Characteristic Community Structure of Florida's Subtropical Wetlands: The Florida Wetland Condition Index

Kelly Chinners Reiss[1*], Mark T. Brown[1], Charles R. Lane[1,2]

[1] Howard T. Odum Center for Wetlands, University of Florida
PO Box 116350
Gainesville, Florida 32611-6350
Phone: 352-392-2425
[*]kcr@ufl.edu

[2] Current Address
US Environmental Protection Agency, Office of Research and Development, National Exposure Research Laboratory, 26 W. Martin Luther King Dr., MS 642, Cincinnati OH 45268.

ABSTRACT

Depending on the classification scheme there are between 10 and 45 different wetland types in Florida. Land use and land cover change has a marked affect on wetland condition, and different wetland types are affected differentially depending on many abiotic and biotic variables. To assess wetland condition, we have developed a Florida Wetland Condition Index (FWCI) composed of indicators of community structure in the diatom, macrophyte, and macroinvertebrate assemblages for 216 wetlands (n = 74 depressional marsh, n = 118 depressional forested, n = 24 flowing water forested wetlands). Depressional wetlands located along a human disturbance gradient throughout Florida were sampled for each assemblage as well as water and soil physical/chemical measures (e.g., specific conductance, pH, total phosphorus). Flowing water wetlands were sampled for macrophytes. The Landscape Development Intensity Index (LDI) was used to quantify the human disturbance gradient. In general, human disturbance in adjacent areas had the greatest impact on depressional herbaceous wetlands, followed by depressional forested wetlands. Forested flowing water wetlands (i.e., forested strands and floodplain wetlands) were less affected by local conditions, with most of their changes in biotic integrity correlated with alterations at the larger watershed scale. Predictive models of ecosystem change can be developed based on changes from human disturbance and effects on the community structure of biotic assemblages.

KEY WORDS: biological integrity; Florida Wetland Condition Index; human disturbance; Landscape Development Intensity index; wetland condition

INTRODUCTION

With even casual observation, it is apparent that ecosystem condition changes with increasing levels of human development intensity. The extent of change is observably related to the magnitude of human activity. An estimate from 2000 suggested 10-11% of Florida's land area was in urban development, a number that continues to increase (Reynolds 2001). Projections for future land conversion include an additional 2.8 million ha (7 million ac) converted to urban development in Florida through 2060 (Zwick and Carr 2006); however, such land use conversion is not limited to Florida. Projections for continued increases in world population from 6 billion in 1999 to 9 billion in 2042 (US Census Bureau 2007), coupled with the reality that more than half of the world population will be living in urban areas sometime in 2008 (UNFPA 2007) focus the debate on the condition of natural ecosystems within the context of a human land use development gradient.

Wetlands and aquatic resources are generally susceptible to human development activities. Because of differences in the extent of land use activities and the different spatial extent of wetlands, wetlands of different types are affected differentially depending on many variables including biotic and abiotic factors. While previous research has identified ecosystem responses to human induced changes such as increased nutrients (e.g., Nessel et al. 1982; Lemlich and Ewel 1984) or altered hydrology (e.g., Lugo and Brown 1986; Young et al. 1995) few have studied the complex response of ecosystems resultant from the combined effects of anthropogenic development. Our study aims at understanding the characteristic community structure of Florida's freshwater subtropical wetlands through an analysis of the diatom, macrophyte, and macroinvertebrate communities using the Florida Wetland Condition Index

(FWCI) and water and soil physical/chemical factors along a human disturbance gradient using the Landscape Development Intensity (LDI) index (Brown and Vivas 2005).

Florida Wetland Condition Index

Depending on classification scheme there are between 10 and 45 different types of wetlands in peninsular Florida (e.g., Cowardin et al. 1979; Brinson 1993), which makes a universal FWCI difficult. Instead, we focused on three types of freshwater wetland, including depressional marshes, depressional forested wetlands, and flowing water forested wetlands (i.e., forested strands and floodplain wetlands), as approximately 23% of Florida (3.6 million ha) has been classified as inland freshwater wetlands by the National Wetlands Inventory (Doherty et al. 2000). To describe wetland community structure, we developed a group of indices of biotic integrity (IBIs) called the Florida Wetland Condition Index (FWCI), which relies on biological indicators described through quantitative metrics. Karr and Chu (1997) defined metrics as biological attributes that have a consistent and predictable response to anthropogenic activities. Anthropogenic activities can alter the condition of wetland ecosystems by causing one or more of the following conditions: eutrophication, contaminant toxicity, acidification, salinization, sedimentation, burial, thermal alteration, vegetation removal, turbidity, shading, dehydration or inundation, and/or habitat fragmentation (Danielson 1998).

We have adopted the working definitions of the United States Environmental Protection Agency (USEPA) 2011 National Wetlands Condition Assessment (NWCA) for condition (i.e., current state of a resource compared to reference standards for physical, chemical, and biological characteristics) and ecological integrity (i.e., condition of an unimpaired ecosystem as measured by combined chemical, physical, and biological attributes). This builds upon the earliest definition of integrity from Karr and Dudley (page 55, 1981), defining integrity 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."

IBIs have been developed for a number of different species assemblages including diatoms (e.g., Fore and Grafe 2002, Atazadeh et al. 2007, Lane and Brown 2007); macrophytes (e.g., Galatowitsch et al. 1999, Mack 2001, Miller at al. 2006); macroinvertebrates (e.g., Kerans and Karr 1994, Barbour et al. 1996); fish (e.g., Schulz et al. 1999, Uzarski et al. 2005); and birds (O'Connell et al. 1998, Glennon and Porter 2005). Such indices have been widely applied to ecosystems throughout Europe (e.g., Kelly and Whitton 1998, Angermeier and Davideanu 2004, Ferreira et al. 2005) and the United States (e.g., Karr 1981, Fore and Grafe 2002, Southerland et al. 2007). The primary aim of such IBIs is to detect changes in abundance, structure, and diversity of the target species assemblage(s). For the FWCI, describing the community composition of three separate species assemblages (i.e., diatoms, macrophytes, and macroinvertebrates), water and soil physical/chemical measures, and characterizing the wetlands along a human disturbance gradient provided a comprehensive picture of the ecological condition of Florida wetlands.

Landscape Development Intensity Index

The underlying concept behind calculating the LDI, which quantifies the nonrenewable energy use per unit area in the surrounding landscape, stems from earlier works by Odum (1996), who pioneered emergy analysis for environmental accounting. Emergy is an environmental accounting term referring to expressing energy use in solar equivalents (Odum 1996). The LDI scale encompasses a gradient from completely natural to highly developed land use intensity, based on the nonrenewable empower density (Table 1), which is defined by the amount of nonrenewable energy use for a given land use (Brown and Vivas 2005).

The LDI 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 (e.g., 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 scores. Undeveloped land uses (e.g., wetlands, lakes, upland forests) have zero values, based on no use of nonrenewable energy in these ecosystems. Other studies have successfully correlated the LDI with wetland condition (e.g., Mack 2006).

Project Overview

Over 30 years ago, the federal Water Pollution and Control Act (later renamed the Clean Water 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) and assumed losses in ecosystem integrity if the criteria were exceeded (Danielson 1998). The 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).

Following this effort, the purpose of this study was to develop an FWCI based on field sampling and data processing from 1998-2005 and to characterize the community structure and response of several types of Florida's subtropical wetlands to human disturbance. Future application of the FWCI includes a biological monitoring and assessment tool to better understand, monitor, restore, and manage wetlands of different types – an important factor considering that while over half of Florida's 8.2 million ha of wetlands have been lost, 3.6 million ha of varying wetland types yet remain (Dahl 1990, Doherty et al. 2000). Attention has started to focus on the ecosystem services (e.g., water storage, aesthetics) wetlands provide according to wetland condition and landscape setting (e.g., Ehrenfeld 2004), and on tools available to assist in resource management decisions for wetland protection.

METHODS

Study Sites

Sample wetlands (n = 216 total; n = 74 depressional marsh, n = 118 depressional forested, n = 24 flowing water forested wetlands) were selected spatially throughout Florida (latitude 31.0ºN-26.0ºN; longitude 80.1ºW-87.5ºW, Figure 1). Field research spanned five growing seasons (roughly May-October) from 1999-2003 (n = 35 depressional marshes in 1999;

n = 39 depressional marshes in 2000; n = 72 depressional forested in 2001; n = 46 depressional forested in 2002; n = 24 flowing forested wetlands in 2003). Wetlands were categorized by generalized *a priori* land use categories as low development intensity wetlands (i.e., reference) or high development intensity wetlands (i.e., agriculture or urban) based on dominant surrounding land use.

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, upland) and wetland hydrologic indicators (e.g., lichen lines, moss collars) (Tobe et al. 1998; USDA 2002). At depressional wetlands, 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 (Lane et al. 2003; Reiss 2006). At flowing forested wetlands, four transects were established perpendicular to the hydrologic flow of the system extending from the wetland/upland boundary to the middle of the channelized flow at 25 m intervals (Reiss and Brown 2005). The first transect was located 25 m upstream of the nearest Florida Department of Environmental Protection (FDEP) Stream Condition Index (SCI) sampling point to correlate measures of wetland forest condition (i.e., the FWCI) adjacent to the channelized water course to in-stream measures of macroinvertebrate community condition (i.e., the SCI). As SCI values were only available for flowing water forested wetlands and not depressional wetlands, they have not been included in this manuscript.

Community Structure

The community structure of the diatom, macrophyte, and macroinvertebrate assemblages was characterized for development of the FWCI and soil and water parameters were measured and collected for depressional systems (i.e., herbaceous and forested isolated wetlands). Diatoms and macroinvertebrates and water parameters were only collected in hydrated (>10 cm) systems. Only macrophytic assemblages were sampled for flowing water systems, and no water or soil physical or chemical parameters were measured. Landscape indicators were measured for all systems, as described below.

Diatoms

At depressional marshes (n = 69), 10 epiphytic diatom samples were distributed at 10 unique locations in the wetland by snipping representative submersed and floating vegetation and placing the samples into a 3.8L Ziploc bag. Approximately 0.5 L of wetland water was added and the bag gently kneaded to mobilize the epiphyton. A 10 mL sample was obtained using a bulb pipette rinsed with wetland water and placed into a 100 mL composite sampling jar.

At depressional forested wetlands (n = 50), 10 benthic diatom samples were taken throughout the flooded portion of the wetland. A hollow cylinder was placed on the soil surface to isolate an area of substrate with an approximate surface area of 28 cm^2. A bulb pipette was swirled to loosen the top 0.5 cm layer at the soil surface-water interface, and a 10 mL sample was extracted. This was repeated 10 times throughout the wetland, resulting in a final composite sample volume of 100 mL.

All samples were preserved in the field using 5 mL of M3 fixative (Clesceri et al. 1995). Samples were homogenized prior to sub-sampling for laboratory identification. Sub-samples were digested following Hasle and Fryxell (1970), which removed the organic matter from the diatom frustules to aide in identification. Following rinsing with distilled water, the digested sub-samples were mounted on microscope slides using Naphrax (Northern Biological Supplies Limited, Ipswich, England). Following FDEP procedures (SOP #AB-03.1, available at www.dep.state.fl.us/labs/sop/), 250 diatom valves for epiphytic samples and 500 valves for benthic samples were then counted along microscope transects and identified to the lowest taxonomic level possible (usually species). If the target goal of valves was achieved, identification continued until the end of the transect; however, if the target goal could not be identified within a one hour time frame, identification was considered limited by the sample.

Macrophytes

Presence/absence vegetation data were collected along each belt transect, which were subdivided into 1 m wide by 5 m long quadrats. Living macrophytes rooted within each quadrat were identified to the lowest taxonomic level possible (usually species). Additional species characteristics were collected for use in developing potential biological indicator metrics, including growth form (i.e., aquatic, fern, grass, herb, sedge, shrub, tree, vine) and category (e.g., 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 2002; Wunderlin and Hansen 2003).

Macroinvertebrates

Macroinvertebrates were collected using standard U.S. 30-mesh D-frame dip-nets (n = 70 depressional marshes; n = 79 depressional forested wetlands). Twenty sweeps were collected proportionally divided according to different vegetation/habitat zones present. A single dip-net sweep was one net width and two net lengths covering an area of 0.5 m^2. The content of each sweep was deposited into a 3.8 L plastic jar. When all 20 sweeps were completed, the sample was preserved with buffered formalin at a rate of approximately 10% of the sample volume. Samples were delivered to the FDEP Central Laboratories for enumeration and identification. Following receipt, the material was sieved and washed following FDEP Standard Operating Protocols and placed on a pan with twenty-four numbered cells. Eight random cells were selected (a third of the sample) and placed in another numbered tray. From the second numbered tray a single sub-sample was taken and all organisms enumerated and identified. If fewer than 100 organisms were encountered, a second randomly chosen cell from the second numbered tray was selected and all organisms within that cell enumerated and identified and added to the total of the first count. Identifications by FDEP were made to the lowest taxonomic level possible (usually genus) following FDEP Standard Operating Procedure #IZ-06 (available at: ftp://ftp.dep.state.fl.us/pub/labs/lds/sops/3716.pdf).

Florida Wetland Condition Index

Metrics were developed for depressional wetlands using diatoms, macroinvertebrates, and macrophytes. Metrics of flowing water systems condition were developed using macrophytes. Metrics for the FWCI were developed using literature sources for diatoms (e.g., Bahls 1993, van Dam et al. 1994, USEPA 2002b, Fore and Grafe 2002), macrophytes (e.g., Wilhelm and Ladd 1988, Adamus 1996, Mack 2001), and macroinvertebrates (e.g., Barbour et al. 1996, Gerritsen and White 1997, USEPA 2002a). Iterative tests of the dataset were conducted to ascertain distinctions between *a priori* land use categories (Mann-Whitney U-test, $\alpha < 0.05$), and correlation (Spearman $r \geq |0.30|$, $\alpha < 0.05$) were made with the Landscape Development Intensity index (Brown and Vivas 2005). Metric evaluations were conducted with SAS (SAS Institute, Cary N.C., version 8.02), Minitab (Version 13.1, Minitab Statistical Software), and Analyse-It (Analyse-It software v. 1.67 for Microsoft Excel).

Sensitive and tolerant indicator taxa lists were determined using Indicator Species Analysis in PCORD (Version 4.1 from MJM Software, Gleneden Beach, Oregon). Many diatom metrics were based on established ecological indicator values from published tolerances to particular physical/chemical condition for each diatom species identified using a coded checklist of autecological guilds from Bahls (1993) and van Dam et al. (1994). Bahls (1993) categorized diatoms in Montana streams as very tolerant, tolerant, or sensitive to pollution based on initial classifications of Lange-Bertalot (1979) and Lowe (1974). van Dam et al. (1994) provided a list of attributes describing the tolerance of European diatoms to varying pH, salinity, dissolved oxygen content, nutrient enrichment, and saprobic conditions, which Fore (2004) and Fore and Grafe (2002) used to develop indicators for streams in Mid-Atlantic states and large rivers in Idaho, respectively.

Metrics were explored in five main categories: 1) Tolerance metrics determined with indicator species or established index values; 2) Autecological metrics that explored a previously described relationship between taxa and an environmental gradient; 3) Community structure metrics that explored taxonomic structure; 4) Community balance metrics with calculated values, such as evenness or dominance; and 5) Functional group metrics related to feeding behavior.

Further details are provided in Lane et al. (2003), Reiss (2004), Reiss and Brown (2005), Reiss (2006), Lane (2007), and Lane and Brown (2006, 2007).

Wetland FWCI scores were calculated as the percent of the reference standard condition based on the highest possible score for each FWCI, allowing for comparison among the different wetland types and species assemblages. A wetland receiving 100% for any given FWCI would be considered to have a high wetland condition; whereas lower percentages would reflect reduced wetland condition. Comparisons among *a priori* land use categories for low development intensity (i.e., reference wetlands) and high development intensity (i.e., agriculture and urban wetlands) using the Mann-Whitney U test (α=0.05) were completed using Analyse-It (Analyse-It software v. 1.67 for Microsoft Excel). For depressional wetlands with FWCI scores for all three assemblages, Spearman correlation was used to test the direction and strength of the relationship among two assemblages at a time using Minitab (Version 13.1, Minitab Statistical Software).

Water and Soil Measurements

Water and soil physical/chemical measurements were taken at depressional marshes (n = 71) and depressional forested wetlands (n = 75). 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 temperature and dissolved oxygen were taken in the field using a YSI 55 Dissolved Oxygen meter. Water samples analyzed for color (PCU; EPA 110.2), turbidity (NTU; EPA 180.1), pH (150.1), and specific conductance (umhos/cm; EPA 120.1) were placed on ice in the field and maintained at 4 ^{o}C. Samples analyzed for ammonia-nitrogen (mg N/L; EPA 350.1), nitrate and nitrite-nitrogen (mg N/L; EPA 353.2), total Kjeldahl nitrogen (mg N/L; TKN; EPA 351.2), and total phosphorus (mg P/L; TP; EPA 365.4) (USEPA 1993) were preserved in the field with acid (2 mL 1:1 H_2SO_4 per sample), placed in a cooler with the above sample, and shipped to the FDEP Central Chemistry Laboratory, Tallahassee, Florida (SOP #TA-06.04-5). Original data were archived in the FDEP STORET database and at the Howard T. Odum Center for Wetlands, University of Florida, Gainesville, Florida.

Soil cores were taken from depressional marshes (n = 74) and depressional forested wetlands (n = 118) 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. Cores for each site were homogenized into a composite sample per site per vegetation zone and stored on ice in the field until refrigeration was available. Soil processing for depressional marshes for total phosphorous (TP, mg/kg; USEPA 365.4), total nitrogen (TN), total carbon (TC), organic matter (OM), and pH was completed by the University of Florida Soil and Water Science Department. In the lab, pH readings were taken and subsamples were dried to a constant weight and finely ground (<0.2 mm). Percent TN and TC were obtained from the ground samples utilizing a Carlo-Erba NA-1500 CNS Analyzer (Haak-Buchler Instruments, Saddlebrook, NJ). Soil processing for depressional forested wetlands included soil moisture and organic matter (Gardner 1986) analyzed at the University of Florida Howard T. Odum Center for Wetlands, and TKN (USEPA 1993) and TP (USEPA 1979) processed through the Institute of Food and Agricultural Sciences Analytical Research Laboratory, Gainesville, Florida.

Comparisons among *a priori* land use categories for reference and agriculture depressional marshes (Mann-Whitney U test, α=0.05) and reference, agriculture, and urban depressional forested wetlands (Fisher's LSD pair wise comparison, $\alpha=0.05$) were completed using Analyse-It (Analyse-It software v. 1.67 for Microsoft Excel).

Landscape Development Intensity Index

The Landscape Development Intensity (LDI) index was used as an independent measure of anthropogenic activity in the landscape surrounding each study wetland to quantify the human disturbance gradient. Wetlands were delineated from aerial images and a surround zone was delineated in ArcView GIS 3.2 (Environmental Systems Research Institute, Inc. 1999). For depressional wetlands a 100 m zone was constructed around the edge of each wetland. For flowing forested wetlands a 200 m zone upstream of the downstream transect was constructed perpendicular to the channelized flow. The boundary of the flowing forested wetland 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. Land uses within the surrounding wetland zones were delineated based on digital orthophoto imagery available from Labins, The Land Boundary Information System from the FDEP (http://www.labins.org/2003/), and were updated during site visits to reflect any changes in land use since the images were recorded. The LDI index value was calculate for each wetland as:

$$LDI = 10 \bullet \log (empPD_{Total}/emPD_{Ref}) \qquad \text{(Eq. 1)}$$

where LDI is the Landscape Development Intensity index for a given wetland assessment area, in this case the zone around each wetland; $empPD_{Total}$ is the total empower density (including the background environment); and $emPD_{Ref}$ is the empower density of the background environment (Vivas 2007). The total empower density ($empPD_{Total}$) was calculated as:

$$emPD_{total} = emPD_{Ref} + \sum(LU_i \bullet emPD_i) \qquad \text{(Eq. 2)}$$

where LU_i is the fraction of the total surrounding wetland zone in land use i and $emPD_i$ is the nonrenewable empower density for land use i (Table 1). This is an updated modification of the LDI published by Brown and Vivas (2005), and was developed by Vivas (2007). The non-parametric Spearman correlation was used to test the direction and strength of the relationship between the FWCI and LDI index scores using Analyse-It (Analyse-It software v. 1.67 for Microsoft Excel).

RESULTS

Metric Development

Metrics developed independently for depressional marshes (diatoms, macrophytes, and macroinvertebrates; Lane et al. 2003), depressional forested wetlands (diatoms, macrophytes, and macroinvertebrates, Reiss 2004), and flowing water forested wetlands (macrophytes; Reiss and Brown 2005) are summarized below. Consult the original sources for additional information.

The diatom FWCI for depressional marshes and depressional forested wetlands shared seven metrics (Table 2). Sensitive and tolerant indicator taxa were specific to wetland type

(Lane et al. 2003, Reiss 2004, Reiss and Brown 2005). Five additional metrics shared by depressional wetlands were based on established ecological indicator values from published tolerances to particular physical/chemical condition for each diatom species (i.e., pollution tolerant, tolerant of elevated nitrogen, meso- and poly-saprobous, tolerant of elevated pH, and sensitive to low dissolved oxygen). The depressional marsh diatom FWCI had seven additional metrics based on published tolerances to particular physical/chemical conditions.

Three metrics (sensitive indicator taxa, tolerant indicator taxa, and exotic taxa) occurred within the macrophyte FWCI for all three wetland type (see Table 2). Two additional metrics for the depressional marshes included the ratio of annual to perennial species and the mean Coefficient of Conservatism (CC) score. The CC score was derived from the Floristic Quality Assessment index (FQAI) developed for Florida (Lane et al. 2003, Cohen et al. 2004, Reiss 2004, Reiss and Brown 2005). Additional metrics for both depressional forested and flowing forested wetland FWCIs were native perennial species and FQAI score (i.e., sum CC scores divided by total species richness). A sixth metric for the depressional forested wetlands was the wetland status metric, based on obligate and facultative wetland species.

The macroinvertebrate FWCI shared two metrics for depressional marshes and depressional forested wetlands, tolerant indicator taxa and sensitive indicator taxa, though indicator species lists were developed specific to wetland type. The depressional marsh macroinvertebrate FWCI had three additional metrics: predator functional feeding group; order Odonata (dragonflies and damselflies; class Insecta); and Orthocladinae tribe (midges, class Insecta, Chironomidae family). The depressional forested macroinvertebrate FWCI had four additional metrics: Florida Index (based on pollution tolerance); order Mollusca (including Bivalva, Gastropoda, and Plecypoda); family Noteridae (order Coleoptera, class Insecta, phylum

Arthropoda); and scraper functional feeding group (included macroinvertebrates that scrape periphyton from mineral and organic surfaces and those that browse or graze algal materials).

Florida Wetland Condition Index

Low development intensity wetlands (i.e., *a priori* reference wetlands) had significantly lower FWCI scores as a percent of the reference standard condition for depressional wetlands than high development intensity wetlands (i.e., *a priori* agriculture and urban wetlands) for all three species assemblages (Mann-Whitney U test, $\alpha < 0.001$); however, macrophyte FWCI scores as a percent of the reference standard condition were not significantly different for flowing forested wetlands (Figure 2). While the flowing forested wetlands only had a macrophyte FWCI, the flowing forested wetlands surrounded by land uses with high human development intensity appeared to have the highest FWCI scores as a percent of the reference standard condition when compared to depressional wetlands. FWCI scores for depressional marshes surrounded by high development intensity were lower than those for depressional forested wetlands or flowing forested wetlands.

A comparison of the diatom, macrophyte, and macroinvertebrate FWCI scores for depressional wetlands with FWCI scores for all three assemblages for depressional marshes (pair wise Spearman $r \geq 0.60$, $p < 0.001$) and depressional forested wetlands (pair wise Spearman $r \geq 0.63$, $p < 0.001$) revealed a strong correlation among the three different assemblages regarding wetland condition (Table 3). Instances when the diatom, macrophyte, and macroinvertebrate FWCI scores were not in agreement provided insight into the differential effects of temporal and spatial changes and the variability in response for each assemblage (Figure 3). For example, three depressional marshes surrounded by agricultural land use activities had fairly low wetland condition according to the diatom FWCI (range 2.1-6.4%) and the macrophyte FWCI (range 0.0-12.0%), yet had relatively high wetland condition based on the macroinvertebrate FWCI (range 54.0-80.0%). In another example, an *a priori* classified high development intensity depressional forested wetland showed relatively low wetland condition based on the diatom FWCI (21.6%) and the macroinvertebrate FWCI (39.0%), yet relatively high wetland condition based on the macrophyte FWCI (69.2%).

Soil and Water Parameters

A comparison of the water and soil physical/chemical parameters for depressional marshes and depressional forested wetlands showed that some measures were different between *a priori* land use categories (Table 4). Agricultural depressional marshes had significantly higher (Mann-Whitney U test, $\alpha=0.05$) water column pH, specific conductance, ammonia-nitrogen, total Kjeldahl nitrogen (TKN), and total phosphorus (TP) than reference depressional marshes. Among the three *a priori* land use categories for depressional forested wetlands, reference wetlands had significantly higher water column dissolved oxygen, lower pH, lower turbidity, and lower TP, than agriculture and urban wetlands (Fisher's LSD pair wise comparison, $\alpha=0.05$). Additionally, specific conductivity was significantly lower for reference than urban depressional forested wetlands. Similar to the depressional marshes, water column ammonia-nitrogen and TKN were significantly lower for reference than agriculture depressional forested wetlands.

Differences for soil measurements included lower soil pH and TP for reference than agriculture depressional marshes; higher soil moisture for reference than agriculture depressional forested wetlands; lower soil organic matter for agriculture than urban depressional forested wetlands; and lower TP for reference than agriculture depressional forested wetlands.

LDI

Diatom, macrophyte, and macroinvertebrate FWCI scores for depressional wetlands were significantly negatively correlated with LDI index scores (Spearman correlation $r \geq |0.25|$, $\alpha < 0.05$), where low LDI index numbers represent natural lands and higher numbers represent more developed human land use activities; however, the macrophyte FWCI scores for flowing forested wetlands were not significantly correlated with the LDI index scores (Table 5). The highest LDI for depressional marshes was 18.1; whereas the range for depressional forested wetlands was 0.0-34.1 and 0.0-29.3 for flowing forested wetlands.

DISCUSSION

Through the development of the FWCI, two main conclusions have been drawn. First, in general from a review of the water and soil physical/chemical measures and FWCI scores, human activity in adjacent areas has the greatest impact on depressional herbaceous wetlands, followed by depressional forested wetlands, as described below. Second, models of change in ecosystem condition can be developed based on changes in wetland condition from human disturbance and effects on the community structure of biotic assemblages.

Influence of Human Activity on Wetland Condition

In general, human disturbance in areas adjacent to wetlands had the greatest impact on depressional herbaceous wetlands, followed by depressional forested wetlands. Due to their spatial location, small size, and irregular distribution in the landscape, the number of depressional wetlands has been greatly reduced due to filling or destruction (Kirkman et al. 1998). The higher impact on depressional marshes is likely a factor of the ease of encroaching into and filling herbaceous wetlands and that many forested wetlands are bordered by marsh or wet prairie, affording some buffering from human land use development activities. Forested flowing water wetlands were less affected by local conditions, with most of their changes in ecological condition correlated with alterations at the larger watershed scale.

Depressional marshes, also called palustrine emergent marshes, account for greater than 34% of Florida wetlands (Doherty et al. 2000). Such wetlands are characteristically found in topographic lows in the landscape resulting in an accumulation of nutrients, metals, and toxins from up-slope sources as rainfall carries these contaminants into the wetland (Kirkman et al. 1998, Mitsch and Gosselink 2000). The accumulation of such contaminants can result in changes in structure, function, and processes and alteration in the floral and faunal composition can be detected. The shorter turnover time associated with herbaceous vegetation perhaps partially accounts for the greatest impacts being detected in depressional marshes.

An important aspect of our study is that depressional marshes occurred in the lowest LDI index range (LDI 0.0 – 18.1) as compared to the forested wetlands (LDI 0.0 – 34.1). This corresponds to the *a priori* land use categorization in which no depressional marshes were sampled in urban dominated land use settings. The strong influence of agricultural activities on depressional marshes and the lack of marshes in urban settings suggest that the direct and indirect effects from agricultural activities may have a larger influence on wetland condition than the higher energy intense urban land uses. For example, in the Prairie Pothole region Euliss and Mushet (1999) found that the pesticides and fertilizers used in agricultural practices have a direct effect on aquatic invertebrate community composition and function in wetland systems. Aside from fertilizers, additional nutrients are added to wetlands through agricultural activities such as grazing. In south Florida rangelands, Tanner et al. (1984) found that cattle spent the majority of time in the fall in herbaceous wetlands and in the summer in upland/herbaceous wetland ecotones, showing increased nutrient loading from cattle defecation/urination. Further studies show an influx of nutrient and sediment from cattle destruction of vegetative buffers (e.g., *Serenoa repens* (W. Bartram) Small, *Hypericum* spp.) fringing depressional marshes (Winchester et al. 1985, Vulink et al. 2000).

Changes in macrophyte and amphibian community structure and composition have also been noted through the direct physical impact of cattle trampling wetland vegetation and soil disturbance (Jansen and Healey 2003, Coles-Ritchie 2007) and to selective grazing by cattle (van Oene et al. 1999, Vulink et al. 2000). Though a recent study by Marty (2005) suggests that at reasonable densities, cattle may help maintain native plant and aquatic community diversity. Depressional forested agricultural wetlands typically received lower LDI index scores than depressional forested urban wetlands; however in the case of wetlands with active cattle grazing within the wetland ecosystem, FWCI scores were lower for grazed wetlands than for urban wetlands. Because of the longer response time of perennial macrophytes, particularly in the case of the woody mid- and over-story species, changes in the community composition of forested wetlands may be delayed.

Models of Change in Ecosystem Condition

To meet the goals of the Clean Water Act, methods to evaluate the relative condition of wetland ecosystems have to be developed. The FWCI is one such method that provides an assessment of wetland condition that can further be used to compare changes over time, restoration success, or appropriateness for wetland mitigation. While the FWCI can not be used to predict changes in the physical and chemical parameters of a wetland, its strength lies in providing an overview of ecological wetland condition 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: dissolved oxygen, pH, turbidity, specific conductivity, ammonia-nitrogen, TKN, and TP; and soil: pH, moisture, organic matter, and TP). When soil and water parameters were used to explain variation in the community composition of each assemblage, water and/or soil pH was universally identified as an important driving variable (Lane et al. 2003, Reiss 2004). Additionally, total phosphorus concentrations explained some of the variance in community composition. Perhaps land management strategies could focus on limiting activities in the surrounding landscape that influence changes to water and/or soil pH and total phosphorus loading to wetlands in order to promote protection of wetland ecological integrity.

In a study of diatom community measures for freshwater lakes, Pither and Aarssen (2005) noted that diatom richness was highest at circum-neutral pH levels, and decreased for both acidic and alkaline conditions. The pH range in our study was fairly acidic (depressional marshes range 3.4-8.1 pH; depressional forested wetlands range 3.8-7.8 pH) with *a priori* reference wetlands having lower pH values, which could have a direct effect on the concentration and form of bio-available constituents (USEPA 2002b). In a study of Maine wetlands of low (pH < 5.0) and high (pH > 5.5) pH, Woodock et al. (2005) found that macrophyte richness was significantly greater and richness of chironomid larvae (a common macroinvertebrate in the order Diptera) was lower in low pH wetlands. They further suggested that community structure was related to pH. That *a priori* agriculture and urban wetlands had higher pH values may suggest that runoff and transportation of agrochemicals into receiving water bodies such as wetlands may affect not only the nutrient levels in the receiving waters, but also the pH of the system (Fore and Grafe 2002).

Nutrient limitations are important direct considerations for diatom and macrophyte community composition; however, increases in nutrient levels can also influence changes throughout the trophic dynamics of wetland ecosystems. For instance, increased nutrient loading can affect dissolved oxygen levels within the water body as chemical and biological oxygen demand may surpass available oxygen levels. This would drastically affect the three assemblages sampled in our study (i.e., diatoms, macrophytes, macroinvertebrates), as well as most other organisms within the system. Chemical reactions to anoxia, such as the release of bound phosphorous from the soil, could further alter community composition. In a recent synthesis of published studies, Elser et al. (2007) suggest that simultaneous loading of nutrients (i.e., nitrogen and phosphorus) produces strongly positive synergistic responses resulting in changes in ecosystems and that ecosystem conservation and resource management should focus on nutrient reduction. As the three community assemblages included in this study (i.e., diatoms, macrophytes, macroinvertebrates) represent primary, secondary, and tertiary trophic classes, structural components of the wetland itself, and a portion of the landscape food web (e.g., food source for passerines and wading birds, reptiles and amphibians, small mammals), changes in wetland management strategies to decrease the effect of the controlling environmental variables (i.e., water and/or soil pH and total phosphorus) would be advantageous to meeting the goals of the Clean Water Act.

While agreement in the ranking of the biological condition of study wetlands using the FWCIs was anticipated, discrepancies among the ranking from the different assemblages may provide great insight into wetland condition as different species assemblages respond to changes in driving energies over different time scales. There was variation among the ranking of wetlands for the diatom, macrophyte, and macroinvertebrate FWCIs; though there were no extreme outliers when the three assemblages were compared. While the low development intensity (i.e., *a priori* reference) wetlands were generally differentiated from the high development intensity (i.e., agriculture and urban) wetlands, differences within the high development intensity *a priori* categories (i.e., agriculture and urban land uses) were not as apparent.

As a measurement of human development intensity and landscape modification, the LDI was expected to correlate with changes in community composition and gradients of water and soil physical/chemical measures. Results suggested that the LDI was able to couple the disparate effects of human landscape modification, such as altered hydrology (strongly affecting diatoms and macroinvertebrates) or trampling/selective herbivory (strongly affecting macrophytes), into a single value, which was assumed to be manifested by changes in community composition. Similarly, other studies have found a correlation among plant and animal communities to land use intensity (Jansen and Healey 2003, Lane and Brown 2006, Dorman et al. 2007). The benefit of using the LDI index as a measure of land use intensity is two-fold. First, the LDI index provided a means of remote GIS based assessment of wetland condition. Second, the LDI index provided an objective, repeatable measure of the human disturbance gradient.

Conclusion

The variable turnover times and sensitivities of the three assemblages (i.e., diatoms, macrophytes, macroinvertebrates) suggest that multiple assemblage specific multi-metric FWCIs have more merit than an FWCI based on a single assemblage, although there is a trade off in resources utilized with additional assemblages that need be considered by resource managers. Further, the FWCI would benefit from an expansion of assemblages (e.g., birds, amphibians), an expansion of freshwater wetland types (e.g., bayhead, Everglades flats, wet prairie), an expansion to coastal wetland communities (e.g., mangrove swamps, salt marshes), and/or further consideration of regionalization (though see Lane et al. 2003, Reiss 2004). Revisiting sites, increasing the dataset through the addition of new sites, and reevaluating the metrics developed based on additional sampling effort are the hallmarks of a thorough validation and calibration program, and should be incorporated into the next phase of the FWCI. With regular biological monitoring it may be possible to further explore the temporal effects of changing human development activities on wetland condition through changes in the community composition of different species assemblages. Perhaps through the application of assessment tools such as the FWCI and the LDI the remaining wetlands across the Florida landscape and beyond can be afforded protection from the assaults of human actions through an understanding of expected ecosystem change and identification of management or regulatory actions that could be implemented to fulfill the obligations of the Clean Water Act, specifically to protect and restore the chemical, physical, and biological integrity of the Nation's waters, including wetlands.

ACKNOWLEDGEMENTS

This project was funded in part by a Section 319 Nonpoint Source Management grant from the U.S. Environmental Protection Agency through a contract with the Florida Department of Environmental Protection (FDEP). Any opinions, findings, and conclusions or recommendations expressed in this material are those of the author(s) and do not necessarily reflect the views of the FDEP. FDEP staff provided support for this research (particularly R. Frydenborg, A. O'Neal, E. McCarron, J. Espy, T. Frick, J. Jackson, E. Miller, J. Richardson, and L. Wolf). R.J. Stevenson, Michigan State University, and M. Hein of Water and Air Research identified the diatoms. We thankfully acknowledge the input of 12 botanists (G. Anglin, A. Arcuri, D. Austin, K. Bradley, K. Burks, D. Hall, A. O'Neal, J. Poppleton, N. Raymond, B. Tatje, J. Tobe, and W. Zomlefer) who contributed to different surveys for the Floristic Quality Assessment Index. We acknowledge the systems ecology research group at the Howard T. Odum Center for Wetlands. Assistance in field-data collection, laboratory analysis, data entry, and/or feedback on statistical analyses from E. Bardi, S. Carstenn, M. Cohen, A. Davanzo, M. Friedman, K. Jackson, A. Moore, M. Murray-Hudson, J. Reilly-Brown, V. Rumancik, E. Spurrier, J. Surdick, M.B. Vivas, and M. Yonteck. We are particularly thankful and indebted to the land owners and managers who allowed access to the wetlands throughout Florida. Although this work was reviewed by the USEPA and approved for publication, it may not reflect official Agency policy.

REFERENCES

Adamus PR (1996) Bioindicators for assessing ecological integrity of prairie wetlands. EPA600-R- 96/082. National Health and Environmental Effects Laboratory, Western Ecology Division, United States Environmental Protection Agency, Corvallis, Oregon

Angermeier PL, Davideanu G (2004) Using fish communities to assess streams in Romania: initial development of an Index of Biotic Integrity. Hydrobiologia 511: 65-78

Atazadeh I, Sharifi M, Kelly MG (2007) Evaluation of the Trophic Diatom Index for assessing water quality in River Gharasou, western Iran. Hydrobiologia 589(1): 165-173

Bahls L (1993) Periphyton bioassessment methods for Montana streams. Water Quality Bureau, Department of Health and Environmental Science, Helena, Montana

Barbour MT, Gerritsen J, White JS (1996) Development of the Stream Condition Index (SCI) for Florida. A Report to the Florida Department of Environmental Protection, Stormwater and Nonpoint Source Management Section. Tetra Tech, Inc. Owing Mills, Maryland

Beck WM (1954) Studies in stream pollution biology I: a simplified ecological classification of organisms. Quarterly Journal of the Florida Academy of Science 17(4): 211-227

Brinson MM (1993) A hydrogeomorphic classification for wetlands. US Army Corps of Engineers, Waterways Experiment Station, Wetlands Research Program Technical Report WRP-DE-4. Available via http://el.erdc.usace.army.mil/wetlands/pdfs/wrpde4.pdf. Cited 14 March 2008.

Brown MT, Vivas MB (2005) A landscape development intensity index. Environmental Monitoring and Assessment 101: 289–309

Clesceri LS, Greenberg AE, Easton AD (1995) Standard methods for the examination of water and wastewater, 20th edition. American Public Health Association, Washington, DC.

Cohen MJ, Carstenn S, Lane CR (2004) Floristic quality indices for biotic assessment of depressional marsh condition in Florida. Ecological Applications 14(3): 784-794

Coles-Ritchie MC, Roberts DW, Kershner JL, Henderson RC (2007) Use of a wetland index to evaluate changes in riparian vegetation after livestock exclusion. Journal of the American Water Resources Association 43(3): 731-743

Cowardin LM, Carter V, Golet FC, LaRoe ET (1979) Classification of wetlands and deepwater habitats of the United States. US Department of the Interior, Fish and Wildlife Service, Washington, D.C. Jamestown, ND: Northern Prairie Wildlife Research Center Home Page. Available via http://www.npwrc.usgs.gov/resource/1998/classwet/classwet.htm (Version 04DEC98)

Dahl TE (1990) Wetlands - Losses in the United States, 1780's to 1980's. Fish and Wildlife Service Report to Congress, Washington DC

Danielson TJ (1998) Indicators for monitoring and assessing biological integrity of inland freshwater wetlands. EPA 843-R-98-002. Wetlands Division Office of Water, United States Environmental Protection Agency, Washington, D.C.

Doherty SM, Cohen MJ, Lane CR, Line L, Surdick J (2000) Biological criteria for inland freshwater wetlands in Florida: a review of technical and scientific literature (1990-1999). Report to the United States Environmental Protection Agency, Center for Wetlands, University of Florida, Gainesville, Florida

Dorman CF, Schweiger O, Augenstein I, Bailey D, Billeter R, de Blust G, DeFilippi R, Frenzel M, Hendrickx F, Herzog F, Klotz S, Liira J, Maelfait J, Schmidt T, Speelmans M, van Wingerden WKRE, Zobel M (2007) Effects of landscape structure and land-use intensity on similarity of plant and animal communities. Global Ecology & Biogeography 16(6): 774-787

Ehrenfeld JG (2004) The expression of multiple functions in urban forested wetlands. Wetlands 24(4): 719-733

Elser JJ, Bracken MES, Cleland EE, Gruner DS, Harpole WS, Hillebrand H, Ngai JT, Seabloom EW, Shurin JB, Smith JE (2007) Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. Ecology Letters 10(12): 1135-1142

Euliss NH, Mushet DM (1999) Influence of agriculture on aquatic invertebrate communities of temporary wetlands in the Prairie Pothole region of North Dakota, USA. Wetlands 19(3): 578-583

Ferreira MT, Rodríguez-González PM, Aguiar FC, Albuquerque A (2005) Assessing biotic integrity in Iberian rivers: development of a multimetric plant index. Ecological Indicators 5(2): 137-149

Fore LS (2004) Development and testing of biomonitoring tools for macroinvertebrates in Florida streams. Statistical Design, Seattle, Washington. A report for the Florida Department of Environmental Protection, Tallahassee, Florida

Fore LS, Grafe C (2002) Using diatoms to assess the biological condition of large rivers in Idaho (U.S.A.). Freshwater Biology 47: 2015-2037

Galatowitsch SM, Whited DC, Tester JR (1999) Development of community metrics to evaluate recovery of Minnesota wetlands. Journal of Aquatic Ecosystem Stress and Recovery 6: 217-234

Gardner WH (1986) Water content. In: Methods of soil analysis: part 1 physical and mineralogical methods, 2nd edition. American Society of Agronomy, Inc. and Soil Science Society of America, Inc., Madison, Wisconsin, p 493

Gerristen J, White J (1997) Development of a biological index for Florida lakes. A Report to the Florida Department of Environmental Protection. Tetra Tech, Inc. Owing Mills, Maryland

Glennon MJ, Porter WF (2005) Effects of land use management on biotic integrity: an investigation of bird communities. Biological Conservation 126(4): 499-511

Godfrey RK, Wooten JW (1981) Aquatic and wetland plants of the southeastern United States. University of Georgia Press, Athens, Georgia

Hasle GR, Fryxell GA (1970) Diatoms: cleaning and mounting for light and electron microscopy. Transactions of the American Microscopy Society 89 (4): 469-474

Jansen A, Healey M (2003) Frog communities and wetland condition: relationships with grazing by domestic livestock along an Australian floodplain river. Biological Conservation 109: 207-219

Karr JR (1981) Assessment of biotic integrity using fish communities. Fisheries 6: 21-27

Karr JR, Chu EW (1997) Biological monitoring and assessment: using multimetric indexes effectively. EPA 235-R-97-001. University of Washington, Seattle, Washington

Karr JR, Dudley DR (1981) Ecological perspectives on water quality goals. Environmental Management 5: 55-68

Kelly MG, Whitton BA (1998) Biological monitoring of eutrophication in rivers. Hydrobiologia 384: 55-67

Kerans BL, Karr JR (1994) A benthic index of biotic integrity (B-IBI) for rivers in the Tennessee valley. Ecological Applications 4(4): 768-785

Kirkman LK, Drew MB, West LT, Blood ER (1998) Ecotone characterization between upland longleaf pine/wiregrass stands and seasonally ponded isolated wetlands. Wetlands 18(3): 346-364

Lane CR (2007) Assessment of isolated wetland condition in Florida using epiphytic diatoms at genus, species, and subspecies taxonomic resolution. EcoHealth 4: 219-230

Lane CR, Brown MT, Murray-Hudson M, Vivas MB (2003) The wetland condition index: biological indicators of wetland condition for isolated depressional herbaceous wetlands in Florida. A Report to the Florida Department of Environmental Protection. Howard T. Odum Center for Wetlands, University of Florida

Lane CR, Brown MT (2006) Energy-based land use predictors of proximal factors and benthic diatom composition in Florida freshwater marshes. Environmental Monitoring and Assessment 117: 433-450

Lane CR, Brown MT (2007) Diatoms as indicators of isolated herbaceous wetland condition in Florida, U.S.A. Ecological Indicators 7: 521-540

Lange-Bertalot H (1979) Pollution tolerance of diatoms as a criterion for water quality estimation. Nova Hedwigia 54: 285-304

Lemlich SK, Ewel KC (1984) Effects of wastewater disposal on growth rates of cypress trees. Journal of Environmental Quality 13(4): 602-604

Lowe RL (1974) Environmental requirements and pollution tolerance of freshwater diatoms. U.S. Environmental Protection Agency, National Environmental Research Center, Office of Research and Development. EPA-670/4-74-005. Cincinnati, Ohio

Lugo AE, Brown SL (1986) The Ocklawaha River forested wetlands and their response to chronic flooding. In: Ewel KC, Odum HT (eds) Cypress swamps. University Presses of Florida, Gainesville, Florida, p 365

Mack J (2001) Vegetation Index of Biological Integrity (VIBI) for wetlands: ecoregional, hydrogeomorphologic, and plant community comparisons with preliminary wetland aquatic life use designations. Final Report to the United States Environmental Protection Agency Grant No. CD985875, Volume 1. Wetland Ecology Group, Division of Surface Water, Ohio Environmental Protection Agency, Columbus Ohio, USA. Available via http://www.epa.state.oh.us/dsw.wetlands/wetlands_bioasses.html Cited 10 April 2005

Mack J (2006) Landscape as a predictor of wetland condition: an evaluation of the Landscape Development Index (LDI) with a large reference wetland dataset from Ohio. Environmental Monitoring and Assessment 120(1-3): 221-241

Marty JT (2005) Effects of cattle grazing on diversity in ephemeral wetlands. Conservation Biology 19(5): 1626-1632

Miller SJ, Wardrop DH, Mahaney WM, Brooks RP (2006) A plant-based index of biological integrity (IBI) for headwater wetlands in central Pennsylvania. Ecological Indicators 6(2): 290-312

Mitsch WJ, Gosselink JG (2000) Wetlands. John Wiley and Sons, New York, New York

Nessel JK, Ewel KC, Burnett MS (1982) Wastewater enrichment increases mature pondcypress growth rates. Forest Science 28(2):400-403

O'Connell TJ, Jackson LE, Brooks RP (1998) A bird community index of biotic integrity for the mid-Atlantic highlands. Environmental Monitoring and Assessment 51: 145-156

Odum HT (1996) Environmental accounting: emergy and environmental decision making. John Wiley and Sons, New York

Pither J, Aarssen LW (2005) The evolutionary species pool hypothesis and patterns of freshwater diatom diversity along a pH gradient. Journal of Biogeography 32: 503-513

Reiss KC (2004) Developing biological indicators for isolated forested wetlands in Florida. Dissertation, University of Florida

Reiss KC (2006) Florida Wetland Condition Index for depressional forested wetlands. Ecological Indicators 6: 337-352

Reiss KC, Brown MT (2005) -Pilot Study- The Florida Wetland Condition Index (FWCI): preliminary development of biological indicators for forested strand and floodplain wetlands. A Report to the Florida Department of Environmental Protection. Howard T. Odum Center for Wetlands, University of Florida, Gainesville, Florida

Reynolds, JE (2001) Urbanization and land use change in Florida and the South. Current Issues Associated with Land Values and Land Use Planning, Proceedings of a Regional Workshop SERA-IEG-30. Sothern Rural Development Center (SRDC) Series #220. Available via http: http://srdc.msstate.edu/publications/220.htm. Cited 12 March 2008

Schulz EJ, Hoyer MV, Canfield DE, Jr. (1999) An index of biotic integrity: a test with limnological and fish data from sixty Florida lakes. Transactions of the American Fisheries Society 128: 564-577

Southerland MT, Rogers GM, Kline MJ, Morgan RP, Boward DM, Kazyak PF, Klauda RJ, Stranko SA (2007) Improving biological indicators to better assess the condition of streams. Ecological indicators 7(4): 751-767

Tanner GW, Sandoval LD, Martin FG (1984) Cattle behavior on a South Florida range. Journal of Range Management 37(3): 248-251

Tobe JD, Craddock Burks K, Cantrell RW, Garland MA, Sweeley ME, Hall DW, Wallace P, Anglin G, Nelson G, Cooper JR, Bickner D, Gilbert K, Aymond N, Greenwood K, Raymond N (1998) Florida wetland plants: an identification manual. Florida Department of Environmental Protection, Tallahassee, Florida

USDA United States Department of Agriculture, Natural Resource Conservation Service (2002) The PLANTS Database, Version 3.5. National Plant Data Center, Baton Rouge, Louisiana, USA. Available via http://plants.usda.gov. Cited 2001-2004

USEPA United States Environmental Protection Agency (1979) 351.2. Nitrogen, Kjeldahl, total (colorimetric; semi-automated digester, AAII). CAS # N Nitrogen 7727-37-9. EPA/600/4-79/020 Methods for the chemical analysis of water and wastes. Available via http://www.epa.gov/epahome/index/ Cited 15 June 2004

USEPA United States Environmental Protection Agency (1990) Feasibility Report on Environmental Indicators for Surface Water Programs. Office of Water Regulations and Standards and Office of Policy, Planning and Evaluation, Washington, D.C.

USEPA United States Environmental Protection Agency (1993) 365.1 Determination of phosphorus by semi-automated colorimetry methods for the determination of inorganic substances in environmental samples. EPA-600-R-93-100. Available via http://www.epa.gov/epahome/index/ Cited 10 January 2004

USEPA United States Environmental Protection Agency (2002a) Methods for Evaluating Wetland Condition: Developing and Invertebrate Index of Biological Integrity for Wetlands. EPA-822-R-02-019. Office of Water, U.S. Environmental Protection Agency. Washington, D.C.

USEPA United States Environmental Protection Agency (2002b) Methods for Evaluating Wetland Condition: Using Algae to Assess Environmental Conditions in Wetlands. EPA-822-R-02-021. Office of Water, U.S. Environmental Protection Agency. Washington, D.C.

UNFPA United Nations Population Fund (2007) State of the world population 2007: unleashing the potential of urban growth. Available via http://www.unfpa.org/swp/. Cited 2 March 2008

US Census Bureau (2007) Total midyear population for the world: 1950-2050. Data from International Data Base, updated 07-16-2007. Available via http://www.census.gov/ipc/www/idb/worldpop.html. Cited 12 March 2008

Uzarski DG, Burton TM, Cooper MJ, Ingram JW, Timmermans STA (2005) Fish habitat use within and across wetland classes in coastal wetlands of five Great Lakes: development of a fish-based index of biotic integrity. Journal of Great Lakes Research 31(suppl 1): 171-187

van Dam H, Mertens A, Sinkeldam J (1994) A coded checklist and ecological indicator values of freshwater diatoms from the Netherlands. Netherlands Journal of Aquatic Ecology 28: 117-133

van Oene H, Mieleke van Deursen EJ, Berendse F (1999) Plant-herbivore interaction and its consequence for succession in wetland ecosystems: a modeling approach. Ecosystems 2: 122-138

Vivas MB (2007) Development of an index of landscape development intensity for predicting the ecological condition of aquatic and small isolated palustrine wetland systems in Florida. Dissertation, University of Florida

Vulink JT, Drost HJ, Jans L (2000) The influence of different grazing regimes on Phragmites- and shrub vegetation in the well-drained zone of a eutrophic wetland. Applied Vegetation Science 3: 73-80

Wilhelm G, Ladd D (1988) Natural Area Assessment in the Chicago Region. In: Transactions of the 53rd North American Wildlife and Natural Resource Conference, Louisville, Kentucky, Wildlife Management Institute, Washington D.C., p 361

Winchester BH, Bays JS, Higman JC, Knight RL (1985) Physiography and vegetation zonation of shallow emergent marshes in southwestern Florida. Wetlands 5: 99-118

Woodcock T, Longcore J, McAuley D, Mingo T, Bennatti CR, Stomborg K (2005) The role of pH in structuring communities of Maine wetland macrophytes and Chironomid larvae (Diptera). Wetlands 25(2): 306-316

Wunderlin RP (1998) Guide to the vascular plants of Florida. University Press of Florida, Gainesville, Florida,

Wunderlin RP, Hansen BF (2003) Atlas of Florida vascular plants. From Landry SM, Campbell KN, application development. Florida Center for Community Design and Research. Institute for Systematic Botany, University of South Florida. Available via http://www.plantatlas.usf.edu/ Cited 2003-2008

Young PJ, Keeland BD, Sharitz RR (1995) Growth response of baldcypress [*Taxodium distichum* (L.) Rich.] to an altered hydrologic regime. American Midland Naturalist 133(2): 206-212

Zwick, PD, MH Carr (2006) Florida 2060: a population distribution scenario for the state of Florida. Geoplan Center at the University of Florida. Available via http://www.1000friendsofflorida.org/Publications/main.asp. Cited 10 March 2008

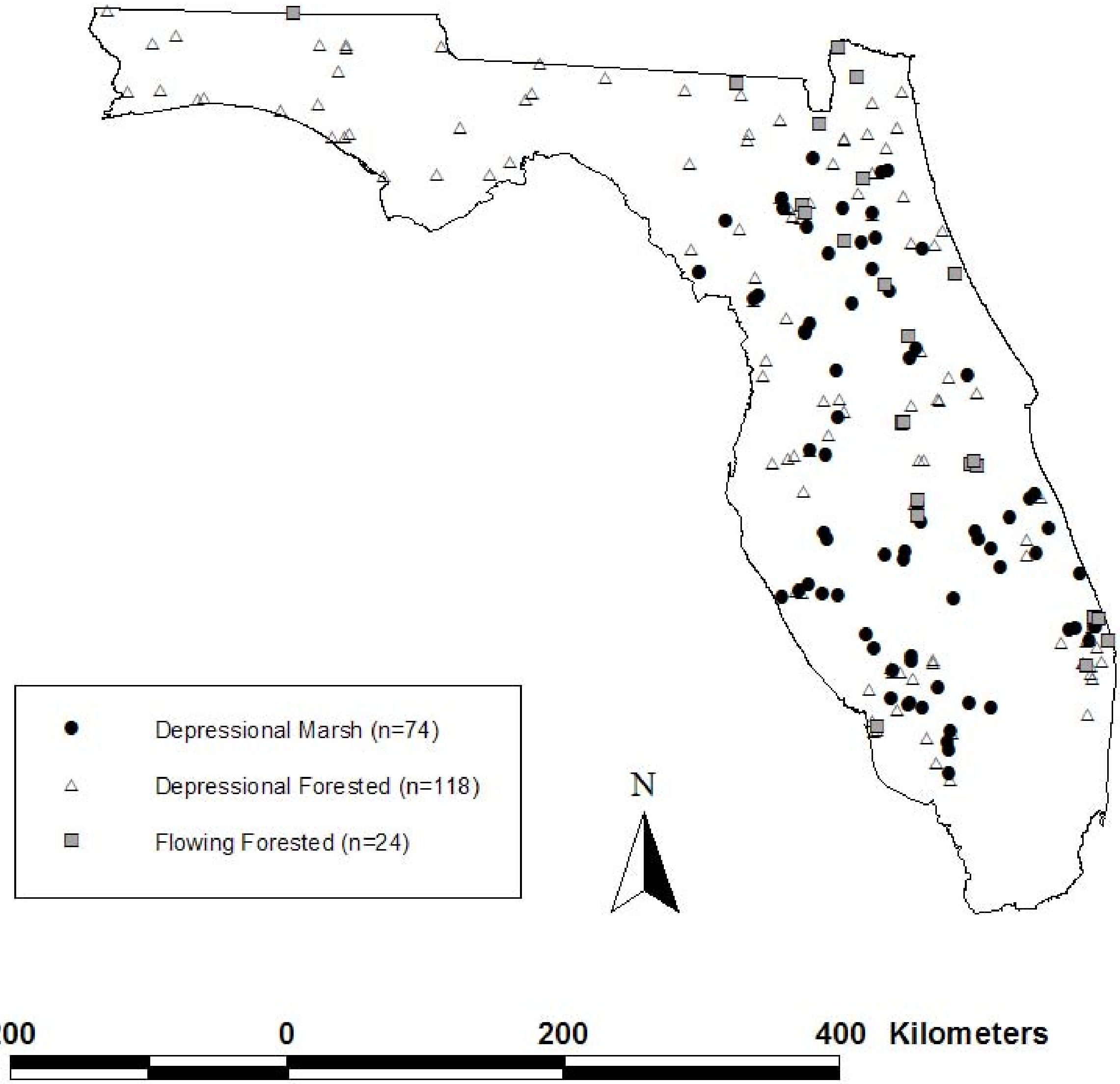
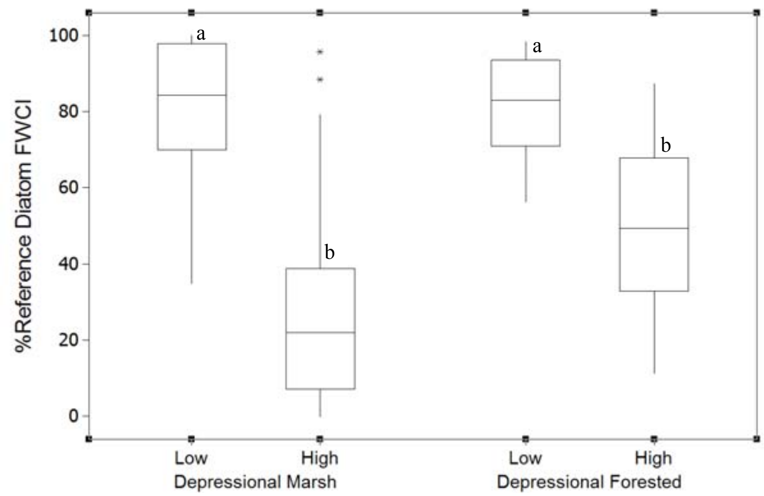

Depressional Marsh (n=74)
Depressional Forested (n=118)
Flowing Forested (n=24)
N
200
0
200
400 Kilometers

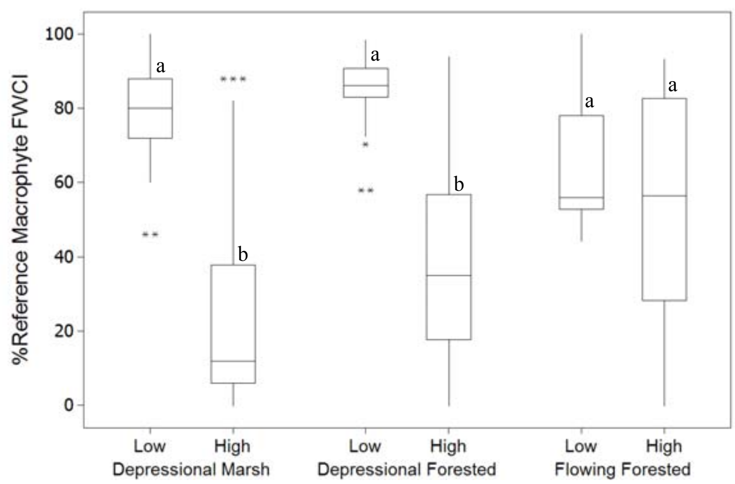
Figure 1.

(a)



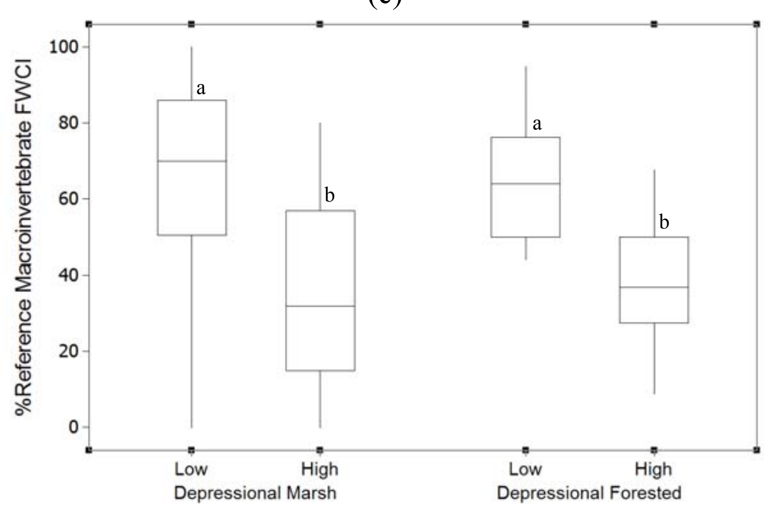
%Reference Diatom FWCI
0
20
40
60
80
100
a
b
a
b
Low
High
Depressional Marsh
Low
High
Depressional Forested

(b)



%Reference Macrophyte FWCI
0
20
40
60
80
100
a
b
a
b
a
a
Low
High
Depressional Marsh
Low
High
Depressional Forested
Low
High
Flowing Forested

(c)



%Reference Macroinvertebrate FWCI
0
20
40
60
80
100
a
b
a
b
Low
High
Depressional Marsh
Low
High
Depressional Forested

Figure 2

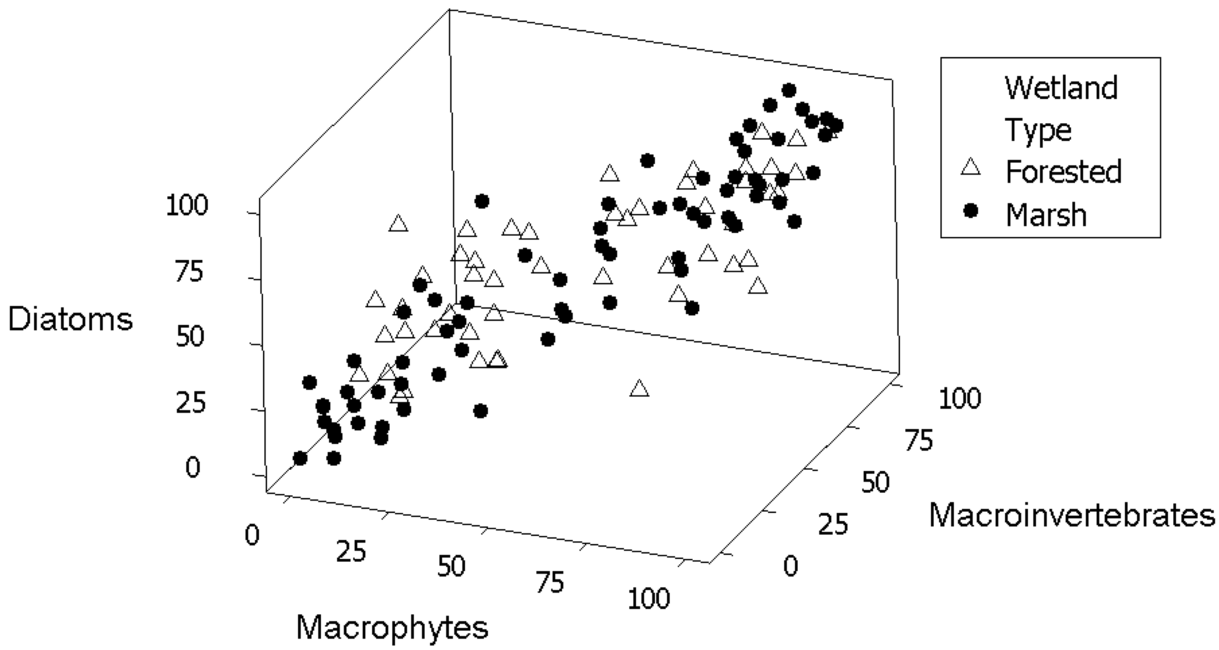

Wetland
Type
Forested
Marsh
Diatoms
100
75
50
25
0
0
25
50
75
100
Macrophytes
100
75
50
25
0
Macroinvertebrates

Figure 3.

Figure Legends

Figure 1. Location of 216 subtropical Florida wetlands (n = 74 depressional marsh, n = 118 depressional forested, n = 24 flowing forested wetlands).

Figure 2. Box plot comparisons of wetland Florida Wetland Condition Index (FWCI) scores as a percent of the reference standard wetland condition for the (a) diatom, (b) macrophyte, and (c) macroinvertebrate community assemblages. Wetlands are distinguished by wetland type (depressional marsh, depressional forested, flowing forested) and by *a priori* land use classification including wetlands characterized by natural lands and low development intensity (Low) in the surrounding landscape (i.e., reference wetlands) and wetlands characterized by high human development intensity (High) in the surrounding landscape (i.e., agriculture and/or urban wetlands). The box portion of the plot represents the interquartile range from the first quartile (bottom horizontal line) to the third quartile (top horizontal line). The line drawn through the box represents the median. The vertical whiskers represent the highest and lowest values in the dataset, excluding outliers which are marked with an asterisk (*). Boxes with similar letters within each wetland type were not significantly different (Mann-Whitney U test, $\alpha=0.05$).

Figure 3. Comparison of wetlands with FWCI scores for all three assemblages (i.e., diatoms, macrophytes, macroinvertebrates) for depressional marshes (●) and depressional forested wetlands (Δ). Axes represent the percent of the FWCI score for each wetland as a percentage of the reference standard wetland condition.

Table 1. Non-renewable areal empower intensity (sej/ha/yr) used to calculate the Landscape Development Intensity (LDI) index. The LDI column represents the calculated LDI if all of the land use within the zone used to calculate the LDI is of a single land use.

Land Use	Non-Renewable Areal Empower Intensity (E15 sej/ha/yr)	LDI
Natural Land / Open Water	0.00	0.00
Pine Plantation	0.51	1.00
Open Space / Recreational - Low Intensity	0.52	1.02
Unimproved Pastureland (with livestock)	0.53	1.04
Improved Pasture (no livestock)	2.02	3.07
Pasture (with livestock) - Low Intensity	3.38	4.34
Pasture (with livestock) - High Intensity	5.93	6.03
Open Space / Recreational - Medium Intensity	6.06	6.10
Citrus	7.76	6.94
General Agriculture	15.10	9.38
Row Crops	20.30	10.53
Agriculture (dairy farm) - High Intensity	50.40	14.25
Open Space / Recreational – High Intensity	123.00	18.02
Single Family Residential – Low Density	197.50	20.05
Transportation – 2 Lane Highway	308.00	21.97
Single Family Residential – Medium Density	658.33	25.25
Single Family Residential – High Density	921.67	26.71
Transportation 4 Lane Highway – Low Intensity	2533.70	31.10
Multi-Family Residential - Low Density	4213.33	33.30
Institutional	4042.20	33.12
Transportation 4 Lane Highway – High Intensity	5020.00	34.06
Commercial (Comm Strip) - Low Intensity	5173.40	34.19
Industrial	5210.60	34.23
Commercial (Mall) - High Intensity	8372.40	36.28
Multi-family residential (High rise)	12771.67	38.12
Central Business District (Avg 2 stories)	16150.30	39.14
Central Business District (Avg 4 stories)	29401.30	41.74

Table 2. Florida Wetland Condition Index (FWCI) metrics for depressional marshes, depressional forested wetlands, and flowing forested wetlands.

Diatom FWCI Metrics

Shared Metrics	**Depressional Marshes**	**Depressional Forested**	**Flowing Forested**
Sensitive Indicator Taxa	Sensitive Elevated pH (1)*	No additional metrics	No diatom FWCI
Tolerant Indicator Taxa	Sensitive Elevated Salinity (1)*		
Pollution Tolerant (1)*	Tolerate Elevated Salinity (3)*		
Tolerate Elevated Nitrogen (3)*	Sensitive Elevated Nitrogen (1)*		
Meso- & Polysaprobous (4)*	Tolerate Low Dissolved Oxygen (4)*		
Tolerant Elevated pH (3)*	Oligotrophic (1&2)*		
Sensitive Low Dissolved Oxygen (1)*	Eutrophic (5)*		

Macrophyte FWCI Metrics

Shared Metrics	**Depressional Marshes**	**Depressional Forested**	**Flowing Forested**
Sensitive Indicator Taxa	Ratio of Annual to Perennial Species	Native Perennial Species	Native Perennial Species
Tolerant Indicator Taxa	Mean Coefficient of Conservatism Score	Floristic Quality Assessment Index (FQAI) Score	Floristic Quality Assessment Index (FQAI) Score
Exotic Species		Wetland Indicator Status	

Macroinvertebrate FWCI Metrics

Shared Metrics	**Depressional Marshes**	**Depressional Forested**	**Flowing Forested**
Sensitive Indicator Taxa	Predators	Florida Index Score^	No macroinvertebrate FWCI
Tolerant Indicator Taxa	Odonata	Mollusca	
	Orthocladiinae	Noteridae	
		Scrapers	

*Numbers in parentheses refer to Bahls (1993) classes for pollution tolerance and van Dam et al. (1994) classes for nitrogen, saprobity, pH, dissolved oxygen, and salinity.

^Florida Index score from Beck 1954; USEPA 2002a.

Table 3. Spearman correlations (Spearman's *r*) among depressional wetlands with FWCI scores for all three assemblages. All correlations were significant at $\alpha < 0.001$.

Depressional Marsh Systems (n=69)		
	% Reference Diatom FWCI	% Reference Macrophyte FWCI
% Reference Diatom FWCI	1.00	0.89
% Reference Macrophyte FWCI	0.89	1.00
% Reference Macroinvertebrate FWCI	0.60	0.67

Depressional Forested Systems (n=50)		
	% Reference Diatom FWCI	% Reference Macrophyte FWCI
% Reference Diatom FWCI	1.00	0.70
% Reference Macrophyte FWCI	0.70	1.00
% Reference Macroinvertebrate FWCI	0.63	0.73

Table 4. Water physical/chemical measures for Florida depressional marshes and depressional forested wetlands by *a priori* land use categories (i.e., reference, agriculture, urban). Values represent mean (standard deviation).

Wetland Water	Depressional Marsh*		Depressional Forested^		
	Reference n = 34	Agriculture n = 37	Reference n = 28	Agriculture n = 21	Urban n = 26
Dissolved Oxygen (mg/L)	4.5 (4.7)[a]	4.1 (3.3)[a]	2.9 (1.7)[a]	1.6 (0.9)[b]	1.9 (1.1)[b]
Temperature (°C)	25.8 (5.1)[a]	26.9 (5.5)[a]	26.2 (2.8)[a]	25.2 (1.9)[a]	24.9 (2.4)[a]
Color (PCU)	289 (226)[a]	598 (1338)[a]	285 (178)[a]	346 (204)[a]	198 (129)[a]
pH	5.1 (1.0)[a]	6.3 (1.0)[b]	5.2 (1.2)[a]	6.2 (0.8)[b]	6.4 (1.0)[b]
Turbidity (NTU)	2.1 (2.0)[a]	163.7 (920.0)[a]	3.8 (4.2)[a]	17.7 (40.7)[b]	9.5 (11.9)[b]
Specific Conductance (μmhos/cm)	76 (85)[a]	247 (288)[b]	81 (48)[a]	136 (134)[ab]	231 (175)[b]
Ammonia-Nitrogen (mg N/L)	0.17 (0.57)[a]	1.38 (7.88)[b]	0.15 (0.33)[a]	0.33 (0.57)[b]	0.19 (0.27)[ab]
Nitrate/Nitrite-Nitrogen (mg N/L)	0.01 (0.02)[a]	0.01 (0.04)[a]	0.09 (0.37)[a]	0.01 (0.01)[a]	0.02 (0.03)[a]
Total Kjeldahl Nitrogen (mg N/L)	1.90 (1.28)[a]	5.37 (17.74)[b]	1.93 (1.24)[a]	3.17 (2.20)[b]	1.84 (1.06)[ab]
Total Phosphorus (mg P/L)	0.04 (0.03)[a]	1.75 (7.45)[b]	0.08 (0.11)[a]	0.81 (1.38)[b]	0.23 (0.26)[b]
Wetland Soil	Depressional Marsh*		Depressional Forested^		
	Reference n = 36	Agriculture n = 38	Reference n = 37	Agriculture n = 40	Urban n = 41
pH	4.8 (0.9)[a]	5.3 (0.9)[b]			
Soil Moisture (%)			61 (20)[a]	46 (17)[b]	55 (22)[ab]
Organic Matter (%)	23 (25)[a]	24 (24)[a]	40 (25)[ab]	30 (17)[a]	41 (28)[b]
Total Carbon (%)	15 (15)[a]	16 (14)[a]			
Total Nitrogen (%)	45 (183)[a]	30 (101)[a]			
Total Kjeldahl Nitrogen (mg N/g soil)			6.76 (3.68)[a]	5.53 (3.30)[a]	6.70 (4.75)[a]
Total Phosphorus (mg P/g soil)	0.12 (0.20)[a]	0.29 (0.24)[b]	0.38 (0.28)[a]	0.91 (1.27)[b]	0.53 (0.31)[ab]

*Categories with similar letters were not significantly different (Mann-Whitney U test, α=0.05).
^Categories with similar letters were not significantly different (Fisher's LSD pair wise comparison, α=0.05).

Table 5. Spearman correlations (r) between the Landscape Development Intensity (LDI) index with scores for each assemblage (i.e., diatom, macrophyte, and macroinvertebrate) for the Florida Wetland Condition Index (FWCI).

	Depressional Marshes			Depressional Forested			Flowing Forested		
FWCI	n	r	p	n	r	p	n	r	p
Diatom	69	-0.28	0.02	50	-0.59	<0.001			
Macrophyte	74	-0.29	0.01	118	-0.61	<0.001	24	-0.18	ns
Macroinvertebrate	70	-0.25	0.03	79	-0.52	<0.001			

ns = not significant ($\alpha > 0.05$)