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Florida Wetland Condition Index for depressional forested wetlands

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Abstract

Increasingly in the last decade biological monitoring and assessment have been used by federal and state agencies to assess water quality standards as required under the Clean Water Act. These efforts have led to the development of indices of biological integrity (often referred to as IBIs). Many states have created multi-metric indices, incorporating individual metrics into a quantitative value of community condition or biological integrity. The primary objective of this study was to develop the Florida Wetland Condition Index (FWCI) as a tool to evaluate the biological integrity of Florida depressional freshwater forested wetlands.

Vegetative community composition and chemical and physical water and soil parameters were measured at 118 wetlands throughout Florida. An independent measure of the human disturbance gradient, the Landscape Development Intensity (LDI) index, which is based on the use of nonrenewable energy within a 100 m buffer around a wetland, was calculated. Six macrophyte community composition metrics were selected for inclusion in the FWCI based on the strength of correlation with LDI (P < 0.01) and differentiation between low (LDI < 2.0) and high (LDI \geq 2.0) LDI groups (P < 0.01). The metrics included tolerant indicator species, sensitive indicator species, exotic species, floristic quality assessment index, native perennial species, and wetland status species. Metrics were scaled between 0 and 10, with 10 representing the reference wetland condition (correlating to wetlands in undeveloped landscapes). Scaled metrics were then added together to create the FWCI, with values ranging from 0 to 60. The FWCI was significantly correlated with LDI (P < 0.001). In addition, significantly differentiated among sample wetlands categorized by low and high LDI groups (P < 0.001). In addition, significant correlations were found among the six metrics, FWCI, and LDI with measured chemical and physical water and soil parameters, including water column pH, turbidity, ammonia-nitrogen concentration. The primary efficacy of the FWCI was the calculation of a quantitative value of biological integrity for wetlands across a gradient of anthropogenic land use activities, which can be used objectively to assess water quality standards of Florida wetlands.

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Keywords: Biological integrity; Florida Wetland Condition Index; Forested wetlands; Landscape Development Intensity; Urban wetlands

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1. Introduction

Since the time of human settlement the geographic area of wetlands in the United States has been reduced and altered by anthropogenic development, with some wetlands being filled or drained and others being left as fragmented habitat in a human influenced landscape matrix. Understanding the condition of remnant and altered wetlands has been of increasing concern under regulatory guidelines since the passage of the Clean Water Act. As a first step, defining wetland condition requires a definition of biological reference wetland condition to establish a point of comparison. The concept of reference condition has been defined as biological integrity (Karr, 1993), or the ability of an ecosystem to sustain the community composition, structure, and function characteristic of an otherwise natural or undisturbed ecosystem.

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 lost 46% of the pre-1780s wetland area (Dahl, 2000; Mitsch and Gosselink, 1993). Throughout the continental United States, similar trends were apparent, with a drastic decline in the surface area of wetlands. 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 exposed to the effects of land use practices such as herbicide and pesticide application, irrigation, livestock watering and wastes, soil erosion, and deposition. An additional 17% of the remaining wetlands 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.

Biological monitoring and assessment research has begun to address the question of biological integrity of wetlands influenced by various anthropogenic land use activities. The primary aim of biological monitoring and assessment is to detect changes in abundance, structure, and diversity of a target species assemblages as compared to the reference condition. A variety of assemblages have been used in biological assessments, including diatoms (Fore and Grafe, 2002); macrophytes (Galatowitsch et al., 1999; Gernes and Helgen, 1999; Mack, 2001; Lane, 2003); macroinvertebrates (Kerans and Karr, 1994; Barbour et al., 1996); amphibians (Micacchion, 2004); fish (Schulz et al., 1999); and birds (O'Connell et al., 1998). Biotic indices have been applied to ecosystems throughout the United States (i.e. Karr, 1981; Lenat, 1993; Lane et al., 2002).

Macrophytes, defined as emergent, submergent, or floating plants (USEPA, 1998), are one of the most obvious and easily identifiable assemblages in the landscape. The macrophyte assemblage plays 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 (Cronk and Fennessy, 2001). The spatial distribution of macrophytes in the landscape results from a multitude of factors, including substrate type, water chemistry, and hydroperiod, as well as other larger scale factors such as available seed source and climatic conditions. Crowder and Painter (1991) state that a lack of macrophytes where they are otherwise expected to grow can be indicative of 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 specific macrophyte species may signify increased nutrient loading (USEPA, 1998). Fennessy et al. (2001) assert that the community composition of wetland vegetation typifies the chemical, physical, and biological dynamics of a wetland in time and space.

Numerous means of quantifying a human disturbance gradient have been used in parallel with biotic indices as corroborative confirmation of measured biological integrity (i.e. Ohio Rapid Assessment Method (ORAM) v 5.0 and Vegetation Indices of Biotic Integrity (VIBI), Mack et al., 2000; Landscape Development Intensity (LDI) index and Wetland Condition Index (WCI) for depressional herbaceous wetlands, Lane, 2003). The Landscape Development Intensity (LDI) index (Brown and Vivas, 2005) is one such measure of anthropogenic influence that provides an independent, quantitative. and reproducible measure of the human disturbance gradient. The underlying concept behind calculating the LDI index (quantifying the nonrenewable energy use per unit area in the surrounding landscape expressed in emergy terms) stems from earlier works by Odum (1995), who pioneered emergy analysis for environmental accounting. (Emergy is an established 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 correlates to limited development in the surrounding landscape and the maintenance of ecosystem structure and function, even when stressors 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 wetland buffer multiplied times the LDI coefficient, which is defined by the amount of nonrenewable energy use for a given land use (Reiss, 2004; 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 (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, with the highest LDI coefficient being a 10.0 correlating to Central Business District land use. 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.

The creation of the Florida Wetland Condition Index (FWCI), a multi-metric index of biological integrity, for isolated depressional forested wetlands based on the macrophyte assemblage was the primary goal of this research. The LDI was used as the human disturbance scale in the development of the FWCI. As a secondary objective, chemical and physical water and soil parameters were correlated with measured biological integrity to correlate the findings from the chemical, physical, and biological measures of forested wetlands along a disturbance gradient.

2. Methods

Sample wetlands (n = 118) were selected spatially throughout Florida (latitude 26.0°N–31.0°N; longitude 80.1°W–87.5°W, Fig. 1) so that a nearly equal number was sampled within four Florida ecoregions (Lane, 2000). Wetlands ranged in size from 0.1 to 2.1 ha (mean 0.6 ± 0.4 ha). Field research spanned two growing seasons with 72 wetlands sampled between May and September in 2001 and an additional 46 wetlands sampled between May and October in 2002. Wetlands were categorized by generalized a priori land use categories (reference, agricultural, or urban based on the dominant



Fig. 1. Site locations for 118 isolated depressional forested wetlands in Florida. Sites were sorted by a priori land use category (reference, agricultural, urban). Ecoregion boundaries from Lane (2000).

surrounding land use) and ecoregion (panhandle, north, central, south; Lane, 2000).

Anthropogenic activity in the landscape was quantified using the Landscape Development Intensity (LDI) index. The following procedure was used to calculate LDI scores: (1) sample wetlands were delineated from 1999 aerial images (available from Labins from the Florida Department of Environmental Protection), and a 100 m buffer was delineated around the edge of each wetland in ArcView GIS 3.2 (Environmental Systems Research Institute, Inc., 1999), (2) land uses within the 100 m buffer were digitized based on aerial images and updated during the site visit to reflect any changes since 1999, and (3) the following equation was applied to calculate LDI for each study wetland:

$$\mathrm{LDI}_{\mathrm{total}} = \sum \% \, \mathrm{LU}_i \times \mathrm{LDI}_i \tag{1}$$

where LDI_i is the LDI coefficient (Reiss, 2004; Brown and Vivas, 2005) for a particular land use, *i*, based on the amount of nonrenewable energy use per unit area in the surrounding landscape, and %LU_i is the percent of land use within a 100 m buffer around the study wetland.

2.1. Field data collection

At each study wetland, the upland/wetland boundary was delineation based on plant species composition and wetland status, and hydrologic indicators (Tobe et al., 1998; USDA, 2002). Four 1 m wide belt transects spanning the entire length of the wetland from the upland/wetland boundary were lain following north/south and east/west cardinal directions, thus meeting at the center of each wetland. Presence/ absence vegetation data were collected along each belt transect, which were subdivided into 1 m wide \times 5 m long quadrats. Living macrophytes rooted within each quadrat were identified to the lowest taxonomic level possible. Additional characteristics were collected for use in developing potential biological indicator metrics, including growth form (aquatic, fern, grass, herb, sedge, shrub, tree, or vine) and category (annual or perennial, evergreen or deciduous, indigenous or exotic). The timeline for determining the exotic status of a species was set near the beginning of European settlement in North America (Tobe et al., 1998; Wunderlin, 1998; USDA, NRCS, 2002; Wunderlin and Hansen, 2003).

The Florida specific wetland indicator status was determined for five potential wetland status classifications: obligate, facultative wetland, facultative, facultative upland, and upland (Tobe et al., 1998; Wunderlin and Hansen, 2003). If a species was not listed with a Florida wetland indicator status, the National Wetlands Inventory wetland indicator status for the United States was used (Godfrey and Wooten, 1981; Wunderlin, 1998; USDA, NRCS, 2002). When information was still unavailable in published literature for species encountered, Florida botanists (who also participated in the Floristic Quality Assessment Index) were consulted to determine wetland indicator status. Original data were archived at the Howard T. Odum Center for Wetlands, University of Florida, Gainesville, Florida.

A grab water sample was taken in the deepest pool of each wetland provided standing water was present in at least 50% of the wetland area with a minimum water depth of at least 10 cm. Water samples (n = 75)analyzed for color (EPA 110.2), turbidity (EPA 180.1), pH (150.1), and specific conductance (EPA 120.1) were placed on ice and maintained at 4 °C. Samples analyzed for ammonia-nitrogen (EPA 350.1), nitrate/ nitrite-nitrogen (EPA 353.2), total Kjeldahl nitrogen (TKN) (EPA 351.2), and total phosphorus (TP) (EPA 365.4) (USEPA, 1983) were preserved in the field with acid (2 mL H₂SO₄ per sample), placed in a cooler with the above sample, and shipped to the Florida Department of Environmental Protection Central Chemistry Laboratory, Tallahassee, Florida. Original data were archived in the Florida Department of Environmental Protection STORET database and at the Howard T. Odum Center for Wetlands, University of Florida, Gainesville, Florida.

Soil cores were taken from sample wetlands using a 7.6 cm diameter PVC pipe driven 10 cm into the soil at the midpoint of each transect, for a total of four samples at each site. The cores were then homogenized into a composite sample per site and preserved on ice. Analyses for soil moisture and organic matter (Gardner, 1986) were completed in house. TKN (USEPA, 1993) and TP (USEPA, 1979) were analyzed at the Institute of Food and Agricultural Sciences Analytical Research Laboratory, Gainesville, Florida. Original data were archived at the

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

2.2. Floristic Quality Assessment Index

A Floristic Quality Assessment Index (FQAI) was developed for Florida isolated depressional forested wetlands following Wilhelm and Ladd (1988). The FQAI score for an individual wetland was calculated as:

$$FQAI = \frac{\sum (C_1 + C_2 + \dots + C_n)}{N}$$
(2)

where C is the Coefficient of Conservatism score (C of C), and N the species richness. This equation was considered a modified FQAI because previous studies used the square root of native species richness as the denominator. Assumptions of 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 C of C (or FQAI as calculated in this study) than with the traditional FOAI equation (using the square root of 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 C of C with the human disturbance gradient (measured with LDI).

Five Florida botanists participated in the FQAI survey, including Guy Anglin, David Hall, Ashley O'Neal, Nina Raymond, and John Tobe. Each botanist was sent a complete list of species identified in wetlands sampled in the 2001 field season (n = 482) species), and scored each species based on its faithfulness to Florida depressional forested wetlands. After the 2002 field season, one botanist (David Hall) scored the additional 79 species not previously encountered in the 2001 sample season, raising the number of species with C of C scores to 561 species (Reiss, 2004). Potential C of C scores ranged from zero (exotic and native species that act as opportunistic invaders, included species that commonly occur in disturbed ecosystems) to 10 (species that occurred within a narrow set of stable ecological conditions and characteristic of a stable, reference ecosystem).

Species with low C of C scores were considered tolerant of many disturbances, whereas species with high C of C scores were considered to occur within a narrow set of stable ecological condition.

2.3. Data analysis

Water and soil parameters were compared among wetlands within three a priori land use categories (reference, agricultural, and urban) using Fisher's LSD pair wise comparison (Minitab Statistical Software, State College, Pennsylvania, 2000). Candidate macrophyte metrics were calculated at the statewide scale using all 118 sample wetlands, and also at the regional scale using only those wetlands within each ecoregion (Lane, 2000). Each candidate metric was constructed in multiple forms including the number (N), percent (P), proportion (A), and frequency of occurrence (F). The number metric (N) referred to a straight count of species fitting the particular metric category. The percent metric (P) was calculated as the number metric (N) divided by the species richness (R)for each sample wetland:

$$P_i = \frac{N_i}{R_i} \tag{3}$$

where *i* represented a sample wetland. The proportion metric (*A*) referred to the sum of the total number of species designated by the metric category in each quadrat for each respective sample wetland (*m*) divided by the total number of all species occurrences at a wetland (*M*):

$$A_i = \sum \left(\frac{m_i \text{ occurrence of metric species}}{\sum M_i \text{ all species occurrences}} \right)$$
(4)

The frequency of occurrence metric (F) was calculated as the number of quadrats a particular category of species occurred in (q) divided by the total number of quadrats sampled at each wetland (Q):

$$F_i = \frac{q_i}{Q_i} \tag{5}$$

Sample wetlands were categorized into groups and analyzed with Indicator Species Analysis (ISA) in PCORD (1999) (MJM Software, Glenden Beach, Oregon), which evaluates the abundance and faithfulness of individuals in a defined group (McCune and Grace, 2002). ISA can be used to detect and describe the value of individuals indicative of environmental conditions. It requires a priori groups and data on the abundance or presence of individuals in each group. Calculated indicator values range from zero (no indication) to 100 (a perfect indication of a particular group). Wetlands were assigned groups based on different LDI index values for sensitive (LDI < 2.0) and tolerant (LDI > 4.0) analyses. This ensured that the pool of least impacted wetlands (LDI < 2.0) were used to define sensitive species, whereas the pool of wetlands surrounded by higher intensity land uses (LDI > 4.0), or the more impacted wetlands, were used to define tolerant indicator species. Indicator values were calculated and tested for statistical significance (P < 0.10) using a Monte Carlo randomization technique with 1000 randomized runs.

Candidate metrics were selected for inclusion in the FWCI if they satisfied three criteria: (1) correlated with the LDI according to the strength and significance of the Spearman's rank correlation coeffidisplayed visually cient: (2)distinguishable correlations with LDI in scatter plots; and (3) showed a significant difference between low (LDI < 2.0) and high (LDI \geq 2.0) LDI groups tested with the Mann-Whitney U-test. The FWCI was composed of individual metrics, which were scaled and added together. Metric scoring was based on an approach modified from the Florida Stream Condition Index (Fore, 2003). Metrics with a skewed distribution were natural log transformed to improve the distribution. The 5th to 95th percentile values of each metric were normalized from zero to 10, with 10 always representing reference biological wetland condition. Metric scores for each sample wetland were added together to create the FWCI (Reiss, 2004).

An agglomerative cluster analysis (PCORD, MJM Software, Glenden Beach, Oregon) was used to determine wetland clusters based on macrophyte community composition. A dissimilarity matrix was constructed using the Sørensen distance measure and the flexible beta ($\beta = -0.25$) linkage method. The resulting dendrogram was pruned to maintain the smallest number of significantly different clusters based on Fisher's LSD pair wise comparison (P < 0.05) of FWCI scores (Minitab Statistical Software, State College, Pennsylvania). The macrophyte metrics, FWCI and LDI were correlated with measured water and soil parameters using Spearman's rank correlation coefficient (Analyse-it Software, Leeds, England, 1997–2003).

3. Results

3.1. Water and soil

Water and soil samples were analyzed for 75 and 118 sample wetlands, respectively. Table 1 shows values (mean \pm the standard deviation) for water and soil parameters for a priori land use categories (reference, agricultural, and urban). Means with similar letters were not significantly different (Fisher's LSD pair wise comparison, $\alpha = 0.05$), including water nitrate/nitrite-nitrogen concentration and soil total Kjeldahl nitrogen (TKN) concentration. Reference wetlands had significantly different water column turbidity, pH, and total phosphorus (TP) concentration, than agricultural and urban wetlands. Whereas, the water color of urban wetlands was significantly different from reference and agricultural wetlands. Specific conductivity was significantly different between reference and urban wetlands; while water ammonia-nitrogen (mg N/L), water TKN (mg N/L), soil moisture, and soil TP (mg P/g soil) were significantly different between reference and agricultural wetlands. Soil organic matter was significantly different between agricultural and urban wetlands.

3.2. Macrophyte community composition

Statewide 605 plant species, representing 323 genera and 126 families were identified at the 118 sample wetlands. Taxodium ascendens (pond-cypress) occurred most often, and was rooted within the vegetation quadrats at 93% of the study wetlands. The second most abundant species was Myrica cerifera (wax myrtle) found in 64% of the study wetlands. The most common fern was Woodwardia virginica (Virginia chain fern) found at 53% of the wetlands: the most common vine was Toxicodendron radicans (Eastern poison ivy) also found at 53% of the wetlands; and the most common graminoid was Panicum hemitomon (maidencane) found at 50% of the wetlands. Of the species encountered, only 130 species (22%) occurred at a minimum of 5% of the sample wetlands $(n \ge 6)$. Approximately one-third of

343

	Land use category			
	Reference*	Agricultural*	Urban [*]	
Water parameters				
Color (PCU)	285 ± 178 a	346 ± 204 a	$198\pm129~\mathrm{b}$	
Turbidity (NTU)	3.8 ± 4.2 a	$17.7\pm40.7~\mathrm{b}$	$9.5\pm11.9~\mathrm{b}$	
pH	5.2 ± 1.2 a	6.2 ± 0.8 b	$6.4\pm1.0~\mathrm{b}$	
Specific conductivity (umhos/cm)	81 ± 48 a	136 ± 134 ab	$231\pm175~\mathrm{b}$	
Ammonia-nitrogen (mg N/L)	0.15 ± 0.33 a	0.33 ± 0.57 b	$0.19\pm0.27~\mathrm{ab}$	
Nitrate/nitrite-nitrogen (mg N/L)	0.09 ± 0.37 a	0.01 ± 0.01 a	0.02 ± 0.03 a	
TKN nitrogen (mg N/L)	1.93 ± 1.24 a	$3.17\pm2.20~\mathrm{b}$	$1.84 \pm 1.06 \text{ ab}$	
TP (mg P/L)	0.08 ± 0.11 a	$0.81\pm1.38~\mathrm{b}$	$0.23\pm0.26~\text{b}$	
Soil parameters				
Moisture (%)	61 ± 20 a	$46\pm17~\mathrm{b}$	55 ± 22 ab	
Organic matter (%)	$40\pm25~\mathrm{ab}$	$30\pm17~\mathrm{a}$	41 ± 28 b	
TKN nitrogen (mg N/g soil)	6.76 ± 3.68 a	5.53 ± 3.30 a	6.70 ± 4.75 a	
TP (mg P/g soil)	0.38 ± 0.28 a	$0.91\pm1.27~\mathrm{b}$	0.53 ± 0.31 ab	

Table 1

Chemical and physical water and soil parameter comparisons among 3 a priori land use categories

Values represent the mean \pm standard deviation.

^{*} Categories with similar letters were not significantly different (Fisher's LSD pair wise comparison, $\alpha = 0.05$).

the species identified (202 species or 33.5%) were rooted in only one vegetation quadrat.

3.3. Metric development

In the context of this study, metrics were defined as biological attributes, which have a consistent and predictable response to anthropogenic activities (Karr and Chu, 1997). Six metrics were selected for inclusion in the FWCI, including tolerant and sensitive indicator species; Floristic Quality Assessment Index (FQAI); exotic species; native perennial species; and wetland status species (Table 2). Tolerant indicator and exotic species increased with increasing development intensity; whereas, sensitive indicator species, FQAI, native perennial species, and wetland status species displayed the opposite trend. Metrics were

Table 2 Matrice included in the Florida Watland Condition Index (FWCI)

significantly correlated with LDI (Table 3) and significantly differentiated between LDI groups (Table 4).

Shrub and tree species were included in the ISA for both tolerant and sensitive metrics (Table 5). 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 integrity of forested wetlands and excluding species of the tree and shrub layers would seemingly underscore their importance. However, trees comprised only a small percentage of the identified tolerant and sensitive indicator species. Three percent of tolerant indicator species were trees, 9% shrubs, 14% vines, and 74% herbaceous (including herbs, sedges, grasses, etc.). The two tree tolerant indicator species included the hardwood *Acer rubrum*

Methes included in the Florida wetland Condition index (FwCl)						
Metric	Calculation	Trend along human disturbance gradient	Explanation			
Tolerant indicator species	Percent	Increase	Calculated using ISA and associated with wetlands with an $LDI > 4.0$			
Sensitive indicator species	Percent	Decrease	Calculated using ISA and associated with wetlands with an $LDI < 2.0$			
FQAI	Mean	Decrease	Calculated as the sum of the Coefficient of Conservatism score for each species divided by species richness			
Exotic species	Percent	Increase	The timeline for determining the exotic status of a species was set near the beginning of European settlement in North America			
Native perennial species	Percent	Decrease	Species need be both native and perennial to be included			
Wetland status species	Percent	Decrease	Species categorized as obligate or facultative wetland status			

Table 3 Regional correlations between the Florida Wetland Condition Index (FWCI) and six metrics with the Landscape Development Intensity (LDI) index

	Panhandle	North	Central	South
FWCI	-0.74	-0.74	-0.73	-0.88
Tolerant indicator species	0.72	0.80	0.68	0.86
Sensitive indicator species	-0.68	-0.78	-0.72	-0.71
Modified FQI	-0.68	-0.75	-0.68	-0.86
Exotic species	0.72	0.65	0.70	0.80
Native perennial species	-0.67	-0.65	-0.66	-0.80
Wetland status species	-0.60	-0.55	-0.39^{a}	-0.69

Values represent Spearman's rank correlation coefficients.

^a Correlation was significant at P < 0.05 level, all other correlations were significant at the P < 0.001 level.

(red maple) and exotic *Sapium sebiferum* (Chinese tallowtree). The six shrub statewide tolerant indicator species were *Aster carolinianus* (climbing aster), *Rubus argutus* (sawtooth blackberry), *R. trivialis* (southern dewberry), and *Sambucus canadensis* (elderberry), and also two exotic species *Ligustrum sinense* (Chinese privet) and *Ludwigia peruviana* (Peruvian primrosewillow).

For the tolerant indicator species, a central ecoregion agricultural wetland (CA2) had the highest percent tolerant indicator species (72%). [Site codes refer to the regional location of a wetland (p = panhandle; n = north; c = central; s = south); the category of surrounding land use (r = reference, a = agricultural, u = urban; and the numeric order of sampling within that category.] Ninety-three percent of the wetlands in the low LDI group had less than 20% statewide tolerant indicator species. Three outliers included SA8 (32%), CA8 (26%), and PR7 (25%). In the high LDI group, 73% of the wetlands had over 20% tolerant indicator species. In contrast, 85% of wetlands in the low LDI group had over 20% sensitive indicator species; whereas, 86% of wetlands in the high LDI group had less than 20% indicator species.

Wetlands in the panhandle (maximum FQAI = 6.25) and north (maximum FQAI = 5.95) ecoregions had higher FQAI scores versus wetlands in the central (maximum FQAI = 4.93) and south (maximum FQAI = 5.24) ecoregions. Statewide the FQAI was significantly correlated with LDI and there was a significant difference between FQAI scores between low and high LDI groups (P < 0.001). In the low LDI groups (LDI < 2.0), 87.5%, 100%, 92%, and

Table 4

Statewide comparisons among six metrics for wetlands in low (LDI < 2.0) and high $(LDI \ge 2.0)$ Landscape Development Intensity (LDI) index groups

Metric	Low LDI	High LDI	Wa
Tolerant indicator species	7.8 ± 7.8	31.2 ± 14.7	1116.5
Sensitive indicator species	39.5 ± 16.7	9.4 ± 10.1	3665.0
Modified FQI	4.81 ± 0.62	3.62 ± 0.80	3771.0
Exotic species	3.0 ± 3.6	14.3 ± 10.6	1379.0
Native perennial species	92.7 ± 4.5	79.7 ± 12.0	3453.0
Wetland status species	72.0 ± 9.8	54.1 ± 12.5	3612.0

Values represent the mean metric value \pm standard deviation.

^a The Mann–Whitney U-test statistic, all reported values were significant (P < 0.001).

83% of the wetlands had an FQAI score greater than 4.00 in the panhandle, north, central, and south ecoregions, respectively. Wetlands with an FQAI score less than 4.00 accounted for 70% 82%, 74%, and 87.5% of the wetlands in the high LDI group (LDI \geq 2.0) in the panhandle, north, central, and south ecoregions, respectively.

In total 113 of the 605 species identified (19%) were categorized as exotic species. Statewide, the percent exotic species was significantly correlated with development intensity in the surrounding landscape. The north ecoregion hosted the wetland with the greatest percent exotic species, NA1 (52.6%). NA1 was surrounded by a research facility growing experimental pasture grasses, potentially biasing the high percent exotic species present at this study wetland. The wetland with the second highest percent exotic species was SU8 (38.5%), a wetland embedded in residential and commercial urban land use. One apparent outlier in the low LDI group was SU4 (18.4%). All remaining wetlands in the low LDI group (n = 40) had less than 10% exotic species. Only six exotic species were found in all four ecoregions including Commelina diffusa (common dayflower), Cuphea carthagenensis (Columbian waxweed), Cynodon dactylon (Bermudagrass), Kyllinga brevifolia (shortleaf spikesedge), Ludwigia peruviana (Peruvian primrosewillow), and Paspalum notatum (Bahiagrass), and just 14 exotic species occurred in three of the four ecoregions.

The percent native perennial species decreased with increasing development intensity in the surrounding landscape. Of the 605 macrophyte species identified,

Table 5

Statewide indicator species for depressional forested wetlands

Tolerant indicator species			
Acer rubrum	Diodia virginiana	Lygodium japonicum	Rubus trivialis
Alternanthera philoxeroides	Dioscorea bulbifera	Melothria pendula	Sacciolepis indica
Amaranthus spinosus	Echinochloa colona	Micranthemum umbrosum	Sambucus canadensis
Ampelopsis arborea	Eclipta prostrata	Momordica charantia	Sapium sebiferum
Aster carolinianus	Eupatorium capillifolium	Oxalis corniculata	Senna obtusifolia
Axonopus fissifolius	Galium hispidulum	Parthenocissus quinquefolia	Sesbania vesicaria
Boehmeria cylindrical	Galium tinctorium	Paspalum notatum	Setaria parviflora
Carex longii	Hymenachne amplexicaulis	Paspalum urvillei	Sida rhombifolia
Colocasia esculenta	Juncus effusus	Phyla nodiflora	Smilax pumila
Commelina diffusa	Kyllinga brevifolia	Phyllanthus urinaria	Solanum carolinense
Cuphea carthagenensis	Leersia hexandra	Phytolacca americana	Solidago stricta
Cynodon dactylon	Lepidium virginicum	Polygonum hydropiperoides	Sporobolus indicus
Cyperus croceus	Ligustrum sinense	Polygonum punctatum	Stenotaphrum secundatum
Cyperus lanceolatus	Lonicera japonica	Polypremum procumbens	Toxicodendron radicans
Cyperus polystachyos	Ludwigia peruviana	Proserpinaca palustris	Trifolium repens
Cyperus retrorsus	Ludwigia repens	Richardia brasiliensis	Vitis rotundifolia
Cyperus virens	Luziola fluitans	Rubus argutus	Wedelia trilobata
Digitaria ciliaris		-	
Sensitive indicator species			
Amphicarpum muhlenbergianum	Hypericum fasciculatum	Panicum hemitomon	Sagittaria graminea
Andropogon virginicus	Hypericum myrtifolium	Panicum rigidulum	Sagittaria lancifolia
Aristida beyrichiana	Hyptis alata	Panicum tenerum	Scleria baldwinii
Aristida purpurascens	Ilex glabra	Pinus elliottii	Scleria georgiana
Cladium jamaicense	Ilex myrtifolia	Pinus palustris	Scleria triglomerata
Coelorachis rugosa	Ipomoea saggitate	Pluchea foetida	Serenoa repens
Cyperus haspan	Lachnanthes caroliana	Pluchea rosea	Spartina bakeri
Drosera brevifolia	Lophiola aurea	Polygala cymosa	Stillingia aquatica
Erianthus giganteus	Ludwigia linifolia	Polygala lutea	Syngonanthus flavidulus
Eriocaulon compressum	Lycopodiella alopecuroides	Proserpinaca pectinata	Utricularia purpurea
Eriocaulon decangulare	Lyonia lucida	Rhexia alifanus	Vaccinium corymbosum
Eupatorium leptophyllum	Nymphaea odorata	Rhexia lutea	Xyris ambigua
Eupatorium mohrii	Nymphoides aquatica	Rhexia mariana	Xyris caroliniana
Fuirena scirpoidea	Panicum ensifolium	Rhynchospora inundata	Xyris elliottii
Gratiola ramosa	Panicum erectifolium	Sabatia bartramii	Xyris jupicai
Hypericum chapmanii			- • •

Tolerant indicator species (T) were associated with wetlands in developed landscapes (LDI > 4.0), whereas sensitive indicator species (S) were associated with wetlands in undeveloped landscapes (LDI < 2.0). Indicator species were determined using Indicator Species Analysis in PCORD.

427 were classified as native perennial species (71%). Statewide 78% of the wetlands in the low LDI group (LDI < 2.0) had greater than 90% native perennial species identified at each wetland; whereas, 75% of the wetlands in the high LDI group (LDI \geq 2.0) had at least 10% non-native or non-perennial species.

Of the macrophyte species identified, 56% were categorized as "wetland status" species, including 160 designated as obligate and 180 designated as facultative wetland species. An additional 137 facultative, 62 facultative upland, and 49 upland species were identified in the study wetlands. Seventeen species

were not categorized by wetland status. Statewide 90% of the wetlands in the low LDI group (LDI < 2.0) had greater than 60% wetland status species identified at each wetland; whereas, 75% of the wetlands in the high LDI group (LDI \geq 2.0) had less than 60% wetland status species identified at each wetland.

3.4. Florida Wetland Condition Index

Metrics were constructed and scored according to a statewide (n = 118) and regional (panhandle n = 28; north n = 31; central n = 31; south n = 28) approach.

Because scoring was based on a continuous spread between the lower 5th and upper 95th percentiles for a given metric, differences would be expected if wetlands within an ecoregion consistently had lower values in any particular metric category. While the linear correlation between the statewide and regional scoring approaches was highly significant ($R^2 = 0.97$) and reference wetlands (those with the highest FWCI scores) in the panhandle and north ecoregions received similar scores for both statewide and regional approaches, reference wetlands in the central and south ecoregions consistently received lower FWCI scores in the statewide scoring approach (Fig. 2). Further results and discussion regarding the FWCI adopted the regional scoring approach.

The six metrics included in the FWCI were tolerant and sensitive indicator species, Floristic Quality Assessment Index (FQAI), exotic species, native perennial species, and wetland status species. FWCI scores decreased with increasing development intensity in all four ecoregions (Fig. 3). While the a priori reference and agricultural wetlands received scores as expected, a few urban wetlands with higher LDI scores received higher FWCI scores than anticipated. However, correlations between FWCI and six metrics with LDI were strong (|r| > 0.50, P < 0.01) for all of



Fig. 2. The Florida Wetland Condition Index (FWCI) was scored using two approaches, for the statewide dataset (n = 118) and separately for each ecoregion (panhandle n = 28; north n = 31; central n = 31; south n = 28). Scoring approaches were highly correlated ($R^2 = 0.97$), though central and south reference wetlands consistently received lower FWCI scores when scored using the statewide approach.

metrics (Table 3), except for the wetland status species metric in the central ecoregion, which was significantly correlated with LDI at a lower significance level (|r| = 0.39, P < 0.05).

Cluster analysis determined five categories of wetlands based on macrophyte community composition for the statewide data set. Clusters were roughly explained by ecoregions and a priori land use categories, including: (1): northern reference; (2): southern reference; (3): northern developed land use; (4): southern developed land use; and (5): statewide cattle land use (Fig. 4). Based on FWCI scores, clusters 1 and 2 were not significantly different from one another, but were significantly different from clusters 3, 4, and 5 (P < 0.05). Clusters 3 and 4 were not significantly different from cluster 5 was significantly different from all other clusters.

Eight of the measured chemical and physical water and soil parameters were strongly correlated with the six metrics, FWCI, or LDI (Table 6). Water column pH, turbidity, TP, and soil moisture were significantly correlated for all comparisons (P < 0.05); soil TKN was significantly correlated with all comparisons (P < 0.05), except sensitive indicator species. The FWCI was correlated with all eight measured variables, whereas the LDI was significantly correlated for only seven of the eight (excluding soil organic matter). Four measured water parameters were not significantly correlated with any of the six metrics, FWCI, and LDI scores, including water color, specific conductivity, nitrate/nitrite-nitrogen concentration, and TKN concentration.

4. Discussion

It has been suggested 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. For the purposes of this study, biological integrity has been defined quantitatively with the FWCI. The FWCI incorporated six metrics from the macrophyte species assemblage into a single quantitative value of wetland



Fig. 3. Florida Wetland Condition Index (FWCI) scores decreased with increasing Landscape Development Intensity (LDI) for wetlands in the (A) panhandle; (B) north; (C) central; and (D) south ecoregions.

condition. Correlations between the FWCI and the intensity of development in the surrounding landscape (based on the use of nonrenewable energy and calculated with the Landscape Development Intensity (LDI) index) suggest that predicted changes in macrophyte community composition were captured by the FWCI. Each of the six FWCI metrics addressed some disparity from the assumed range of the reference condition.

The tolerant indicator species metric demonstrated the strongest correlation with LDI, suggesting that the presence of a suite of species characteristic of wetlands with impaired biological integrity may be the single most effective means of identifying changes in community composition. The isolated forested wetlands sampled were influenced by various anthropogenic activities (from direct herbivory and trampling by domestic cattle, to increased nutrients from agricultural or stormwater run-off, to hydrological impoundments or drainage), yet despite the vast differences in surrounding land uses, the community composition of these wetlands was similar enough to detect a universal suite of tolerant indicator species.

One strength of the FWCI lies in providing an overview of biological integrity through the integration of changes in community composition from cumulative effects. Among a priori land use categories, differences in water and soil parameters were apparent (including water: color, turbidity, pH, specific conductivity, ammonia-nitrogen, TKN, TP; soil: moisture, organic matter, and TP). The strongest correlations of water and soil parameters with metrics, FWCI, and LDI provide insight into the influence of specific environmental parameters on macrophyte



Fig. 4. Statewide comparison of Florida Wetland Condition Index (FWCI) scores for wetlands in five clusters based on macrophyte community composition. Clusters 1 and 2 are comprised of wetlands surrounded by low development intensity, whereas clusters 3 and 4 represent wetlands surrounded by high development intensity. Cluster 5 is comprised mainly of wetlands surrounded by cattle operations. Clusters with similar letters were not significantly different (Fisher's LSD pair wise comparison, P < 0.05). (\blacksquare) Represent mean and horizontal lines represent median FWCI per cluster; boxes represent 25th to 75th percentile; vertical lines represent outliers.

community composition and the impacts from surrounding land use activities. While the FWCI cannot be used to predict changes in the chemical and physical parameters of a wetland, its strength is found in providing an overview of biological integrity through the integration of changes in community composition from cumulative effects. Perhaps initial preservation and restoration strategies could focus on limiting activities that influence changes to water column pH, turbidity, or total phosphorus loading to wetlands in order to promote biological integrity. Though it is unclear whether changes in these parameters alone would produce the low biological integrity detected.

4.1. Merits of regionalization

While the ease and utility of a single statewide FWCI would seemingly prevail over an ecoregional approach, the necessity of scoring each ecoregion based on the best possible reference conditions (Karr and Chu, 1999) cannot be overlooked. Regionalization of biological indices has been suggested throughout the literature. The main reason for classification is to compare "like to like" (Gerristen et al., 2000); that is, to reduce the noise in background variability in biological data. 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 ecoregions were seemingly the best possible examples of reference type conditions, these wetlands may be more impacted by secondary and cumulative effects (i.e. development in the surrounding landscape due to drainage

Table 6

Correlation of water and soil parameters with six metrics, Florida Wetland Condition Index (FWCI), and Landscape Development Intensity (LDI) index

	Water column			Soil				
	рН	Turbidity ^a NTU	Ammonia ^a (mg N/L)	TP ^a (mg P/L)	Moisture ^b (%)	Organic matter ^b (%)	TKN ^a (mg N/L)	TP ^a (mg P/L)
Metrics								
Tolerant indicator species	0.48^{**}	0.33**		0.53**	-0.35^{**}	-0.26^{**}	-0.20^{*}	0.38^{**}
Sensitive indicator species	-0.58^{**}	-0.29^{*}		-0.56^{**}	0.25^{**}			-0.38^{**}
FQAI	-0.54^{**}	-0.29^{*}		-0.48^{**}	0.39**	0.41**	0.33**	-0.25^{**}
Exotic species	0.58^{**}	0.38**		0.36**	-0.32^{**}	-0.35^{**}	-0.30^{**}	
Native perennial species	-0.53^{**}	-0.29^{*}	-0.25^{*}	-0.42^{**}	0.38**	0.38^{**}	0.31**	
Wetland status species	-0.40^{**}	-0.26^{*}		-0.38^{**}	0.45**	0.38**	0.35**	
FWCI	-0.56^{**}	-0.35^{**}	-0.23^{*}	-0.49^{**}	0.41**	0.35**	0.29**	-0.28^{**}
LDI	0.39**	0.30**	0.25^{*}	0.48^{**}	-0.22^{*}		-0.19^{*}	0.23*

Values represent Spearman's rank correlation coefficients.

^a Environmental variables were transformed by taking the natural log.

^b Environmental variables were transformed by taking the arcsine square root.

* Significant at the P < 0.05 level.

** Significant at the P < 0.01 level.

improvements) than their panhandle and north ecoregion counterparts. There were clear differences in FWCI scores for reference wetlands (LDI < 2.0) between statewide and regional scoring approaches, which can be seen when looking at the metric calculations for each ecoregion. For example, it was noted that there was a difference in the maximum FOAI scores of nearly one FOAI unit between the panhandle and north (higher maximum FOAI scores) and central and south (lower maximum FOAI scores) ecoregions. This suggests that the defined biological integrity of reference wetlands (correlating to those wetlands with higher FQAI score) in the south and central ecoregions would be lower than the biological integrity of reference wetlands in the panhandle and north ecoregions when scoring was completed with the statewide approach. However, when FWCI scoring was standardized among ecoregions, the assumed biological integrity of reference wetlands in all ecoregions was weighted equally. Arguments could easily be made supporting both scoring approaches (statewide or regional). However, the intent this research was to assess the current biological integrity of Florida depressional forested wetlands, and as such the comparison was made against the current day reference standard of biological integrity determined within each ecoregion. The built-in bias of a statweide approach where scoring a wetland lower simply because of its location in the south or central ecoregion suggests a lower maximum biological integrity was possible. While this may be an acceptable assumption when comparing current and historic conditions of biological integrity (assuming that historic conditions in the south and central ecoregions were comparable to those of the north and panhandle ecoregions), the premise of the FWCI was to describe the current gradient of biological integrity for Florida wetlands. Scoring the FWCI with a regional approach accounts for natural differences (i.e. climactic) and human induced alterations including secondary and cumulative effects (i.e. increased development intensity due to increased drainage of the landscape) characteristic of wetlands in different ecoregions.

4.2. Variation of biological integrity by land use

Wetland clusters based on macrophyte community composition suggested that differences in some agricultural and urban wetlands may be too subtle to detect with macrophyte community composition data alone. Furthermore, greater variability in the macrophyte assemblage of reference wetlands (Brinson and Rheinhardt, 1999 and our data) as compared to that of agricultural and urban wetlands suggested that anthropogenic induced perturbations to wetlands may result in a convergence of species present. Indeed, the decreased variability of macrophyte community composition among agricultural and urban wetlands suggests that as the degree of human disturbance increases, wetland macrophyte community composition may converge so that highly disturbed wetlands have less compositional variability than their reference wetland counterparts.

Urban wetlands visually appear to exhibit a different vector of change than do agricultural wetlands. Anthropogenic influences to urban study wetlands often entailed inflow from stormwater runoff and the dissolved and suspended contaminants that are carried in from vehicles and other urban polluters; whereas anthropogenic influences to agricultural wetlands often included pesticide and fertilizer inflows, direct herbivory from cattle, or increased nutrients from animal husbandry operations (cattle, chicken, etc.). However neither the FWCI nor the LDI significantly differentiate between agricultural and urban wetlands. Within the LDI scale, LDI coefficients for agricultural land uses ranged from 2.6 (unimproved pasture) to 6.6 (high intensity agriculture) (Brown and Vivas, 2005). Urban LDI coefficients overlap that range as Open Space/ Recreational land uses ranged from 2.1 (low intensity) to 4.8 (middle intensity) to 6.9 (high intensity). Urban wetlands in general had the widest range of scores for both LDI and FWCI.

The main conclusion drawn from the FWCI is that both agricultural and urban wetlands have lowered biological integrity as compared to reference wetlands. However, this statement is not meant to imply that these wetlands lack value, as they provide important services and do work in the environment. Wetlands embedded in a developed landscape matrix provide an abundance of potential services. For example, they may store and purify stormwater, process nutrients and toxins (perhaps acting as a sink and protecting hydrologically connected systems), provide habitat for local wildlife and migratory species, produce oxygen, filter the air, provide noise abatement, and act as refugia for urban ecologists. Specifically in the case of urban wetlands, there is a debate as to the value of small remnant wetlands embedded within highly developed landscape matrices. While wetlands do exist in highly urbanized areas, they do not appear to closely resemble wetlands in undeveloped landscapes according to either the LDI or FWCI methods of determining biological integrity.

Under current Florida law, mitigation ratios for urban wetlands will be small, and some people may question the idea of keeping urban wetlands of substandard biological integrity on expensive real estate parcels. Perhaps mitigating off-site into near-by areas with low development intensity would improve the chances of creating or preserving a wetland with the possibility of successfully meeting mitigation criteria. However, offsite mitigation undervalues the services provided by urban wetlands. Urban wetlands clearly provide some function, and perchance they are doing more work processing nutrients, storing urban stormwater run-off, and storing toxins, than wetlands in undeveloped landscapes. While the FWCI scores for urban wetlands reflect lowered biological integrity, perhaps having 30-70% on average of the biological integrity of reference wetland is more important than having no wetland in the landscape and therefore no functional capacity (i.e. stormwater treatment, flood abatement).

4.3. 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 chemical and physical water and soil parameters; and the 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 could improve the scoring criteria of the FWCI based on ecoregions for metrics such as indicator species analysis.

4.4. Conclusions

The use of the macrophyte assemblage for a biological assessment of Florida isolated depressional

forested wetland provided a useful tool for detecting changes in biological integrity associated with changes in the surrounding Landscape Development Intensity. While richness and diversity measures of macrophyte community composition were not particularly sensitive to changes in Landscape Development Intensity, biological indicators along with physical and chemical parameters were. In fact, the strong correlation between the landscape scale human disturbance gradient (LDI) and the local wetland scale 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 Wetland Condition Index for Florida depressional herbaceous wetlands (Lane, 2003), a multi-metric multi-assemblage FWCI could be constructed for all freshwater wetlands throughout the state of Florida, with specific indicator species and metric scores based on wetland type and Florida ecoregions. While the FWCI suggested low biological integrity of both agricultural and urban wetlands, these wetlands provide services and do work in the environment. Therefore, 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 Florida wetlands.

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