tested for an association between two related variables, and was a non-parametric alternative to the Pearson correlation.

Macrophyte metrics were adopted from previous FWCI studies (Lane et al. 2003; Reiss and Brown 2005), included tolerance (tolerant species; sensitive species), exotic, FQAI, and longevity and plant growth form (native perennial). Additional metrics were explored for summary statistics (richness, evenness, and diversity), as well as dominance, annual to perennial ratio, wetland indicator status, native evergreen, native deciduous, and basal area (total, deciduous, or evergreen).

Tolerance metrics were calculated with Indicator Species Analysis (ISA) in PCORD (Version 4.1 from MJM Software, Gleneden Beach, Oregon). Sample wetlands were categorized into low and high LDI groups and analyzed with ISA, which evaluates the abundance and faithfulness of species in a defined group (McCune and Grace 2002). ISA is used to detect and describe the value of species indicative of environmental conditions. It requires *a priori* groups and data on the presence of taxa in each group. Groups are commonly defined by categorical environmental variables, levels of disturbance, experimental treatments, presence and absence of a target species, or habitat types (McCune and Grace 2002). Calculated indicator values are based on two standards, faithfulness and exclusion. Faithfulness is defined mathematically by a particular species always being present in a particular group. Additionally, the perfect indicator species is exclusive to that group, meaning it never occurs in other groups. Indicator values range from 0 (no indication) to 100 (a perfect indication of a particular group).

Multiple ISA iterations were conducted to determine sensitive and tolerant indicator species. Sample wetlands were categorized based on 1995 LDI_F/wo_200m scores, and ISA calculations were completed for consecutive LDI breaks from 1.0 through 3.5 at each 0.5 increment using the presence/absence of taxa at each wetland. In total ISA was run six times. Species significantly associated ($p < 0.10$) with the higher LDI group (LDI > 1.0; LDI > 1.5; LDI > 2.0; LDI > 2.5; LDI > 3.0; LDI > 3.5) for each iteration were included in the overall list of tolerant indicator species; species significantly associated ($p < 0.10$) with the lower LDI group (LDI = 1.0; LDI ≤ 1.5; LDI ≤ 2.0; LDI ≤ 2.5; LDI ≤ 3.0; LDI ≤ 3.5) for each iteration were included in the overall list of sensitive indicator species. Exotic species were excluded from the sensitive indicator species list, as this list was constructed to represent reference biological condition, which

should exclude the presence of nonnative species. A random number seed was used for each ISA iteration.

Calculated indicator values were tested for statistical significance using a Monte Carlo randomization technique with 1000 randomized runs. Indicator species categorized as tolerant species were associated with a higher LDI group; indicator species categorized as sensitive species were associated with a lower LDI group. Tolerant and sensitive indicator species lists were compiled based on a collection of significant indicator species at each incremental LDI break. This method was employed due to the small sample size of wetlands available to construct indicator species lists. The proportion indicator species metrics were calculated for each wetland as the number of indicator species (either tolerant or sensitive) divided by species richness.

The proportion exotic species metric was calculated as the number of species that were exotic to Florida divided by species richness. The timeline for determining the exotic status of a species was set near the beginning of European settlement in North America. Sources consulted to determine the exotic status of a species included Godfrey and Wooten (1981), Tobe et al. (1998), Wunderlin (1998), USDA NRCS (2002), and Wunderlin and Hansen (2003).

FQAI scores for each wetland were calculated based on the sum of CC scores for each species identified divided by total species richness at each wetland (Eq. 2-2). FQAI scores had the potential to range from 0.0-10.0. Wetlands with higher FQAI scores represented wetlands with species indicative of more stable environmental conditions.

The proportion native perennial species was calculated as the number of native perennial species encountered divided by wetland species richness. Each species was categorized as native or exotic and annual or perennial according to Godfrey and Wooten (1981), Tobe et al. (1998), Wunderlin (1998), USDA NRCS (2002), and Wunderlin and Hansen (2003). Wienhold and Van der Valk (1989) and Ehrenfeld and Schneider (1991) determined that disturbance often favors annual species over perennial species and promotes the invasion of nonnative perennials in wetlands. Galatowitsch et al. (2000) found that while native perennial cover was reduced in wetlands impacted by cultivation, the occurrence of introduced perennials rather than annuals increased in stormwater impacted wetlands.

Florida Wetland Condition Index

The Florida Wetland Condition Index (FWCI) consists of individual metrics, which were scaled and added together. Metric scoring was based on an approach from the Stream Condition Index (SCI), a Florida based biological index of the macroinvertebrate assemblage used to discern stream condition (Fore 2004) and also used in the development of the FWCI for depressional forested wetlands (Reiss 2004). Metrics were natural log transformed to improve the normality of distribution. The 5^{th} to 95^{th} percentile values of each metric were normalized from 0 to 10, with 10 always representing the reference biological wetland condition.

An agglomerative cluster analysis in PCORD was used to determine wetland groups (McCune and Grace 2002), for wetlands with similar macrophyte community composition. The dissimilarity matrix was constructed using the Sørensen distance measure and the flexible beta linkage method (β = -0.25), which is a flexible clustering

setting designed to reduce chaining in the dendrogram. The resulting dendrogram was pruned using ISA, which provided an objective, quantitative means of selecting the optimum number of clusters representing the most ecologically meaningful point of pruning (McCune and Grace 2002). Group membership determined in each step of the agglomerative cluster analysis was used as the grouping variable for ISA. Using the species by site presence/absence matrix, indicator values were calculated for each species occurring in three or more sites for each step of the dendrogram. This is an entirely separate application of ISA as that used to construct the metrics for tolerant and sensitive indicator species. This method of using ISA to determine the most appropriate and meaningful point on a dendrogram for understanding a significant number of groups for a data set is described more completely by McCune and Grace (2002). The resulting p-values from the randomized Monte Carlo tests were averaged among all species at each step of the dendrogram, and the cluster step with the lowest average p-value was used to determine the level in the dendrogram yielding the greatest ecologically meaningful information (McCune and Grace 2002). FWCI scores and LDI index values among clusters were then compared using Fisher's LSD pair wise comparison ($p < 0.05$) available in Minitab.

To determine how the biological data compared to the intensity of landscape development, Spearmans rank correlation was used to compare individual metrics and the FWCI with the multiple LDI calculations. For forested floodplain wetlands, comparisons of the FWCI and LDI_F/wo_200m were made with the HDG and SCI scores using the non-parametric Spearmans rank correlation coefficient (HDG and SCI scores were not available for forested strands). The LDI_F/wo_200m and HDG represented measures of anthropogenic influence; whereas the FWCI and SCI represented biological indices based on the macrophyte community composition in the forested wetland and macroinvertebrate community composition in the channelized stream flow, respectively.

CHAPTER 3
RESULTS

Gradients of Anthropogenic Activity

Two measures of anthropogenic activity in the landscape were correlated to physical, chemical, and biological measures of wetland condition, including the Landscape Development Intensity (LDI) index and the Human Disturbance Gradient (HDG).

Landscape Development Intensity

LDI index calculations were completed for the forested strand (Table 3-1) and floodplain (Table 3-2) wetlands. Transect (LDI_T) and feature (LDI_F) scale calculations were calculated for all 24 wetlands using 1995 land use coverages and 2000 land use coverages for wetlands located within the boundary of the St Johns River Water Management District (SJRWMD). Land uses in the 100 m buffers for LDI_T calculations were delineated from digital orthophoto imagery and assigned land use categories based on ground truthed land use maps drawn during the field site visit. Watershed (WS) scale LDI calculations were completed for the 14 forested floodplain wetlands.

A comparison between LDI calculations including (1995_LDI_T/w_100m) and excluding (1995 LDI_T/wo_100m) wetland area (Figure 3-1) show that calculations excluding the wetland area (1995 LDI_T/wo_100m) for agricultural and urban wetlands were consistently higher than those including the wetland area (1995 LDI_T/w_100m) for the forested wetlands, as can be seen by the values generally occurring above the 1:1 line. Four reference wetlands received a score of 1.0 using both calculation methods. In effect the calculation of LDI_T/w compensates for differences in wetland area between study wetlands. However, this difference in area may or may not correlate to differences in biological response such as changes in macrophyte community composition to anthropogenic activities in the landscape.

The correlation between 1995 LDI calculations at the transect and feature scales with 200 m buffers (Figure 3-2) was low ($R^2 = 0.32$). Because the forested strand and floodplain wetlands involve channelized flowing water, it is anticipated that the feature scale LDI would better reflect the biological response signal of organisms that are affected by upstream anthropogenic activities. Three wetlands, one in each *a priori* land use category, received identical scores for the transect (1995 LDI_T/wo_200m) and feature scales (1995 LDI_F/wo_200m). Ten wetlands received higher 1995 LDI_F/wo_200m than 1995 LDI_T/wo_200m scores, including six reference, one agricultural, and three urban wetlands; whereas the remaining 11 wetlands received higher 1995 LDI_T/wo_200m scores, including five agricultural and six urban wetlands.

Watershed scale LDI calculations using equal weighting for all area within a watershed, excluding the wetland area, (1995 LDI_WS_ED/wo) and a distance weighted

Table 3-1. Landscape Development Intensity (LDI) index scores for 10 forested strands using 1995 and 2000 land use coverages. LDI calculations were completed at the transect (T) and wetland feature (F) scale, and including (/w) and excluding (/wo) the wetland area. See text for further details of individual calculations.

	FS1	FS2	FS3	FS4	FS5	FS6	FS7	FS8	FS9	FS10
A Priori Group	R	A	A	U	U	U	U	A	R	R
1995 LDI_T/w_100m	1.2	2.4	2.3	1.2	1.9	3.7	1.0	2.6	1.3	1.2
1995 LDI_T/wo_100m	1.3	3.5	3.3	1.4	2.3	4.6	1.0	3.5	1.4	1.2
1995 LDI_T/w_200m	1.3	2.6	2.4	1.2	2.1	3.5	1.0	2.9	1.4	1.0
1995 LDI_T/wo_200m	1.3	3.2	3.5	1.3	3.5	3.9	1.0	3.2	1.5	1.0
1995 LDI_F/w_200m	2.6	2.5	3.1	1.9	3.0	3.0	1.0	3.0	1.7	1.0
1995 LDI_F/wo_200m	3.0	2.9	3.1	1.9	3.5	3.2	1.0	3.1	2.1	1.0
2000 LDI_T/w_100m	1.2	2.6	2.3	4.3	5.4	5.8	1.0	6.1	1.0	1.0
2000 LDI_T/wo_100m	1.2	3.8	2.5	8.0	7.0	7.5	1.0	8.0	1.0	1.2
2000 LDI_T/w_200m	1.5	2.6	2.3	na	na	na	na	6.9	1.4	na
2000 LDI_T/wo_200m	1.5	3.2	2.3	na	na	na	na	7.4	1.6	na
2000 LDI_F/w_200m	2.5	2.5	2.6	na	na	na	na	6.9	1.7	na
2000 LDI_F/wo_200m	2.9	2.9	3.0	na	na	na	na	7.2	2.1	na

na – 2000 land use coverages were not available for these wetlands.

Table 3-2. Landscape Development Intensity (LDI) index scores for 14 forested floodplain wetlands using 1995 and 2000 land use coverages. LDI calculations were completed at the transect (T), wetland feature (F), and watershed (WS) scale, and including (/w) and excluding (/wo) the wetland area. See text for further details of individual calculations.

Site Code	FF1	FF2	FF3	FF4	FF5	FF6	FF7	FF8	FF9	FF10	FF11	FF12	FF13	FF14
A Priori Land Use	U	R	U	A	U	R	R	U	U	U	A	A	R	A
1995 LDI_T/w_100m	1.9	1.0	5.2	1.5	1.9	1.1	1.6	3.2	1.0	4.8	1.2	4.2	1.0	1.2
1995 LDI_T/wo_100m	3.2	1.0	7.6	2.2	3.3	1.1	2.1	4.4	1.0	6.9	1.6	6.3	1.0	1.5
1995 LDI_T/w_200m	2.2	1.0	5.9	1.2	2.3	1.0	1.0	3.3	1.1	4.0	1.3	4.3	1.0	1.7
1995 LDI_T/wo_200m	3.1	1.0	7.1	1.3	3.1	1.0	1.0	3.9	1.1	4.7	1.6	6.5	1.0	2.1
1995 LDI_F/w_200m	1.5	1.1	3.2	1.6	1.7	1.4	1.9	2.0	1.5	5.7	1.6	2.4	1.9	1.7
1995 LDI_F/wo_200m	2.0	1.2	3.7	2.3	1.9	1.4	2.5	2.5	1.8	7.1	1.6	2.8	2.1	1.8
1995 LDI_WS_ED/w	2.4	1.5	3.8	2.2	1.8	2.7	3.1	2.2	1.7	2.1	1.4	1.9	1.8	2.0
1995 LDI_WS_ED/wo	2.5	1.5	2.9	2.3	1.9	2.7	3.4	2.5	1.7	2.1	1.4	1.9	1.8	2.1
1995 LDI_WS_DW_lin	2.2	1.4	2.2	2.2	2.1	2.7	2.8	2.3	1.5	2.2	1.4	2.0	1.7	2.0
1995 LDI_WS_DW_exp	2.3	1.4	2.7	2.2	2.1	2.6	2.6	2.6	1.5	2.8	1.4	2.8	1.6	1.9
2000 LDI_T/w_100m	1.9	1.0	5.2	1.5	1.9	1.1	1.6	3.3	1.0	5.6	1.2	2.9	1.0	1.2
2000 LDI_T/wo_100m	3.2	1.0	7.6	2.2	3.3	1.1	2.1	4.5	1.0	8.1	1.6	5.2	1.0	1.5
2000 LDI_T/w_200m	2.2	1.0	5.9	1.0	2.2	na	na	3.4	1.3	na	na	4.1	na	na
2000 LDI_T/wo_200m	3.1	1.0	7.1	1.1	3.1	na	na	4.8	1.4	na	na	5.6	na	na
2000 LDI_F/w_200m	1.5	1.1	3.0	1.6	1.8	na	na	2.0	1.5	na	na	2.3	na	na
2000 LDI_F/wo_200m	2.0	1.3	3.5	2.2	2.0	na	na	2.5	1.7	na	na	2.7	na	na
2000 LDI_WS_ED/w	2.3	1.5	2.3	2.2	1.7	na	na	2.3	1.6	na	na	1.9	na	na
2000 LDI_WS_ED/wo	2.4	1.5	2.3	2.3	1.8	na	na	2.6	1.8	na	na	1.9	na	na
2000 LDI_WS_DW_lin	2.2	1.5	2.4	2.2	1.9	na	na	2.3	1.7	na	na	2.0	na	na
2000 LDI_WS_DW_exp	2.2	1.4	2.8	2.2	1.9	na	na	2.6	1.6	na	na	2.8	na	na

na – 2000 land use coverages were not available for these wetlands.

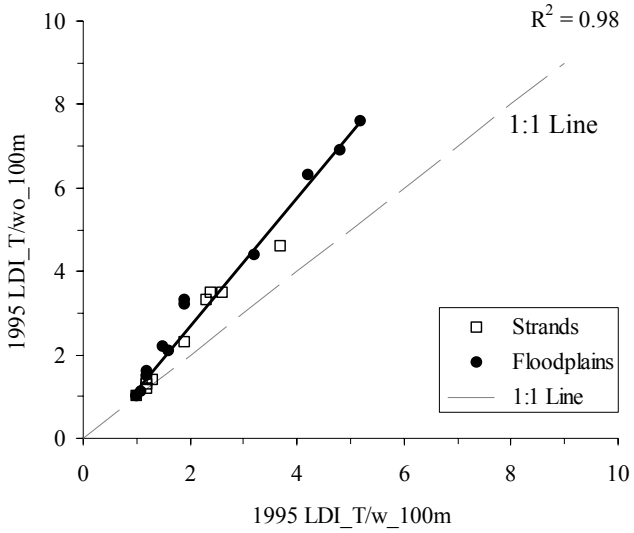


Figure 3-1. Comparison between the LDI calculations including (1995_LDI_T/w_100m) and excluding (1995 LDI_T/wo_100m) wetland area (R^2 = 0.98). Calculations excluding the wetland area (1995 LDI_T/wo_100m) for agricultural and urban wetlands were consistently higher than those including the wetland area (1995 LDI_T/w_100m) for the forested wetlands. Four reference wetlands received a score of 1.0 using both calculation methods.

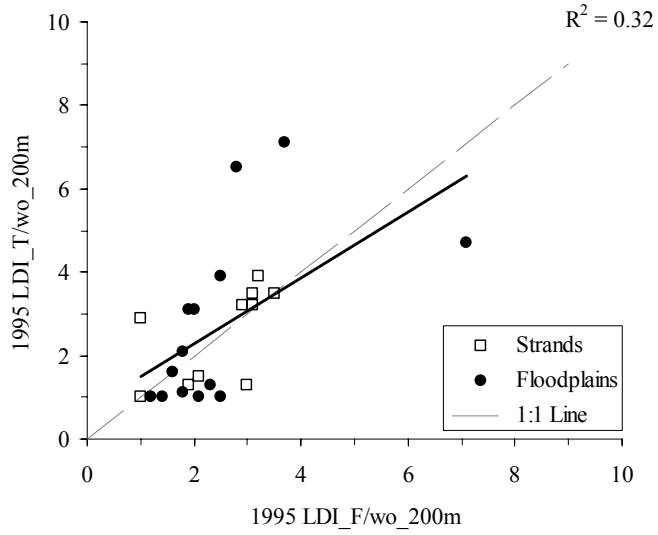


Figure 3-2. Comparison between LDI calculations at the transect (1995_LDI_T/wo_200m) and feature (1995 LDI_F/wo_200m) scale for 24 forested wetlands (R^2 = 0.32).

approach using linear weighting (1995 LDI_WS_DW_lin) showed a stronger correlation ($R^2 = 0.82$) for the 14 forested floodplain wetlands (Figure 3-3).

LDI calculations were also completed for multiple years, using land use/land cover data from 1995 and 2000. Comparison between 1995 and 2000 LDI calculations at the feature scale (without the wetland area) for 13 forested wetlands showed a moderate correlation ($R^2 = 0.41$) (Figure 3-4). However, 12 of the wetlands had nearly equal LDI scores for the 1995 and 2000 LDI_F/wo_200m calculations with one clear disagreement, FS8, which had a significant area of the 200m buffer categorized as Low Intensity Pasture (with livestock) in 1995 and Institutional in 2000. The land use was not actually changed, only the classification on the GIS coverage was changed, as the property is part of the University of Florida's (thus the Institutional classification) Beef Research Unit (thus the Low Intensity Pasture (with livestock) classification). Removing FS8 because of erroneous differences in the land use/land cover coverages, the remaining 12 wetlands had strongly correlated 1995 and 2000 LDI_F/wo_200m calculations ($R^2 = 0.99$ without FS8).

LDI_F/wo_200m LDI calculations for 1995 land use/land cover were selected for further analysis for four reasons. First, 1995 land use/land cover data were available for all of the 24 forested wetlands sampled. Second, calculations for 13 forested wetlands having both 1995 and 2000 LDI_F/wo_200m calculations showed similar values as described above (Figure 3-4). Using the older land use/land cover data from 1995 may have limited implications in changes to LDI scores as changes in land use are localized to areas with recent development. In these areas it may be argued that more recent coverages are needed, however at the statewide scale the 1995 land use coverage provides a baseline for LDI calculations. We are not suggesting that the most recent and complete coverage should not be used, but simply that the LDI calculation should not be disregarded because the coverage is ten years old. Third, because the 24 wetlands surround channelized flowing water, the feature scale LDI calculations were used to capture the human activity influencing the upstream wetlands. Fourth, the LDI_T_100m calculations included hand delineation of land use according to interpretation of digital orthophoto imagery and ground truthing; however, the application of LDI as a remote (GIS computer based) tool for assessing human activity in the landscape, using widely available coverages, was an important consideration for future application.

Human Disturbance Gradient

Thirteen of the forested floodplain wetlands sampled corresponded to storet stations established by FDEP for water quality monitoring. Data were available for the four components of the Human Disturbance Gradient (HDG), including ammonia concentration, hydrologic condition, habitat assessment (habitat condition index), and LDI for the buffer (Table 3-3). Using the scoring criteria established by FDEP (Table 1-2), each of the HDG components were scored individually and summed to get the HDG for each storet station. HDG scores were adopted directly from previous FDEP assessments, and no scoring was done directly for this research. HDG scores potentially range from 0-9, with 0 representing the reference condition and 9 representing a highly impaired condition. The maximum HDG score for the 13 forested floodplain wetlands was a 4 at the urban wetland FF10 (storet 28020234) for the most recent sampling period

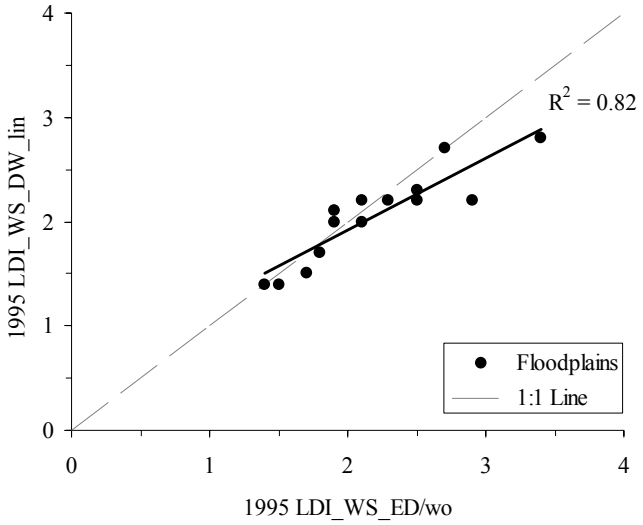


Figure 3-3. Comparison between LDI calculations at the watershed scale using equal weighting for all area within a watershed (1995 LDI_WS_ED/wo) and a distance weighted approach using linear weighting (1995 LDI_WS_DW_lin) (R^2 = 0.82).

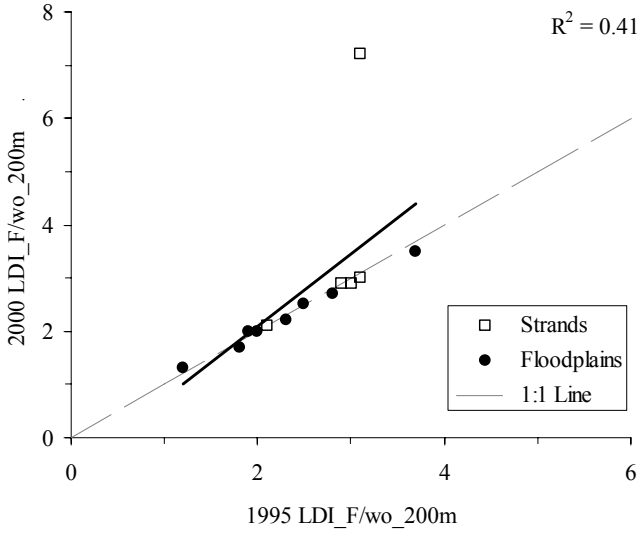


Figure 3-4. Comparison between 1995 and 2000 LDI calculations at the feature scale (without the wetland area) for 13 forested wetlands (R^2 = 0.41).

Table 3-3. Human Disturbance Gradient (HDG) for 13 storet stations, which correspond with the forested floodplains sampled. Some forested floodplain wetlands have multiple HDG calculations for different sampling dates (1992-2000). The four components of the HDG were also provided, including ammonia concentration, Hydrologic Condition, Habitat Index, and LDI buffer. HDG scoring criteria were presented in Table 1-2.

Code	STORET	HDG Date	HDG	Ammonia (NH_3) (mg N/L)	NH_3 SCORE	Hydrologic Condition	Hydro SCORE	Habitat Assessment	Habitat SCORE	LDI Buffer	LDI_BF SCORE
FF1	20020404	8/15/2000	0	*	0	4	0	84	0	1.38	0
FF1	20020404	8/15/2000	0	0.04	0	4	0	84	0	1.38	0
FF2	20010454	7/19/1999	0	0.01	0	1	0	88	0	1.06	0
FF3	27010580	8/22/2000	2	0.10	1	4	0	58	1	1.79	0
FF4	20010455	2/23/1999	0	- -	- -	2	0	84	0	1.02	0
FF5	19010099	8/17/1993	1	*	0	3	0	65	1	1.27	0
FF5	19010099	3/5/1998	0	- -	- -	3	0	88	0	1.27	0
FF5	19010099	2/1/2000	0	0.04	0	3	0	91	0	1.27	0
FF5	19010099	7/9/1996	0	0.02	0	3	0	90	0	1.27	0
FF5	19010099	8/18/1992	0	- -	- -	3	0	91	0	1.27	0
FF6	26011019	3/8/1999	0	- -	- -	3	0	86	0	1.51	0
FF7	26011020	7/20/1999	0	*	0	2	0	89	0	1.04	0
FF8	19020027	7/15/1996	1	0.08	0	7	1	88	0	1.40	0
FF9	20030246	4/17/1997	0	*	0	2	0	99	0	1.21	0
FF10	28020234	10/9/2000	4	0.12	1	7	1	81	0	4.73	2
FF10	28020234	9/11/1996	3	0.06	0	7	1	86	0	4.73	2
FF11	21010032	1/17/1995	0	- -	- -	3	0	97	0	1.16	0
FF12	19010072	3/22/1999	0	*	0	2	0	86	0	1.56	0
FF14	32010024	7/14/1998	0	0.02	0	5	0	75	0	1.33	0

- - no data available

*below detection limits

(10/9/2000). [Multiple data points were available for many of the SCI points used in this research project, and the most complete data from the most recent sampling period was used for correlation purposes.] The moderate to low range of scores suggests that the wetlands sampled represented wetlands on the lower scale of impairment. This may be an accurate statement when considering that many stream sampling locations with SCI data were visited during site reconnaissance, but found unacceptable for this study because the floodplain wetland had been completed altered, often because the area had been sodded or paved as part of adjacent human activity in the surrounding landscape. These sites, considered highly impaired on the SCI, were not considered for macrophyte sampling and FWCI development, as they no longer hosted wetland vegetation.

Water Quality

Water quality data (chemical and physical parameters) were obtained for 13 of the forested floodplain wetlands sampled that corresponded to storet stations. Water parameters available included temperature (^{o}C), pH, specific conductivity (umhos/cm), dissolved oxygen (mg O_2/L), turbidity (NTU), Total Kjeldahl Nitrogen (mg/L), ammonia (as mg N/L), and total phosphorus (mg P/L) (Table 3-4). FF5 an urban wetland had the lowest values for temperature (7.1 ^{o}C), specific conductivity (49 umhos/cm), and the highest value for dissolved oxygen (10.2 mg O_2/L). In contrast FF8, another urban wetland, had the highest values for turbidity (5.7 NTU), Total Kjeldahl Nitrogen (1.50 mg N/L), and total phosphorus (0.28 mg P/L). A third urban wetland FF10 had the lowest value for dissolved oxygen (2.6 mg O_2/L), and the highest values for temperature (25.8 ^{o}C) and ammonia nitrogen (0.120 mg N/L).

Data Analysis

Statewide, 24 wetlands were sampled with 283 macrophyte species, representing 190 genera and 97 families identified. The most abundant species was the vine *Vitis rotundifolia* (muscadine), which was found rooted within vegetation quadrats at 23 of 24 study wetlands. The second most abundant species was the tree *Acer rubrum* (red maple) found at 75% of the study wetlands. The most common fern was *Woodwardia virginica* (Virginia chain fern) found at 54% of the wetlands; the most common graminoid was *Panicum commutatum* (variable witchgrass) found at 46% of the wetlands. *Boehmeria cylindrica* (false nettle) was the most common herbaceous species, found at 63% of the wetlands; the two most common shrubs were *Cephalanthus occidentalis* (buttonbush) and *Myrica cerifera* (wax myrtle), each found at 71% of the wetlands; the second most common tree was *Sabal palmetto* (cabbage palm) found at 71% of the wetlands; and the second most common vine was *Toxicodendron radicans* (Eastern poison ivy) found at71% of the wetlands. Of the species encountered, 110 species (39%) occurred at a minimum of three sample wetlands (12.5%). Nearly one-fifth of the species identified (48 species or 17%) were identified at only two sample wetlands; approximately two-fifths of the species identified (126 species or 44%) were found at only one wetland. In total the forested strand wetlands hosted 161 species, and the forested floodplain wetlands hosted 214 species. The forested strand and floodplain wetlands had 91 species in common.

Table 3-4. Water quality (chemical and physical parameters) for 13 storet stations, which correspond with the forested floodplains sampled. Three forested floodplain wetlands (FF1, FF5, and FF10) have multiple data from different sampling dates (1993-2000).

Code	Date	STORET	Temperature °C	pH	Specific Conductivity umhos/cm	Dissolved Oxygen mg /L	Turbidity NTU	Total Kjeldahl Nitrogen mg/L	Ammonia Nitrogen mg/L	Total Phosphorus mg/L
FF1	8/15/2000	20020404	23.7	6.60	121	6.2	- -	0.75	0.004	0.08
FF1	8/15/2000	20020404	23.7	6.61	121	6.2	1.2	0.75	0.038	0.08
FF2	7/19/1999	20010454	23.2	7.10	1580	6.8	1.8	0.27	0.012	0.04
FF3	8/22/2000	27010580	25.0	7.40	746	2.7	- -	0.62	0.100	0.13
FF4	2/23/1999	20010455	12.8	6.90	230	7.2	- -	- -	- -	- -
FF5	8/17/1993	19010099	25.0	6.36	70	- -	3.2	0.34	0.000	0.04
FF5	3/5/1998	19010099	- -	- -	- -	- -	- -	0.47	- -	0.02
FF5	2/1/2000	19010099	7.1	5.10	75	10.2	1.9	0.25	0.039	0.02
FF5	7/9/1996	19010099	24.7	5.25	49	5.3	- -	0.66	0.024	0.02
FF6	3/8/1999	26011019	18.5	6.90	219	7.0	- -	- -	- -	- -
FF7	7/20/1999	26011020	25.6	6.72	172	6.6	1.5	0.26	0.000	0.00
FF8	7/15/1996	19020027	25.7	5.90	101	4.2	5.7	1.50	0.080	0.28
FF9	4/17/1997	20030246	16.7	5.60	68	7.7	3.5	0.46	0.000	0.00
FF10	10/9/2000	28020234	24.6	7.20	399	2.6	2.0	0.88	0.120	0.18
FF10	9/11/1996	28020234	25.8	7.40	451	3.1	- -	0.75	0.059	0.11
FF11	1/17/1995	21010032	11.6	3.97	79	8.3	- -	- -	- -	- -
FF12	3/22/1999	19010072	16.4	6.10	77	6.8	- -	0.46	0.000	0.05
FF14	7/14/1998	32010024	23.3	7.70	189	7.1	3.0	0.12	0.017	0.02

- - no data available

Summary Statistics

Species richness (R), species evenness (E), Shannon diversity (H'), Simpson diversity (D), and Whittaker's beta diversity (βW) were calculated based on the macrophyte assemblage for each sample wetland. Species richness ranged from 21 species at FF2 (a reference floodplain forested wetland), to 77 species at FF5 (a wetland embedded in an urbanized landscape). Species evenness was consistent among all *a priori* land use groups, and species evenness only showed differences at the hundred thousandth decimal place, explaining why reported values were identical for two decimal places. Shannon diversity ranged from 3.05 to 4.34 at FF2 and FF5, respectively; these were the same wetlands with the lowest (21) and highest (77) species richness, respectively. Similarly, FF2 had the lowest Simpson diversity index value (0.95), and FF5 and FF8 (another urban wetland) shared the highest Simpson diversity index value (0.99). Whittaker's beta diversity ranged from a low of 1.8 at FF6 (a reference wetland), to a high of 7.4 at FF4 (an agricultural wetland). A complete table of summary statistics for each wetland is presented in Appendix C.

Table 3-5 summarizes comparisons of mean richness, evenness, and diversity calculations by *a priori* land use category. Urban wetlands had the mean greatest species richness (50 ± 17) followed by agricultural wetlands (41 ± 10). This trend was evident for both Shannon diversity, with urban wetlands having greater mean values (3.86 ± 0.35). Whittaker's beta diversity was greatest in agricultural wetlands (4.94 ± 1.07). Beta and gamma diversity were calculated for overall *a priori* land use categories, with urban wetlands having the highest beta and gamma diversity (4.0 and 201, respectively). None of the summary statistics were significantly different among *a priori* land use categories (Fisher's LSD pair wise comparison, $\alpha = 0.05$) as the standard deviations overlap for most of the summary statistics.

Similarly, summary statistics (including species richness, species evenness, Shannon diversity, and Whittaker's beta diversity) were not significantly different between low (1995 LDI_F/wo_200m < 2.0) and high (1995 LDI_F/wo_200m ≥ 2.0) LDI groups (Mann-Whitney U-Test; Table 3-6). However, wetlands in the high LDI group had higher (though not statistically significant) mean species richness (47 ± 17), Shannon diversity (3.80 ± 0.34), Whittaker's beta diversity (4.56 ± 1.44), and overall beta (4.9) and gamma (233) diversity.

Regional Compositional Analysis

MRPP was calculated across all wetlands grouped according to wetland regions (Lane 2000), as well as for multiple pair wise comparisons. Sample size limitations prevented all possible wetland region pair wise comparisons when a minimum of two wetlands per wetland region and land use category were not available. Results for the MRPP tests included the test statistic (T), chance-corrected within-group agreement (A), and significance value (p) (Table 3-7). Only the MRPP pair wise comparison of the panhandle and north wetland regions for the compiled pool of all wetlands (including *a priori* reference, agricultural, and urban) was not significant at the $\alpha = 0.05$ level, suggesting that there were regionally significant differences among species composition across all wetland regions. Of the possible pair wise comparisons within *a priori* land use categories, the MRPP analysis did not suggest a difference among species composition in north and central reference wetlands or agricultural wetlands for all

Table 3-5. Richness, evenness, and diversity of the macrophyte assemblage among *a priori* land use categories.

	Reference	Agricultural	Urban
Species richness (R)	36 ± 11[a]	41 ± 10[a]	50 ± 17[a]
Species evenness (E)	1.00 ± 0.00[a]	1.00 ± 0.00[a]	1.00 ± 0.00[a]
Shannon diversity (H')	3.55 ± 0.32[a]	3.69 ± 0.25[a]	3.86 ± 0.35[a]
Simpson's diversity (D)	0.97 ± 0.01[a]	0.97 ± 0.01[a]	0.98 ± 0.01[a]
Whittaker's Beta diversity (βW)	3.53 ± 1.03[a]	4.94 ± 1.07[a]	4.57 ± 1.38[a]
Beta diversity	3.7	3.7	4.0
Gamma diversity	134	153	201

Categories with similar letters were not significantly different (Fisher's LSD, α=0.05)

wetland regions. However, all MRPP tests among urban wetlands suggested a difference in the species composition of wetlands among the wetland regions (multiple comparison panhandle versus north versus central versus south; pair wise north versus central; and pair wise north versus south). While limitations to interpretation are apparent due to small sample sizes within each wetland region, the differences in species composition among urban wetland showed the greatest variability.

The MRPP analysis was repeated using wetlands grouped according to bioregions (Griffith et al. 1994). Of the forested wetland sampled, only one was located in the panhandle bioregion and one in the Everglades bioregion. All of the reference wetlands (n=7) were located in the peninsula bioregion. Thus the only comparisons possible were between *a priori* agricultural and urban wetlands in the northeast and peninsula ecoregions, limiting the MRPP analysis to two comparisons. A significant difference in species composition was not found among agricultural or urban wetlands in the northeast and peninsula bioregions (Table 3-7).

Table 3-6. Richness, evenness, and diversity of the macrophyte assemblage between low (LDI < 2.0) and high (LDI ≥ 2.0) LDI groups.

	Low LDI	High LDI	W^	p`
Species richness (R)	39 ± 11	47 ± 17	120	0.32
Species evenness (E)	1.00 ± 0.00	1.00 ± 0.00	153	0.38
Shannon diversity (H')	3.62 ± 0.31	3.80 ± 0.34	120	0.32
Simpson's diversity (D)	0.97 ± 0.01	0.98 ± 0.01	120	0.32
Whittaker's Beta diversity (βW)	4.16 ± 1.44	4.56 ± 1.44	123	0.42
Beta diversity	4.3	4.9		
Gamma diversity	169	233		

W^ = Mann-Whitney U-test statistic p` = significance value

Table 3-7. Macrophyte community composition similarity among wetland regions (Lane 2000) and bioregions (Griffith et al. 1994) with MRPP.

	Sites (n)	T^	A`	p#
Florida Ecoregions (Lane 2000)				
All wetlands				
All regions (P vs N vs C vs S)	24	-4.2	0.07	0.00*
Panhandle vs north	12	0.2	-	0.56
Panhandle vs central	10	-1.9	0.04	0.04*
Panhandle vs south	6	-2.4	0.12	0.02*
North vs central	18	-3.2	0.04	0.01*
North vs south	14	-3.7	0.07	0.01*
Central vs south	12	-4.0	0.07	0.00*
Reference wetlands				
North vs central	6	-1.2	0.04	0.10
Agricultural wetlands				
All regions (P vs N vs C vs S)	7	0.4	-	0.65
Urban wetlands				
All regions (P vs N vs C vs S)	10	-4.4	0.18	0.00*
North vs central	7	-2.8	0.13	0.01*
North vs south	8	-3.9	0.15	0.01*
Bioregions (Griffith et al. 1994)				
Agricultural wetlands				
Northeast vs peninsula	6	-1.5	0.12	0.07
Urban Wetlands				
Northeast vs peninsula	9	-1.6	0.05	0.07

* A high |T| value and significant p-value ($p<0.05$) suggests a difference in species composition.
^ T = the MRPP test statistic
` A = the chance corrected within-group agreement
p = the significance value.

Community Composition

Macrophyte community composition was summarized in a 2-dimensional nonmetric multidimensional scaling (NMDS) ordination to explore gradients in macrophyte community composition (Figure 3-5). The final solution had an overall stress of 17.8 with a final stability of 0.0002, which is considered a fair stress value useful for ordinations with community data sets (Kruskal 1964; Clarke 1993; McCune and Grace 2002). Axis 1 explained 53.3% variance, and axis 2 explained 29.2% additional variance. No measured variables were correlated with the ordination axes, including 1995 LDI_F/wo_200m, species richness, Shannon diversity, Simpson diversity, Whittaker's beta diversity, and Julian date.

Wetlands appeared to be broadly grouped on axis 1 according to wetland region, and perhaps the gradients of latitude and longitude would have been appropriate

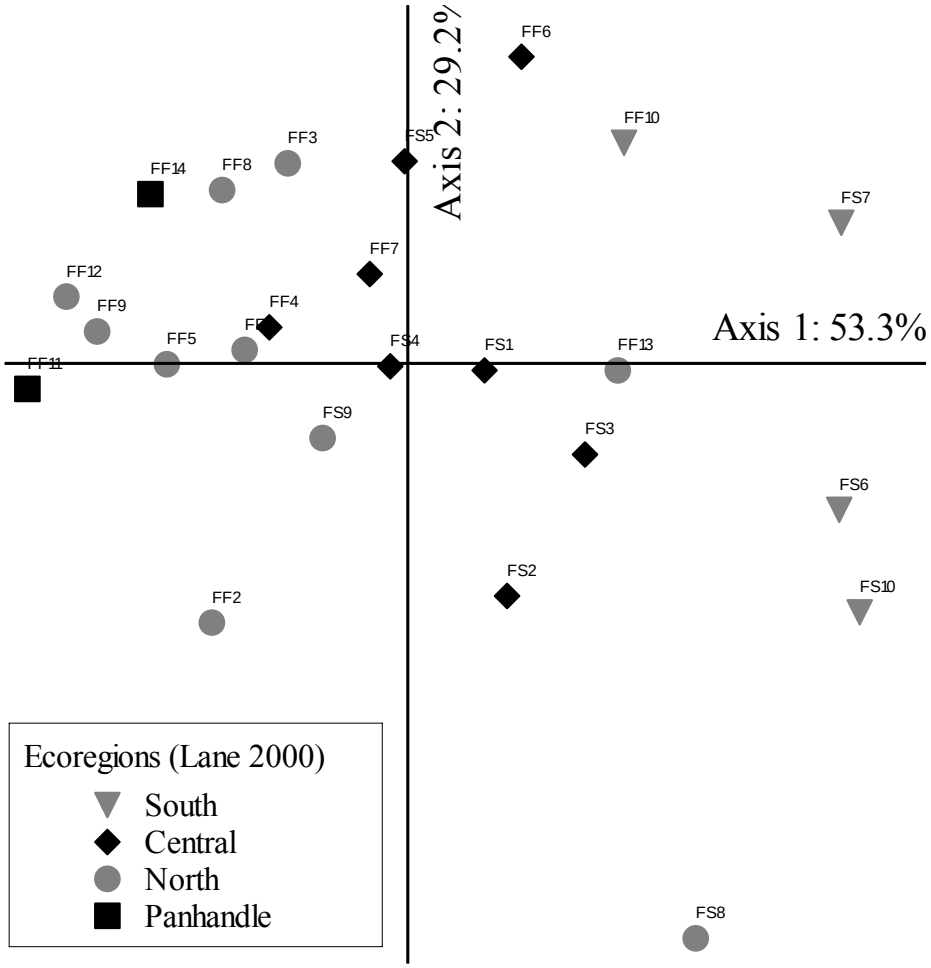


Figure 3-5. NMDS ordination bi-plot of 24 sample wetlands in macrophyte species space; wetlands are labeled according to site code; symbols correlate to wetland regions (Lane 2000). Axis 1 explained 53.3% variance; axis 2 explained an additional 29.2% variance.

explanatory variables for the ordination axes. For purposes of speculation, the second axis may be correlated to wetland type, as only two of the forested strands (FS5 and FS7) fell above the horizontal line of axis 2 at the center of the plot; whereas the remaining eight strands fell on or below the horizontal line. In contrast, only three of the forested floodplain wetlands (FF2, FF11, and FF13) fell below the horizontal line; while, the remaining 11 floodplain wetlands fell on or above the horizontal line. Perhaps there were distinct differences in the macrophyte community composition between wetland types (strand versus floodplain) detected with the NMDS, which could be supported by the observation that the forested strand and floodplain wetlands in this study shared just 91 species. This correlates to 43% of the species identified at any of the 14 forested floodplain wetlands also occurring at any of the 10 forested strand wetlands, which had fewer species identified (161 species) meaning that 57% of the species identified within the forested strands also occurred in at least one floodplain wetland. However,

considering that 126 species (45% of the entire 283 species identified for this study) were only sampled at one of the 24 wetlands included in this study, perhaps the dissimilarity of species composition among wetland type (strand versus floodplain) is not as significant as some other unmeasured environmental variable. A meaningful grouping of wetlands based on *a priori* land use classification was not apparent.

Metric Selection

Five metrics that were significantly correlated with LDI (Spearmans correlation coefficient $|r| \geq 0.50$, $p < 0.05$) were selected for inclusion in the preliminary FWCI for forested strand and floodplain wetlands (Table 3-8). Metrics selected for inclusion were the proportion tolerant indicator species (TOL); proportion sensitive indicator species (SEN); Floristic Quality Assessment Index (FQAI); proportion exotic species (EX); and proportion native perennial species (NP). The proportion of tolerant indicator species and proportion exotic species increased with increasing development intensity; whereas, the proportion sensitive indicator species, FQAI, and proportion native perennial species decreased with increased landscape development intensity. All metrics significantly ($p < 0.05$) differentiated between low (LDI<2.0) and high (LDI≥2.0) 1995 LDI_F/wo_200m groups (Table 3-9).

Tolerance metrics

Tolerant and sensitive indicator species were determined statewide using Indicator Species Analysis (PCORD). Table 3-10 provides a list of 19 tolerant indicator species. Figure 3-6 shows a scatter plot of proportion tolerant indicator species versus 1995 LDI_F/wo_200m. The proportion tolerant indicator species increased with increasing development intensity. FF10 (an urban floodplain forest in the south wetland region (Lane 2000) or Everglades bioregion (Griffith et al. 1994) had the highest proportion tolerant indicator species (0.29), followed by FS5 (0.28) and FF7 (0.25), which were both in the central wetland region (Lane 2000) or peninsula bioregion (Griffith et al. 1994). Two wetlands in the peninsula bioregion (Griffith et al. 1994) had less than 0.05 proportion tolerant indicator species, including FS10 a reference strand in the south wetland region (Lane 2000) and FF2 a reference floodplain in the north wetland region (Lane 2000).

Table 3-8. Spearmans correlation coefficients for macrophyte metrics and FWCI with 1995 LDI_F/wo_200m.

Metric	Spearman's r	p-value
Proportion tolerant indicator species	0.65	0.001
Proportion sensitive indicator species	-0.83	<0.001
FQAI score	-0.50	0.012
Proportion exotic species	0.54	0.006
Proportion native perennial species	-0.56	0.004
FWCI	-0.75	<0.001

Table 3-9. Comparisons among five macrophyte metrics and the FWCI between low (LDI < 2.0) and high (LDI ≥ 2.0) LDI groups (LDI_F/wo_200m).

Metric	Low LDI	High LDI	W^	p`
Tolerant indicator species	0.11 ± 0.06	0.19 ± 0.06	115.5	0.007
Sensitive indicator species	0.15 ± 0.08	0.05 ± 0.05	14	0.001
FQAI score	4.72 ± 0.68	4.10 ± 0.64	34	0.036
Exotic species	0.03 ± 0.04	0.10 ± 0.08	109	0.022
Native perennial species	0.95 ± 0.06	0.86 ± 0.11	34	0.036
FWCI	38.33 ± 7.91	21.54 ± 12.00	16	0.001

Values represent the mean ± standard deviation
W^ = the Mann-Whitney U-Test statistic
p` = the significance value

Table 3-11 provides a list of the 16 sensitive indicator species. Figure 3-7 shows that the proportion sensitive indicator species decreased with increasing development intensity. The wetland with the highest proportion sensitive indicator was FF2 (0.38; a reference wetland), followed by and FF9 (0.18; an urban wetland) and FS7 (0.17; a reference wetland). Three wetlands had no sensitive indicator species including two in the central wetland region (Lane 2000) or peninsula bioregion (Griffith et al. 1994) (the urban strand FS5 and the agricultural strand FS3) and one wetland in the south wetland region (Lane 2000) or Everglades bioregion (Griffith et al. 1994) (FF10).

Shrub and tree species were included in the ISA for both tolerant and sensitive metrics. Metrics developed based on the macrophyte community composition included woody species rooted within the sampling quadrats, as structure was thought to play an important role in the biological condition of flowing water forested wetlands. Excluding the tree and shrub layers would seemingly underscore the importance of these woody species. In fact, tree and shrub species comprised 52.6% of the tolerant and 62.5% of the sensitive indicator species lists (Tables 3-10 and 3-11). The five tolerant indicator tree species were *Acer rubrum* (red maple), *Ilex cassine* (dahoon holly), *Liquidambar styraciflua* (sweetgum), *Nyssa sylvatica* var. *biflora* (swamp tupelo), and *Prunus caroliniana* (Carolina laurelcherry) (Table 3-10); the five tolerant indicator shrub species were *Callicarpa americana* (American beautyberry), *Cephalanthus occidentalis* (buttonbush), *Ludwigia peruviana* (water-primrose), *Sambucus canadensis* (elderberry), and *Viburnum obovatum* (Walter viburnum). Vines also comprised a high percentage of the tolerant indicator species list, including *Berchemia scandens* (rattan vine), *Smilax auriculata* (earleaf greenbrier), *Smilax tamnoides* (bristly greenbrier), and *Toxicodendron radicans* (Eastern poison ivy).

Of the sensitive indicator species, 43.8% were shrubs, 18.8% trees, 18.8% herbs, 12.5% graminoids, 6.3% ferns, and 0% vines (Table 3-11). The three sensitive indicator tree species included *Fraxinus caroliniana* (Carolina ash), *Persea palustris* (swamp bay) and *Pinus elliottii* (slash pine). Seven shrub species were categorized as sensitive indicator species, including *Agarista populifolia* (Florida hobble-bush), *Hypericum hypericoides* (St. Andrew's cross), *Ilex coriacea* (bay-gall holly), *Lyonia lucida* (fetter-

Table 3-10. Tolerant indicator species for forested strand and floodplain wetlands.

Botanical Name	Common Name	LDI Break	(Indicator Value, p-value)	Growth Form
Acer rubrum	Red Maple	1.0	(81.8, 0.057)	Tree
		1.5	(90, 0.005)	
Berchemia scandens	Rattan Vine	3.5	(58.3, 0.062)	Vine
Bidens alba	Hairy Beggar-Ticks	3.5	(54.9, 0.097)	Herb
Callicarpa americana	American Beautyberry	3.5	(75, 0.053)	Shrub
Cephalanthus occidentalis	Buttonbush	1.0	(77.3, 0.071)	Shrub
		1.5	(61, 0.072)	
Commelina diffusa	Dayflower	2.0	(50.6, 0.034)	Herb
		3.0	(67.3, 0.015)	
Hydrocotyle verticillata	Pennywort	3.0	(43.7, 0.068)	Herb
Ilex cassine	Dahoon Holly	1.5	(55, 0.095)	Tree
Liquidambar styraciflua	Sweetgum	1.5	(60, 0.092)	Tree
Ludwigia peruviana	Water-Primrose	3.5	(54.9, 0.087)	Shrub
Nyssa sylvatica var. *biflora*	Swamp Tupelo	1.5	(60, 0.092)	Tree
Panicum rigidulum	Red-Top Panicum	3.0	(47.4, 0.031)	Graminoid
Prunus caroliniana	Carolina Laurelcherry	2.5	(27.3, 0.092)	Tree
Rhynchospora miliacea	Millet Beakrush	3.0	(43.7, 0.072)	Graminoid
Sambucus canadensis	Elderberry	2.5	(27.3, 0.09)	Shrub
Smilax auriculata	Earleaf Greenbrier	3.5	(72.4, 0.086)	Vine
Smilax tamnoides	Bristly Greenbrier	3.0	(37.7, 0.051)	Vine
Toxicodendron radicans	Eastern Poison Ivy	1.0	(77.3, 0.07)	Vine
		1.5	(61, 0.068)	
		2.0	(57.3, 0.069)	
Viburnum obovatum	Walter viburnum	3.5	(58.3, 0.052)	Shrub

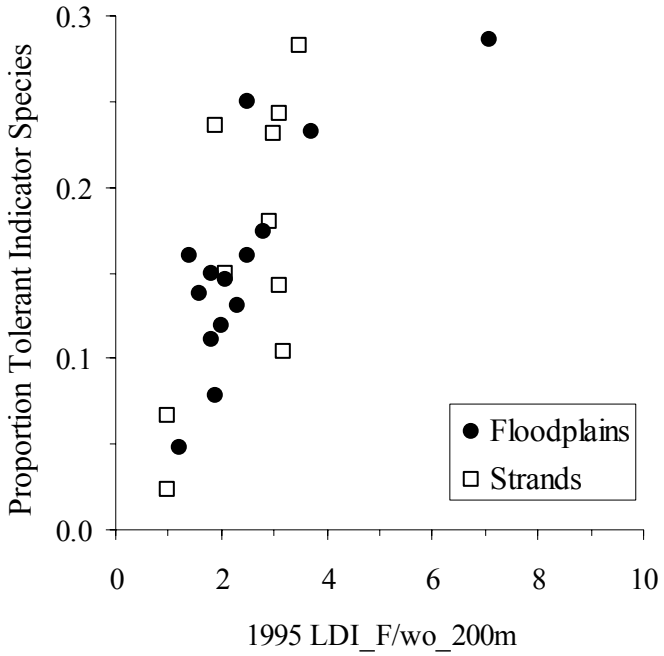


Figure 3-6. The proportion of tolerant indicator species at wetlands increased with increasing development intensity (LDI).

bush), *Rhododendron viscosum* (swamp azalea), *Vaccinium arboreum* (sparkleberry), and *Vaccinium corymbosum* (highbush blueberry). The two sensitive indicator graminoid species were *Cladium jamaicense* (saw-grass) and *Panicum hemitomon* (maidencane).

Floristic Quality Assessment Index metric

Wetland FQAI scores decreased with increasing 1995 LDI_F/wo (Figure 3-8). Of the 283 species identified in the flowing water wetlands, 17 were assigned coefficient of conservatism (CC) scores of zero, and 11 additional species received scores less than 1.0. Of the species receiving a zero CC score, nine (53%) were listed as Category I invasive exotics, and two (12%) were listed as Category II invasive exotics (EPPC 2003). [The rankings of Category I or II invasive exotics are from the Florida Exotic Pest Plant Council (EPPC), which focuses on identifying exotic pest species. Category I species include those invasive exotic species considered responsible for changes to native plant communities through the displacement of natives, changes in community structure or ecological functions, and hybridizing with natives, based on documented ecological damage (EPPC 2003). Category II species have not yet altered native plant communities, but have increased in abundance or frequency and may become ranked as Category I with confirmed ecological damage (EPPC 2003).]

The species with the highest CC score was *Pieris phyllyreifolia* (climbing fetter-bush) (9.5), followed by *Panicum abscissum* (cut-throat grass) (9.22), *Taxodium ascendens* (pond cypress) (8.8), *Asplenium heterochroum* (bicolored spleenwort) (CC = 8.5), and *Pinckneya bracteata* (fever-tree) (8.3). A complete list of CC scores is available in Appendix B. Wetland FQAI scores were significantly correlated with LDI

Table 3-11. Sensitive indicator species for forested strand and floodplain wetlands.

Botanical Name	Common Name	LDI Break	(Indicator Value, p-value)	Growth Form
Agarista populifolia	Florida Hobble-Bush	2.5	(30.8, 0.092)	Shrub
Centella asiatica	Coinwort	2.5	(38.6, 0.082)	Herb
Cladium jamaicense	Saw-Grass	1.5	(45.5, 0.059)	Graminoid
Eupatorium capillifolium	Dog Fennel	1.0	(88, 0.034)	Herb
Fraxinus caroliniana	Carolina Ash	2.5	(38.6, 0.064)	Tree
Hypericum hypericoides	St. Andrew's Cross	2.5	(30.8, 0.095)	Shrub
Ilex coriacea	Bay-Gall Holly	2.5	(38.5, 0.037)	Shrub
Lyonia lucida	Fetter-Bush	2.5	(53.6, 0.017)	Shrub
		3.0	(52.9, 0.054)	
Panicum hemitomon	Maidencane	2.0	(38.6, 0.042)	Graminoid
Persea palustris	Swamp Bay	2.5	(61.2, 0.006)	Tree
Phlebodium aureum	Golden Polypody	1.0	(95.7, 0.008)	Fern
		1.5	(45.5, 0.065)	
Pinus elliottii	Slash Pine	1.0	(95.7, 0.008)	Tree
		1.5	(45.5, 0.065)	
Rhododendron viscosum	Swamp Azalea	2.5	(38.6, 0.081)	Shrub
Saururus cernuus	Lizard's Tail	3.0	(64.7, 0.01)	Herb
Vaccinium arboreum	Sparkleberry	2.5	(30.8, 0.098)	Shrub
Vaccinium corymbosum	Highbush Blueberry	3.0	(52.9, 0.038)	Shrub

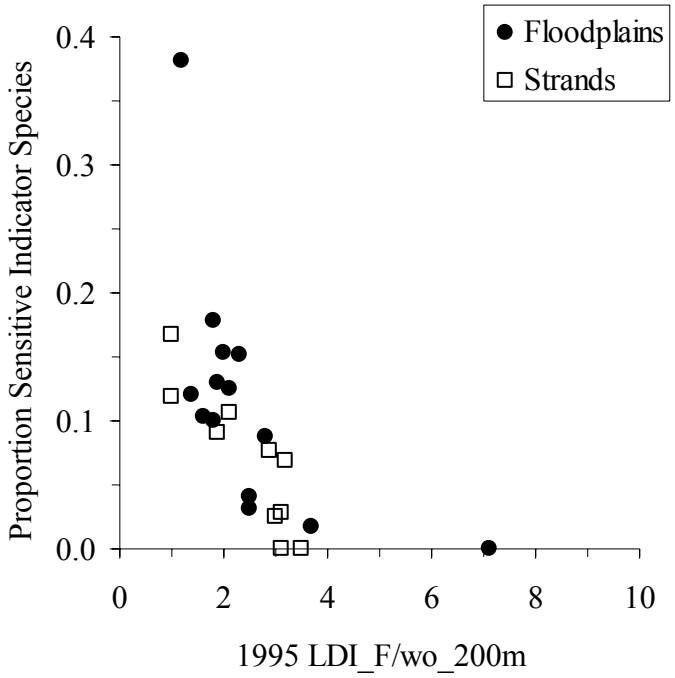


Figure 3-7. The proportion sensitive indicator species at wetlands decreased with increasing development intensity (LDI).

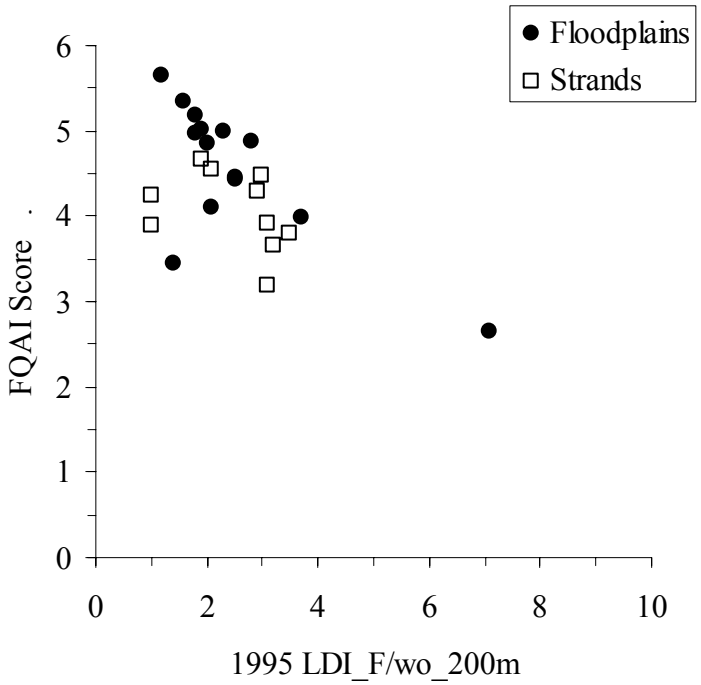


Figure 3-8. FQAI scores decreased with increasing landscape development intensity.

gradient (|r| = 0.50; p = 0.012; Table 3-8). When wetlands were divided into two groups based on 1995 LDI_F/wo_200m (LDI < 2.0 and LDI ≥ 2.0), there was a significant difference between FQAI scores (U = 34; p < 0.05) (Table 3-9).

The range of wetland FQAI scores was 3.0, though the scale of species CC scores ranged from 0.0-9.5. The wetland with the highest FQAI was FF2 (5.6), a reference floodplain forest in the north (Lane 2000) or peninsula stream (Griffith et al. 1994) ecoregions. The wetland receiving the lowest FQAI score was FF10 (2.7), an urban floodplain forest in southwest Florida in the south wetland region (Lane 2000) or Everglades bioregion (Griffith et al. 1994). Three wetlands in the low LDI group with low FQAI scores included FS7 (4.3), FS10 (3.9), and FF6 (3.4). Sixty-seven percent of the wetlands in the low LDI group had an FQAI score greater than 4.5; whereas 73% of wetlands in the high LDI group had an FQAI score less than 4.5.

Exotic species metric

The proportion of exotic species at a wetland was significantly correlated with the gradient of development intensity in the surrounding landscape (|r| = 0.54, p = 0.006; Table 3-8). Figure 3-9 shows that the proportion of exotic species increased with increasing LDI. The south wetland region (Lane 2000; or Everglades bioregion from Griffith et al. 1994) hosted the wetland with the greatest proportion exotic species, FF10 (0.29), which also received the lowest FQAI score (2.7). The wetland with the second highest proportion exotic species was FS3 (0.18), an agricultural strand surrounded by cattle pasture. Six floodplain wetlands had no exotic species present, including two reference wetlands (FF2 and FF7), two agricultural wetlands (FF4 and FF11), and two urban wetlands (FF5 and FF9).

The proportion of exotic species at wetlands was significantly different between low and high LDI groups (W = 109, p = 0.022; Table 3-9). Table 3-12 lists the 35 exotic species encountered throughout the forested strand and floodplain wetlands. The most common exotic species was the herbaceous species *Commelina diffusa* (dayflower) found at 10 (42%) of the 24 forested wetlands, which was also categorized as a tolerant indicator species (Table 3-10). Sixty-nine percent of the exotic species (n=24) were found at only one forested wetland. Thirty-seven percent of the species (n=13) were listed as Category I invasive exotic species, and 14% (n=5) were listed as Category II invasive exotic species (EPPC 2003). Forty-three percent of the exotic species were herbs, 20% vines, 14% shrubs, 11% trees, 9% graminoids, and 3% ferns.

Native perennial species metric

Of the 283 macrophyte species identified, 234 (83%) were classified as native perennials. Figure 3-10 shows that the proportion of native perennial species at a wetland decreased with increasing development intensity. The native perennial species metric was significantly correlated with LDI (Spearman |r| = 0.56, p = 0.004; Table 3-8); and there was a significant difference between the proportion native perennial species at low and high LDI group wetlands (W = 34, p = 0.036; Table 3-9). FF10 had the lowest proportion native perennial species (0.60). Four wetlands hosted entirely native perennial species, including one reference wetland (FF2), two agricultural wetlands (FF4 and FF11), and one urban wetland (FF9).

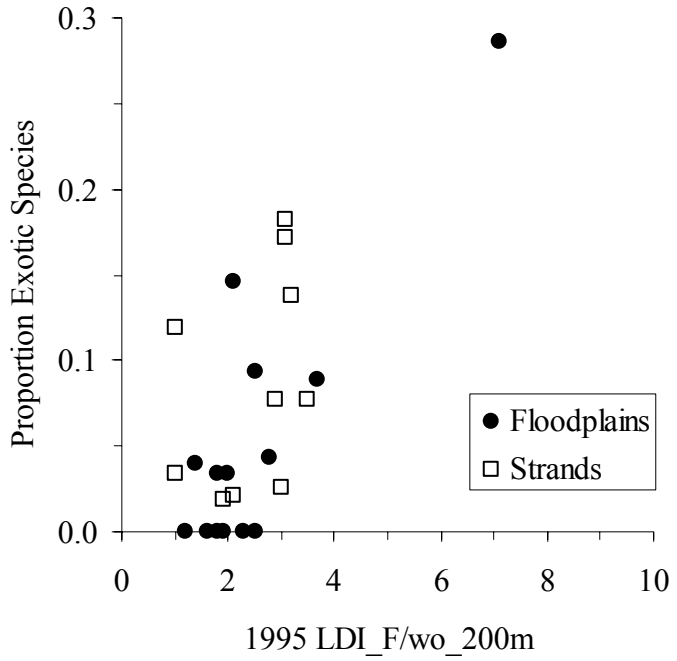


Figure 3-9. The proportion of exotic species at a wetland increased with increasing development intensity.

Florida Wetland Condition Index

Five metrics were included in the preliminary FWCI for forested strand and floodplain wetlands, including proportion tolerant indicator species, proportion sensitive indicator species, FQAI score, proportion exotic species, and proportion native perennial species. Metrics were natural log transformed to improve normality. Metric scoring was done on a continuous scale spreading between the 5th to 95th percentiles of the values for the sampled wetlands. Scores for each metric were then added together to create the preliminary forested strand and floodplain FWCI with a scale of 0-50, with 50 representing the reference condition of high biological integrity. Appendix D provides metric scoring criteria.

Figure 3-11 shows that FWCI scores decreased with increasing development intensity. Correlations between macrophyte metrics and FWCI with LDI were significant ($p < 0.05$) for all of the metrics and the FWCI ($|r| = 0.75$, $p < 0.001$) (Table 3-8). One wetland, FF2 a reference forested floodplain wetland, received a perfect 50 on the FWCI scale, also receiving a perfect 10 score for all five metrics. One wetland scored the lowest potential score of zero, FF10 which was an urban forested floodplain wetland in the south wetland region (Lane 2000) or Everglades bioregion (Griffith et al. 1994). All of the wetlands in the low LDI group scored above the midpoint of 25 on the FWCI scale; while 64% of wetlands in the high LDI group scored below 25.

Table 3-12. Exotic species identified at 24 forested strand and floodplain wetlands.

Botanical Name	Common Name	EPPC	Growth Form
Abrus precatorius	Rosary Pea	I	Vine
Alternanthera philoxeroides	Alligator Weed	II	Herb
Alternanthera sessilis	Sessile Alligator Weed		Herb
Begonia cucullata	Wax Begonia	II	Herb
Cinnamomum camphora	Camphor Tree	I	Tree
Colocasia esculenta	Elephant's Ear	I	Herb
Commelina communis	Asiatic Dayflower		Herb
Commelina diffusa	Dayflower		Herb
Cuphea carthagenensis	Columbia waxweed		Herb
Cynodon dactylon	Bermudagrass		Graminoid
Cyperus difformis	Variable Flatsedge		Graminoid
Emilia fosbergii	Florida Tasselflower		Herb
Eugenia uniflora	Surinam Cherry	I	Shrub
Koelreuteria elegans	Flamegold	II	Tree
Ligustrum sinense	Chinese Privet	I	Shrub
Lonicera japonica	Japanese Honeysuckle	I	Vine
Ludwigia peruviana	Water-Primrose		Shrub
Lygodium japonicum	Japanese Climbing Fern	I	Vine
Lygodium microphyllum	Small-leaf Climbing Fern	I	Vine
Melaleuca quinquenervia	Punk Tree; Melaleuca	I	Tree
Merremia dissecta	Alamo Vine		Vine
Mimosa pigra	Black Mimosa	I	Shrub
Momordica charantia	Balsampear		Vine
Oeceoclades maculata	Monk Orchid		Herb
Paspalum notatum	Bahiagrass		Graminoid
Phyllanthus urinaria	Water Leafflower		Herb
Sapium sebiferum	Chinese Tallowtree	I	Tree
Schefflera actinophylla	Australian Umbrella Tree	I	Tree
Schinus terebinthifolius	Brazilian Pepper	I	Shrub
Thelypteris dentata	Downy Maiden Fern		Fern
Trifolium repens	White Clover		Herb
Urena lobata	Caesarweed	II	Herb
Wisteria sinensis	Chinese Wisteria	II	Vine
Xyris jupicai	Richard's Yellow-Eyed-Grass		Herb
Youngia japonica	Oriental False Hawksbeard		Herb

Exotic Pest Plant Council (EEPC) categories from EEPC (2003).

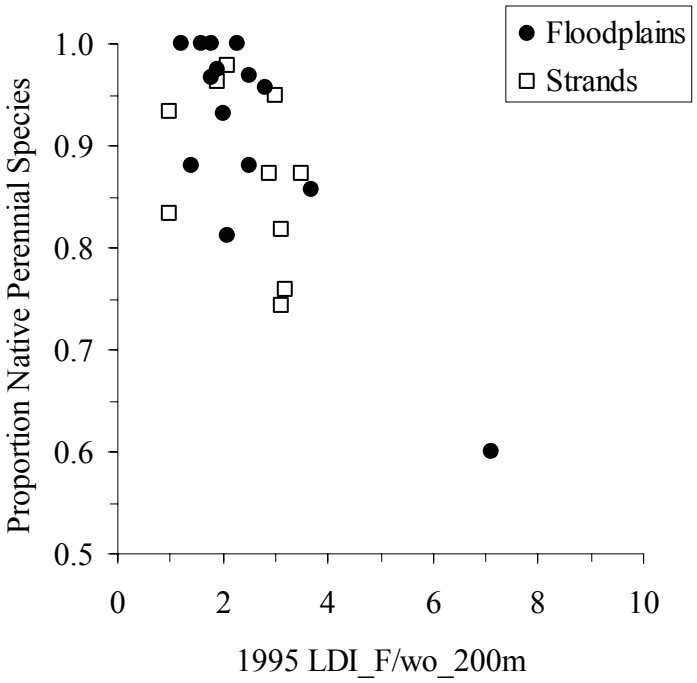


Figure 3-10. The proportion of native perennial species decreased with increasing development intensity (LDI).

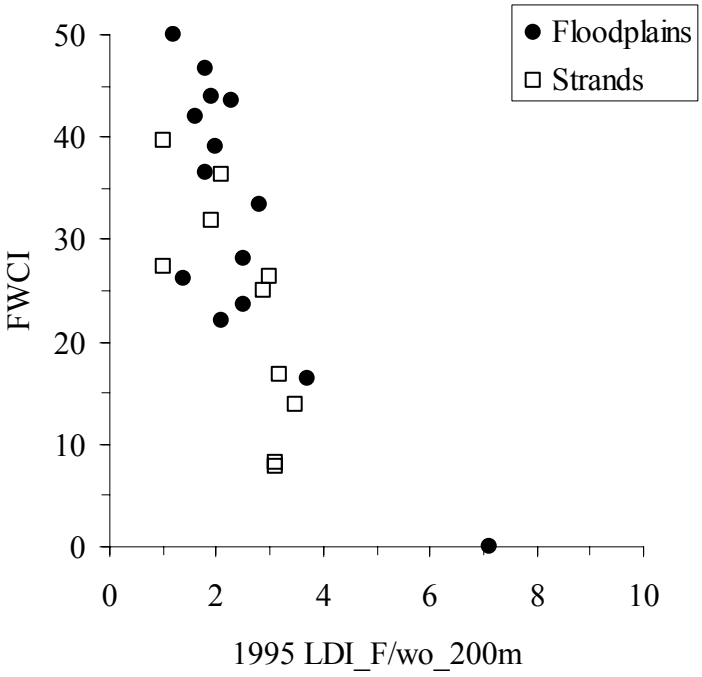


Figure 3-11. Forested Wetland Condition Index (FWCI) scores decreased with increasing development intensity (LDI).

Cluster Analysis

Cluster analysis determined wetland groupings based on macrophyte community composition using a species by site presence/absence matrix. Based on the lowest average p-value for all species from the randomized Monte Carlo tests used in the ISA analysis, the most ecologically meaningful cluster for dendrogram pruning was at cluster step 5 (Figure 3-12) with the lowest average species p-value of 0.27. The highest number of significant ($p < 0.05$) indicator species was found at cluster step 3, which had 37 significant indicator species (Figure 3-13). Cluster steps 4 and 5 had the second highest number of significant indicator species at 31. Exploration of the sample sites assigned to the groups at cluster step 3 suggested an association between group membership and spatial distribution of the wetlands throughout Florida. [Note: ISA used to determine the most ecologically meaningful clusters for dendrogram pruning is separate from ISA used in determining tolerant and sensitive indicator species. For ISA used to determine meaningful clusters for dendrogram pruning, group membership was assigned based on the wetland groupings established through cluster analysis. For ISA used to determine sensitive and tolerant indicator species, group membership was assigned based on the calculated LDI score for each wetland.] The groups defined through the agglomerative cluster analysis at cluster step 5 were roughly defined by wetland regions (Lane 2000), bioregions (Griffith et al. 1994), and *a priori* land use categories, including:

Cluster 1: FF2, a north wetland region, peninsula bioregion, reference floodplain;

Cluster 2: Nine floodplains, including two in the panhandle, six in the north, and one in the central wetland regions; one in the panhandle, five in the northeast, and three in the peninsula bioregions; and four agricultural and five urban wetlands;

Cluster 3: Five strand and two floodplain wetlands; one in the north and six in the central wetland regions; all seven in the peninsula bioregion; four reference, one agricultural, and two urban wetlands;

Cluster 4: Three strand and two floodplains, including one in the north and four in the south wetland regions; four in the peninsula and one in the Everglades bioregions; three reference and two urban wetlands;

Cluster 5: Two floodplain wetlands with one in the north and one in the central wetland regions; both in the peninsula bioregion; and both agricultural wetlands.

Wetland groups based on the third cluster step combined the aforementioned groups from the cluster step 5, so that:

Cluster 1: Combines Clusters 1 & 2

Cluster 2: Composed of Cluster 3

Cluster 3: Combines Clusters 4 & 5

It appear that the initial grouping for cluster step three was based on spatial location and that further cluster steps separate out wetlands based on impairment. Figure 3-14 shows that based on wetland groups from cluster step 3, FWCI scores for wetlands in Cluster 1 (37.5 ± 10.5) were significantly different from wetlands in Cluster 2 (24.4 ± 10.0) and Cluster 3 (19.8 ± 13.1) ($p < 0.05$). Wetland FWCI scores for Cluster 2 and Cluster 3 were not significantly different from one another. Regionalization was apparent in the wetland groupings of Cluster 3 (at cluster step 3); as Cluster 3 was comprised of the all of

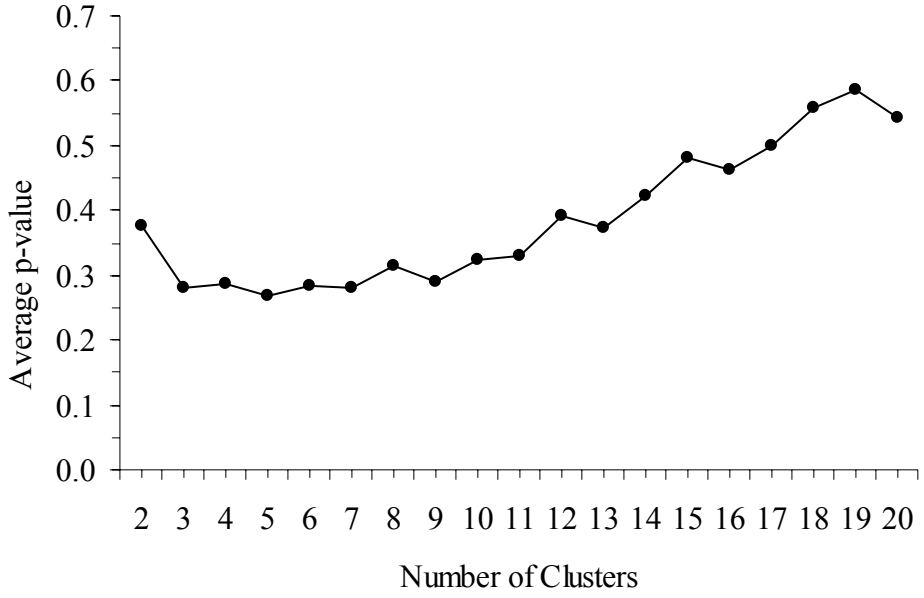


Figure 3-12. Change in average species p-value from the randomized Monte Carlo tests at each step in clustering. The minimum average p-value (0.27) was found at cluster step 5.

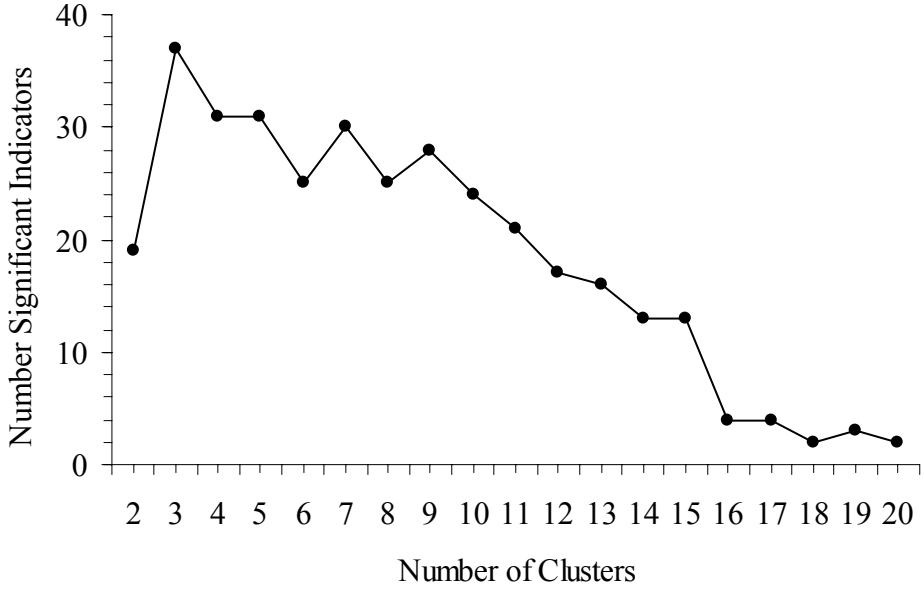


Figure 3-13. Change in the number of significant indicator species from the indicator species analysis performed at each step in clustering. The maximum number of significant indicator species (37) was found at cluster step 3 ($p < 0.05$), followed by 31 significant indicator species at cluster steps 4 and 5.

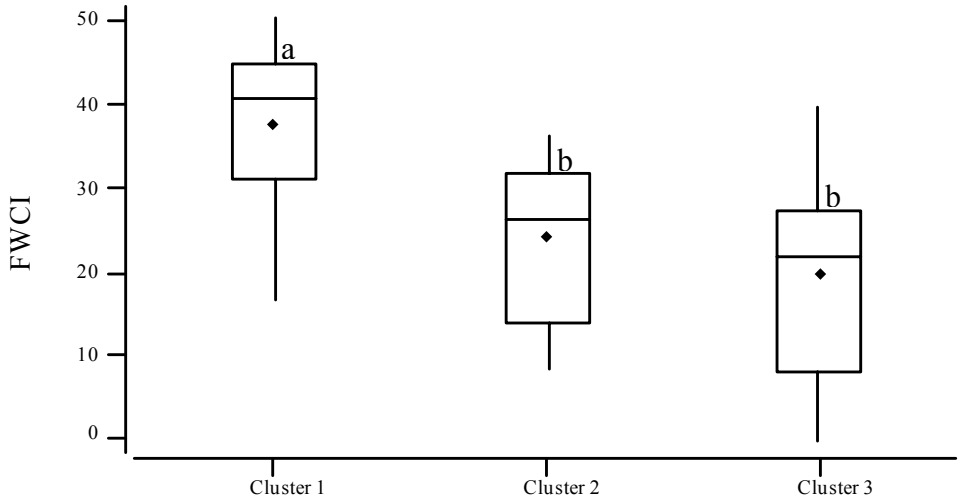


Figure 3-14. FWCI scores for three wetland clusters based on macrophyte community composition. Boxes represent 25th and 75th quartiles, lines represent median FWCI scores per cluster, dots represent the mean, and vertical lines represent the range. Clusters with similar letters were not significantly different ($p<0.05$).

the south wetland region wetlands (Lane 2000), including the only wetland sampled in the Everglades bioregion (Griffith et al. 1994). Table 3-13 provides means and standard deviations for cluster FWCI and LDI_F/wo_200m scores. LDI_F/wo_200m scores were not significantly different among clusters.

Landscape Development Intensity Index and the Florida Wetland Condition Index

Nearly all of the correlation (110 of 120, or 92%) between the five macrophyte metrics and the FWCI with the 20 variations of LDI were significantly correlated at the flexible $p < 0.10$ level (Table 3-14). Additionally, nearly half of the comparisons (58 of 120, or 48.3%) were significantly correlated at the strictest significance level ($p < 0.01$). Three of the LDI calculations (1995 LDI_F/w_200m, 2000 LDI_F/w_200m, and 2000 LDI_F/wo_200m) were significant at the strictest significance level ($p < 0.01$) for all five metrics and the FWCI, which should not be surprising given that metrics were selected for inclusion in the FWCI based on correlations with the 1995 LDI_F/wo_200m (using Spearman correlation with LDI, visually distinguishable and ecologically meaningful pattern when graphed with LDI, and differentiation among LDI groups with the Mann Whitney U-test).

The proportion tolerant indicator species metric was significantly correlated with 1995 and 2000 LDI_F (wetland feature scale) calculations ($p < 0.01$). However, the proportion tolerant indicator species metric was not significantly correlated at the strictest significance level ($p < 0.01$) for any of the 1995 or 2000 LDI_T (transect scale)

Table 3-13. FWCI scores and LDI values for wetland clusters (at the third cluster step) based on macrophyte community composition.

Wetland Clusters	FWCI	LDI_F/wo_200m
1	37.5 ± 10.5^{a}	2.2 ± 0.7^{a}
2	24.4 ± 10.0^{b}	2.5 ± 0.7^{a}
3	19.8 ± 13.1^{b}	2.9 ± 2.1^{a}

Clusters with similar letters within columns were not significantly different ($p < 0.05$).
LDI_F/wo_200m represents the LDI calculated at the feature scale excluding the wetland area within a 200 m buffer.

calculations. The proportion sensitive indicator species metric was more strongly correlated with the LDI calculations at the transect and feature scales (11 of 12 correlations with $|r| \geq 0.63$, $p < 0.01$, remaining one at $|r| = 0.57$, $p < 0.05$) than the watershed scales (1995 and 2000 LDI_WS_DW_exp at $p < 0.01$; 2000 LDI_WS_DW_lin at $p < 0.05$; 2000 LDI_WS_ED/wo not significant; remaining four LDI_WS correlations at $p < 0.10$).

The FQAI score metric was not significantly correlated with three of the 1995 transect level LDI calculations, but was strongly ($|r| > 0.66$) and significantly ($p < 0.01$) correlated with all of the 1995 and 2000 watershed scale LDI calculations. Watershed scale LDI calculations were completed for floodplain forested wetlands only, leaving speculation as to whether these results suggest that FQAI scoring reflect landscape level anthropogenic activity (e.g. exotic species with low CC scores entering a system due to anthropogenic activities) or whether FQAI scoring was biased towards larger flowing water systems such as floodplain forests. The proportion exotic species and proportion native perennial species metrics were more strongly correlated with the feature scale LDI calculations than transect or watershed scales. In fact, three watershed 1995 LDI calculations were not significant with the proportion exotic species or the proportion native perennial species metrics, including 1995 LDI_WS_ED/w, 1995 LDI_WS_ED/wo, and 1995 LDI_WS_DW_lin.

The strongest correlation between the FWCI and LDI was with the 2000 LDI_WS_DW_exp ($|r| = 0.95$; $p < 0.01$), though the correlation with 2000 LDI_F/wo_200m ($|r| = 0.94$; $p < 0.01$) was also remarkably strong. The two correlations with the highest Spearmans correlation coefficients were between the proportion tolerant indicator species metric and the 2000 LDI_WS_DW_exp and between the FQAI score metric and the 2000 LDI_WS_DW_lin ($|r| = 0.96$; $p < 0.01$).

Table 3-14. Correlations of metrics and FWCI scores with 20 variations of the LDI index. Differences in LDI calculations include 1995 or 2000 land use; transect (T), feature (F), or watershed (WS) scale buffers; including (/w) or excluding (/wo) the wetland area; 100 (100m) or 200 meter (200m) buffers; equal distance (ED) or distance weighted (DW); and linear (lin) or exponential (exp) weighting. Column headings refer to the five metrics and the FWCI: proportion tolerant indicator species (TOL), proportion sensitive indicator species (SEN), Floristic Quality Assessment Index (FQAI), proportion exotic species (EX), and proportion native perennial species (NP).

LDI	n =	TOL		SEN		FQAI		EX		NP		FWCI	
1995 LDI_T/w_100m	24	0.44	**	-0.68	*	-0.39	***	0.51	**	-0.48	**	-0.58	*
1995 LDI_T/wo_100m	24	0.41	**	-0.65	*	ns		0.46	**	-0.43	**	-0.52	*
1995 LDI_T/w_200m	24	0.38	***	-0.63	*	ns		0.49	**	-0.44	**	-0.49	**
1995 LDI_T/wo_200m	24	0.45	**	-0.69	*	ns		0.51	**	-0.44	**	-0.52	**
1995 LDI_F/w_200m	24	0.72	*	-0.90	*	-0.57	*	0.61	*	-0.63	*	-0.79	*
1995 LDI_F/wo_200m	24	0.66	*	-0.83	*	-0.50	**	0.54	*	-0.55	*	-0.75	*
1995 LDI_WS_ED/w	14	0.60	**	-0.49	***	-0.71	*	ns		-0.50	***	-0.57	**
1995 LDI_WS_ED/wo	14	0.59	**	-0.49	***	-0.67	*	ns		ns		-0.53	**
1995 LDI_WS_DW_lin	14	0.58	**	-0.49	***	-0.69	*	ns		ns		-0.53	***
1995 LDI_WS_DW_exp	14	0.80	*	-0.72	*	-0.75	*	0.60	**	-0.65	**	-0.70	*
2000 LDI_T/w_100m	24	0.51	**	-0.72	*	-0.49	**	0.48	**	-0.54	*	-0.63	*
2000 LDI_T/wo_100m	24	0.51	**	-0.72	*	-0.42	**	0.44	**	-0.49	**	-0.59	*
2000 LDI_T/w_200m	13	0.54	***	-0.69	*	-0.74	*	0.83	*	-0.85	*	-0.81	*
2000 LDI_T/wo_200m	13	ns		-0.57	**	-0.66	**	0.73	*	-0.77	*	-0.72	*
2000 LDI_F/w_200m	13	0.78	*	-0.93	*	-0.88	*	0.79	*	-0.85	*	-0.91	*
2000 LDI_F/wo_200m	13	0.82	*	-0.92	*	-0.91	*	0.82	*	-0.84	*	-0.94	*
2000 LDI_WS_ED/w	8	0.73	**	-0.68	***	-0.95	*	0.77	**	-0.83	**	-0.88	*
2000 LDI_WS_ED/wo	8	0.65	***	ns		-0.86	*	0.73	**	-0.69	***	-0.78	**
2000 LDI_WS_DW_lin	8	0.83	**	-0.80	**	-0.96	*	0.77	**	-0.81	**	-0.92	*
2000 LDI_WS_DW_exp	8	0.96	*	-0.88	*	-0.88	*	0.83	**	-0.78	**	-0.95	*

*p<0.01 **p<0.05 ***p<0.10 ns – not significant

Human Disturbance Gradient and Stream Condition Index

Thirteen of the forested floodplain wetlands had Human Disturbance Gradient (HDG) and Stream Condition Index (SCI) scores for sampling events taken within the channelized flow of the forested floodplain wetlands (Table 3-15). Sample dates for the FWCI and LDI were during 2003 (Table 2-1); sample dates for the HDG and SCI ranged from 1995-2000 (Table 3-15). Table 3-15 provides storet identification numbers, HDG and SCI sample dates, *a priori* land use categories, bioregions (Griffith et al. 1994), wetland regions (Lane 2000), FWCI, LDI_F/wo_200m scores, HDG scores, and SCI scores for the 13 forested floodplain wetlands. Data for the most recent and complete sampling event for each wetland, as presented in Table 3-15, was selected for correlation purposes.

The HDG was significantly correlated with the SCI ($|r| = 0.58$, $p<0.05$), LDI_F/wo_200m ($|r| = 0.66$, $p<0.05$), and FWCI ($|r| = 0.74$, $p<0.01$) (Table 3-16). However, the SCI was not significantly correlated with either the LDI_F/wo_200m or the FWCI. The correlation between the LDI_F/wo_200m and FWCI ($|r| = 0.68$, $p<0.05$) (Table 3-16) for the 13 forested floodplain wetlands (only those with HDG and SCI scores), was weaker than the correlation between them for the entire data set of 24 forested flowing water wetlands ($|r| = 0.75$, $p<0.001$) (Table 3-8).

Table 3-15. Forested Wetland Condition Index (FWCI), Landscape Development Intensity Index (LDI_F/wo_200m), Human Disturbance Gradient (HDG), and Stream Condition Index (SCI) data for 13 forested floodplain wetlands.

Site	STORET ID	HDG/SCI Date	Land Use	Bioregion	Wetland Region	FWCI	LDI_F/wo_200m	HDG	SCI
FF1	20020404	8/15/2000	U	Peninsula	North	39.1	2.0	0	80
FF2	20010454	7/19/1999	R	Peninsula	North	50.0	1.2	0	70
FF3	27010580	8/22/2000	U	Peninsula	North	16.4	3.7	2	20
FF4	20010455	2/23/1999	A	Peninsula	Central	43.6	2.3	0	75
FF5	19010099	2/1/2000	U	Northeast	North	44.0	1.9	0	55
FF6	26011019	3/8/1999	R	Peninsula	Central	26.2	1.4	0	80
FF7	26011020	7/20/1999	R	Peninsula	Central	28.2	2.5	0	90
FF8	19020027	7/15/1996	U	Northeast	North	23.6	2.5	1	70
FF9	20030246	4/17/1997	U	Northeast	North	46.6	1.8	0	95
FF10	28020234	10/9/2000	U	Everglades	South	0.0	7.1	4	50
FF11	21010032	1/17/1995	A	Northeast	Panhandle	42.0	1.6	0	25
FF12	19010072	3/22/1999	A	Northeast	North	33.5	2.8	0	85
FF14	32010024	7/14/1998	A	Panhandle	Panhandle	36.5	1.8	0	80

STORET ID refers to Florida Department of Environmental Protection (FDEP) database ID.
HDG/SCI Date refers to original sample date corresponding to HDG and SCI data. FWCI and LDI were calculated for the 2003 site visit.
Land Use - refers to *a priori* land use category (R-reference, A-agricultural, U-urban)
Bioregion from Griffith et al. (1994)
Wetland Region from Lane (2000)
LDI_F/wo_200m refers to the LDI calculated at the feature scale, excluding the wetland area, within a 200 m buffer.

Table 3-16. Correlations among four measures of ecosystem condition or anthropogenic activity, including the Human Disturbance Gradient (HDG), Stream Condition Index (SCI), Landscape Development Intensity Index (1995 LDI_F/wo_200m), and the Florida Wetland Condition Index (FWCI) for freshwater forested floodplain wetlands. Values are Spearmans rank correlation coefficients.

	HDG	SCI	LDI_F
SCI	-0.58**		
LDI_F	0.66**	ns	
FWCI	-0.74*	ns	-0.68**

*p<0.01, **p<0.05, ns=not significant

CHAPTER 4
DISCUSSION

The primary objective of this research was to develop a preliminary Florida Wetland Condition Index (FWCI) for forested strand and floodplain wetlands. Wetland study sites were sought in various *a priori* designated land use categories that included undeveloped, agricultural, and urban land uses. The preliminary FWCI provides a quantitative measure of the biological integrity of forested strand and floodplain wetlands in Florida. Comprised of five metrics, the preliminary FWCI was developed based on the community composition of the macrophyte species assemblages (Table 4-1). Metrics were selected for inclusion in the FWCI based on the correlation (nonparametric Spearman correlation coefficient) of each metric with the Landscape Development Intensity (LDI), an independent measure of anthropogenic activity in the landscape calculated for each wetland (Brown and Vivas 2005); based on a metrics visually distinguishable correlation with LDI in a scatter plot; and based on a statistical difference of metric values between low and high LDI groups (Mann-Whitney U-test). The FWCI was composed of individual metrics, which were scaled and added together, creating the preliminary forested strand and floodplain wetland FWCI (0-50 scale), with the highest score of 50 reflecting the highest biological integrity and the lowest score of zero reflecting a lack of biological integrity or no similarity to the reference wetland condition.

The contribution of this research to our understanding of changes in the macrophyte community composition of forested strand and floodplain wetlands in relation to differing anthropogenic activities in the surrounding landscape can be summarized in five main points:

1. Five macrophyte based metrics including proportion tolerant indicator species, proportion sensitive indicator species, Floristic Quality Assessment Index (FQAI) score, proportion exotic species, and proportion native perennial species, were useful biological indicators for defining biological integrity for forested strand and floodplain wetland vegetation;
2. Vegetation richness, evenness, and diversity were not sensitive to *a priori* land use categories or development intensities in the surrounding landscape for forested strand and floodplain wetlands;
3. The Landscape Development Intensity (LDI) index was a useful tool correlating with the measured biological condition of vegetation for forested strand and floodplain wetlands;
4. Regional species lists for metrics would enhance the forested strand and floodplain Florida Wetland Condition Index (FWCI);
5. An FWCI with a set of core metrics could be developed for Florida freshwater wetlands, which includes separate species lists for indicator species by wetland type and ecoregions and separate Floristic Quality Assessment Index (FQAI) scores for species by wetland type.

Table 4-1. The five metrics of the preliminary Florida Wetland Condition Index for freshwater forested strand and floodplain wetlands based on the macrophyte species assemblage.

FWCI Metrics
1. Proportion Tolerant Indicator Species
2. Proportion Sensitive Indicator Species
3. Floristic Quality Assessment Index Score
4. Proportion Exotic Species
5. Proportion Native Perennial Species

Describing Biological Integrity

Biological indicators were useful in determining the biological integrity of freshwater forested strand and floodplain wetlands. For the purposes of this study, biological integrity has been defined quantitatively with the FWCI. The FWCI incorporated five metrics from the macrophyte species assemblages (Table 4-1). Strong correlations between the FWCI and the intensity of development in the surrounding landscape (based on the use of nonrenewable energy and calculated with the LDI) suggest that changes in macrophyte community composition quantified as metrics were captured by the LDI. It has been proposed that organisms respond to environmental gradients by colonizing a range of feasible conditions beyond which the organisms fail to persist (ter Braak 1987). By selecting species that occur throughout the range of measurable environmental parameters, the FWCI defined and detected deviations from the condition of reference wetlands based on macrophyte community composition. Each of the FWCI metrics addressed some disparity from the assumed range of feasible conditions. The proportion sensitive indicator species metric showed the strongest correlation with LDI, suggesting that the presence of a suite of taxa characteristic of wetlands with high biological integrity may be the most effective means of identifying changes in macrophyte community composition in freshwater forested strand and floodplain wetlands associated with changes in anthropogenic activities.

Richness, Evenness, and Diversity

Measures of richness, evenness, and diversity of the macrophyte assemblage were not sensitive to differences in land use or development intensity in the surrounding landscape within the forested strand and floodplain wetlands. Perhaps due to a limited sample size and high variability inherent in the landscape (e.g. regional differences, land use differences, etc.) no statistical test on summary statistics produced statistically significant results. Nevertheless, wetlands in the high LDI group (LDI $\geq$ 2.0) had higher (though not statistically significant) mean species richness and diversity (Shannon diversity, Whittaker's beta diversity, and overall beta and gamma diversity). Species evenness and Simpson's diversity were remarkably similar among the sample wetlands; in effect no differences were detectable.

General ecological theory predicts a decrease in plant diversity resulting from an increase in anthropogenic assaults such as grazing (Blanch and Brock 1994; Grace and Jutila 1999) and nutrient enrichment (Bedford et al. 1999), though the forested strand and floodplain wetlands displayed the opposite trend. Mitsch and Gosselink (1993) report that freshwater forested wetlands have low species diversity as compared to other ecosystems, so perchance macrophyte species that entered wetlands in developed landscapes were merely taking advantage of available habitat and in fact increased the overall species diversity. However, Ewel (1990) notes that due to high topographic and soil variability, river swamps may be the most diverse type swamp in Florida. Clearly there is some uncertainty within the published literature as to anticipated and abnormal diversity for Florida forested strand and floodplain wetlands.

Many of the species entering wetlands in developed landscapes were categorized as exotic species, and the increased incidence of exotic species has long been associated with disturbed ecosystems (Galatowitsch 1999b; Cronk and Fennessy 2001). An increase in the frequency of exotic species has been attributed to drainage and hydrologic alterations (Hobbs and Heunneke 1992; David 1999; Galatowitsch et al. 1999b), increased human development (Cronk and Fennessy 2001), and ecosystem scale alterations such as clear-cut harvests (Devine 1998). Within the study wetlands, the proportion of exotic species increased with increasing development intensity in the surrounding landscape. It appears that the influx of exotic species added to rather than diminished the species richness and diversity within the freshwater forested strand and floodplain wetlands.

As such, the presence of exotic species alone may not be an ideal indicator of biological integrity. While many ecological theories have been established suggesting that the presence or occurrence of exotic species increases with anthropogenic disturbance (Cronk and Fennessy 2001; Galatowitsch 1999b), there was a inconsistent pattern of occurrence of exotic species in the forested strand and floodplain wetlands sampled. For example, six of the freshwater forested floodplain wetlands, including two reference, two agricultural, and two urban wetlands, hosted zero exotic species. That four floodplain wetlands in developed landscapes (including two agricultural and two urban wetlands) hosted no exotic species was somewhat contradictory to the theories of increased exotic species occurrence in disturbed ecosystems. However, for the complete dataset, the trend of an increase in the proportion of exotic species with increasing landscape development intensity held.

Some concerns arise considering the discrepancies on the exotic status of some species. For example, not all of the species listed by the Exotic Pest Plant Council (EPPC) as invasive exotics altering native plant communities (Category I) or invasive exotics increasing in frequency or abundance with the potential to alter native plant communities (Category II) (EPPC 2003) received an FQAI score of 0.0 corresponding to species that act as opportunistic invaders, including species that commonly occur in disturbed ecosystems. The disagreement on the status of a species as an invasive exotic translates into disagreement as to the meaning of a particular exotic species occurring within an ecosystem. As a case in point, two exotic species were included as tolerant indicator species including *Commelina diffusa* (dayflower) and *Ludwigia peruviana* (water-primrose). Neither of the two tolerant indicator species that are also exotic species was listed as an EPPC Category I or II species. In fact, none of the 13 Category I

invasive exotic species identified in the forested strand and floodplain wetlands was categorized as a tolerant indicator species.

The intent of this research was to assess the current biological integrity of Florida freshwater forested flowing water wetlands in order to develop a quantitative wetland condition index. As such the comparison was made against the current day reference standard of biological integrity apparent in the reference strand and floodplain wetlands sampled. While we may innately believe that reference wetlands host no exotic species, we found that five of nine wetlands surrounded by low development intensity (56%) hosted at least one exotic species. Increasingly, it may become apparent that even reference wetlands with the highest current standard of biological integrity may host some proportion of exotic species. This may be even more apparent in the southern half of the state, where drainage and development have altered nearly all of the Florida landscape. In fact, the south wetland region reference wetland (FS10) had the highest proportion exotic species of all wetlands in the low LDI group by nearly 10%. Consequently, establishing a baseline for the reference condition of biological integrity is crucial for the application of indices of biotic integrity. Consensus should be reached as to whether scientists proceed by establishing a present day baseline for future assessments or by maintaining and updating a moving baseline for future application of the FWCI. Implications for both methods are complex.

Measuring Anthropogenic Activity

The variable sensitivities of three different independently derived indices compared to the forested strand and floodplain FWCI, including the Landscape Development Intensity index (LDI; Lane et al. 2003; Brown and Vivas 2005), the Human Disturbance Gradient (HDG; Fore 2004), and the Stream Condition Index (SCI; Fore 2004), suggest that multiple measures of biological integrity may be more effective at describing ecosystem wide biological integrity than any single measure based on an individual species assemblage or surrounding land use activity. Measurements of anthropogenic activity such as the LDI and HDG seek to describe ecosystem biological integrity from the perspective of outside anthropogenic influences which act to alter an ecosystem. While the LDI was used as a remote based measure of human development intensity, the HDG used both remote and local conditions integrating four components ranging from the in-stream chemical water quality, habitat structure, and hydrologic alteration, to a remote landscape assessment (using a form of LDI). For the HDG each of the four components was given equal weighting in determining the influence of anthropogenic activities on the biological integrity of an ecosystem. The LDI and HDG were strongly correlated, suggesting that the LDI alone may capture the influence from human development intensity on ecosystem biological integrity without the need for additional sampling and sample processing associated with in-stream chemical water quality, habitat structure, and hydrologic alteration.

The FWCI was strongly correlated with both the HDG and the LDI, though the significance of correlation was slightly stronger with the HDG than the LDI. The caveat here was that only three of the thirteen floodplain wetlands had an HDG greater than zero, and because these scores were tested using the non-parametric Spearmans rank correlation coefficient test the strong correlations may simply be a factor of zero HDG

values. In fact, our findings of strong correlations of LDI and HDG must be interpreted with caution because so few of the wetlands received HDG values greater than zero (an HDG score of zero represents no detectable human induced disturbance). This was likely due to the fact that the HDG and SCI were calculated in the channelized water course, whereas the LDI and FWCI were calculated for the surrounding floodplain forest. Many of the streams with low SCI and high HDG scores (considered those with low biological integrity) were not sampled for the FWCI and LDI as no floodplain forest remained, with the banks of the channelized water course being either sodded and mowed or paved. Therefore we should cautiously interpret correlations and discrepancies between the SCI (no significant correlations with FWCI or LDI) and HDG with the FWCI and LDI because the range of impaired conditions used to develop the FWCI was much narrower than that used to develop the SCI.

The lack of correlations of SCI with both FWCI and LDI, suggest that in-stream macroinvertebrate based measures of biological condition and surrounding forested wetland macrophyte based measures of biological condition did not respond in a consistent manner to changes in anthropogenic activity. Using both the in-stream macroinvertebrate SCI biological assessment and the surrounding wetland macrophyte FWCI biological assessment methods may provide a more complete picture of the overall condition of a wetland and associated stream at a particular spatial location. While agreement in the ranking of the biological condition of study wetlands using the FWCI and SCI was anticipated, discrepancies among the ranking from the different assemblages may provide great insight into biological condition as different species assemblages respond to changes in anthropogenic activities and the associated changes in inflows (e.g. nutrient enrichment) over different time scales. Additionally, use of the forested strand and floodplain FWCI may lead to specific conclusions as to the biological condition of local or nearby anthropogenic activity, while use of the SCI may enhance understanding of larger watershed scale influences from anthropogenic activity (i.e. due to the convergence of surface water within the watershed associated with stream flow).

By correlating the community composition based FWCI and the landscape based LDI, we were better able to understand anthropogenic influence on biological integrity on forested wetland ecosystems. We found that the LDI index was a useful tool in approximating biological integrity. Its primary power was in the reproducible, objective, and quantitative methods employed to obtain a score based on the use on non-renewable energy in the surrounding landscape. A second strength of the LDI index was apparent in the practical application of a remote GIS based method of describing ecosystem condition as a starting point to identify potential areas for further biological sampling.

Regionalization of the Florida Wetland Condition Index

Pronounced differences in the local climate across Florida, such as differences in the amount of annual rainfall, seasonal maximum temperatures, and number of freeze days (Fernald and Purdum 1992; Lane 2000) and the broad latitudinal and longitudinal ranges of sample wetlands (26.3ºN -30.8ºN latitude, 80.1ºW-82.1ºW longitude), suggest differences would be found among macrophyte community composition in Florida wetlands. In fact, compositional differences of the macrophyte species assemblages were

found among Lane's (2000) Florida wetland regions. However, the study sample size limited the development of regional indicator species and metric scoring criteria.

Wetland clusters based on macrophyte community composition (species presence by site) roughly correlated with wetland spatial distribution throughout the state. The distribution of wetlands in the cluster groups suggests that wetlands located in the northern area of Florida may have higher biological integrity than wetlands in the central or southern peninsula given a statewide scoring approach of the preliminary forested strand and floodplain FWCI. Accordingly, most of the human development in Florida has occurred along the east and west coastal areas of peninsular Florida (Fernald and Purdum 1992), suggesting that while the reference wetlands selected in the south and central wetland regions were selected as the best possible examples of reference type conditions, they may be more affected by development in the surrounding landscape (such as compounded secondary effects) than their panhandle and north wetland region reference counterparts. While the ease and utility of a single statewide FWCI would seemingly prevail over regional indicator species and metric scoring criteria, the necessity of scoring each wetland region based on the best possible reference conditions cannot be overlooked (as suggested by Karr and Chu 1999).

Regionalization of biological indices has been suggested throughout the literature. One of the main premises behind indices of biological integrity is a comparison of "like to like" (Gerristen et al. 2000), that is, to reduce the noise in background variability in biological data, which could be accomplished through regionally based indices. The lower FWCI scores for wetlands in the central and southern peninsula may also be a factor of the smaller sample size for wetlands located in those wetland regions. Since the macrophyte community composition was found to be different between wetland regions, wetlands in the south and central wetland regions were underrepresented and therefore had less influence in the indicator species analysis and the metric scoring criteria. A larger sample size would improve uncertainty related to questions surrounding the biological integrity of wetlands in the south and central wetland regions.

Florida Wetland Condition Index Independent of Wetland Type

Recent works by Lane et al. (2003) and Reiss and Brown (2005) presented a five metric FWCI for isolated depressional herbaceous wetlands and a six metric FWCI for isolated depressional forested wetlands in Florida based on the community composition of the macrophyte assemblage. The depressional herbaceous FWCI was created based on a sample size of 75 freshwater marshes surrounded by reference (n=34) and agricultural (n=40) land uses throughout peninsular Florida. The depressional forested FWCI was created based on a sample size of 118 freshwater wetlands surrounded by reference (n=37), agricultural (n=40) and urban (n=41) land uses throughout the extent of Florida. The five metrics of the preliminary FWCI for forested strand and floodplain wetlands were nearly identical to the five metrics of the depressional herbaceous FWCI (with an adaptation from annual to perennial ratio from the depressional herbaceous FWCI changed to the proportion native perennial species for the depressional forested and forested strand and floodplain FWCIs) and similar to five of the depressional forested FWCI metrics.

The five macrophyte metrics included in the three wetland type FWCIs (depressional herbaceous, depressional forested, forested strand and floodplain) were tolerant and sensitive indicator species, FQAI score, exotic species, and native perennial species (annual to perennial ratio in the depressional herbaceous FWCI). Tolerant and sensitive indicator species lists were constructed independently for each wetland type. Of the 16 tolerant indicator species for flowing water wetlands, five also occurred as tolerant indicator species for depressional forested wetlands (Reiss and Brown 2005). Two of those species, *Commelina diffusa* (dayflower) and *Ludwigia peruviana* (water-primrose) also occurred as tolerant indicator species for depressional herbaceous wetland (Lane et al. 2003). The additional three overlapping tolerant indicator species between the flowing water and depressional forested wetlands were *Acer rubrum* (red maple), *Sambucus canadensis* (elderberry), and *Toxicodendron radicans* (Eastern poison ivy). One tolerant indicator species for flowing water wetlands, *Nyssa sylvatica* var. *biflora* (swamp tupelo) was found to be a sensitive indicator species for depressional herbaceous wetlands (Lane et al. 2003); and the tolerant indicator species for forested strand and floodplain wetlands *Panicum rigidulum* (red-top panicum) was found to be a sensitive indicator species for both depressional herbaceous and forested wetlands. Clearly a larger sample size and refinement of indicator species analysis for the forested strand and floodplain wetlands FWCI could reduce the inconsistencies among indicator species.

Similarly, four sensitive indicator species were common among depressional herbaceous, depressional forested, and forested strand and floodplain wetlands, including *Cladium jamaicense* (saw-grass), *Lyonia lucida* (fetter-bush), *Panicum hemitomon* (maidencane), and *Pinus elliottii* (slash pine). *Vaccinium corymbosum* (highbush blueberry) was also considered a sensitive indicator species for both depressional herbaceous and forested strand and floodplain wetlands. However, three sensitive indicator species for forested strand and floodplain wetlands were listed as tolerant indicator species for depressional forested (*Saururus cernuus*, lizard's tail) or depressional herbaceous and forested (*Centella asiatica*, coinwort; *Eupatorium capillifolium*, dog fennel) wetlands.

The three additional metrics included in the FWCIs were Floristic Quality Assessment Index score, exotic species, and native perennial species (modified from the depressional herbaceous wetland FWCI that used annual to perennial ratio). The variant of the annual to perennial ratio (as native perennial species) was used to account for variable conditions at urban wetlands, which were not included in the study of depressional herbaceous wetlands but were studied in both the depressional forested and forested strand and floodplain wetland research. Wienhold and Van der Valk (1989) and Ehrenfeld and Schneider (1991) determined that disturbance often favors annual species over perennial species and promotes the invasion of nonnative perennials in wetlands. Galatowitsch et al. (2000) found that while native perennial cover was reduced in wetlands impacted by cultivation, the occurrence of introduced perennials rather than annuals increased in stormwater impacted wetlands. The sixth depressional forested FWCI metric was the wetland status species (including both obligate and facultative wetland species), which was not included for the preliminary forested strand and floodplain wetland FWCI as it did not meet selection criteria for inclusion.

Overall, the depressional herbaceous, depressional forested, and flowing water forested wetland FWCIs included five similar metrics. Perhaps the strong similarity of

metrics selected for inclusion in the FWCIs suggests that a universal assessment index with core metrics could be constructed regardless of wetland type. However, it would likely be necessary to maintain independent indicator species lists and metric scoring criteria for different wetland types within each wetland region.

Limitations and Further Research

Generally wetlands were visited only once, with a complete sample effort lasting just one day, which provided a mere snapshot of wetland condition. Visiting these wetlands only once did not allow insight into seasonal or yearly variations in the macrophyte assemblage; and the preliminary forested strand and floodplain FWCI would benefit from inter-seasonal validation. The FWCI would also benefit from validation based on a new set of wetlands to test the repeatability of this index. A larger sample size of wetlands will improve the scoring criteria of the FWCI based on wetland regions for metrics such as indicator species analysis. Funding for additional wetland sampling and FWCI refinement is in the application phase. Regionalization appears to be an important next step in refining the FWCI for all wetland types, as this study was limited to a statewide approach due to small sample sizes within each wetland region.

Conclusions

The use of the macrophyte assemblage for a biological assessment of Florida freshwater flowing water forested wetlands provided a useful tool for detecting changes in biological integrity associated with changes in anthropogenic activity. While richness and diversity measures of macrophyte community composition were not particularly sensitive to changes in landscape development intensity, metrics used as biological indicators of changes in macrophyte community composition were. In fact, the strong correlation between the landscape scale human disturbance gradient (LDI) and the local scale wetland condition index of biological integrity (FWCI) demonstrated the potential value of using the LDI index as an initial indication of biological integrity, which can be further tested with chemical and physical parameters and compared against assemblage specific biological indices.

Due to similarities with metrics from the FWCIs for depressional herbaceous (Lane et al. 2003) and depressional forested (Reiss and Brown 2005) wetlands, a multi-metric multi-assemblage FWCI could be constructed for all freshwater wetlands throughout the state of Florida, with a set of core metrics and specific indicator species and metric scores based on wetland type within the wetland regions. While the forested strand and floodplain FWCI for flowing water systems can not be used to predict changes in the physical and chemical parameters of a wetland, its strength lies in providing an overview of biological integrity through the integration of changes in macrophyte community composition from cumulative effects. The quantitative score of biological integrity established through the FWCI should not be used as a surrogate for wetland value, but as an objective, quantitative means of comparing changes in community composition along gradients of human development intensity, which can be used objectively to assess the biological integrity of Florida's wetlands.

APPENDIX A
STANDARD OPERATING PROCEDURES

CHECKLIST OF MATERIALS/FIELD EQUIPMENT

Miscellaneous

- ✓ SOPs
- ✓ Large cooler with frozen ice bottles for unknown vegetation
- ✓ Waders
- ✓ Garmin III - GPS unit
- ✓ Florida Gazetteer
- ✓ Machete
- ✓ Aerial photo & FLUCCS codes of site

Vegetation Transects

- ✓ 2-3 100m transect tapes
- ✓ 1 m PVC with distance marks (cm)
- ✓ 2-3 compasses
- ✓ Clipboards
- ✓ Field data sheets – a minimum of 10 per site
- ✓ Site Characterization & WRAP sheets – 1 per person per site
- ✓ Pencils
- ✓ Sharpie
- ✓ Bag for unknown plants
- ✓ Masking tape
- ✓ Field ID manuals
- ✓ Prism for basal area
- ✓ Hand lens
- ✓ Index cards

SOPS FOR FORESTED STRAND WETLANDS: VEGETATION

1. Note the direction of flow through the landscape.
2. Locate a line running through the center of the strand along the flow gradient. This is the center-line.
3. Randomly select a starting point for the initial transect. Each consecutive transect will begin approximately 25 m upstream of the initial transect, so that a stretch of approximately 100 m will be sampled along the length of the strand. Run transects perpendicular to the main channelized flow.
4. At the beginning of each transect, delineate the edge of the wetland using a combination of wetland plants and hydrologic indicators. Be conservative on the side of the wetland.
5. Establish the transect using a meter tape and a compass. Each transect will start with 0 meters at the wetland edge and run into the center-line (established in step 2).
6. Use a separate field data sheet for each transect. If the number of species located on a transect exceeds the number of columns on the data sheet, start a new data sheet. Be thorough in completing field data sheets including information on site, transect direction, date, and data recorder. Specify if there are multiple field data sheets for a single transect.
7. Create quadrats that are 0.5 m on either side of the transect (1-m wide) and 5-m long, record all species rooted within these elongated quadrats.
8. Plant species names are recorded on the data sheets using the full genus and species names. Each unknown species is given a unique ID code using the transect number (ex. 1-1, 1-2, 1-3, 2-1, 2-2, etc.).
9. Collect voucher specimens for all unknown species being sure to get plant inflorescence and roots, tag samples with properly labeled masking tape, and put into a labeled collection bag. Note the color of the inflorescence on the label, as the flowers often do not preserve well. Index cards can be used to protect especially sensitive parts. When vegetation sampling is complete, store the collection bag in a cooler on ice until identification can be completed.
10. Voucher specimens are identified in the field on the day of sampling. Unidentified plants will be placed in a plant press for further clarification and identification. Plant nomenclature follows FDEP's *Florida Wetland Plant Identification Manual* (Tobe et al. 1998). If time prohibits immediate pressing, unknown plants should be stored in the cooler.
11. At each 10 meters along each transect, (i.e. 10 m, 20 m, etc.), tree basal area will be recorded. Use the data sheet for basal area, and record basal area per species using variable area plots and a 10 factor prism. Hold the prism at eye level, with a bent elbow. Looking through the prism count the number of trees per species that fall within the variable area plot. The prism shall be centered over the sampling point at all times, with the field person rotating around the prism so that the entire circular area (360^{o}) around the point of sampling is included.

Forested Strand Wetlands - Field Data Sheet - Transects, 1 x 5 m quadrat presence - UF Center for Wetlands

Site: **Transect Number:**

Date: **Data Recorder:**

Species																									
0-5 m																									
5-10																									
10-15																									
15-20																									
20-25																									
25-30																									
30-35																									
35-40																									
40-45																									
45-50																									
50-55																									
55-60																									
60-65																									
65-70																									
70-75																									
75-80																									
80-85																									
85-90																									
90-95																									
95-100																									

Forested Strand Wetlands - Field Data Sheet - Basal Area - UF Center for Wetlands

Site: **Transect Direction:**

Date: **Data Recorder:**

Species												
10 m												
20 m												
30 m												
40 m												
50 m												
60 m												
70 m												
80 m												
90 m												
100 m												

SOPS FOR FORESTED FLOODPLAIN WETLANDS: VEGETATION

1. Note the direction of flow through the landscape.
2. Locate a line running through the center of the strand along the flow gradient. This is the center-line.
3. Randomly select a starting point for the initial transect, preferably this point with coincide with a Stream Condition Index (SCI) sample point. Each consecutive transect will begin approximately 25 m upstream of the initial transect, so that a stretch of approximately 100 m will be sampled along the length of the strand. Run transects perpendicular to the main channelized flow.
4. Delineate the wetland line using a combination of wetland plants and hydrologic indicators. Be conservative on the side of the wetland.
5. Establish the transect using a meter tape and a compass. Transects will be limited to a maximum length of 50 m. The first transect will begin with 0 meters at the wetland edge and run towards the center-line (established in step 2). The second transect will begin 25 m upstream of the first transect. Transect 2 begins at the edge of the channelized flow and runs through the wetland perpendicular to the flow for a maximum of 50 m. Repeat placement for transects 3 and 4.
6. Use a separate field data sheet for each transect. If the number of species located on a transect exceeds the number of columns on the data sheet, start a new data sheet. Be thorough in completing field data sheets including information on site, transect direction, date, and data recorder. Specify if there are multiple field data sheets for a single transect.
7. Create quadrats that are 0.5 m on either side of the transect (1-m wide) and 5-m long, record all species rooted within these elongated quadrats.
8. Plant species names are recorded on the data sheets using the full genus and species names. Each unknown species is given a unique ID code using the transect number (ex. 1-1, 1-2, 1-3, 2-1, 2-2, etc.).
9. Collect voucher specimens for all unknown species being sure to get plant inflorescence and roots, tag samples with properly labeled masking tape, and put into a labeled collection bag. Note the color of the inflorescence on the label, as the flowers often do not preserve well. Index cards can be used to protect especially sensitive parts. When vegetation sampling is complete, store the collection bag in a cooler on ice until identification can be completed.
10. Voucher specimens are identified in the field on the day of sampling. Unidentified plants will be placed in a plant press for further clarification and identification. Plant nomenclature follows FDEP's *Florida Wetland Plant Identification Manual* (Tobe et al. 1998). If time prohibits immediate pressing, unknown plants should be stored in the cooler.
11. At each 10 meters along each transect, (i.e. 10 m, 20 m, etc.), tree basal area will be recorded. Use the data sheet for basal area, and record basal area per species using variable area plots and a 10 factor prism. Hold the prism at eye level, with a bent

elbow. Looking through the prism count the number of trees per species that fall within the variable area plot. The prism shall be centered over the sampling point at all times, with the field person rotating around the prism so that the entire circular area (360^{o}) around the point of sampling is included.

Forested Floodplain Wetlands - Field Data Sheet - Transects, 1 x 5 m quadrat presence - UF Center for Wetlands

Site: **Transect Number:**

Date: **Data Recorder:**

Species																									
0-5 m																									
5-10																									
10-15																									
15-20																									
20-25																									
25-30																									
30-35																									
35-40																									
40-45																									
45-50																									

Forested Floodplain Wetlands - Field Data Sheet - Basal Area - UF Center for Wetlands

Site: **Transect Direction:**

Date: **Data Recorder:**

Species												
10 m												
20 m												
30 m												
40 m												
50 m												

APPENDIX B
COEFFICIENT OF CONSERVATISM SCORES

Table B-1. Coefficient of Conservatism (CC) scores for macrophyte species identified in forested strand and floodplain wetlands in Florida. CC scores were assigned from FQAI surveys from isolated depressional forested wetlands (Reiss and Brown 2005) first, followed by isolated depressional herbaceous wetlands (Lane et al. 2003). Species without scores from previous FQAI studies were scores according to their faithfulness and fidelity to freshwater flowing water wetlands (strands and floodplains) in 2003 by five expert Florida botanists (Tony Arcuri, Dan Austin, David Hall, Nina Raymond, and Bruce Tatje).

Species	CC Score	1999/2000 Herbaceous	2001/2002 Forested	2003 Flowing
Abrus precatorius	0.0			1
Acer rubrum	5.2		1	
Acer saccharum	6.0			1
Agarista populifolia	6.5			1
Alnus serrulata	6.0			1
Alternanthera philoxeroides	0.0		1	
Alternanthera sessilis	0.7		1	
Amaranthus australis	2.6		1	
Ambrosia artemisiifolia	0.7		1	
Ampelopsis arborea	3.3		1	
Amphicarpum muhlenbergianum	5.0		1	
Andropogon virginicus	2.6		1	
Annona glabra	6.8		1	
Apios americana	3.1		1	
Ardisia escallonioides	7.0			1
Arisaema triphyllum	6.5			1
Aronia arbutifolia	5.7		1	
Arundinaria gigantea	5.3			1
Asimina parviflora	5.0			1
Asimina reticulata	4.4			1
Asplenium heterochroum	8.5			1
Aster elliottii	4.2		1	
Bacopa caroliniana	6.0		1	
Bacopa monnieri	4.3		1	
Begonia cucullata	1.5		1	
Berchemia scandens	5.1		1	
Bidens alba	1.0		1	
Bidens mitis	3.8		1	
Bignonia capreolata	4.4			1
Blechnum serrulatum	5.5		1	
Boehmeria cylindrica	4.5		1	
Bumelia lycioides	6.0		1	
Callicarpa americana	2.4		1	
Campsis radicans	3.3		1	
Carex albolutescens	3.6		1	

Species	CC Score	1999/2000 Herbaceous	2001/2002 Forested	2003 Flowing
Carex crus-corvi	6.0			1
Carex leptalea	6.8			1
Carex lupulina	6.3			1
Carex stipata	4.5	1		
Carpinus caroliniana	6.8			1
Carya glabra	5.8			1
Celtis laevigata	5.0		1	
Centella asiatica	1.9		1	
Cephalanthus occidentalis	6.0		1	
Chasmanthium laxum	6.0			1
Chasmanthium nitidum	6.3			1
Chrysobalanus icaco	6.3		1	
Cinnamomum camphora	0.2		1	
Cladium jamaicense	5.5		1	
Clematis crispa	5.5			1
Colocasia esculenta	0.0		1	
Commelina communis	2.5			1
Commelina diffusa	1.7		1	
Conyza canadensis	0.3		1	
Cornus foemina	6.6			1
Crataegus marshallii	6.3			1
Crinum americanum	7.6		1	
Cuphea carthagenensis	1.4		1	
Cynodon dactylon	0.0		1	
Cyperus difformis	2.0			1
Cyperus globulosus	1.8		1	
Cyperus haspan	2.6		1	
Cyperus ligularis	3.2			1
Cyperus odoratus	3.6		1	
Cyrilla racemiflora	4.5		1	
Decumaria barbara	6.3			1
Dichondra carolinensis	1.9		1	
Dichromena colorata	5.5		1	
Digitaria serotina	1.8		1	
Diodia virginiana	2.4		1	
Dioscorea floridana	5.5			1
Diospyros virginiana	4.0		1	
Eclipta alba	1.7		1	
Emilia fosbergii	0.5			1
Erechtites hieraciifolius	2.1		1	
Erianthus giganteus	6.0		1	
Eryngium baldwini	4.4			1
Eryngium prostratum	4.0		1	
Erythrina herbacea	4.0			1
Eugenia uniflora	1.3			1
Euonymus americanus	5.5			1
Eupatorium capillifolium	0.5		1	
Eupatorium mikanioides	5.5			1

Species	CC Score	1999/2000 Herbaceous	2001/2002 Forested	2003 Flowing
Eupatorium perfoliatum	5.9	1		
Eustachys petraea	2.0			1
Fagus grandifolia	7.5			1
Fraxinus caroliniana	7.1		1	
Galium tinctorium	3.1		1	
Gaylussacia dumosa	5.4	1		
Gaylussacia frondosa	6.7		1	
Gelsemium sempervirens	4.0		1	
Gordonia lasianthus	6.7		1	
Gratiola hispida	6.0			1
Hamamelis virginiana	6.5			1
Hibiscus coccineus	6.6			1
Hydrocotyle bonariensis	3.3		1	
Hydrocotyle ranunculoides	3.1		1	
Hydrocotyle umbellata	2.9		1	
Hydrocotyle verticillata	3.1		1	
Hypericum brachyphyllum	6.8		1	
Hypericum hypericoides	4.0		1	
Hypericum myrtifolium	5.5		1	
Hypericum tetrapetalum	5.0			1
Hypoxis curtissii	5.7			1
Hyptis alata	4.3		1	
Ilex cassine	8.1		1	
Ilex coriacea	6.0		1	
Ilex glabra	4.3		1	
Ilex opaca var. opaca	6.0		1	
Ilex vomitoria	4.8		1	
Ipomoea hederifolia	2.0			1
Ipomoea pandurata	4.8			1
Ipomoea sagittata	5.4		1	
Itea virginica	7.9		1	
Juncus effusus	1.9		1	
Juncus megacephalus	3.3		1	
Juncus polycephalos	3.3		1	
Juniperus virginiana	5.2		1	
Justicia ovata	5.5		1	
Koelreuteria elegans	2.0			1
Lachnanthes caroliana	3.1		1	
Lemna minor	1.0		1	
Leucothoe axillaris	7.0			1
Leucothoe racemosa	6.2		1	
Ligustrum sinense	0.0		1	
Lindernia grandiflora	3.6		1	
Liquidambar styraciflua	3.3		1	
Liriodendron tulipifera	6.8			1
Lobelia cardinalis	6.8			1
Lonicera japonica	0.0		1	
Ludwigia maritima	3.3		1	

Species	CC Score	1999/2000 Herbaceous	2001/2002 Forested	2003 Flowing
Ludwigia palustris	4.0		1	
Ludwigia peruviana	1.2		1	
Ludwigia pilosa	5.8			1
Ludwigia repens	2.9		1	
Lycopus rubellus	5.2		1	
Lygodium japonicum	0.0		1	
Lygodium microphyllum	0.0		1	
Lyonia fruticosa	6.0			1
Lyonia lucida	6.0		1	
Lyonia mariana	6.8			1
Magnolia grandiflora	6.2			1
Magnolia virginiana var. australis	8.1		1	
Matelea floridana	6.7		1	
Melaleuca quinquenervia	0.0		1	
Merremia dissecta	0.3			1
Micranthemum glomeratum	4.0			1
Micranthemum umbrosum	4.3		1	
Micromeria brownei	4.8		1	
Mikania scandens	2.4		1	
Mimosa pigra	0.7			1
Mitchella repens	6.7		1	
Momordica charantia	0.0		1	
Morus rubra	4.4			1
Myrica cerifera	3.1		1	
Myrsine guianensis	5.2		1	
Nephrolepis exaltata	3.8			1
Nyssa ogeche	7.0			1
Nyssa sylvatica var. biflora	7.4		1	
Oeceoclades maculata	0.4		1	
Oplismenus setarius	3.3		1	
Orontium aquaticum	7.6			1
Osmunda cinnamomea	5.5		1	
Osmunda regalis	6.9		1	
Oxypolis filiformis	6.7		1	
Panicum abscissum	9.2	1		
Panicum anceps	4.3			1
Panicum commutatum	4.5		1	
Panicum dichotomum	4.0		1	
Panicum ensifolium	5.0		1	
Panicum erectifolium	5.7		1	
Panicum hemitomon	5.0		1	
Panicum rigidulum	4.5		1	
Panicum spretum	5.4		1	
Panicum tenue	4.2		1	
Parietaria praetermissa	3.0			1
Parthenocissus quinquefolia	3.0		1	
Paspalum conjugatum	3.1		1	
Paspalum notatum	0.0		1	

Species	CC Score	1999/2000 Herbaceous	2001/2002 Forested	2003 Flowing
Passiflora incarnata	3.0		1	
Peltandra virginica	5.8			1
Persea borbonia	6.3		1	
Persea palustris	7.4		1	
Phlebodium aureum	6.8		1	
Phyla nodiflora	1.4		1	
Phyllanthus urinaria	0.0		1	
Phytolacca americana	1.2		1	
Pieris phyllyreifolia	9.5		1	
Pinckneya bracteata	8.3			1
Pinus clausa	5.6		1	
Pinus elliottii	4.0		1	
Pinus taeda	3.3		1	
Pluchea foetida	3.8		1	
Pluchea rosea	3.6		1	
Polygala rugelii	8.2	1		
Polygonum densiflorum	5.3	1		
Polygonum hirsutum	8.2	1		
Polygonum hydropiperoides	2.6		1	
Pontederia cordata	5.0		1	
Proserpinaca palustris	3.8		1	
Prunus caroliniana	3.0		1	
Prunus serotina	3.6		1	
Psychotria nervosa	5.2			1
Psychotria sulzneri	5.5			1
Pteridium aquilinum	3.6		1	
Ptilimnium capillaceum	3.1		1	
Quercus laurifolia	3.6		1	
Quercus michauxii	5.7			1
Quercus nigra	2.1		1	
Quercus virginiana	4.2		1	
Rhododendron canescens	6.8			1
Rhododendron viscosum	7.6		1	
Rhus copallinum	2.4		1	
Rhynchospora baldwinii	5.7			1
Rhynchospora inundata	6.0		1	
Rhynchospora microcephala	4.8		1	
Rhynchospora miliacea	7.1		1	
Rubus argutus	2.1		1	
Rubus trivialis	1.9		1	
Ruellia caroliniensis	4.3		1	
Rumex verticillatus	4.8			1
Sabal minor	6.2			1
Sabal palmetto	4.5		1	
Sabatia calycina	6.2			1
Sagittaria filiformis	6.0			1
Sagittaria lancifolia	4.5		1	
Sagittaria latifolia	5.0		1	

Species	CC Score	1999/2000 Herbaceous	2001/2002 Forested	2003 Flowing
Salvia lyrata	3.2			1
Sambucus canadensis	1.7		1	
Samolus valerandi	5.6			1
Sanicula canadensis	5.7			1
Sapium sebiferum	0.0		1	
Saururus cernuus	5.5		1	
Schefflera actinophylla	0.5			1
Schinus terebinthifolius	0.0		1	
Scleria triglomerata	4.8		1	
Serenoa repens	4.5		1	
Setaria geniculata	3.1		1	
Sida rhombifolia	1.0		1	
Smilax auriculata	3.8		1	
Smilax bona-nox	2.6		1	
Smilax glauca	3.3		1	
Smilax laurifolia	5.2		1	
Smilax pumila	6.0	1		
Smilax smallii	4.5		1	
Smilax tamnoides	3.6		1	
Smilax walteri	6.0		1	
Solidago fistulosa	3.6		1	
Sparganium americanum	6.7			1
Sporobolus floridanus	7.1		1	
Stenotaphrum secundatum	0.8		1	
Stillingia aquatica	7.4		1	
Symplocos tinctoria	6.0			1
Taxodium ascendens	8.8		1	
Taxodium distichum	7.2	1		
Thelypteris dentata	3.4			1
Thelypteris hispidula	4.5		1	
Thelypteris palustris	5.8			1
Tilia americana	5.5		1	
Toxicodendron radicans	1.9		1	
Triadenum virginicum	5.0		1	
Trichostema dichotomum	4.5			1
Trifolium repens	0.0		1	
Tripsacum dactyloides	4.0		1	
Ulmus americana	7.4		1	
Urena lobata	0.0		1	
Vaccinium arboreum	6.4		1	
Vaccinium corymbosum	5.7		1	
Vaccinium stamineum	5.8			1
Vaccinium tenellum	5.5			1
Viburnum dentatum	6.0			1
Viburnum nudum	5.0			1
Viburnum obovatum	4.7			1
Viola affinis	5.5			1
Vitis aestivalis	2.9		1	

Species	CC Score	1999/2000 Herbaceous	2001/2002 Forested	2003 Flowing
Vitis cinerea	2.0		1	
Vitis rotundifolia	2.1		1	
Vitis shuttleworthii	3.5			1
Wisteria sinensis	1.0			1
Woodwardia areolata	5.7		1	
Woodwardia virginica	4.8		1	
Xyris jupicai	1.7		1	
Youngia japonica	0.0		1	

APPENDIX C
SUMMARY STATISTICS

Table C-1. Summary statistics of richness (R), evenness (E), Shannon diversity (H'), Simpson diversity (D), and Whittaker's beta diversity (β_W) for the macrophyte assemblage (species level).

Site	R	E	H'	D	βw
FF1	59	1.0001	4.08	0.98	5.88
FF2	21	1.0002	3.05	0.95	3.73
FF3	56	0.9999	4.03	0.98	4.86
FF4	46	1.0001	3.83	0.98	7.36
FF5	77	1.0000	4.34	0.99	6.82
FF6	25	1.0000	3.22	0.96	1.78
FF7	32	1.0001	3.47	0.97	3.49
FF8	75	0.9999	4.32	0.99	6.11
FF9	45	1.0001	3.81	0.98	3.90
FF10	35	0.9999	3.56	0.97	3.18
FF11	29	0.9999	3.37	0.97	4.73
FF12	46	1.0001	3.83	0.98	5.15
FF13	48	0.9999	3.87	0.98	4.93
FF14	60	0.9999	4.09	0.98	5.94
FS1	39	1.0001	3.66	0.97	2.68
FS2	39	1.0001	3.66	0.97	4.90
FS3	33	1.0001	3.50	0.97	4.45
FS4	55	0.9999	4.01	0.98	5.03
FS5	39	1.0001	3.66	0.97	3.91
FS6	29	0.9999	3.37	0.97	3.02
FS7	30	0.9999	3.40	0.97	2.96
FS8	35	0.9999	3.56	0.97	2.12
FS9	47	1.0000	3.85	0.98	3.73
FS10	42	1.0001	3.74	0.98	4.43

APPENDIX D
METRIC SCORING FOR THE MACROPHYTE FLORIDA WETLAND CONDITION INDEX FOR FLOWING WATER SYSTEMS

1. Calculate values for the 5 metrics:
   1 - Proportion tolerant indicator species
   2 – Proportion sensitive indicator species
   3 – FQAI score
   4 – Proportion exotic species
   5 – Proportion native perennial species

2. Take the natural log of metrics to improve distribution.
   = ln (metric value + 1)
   1 is added to avoid errors related to taking the natural log of a zero value

3. Use the scoring equations to normalize scores between 0 and 10.
   Metrics that increase with increasing LDI – tolerant, exotic metrics:
   = 10 - ((metric - 5^{th} percentile) * (10 / (95^{th} percentile - 5^{th} percentile)))

   Metrics that decrease with increasing LDI – sensitive, FQAI, native perennial metrics:
   = ((metric - 5^{th} percentile) * (10 / (95^{th} percentile - 5^{th} percentile)))

Below are the 5^{th} and 95^{th} percentiles for each metric (transformed values are presented, see step 2 above):

	5^{th} Percentile	95^{th} Percentile
Proportion tolerant indicator species	0.05	0.24
Proportion sensitive indicator species	0.00	0.16
FQAI score	1.44	1.84
Proportion exotic species	0.00	0.17
Proportion native perennial species	0.56	0.69

4. Rescore, so that the metrics in the outer 5^{th} percentiles receive scores of 0 or 10.

   = IF (score < 0, 0, (IF (score >= 10, score, 10)))

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