DEVELOPING BIOLOGICAL INDICATORS FOR ISOLATED FORESTED WETLANDS IN FLORIDA

By

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This document is dedicated to my family and friends:
Lena, Lucky, Casey, Nana, Ben, Krista, Wendy, and Mike.

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By

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The Wetland Condition Index (WCI) provided a quantitative measure of the biological integrity of isolated forested wetlands in Florida. Environmental parameters and community composition of the diatom, macrophyte, and macroinvertebrate assemblages were sampled in 118 isolated forested wetlands throughout Florida to answer the overall question: can changes in the biotic components of pondcypress wetlands (such as the community composition of the diatom, macrophyte, and macroinvertebrate assemblages) be related to changes in development intensity in the landscape immediately adjacent to and surrounding them. While richness, evenness, and diversity measures were not sensitive to changes in landscape development intensity, biological indicators along with physical and chemical parameters were useful in defining biological integrity.

Differences in diatom, macrophyte, and macroinvertebrate community composition were explored in nonmetric multidimensional scaling (NMS) ordinations. Water-column

pH was correlated with the community composition of all 3 assemblages. Each assemblage was used to construct the WCI for isolated forested wetlands in Florida, which included 19 total metrics (7 diatom metrics, 6 macrophyte metrics, and 6 macroinvertebrate metrics). All metrics were significantly correlated (Spearman's correlation coefficient, $p < 0.05$) with the Landscape Development Intensity (LDI) index, a measure of the use of nonrenewable energy in the surrounding landscape.

While the WCI 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 WCI should not be used as a surrogate for wetland value, but rather as an objective, quantitative means of comparing changes in community composition along gradients of landscape development intensity. In the future, an integrative multi-metric multi-assemblage WCI could be constructed for wetlands throughout the state, with lists of indicator species and metric scores dependent on Florida ecoregions and specific to wetland type.

CHAPTER 1
INTRODUCTION

With even casual observation, it is apparent that ecosystems change with increasing levels of human development. The extent of change is observably related to the magnitude of human activity. While previous research has identified ecosystem responses to human induced changes such as increased nutrients (Nessel et al. 1982; Lemlich and Ewel 1984; Devall 1998) or altered hydrology (Marois and Ewel 1983; Lugo and Brown 1986; Young et al. 1995) few have studied the amalgamated response of ecosystems resultant from the combined effects of anthropogenic development. This is especially true for wetlands located within urban settings. Our study aimed at understanding the effects of landscape development (from anthropogenic activities), and focused on one specific ecosystem type, the isolated pondcypress dome.

Statement of the Problem

Pondcypress domes are isolated depressional forested wetlands. These historically nutrient-poor ecosystems occur throughout Florida and the southeastern United States coastal plain. Primary driving energies include inputs from rainfall and localized run-off. Because of their position in the landscape, pondcypress domes have specific hydrologic and nutrient regimes that regulate species composition. Conversion of lands adjacent to and surrounding isolated wetlands to more intensively managed land uses may alter the driving energies of the wetland; and since driving energies are fundamental to ecosystem organization, changes in inflows may influence the rates and direction of processes and ultimately system organization. The changes resultant from modification of the driving

energies may in turn be manifest in detectable differences in the biotic components. Change is further defined as a detectable difference between current conditions from the reference condition. Reference condition is defined as the condition of wetlands surrounded by undeveloped landscapes and without apparent human induced alterations. While it may be difficult to discern the exact causal agent of such changes, it may be possible to detect differences in various biotic assemblages of ecosystems and relate them to surrounding land use intensity.

Our study focuses on understanding the changes in the biotic assemblages that occur in isolated pondcypress wetlands resulting from different land uses in the landscapes surrounding them. The major question addressed in this dissertation is: can the changes in biotic components of pondcypress wetlands (such as the community composition of the diatom, macrophyte, and macroinvertebrate assemblages) be related to changes in development intensity in the landscape immediately adjacent to and surrounding the wetlands. From this main question several secondary questions arise. Are there differences among pondcypress domes surrounded by different land uses? If change does occur, what describes the change in pondcypress communities? Biological signals may be apparent in the diatom, macrophyte, and macroinvertebrate community composition and in the abiotic components. Can the presence of particular diatoms, macrophytes, or macroinvertebrates be used as an indication of change? Are differences in physical and chemical water and soil criteria detectable in wetlands surrounded by different land uses? The extent of change of wetland biota may be an indicator of change in community structure and thus indicative of what has been termed ecosystem integrity (Karr and Dudley 1981). By analyzing changes in multiple assemblages and relating

them to the intensity of land use surrounding each wetland, new insights concerning the effects of land use on wetland structure and integrity may be generated.

Defining Ecosystem Integrity

Determining ecosystem integrity through the use of biological indicators requires an accepted definition of integrity. Karr and Dudley (page 55, 1981) defined 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." This definition requires two things: a definition of the natural habitat (or reference condition) and appropriate regionalization. Gerristen et al. (2000) concur that biological assessment relies on a characterization of the reference condition.

Historical Perspective

Over 30 years ago, the passage of the Water Pollution and Control Act (later referred to as the Clean Water Act, 1972) required states to "restore and maintain the chemical, physical, and biological integrity of the Nation's waters" (USEPA 1990). This legislation included establishing water-quality standards for all waters within state boundaries, including wetlands. 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 assuming losses in ecosystem integrity if the criteria were exceeded (Danielson 1998a).

Water Quality Criteria

There are several shortcomings in deriving ecosystem integrity based on exceeding established limits for chemical and physical parameters. Such criteria have been

considered rudimentary in their ability to reflect more than the temporal concentration within a water body (Karr 1993). For instance, the use of toxicity parameters for determining ecosystem integrity may falsely indicate high ecosystem integrity simply because a single toxicity parameter went undetected. This same water body could have undesirable levels of other nontarget toxics or metals; or be physically altered so that it no longer resembles a fully functioning water body (Karr and Chu 1997). Furthermore, chemical and physical sampling may not occur during specific loading events and may therefore incompletely describe the biological and ecological condition of the system. Adams (2002) points out that other environmental factors (such as sedimentation, alterations to habitat, varying temperature and oxygen levels, and changes in ecological aspects like food availability and predator-prey relationships) are not reflected with chemical criteria alone. James and Kleinow (1994) add that different organisms respond in different ways to the amount, persistence, and exposure of xenobiotics (chemical compounds otherwise foreign to an organism); and single-valued chemical and physical criteria of water quality may overlook important biological implications.

Alternatively, biological indicators integrate the spatial and temporal effects of the environment on resident organisms, and are suitable for assessing the possible effects of multifaceted changes in aquatic ecosystems (Adams 2002). Adams (2002) and Karr and Chu (1997) note that biological indicators signal changes in the environment that might otherwise be overlooked or underestimated by methods that depend on chemical criteria alone. The underlying support for using biological indicators is that organisms have an intricate relationship with their environment, which reflects current and cumulative ecosystem conditions (Karr 1981). Biological indicators reflect chemical exposure and

also integrate changes in the community composition of the ecosystem (from physical, chemical, and biological changes) (Adams 2002).

The United States Environmental Protection Agency (USEPA) recognized the potential of biological criteria to assess water-quality standards and in the late 1980s required states to use biological indicators to accomplish the goals of the Clean Water Act (USEPA 1990). In effect, biological assessment has evolved into one of the standard monitoring tools of water resource-protection agencies over the past 2 decades (Gerristen et al. 2000). Biological criteria and monitoring programs through the USEPA have been created for lakes and streams throughout the United States (Barbour et al. 1996a; Karr and Chu 1999; Gerristen et al. 2000), and more recently efforts to assess wetland condition have been initiated (USEPA 2002a). Currently wetland bioassessment programs are in place or being developed in 14 states (USEPA 2003).

Biological Indicators of Ecosystem Integrity

Biological monitoring to assess ecosystem condition has been applied widely in ecological research. One trend in biological monitoring has led to the development of indices of biological integrity (often referred to as IBIs), for different species assemblages including diatoms (Fore and Grafe 2002); macrophytes (Galatowitsch et al. 1999a; Gernes and Helgen 1999; Mack 2001; Lane 2003); macroinvertebrates (Kerans and Karr 1994; Barbour et al. 1996b); fish (Schulz et al. 1999); and birds (O'Connell et al. 1998). Such indices have been applied to ecosystems throughout the world including in Europe (Kelly and Whitton 1998); Japan (Mack 2001); widely throughout the United States (Karr 1981; Lenat 1993; Fore and Grafe 2002; Lane et al. 2002); and is beginning in Australia by J.E. Ling of the Royal Botanical Gardens, University of Western Sydney. The primary aim of biological monitoring is to detect changes in abundance, structure,

and diversity of target species assemblages. Danielson (1998a) notes that biological signals are effective mainly because biological monitoring incorporates changes from various collective constant or pulsing sources.

Many studies have created multimetric indices of biological condition, incorporating individual metrics into a quantitative value of community condition or ecosystem integrity. Karr and Chu (1997) defined metrics as biological attributes that have a consistent and predictable response to anthropogenic activities. Anthropogenic activities can alter the integrity 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 1998a).

Diatoms as biological indicators

Diatoms are unicellular or colonial algae with siliceous bodies. They are an important basis of wetland food webs; and because they drive many wetland functions through their primary production, they are considered valuable in wetland biological assessment (Cronk and Fennessy 2001; Stevenson 2001). The USEPA (2002b) described six fundamental ecosystem functions of algae within water bodies:

- Providing a food source for organisms at higher trophic levels
- Contributing to nutrient and biogeochemical cycling
- Oxygenating the water column
- Regulating water chemistry
- Creating habitat for other organisms
- Acting as physical barriers to erosion

Because of their rapid turnover times, algae have a short response time to perturbations including nutrient and toxic contaminant inputs; and algae continue production throughout the winter, taking advantage of available nutrients when higher

plants are dormant (Cronk and Fennessy 2001). While the standing stock of algae is typically lower than that of the macrophyte assemblage, algae can constitute a higher proportion of primary productivity within an aquatic community (Cronk and Fennessy 2001). These factors and others contribute to the utility of the algal assemblage for biological assessment. Among the main advantages of using algae for biological assessment include the high diversity within the algal community (particularly of diatom species) in aquatic environments (Stevenson 2001). There is a depth of knowledge as to the sensitivity of many species to different environmental conditions based on their autecological characteristics, including two published tables of autecological relationships by van Dam et al. (1994) and Bahls (1993). Additionally, the rapid-response time of the algal community to changing environmental conditions is a major advantage to their use as biological indicators (Cronk and Fennessy 2001), as well as an overlap in the species present among different aquatic environments (van Dam et al. 1994; Fore and Grafe 2002). Diatoms in particular are considered easy to identify based on well-established taxonomic keys of their decay resistant siliceous structures (Stevenson et al. 1999), and there are well-tested protocols for sampling aquatic habitats (Goldsborough 2001).

Few significant disadvantages of using algae in biological assessment methodologies have been described. Among them is the necessity of a high-powered microscope for identification (Doherty et al. 2000); although identification is relatively easy, and good taxonomic keys have been established (Stevenson et al. 1999). Additionally, while most algae are not readily motile, wind and current translocation can complicate assessments based on scales of anthropogenic activity in the surrounding

landscape. The third noted disadvantage includes natural seasonal variations in abundance and morphology (Vymazal and Richardson 1995).

Overall, algae are considered a valuable assemblage for assessing the biological condition of wetlands. In particular, diatoms are noted as a useful assemblage (Stevenson 2001; Doherty et al. 2000). Previous research has correlated the response of diatoms in streams, lakes, and wetlands, to changes in surrounding land use and to changes in water-column characteristics including nutrient loading (van Dam et al. 1994); pH (Pan and Stevenson 1996); heavy metal loading (Charles et al. 1996); and saprobity levels (Lange-Bertalot 1979). The USEPA (2002b) reported that diatoms are one of the most commonly used assemblages in aquatic ecosystems for assessing biological, physical, and chemical conditions.

Research correlating changes in the diatom community composition to changes in their aquatic environment has been undertaken for isolated freshwater marshes in Florida (Lane 2003); large rivers in Idaho (Fore and Grafe 2002); streams (Barbour et al. 1999; Winter and Duthie 2000; Munn et al. 2002); depressional wetlands in Michigan (Pan and Stevenson 1996; Stevenson et al. 1999); prairie potholes (Adamus 1996); Mid-Atlantic streams (Pan et al. 1996); the Florida Everglades (Raschke 1993); and Florida lakes (Whitmore 1989). Most of the quantitative biological indices based on diatom community composition have been constructed for rivers and streams (Bahls 1993; Stevenson and Wang 2001).

In a study of isolated freshwater marshes in peninsular Florida, Lane (2003) incorporated fourteen metrics into the Diatom Index of Wetland Condition (DIWC). These included tolerant indicator species, sensitive indicator species, diatoms requiring

low pH, requiring low salinity, tolerant of high salinity, tolerant of high pH, sensitive to high nitrogen, tolerant of high nitrogen, requiring elevated dissolved oxygen, tolerant of low dissolved oxygen, meso- and polysaprobous diatoms, characteristic of oligotrophic environments, characteristic of eutrophic environments, and pollution-tolerant diatoms. Environmental parameters correlating with diatom community composition included specific conductivity, water-column pH, water ammonia-nitrogen concentration, water total Kjeldahl nitrogen (TKN) concentration, water total phosphorus concentration (TP), soil pH, and soil TP (Lane 2003).

In a study of lotic (relating to moving water) systems in the Mid-Atlantic States, Pan et al. (1996) found the strongest correlation with diatom community composition and changes in water-column pH. Additional water-column parameters correlating with diatom community composition included turbidity, aluminum concentration, chlorine concentration, TP, total suspended solids, and dissolved organic-carbon concentration. Similarly, in a study of emergent permanently flooded floodplain wetlands in western Kentucky, Pan and Stevenson (1996) found significant correlations between diatom community composition and 8 water variables, including alkalinity, conductivity, ammonia-nitrogen concentration, pH, silicon concentration, nitrate-nitrogen concentration, chlorine concentration, and TP. Another study of streams in Michigan also correlated the response of diatom community composition to different land use and water physical and chemical parameters (Stewart et al. 1999). These findings found that the algal assemblage was useful in reflecting changes in the water environment.

Macrophytes as biological indicators

Wetland macrophytes are defined as aquatic emergent, submergent, or floating plants growing in or near water (USEPA 1998); and are described as distinguishing

landscape features. The spatial distribution of macrophytes in the landscape occurs according to a multitude of factors, including hydroperiod, water chemistry, and substrate type, as well as other broader factors such as available seed source and climate. Fennessy et al. (2001) state that the community composition of wetland macrophytes typifies the physical, chemical, and biological wetland dynamics in time and space. Macrophytes play a vital role in supporting the structure and function of wetlands by providing food and habitat for other assemblages including algae, macroinvertebrates, fish, amphibians, reptiles, birds, and mammals.

Macrophyte populations can be used as diagnostic tools to assess other aspects of the wetland environment. Crowder and Painter (1991) state that a lack of macrophytes where they are otherwise expected to grow suggests reduced wildlife populations from lack of food or cover; and/or water quality concerns such as toxic chemical constituents, increased turbidity, or increased salinity. In contrast, an overgrowth of particular macrophytes may signify increased nutrient loading (USEPA 1998).

Many advantages of studying macrophytes as indicators of wetland condition have been noted, including their large, obvious size; ease of identification, to at least some useful taxonomic level; known response to toxicity tests; and general lack of ability to move to avoid unfavorable conditions (Danielson 1998a; Cronk and Fennessy 2001). Additionally, macrophytes readily respond to changes in nutrient, light, toxic contaminant, metal, herbicide, turbidity, water, and salt levels. They can also be sampled in the field with transects, or from the office with aerial photography; and well-established field methods of sampling macrophytes exist (USEPA 2003). Furthermore, the USEPA (2003) states that macrophytes do not require laboratory analysis, can easily

be used for calculating simple abundance metrics, and are superb integrators of environmental condition. In general, macrophytes represent a useful assemblage for describing wetland condition (Mack 2001). Schindler (1987) alleged that macrophytes can provide a more integrated picture of wetland function than static measures such as nutrient cycling, productivity, decomposition, or chemical and physical composition.

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

Previous biological assessment studies have included unique and varied macrophyte metrics dependent on wetland type and bioregion. Lane (2003) calculated 5 macrophyte metrics for inclusion in the marsh Vegetative Index of Wetland Condition (VIWC). The 5 core metrics of the VIWC included tolerant indicator species, sensitive indicator species, exotic species, annual to perennial ratio, and average Coefficient of Conservatism score. In Minnesota, Vegetative Indices of Biotic Integrity (V-IBIs) have been created for 8 wetland types (Galatowitsch et al. 1999a). Macrophyte metrics varied depending on wetland type, and included 15 metrics for high-order river floodplain wetlands, 12 for low-order river floodplain wetlands, 8 for mid-order river floodplain

wetlands, 7 for calcareous littoral wetlands, 6 for noncalcareous littoral wetlands, 7 for wet prairie-sedge meadows, 4 for forest glacial marshes, and 1 single metric for prairie glacial marshes. Another comprehensive biological assessment used to construct multimetric indices of biotic integrity for Ohio wetlands was designed by Mack et al. (2000). Separate biological multimetric indices were developed for emergent, forested, and shrub wetlands. Twelve metrics were incorporated, including *Carex* species, dicot species, shrub species, hydrophyte species, Rosaceae species, Floristic Quality Assessment Index, tolerant species, intolerant species, invasive graminoids, shrub density, small-tree density, and maximum importance value.

The Floristic Quality Assessment Index (FQI) has been included in many of the multimetric indices created for the macrophyte assemblage. The concept of FQI was developed by Wilhelm and Ladd (1988) for vegetation around Chicago, Illinois. This method of scoring plant species based on expert botanist opinion has been used in Michigan (Herman et al. 1997), Ohio (Andreas and Lichvar 1995; Fennessy et al. 1998; Mack 2001), Ontario (Francis et al. 2000), North Dakota (Mushet et al. 2002), and Florida (Lane 2003; Cohen et al. 2004). The FQI provides a quantitative means of assessing the fidelity of a plant to a particular environment through the Delphi technique (Kent 2000), where individual botanists assign coefficients to each species, and then reevaluate their scores based on the group mean scores. This technique assumes that the collective decision by a group of expert botanists is more accurate than the professional judgment of one individual (Kent 2000).

Macroinvertebrates as biological indicators

Biological assessment based on the macroinvertebrate assemblage has been widely applied for indications of environmental quality, and often more specifically water

quality (Lenat 1993; Cummins and Merritt 2001). Invertebrates are participants in many fundamental ecological processes, including the breakdown of organic matter and recycling of nutrients; and invertebrates are a vital component of the food web, making up a large portion of the diets of other organisms (such as fish, amphibians, and birds) (Cummins and Merritt 2001; Helgen 2001). As such, Voshell (2002) recognizes that freshwater invertebrates have been used more often than any other group of organisms for assessing freshwater ecosystems.

A great deal is known about the specific ecology of lotic (relating to moving waters such as streams) macroinvertebrates; less is known about those found primarily in lentic (relating to still waters such as wetlands) environments. Williams and Feltmate (1992) noted that, while not well studied, the communities of aquatic insects in wetlands include species from most of the major aquatic groups. The community composition of wetland macroinvertebrates differs from that of flowing waters, because of differences in substrate, dissolved-oxygen level in the water column, hydroperiod, and annual water fluctuations. Macroinvertebrates have been useful indicators of environmental condition in streams; and Karr and Chu (1997) speculate macroinvertebrates also may be appropriate indicators of environmental integrity in wetlands.

Since 1997, the use of the macroinvertebrate assemblage for biological assessments has been initiated in 48 states for lakes and streams (Karr and Chu 1999). Macroinvertebrate-based wetland biological assessment methodologies have been initiated in many states, including Florida (Lane 2003), Minnesota, Montana, North Dakota, and Ohio (Danielson 1998b). Within the state of Florida and throughout the southeastern Coastal Plain, ecological research on the macroinvertebrate community has

included many ecosystem types from isolated marshes of peninsular Florida (Kushlan 1990; Lane 2003); isolated wetlands in south Florida (Stansly et al. 1997); amphipods in southeastern wetlands (Pickard and Benke 1996); nontidal wetlands (Batzer and Wissinger 1996); sloughs of the northern Everglades (Rader and Richardson 1992; Rader and Richardson 1994); floating islands in Orange Lake in north central Florida (Haag et al. 1987); and bottomland hardwood swamps (Wharton et al. 1982).

Doherty et al. (2000) conclude that the structure and function of the macroinvertebrate community accurately reflects the biological condition of a wetland, and that the macroinvertebrate community composition changes in predictable ways with increased human influence. Because wetland macroinvertebrates complete part or all of their lives in the wetland, they are directly exposed to conditions in the wetland water and soils (Merritt and Cummins 1996; Helgen 2001). Also, because of the short length of their life cycles (compared to most macrophytes and vertebrates), Stansly et al. (1997) noted that macroinvertebrates respond quickly to changes in the physical, chemical, or biological parameters of their host environment. Their quick response time, reliance on water (both for the water quality and duration of inundation), and ease of collection make macroinvertebrates a favorable assemblage for use as biological indicators. Noted disadvantages to using macroinvertebrates include the amount of time and knowledge necessary for identification to lower taxonomic levels (Cummins and Merritt 2001).

The Florida Department of Environmental Protection (FDEP) has initiated the development of biological indices based on the macroinvertebrate assemblage for freshwater bodies in Florida. Macroinvertebrate-based biological indices have been created for isolated marshes through the Macroinvertebrate Index of Wetland Condition

(MIWC; Lane 2003), for Florida streams through the Stream Condition Index (SCI; Barbour et al. 1996a; Fore 2003), for surface waters in south Florida canals using SCI protocol (Snyder et al. 1998), for the evaluation of restoration in the Kissimmee River Basin (Merritt et al. 1996), and in freshwater lakes through the Lake Condition Index (LCI; Gerristen and White 1997). Different core metrics comprise each multimetric biological index.

Lane (2003) incorporated 5 core metrics as biological indicators of wetland condition for isolated marshes in the MIWC, including sensitive taxa, tolerant taxa, predators, Odonata, and Orthocladiinae. The SCI was developed with 7 core metrics, including taxa richness, EPT richness (Ephemeroptera, Plecoptera, Trichoptera), Florida Index, percent dominant, Chironomid taxa richness, suspension-filter feeders, and Diptera (Barbour et al. 1996b). Similarly, the LCI incorporated 7 metrics, including taxa richness, Shannon diversity, Hulbert index, ETO taxa (Ephemeroptera, Trichoptera, and Odonata), percent dominance, filter feeders, and gatherers (Gerristen and White 1997).

Numerous studies have documented the response of the benthic macroinvertebrate community to anthropogenic activities. Two primary areas of research include changes in trophic state, and additions of stormwater and wastewater. Gerristen and White (1997) and Cairns and Pratt (1993) found that the benthic macroinvertebrate community composition responded to changes in trophic status. In the northern Everglades, Rader and Richardson (1994) found that macroinvertebrates responded to nutrient enrichment with a greater number of Coleopteran species present (especially those in the families Hydrophilidae and Dystcidae) in nutrient-enriched and intermediate areas than in nonenriched areas. With shifts in trophic status, the structure of other assemblages also

changed, affecting the benthic macroinvertebrate community composition. For example, Adamus and Brandt (1990) found that shading from dense stands of emergent vegetation altered the distribution among functional feeding groups by limiting the production of benthic algae (thus favoring detritovores over grazers). De Szalay and Resh (1996) similarly found that increased shading caused fine particulate organic matter to settle out, making rich detritus accessible to support a large population of benthic macroinvertebrate detritovores.

While adding stormwater and wastewater alters the natural hydrology of an isolated wetland, it also increases the inflow of nutrients, sediments, and toxic metals. Harris and Vickers (1984) found that adding wastewater to cypress domes shifted the macroinvertebrate community toward a less-complex trophic structure. Similarly, when wastewater was directed into Florida cypress domes, *Lemna* spp. (duckweed) mats covered the water surface, blocking sunlight from the water column, and creating anoxic conditions (Dierberg and Brezonik 1984). This reduced the diversity and biomass of benthic invertebrates, leaving only a few pollution-tolerant organisms (Brightman 1984). In Florida streams, Barbour et al. (1996b) found that the occurrence of tubificid oligochaetes increased with organic enrichment.

Other studies have focused on the effects of adding stormwater to freshwater wetlands. Freshwater marshes (in Savannas Preserve State Park, Florida) receiving stormwater additions showed increased phosphorus levels, lowered oxygen levels, increased water-column pH and hardness, and a change in the macroinvertebrate community toward pollution-tolerant species and those intolerant of the typical acidic and oligotrophic environment (Graves et al. 1998). Barbour et al. (1996b) reported that some

chironomids of the family Orthocladiinae, including those in the genus *Cricotopus*, were found to be tolerant of metal pollution; while other Orthocladiinae, including *Rheocricotopus* spp. and *Corynoneura* spp., were thought to be sensitive to metal pollution in Florida streams.

Adding wastewater or stormwater also alters the water level and hydroperiod of cypress domes. Toth (1993) reported a response of the macroinvertebrate community to water-level manipulation in a Kissimmee River demonstration project. Hydroperiod had a distinct influence on community composition; as some macroinvertebrates either temporarily relocated or coped with behavioral and biological adaptations to changing water conditions. Macroinvertebrates with adaptations to wetland hydroperiods demonstrate both behavioral and physiological adaptations for draw-down conditions. For example, in south Florida hydric flatwoods, Gore et al. (1998) found that *Crangonyx* spp. and several other aquatic insects burrowed into moist sediments to avoid desiccation.

Some macroinvertebrates are thought to be indicative of water level and seasonality, with *Caenis* spp., *Anaz* spp., *Libellula* spp., and *Pantala* spp. indicative of persistent water; some Chironomus, some Tanytarsus, *Beardius* spp., and *Zavreliella marmorata*, indicative of permanent standing water; and *Ablabesmyia rhamphe* grp., *Krenopelopia* spp., and *Tanytarsus* sp. g. indicative of ephemeral wetlands (Doherty et al. 2000). Stansly et al. (1997) concluded that in isolated wetlands of south Florida, the presence of macroinvertebrates with long life cycles or predatory behavior may indicate hydroperiod stability. Snyder et al. (1998) found that macroinvertebrates with comparatively short life cycles that are capable of rapid colonization were typical of

canals surrounded by urban land uses, whereas the occurrence of macroinvertebrates with longer life cycles were more common in canals surrounded by more natural landscapes.

Review of Isolated Freshwater Forested Wetlands

Throughout the world, wetlands have been categorized in many different ways (Keddy 2000; Kent 2000; Mitsch and Gosselink 1993). Probably the most widely applied classification system in North America is that by Cowardin et al. (1979). Our study focused on what Cowardin et al. (1979) categorized forested palustrine wetlands. More specifically, the wetlands targeted in our study are called pondcypress domes with reference to Vernon (1947), who was the first to name these systems after their characteristic silhouette in the landscape. Cypress domes range in size from less than 1 hectare to more than 10 hectares (Wharton et al. 1977). There is however much application of ecological theories and research from all types of wetlands worldwide, as wetland species share mechanisms to deal with a fluctuating environment by adapting to periodically inundated and often anaerobic conditions.

Figure 1-1 is a systems diagram of the primary components, sources, and flows of a typical cypress dome. Symbols and terminology are from Odum (1994). Appendix A provides an overview of the energy circuit language and symbols used in Figure 1-1. Inflows into the cypress dome are limited to sunlight, wind, water (rain, surface run-off, groundwater), and recruitment of plant and animal species. Water inflow comes almost entirely from rainwater, both as a direct input and as "run-off" from a relatively small watershed, as such these wetlands are often termed "isolated" due to their somewhat limited hydrologic connections.

Standing water is present in most cypress domes much of the year (Odum 1978; Mitsch and Gosselink 1993), and some cypress domes have deep central pools staying

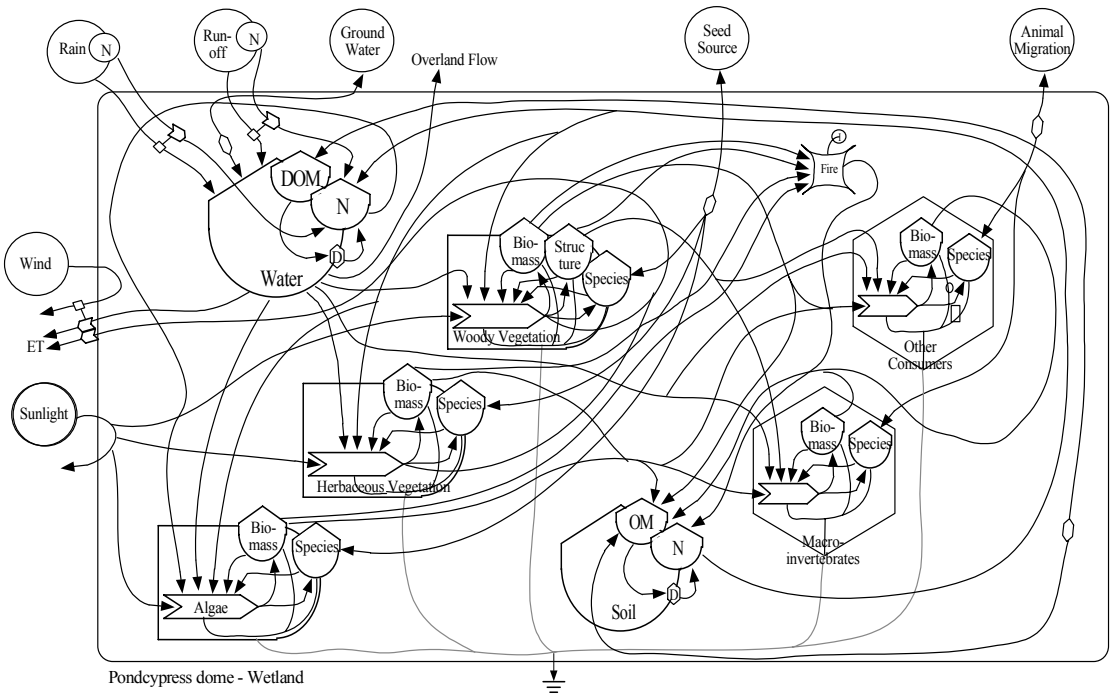


Figure 1-1. Systems diagram showing major sources, storages, and flows of a cypress dome.

wet year round, while others go dry annually. There is also variation in maximum flooding depth and length of standing water between years, which reflects the larger scale climatic and physiogeographic influences to cypress domes. Typically, the wettest period is summer and the driest spring and fall (Mitsch and Gosselink 1993). *Taxodium ascendens* (pondcypress) is the principal tree species in cypress domes (Devall 1998). Other tree species associated with *Taxodium ascendens* include: *Nyssa biflora* (black gum), *Pinus* spp. (many southern pines), *Acer rubrum* (red maple), and *Magnolia virginiana* (sweetbay) (Wilhite and Toliver 1990; Devall 1998). Pomdcypress trees characteristically dominate the center, with the pomdcypress along the edge in competition with other species that are less tolerant of flooded conditions. There is a greater likelihood of fire and a larger number of seedlings in the drier edges (Odum 1978).

The isolated cypress dome is characterized by a tolerance of low nutrient levels and intermittent fire (Brandt and Ewel 1989), with major system inputs limited to rainfall and surface inflows (Mitsch and Gosselink 1993). These ecosystems are considered successionally stable, but may be replaced by other ecosystems with changing environmental conditions, such as decreased water levels (Devall 1998). In drained cypress domes in northern Florida, Marois and Ewel (1983) found an increase in the densities of hardwood and shrub species.

Pondcypress and the typical co-dominate black gum are deciduous, shedding their leaves from October to December. Ewel (1984) and Wharton et al. (1977) found that the perpetuation of cypress domes depends on fluctuating water levels, with a dry period without standing water necessary for cypress generation and higher water levels some time during the year necessary to prevent germination of more terrestrial faster-growing pines and hardwoods that are not tolerant of standing water. Altering the typical hydroperiod of a cypress dome would effect species composition, resulting in encroachment of terrestrial species in a drained cypress dome and a lack of regeneration in an artificially flooded dome.

Anthropogenic activities in the surrounding up-slope landscape can create a wide array of changes to the inflows of these otherwise isolated systems. Figure 1-2 presents a systems diagram of the primary components, sources, and flows of a cypress dome embedded in a developed landscape with variable land uses. The systems boundary reflects a 100 m buffer zone around the isolated wetland. Some of the potential alterations to pondcypress wetlands located within developed land uses include changes

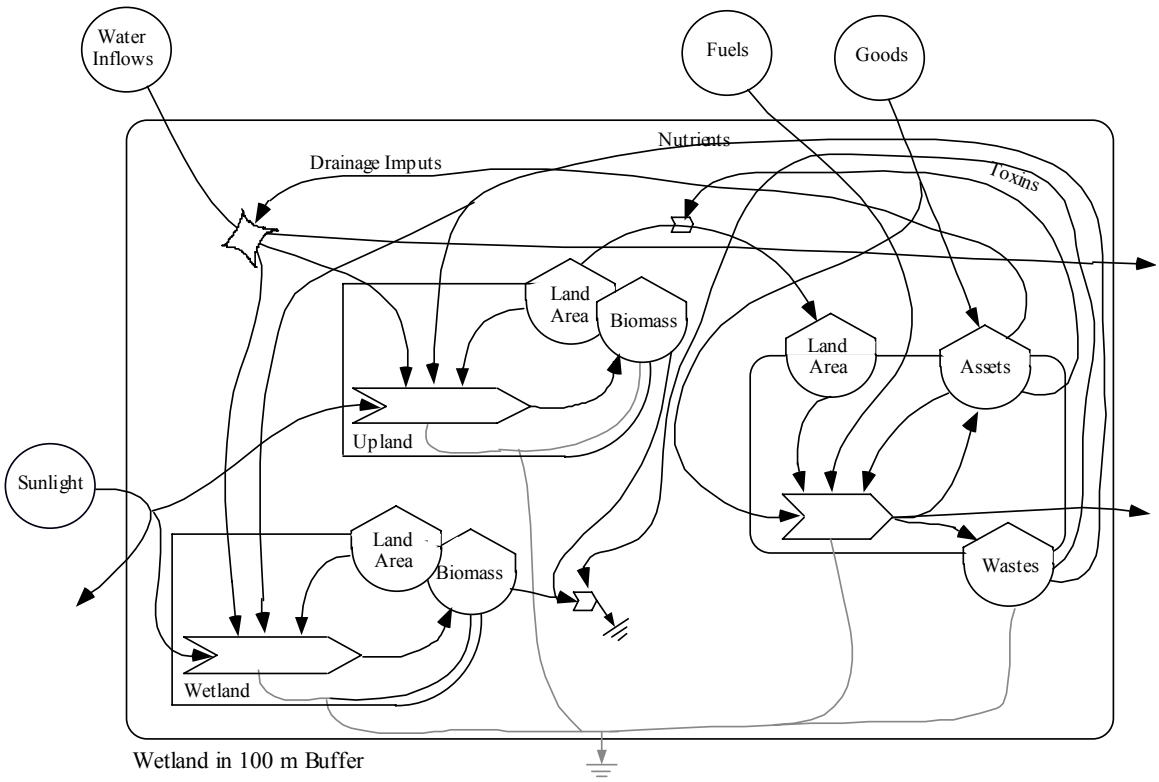


Figure 1-2. Aggregate systems diagram of a cypress dome embedded within a developed landscape. The systems boundary is a 100 meter buffer around the delineated wetland edge, which could be used for the Landscape Development Intensity (LDI) index calculation.

in the seasonality and depth of flooding, increased nutrient inputs, increased toxin inputs, and physical impacts (canals, retaining walls, stormwater box culverts, etc.).

Changes in Hydrology

Figure 1-3 is a systems diagram representing potential hydrologic alterations to a pondcypress ecosystem surrounded by developed land uses. Two important mechanisms of the developed landscape are highlighted. First, increased run-off is considered a factor of the amount of increased impervious surface in the watershed supporting the isolated pondcypress wetland and the amount of rainfall. This would be particularly apparent in an urban landscape, where previously vegetated lands are paved creating increased water flow during rain events, which might otherwise have been intercepted by the vegetation.

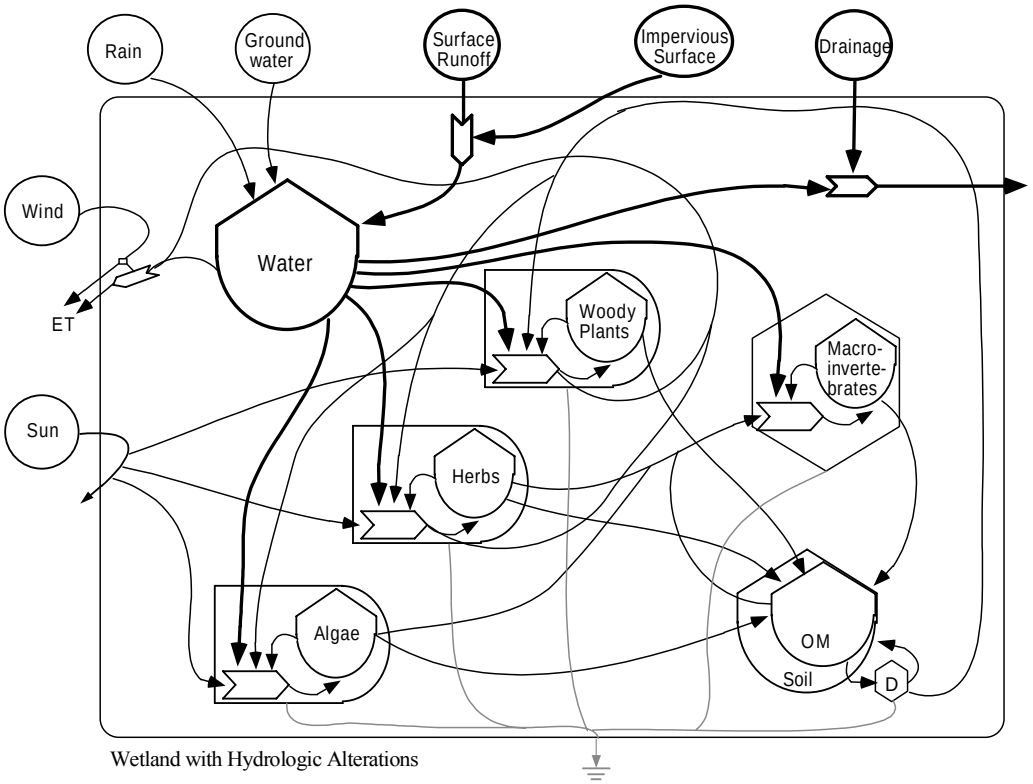


Figure 1-3. Mechanism of altered hydrology of a wetland in a developed landscape. Bold lines represent examples of potential hydrological alterations.

A second mechanism of the developed landscape is an increased outflow of water storage from the wetland resulting from drainage.

In Florida, mature cypress trees are considered the most flood tolerant of all tree species (Harms et al. 1980; Ewel 1990). Past research shows that cypress trees can survive sustained deep flooding (Lugo and Brown 1986; Young et al. 1995), but they also found decreased growth rates and no evidence of regeneration, suggesting that while mature cypress ecosystems may be able to withstand some threshold level of long-term flooding, regeneration may be impeded which is otherwise necessary to ensure the long-term survival of the ecosystem. Ultimately, removing the structure of the wetland will predictably alter other ecosystem components. For example, by removing the tree

canopy, algae may flourish at the water surface where it would have otherwise been shaded out. This would in term alter the food available for the fauna and decrease available sunlight at the soil surface. Ewel (1990) reported that increasing the length of flooding would also affect soil aeration and the ability of other plants to survive and reproduce.

Lugo and Brown (1986) looked at the response of floodplain tree species to sustained increases in water depth after the damming of the Ocklawaha River in Florida. They found that while the larger trees survived some depths of flooding, in the deepest areas, where mean water depth was 1 m, there was 100% tree mortality within 5 years of flooding. In the control system, tree mortality was less than 1% per year. Additionally, flooded trees responded with a dieback of terminal branches, loss of leaves, reduction in leaf size, and loss of color brightness in leaves (Lugo and Brown 1986). Among the trees in the Ocklawaha River floodplain, 25-53 cm was the threshold flooding depth, beyond which tree mortality sharply increased. In a different study, Young et al. (1995) found that the annual radial growth of *Taxodium distichum* (baldcypress) significantly increased for 4 years after flooding, followed by declining growth in the subsequent 16 years. The researchers offered two potential explanations for the initial growth increases: decreased competition due to the death of less flood tolerant species, or increased nutrient levels immediately following flooding.

Marois and Ewel (1983) studied the effects of ditches and berms on 15 cypress domes situated within an intensively managed slash pine plantation. In the cypress domes not ditched and bermed, the lengths of flooding and mean water depth were generally greater. Alternatively in the drier ditched and bermed cypress domes the

density of hardwoods, shrubs, and vines increased. They concluded that while cypress tree growth increased in the years directly following the drying of the domes, cypress regeneration might be inhibited due to changes in vegetative species composition, soil chemistry, and hydrology.

Cypress seeds require soaking in water in order to germinate (Demaree 1932), so altering the seasonality and decreasing the depth of flooding in a cypress dome may inhibit or seriously diminish potential germination. The higher density of hardwoods, shrubs, and vines may also inhibit cypress regeneration by blocking sunlight from reaching the forest floor. Marois and Ewel (1983) found the highest percentage of light transmittance in unaltered domes. The cypress domes with greater light also had an abundance of grasses and sedges. Ewel (1990) noted that drainage allows species with low flood tolerance to become established, resulting in an increased density of shrubs and hardwoods, poor cypress regeneration, increased fire potential, and a dramatic shift to arboreal species from aquatic and wading fauna (Marois and Ewel 1983; Harris and Vickers 1984). More specifically, Marois and Ewel (1983) found broadleaved predominantly evergreen mid-story plants (such as *Ilex cassine*, *Lyonia lucida*, *Magnolia virginiana*, and *Persea palustris*), became more common in swamps when water levels were lowered. Harris and Vickers (1984) speculated that shifting species in the vegetation layer equates to altered structure and habitat for fauna which affects organisms in all other trophic levels.

Decreasing the mean water level could cause changes in the community composition of many species that rely on cypress domes for regeneration. Benthic invertebrates may not withstand increased dry periods, and reproduction may be difficult

or unattainable. Often forming the base of swamp food chains (Ewel 1990), eliminating or decreasing the population of benthic macroinvertebrates could have repercussions throughout the food chain. Fish, amphibians, and reptiles may be eliminated from cypress domes with decreased hydroperiods due to reproduction difficulties or altered food availability (Means et al. 1998; Ewel 1990).

Increased Inflows of Nutrients and/or Toxins

In undeveloped landscapes cypress domes receive limited nutrient inputs from rainwater and surface water run-off (Wharton et al. 1977). Figure 1-4 is a systems diagram of a pondcypress dome receiving increased nutrients and/or toxins. The inflow of nutrients and toxins may come from both point (stormwater and wastewater additions) and non-point (run-off) sources. An increase in run-off from impervious surfaces in the surrounding landscape may carry an increased loading of nutrients and toxins (Harper 1994). Surface run-off carrying fertilizer used on agricultural crops or home lawns are examples of non-point source contributions. Figure 1-4 shows that as nutrients flow into the wetland the growth of living biomass increases, and that nutrients accumulate in the water and soil organic matter storages. Conversely, as toxins flow into and accumulate in the wetland, there is a deleterious effect on biomass.

Two nutrients of primary importance in pondcypress domes are phosphorus and nitrogen. These are represented in the grouped "N" nutrient pool sources and storage tanks in Figures 1-4. Phosphorus, an element critical to plant growth, is mostly bound into forms unavailable to plants at pH levels below 5.7 (Brady and Weil 2004), higher than the average pH of cypress domes embedded in undeveloped landscapes (Coultas and Duever 1984) Phosphorus is known to accumulate in the clay layers found beneath cypress domes, which makes cypress dome ecosystems dependent on a constant input of

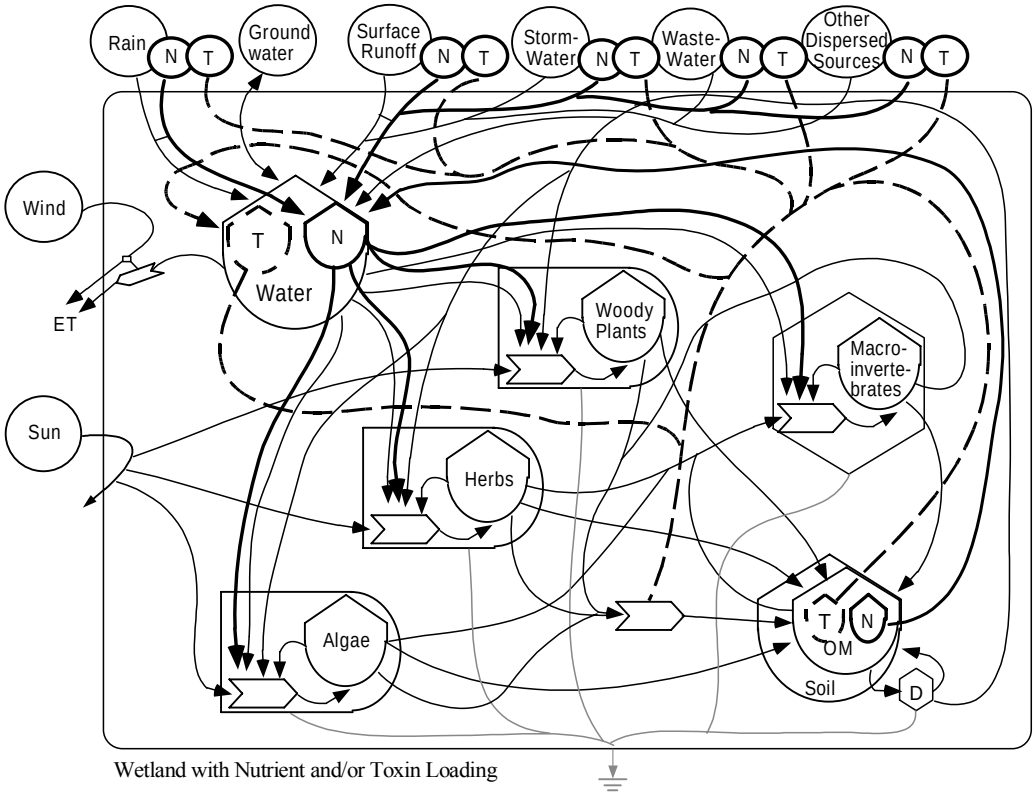


Figure 1-4. Increased nutrients and/or toxin inflows into a wetland from the surrounding developed landscape. The character "N" represents a pooled nutrient tank, and "T" represents a pools toxins tank.

available phosphorus from rainfall. Raising the pH of the wetland should increase the concentration of available phosphorus. In contrast, nitrogen does not accumulate in the clay layer or organic sediments at the bottom of cypress domes due to denitrification processes, and the rate of the nitrogen cycle seems dependent on the cycling of decomposition of organic matter (Wharton et al. 1977).

Previous studies in cypress domes show that cypress trees respond to increased nutrient loading with increased tree growth rates (Nessel et al.1982; Lemlich and Ewel 1984). Nessel et al. (1982) measured phosphorus concentrations in live cypress needles at a cypress dome embedded in silvicultural land use and a cypress strand receiving

sewage for more than 40 years. Cypress needles in the silvicultural wetland had a lower average phosphorus concentration than cypress needles in the wetland receiving wastewater. Additionally, the top 20 cm of sediments in the wetland receiving wastewater had nearly 5.5 times as much phosphorus per m^2 as in the silvicultural wetland. They concluded that the trees in the cypress strand were in fact responding to the increased nutrient inputs (from the sewage) with increased growth rates.

Historically cypress domes were oligotrophic systems (Mitsch and Gosselink 1993). Ewel (1984) and Harris and Vickers (1984) found that an increase in dissolved nutrients led to the development of thick mats of *Lemna* spp., *Spirodela* spp., and/or *Azolla* spp. on the water surface. Ewel (1984) also noted that a nutrient enriched cypress dome had similar understory species composition compared to a cypress dome not receiving wastewater, however the leaf area was significantly higher in the nutrient enriched cypress dome. Changes in the understory vegetation (from increased leaf area or covering of the water surface with a layer of vegetation) can have an effect on other trophic levels within the ecosystem. In cypress domes receiving wastewater additions, Harris and Vickers (1984) reported an increase in numbers of invertebrates and amphibians, however they also noted a shift in the invertebrate taxa and a high larval mortality of amphibians suggesting the fauna in the nutrient enriched dome were different from the control wetland.

Physical Disturbance

Figure 1-5 is a systems diagram of the physical influences to a pondcypress wetland in a developed landscape. Examples of physical changes include the trampling and grazing of domestic cattle, rooting of feral pigs, barriers of roads and retention walls,

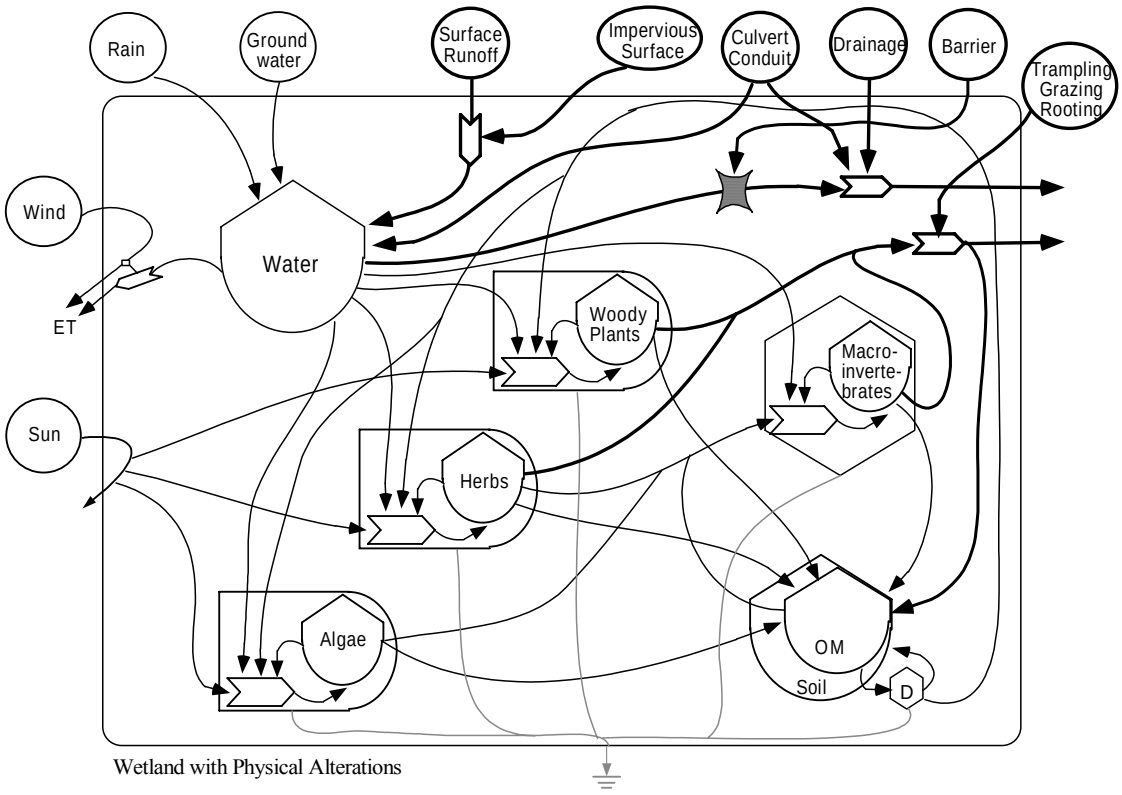


Figure 1-5. Potential physical alterations to a pondcypress wetland. Bold lines highlight physical alterations such as impervious surface, culverts, drainage mechanisms, barriers, and trampling, grazing, and rooting by animals.

and conduits such as stormwater culverts. The trampling, grazing, and rooting by animals is depicted as a drain on the biomass of herbaceous and woody plants and macroinvertebrates. Barriers to water flow and impoundments are represented through a switch operation, showing that when a barrier is constructed it acts as a control over the level of water in the storage tank. Additionally, conduits act in two opposing ways by either increasing the flow of water into the system or by helping to drain water from the system, depending on the construction design. Another important component is the increased flow of surface run-off from impervious surface outside of the wetland system boundary.

Little research has been able to quantify the effects of physical modifications to wetlands. Findlay and Houlahan (1997) used existing biological surveys for 30 Ontario, Canada, wetlands, comparing the species richness of plants, amphibians, birds, and reptiles with wetland areas, density of paved roads, and percent forest cover. They found a negative correlation between wetland species richness and density of paved roads on lands within 2 km of the wetland. They concluded that increasing the density of paved road surface or decreasing the forest cover by 20% within 2 km surrounding a wetland would pose significant risks to the biodiversity of the wetland and be as detrimental as losing 50% of the wetland itself, in terms of loss of species richness.

Another physical modification to cypress domes is removing a portion of the canopy layer. Florida has a long history of timber harvesting, and Ewel (1990) suggested that nearly all of the cypress domes in north Florida have been logged since the late 1800s. Studies showed that logged cypress domes maintained their defining characteristics after regeneration (Terwilliger and Ewel 1986; Ewel et al. 1989); however, during regeneration, there were shifts in the flora and fauna of logged wetlands. Physical modifications such as roads, canals, and stormwater culverts also act as direct conduits for the introduction of exotic species (Frappier and Eckert 2003).

Quantifying Anthropogenic Influence

Wetlands occupy a large portion of the Florida landscape. An estimate from the 1780s reported 8,225,000 ha of wetlands in Florida (Dahl 2000). By the mid-1980s, the National Wetlands Inventory estimated Florida had 4,467,000 ha of wetlands remaining, translating into a loss in Florida of 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% of those wetlands sampled were adjacent to agricultural lands and therefore potentially affected by land use practices such as herbicide and pesticide application, irrigation, livestock watering and wastes, soil erosion, and deposition. An additional 17% 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 are 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. Spanning the populated coasts from Jacksonville to Miami on the east coast and from Naples to Tampa along the west coast, most coastal counties are reported to have high wetland loss of non-tidal freshwater wetlands from 1986 to 1997 (Dahl 2000). Dahl (2000) suggested that many of these wetlands were harvested and returned as shrub wetlands.

Anthropogenic activities can influence an array of changes in surrounding ecosystems. There have been numerous attempts at quantifying anthropogenic influence based on varying scales. Three primary indices of anthropogenic influences were incorporated throughout our study to compare wetland condition, including the Landscape Development Intensity (LDI) index (Brown and Vivas 2004), the Wetland Rapid Assessment Procedure (WRAP; Miller and Boyd 1999), and the Minnesota disturbance index (Gernes and Helgen 1999).

Landscape Development Intensity Index

The Landscape Development Intensity (LDI) index can be used as an index of human activity based on a development intensity measure derived from nonrenewable energy use in the surrounding landscape. The underlying concept behind calculating the LDI (quantifying the nonrenewable energy use per unit area in the surrounding landscape 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 Vivas (2004) suggest that landscape condition 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. The LDI scale encompasses a gradient from completely natural to highly developed land use intensity. More intense activities such as highways and multi-family residential land uses receive higher LDI scores. Natural landscapes such as wetlands, lakes, and upland forests receive a 1.0, the lowest possible LDI score, based on no use of nonrenewable energy in these ecosystems.

The LDI is calculated based on the percent of the area in a particular land use times the Landscape Development Coefficient (LDC), which is defined by the amount of nonrenewable energy use. The LDC coefficient does not account for any individual causal agent directly, but instead, may represent the combined actions 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.

Wetland Rapid Assessment Procedure

The Wetland Rapid Assessment Procedure (WRAP) attempts to provide accurate and consistent evaluations of wetland sites, and relied on an evaluator with an adequate understanding of the functions of and species found throughout Florida ecosystems (Miller and Gunsalus 1997). WRAP consists of a qualitative score describing the functional capacity of a wetland. Scores ranged from 0.0 to 3.0, in 0.5 increments. The 6 scoring categories for WRAP include: 1) Wildlife utilization; 2) Overstory/shrub canopy; 3) Vegetative ground cover; 4) Adjacent upland support/buffer; 5) Field indicators of wetland hydrology; and 6) Water quality input and treatment. A score of 3.0 indicates an "intact" wetland, whereas a score of 0.0 indicates a wetland with a reduced functional capacity (Miller and Boyd 1999).

Minnesota Disturbance Index

The Minnesota disturbance index is considered a gradient of human disturbance, or a measure of land use disturbance based on investigator knowledge, observations, and best professional judgment about the degree of influence to the ecosystem. Gernes and Helgen (1999) used the Minnesota disturbance index as a baseline for creating an index of vegetative biotic integrity for depressional wetlands. There are 2 primary categories and 3 secondary categories used to calculate the Minnesota disturbance index score. The primary categories include stormwater and agricultural influence, and are weighted twice as high as the secondary categories. Wetlands receive scores assigned according to significantly affected (S = 8), moderately affected (M = 4), least affected (L = 2), and not applicable (NA = 0) depending on the scorers opinion as to the degree of influence. Wetlands only receive a score in 1 of the primary categories, and reference wetlands receive a score of 0 in both primary categories. The 3 secondary categories include

hydrologic/miscellaneous influence, historical influence, and buffer, receiving scores of significantly affected (S = 3), moderately affected (M = 2), least affected (L = 1), and not applicable (NA = 0). Wetlands can receive scores in all of the secondary categories, with possible scores ranging from 0 to 17.

Plan of Study

Physical and chemical environmental parameters and the community composition of diatoms, macrophytes, and macroinvertebrates were sampled in isolated forested wetlands throughout Florida to answer the overall question: can the changes in biotic components of pondcypress wetlands (such as the community composition of the diatom, macrophyte, and macroinvertebrate assemblages) be related to changes in development intensity in the landscape immediately adjacent to and surrounding them. Wetland study sites were sought in various landscape settings that included natural, agricultural, and urban land uses. Three independent measures of anthropogenic influence were calculated for each wetland including LDI, WRAP, and the Minnesota disturbance index. Compositional differences among the diatom, macrophyte, and macroinvertebrate assemblages were identified and related to the 3 measures of anthropogenic influence. Each assemblage was used to construct the Wetland Condition Index (WCI) for isolated forested wetlands in Florida.

CHAPTER 2
METHODS

Biological, physical, and chemical parameters were sampled in 118 forested wetlands less than 2 ha in size. This chapter describes site selection, calculations of landscape development intensity, field-data collection, and laboratory analyses. Statistical analyses are described for each assemblage and for the creation of the Wetland Condition Index (WCI).

Site Selection

Field research spanned two growing seasons with 72 wetlands sampled between May-September in 2001 and an additional 46 wetlands sampled between May-October in 2002. Figure 2-1 shows the location of the 118 sample wetlands indicated by generalized *a priori* land use categories (reference, agricultural, urban). Hereafter wetlands embedded in primarily undeveloped landscapes were called reference; wetlands embedded in primarily agricultural land uses were called agricultural wetlands; and wetlands embedded in primarily urban land uses were called urban wetlands.

Random site selection was not feasible given the necessity of obtaining permission to access private lands and the non-random pattern of land development in Florida. Site selection for agricultural wetlands was accomplished with the aide of the Natural Resources Conservation Service under the United States Department of Agriculture and University of Florida Institute of Food and Agricultural Sciences extension agents. Sample wetlands were targeted spatially throughout Florida, so that a nearly equal distribution of wetlands was sampled within each of the 4 Florida ecoregion (panhandle

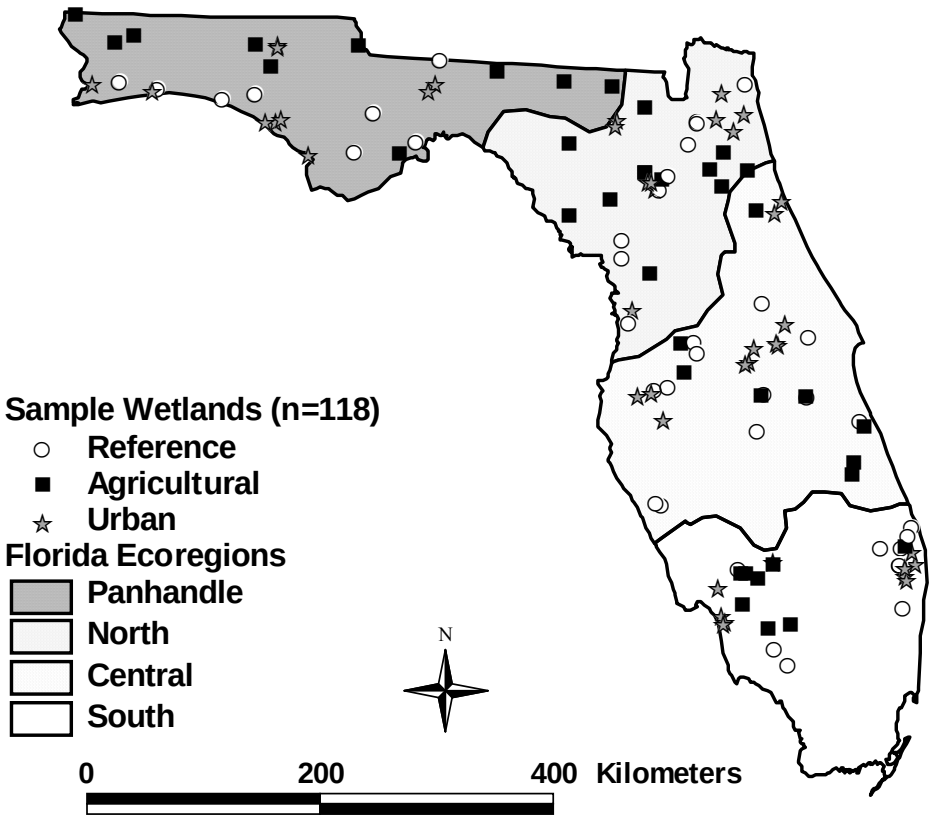


Figure 2-1. Study site location of 118 isolated forested wetlands in Florida. The state of Florida was separated into four ecoregions (Lane 2000). Sample wetlands were designated by *a priori* surrounding land use categories: o reference, ■ agricultural, or ★ urban.

n=28; north n=31; central n=31; south n=28). Boundaries of the Florida ecoregions were determined with a hydrologic model by Lane (2000). Florida freshwater palustrine wetlands were classified using a hierarchical classification technique, and physical (surficial geology, soils, digital elevation model, slope) and climatic (precipitation, potential evapotranspiration, runoff, annual days of freezing) variables were tested for correlation with wetland clusters. Final ecoregion boundaries were based on a spatial water balance model.

The number of wetlands sampled per *a priori* land use category per region varied, with 28 wetlands in the south (n = 9 reference, n = 9 agricultural, n = 10 urban), 31 in the

central (n = 11 reference, n = 9 agricultural, n = 11 urban), 31 in the north (n = 9 reference, n = 12 agricultural, n = 10 urban), and 28 in the panhandle (n = 8 reference, n = 10 agricultural, n = 10 urban) ecoregions. All 72 wetlands sampled in 2001 and 39 (of the 46) wetlands sampled in 2002 were dominated by *Taxodium ascendens*, and were considered pond-cypress domes. The remaining 7 wetlands sampled in 2002 had a canopy layer comprised of a mixture of species that was not dominated by *Taxodium ascendens* alone. These wetlands exhibited the characteristic depressional shape of cypress domes in the landscape and from GIS based aerial photography and were included in the sampling pool as potential variants of cypress domes. Wetlands surrounded by natural landscapes were generally located on conservation lands including state and national parks and forests, county and city lands, and private conservation tracts. Wetlands currently surrounded by cattle pasture, row crops, citrus, and silvicultural land uses were included in the agricultural *a priori* land use category. Urban wetlands located in an urban land use matrix for the longest period of time were given priority for sampling. However, due to the widespread historic loss of wetlands throughout Florida (FDNR 1988) and early incentives to drain swamplands, few pondcypress domes were found in the oldest urban areas. Many of the urban wetlands sampled were suspected to previously been embedded in agricultural land uses.

Table 2-1 provides some general information about each sample wetland, including sample date, surrounding land use, and land ownership. The sample date provided is the earliest sample date, and correlates to macrophyte sampling. A minimum water level of 10 cm was standardized to ensure sampling did not occur immediately following a small rain event, or too soon after initial hydration for the growing season, which would not

Table 2-1. Surrounding land use, land ownership, and sample date for 118 study wetlands in Florida.

Site Code*	Sample Date	Surrounding Land Use^	Land Owner	Site Code*	Sample Date	Surrounding Land Use^	Land Owner
SA1	6/5/01	Cattle & Crops	Public	CR3	6/20/01	State Park	Public
SA2	6/6/01	Citrus	Private	CR4	8/10/01	WMD	Public
SA3	6/27/01	Cattle	Public	CR5	8/13/01	State Park	Public
SA4	7/30/01	Crops	Private	CR6	8/15/01	State Forest	Public
SA5	7/31/01	Cattle & Crops	Public	CR7	5/30/02	City Owned	Public
SA6	9/5/01	Cattle	Private	CR8	7/2/02	State Forest	Public
SA7	7/31/02	Woodland	Public	CR9	7/11/02	State Preserve	Public
SA8	7/31/02	County Park	Public	CR10	10/9/02	State Park	Public
SA9	8/1/02	Cattle	Private	CR11	10/9/02	State Park	Public
SR1	6/28/01	County Park	Public	CU1	5/31/01	Univ. Campus	Public
SR2	7/3/01	State Park	Public	CU2	6/15/01	Residential	Private
SR3	7/24/01	State Reserve	Public	CU3	7/16/01	Commercial	Private
SR4	8/1/01	National Park	Public	CU4	8/14/01	Road Side	Private
SR5	8/21/01	State Preserve	Public	CU5	9/11/01	Road Side	Private
SR6	9/18/01	NWR	Public	CU6	9/12/01	Golf Course	Private
SR7	7/15/02	County Park	Public	CU7	5/30/02	City Owned	Public
SR8	7/17/02	County Airport	Public	CU8	7/1/02	Industrial	Private
SR9	7/24/02	County Park	Public	CU9	7/8/02	Commercial	Private
SU1	6/6/01	Resid. & Golf	Private	CU10	8/7/02	Park	Public
SU2	6/29/01	School Campus	Public	CU11	8/8/02	Park	Public
SU3	7/4/01	Residential	Public	NA1	5/21/01	Cattle	Public
SU4	8/22/01	Residential	Private	NA2	6/4/01	Cattle	Private
SU5	8/23/01	Industrial	Private	NA3	6/19/01	Silviculture	Public
SU6	9/30/01	Industrial	Private	NA4	7/20/01	Crops	Private
SU7	7/16/02	Commercial	Private	NA5	7/27/01	Cattle	Private
SU8	7/16/02	Comm. & Resid.	Private	NA6	7/31/01	Silv., Cat.,Crops	Private
SU9	7/23/02	Residential	Private	NA7	5/22/02	Crops	Public
SU10	7/30/02	Roads & Canals	Public	NA8	5/21/02	Silviculture	Private
CA1	5/23/01	Crops	Private	NA9	6/10/02	Silviculture	Ease.
CA2	5/30/01	Cattle	Private	NA10	7/12/02	Silviculture	Ease.
CA3	6/7/01	Pullet Farm	Private	NA11	7/24/02	Cattle	Public
CA4	6/21/01	Cattle	Public	NA12	7/26/02	Cattle & Crops	Public
CA5	7/10/01	Cattle	Private	NR1	5/26/01	University Land	Public
CA6	7/23/01	Citrus	Private	NR2	6/18/01	City Park	Public
CA7	7/3/02	Silv. & Cattle	Public	NR3	7/10/01	State Forest	Public
CA8	7/19/02	Dairy Farm	Public	NR4	7/11/01	WMD	Public
CA9	7/24/02	Citrus	Private	NR5	8/6/01	Military	Private
CR1	5/30/01	Conserv. Tract	Private	NR6	8/21/01	State Park	Public
CR2	6/14/01	Conserv. Tract	Private	NR7	5/28/02	State Park	Public

Table 2-1. Continued

Site Code*	Sample Date	Surrounding Land Use^	Land Owner	Site Code*	Sample Date	Surrounding Land Use^	Land Owner
NR8	8/5/02	State Park	Public	PA9	8/13/02	Row Crops	Public
NR9	8/29/02	State Forest	Public	PA10	8/14/02	Silviculture	Public
NU1	5/22/01	Road Side	Private	PR1	6/15/01	National Forest	Public
NU2	6/11/01	Resid. & Golf	Private	PR2	7/3/01	WMD	Public
NU3	6/26/01	Residential	Private	PR3	7/4/01	Military	Public
NU4	6/27/01	Residential	Private	PR4	8/9/01	State Forest	Public
NU5	6/28/01	Residential	Private	PR5	8/10/01	State Forest	Public
NU6	8/1/01	Resid. & Instit.	Private	PR6	8/18/01	National Forest	Public
NU7	5/15/02	Comm. & Residential	Private	PR7	6/4/02	Conservation Tract	Private
NU8	6/3/02	Residential & Golf	Private	PR8	8/7/02	NWR	Public
NU9	6/12/02	Industrial	Private	PU1	6/14/01	Residential	Private
NU10	7/29/02	Resid. & Instit.	Private	PU2	7/5/01	Residential	Private
PA1	5/24/01	Cattle	Private	PU3	8/17/01	Resid. & Comm.	Private
PA2	5/29/01	Cattle	Private	PU4	8/17/01	Residential & Park	Private
PA3	7/3/01	Crops/Turf Grass	Public	PU5	9/28/01	Comm. & Silv.	Private
PA4	7/2/01	Crops	Private	PU6	9/29/01	Commercial	Private
PA5	8/8/01	Cattle	Private	PU7	6/18/02	Resid. & Orchard	Private
PA6	8/9/01	Cattle	Private	PU8	6/19/02	Indust. & Silv.	Private
PA7	6/5/02	Cattle	Private	PU9	6/20/02	Residential	Private
PA8	8/8/02	Silviculture	Public	PU10	7/25/02	Institutional	Private

*Site Codes correspond to the region, land use category, and sample order: S = south, C = central, N = north, and P = panhandle; R = reference, A = agriculture, and U = urban.
^Surrounding Land Use abbreviations: NWR = National Wildlife Refuge; WMD = Water Management District; Resid. = Residential; Cat. = Cattle; Comm. = Commercial; Instit. = Institutional; Crops = Row Crops; Silv. = Silviculture; Ease. = Easement.

allow the biological assemblages dependent on inundation time to respond. Wetlands sampled without sufficient standing water were revisited later in the field season once the wetlands held at least 10 cm of water. Table 2-2 identifies data collected at each wetland. Site codes reflect the ecoregion (S = south; C = central; N = north; P = panhandle), land use category (R = reference; A = agricultural; U = urban), and the order they were sampled. Site codes were assigned to preserve the anonymity of individual land owners.

Gradients of Landscape Development Intensity

Three independent indices of anthropogenic activity in the landscape were calculated for the study wetlands including the Landscape Development Intensity (LDI)

Table 2-2. Field-data collected at 118 sample wetlands.

Site	Soil	Water	Diatoms	Macrophytes^	Macro-invertebrates	Site	Soil	Water	Diatoms	Macrophytes^	Macro-invertebrates	Site	Soil	Water	Diatoms	Macrophytes^	Macro-invertebrates
SA1	✓			✧		CR4	✓	✓	✓	◆	✓	NU1	✓			✧	
SA2	✓	✓	✓	◆	✓	CR5	✓	✓	✓	◆	✓	NU2	✓	✓	✓	✧	✓
SA3	✓	✓	✓	✧	✓	CR6	✓	✓	✓	◆	✓	NU3	✓			✧	
SA4	✓	✓	✓	◆	✓	CR7	✓			✧		NU4	✓	✓	✓	✧	✓
SA5	✓	✓	✓	◆	✓	CR8	✓	✓	✓	◆	✓	NU5	✓	✓	✓	✧	✓
SA6	✓	✓	✓	◆	✓	CR9	✓	✓	✓	◆	✓	NU6	✓	✓	✓	✧	✓
SA7	✓	✓	✓	◆	✓	CR10	✓	✓	✓	◆	✓	NU7	✓			✧	
SA8	✓	✓	✓	◆	✓	CR11	✓	✓	✓	◆	✓	NU8	✓			✧	
SA9	✓	✓	✓	◆	✓	CU1	✓	✓	✓	✧	✓	NU9	✓			✧	
SR1	✓	✓	✓	◆	✓	CU2	✓			✧		NU10	✓	✓	✓	◆	✓
SR2	✓	✓	✓	◆	✓	CU3	✓	✓	✓	◆	✓	PA1	✓			✧	
SR3	✓	✓	✓	◆	✓	CU4	✓			✧		PA2	✓	✓	✓	✧	✓
SR4	✓	✓	✓	◆	✓	CU5	✓	✓	✓	◆	✓	PA3	✓	✓	✓	◆	✓
SR5	✓	✓	✓	◆	✓	CU6	✓	✓	✓	◆	✓	PA4	✓			✧	✓
SR6	✓		✓	◆	✓	CU7	✓	✓	✓	✧	✓	PA5	✓	✓	✓	◆	✓
SR7	✓	✓	✓	◆	✓	CU8	✓	✓	✓	◆	✓	PA6	✓	✓	✓	◆	✓
SR8	✓	✓	✓	◆	✓	CU9	✓	✓	✓	◆	✓	PA7	✓			✧	
SR9	✓	✓	✓	◆	✓	CU10	✓	✓	✓	◆	✓	PA8	✓			✧	
SU1	✓	✓	✓	✧	✓	CU11	✓		✓	✧		PA9	✓			✧	
SU2	✓	✓	✓	✧	✓	NA1	✓			✧		PA10	✓			✧	
SU3	✓	✓	✓	◆	✓	NA2	✓			✧		PR1	✓		✓	◆	✓
SU4	✓	✓	✓	◆	✓	NA3	✓			✧		PR2	✓			✧	
SU5	✓	✓	✓	◆	✓	NA4	✓		✓	◆	✓	PR3	✓			✧	
SU6	✓	✓	✓	◆	✓	NA5	✓			✧		PR4	✓	✓	✓	◆	✓
SU7	✓	✓	✓	◆	✓	NA6	✓	✓	✓	◆	✓	PR5	✓	✓	✓	◆	✓
SU8	✓	✓	✓	◆	✓	NA7	✓			✧		PR6	✓	✓	✓	◆	✓
SU9	✓		✓	◆	✓	NA8	✓			✧		PR7	✓	✓	✓	◆	✓
SU10	✓			✧		NA9	✓			✧		PR8	✓	✓	✓	◆	✓
CA1	✓			✧		NA10	✓	✓	✓	◆	✓	PU1	✓	✓		✧	
CA2	✓	✓	✓	✧	✓	NA11	✓	✓	✓	◆	✓	PU2	✓			✧	
CA3	✓	✓	✓	✧	✓	NA12	✓			✧		PU3	✓	✓	✓	◆	✓
CA4	✓	✓	✓	✧	✓	NR1	✓			✧		PU4	✓	✓	✓	◆	✓
CA5	✓	✓	✓	◆	✓	NR2	✓	✓	✓	✧	✓	PU5	✓			✧	
CA6	✓		✓	◆	✓	NR3	✓	✓	✓	✧	✓	PU6	✓	✓		✧	
CA7	✓	✓	✓	◆	✓	NR4	✓	✓	✓	◆	✓	PU7	✓			✧	
CA8	✓	✓	✓	✧	✓	NR5	✓			✧		PU8	✓			✧	
CA9	✓		✓	◆	✓	NR6	✓	✓	✓	◆	✓	PU9	✓			✧	
CR1	✓	✓	✓	✧		NR7	✓			✧		PU10	✓	✓	✓	◆	✓
CR2	✓			✧		NR8	✓	✓	✓	◆	✓						
CR3	✓	✓	✓	✧	✓	NR9	✓	✓	✓	◆	✓						

✓ = Data collected

^ ◆-sampled with >10 cm standing water; ✧-sampled with < 10 cm standing water

index, the Wetland Rapid Assessment Procedure (WRAP), and the Minnesota disturbance index. LDI, WRAP, and Minnesota disturbance index scores for each sample wetland are listed in Appendix B. LDI scores were calculated prior to site visits using 1999 digital orthophoto imagery of Florida available from Labins, The Land Boundary Information System from the Florida Department of Environmental Protection (FDEP) (available at http://www.labins.org/2003/index.cfm). Sample wetlands were delineated from aerial images, and a 100 m buffer was constructed around the edge of each wetland in ArcView GIS 3.2 (Environmental Systems Research Institute, Inc. 1999). Different land uses within the 100 m buffer were hand delineated based on the aerial images. Land uses were updated during the site visit to reflect any changes in land use since the 1999 aerial images were recorded. The following equation was used to calculate LDI:

$$LDI = \Sigma (LDC * \% LU) \quad (2\text{-}1)$$

where LDC is the Landscape Development Coefficient for a particular land use based on the amount of nonrenewable energy use per unit area in the surrounding landscape (Table 2-3), and %LU is the percent of a land use within the 100 m buffer of the wetland. The LDC values and LDI equation are based on work by Lane (2003)and Brown and Vivas (2004). Potential LDI scores ranged from a minimum of 1.0 (Natural Land/Open Space) to a maximum of 10.0 (Central Business District).

WRAP was scored during the initial 30 minutes at each study wetland according to descriptions from Miller and Gunsalus (1997). The Minnesota disturbance index was scored (after the field visit) by the field crew leader using information obtained from ArcView GIS 3.2 (Environmental Systems Research Institute, Inc. 1999) and field notes using categories established by Gernes and Helgen (1999).

Table 2-3. Landscape Development Coefficients (LDC) used in the calculation of the Landscape Development Intensity (LDI) index.

Land Use	Nonrenewable Energy Use (E14 solar equivalent joules/ha/yr)	LDC
Natural Land / Open Water	1.0	1.0
Pine Plantation	5.1	2.0
Low Intensity Open Space / Recreational	6.7	2.1
Unimproved Pastureland (with livestock)	8.3	2.6
Improved Pasture (no livestock)	19.5	3.7
Low Intensity Pasture (with livestock)	36.9	4.5
Medium Intensity Open Space / Recreational	51.5	4.8
High Intensity Pasture (with livestock)	65.4	4.9
Citrus	67.3	5.2
Row crops	117.1	5.9
High Intensity Agriculture (dairy farm)	201.0	6.6
Recreational / Open Space (High-intensity)	1077.0	6.9
Single Family Residential (Low-density)	1230.0	7.6
Single Family Residential (Med-density)	2461.5	7.7
Single Family Residential (High-density)	3080.0	8.0
Low Intensity commercial (Comm Strip)	3729.5	8.0
Institutional	3758.0	8.1
Highway (2 lane)	4042.2	8.3
Industrial	5020.0	8.3
Multi-family residential (Low rise)	5210.6	8.7
Highway (4 lane)	7391.5	8.9
High intensity commercial (Mall)	12661.0	9.2
Multi-family residential (High rise)	12825.0	9.2
Central Business District (Avg 2 stories)	16150.3	9.4
Central Business District (Avg 4 stories)	29401.3	10.0

Field-data Collection

A concise summary of field-data collection procedures follows. Appendix C provides more detailed descriptions of field-data collection techniques in the format of Standard Operating Procedures (SOPs) for field use. Field methods are described as transect establishment followed by water, soil, diatom, macrophyte, and macroinvertebrate sampling techniques.

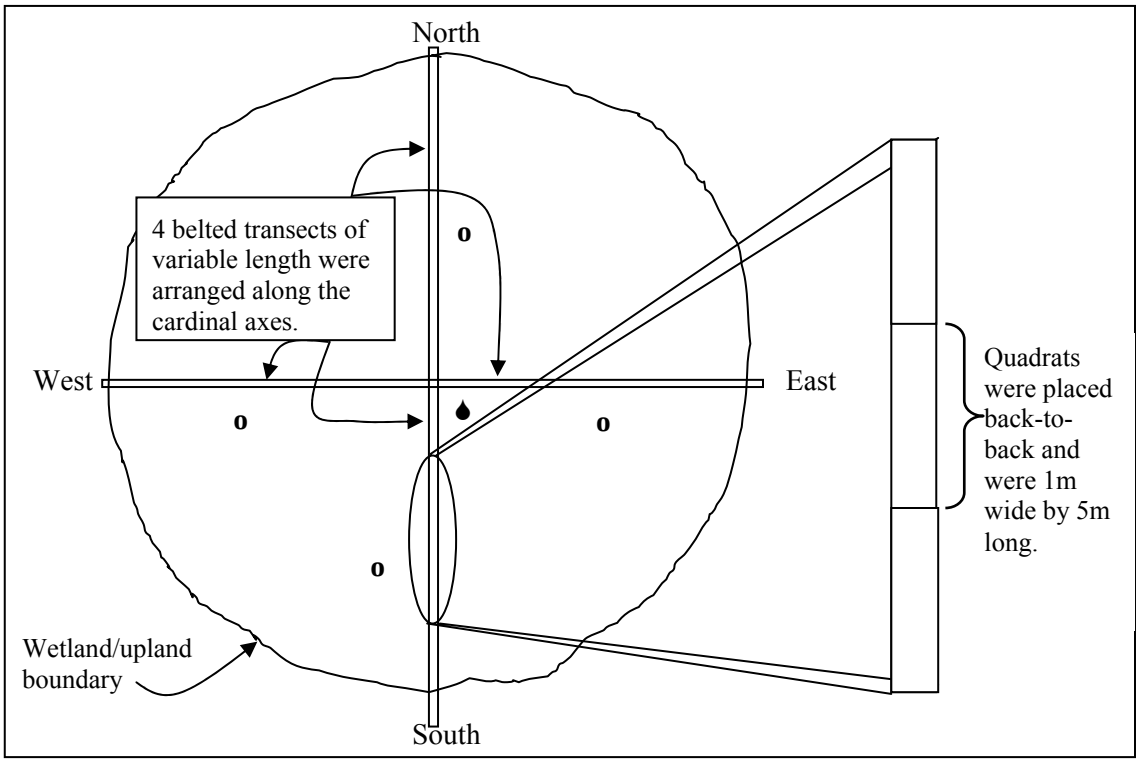


Figure 2-2. Belted transect layout for macrophyte sampling and the location of the water and soil samples. One soil core (**o**) was taken along each transect and compiled for each wetland. One water sample (♦) was taken in the approximate center of each wetland.

Sampling Design

Figure 2-2 shows the positioning of the 4 transects established at each wetland that were situated as perpendicular crossing axes running through the center of each wetland. Transect axes always corresponded to the cardinal directions (north, east, south, west). The wetland/upland boundary was determined based on a combination of wetland plant presence according to wetland plant status (for example, obligate, facultative, or upland from Tobe et al. 1998) and wetland hydrologic indicators.

Water Samples

A grab style water sample was taken in the deepest pool of each wetland when a minimum of 10 cm of standing water was present throughout at least 50% of the wetland

area. The area with the deepest pool often coincided with the center of each wetland as depicted in Figure 2-2. Water samples were collected at 75 wetlands, including 14 in the panhandle ecoregion (reference n = 5; agricultural n = 4; urban n = 5), 14 in the north ecoregion (reference n = 6; agricultural n = 3; urban n = 5), 23 in the central ecoregion (reference n = 9; agricultural n = 6; urban n = 8), and 24 in the south ecoregion (reference n = 8; agricultural n = 8; urban n = 8).

Dissolved oxygen and water temperature were taken on-site using a YSI-55 Dissolved Oxygen hand meter. Grab water samples were sent to the Florida Department of Environmental Protection (FDEP) Central Chemistry Laboratory, Tallahassee, Florida. Analysis included color (EPA 110.2), turbidity (EPA 180.1), pH (150.1), specific conductance (EPA 120.1), ammonia- nitrogen (EPA 350.1), nitrate/nitrite-nitrogen (EPA 353.2), total Kjeldahl nitrogen (EPA 351.2), and total phosphorus (EPA 365.4).

Soil Samples

A composite soil sample was collected at all 118 sample wetlands. Cores were taken using a 7.6 cm diameter PVC pipe driven 10 cm into the soil. One soil core was collected in the approximate middle of each transect (Figure 2-2), and soil cores were homogenized into a composite sample per site. Soil moisture (Gardner 1986), organic matter, total Kjeldahl nitrogen (USEPA 1993), and total phosphorus (USEPA 1979) were analyzed. Nitrogen and phosphorus samples were processed through the Institute of Food and Agricultural Sciences (IFAS) Analytical Research Laboratory, Gainesville, Florida.

Diatoms

Benthic diatom samples were collected at 50 isolated forested wetlands throughout Florida between May-September 2001, as listed in Table 2-2. Figure 2-3 shows the

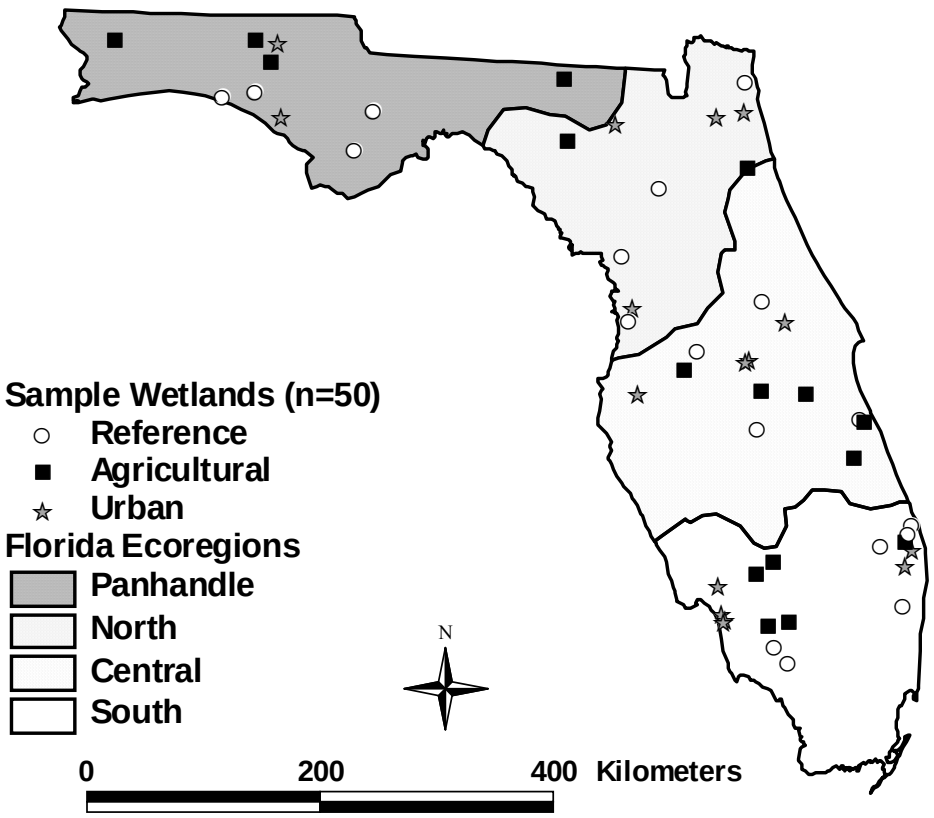


Figure 2-3. Benthic diatom samples were collected at 50 isolated forested wetlands. The state of Florida was separated into four ecoregions (Lane 2000). Sample wetlands were designated by surrounding land use: o reference, ■ agricultural, or ★ urban.

spatial location of the wetlands with benthic diatom samples in Florida. Sites were sampled in the panhandle (n=10), north (n=10), central (n=13), and south (n=17) ecoregions. Sample wetlands were situated in 3 *a priori* described land use categories (reference n=18; agricultural n=16; urban n=16).

A minimum of 10 cm of standing water was necessary for benthic diatom collection. Ten 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 a surface area of 28 cm^2. A bulb pipette was used 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 sample volume of 100 mL. For preservation, 5 mL of M3, an standard preservative for algae samples (APHA 1995), was added to the 100 mL algae sample.

Benthic algae samples were shipped to Michigan State University for identification and enumeration under the supervision of R. J. Stevenson. 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). Five hundred valves were counted along microscope transects and identified to the lowest possible taxonomic level, preferably species (following FDEP SOP#AB-03.1 http://www.dep.state.fl.us/labs/sop).

Macrophytes

Macrophyte vegetation was sampled at all 118 depressional isolated forested wetlands throughout Florida (Figure 2-1). Macrophyte sampling was conducted along the 4 transects situated as perpendicular crossing axes running through the center of each wetland, shown in Figure 2-2. Along each transect, a series of 1 m wide by 5 m long quadrats was established back to back. Living macrophytes rooted within each quadrat were identified to the lowest taxonomic level possible.

Supplementary data

Taxonomic information including species, genus, and family were compiled for all of the macrophytes identified. Additional characteristics were collected for use in developing potential biological indicator metrics, including category (annual or perennial, evergreen or deciduous, indigenous or exotic) and growth form (aquatic, fern, grass, herb,

sedge, shrub, tree, or vine). References specific to Florida were consulted first (Tobe et al. 1998; Wunderlin and Hansen 2003), and additional information was supplemented from other sources (in the following order: Godfrey and Wooten (1981), Wunderlin (1998), and USDA NRCS (2002). When information was still unavailable in published literature for species encountered, Florida botanists (who also participated in the Floristic Quality Assessment Index) were consulted.

Floristic Quality Index

Five Florida botanists agreed to participate in creating a Floristic Quality Assessment Index (FQI) for Florida isolated forested wetlands. The Florida FQI was modeled after an earlier study from Chicago, Illinois, by Willhelm and Ladd (1988), which enlisted botanists to provide quantitative scores for vegetation based on the fidelity of each plant species to a particular environment. The FQI score for an individual wetland was calculated as:

$$\text{Modified FQI} = [\ \Sigma\ (\text{CC for each species present})\ ] / \text{species richness} \qquad (2\text{-}2)$$

where CC = Coefficient of Conservatism score. This equation is considered a modified FQI because previous studies did not account for species richness. The sum of the species CC scores was divided by species richness (Equation 2-2) in this study to account for potential differences in species richness due to differences in ecoregions, *a priori* land use categories, or other unforeseen differences.

CC scores were obtained from the Florida botanists surveyed. Each botanist was sent a complete list of species found in the isolated forested wetlands in the 2001 field season (n = 482 species), and was asked to score each species based on its faithfulness to Florida isolated forested wetlands. After the 2002 field season, one botanist scored the

additional 79 taxa not previously encountered, raising the number of taxa with CC scores to 561 species. Potential CC scores ranged from 0 to 10:

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

Species with low CC scores were considered tolerant of many disturbances, whereas species with high CC scores were considered to occur within a narrow set of stable ecological condition. Appendix D lists the CC scores for 561 macrophytes identified in this study.

Macroinvertebrates

Macroinvertebrates were collected at 79 depressional isolated forested wetlands throughout Florida as shown in Figure 2-4. Field research spanned two growing seasons with 49 wetlands sampled between June-October 2001 and an additional 30 wetlands sampled between June-October 2002. Sites were sampled in the panhandle (n=13), north (n=15), central (n=25), and south (n=26) ecoregions; sample wetlands were situated in 3 *a priori* land use categories (reference n=29; agriculture n=24; urban n=26).

Samples were collected using a U.S. Standard 30 mesh D-frame net. One sweep covered 0.5 m^2 and was measured as 1 net width by 2 net lengths wide, which was repeated 3 times at each location to ensure adequate sampling coverage. Sweeps were always conducted over areas which had not recently been trampled by the field crew. Twenty sweeps were proportioned among major vegetation zones throughout each sample wetland. Sample wetlands generally had between 1 to 3 vegetation zones,

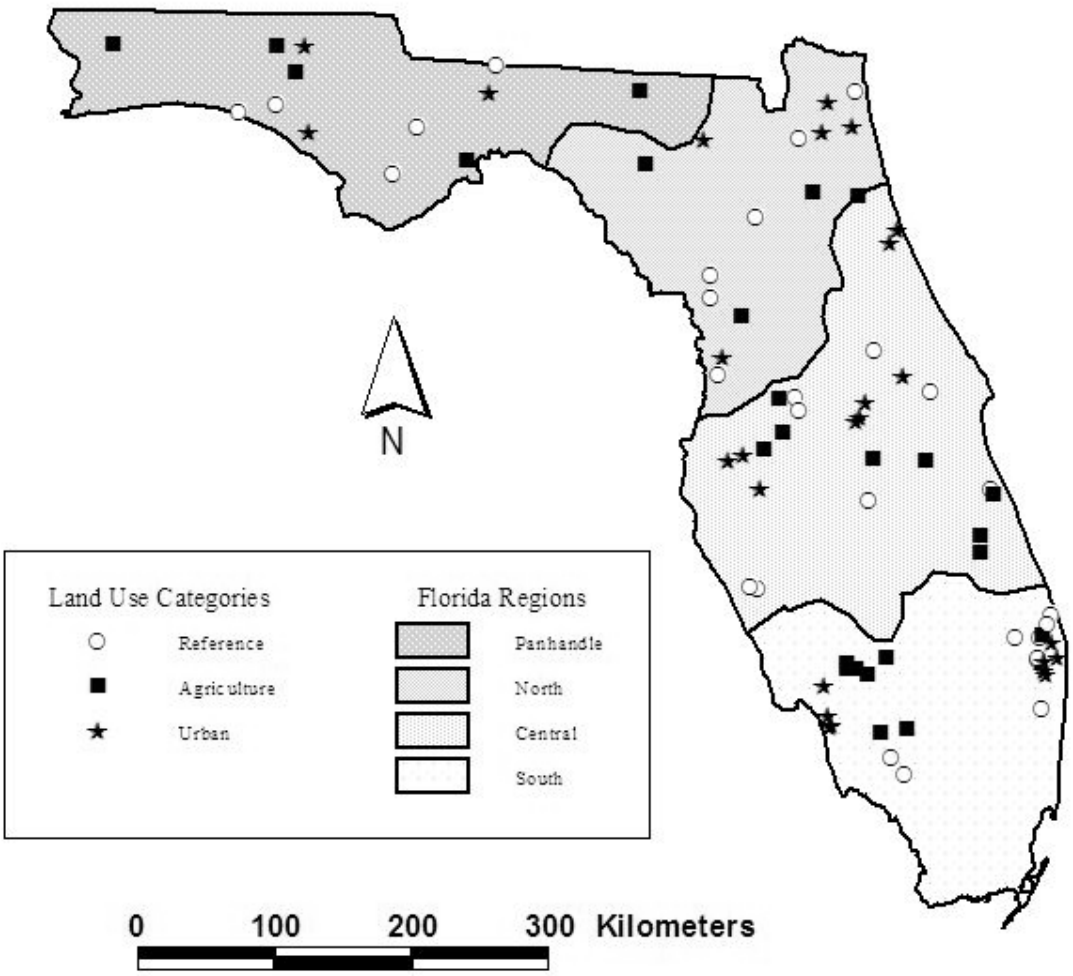


Figure 2-4. Macroinvertebrates were sampled at 79 isolated forested wetlands. The state of Florida was separated into four eco-regions (Lane 2000). Sample wetlands were designated by surrounding land use: o reference, ■ agricultural, or ★ urban.

which were defined by changes in the dominant species cover. When herbaceous plants were included in the sweep area, the bottom of the net was swept from the bottom of the substrate up the plants. In areas with woody plants, the bottom of the net was swept from the substrate up the tree trunk and pieces of woody debris were brushed to remove attached macroinvertebrates. The contents from the sweeps were collected in a 3.8 L

plastic jar and preserved with buffered formalin at a rate of 10% of the sample volume. Appendix C provides a more detailed description of field methods and preservation.

Macroinvertebrate identification was completed at the Florida Department of Environmental Protection (FDEP) Central Laboratory, Tallahassee, Florida, following standard operating procedures (FDEP Standard Operating Procedure #IZ-06 http://www.dep.state.fl.us/labs/sop). Macroinvertebrate samples were sieved, washed, and placed on a pan with 24 individually numbered cells. One-third of the cells were randomly selected and combined in a second numbered tray. A single cell was randomly selected from the second tray for enumeration and identification of macroinvertebrates. When fewer than 100 individuals were encountered in the sample, a second cell was randomly selected from the second tray, and all of the individuals were enumerated and identified. Identification was to the lowest taxonomic level possible.

Data Analysis

Water and Soil Parameters

Water and soil parameters within 3 *a priori* land use categories were compared using Fisher's LSD pair wise comparison in Minitab (Version 13.1, Minitab Statistical Software). The non-parametric Mann-Whitney U-Test was used to discern differences among medians of low and high LDI groups for the water and soil parameters. Wetlands in the low LDI group had a site LDI score less than 2.0, whereas wetlands in the high LDI group had site LDI scores greater than or equal to 2.0, corresponding to a break in the LDC coefficients of undeveloped versus developed land uses (Table 2-3).

To test for multicollinearity among the environmental variables, the variance inflation factor (VIF) and tolerance were calculated using SAS (Version 6 from the SAS Institute, Inc., Cary, North Carolina). Multicollinearity occurred when two or more of the

independent variables exhibited a comparable pattern of correlation with other variables (Zar 1999; Tabachnick and Fidell 1983). To avoid issues with multicollinearity, environmental variables with a VIF greater than 10.0 and a tolerance less than 0.10 (Ott and Longnecker 2001; Pan and Stevenson 1996; ter Braak 1987) were excluded from further analyses.

Summary Statistics

Summary statistics for each assemblage included richness (R), evenness (E), and Shannon diversity (H). For the diatom and macrophyte assemblages summary statistics were calculated at the species level; for the macroinvertebrate assemblage summary statistics were calculated at the genus level. Richness was defined as the total number of distinct taxa encountered within the sample wetland. Evenness was calculated as the Shannon diversity value divided by the natural log of richness:

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

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

$$H = -\Sigma\, p_i * \log(p_i) \quad (2\text{-}4)$$
$$p_i = n_i / N \quad (2\text{-}5)$$

where n_i was the number of occurrences of taxon i, and N was the total number of occurrences of all taxa at a wetland. For the diatom and macroinvertebrate assemblages, the number of occurrence represented the enumeration of the laboratory identified sample and the total number of occurrences represented the sum of the number of occurrences of

all taxa; for the macrophyte assemblage, the number of occurrences represented the number of quadrats a species occurred in, and the total number of occurrences represented the sum of the total number of quadrats of all of the species identified.

For the diatom and macroinvertebrate assemblage, Simpson's index (S) was also calculated as

$$S = 1 - \Sigma (p_i * p_i) \quad (2\text{-}6)$$

where p was defined in Equation 2-5. For the macrophyte assemblage first- and second-order jackknife estimators of species richness ($Jack_1$ and $Jack_2$, respectively) and Whittaker's beta diversity (β_W) were calculated. First- and second-order jackknife estimators were calculated as estimates of true species richness (Colwell and Coddington 1994). Assuming that the sampling effort only measured a portion of the ecosystem, jackknife estimators of species richness provide estimates of actual species richness. The equation for first-order jackknife estimators of species richness is based on the number of species observed (S), the number of species occurring in only one sample unit (rl) (where one sample unit represents one quadrat), and the number of sample units (n) (quadrats):

$$Jack_1 = S + [(rl*(n-1)) / n] \quad (2\text{-}7)$$

The second-order jackknife estimator ($Jack_2$) also incorporated the number of species occurring in exactly two sample units (r2) (quadrats):

$$Jack_2 = S + [(rl*(2n-3)) / n] - [(r2*(n-2)^2) / (n*(n-1))] \quad (2\text{-}8)$$

These estimators of total species richness have shown useful in predicting actual species richness when only a small area of the total ecosystem has been sampled (McCune and Grace 2002).

Whittaker's beta diversity (β_W) was computed as a calculation of overall beta diversity, or the compositional change represented in a sample. Whittaker's beta

diversity was calculated as the number of species at a particular forested wetland (S_c) divided by the average species richness per quadrat (S) minus one:

$$\beta_w = [S_c / S] - 1 \quad (2\text{-}9)$$

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

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

Overall calculations of beta and gamma diversity were calculated for sample wetlands in the 3 *a priori* land use categories. Gamma diversity was calculated as the overall number of taxa encountered at all sample wetlands per *a priori* category. A higher gamma diversity for an *a priori* land use category would suggest a greater difference among the species composition of wetlands within that *a priori* land use category, assuming a similar number of wetlands were sampled within each *a priori* land

use category. Beta diversity was calculated as *a priori* category gamma diversity divided by the average site taxa richness.

Regional Compositional Analysis

The Multi-Response Permutation Procedure (MRPP) was used to test the similarity of community composition for each assemblage among the 4 Florida ecoregions (further application in Zimmerman et al. 1985; McCune et al. 2000; McCune and Grace 2002). MRPP is a nonparametric technique which tests for no difference between groups (the null hypothesis) and is available in PCORD (Version 4.1 from MJM Software, Gleneden Beach, Oregon). It was an appropriate procedure for ecological community data as it does not require distributional assumptions of normality and homogeneity of variances. The Sorensen distance measure was used to calculate the average weighted within-group distance. MRPP provides a test statistic (T), p-value, and chance-corrected within-group agreement (A), which describes within-group similarity. When A equals one, all items are identical within groups, and when A equals zero, differences within-groups equal that expected by chance. Most values of A are less than 0.1 in community ecology (McCune and Grace 2002). MRPP was calculated across all groups (panhandle versus north versus central versus south) as well as for multiple pair wise comparisons (panhandle versus north, panhandle versus central, panhandle versus south, north versus central, north versus south, and central versus south).

Community Composition

Community composition of each assemblage was summarized in a non-metric multidimensional scaling (NMS) ordination to relate changes in community composition with environmental gradients. NMS is an ordination technique designed to compress multi-dimensional space, and is particularly agreeable with ecological data because it

does not rely on linear relationships among variables. This has been described as a compensation for the "zero-truncation problem" through the use of ranked distances and the use of many distance measure (McCune and Grace 2002). The "zero-truncation problem" refers to the extraordinary number of zeros in community ecology data sets. Other ordination techniques depend upon a value for each measured variable for each sample unit. However NMS is useful for presence/absence or abundance data sets, where many species are not present, or receive a zero in the species by site matrix. By ranking the variables, NMS caters to non-parametric, community ecology data sets.

NMS explored the dissimilarities of the community composition of sample wetlands for each assemblage. The Sorensen (Bray-Curtis) distance measure was used for ordination. The dimensionality was chosen based on an initial 6 dimensional run in autopilot mode, which suggested an optimal 3 dimensional solution for the each community composition dataset. To find the optimal 3 dimensional solution, 50 runs with real data and 50 randomized runs were performed with the instability criterion set at 0.00001 and the maximum number of iterations to reach a stable solution set at 500. This procedure was repeated 20 times to insure stability and reproducibility in results. The final run was completed with the starting point set as the results from the best experimental 3 dimensional run, with the lowest stress and best overall fit.

Water and soil parameters, LDI, latitude, and longitude were correlated with the NMS ordination axes with Pearson's correlation coefficients. To improve normality and decrease skewness, 11 water and soil parameters were log (base 10) transformed, including water parameters (dissolved oxygen concentration, temperature, color, turbidity, specific conductivity, ammonia-nitrogen concentration, nitrate/nitrite-nitrogen

concentration, total Kjeldahl nitrogen (TKN) concentration, and total phosphorus (TP) concentration) and soil parameters (TKN and TP concentration). Water pH was not transformed. The remaining soil parameters were measured as percentages, and were transformed by taking the arcsine square root. These included soil moisture and organic matter.

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). Metrics were summarized in 5 main categories:

- tolerance metrics (indicator species or established index values such as the Florida Index)
- autecological (metrics that explore a previously described relationship between a species and an environmental parameter)
- community structure (metrics that explore taxonomic structure)
- community balance (metrics with calculated values, such as evenness or dominance)
- functional group (metrics related to feeding behavior)

Appendix E provides tables of candidate metrics for each assemblage, including 169 candidate diatom metrics (Table E-1), 238 candidate macrophyte metrics (Table E-2), and over 400 candidate macroinvertebrate metrics (Table E-3). Candidate metrics were calculated at the statewide scale for both the diatom and macroinvertebrate assemblages, as sample sizes were limited for regional metric development, particularly in the north and panhandle ecoregions. Candidate macrophyte metrics were calculated at both the regional scale and statewide.

Metrics for the diatom assemblage were calculated as three main forms, including the number (N), percent (P), and abundance (A) based on the single composite sample taken at each sample wetland. The number metric (N) referred to a straight count of

species fitting the particular metric. The percent metric (P) was calculated as the number metric (N) divided by the species richness (R) for each sample wetland.

$$P_i = N_i / R_i \quad (2\text{-}10)$$

where i represents a sample wetland. The abundance metric (A) referred to the sum of the total number of individuals designated by the metric (m) divided by the total number of all individuals identified at a wetland (M):

$$A_i = \Sigma\, m_{i\ \text{occurrence of metric species}} / \Sigma\, M_{i\ \text{all species occurrences}} \quad (2\text{-}11)$$

where i represents a sample wetland. The macroinvertebrate metrics were calculated as the diatom metrics, with the addition of the number of taxa (T), which was calculated as the number of lower taxonomic groups within the metric category.

Macrophyte data were collected within multiple quadrats at each sample wetland, so additional metric forms were possible. Candidate metrics were constructed as the number (N), percent (P), abundance (A), and frequency of occurrence (F). The abundance metric (A) referred to the sum of the total number of species designated by the metric in each quadrat for each respective sample wetland (m) divided by the total number of all species occurrences at a wetland (M). 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).

$$F_i = q_i / Q_i \quad (2\text{-}12)$$

Candidate metrics were accepted if they showed a constant and predictable change along the LDI (Brown and Vivas 2004) according to the strength and significance of the Spearman's correlation coefficient calculated with Analyze-It software v. 1.67 for Microsoft Excel. The Spearman rank correlation tests for an association between 2

related variables, and is a non-parametric alternative to the Pearson correlation. Scatter plots were constructed for each candidate metric versus LDI to ensure correlations were visually distinguishable. The pool of potential candidate metrics was streamlined to reduce the redundancy of selected metrics. Candidate metrics were subjected to a 2-sample t-test to detect differences between low and high LDI groups.

Indicator Species Analysis

For each assemblage, sample wetlands were categorized into two LDI groups and analyzed with Indicator Species Analysis (ISA) in PCORD, which evaluates the abundance and faithfulness of taxa in a defined group (McCune and Grace 2002). ISA can be used to detect and describe the value of taxa indicative of environmental conditions. It requires *a priori* groups and data on the abundance or presence of taxa in each group. These groups are commonly defined by categorical environmental variables, levels of disturbance, experimental treatments, presence and absence of a target species, or habitat types (McCune and Grace 2002). The ISA calculation combines information on the concentration of species abundance and the faithfulness of occurrence of a species in a group. Mathematical equations are available in McCune and Grace (2002) and Dufrêne and Legendre (1997). The calculated indicator species values were based on two standards, faithfulness and exclusion. Faithfulness was defined mathematically by a particular taxa always being present in a particular group. Additionally, the perfect indicator taxa would be exclusive to that group, meaning it never occurred in other groups (Dufrêne and Legendre 1997; McCune and Grace 2002). Calculated indicator values ranged from 0 (no indication), to 100 (a perfect indication of a particular group).

Multiple ISA were conducted to determine sensitive and tolerant indicator taxa for each assemblage. Sample wetlands were categorized based on consecutive LDI breaks

from 1.0 through 7.0 at each 0.25 increment. Each ISA was conducted at each LDI break once using the abundance and once using the presence of taxa at each wetland for each assemblage. The percent sensitive and tolerant indicator taxa at each wetland was calculated and correlated with LDI using Spearman rank correlation. ISA was conducted for each ecoregion (panhandle, north, central, and south) as well as statewide for the macrophyte assemblage. Only statewide analyses were run for the diatom and macroinvertebrate assemblages. Indicator values were calculated and tested for statistical significance using a Monte Carlo randomization technique with 1000 randomized runs. The ISA was used to identify taxa with significant associations to LDI categories. Indicator taxa categorized as tolerant taxa were associated with the higher LDI group; indicator taxa deemed sensitive taxa were associated with the lower LDI group.

In the macrophyte assemblage, the Spearman rank correlation was used to assess differences between statewide and regional indicator species lists for each of the 4 ecoregions. To test for equal application of the statewide indicator species list for each ecoregion, the non-parametric Kruskal-Wallis test was run with Analyse-It software (Ott and Longnecker 2001). Distributional differences were analyzed between *a priori* categories for both tolerant and sensitive indicator taxa among ecoregions.

Diatom metrics

Diatom metrics were created in 3 categories including tolerance, community composition, and autecological metrics (Bahls 1993; van Dam et al. 1994; McCormick and Cairns 1994; Stevenson 2001; Fore and Grafe 2002; Lane et al. 2002; USEPA 2002b; Lane 2003;). Tolerance metrics were created with ISA. Community composition metrics included richness, evenness, and diversity calculations as described above. Autecological metrics were based on previous research that correlated individual diatoms with

morphology, behavior, and the physical and chemical water environment. Diatom species were assigned ecological indicator values using a coded checklist of autecological relationships (Bahls 1993; van Dam et al. 1994). The ecological indicator values from van Dam et al. (1994) included categories diatoms according to water preference, nitrogen metabolism, pH, salinity, dissolved oxygen, saprobic condition, and trophic status. Additional autecological ecological indicator values were adapted from Bahls (1993) analysis of diatoms in Montana streams which included pollution tolerance classifications. Metrics based on morphology and motility were assigned from Stevenson (Lane 2003).

Macrophyte metrics

Macrophyte metrics included tolerance, exotic species, Floristic Quality Index (FQI), longevity, plant growth form, and wetland status metrics (Adamus 1996; Kantrud and Newton 1996; Galatowitsch et al. 1999a; Gernes and Helgen 1999; Carlisle et al. 1999; Fennessy et al. 2001; Mack 2001; and Lane 2003). Tolerance metrics were calculated with ISA. The exotic species metric was calculated as the percent of species that were exotic to Florida divided by the number of species identified at each particular isolated forested wetland. The timeline for determining the exotic status of a species was set near the beginning of European settlement in North America. Many sources were consulted to determine whether a species was considered exotic, including Godfrey and Wooten (1981), Tobe et al. (1998), Wunderlin (1998), USDA NRCS (2002), and Wunderlin and Hansen (2003). For each wetland the modified FQI metric was calculated.

Each species was categorized as native or exotic and annual or perennial (Godfrey and Wooten 1981; Tobe et al. 1998; Wunderlin 1998; USDA NRCS 2002; and

Wunderlin and Hansen 2003). The percent native perennial species metric was calculated as the number of native perennial species encountered divided by the wetland species richness. Wienhold and Van der Valk (1989), Ehrenfeld and Schneider (1991), and Lane (2003) and determined that disturbance often favors annual species over perennial species or promotes the invasion of nonnative perennials in wetlands. Galatowitsch et al. (2000) found that while native perennial cover was reduced in wetlands impacted by cultivation, the occurrence of introduced perennials rather than annuals increased in stormwater impacted wetlands.

The wetland status metric was calculated as the percent of plants classified as obligate or facultative wetland indicator species divided by the number of species at each wetland. Wetland indicator status classifications was from Tobe et al. (1998), USDA NRCS (2002), and Wunderlin and Hansen (2003). There were 5 potential wetland status classifications, including obligate, facultative wetland, facultative, facultative upland, and upland. When possible, the Florida specific wetland indicator status was applied. In some cases, when the Florida wetland indicator status was not available, the National Wetlands Inventory wetland indicator status for the United States was used.

Macroinvertebrate metrics

Candidate metrics for the macroinvertebrate assemblage were constructed in 4 categories, including tolerance, community structure, community balance, and functional group metrics (Lenat 1993; Lenat and Barbour 1994; Kerans and Karr 1994; Wallace et al. 1996; Barbour et al. 1996b; Gerristen and White 1997; Danielson 1998a; Leslie et al. 1999; Galatowitsch et al. 1999a; Smogor and Angermeier 2001; Helgen 2001; Cummins and Merritt 2001; USEPA 2002c; Lane et al. 2002; Lane 2003; Griffith et al. 2003; and Butcher et al. 2003). In addition to ISA, other tolerance candidate metrics were

calculated. Many of the established tolerance metrics were created for flowing waters, which complicated the application of established tolerance values (ex. the Florida Index from Barbour et al. 1996b and Beck 1954; and the Hilsenhoff Biotic Index from Hilsenhoff 1987), since the wetlands sampled in this study were isolated from flowing surface waters (except in extreme high water events when some surface flow may be detectable).

Community structure metrics included richness measures (Danielson 1998a), for example the number of distinct species or specified taxonomic units like the number of families, genera, or species in a collection. Examples of taxa richness metrics include total taxa richness, Ephemeroptera, Plecoptera, and Trichoptera richness; the number of Coleoptera species; or the number of Insecta species (Danielson 1998a).

The use of community balance metrics included some measure of abundance or relative abundance in an attempt to measure the evenness of the macroinvertebrate community (Lenat and Barbour 1994). Examples of community balance metrics include the Shannon diversity index or the percent contribution of the most abundant taxon (Lenat and Barbour 1994).

Macroinvertebrate taxa were grouped based on their functional relationships that overlap taxonomic categorization, including functional feeding groups, habitat groups, and voltinism groups (or life-cycle patterns). Cummins and Merritt (2001) suggest using ratios of numerical abundance or, more favorably, biomass of the various functional groups as indicators of ecosystem attributes, essentially considering the functional groups as surrogates of ecosystem condition. Functional feeding group metrics were based on the morphological structures and behaviors responsible for food acquisition by particular

taxa at a site (Resh and Jackson 1993; Danielson 1998a). As an example, herbivores consume algae and plant material, while predators consume animals, omnivores eat both plant and animal materials, and detritovores consume decomposed particulate material (Helgen 2001).

Wetland Condition Index

Candidate metrics were selected for inclusion in the WCI if they satisfied 3 criteria:

1. Metrics were correlated with the LDI according to the strength and significance of the Spearman's correlation coefficient
2. Displayed visually distinguishable correlations with LDI in scatter plots
3. Showed a significant difference between low and high LDI groups tested with the Mann-Whitney U-test.

A WCI was constructed for each assemblage, including the diatom WCI, the macrophyte WCI, and the macroinvertebrate WCI. Each index was composed of individual metrics specific to the assemblage, which were scaled and added together. Metric scoring was based on an approach modified from the Stream Condition Index, a Florida based biological index of the macroinvertebrate assemblage used to discern stream condition (Fore 2003). Metrics with a skewed distribution were log transformed to improve the distribution. The 5^{th} to 95^{th} percentile values of each metric were normalized from 0 to 10, with 10 always representing the best biological wetland condition. The selected metrics, WCI, and LDI were correlated with water and soil parameters using Spearman's correlation coefficient.

Cluster Analysis

In order to determine whether the WCI provided comparable scores for wetlands with similar community composition within each assemblage, an agglomerative cluster analysis in PCORD was used to determine wetland clusters. A further description is

available in McCune and Grace (2002). The dissimilarity matrix was constructed using the Sorensen distance measure and the flexible beta ($\beta = -0.25$) linkage method, which is a flexible clustering setting designed to reduce chaining in the dendrogram. The resulting dendrogram was pruned to maintain the smallest number of significantly different clusters based on Fisher's LSD pair wise comparison ($p < 0.05$).

Comparisons among Wetland Condition Index Metrics

Metrics selected for inclusion in the WCI were compared using the Pearson correlation coefficient (Analyse-it Software).

CHAPTER 3
RESULTS

Water and Soil Parameters

Water samples were analyzed for 75 wetlands, and soil samples were analyzed for all 118 isolated forested wetlands sampled. Table 3-1 shows mean values ± the standard deviation for water and soil parameters for the 3 *a priori* land use categories (reference, agricultural, and urban). Means with similar letters were not significantly different (Fisher's LSD pair wise comparison, $\alpha = 0.05$). Water temperature, water nitrate/nitrite-nitrogen, and soil TKN were not significantly different among *a priori* land use categories.

Reference wetlands had significantly different dissolved oxygen, turbidity, water pH, and water column total phosphorus, than agricultural and urban wetlands. The water color of urban wetlands was significantly different from reference and agricultural wetlands. Specific conductivity was significantly different between reference and urban wetlands. 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.

Table 3-2 shows the Mann-Whitney U-test results comparing water and soil parameters of low (LDI < 2.0) and high (LDI ≥ 2.0) LDI groups. Dissolved oxygen, turbidity, water pH, water TP, soil moisture, and soil TP were significantly different between LDI groups. Water temperature, specific conductivity, and water ammonia-

Table 3-1. Water and soil parameters among 3 *a priori* land use categories.

	Reference*	Agricultural*	Urban*
Water parameters			
Dissolved oxygen (mg/L)	2.9 ± 1.7[a]	1.6 ± 0.9[b]	1.9 ± 1.1[b]
Temperature (°C)	26.2 ± 2.8[a]	25.2 ± 1.9[a]	24.9 ± 2.4[a]
Color (PCU)	285 ± 178[a]	346 ± 204[a]	198 ± 129[b]
Turbidity (NTU)	3.8 ± 4.2[a]	17.7 ± 40.7[b]	9.5 ± 11.9[b]
pH	5.2 ± 1.2[a]	6.2 ± 0.8[b]	6.4 ± 1.0[b]
Specific conductivity (umhos/cm)	81 ± 48[a]	136 ± 134[ab]	231 ± 175[b]
Ammonia-nitrogen (mg N/L)	0.15 ± 0.33[a]	0.33 ± 0.57[b]	0.19 ± 0.27[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 ± 2.20[b]	1.84 ± 1.06[ab]
Total phosphorus (mg P/L)	0.08 ± 0.11[a]	0.81 ± 1.38[b]	0.23 ± 0.26[b]
Soil parameters			
Moisture (%)	61 ± 20[a]	46 ± 17[b]	55 ± 22[ab]
Organic matter (%)	40 ± 25[ab]	30 ± 17[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]
Total phosphorus (mg P/g soil)	0.38 ± 0.28[a]	0.91 ± 1.27[b]	0.53 ± 0.31[ab]

Values represent the mean ± standard deviation.
*Categories with similar letters were not significantly different (Fisher's LSD pair wise comparison, α=0.05).

nitrogen were significantly different between the LDI groups at the less strict $\alpha = 0.10$ level. Water color, water nitrate/nitrite-nitrogen, water TKN, soil organic matter, and soil TKN were not significantly different between LDI groups. Environmental variables with a VIF greater than 10.0 and a tolerance less than 0.10, including soil organic matter and soil TKN, were excluded from further use to avoid issues with multicollinearity (ter Braak 1987; Pan and Stevenson 1996; Ott and Longnecker 2001).

Diatoms

Statewide 50 wetlands were sampled with 214 diatom taxa identified at the species level or lower. Diatoms identified at the species level represented 98% of the sample. Five diatom species were identified at 50% or more of the sample wetlands ($n \geq 25$) including *Pinnularia subcapitata* (at 66% of the wetlands), *Eunotia bilunaris* (60%), *Nitzschia palea debilis* (60%), *Eunotia incisa* (54%), and *Gomphonema gracile* (50%).

Table 3-2. Water and soil parameters among LDI groups.

	Low LDI	High LDI	W^	p`
Water parameters				
Dissolved oxygen (mg/L)	2.8 ± 1.7	1.8 ± 1.0	1328.5	0.00*
Temperature (°C)	26.2 ± 2.7	24.9 ± 2.2	1242.0	0.06
Color (PCU)	272 ± 180	270 ± 178	1292.5	0.22
Turbidity (NTU)	3.7 ± 4.1	13.5 ± 28.4	953.1	0.02*
pH	5.4 ± 1.3	6.2 ± 0.9	797.5	0.00*
Specific conductivity (umhos/cm)	117 ± 129	177 ± 154	120.0	0.05
Ammonia-nitrogen (mg N/L)	0.15 ± 0.32	0.26 ± 0.43	1007.5	0.07
Nitrate/nitrite-nitrogen (mg N/L)	0.08 ± 0.36	0.01 ± 0.03	837.5	0.78
TKN (mg N/L)	1.89 ± 1.21	2.45 ± 1.77	1155.5	0.81
TP (mg P/L)	0.08 ± 0.11	0.50 ± 0.96	803.5	0.00*
Soil parameters				
Moisture (%)	59 ± 20	51 ± 36	2866.5	0.02*
Organic matter (%)	39 ± 25	36 ± 24	2588.0	0.50
TKN (mg N/g soil)	6.49 ± 3.69	6.25 ± 4.15	2534.0	0.60
TP (mg P/g soil)	0.36 ± 0.28	0.73 ± 0.94	1808.0	0.00*

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

The 3 diatoms identified most often included *Eunotia naegelii*, *Eunotia incisa*, and *Nitzschia palea debilis*. Of the diatoms encountered, 94 taxa (44%) occurred at a minimum of 5% of the sample wetlands ($n \geq 3$). Forty-one percent of the taxa identified (87 taxa) were encountered in only one wetland.

In the panhandle ecoregion, 10 wetlands were sampled with 4 reference, 4 agricultural, and 2 urban wetlands hosting 73 diatom taxa. In the north ecoregion 10 wetlands were sampled (4 reference, 2 agricultural, and 4 urban) with 94 taxa encountered. The central ecoregion included 13 wetlands (5 reference, 4 agricultural, and 4 urban) with 112 taxa sampled. The south ecoregion had 17 sample wetlands (6 reference, 5 agricultural, and 6 urban) with 147 taxa identified.

Summary Statistics

Richness (R), evenness (E), Shannon diversity (H), and Simpson's index (S) were calculated for each sample wetland (Appendix F). Table 3-3 summarizes the

Table 3-3. Diatom richness, evenness, and diversity among *a priori* land use categories.

	Reference	Agricultural	Urban
Species richness (R)	19 ± 8[a]	19 ± 6[a]	22 ± 8[a]
Species evenness (E)	0.74 ± 0.09[a]	0.75 ± 0.08[a]	0.73 ± 0.10[a]
Shannon diversity (H)	2.13 ± 0.49[a]	2.18 ± 0.34[a]	2.21 ± 0.49[a]
Simpson's index (S)	0.80 ± 0.09[a]	0.82 ± 0.07[a]	0.81 ± 0.10[a]
Beta diversity	6.9	6.1	5.8
Gamma diversity	132	117	126

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

richness, evenness, and diversity calculations of the diatom assemblage by *a priori* land use category. Species richness ranged from 9 taxa at CR6 and CU1 to 39 taxa at CU3 (surrounded by commercial and residential land uses). Species evenness ranged from 0.57 at NU5 (an urban wetland), to 0.89 at CA6 (surrounded by citrus groves). Shannon diversity ranged from 1.41 at NU5 to 2.95 at SR5 (surrounded by marsh and flooded flatwoods). Simpson's index was highest at SR5 and SU2 at 0.93, and lowest at NU5 at 0.58. Richness, evenness, Shannon diversity, and Simpson's index were not significantly different among the 3 *a priori* land use categories (Table 3-3) or between LDI groups (Table 3-4). Beta and gamma diversity were similar among *a priori* land use categories. Beta and gamma diversity were higher for the high LDI group (beta diversity of 8.2 and gamma diversity of 167) and lower for the low LDI group (beta diversity of 7.5 and gamma diversity of 145).

Compositional Analysis

MRPP was calculated across all groups (panhandle versus north versus central versus south) as well as for multiple pair wise comparisons (panhandle versus north, panhandle versus central, panhandle versus south, north versus central, north versus

Table 3-4. Mean diatom summary statistics between LDI groups.

	Low LDI	High LDI	W^	p`
Species richness (R)	19 ± 8	20 ± 7	484	0.61
Species evenness (E)	0.73 ± 0.09	0.74 ± 0.09	497	0.80
Shannon diversity (H)	2.13 ± 0.47	2.20 ± 0.43	486	0.64
Simpson's index (S)	0.80 ± 0.09	0.82 ± 0.08	480	0.55
Beta diversity	7.5	8.2		
Gamma diversity	145	167		

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

south, and central versus south) in order to test the similarity of diatom taxa composition among the ecoregions. Table 3-5 shows the results for the MRPP tests, including the test statistic (T), the chance-corrected within-group agreement (A, a measure of within group similarity), and the significance value (p). The global comparison among all wetlands and the 3 *a priori* categories showed that diatom community composition at the species level was significantly different ($\alpha = 0.05$). Within the pair wise comparisons, only the panhandle versus south and north versus south comparisons had significantly different diatom community composition for all land use types.

In the reference wetlands, the south ecoregion had a significantly different diatom community composition compared to both the panhandle and north ecoregions. Similarly, diatom community composition among pair wise comparisons of agricultural wetlands was significant different for the panhandle versus south ecoregions. The only ecoregions with significantly different diatom community composition among urban wetlands were the north and south ecoregions.

Community Composition

Figure 3-1 shows a 2 dimensional bi-plot of the NMS axes used to explore diatom community composition with overlays of significant environmental variables

Table 3-5. Similarity of diatom community composition using MRPP.

	Sites (n)	T^	A`	p#
All wetlands				
All regions (P vs N vs C vs S)	50	-2.2	0.01	0.03*
Panhandle vs north	20	0.5	-0.19	0.67
Panhandle vs central	23	0.9	-0.03	0.81
Panhandle vs south	27	-2.5	0.06	0.02*
North vs central	23	-0.7	0.02	0.20
North vs south	27	-3.9	0.09	0.00*
Central vs south	30	-1.7	0.04	0.06
Reference wetlands				
All regions (P vs N vs C vs S)	18	-1.3	0.09	0.11
Panhandle vs north	8	-0.7	0.07	0.21
Panhandle vs central	8	0.8	-0.07	0.76
Panhandle vs south	10	-2.1	0.16	0.03*
North vs central	8	0.9	-0.08	0.82
North vs south	10	-2.4	0.17	0.02*
Central vs south	10	0.3	-0.02	0.58
Agricultural wetlands				
All regions (P vs N vs C vs S)	16	0.1	-0.01	0.52
Panhandle vs north	6	0.6	-0.06	0.70
Panhandle vs central	9	0.1	-0.01	0.51
Panhandle vs south	9	-2.4	0.16	0.02*
North vs central	7	1.5	-0.19	0.93
North vs south	7	0.2	-0.02	0.52
Central vs south	10	-0.3	0.02	0.38
Urban wetlands				
All regions (P vs N vs C vs S)	16	-0.7	0.05	0.23
Panhandle vs north	6	-0.9	0.15	0.18
Panhandle vs central	6	0.9	-0.07	0.83
Panhandle vs south	8	-0.3	0.02	0.37
North vs central	8	-0.1	0.01	0.37
North vs south	10	-1.9	0.11	0.05*
Central vs south	10	1.1	-0.05	0.86

*A high |T| value and significant p-value ($p<0.05$) suggests a difference in species composition

^ T = the MRPP test statistic

`A = the chance corrected within-group agreement

#p = the significance value

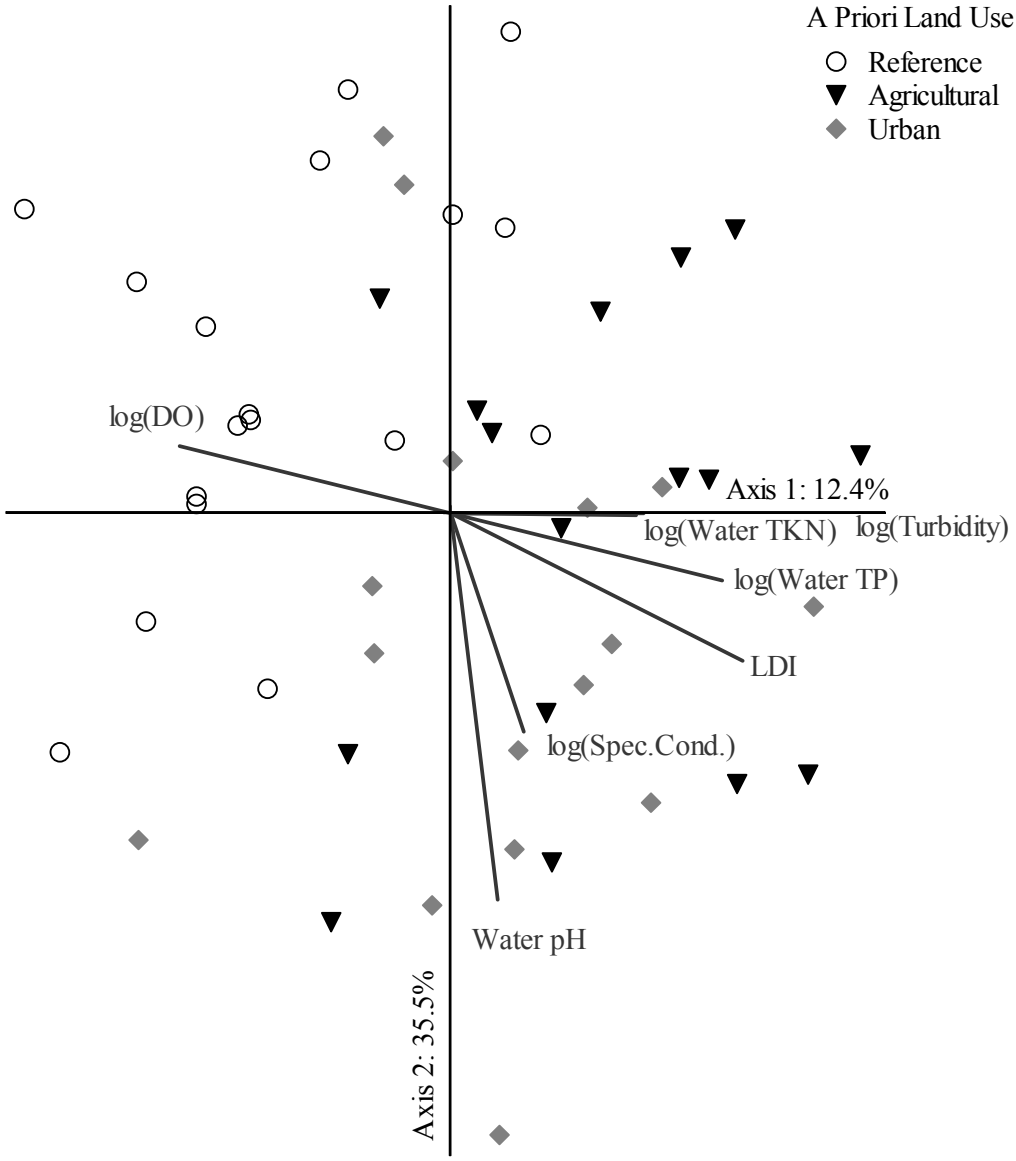


Figure 3-1. NMS ordination bi-plot of 50 wetlands in diatom species space with an overlay of environmental parameters. LDI, dissolved oxygen (DO), turbidity, water pH, specific conductivity (Spec.Cond.), water TKN, and water TP (shown as radiating vectors), were significantly correlated with the NMS axes based on diatom community composition. The length of the vector represents the strength of the correlation, and the angle represents the direction of maximum change. Axis 1 explained 12.4% variance, axis 2 explained 35.5% variance, and axis 3 (not shown) represented an additional 26.7% variance.

Table 3-6. Pearson correlations between environmental parameters and NMS ordination axes based on diatom community composition.

	Axis 1	Axis 2	Axis 3
Incremental r^2	12.4%	35.5%	26.7%
Cumulative r^2	12.4%	47.9%	74.6%
Latitude	0.06	0.12	0.16
Longitude	0.03	0.05	0.04
LDI	0.35	0.18	0.05
Log (DO)	0.33	0.08	0.09
Log (Temperature)	0.03	0.08	0.07
Log(Color)	0.15	0.09	0.17
Log(Turbidity)	0.22	0.01	0.00
pH	0.06	0.47	0.45
Log(Spec.Cond.)	0.09	0.27	0.06
Log(Ammonia-N)	0.13	0.00	0.03
Log (Nitrate/nitrite-N)	0.01	0.01	0.05
Log(TKN)	0.23	0.00	0.01
Log(Water TP)	0.33	0.08	0.00
Arcsin Sqrt (%Soil moisture)	0.11	0.00	0.08
Log(Soil TP)	0.07	0.05	0.02

(transformed), including LDI and 6 water parameters: log(DO), log(turbidity), pH, log(Spec.Cond.), log(TKN), and log(TP). Table 3-6 provides the Pearson's r-squared correlation coefficients between the environmental variables and the NMS ordination axes. A 3 dimensional solution was constructed with an overall stress of 16.3 and a final stability of 0.00001, which is borderline high but an acceptable stress limit for a useful ordination with community data (Kruskal 1964; Clarke 1993; McCune and Grace 2002). Axis 1 explained 12.4% of the variance and was correlated with LDI, log(DO), log(water TKN), and log(water TP). Axis 2 explained 35.5% variance and was correlated with pH and log(Spec.Cond). Axis 3 explained an additional 26.7% variance and was correlated with water column pH.

Metric Selection

Twenty-eight of the candidate metrics were significantly correlated with LDI ($p < 0.05$). Due to the redundant nature of some candidate metrics and strong correlations among metrics (Pearson's $r^2 > 0.9$), 7 metrics which were significant for both the Spearman's correlation coefficient ($|r| > 0.45$, $p < 0.01$) and the Mann-Whitney U-test between LDI groups ($p < 0.10$) were selected for inclusion in the diatom WCI. Table 3-7 provides the Spearman's correlation values between the 7 metrics and LDI.

Metrics selected for inclusion represented 2 of the metric categories, including tolerance metrics and autecological metrics. Tolerance metrics included percent tolerant and sensitive indicator species. The 5 autecological metrics included pollution class 1 (very tolerant to pollution), nitrogen uptake metabolism class 3 (need periodically elevated concentrations of organically bound nitrogen), saprobity class 4 (inhabit aquatic environments with an oxygen saturation between 10-25% and a biological oxygen demand of approximately 13-22 mg/L), pH class 3 (circumneutral, mainly occurring at pH values around 7), and dissolved oxygen class 1 (requiring continuously high dissolved oxygen concentrations near 100%). Pollution class was established by Bahls (1993), and nitrogen metabolism, saprobity, pH, and dissolved oxygen classes were defined by

Table 3-7. Spearman's correlations for 7 diatom metrics and LDI. All correlations were significant ($p < 0.01$).

Diatom Metrics	Spearman's r
Tolerant indicator species	0.65
Sensitive indicator species	-0.60
Pollution class 1	0.52
Nitrogen class 3	0.48
Saprobity class 4	0.48
pH class 3	0.48
Dissolved oxygen class 1	-0.46

Table 3-8. Comparisons among diatom metrics and the diatom WCI for LDI groups.

Metric	Low LDI	High LDI	W^	p`
Tolerant indicator species	1.4 ± 2.5	35.5 ± 15.8	309.0	<0.001
Sensitive indicator species	36.6 ± 25.0	18.0 ± 18.5	719.0	<0.001
Pollution class 1	3.7 ± 6.0	7.6 ± 10.1	337.0	<0.001
Nitrogen class 3	7.7 ± 15.5	20.3 ± 21.5	361.0	0.003
Saprobity class 4	2.7 ± 4.5	25.4 ± 25.6	373.5	0.006
pH class 3	18.3 ± 22.2	14.1 ± 16.6	344.0	0.001
Dissolved oxygen class 1	69.3 ± 24.3	39.9 ± 23.8	658.0	0.004
Diatom WCI	55.3 ± 11.1	45.4 ± 27.1	718.0	<0.001

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

van Dam et al. (1994). Tolerant indicator species, pollution class 1, nitrogen metabolism class 3, saprobity class 4, and pH class 3 increased with increasing development intensity; whereas, sensitive indicator genera and dissolved oxygen class 1 decreased with increasing development intensity. Table 3-8 shows all of the selected metrics differentiated between the 2 LDI groups.

Tolerance metrics

Table 3-9 shows the results of the iterative Indicator Species Analysis (ISA) calculations using species-level abundance data for the diatom assemblage. Tolerant diatom indicator species were established at an LDI break of 5.0, and included 12 species representing 6 genera. Table 3-10 lists the tolerant diatom indicator species. The 3 tolerant indicator species with the highest indicator values were all in the genera *Navicula*, including *N. minima*, *N. confervacea*, and *N. mutica*.

Figure 3-2 shows the percent tolerant diatom indicator species increased with increasing landscape development intensity. Wetlands with the highest percent tolerant indicator species were in the low LDI group, including CR5 and SU3. While CR5 was

Table 3-9. Spearman correlations of diatom indicator species over a range of LDI values. Highlighted areas indicate the LDI value selected for sensitive and tolerant diatom indicator species.

LDI	Low LDI n* =	High LDI n* =	Sensitive r^	Sensitive p`	Tolerant r^	Tolerant p`	No. Sensitive Indicators	No. Tolerant Indicators
1	8	42	-0.48	0.000	0.58	0.000	14	3
1.25	16	34	-0.60	0.000	0.59	0.000	18	8
1.5	19	31	-0.57	0.000	0.58	0.000	14	8
1.75	20	30	-0.57	0.000	0.62	0.000	12	7
2	20	30	-0.57	0.000	0.62	0.000	12	7
2.25	21	29	-0.56	0.000	0.63	0.000	11	9
2.5	21	29	-0.56	0.000	0.63	0.000	11	9
2.75	21	29	-0.56	0.000	0.63	0.000	11	9
3	21	29	-0.56	0.000	0.63	0.000	11	9
3.25	22	28	-0.61	0.000	0.63	0.000	10	10
3.5	22	28	-0.61	0.000	0.63	0.000	10	10
3.75	22	28	-0.61	0.000	0.63	0.000	10	10
4	24	26	-0.56	0.000	0.56	0.000	6	13
4.25	25	25	-0.57	0.000	0.57	0.000	6	8
4.5	26	24	-0.48	0.000	0.56	0.000	5	12
4.75	30	20	-0.55	0.000	0.60	0.000	5	17
5	35	15	-0.52	0.000	0.65	0.000	2	12
5.25	39	11	x	x	0.59	0.000	0	11
5.5	41	9	-0.41	0.003	0.59	0.000	1	14
5.75	42	8	-0.45	0.001	0.54	0.000	1	13
6	42	8	-0.45	0.001	0.54	0.000	1	13
6.25	45	5	x	x	0.55	0.000	0	21
6.5	47	3	x	x	0.41	0.003	0	13
6.75	47	3	x	x	0.41	0.003	0	13
7	48	2	x	x	0.45	0.001	0	23

*n = number of sites
^r = Spearman's r correlation coefficient of indicator species versus LDI
`p = significance value

categorized a reference wetland, it was located in a fragmented state park within a small, highly developed county (Seminole County near Orlando, Florida). SU3 was an urban wetland surrounded by nearly 100 m of marsh that received nutrient enriched water. The 2 wetlands with the highest percent tolerant indicator species in the high LDI group

Table 3-10. Diatom tolerant indicator species. All reported tolerant indicator species calculated at an LDI break of 5.0 were significant ($p < 0.10$).

Indicator Species (LDI>5.0)	Indicator Value	p-value
Cyclotella pseudostelliger	20.0	0.021
Diploneis elliptica	19.7	0.045
Navicula confervacea	45.3	0.006
Navicula minima	48.0	0.050
Navicula mutica	40.5	0.044
Navicula recens	12.9	0.100
Navicula subminuscula	12.5	0.083
Neidium alpinum	20.0	0.021
Nitzschia subacicularis	18.1	0.080
Pinnularia braunii	22.6	0.043
Pinnularia divergentissima	13.3	0.085
Stauroneis kriegeri	13.3	0.085

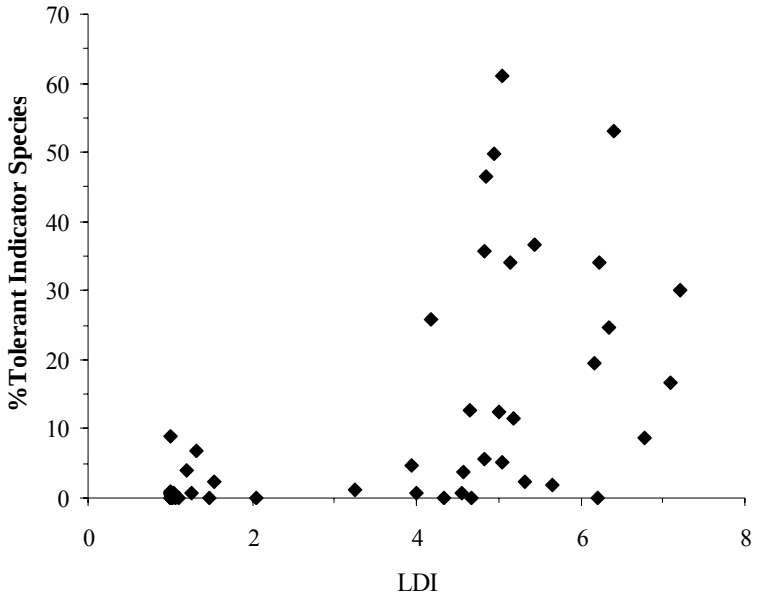


Figure 3-2. Percent diatom tolerant indicator species increased with increasing development intensity (LDI).

included SU1 and CU5. Four agricultural and urban wetlands and 10 reference wetlands had no tolerant diatom indicator species present.

Sensitive diatom indicator species were selected at an LDI break of 1.25, correlating to a break in the natural and developed land uses. Table 3-11 lists the 18 statewide sensitive indicator species. The 5 sensitive indicator species with the highest indicator values included *Eunotia naegelii*, *E. rhomboidea*, *Frustulia rhomboides*, *Anomoeoneis brachysira*, and *Desmogonium rabenhorstianum*. Figure 3-3 shows that the percent sensitive indicator species decreased with increasing development intensity in the surrounding landscape. Eighty-three percent of the high LDI wetlands hosted less than 10% sensitive indicator species. Of these 5 sites with greater than 10% sensitive

Table 3-11. Diatom sensitive indicator species. All reported sensitive indicator species calculated at an LDI break of 1.25 were significant ($p<0.10$).

Indicator Species (LDI<1.25)	Indicator Value	p-value
Anomoeoneis brachysira	39.2	0.008
Cymbella microcephala	12.5	0.097
Desmogonium rabenhorstianum	33.5	0.010
Encyonema silesiacum	24.4	0.100
Eunotia flexuosa	12.5	0.094
Eunotia glacialis	17.0	0.035
Eunotia intermedia	28.1	0.016
Eunotia naegelii	59.2	0.002
Eunotia pectinalis undulata	26.9	0.038
Eunotia rhomboidea	45.9	0.013
Frustulia rhomboides	41.6	0.069
Frustulia rhomboides crassinervia	18.7	0.038
Navicula capitatoradiata	11.5	0.091
Navicula subtilissima	12.5	0.079
Nitzschia nana	17.0	0.064
Nitzschia paleacea	18.7	0.034
Pinnularia streptoraphe	18.7	0.031
Rhopalodia gibba	18.7	0.031

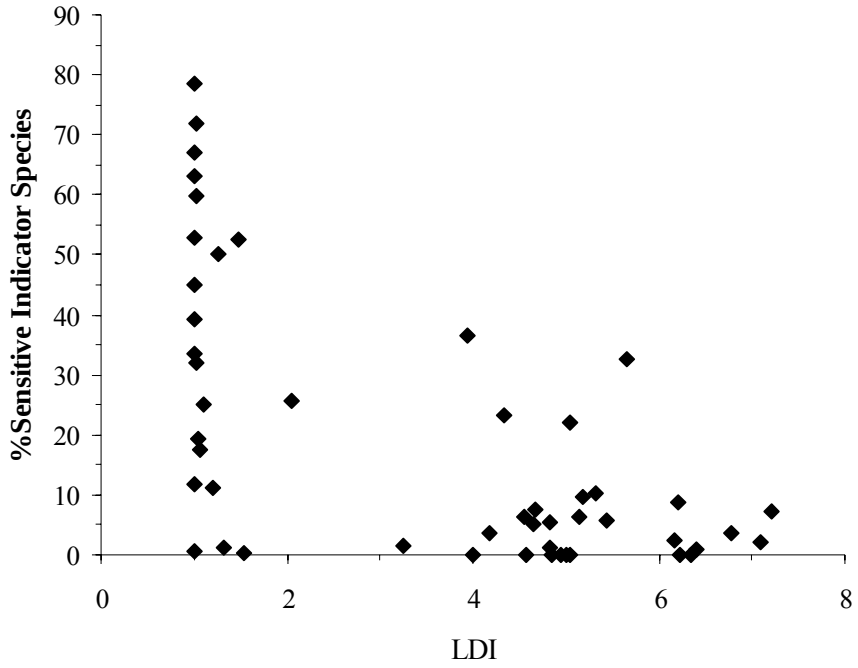


Figure 3-3. Percent diatom sensitive indicator species decreased with increasing development intensity (LDI).

indicator species, 2 were agricultural wetlands embedded in pasture including PA2 and CA4, and 3 were set in recently developed urban landscapes including CU1, NU6, and SU6. Six agricultural and 2 urban wetlands hosted no sensitive indicator species.

Autecological metrics

Between 56-69% of diatoms identified received scores based on established autecological relationships (van Dam et al. 1994; Bahls 1993). Five metrics based on scoring diatoms from a coded checklist describing their autecology were incorporated in the diatom WCI, including the proportion of diatoms in pollution class 1, nitrogen uptake metabolism class 3, saprobity class 4, pH class 3, and dissolved oxygen requirement class 1. Figure 3-4 shows that the proportion of diatoms in pollution class 1 increased with increasing development intensity in the surrounding landscape. Diatoms in pollution class 1 were very tolerant to pollution, as compared to pollution class 2 (moderately

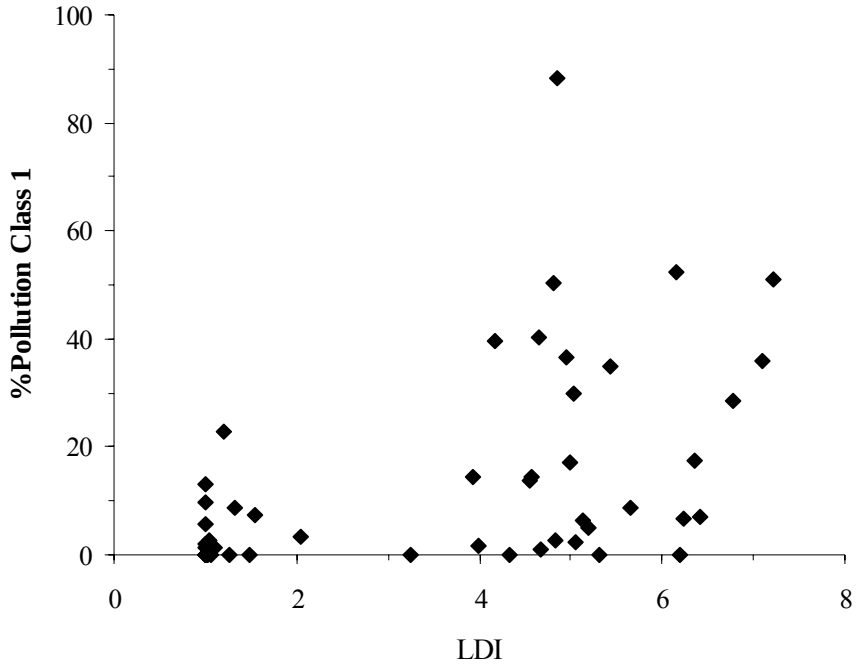


Figure 3-4. Pollution tolerance class 1 diatoms increased with increasing development intensity (LDI).

tolerant) or pollution class 3 (sensitive to pollution; Bahls 1993). A background level of approximately 10% of the diatoms belonging to pollution class 1 distinguishes low and high LDI wetlands. Two exceptions in the low LDI group include CR5 (pollution class 1 = 13%) and SR5 (pollution class 1 = 23%). The wetland with the greatest percent of diatoms in pollution class 1 was CA3 (LDI 4.9, pollution class 1 = 88%), a wetland that received waters carrying wastes from a pullet farm operation.

Figure 3-5 shows that the proportion of diatoms in nitrogen uptake metabolism class 3 increased with increasing development intensity in the surrounding landscape. Membership in nitrogen uptake metabolism class 3 was defined by facultative nitrogen-heterotrophic taxa that need periodically elevated concentrations of organically bound nitrogen (van Dam et al. 1994). Eighty percent of the low LDI group wetlands had less than 10% of the diatoms in nitrogen uptake metabolism class 3. Four outliers in the low

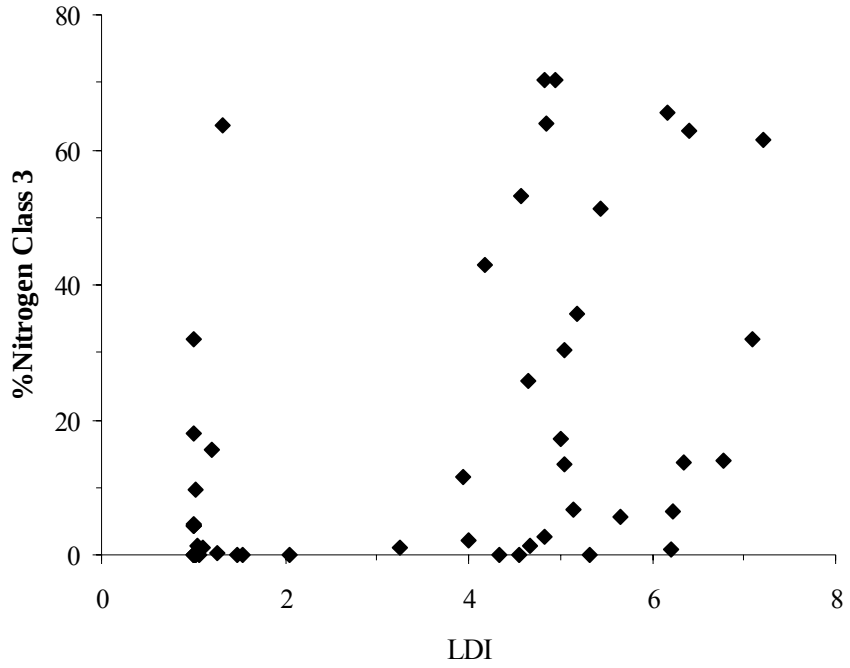


Figure 3-5. Nitrogen uptake metabolism class 3 diatoms increased with increasing development intensity (LDI).

LDI group included SU3, SR6, SR4, and SR5. These 3 southern reference wetland outliers were within state or federal lands protected as part of the Florida Everglades. The urban outlier, SU3 was surrounded by nearly 100 m of marsh that has received nutrient enriched waters since the mid 1970s.

The percent of diatoms in saprobity class 4 increased with increasing landscape development intensity (Figure 3-6). Diatoms characterized as belonging to saprobity class 4 included meso- to poly-saprobous species (inhabit aquatic environments with an oxygen saturation between 10-25 % and a biological oxygen demand (BOD_5^{20}) of 13-22 mg/L) (van Dam et al. 1994). Eighty-five percent of the wetlands in the low LDI group had less than 5% of diatoms in saprobity class 4. The 3 low LDI wetlands with over the 5% threshold included SU3, PR1, and SR6. Over 50% of the wetlands in the high LDI

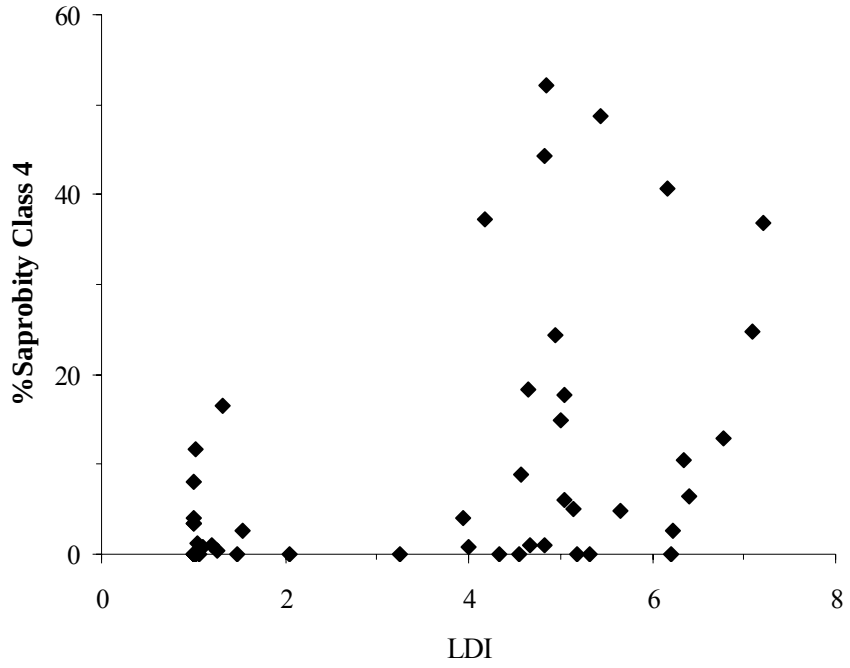


Figure 3-6. Saprobity class 4 diatoms increased with increasing development intensity (LDI).

group had greater than 5% of diatoms in saprobity class 4. CA3 hosted the greatest percent of diatoms in saprobity class 4 (saprobity class 4 = 52%).

Figure 3-7 shows that the percent of diatoms in pH class 3 increased with increasing landscape development intensity. Diatoms in pH class 3 were described as circumneutral (mainly occurring at pH values of approximately 7) (van Dam et al. 1994). Of the wetlands in the low LDI group, 70% had less than 20% of pH class 3 diatoms; whereas, 73% of wetlands in the high LDI group had greater than 20% of diatoms in pH class 3. In the low LDI group, the greatest outlier was SU4 (pH class 3 = 89%).

Diatoms requiring continuously high dissolved oxygen concentrations of approximately 100% saturation (dissolved oxygen class 1) decreased with increasing development intensity in the surrounding landscape (Figure 3-8). In the low LDI group, the same 6 outliers occurred as in the pH class 3 metric; with SU3 having the lowest

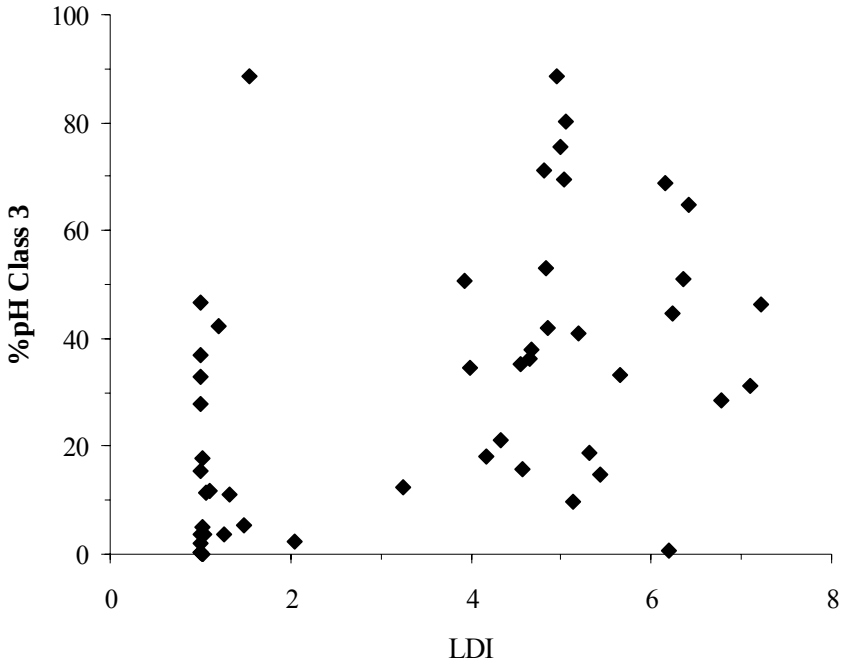


Figure 3-7. The pH class 3 diatoms increased with increasing development intensity (LDI).

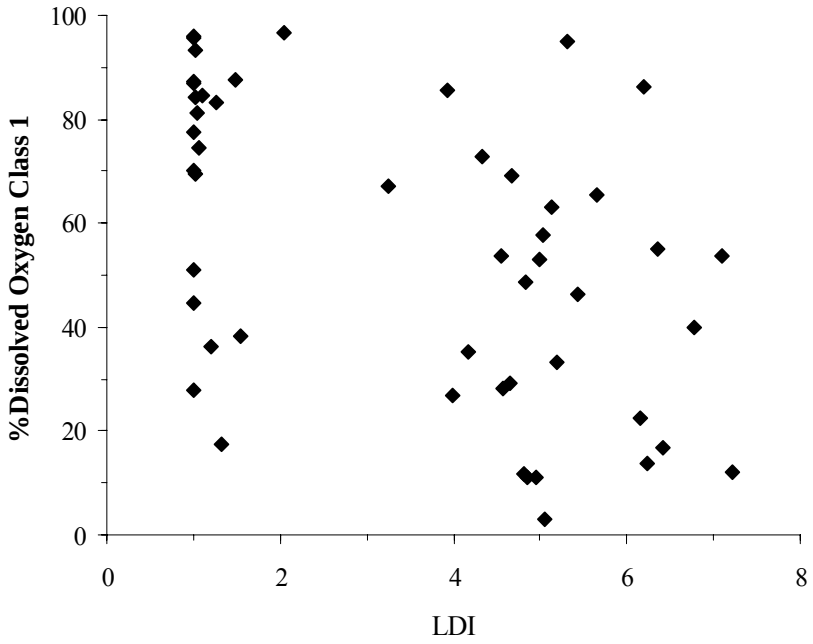


Figure 3-8. Dissolved oxygen class 1 diatoms decreased with increasing development intensity (LDI).

percent diatoms in dissolved oxygen class 1 (18%). Seventy percent of wetlands in the low LDI group had greater than 60% diatoms in dissolved oxygen class 1.

Diatom Wetland Condition Index

The seven metrics described above were scored and added together to create the diatom WCI. Figure 3-9 shows the relationship between the diatom WCI and LDI. Potential scores for the diatom WCI ranged from 0-70, with higher values representing wetlands surrounded by undeveloped landscapes. Actual scores ranged from 8 at CA3 (an agricultural wetland receiving inputs from a spray field associated with pullet farm wastes), to 69 at SR2 (a wetland surrounded by flooded flatwoods and marsh). The next highest scoring wetlands received diatom WCI scores 2 points lower than SR2, with a score of 67 at both NR3 and SR1.

Diatom WCI ranges varied regionally, with the highest scores in each region including PR4 (65), NR3 (67), CR6 (65), and SR2 (69), in the panhandle, north, central, and south ecoregions, respectively. The lowest scores in the panhandle and north ecoregions were for urban wetlands embedded in residential land use, including PU4 (11) and NU2 (24). Wetlands surrounded by agricultural land uses received the lowest scores in the central and south ecoregions, including CA3 (8) and SA4 (16). The diatom WCI was robustly correlated with the LDI index (Spearman correlation $|r| = 0.64$, $p < 0.001$). A Kruskal-Wallis test between median diatom WCI values suggested a significant difference ($H = 20.7$, $p < 0.001$) among wetlands in the 3 *a priori* land use categories.

Cluster Analysis

Cluster analysis determined 4 categories based on diatom community composition. Using site descriptions, clusters were explained by regions, *a priori* land use categories, and water level including: 1: wetlands in the panhandle to central ecoregions with low

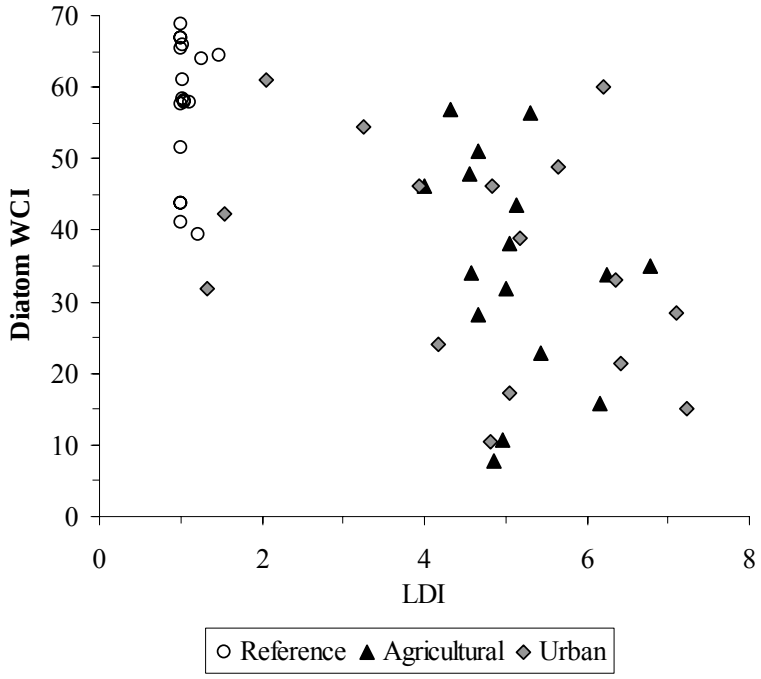


Figure 3-9. Diatom WCI scores decreased with increasing development intensity (LDI). Sample wetlands are designated by *a priori* land use category: reference, agricultural, or urban.

development intensity; 2: wetlands occurring in mixed ecoregions with low development intensity; 3: wetlands within the southern Everglades ; and 4: wetlands within mixed regions surrounded by high development intensity. Figure 3-10 shows that based on the diatom WCI scores, clusters 1 and 2 were not significantly different from one another, but were significantly different from both cluster 3 and cluster 4 ($p<0.05$). Clusters 3 and 4 were significantly different from each other. Table 3-12 provides means and standard deviations for D-WCI scores and LDI of the four diatom based clusters. Cluster 4 had significantly different D-WCI and LDI scores than all other clusters.

Macrophytes

Statewide, 118 wetlands were sampled with 605 species, representing 323 genera and 126 families identified. The most abundant species was *Taxodium ascendens*, which

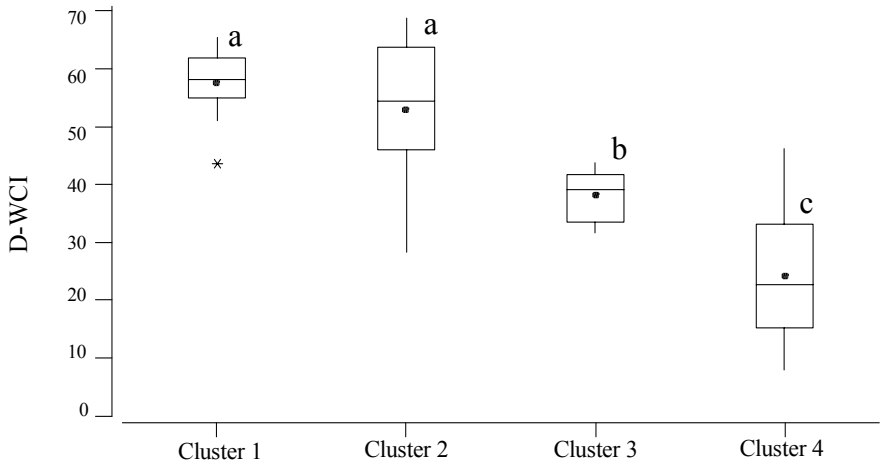


Figure 3-10. Diatom WCI scores for wetland clusters based on diatom community composition. Boxes represent the interquartile range; solid circles represent the mean; middle lines represent the median; whiskers represent the range; asterisks represent outliers (> ±2 standard deviations). Clusters with similar letters were not significantly different (Fisher's LSD, $p<0.05$).

was found rooted within the vegetation quadrats at 93% of the study wetlands. The second most abundant species was *Myrica cerifera* found in 64% of the study wetlands.wetlands. The most common fern was *Woodwardia virginica* found at 53% of the wetlands; the most common vine was *Toxicodendron radicans* also found at 53% of the wetlands; and the most common graminoid was *Panicum hemitomon* found at 50% of the wetlands. Of the species encountered, 130 species (22%) occurred at a minimum of 5% of the sample wetlands ($n \geq 6$). Approximately one-third of the species identified (202 species or 33.5%) were rooted in the vegetation quadrats at only one wetland.

In the panhandle ecoregion, 28 wetlands were sampled hosting 328 species, representing 191 genera and 90 families. In the north ecoregion 31 wetlands were

Table 3-12. Diatom WCI and LDI values for wetland clusters based on diatom community composition.

	Cluster*			
	1	2	3	4
Diatom WCI	57.6 ± 6.4[a]	52.9 ± 11.6[a]	38.1 ± 4.4[b]	24.1 ± 11.7[c]
LDI	2.9 ± 2.1[a]	2.8 ± 2.0[a]	2.4 ± 1.9[a]	5.3± 1.4[b]

* Clusters with similar letters were not significantly different (Fisher's LSD, $p<0.05$).

sampled with 306 species (180 genera and 89 families) encountered. The central ecoregion included 31 wetlands with 329 species (202 genera and 94 families) sampled. The south ecoregion had 28 sample wetlands with 266 species (in 180 genera and 89 families) identified.

Summary Statistics

Species richness (R), first and second order jackknife estimators ($Jack_1$ and $Jack_2$, respectively), species evenness (E), Shannon diversity (H), and Whittaker's beta diversity (β_W) were calculated based on the macrophyte assemblage for each sample wetland (Appendix F). Species richness ranged from 13 species at NA3 (embedded in silvicultural land use), to 77 species at NA12 (surrounded with pasture and row crops). The greatest estimates of species richness were 99 species and 114 species at NA12, for first and second order jackknife estimators, respectively. Sampled species richness at NA12 was 77 species. The lowest estimates of actual species richness were for NR1, with 11 and 15 species estimated with first and second order jackknife estimators, respectively. Sampled species richness for NR1 was 14 species. Species evenness ranged from 0.71 at PR7 (a large, deep water wetland on a private conservation tract), to 0.93 at PU4 (a wetland surrounded by a residential community and park). Shannon diversity ranged from 1.8 to 3.9 at two agricultural wetlands, NA3 and NA12,

respectively, similar to species richness. Whittaker's beta diversity ranged from a low of 0.2 at PU9 (an urban wetland surrounded by residential land use), to a high of 9.4 at CR5 (a deep water wetland on state land).

Table 3-13 summarizes comparisons of mean richness and diversity calculations by *a priori* land use category. Agricultural wetlands had the greatest species richness followed closely by urban wetlands. This same trend was evident for species evenness, with agricultural wetlands having greater species evenness. Diversity indices yielded similar results, with reference wetlands having lower Shannon diversity and Whittaker's beta diversity then both agricultural and urban wetlands. Beta and gamma diversity were calculated for *a priori* land use categories, with urban wetlands having the highest beta diversity and agricultural wetlands the highest gamma diversity. Only Whitaker's beta diversity was significantly different among the *a priori* land use categories (Fisher's LSD pair wise comparison, $\alpha = 0.05$). Species richness and Whittaker's beta diversity were not significantly different between low (LDI < 2.0) and high (LDI ≥ 2.0) LDI groups (Table 3-14); whereas, species evenness and Shannon diversity were significantly different ($p < 0.05$) between LDI groups (Mann-Whitney U-Test).

Compositional Analysis

MRPP was calculated across all groups (panhandle versus north versus central versus south) as well as for multiple pair wise comparisons (panhandle versus north, panhandle versus central, panhandle versus south, north versus central, north versus south, and central versus south). Table 3-15 shows the results for the MRPP tests, including the test statistic (T), chance-corrected within-group agreement (A), and significance value (p). Only 2 of the MRPP comparisons (agricultural wetlands in the panhandle and north ecoregions and central and south ecoregions) were not significant at

Table 3-13. Mean macrophyte richness, evenness, and diversity among *a priori* land use categories.

	Reference	Agricultural	Urban
Species richness (R)	32 ± 11[a]	37 ± 14[a]	36 ± 10[a]
Species evenness (E)	0.85 ± 0.04[a]	0.87 ± 0.04[a]	0.86 ± 0.03[a]
Shannon diversity (H)	2.9 ± 0.3[a]	3.1 ± 0.4[a]	3.1 ± 0.3[a]
Whittaker's Beta diversity (β_W)	4.0 ± 2.9[a]	4.8 ± 1.7[b]	4.3 ± 1.7[ab]
Beta diversity	9.5	10.4	10.5
Gamma diversity	304	383	378

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

the $\alpha = 0.05$ level, suggesting that there were regionally significant differences among species composition across all regions and within *a priori* land use categories.

Community Composition

Macrophyte community composition was summarized in 2 NMS ordinations to relate changes in macrophyte community composition with environmental variables. Figure 3-11 shows a two dimensional bi-plot of the NMS axes used to explore the dissimilarities of macrophyte community composition with overlays of significant environmental variables, including log(soil TP), LDI, latitude, and longitude. Table 3-16 provides the Pearson correlation coefficients between environmental variables and NMS

Table 3-14. Mean macrophyte richness, evenness, and diversity between LDI groups.

	Low LDI	High LDI	W^	p`
Species richness (R)	33 ± 10	37 ± 12	2120	0.07
Species evenness (E)	0.85 ± 0.04	0.87 ± 0.04	2028	0.02*
Shannon diversity (H)	2.9 ± 0.3	3.1 ± 0.4	2062	0.03*
Whittaker's Beta diversity (β_W)	4.1 ± 1.9	4.5 ± 1.7	2131	0.08
Beta diversity	10.2	13.8		
Gamma diversity	338	510		

* Indicates significance at α<0.05
^W = Mann-Whitney U-test statistic
`p = significance value

Table 3-15. Macrophyte community composition similarity among ecoregions with MRPP.

	Sites (n)	T^	A`	p#
All wetlands				
All regions (P vs N vs C vs S)	118	-26.8	0.06	0.00*
Panhandle vs north	59	-7.1	0.03	0.00*
Panhandle vs central	59	-13.1	0.04	0.00*
Panhandle vs south	56	-23.6	0.09	0.00*
North vs central	62	-7.9	0.03	0.00*
North vs south	59	-24.2	0.09	0.00*
Central vs south	59	-11.8	0.04	0.00*
Reference wetlands				
All regions (P vs N vs C vs S)	37	-12.5	0.12	0.00*
Panhandle vs north	17	-5.2	0.08	0.00*
Panhandle vs central	19	-6.8	0.09	0.00*
Panhandle vs south	17	-8.1	0.14	0.00*
North vs central	20	-4.7	0.06	0.00*
North vs south	18	-8.5	0.14	0.00*
Central vs south	20	-7.1	0.07	0.00*
Agricultural wetlands				
All regions (P vs N vs C vs S)	40	-6.6	0.05	0.00*
Panhandle vs north	22	-0.7	0.01	0.21
Panhandle vs central	19	-2.8	0.03	0.01*
Panhandle vs south	19	-8.1	0.10	0.00*
North vs central	21	-2.0	0.02	0.04*
North vs south	21	-7.7	0.09	0.00*
Central vs south	18	-1.4	0.02	0.09
Urban wetlands				
All regions (P vs N vs C vs S)	41	-15.3	0.14	0.00*
Panhandle vs north	20	-6.0	0.07	0.00*
Panhandle vs central	21	-9.0	0.10	0.00*
Panhandle vs south	20	-11.1	0.17	0.00*
North vs central	21	-3.3	0.03	0.00*
North vs south	20	-10.5	0.15	0.00*
Central vs south	21	-8.6	0.09	0.00*

*A high |T| value and significant p-value ($p<0.05$) suggests a difference in species composition.

^ T = the MRPP test statistic

`A = the chance corrected within-group agreement

#p = the significance value.

ordination axes. A 3 dimensional solution was constructed with an overall stress of 20.8 with a final stability of 0.06, which is a fairly high stress limit but considered useful for ordinations with community data sets (Kruskal 1964; Clarke 1993; McCune and Grace 2002). Axis 1 explained 29.1% variance and was correlated with latitude and longitude; axis 2 explained 34.2% variance and was correlated with LDI and log(soil TP). Axis 3 explained an additional 12.1% of the variance and was not correlated with soil parameters.

A second NMS ordination was completed using the macrophyte species composition at the 75 sample wetlands with measured water parameters. Figure 3-12 shows the bi-plot from the NMS ordination. The final stress was 16.6 with a final instability of 0.004. The ordination explained a cumulative 77.4% of the variance in wetland macrophyte community composition. Table 3-17 shows the Pearson r-squared correlation coefficient values for the environmental parameters and the three ordination axes. Axis 1 explained 31.3% of the variance and was correlated with latitude and longitude. Axis 2 was correlated with LDI, water pH, log(water TP), log(soil TP), and log(DO), and explained 25.7% of the variance. Axis 3 explained an additional 20.4% of the variance and was correlated with water pH and Arcsin Sqrt (soil moisture). log(Water N) concentration (ammonia, nitrate/nitrite, and TKN), log(Temperature), log(color), and Log(turbidity) were not strongly correlated with the NMS ordination axes.

Metric Selection

Over 35 of the candidate metrics were significantly correlated with LDI. Due to the redundant nature of some candidate metrics and the multiple forms of calculations (number, percent, proportion, frequency of occurrence), 6 metrics that were significant for both the Spearman's correlation coefficient ($|r|>0.50$, $p<0.001$) and the Mann-

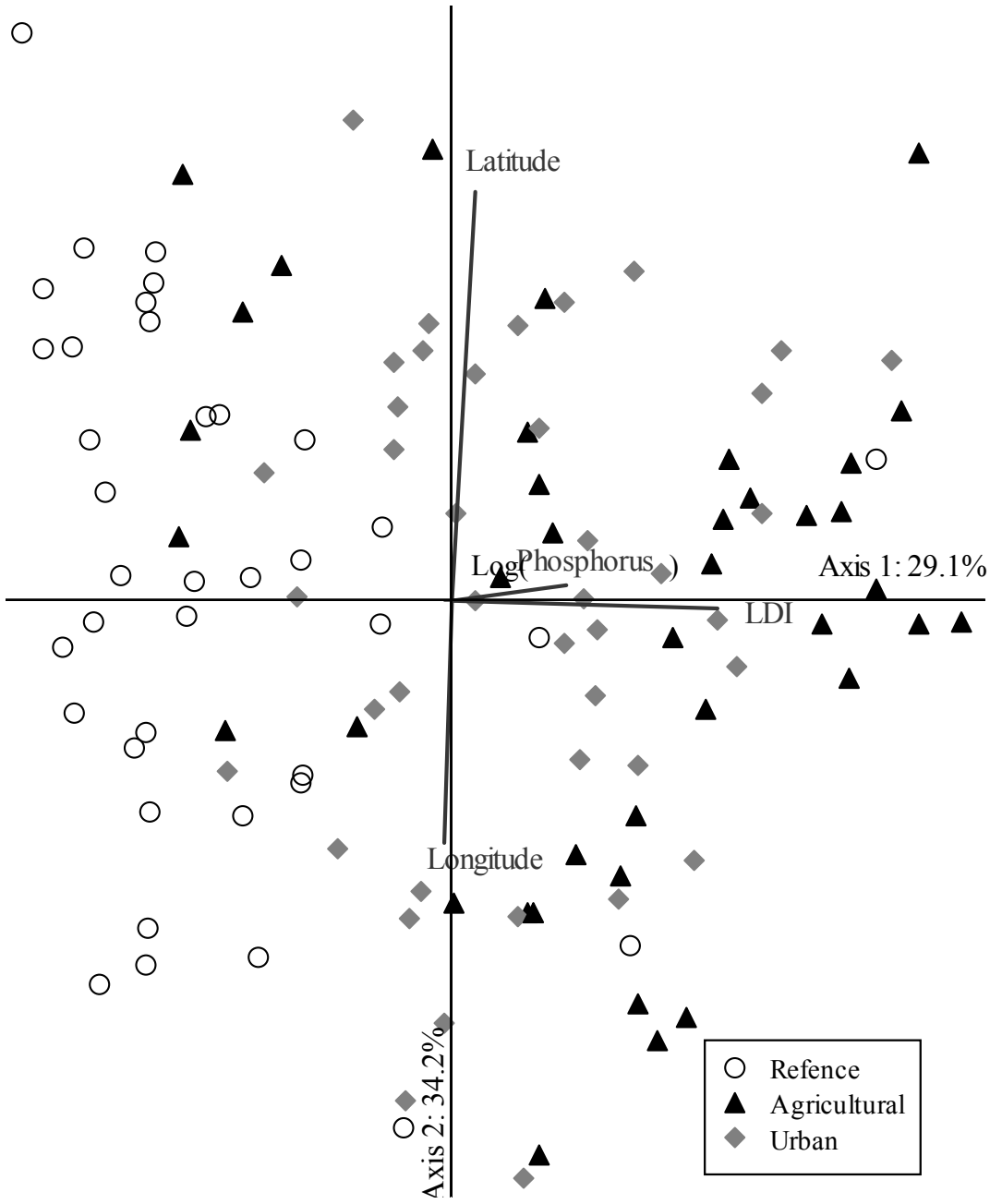


Figure 3-11. NMS ordination bi-plot of 118 sample wetlands in macrophyte species space with an overlay of environmental parameters. Latitude, longitude, LDI, and log(soil TP), shown as radiating vectors, were significantly correlated with NMS axes. Vector length represents the strength of the correlation, and the angle represents the direction of maximum change. Axis 1 explained 29.1% variance, axis 2 explained 34.2% variance, and axis 3 (not shown) represented an additional 12.1% variance.

Table 3-16. Pearson correlations between environmental variables and NMS axes based on macrophyte community composition at 118 wetlands.

	Axis 1	Axis 2	Axis 3
Incremental r^2	29.1%	34.2%	12.1%
Cumulative r^2	29.1%	63.3%	75.4%
Latitude	0.71	0.04	0.00
Longitude	0.42	0.02	0.00
LDI	0.01	0.46	0.01
Arcsin Sqrt (Soil moisture)	0.00	0.11	0.04
Log (Soil TKN)	0.03	0.06	0.00
Log(Soil TP)	0.03	0.20	0.00

macrophyte WCI. Table 3-18 provides the statewide Spearman correlation values Whitney U-test between LDI groups ($p<0.001$) were selected for inclusion in the between the 6 macrophyte metrics and LDI. Metrics selected for inclusion were tolerant and sensitive indicator species; modified Floristic Quality Index (FQI); exotic species; native perennial species; and wetland status species. Tolerant indicator and exotic species increased with increasing development intensity; whereas, sensitive indicator species, modified FQI, native perennial species, and wetland status species decreased with increased landscape development. Table 3-19 shows significant differences of metrics between low and high LDI groups.

Tolerance metrics

Multiple ISAs were completed at different LDI breaks, starting at 1.0 and continuing through 7.0, at 0.25 step increments. Table 3-20 shows the results of the iterative ISA calculations. The greatest number of statewide tolerant indicator species was established at an LDI break of 4.0, and the greatest number of statewide sensitive indicator species was found at an LDI break of 2.0. These break points were used for successive ISA calculations.

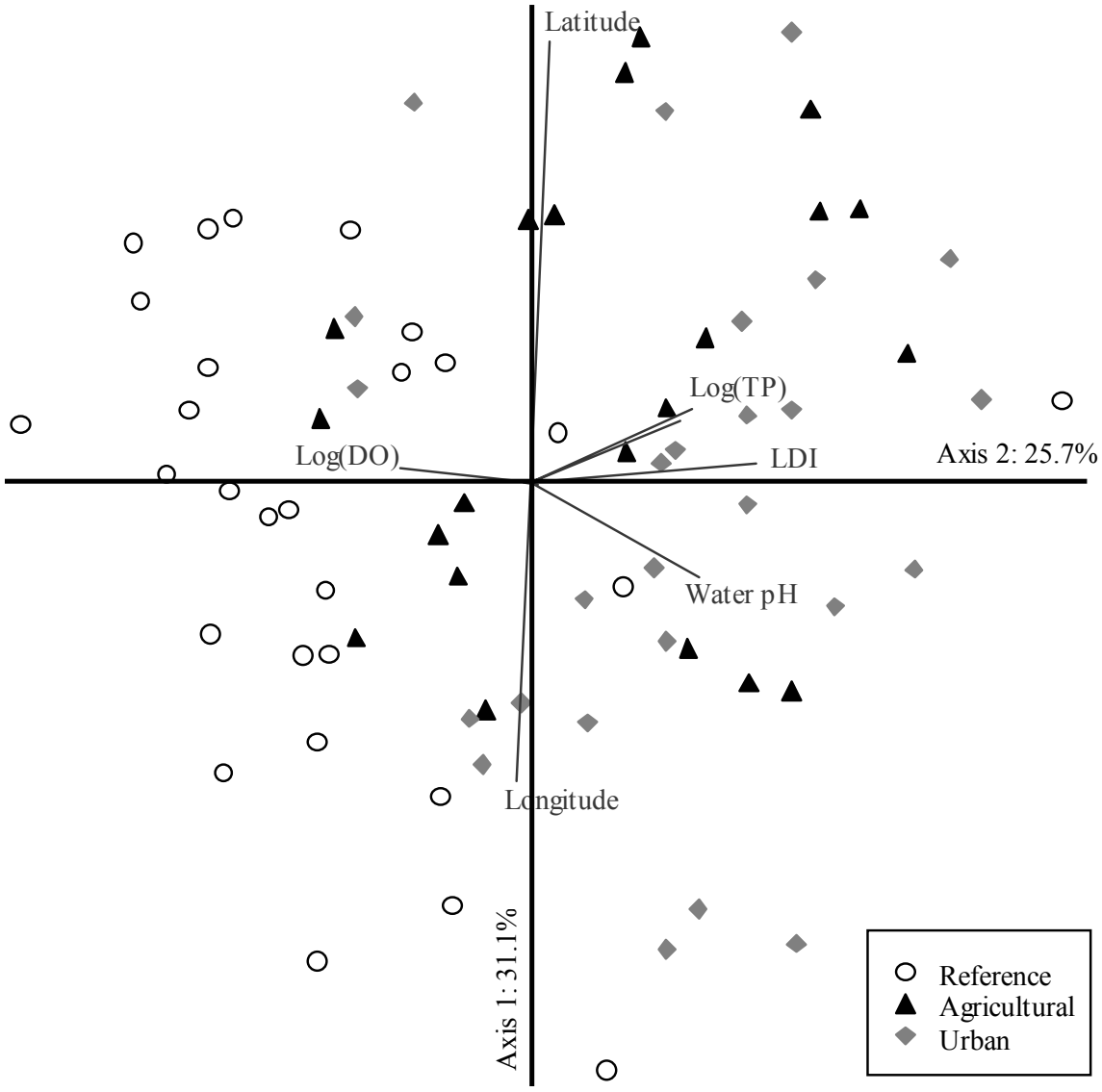


Figure 3-12. NMS ordination bi-plot of 75 sample wetlands in macrophyte species space with an overlay of environmental parameters. Latitude, longitude, LDI, Log(TP), water pH, and Log(DO), shown as radiating vectors, were significantly correlated with NMS axes. Vector length represents the strength of the correlation, and the angle represents the direction of maximum change. Axis 1 explained 31.3% variance, axis 2 explained 25.7% variance, and axis 3 (not shown) represented an additional 20.4% variance.

Table 3-17. Pearson correlations between environmental variables and NMS axes based on macrophyte community composition at 75 wetlands.

	Axis 1	Axis 2	Axis 3
Incremental r^2	31.3%	25.7%	20.4%
Cumulative r^2	31.3%	57.0%	77.4%
Latitude	0.68	0.03	0.17
Longitude	0.46	0.03	0.06
LDI	0.03	0.35	0.11
Water parameters			
Log(DO)	0.02	0.20	0.05
Log(Temperature)	0.02	0.05	0.00
Log(Color)	0.14	0.00	0.05
Log(Turbidity)	0.03	0.06	0.03
pH	0.15	0.26	0.35
Log(Ammonia-N)	0.01	0.04	0.00
Log(Nitrate/nitrite-N)	0.06	0.00	0.06
Log(TKN)	0.02	0.01	0.01
Log(TP)	0.11	0.25	0.06
Soil parameters			
Arcsin Sqrt (Moisture)	0.00	0.00	0.24
Log(TKN)	0.00	0.00	0.16
Log(TP)	0.09	0.23	0.01

ISA calculations were determined for each of the 4 ecoregions and statewide. Table 3-21 provides a list of tolerant indicator species comparing regional and statewide analyses. The same random seed number was used for each ISA. In total the ISA reported 69 statewide tolerant indicator species, and less for each ecoregions with 7, 28, 7, and 12 for the panhandle, north, central, and south ecoregions, respectively. The statewide ISA produced 69 tolerant indicator species and an additional 7 species were included on regional lists, but not the statewide list. No species occurred on the tolerant indicator lists statewide and in all 4 ecoregions. Three species occurred on the statewide tolerant indicator species list and in 3 of the ecoregions, including *Commelina diffusa* (north, central, and south), *Cynodon dactylon* (panhandle, north, central), and *Diodia*

Table 3-18. Spearman correlations between six macrophyte metrics and LDI. All correlations were significant ($p < 0.001$).

Metric	r
Tolerant indicator species	0.75
Sensitive indicator species	-0.66
Modified FQI	-0.71
Exotic species	0.65
Native perennial species	-0.63
Wetland status species	-0.55

virginiana (panhandle, north, south). Seven species occurred on the tolerant indicator species list statewide and in 2 regions, and 24 were listed both statewide and in 1 ecoregion. Thirty-five of the 69 statewide tolerant indicator species (51%) were not listed in any ecoregion. In total, the ecoregions shared more than two-thirds of their listed species with the statewide list: panhandle (100%), north (93%), central (86%), and south (67%). Two species were unique to the north, 1 to the central, and 4 to the south ecoregion tolerant indicator species lists. Figure 3-13 shows the scatter plots of the percent tolerant indicator species versus LDI. The percent tolerant indicator species increased with increasing development intensity. For the statewide tolerant indicator

Table 3-19. Comparisons among 6 macrophyte metrics for LDI groups.

Metric	Low LDI	High LDI	W^	p`
Tolerant indicator species	7.8 ± 7.8	31.2 ± 14.7	1116.5	<0.001
Sensitive indicator species	39.5 ± 16.7	9.4 ± 10.1	3665.0	<0.001
Modified FQI	4.81 ± 0.62	3.62 ± 0.80	3771.0	<0.001
Exotic species	3.0 ± 3.6	14.3 ± 10.6	1379.0	<0.001
Native perennial species	92.7 ± 4.5	79.7 ± 12.0	3453.0	<0.001
Wetland status species	72.0 ± 9.8	54.1 ± 12.5	3612.0	<0.001

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

Table 3-20. Macrophyte ISA calculations were conducted over a range of LDI values. Highlighted areas indicate the LDI value selected for statewide sensitive and tolerant indicator species.

LDI	Low LDI n* =	High LDI n* =	Sensitive r^	Sensitive p`	Tolerant r^	Tolerant p`	No. Sensitive Indicators	No. Tolerant Indicators
1	13	105	-0.52	<0.0001	0.46	<0.0001	34	4
1.25	33	85	-0.67	<0.0001	0.71	<0.0001	55	31
1.5	37	81	-0.66	<0.0001	0.69	<0.0001	58	39
1.75	39	79	-0.66	<0.0001	0.71	<0.0001	59	47
2	41	77	-0.66	<0.0001	0.70	<0.0001	61	43
2.25	48	70	-0.66	<0.0001	0.72	<0.0001	58	60
2.5	48	70	-0.66	<0.0001	0.72	<0.0001	58	60
2.75	49	69	-0.66	<0.0001	0.73	<0.0001	59	56
3	51	67	-0.66	<0.0001	0.73	<0.0001	57	56
3.25	55	63	-0.68	<0.0001	0.73	<0.0001	49	55
3.5	56	62	-0.68	<0.0001	0.74	<0.0001	53	52
3.75	56	62	-0.68	<0.0001	0.74	<0.0001	53	52
4	63	55	-0.65	<0.0001	0.75	<0.0001	35	69
4.25	65	53	-0.62	<0.0001	0.77	<0.0001	34	62
4.5	69	49	-0.63	<0.0001	0.76	<0.0001	32	47
4.75	73	45	-0.65	<0.0001	0.77	<0.0001	25	50
5	82	36	-0.65	<0.0001	0.74	<0.0001	17	41
5.25	91	27	-0.70	<0.0001	0.66	<0.0001	8	30
5.5	97	21	-0.67	<0.0001	0.58	<0.0001	4	17
5.75	99	19	-0.58	<0.0001	0.57	<0.0001	4	17
6	101	17	-0.63	<0.0001	0.63	<0.0001	3	18
6.25	106	12	-0.52	<0.0001	0.53	<0.0001	2	24
6.5	110	8	-0.52	<0.0001	0.56	<0.0001	2	27
6.75	113	5	xx	xx	0.42	<0.0001	0	19
7	114	4	xx	xx	0.14	0.1432	0	14

*n = number of sites.
^r = Spearman's r correlation coefficient of indicator species versus LDI.
`p = significance value.

species, CA2 (tolerant = 72%) had the highest percent statewide tolerant indicator species. Ninety-three percent of the wetlands in the low LDI group had less than 20% statewide tolerant indicator species. Three outliers included SA8 (statewide tolerant = 32%), CA8 (statewide tolerant = 26%), and PR7 (statewide tolerant = 25%). In the high

Table 3-21. Statewide and regional macrophyte tolerant indicator species. Tolerant indicator species calculated at an LDI break of 4.0 were significant ($p < 0.10$).

	Statewide	Panhandl	North	Central	South
No. of Tolerant Species	69	7	28	7	12
Acer rubrum	(28.5, 0.04)		(53.7, 0.01)		
Alternanthera philoxeroides	(11.3, 0.002)				
Amaranthus spinosus	(10.9, 0.01)		(21.4, 0.08)		
Ampelopsis arborea	(18.6, 0.06)				
Aster carolinianus	(9.5, 0.05)				
Axonopus fissifolius	(9.1, 0.02)				
Blechnum serrulatum				(42.4, 0.07)	
Boehmeria cylindrica	(37.9, 0.00)	(57.9, 0.01)	(43.7, 0.04)		
Carex longii	(30.6, 0.00)		(54.3, 0.00)		
Centella asiatica					(52.0, 0.02)
Colocasia esculenta	(5.5, 0.09)				
Commelina diffusa	(44.8, 0.00)		(50.0, 0.00)	(73.3, 0.00)	(44.1, 0.05)
Cuphea carthagenensis	(28.0, 0.00)		(28.6, 0.03)		(50.0, 0.00)
Cynodon dactylon	(29.4, 0.00)	(35.7, 0.05)	(35.7, 0.01)	(28.1, 0.09)	
Cyperus croceus	(9.1, 0.02)				
Cyperus lanceolatus	(7.3, 0.04)				
Cyperus polystachyos	(13.1, 0.01)				
Cyperus retrorsus	(18.2, 0.00)		(35.7, 0.01)	(20.0, 0.10)	
Cyperus virens	(11.3, 0.02)		(42.9, 0.01)		
Digitaria ciliaris	(9.1, 0.02)				
Diodia virginiana	(37.2, 0.00)	(50.0, 0.01)	(40.5, 0.03)		(50.0, 0.06)
Dioscorea bulbifera	(7.3, 0.04)				
Echinochloa colona	(5.5, 0.01)				
Eclipta prostrata	(17.9, 0.01)				
Eupatorium capillifolium	(37.9, 0.01)		(44.1, 0.06)		
Galium hispidulum	(5.5, 0.09)				
Galium tinctorium	(22.6, 0.00)		(51.8, 0.00)	(26.7, 0.05)	
Hymenachne amplexicaulis	(9.1, 0.02)				(25.0, 0.08)
Hypericum mutilum			(28.6, 0.03)		
Juncus effusus	(22.1, 0.00)		(42.9, 0.00)		
Kyllinga brevifolia	(7.3, 0.04)				
Leersia hexandra	(7.3, 0.04)			(20.0, 0.09)	
Lepidium virginicum	(5.5, 0.01)				
Ligustrum sinense	(10.2, 0.08)				
Lonicera japonica	(12.7, 0.05)				
Ludwigia peruviana	(17.7, 0.06)				
Ludwigia repens	(14.3, 0.06)		(21.4, 0.08)		
Luziola fluitans	(5.5, 0.09)				
Lygodium japonicum	(11.9, 0.03)				

Table 3-21. Continued.

	Statewide	Panhandle	North	Central	South
Melaleuca quinquenervia					(36.2, 0.06)
Melothria pendula	(20.8, 0.00)		(28.6, 0.03)		
Micranthemum umbrosum	(5.5, 0.09)				
Momordica charantia	(10.2, 0.08)				
Oxalis corniculata	(18.5, 0.00)		(28.6, 0.04)		
Parthenocissus quinquefolia	(34.6, 0.00)		(40.5, 0.04)		(52.0, 0.01)
Paspalum notatum	(24.4, 0.01)		(30.7, 0.06)		
Paspalum urvillei	(18.5, 0.00)	(35.7, 0.04)	(28.6, 0.03)		
Phyla nodiflora	(15.2, 0.08)				(32.1, 0.09)
Phyllanthus urinaria	(16.7, 0.00)		(21.4, 0.07)		
Phytolacca americana	(26.1, 0.01)	(36.7, 0.09)			
Polygonum hydropiperoides	(21.1, 0.05)	(57.1, 0.00)			
Polygonum punctatum	(28.0, 0.00)	(42.9, 0.02)	(44.7, 0.01)		
Polypremum procumbens	(9.5, 0.06)				
Proserpinaca palustris	(5.5, 0.09)				
Richardia brasiliensis	(5.5, 0.09)				
Rubus argutus	(25.1, 0.07)		(47.1, 0.03)		
Rubus trivialis	(17.7, 0.06)				
Sabal palmetto					(66.7, 0.00)
Sacciolepis indica	(7.3, 0.05)				
Sambucus canadensis	(24.6, 0.00)		(30.7, 0.08)		
Sapium sebiferum	(19.4, 0.02)				
Saururus cernuus			(40.5, 0.05)		
Senna obtusifolia	(9.1, 0.02)				
Sesbania vesicaria	(5.5, 0.01)				
Setaria parviflora	(7.3, 0.04)				
Sida rhombifolia	(20.8, 0.00)		(35.7, 0.01)		
Smilax pumila	(7.3, 0.04)		(21.4, 0.08)		
Solanum carolinense	(9.1, 0.03)				
Solidago stricta	(5.5, 0.09)				
Sporobolus indicus	(7.3, 0.05)				
Stenotaphrum secundatum	(9.5, 0.06)			(26.7, 0.04)	
Toxicodendron radicans	(38.0, 0.03)				
Trifolium repens	(7.3, 0.04)		(21.4, 0.07)		
Urena lobata					(52.0, 0.01)
Vitis rotundifolia	(36.7, 0.01)				(48.5, 0.05)
Wedelia trilobata	(5.5, 0.09)				(25.0, 0.07)

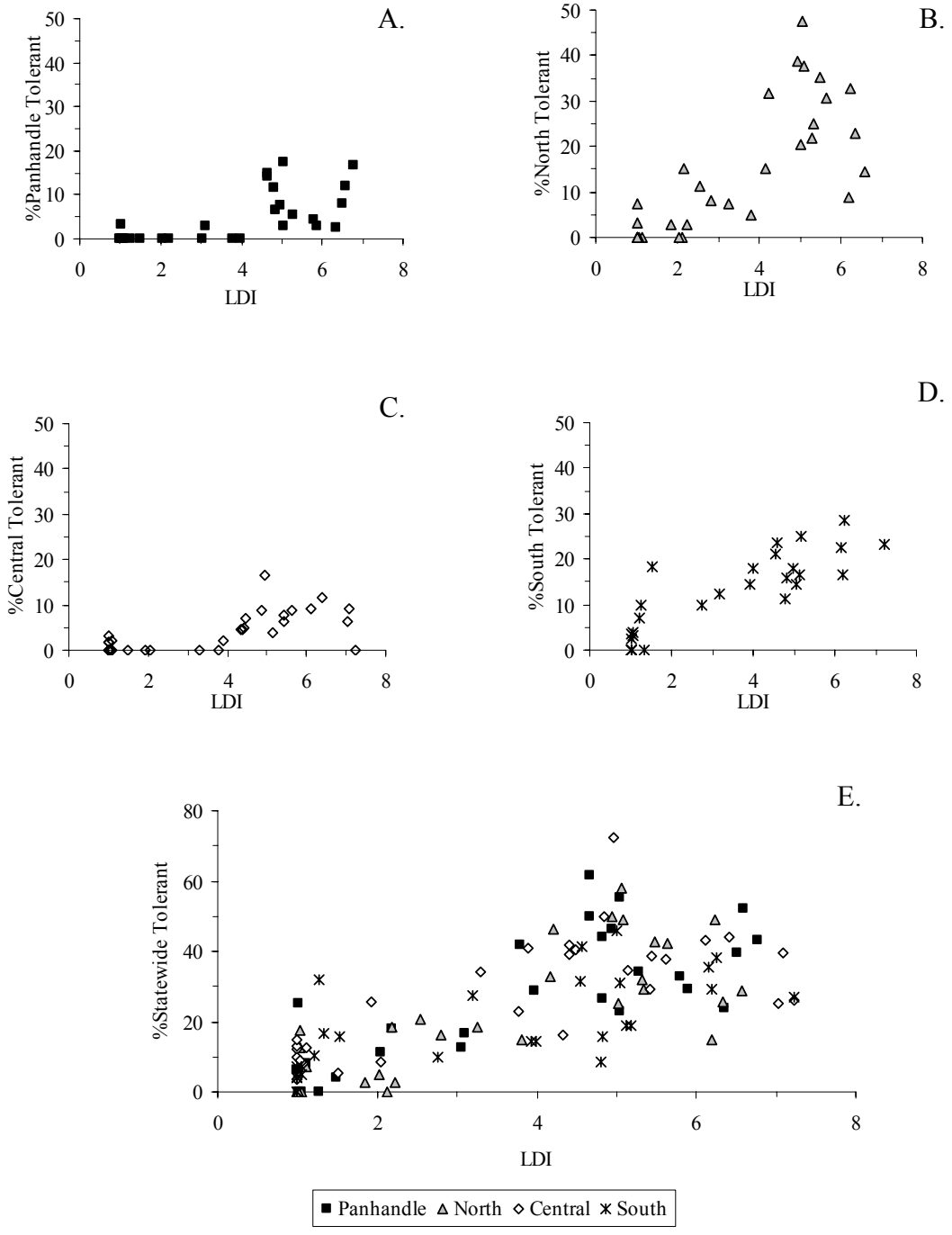


Figure 3-13. Tolerant macrophyte indicator species increased with increasing development intensity (LDI). A) Panhandle tolerant indicator species at panhandle study wetlands. B) North tolerant indicator species at north study wetlands. C) Central tolerant indicator species at central study wetlands. D) South tolerant indicator species at south study wetlands. E) Statewide tolerant indicator species at all study wetlands.

LDI group, 73% of the wetlands had over 20% statewide tolerant indicator species. In the regional ISA calculations, the north ecoregion had the largest percent tolerant indicator species, with NA1 (north tolerant = 47%).

Table 3-22 provides a list of statewide and regional sensitive indicator species. The statewide sensitive indicator species list included 61 species of which 16 were not listed in any of the ecoregions. Two species occurred on all statewide and regional lists, including *Eriocaulon decangulare* and *Panicum erectifolium*. Similarly, 6 species occurred on the statewide list and 3 regional lists including *Andropogon virginicus*, *Aristida purpurascens*, *Ilex glabra*, and *Polygala cymosa* (statewide, panhandle, north, and central); and *Fuirena scirpoidea* and *Pinus elliottii* (statewide, panhandle, central, and south). All 4 ecoregions shared over three-quarters of their species with the statewide list (panhandle = 79%, north = 84%, central = 92%, and south = 85% shared). Six species were unique to the panhandle sensitive indicator species analysis, 3 to the north, 2 to the central, and 2 to the south ecoregion. Figure 3-14 shows that the percent sensitive indicator species, statewide and regionally, decreased with increasing development intensity. Statewide, 85% of wetlands in the low LDI group had over 20% statewide sensitive indicator species; whereas, 86% of wetlands in the high LDI group had less than 20% statewide indicator species.

All of the indicator species metrics were significantly correlated with landscape development intensity. Table 3-23 shows Spearman correlations calculated with both statewide and regional indicator species lists for each ecoregion. There was little difference between the strength of regional and statewide indicator species correlations. Regional indicator species metrics had a stronger correlation value, though all metrics

Table 3-22. Statewide and regional macrophyte sensitive indicator species. Sensitive indicator species calculated at an LDI break of 2.0 were significant ($p<0.10$).

	Statewide	Panhandle	North	Central	South
No. of Sensitive Species	61	28	19	24	13
Amphicarpum muhlenbergianum	(19.3, 0.02)		(52.4, 0.02)		
Andropogon virginicus	(46.9, 0.00)	(62.5, 0.01)	(66.7, 0.01)	(66.5, 0.00)	
Aristida beyrichiana	(8.6, 0.05)	(45.5, 0.01)			
Aristida patula			(22.2, 0.09)		
Aristida purpurascens	(32.9, 0.00)	(50.0, 0.01)	(22.2, 0.08)	(53.5, 0.00)	
Carex verrucosa				(25.0, 0.05)	
Cladium jamaicense	(22.3, 0.01)				(60.6, 0.01)
Coelorachis rugosa	(7.3, 0.05)				
Cyperus haspan	(11.0, 0.02)			(33.3, 0.01)	
Drosera brevifolia	(7.3, 0.06)				
Erianthus giganteus	(17.8, 0.01)				(33.3, 0.02)
Eriocaulon compressum	(17.1, 0.00)	(25.0, 0.07)	(33.3, 0.02)		
Eriocaulon decangulare	(37.8, 0.00)	(37.5, 0.02)	(22.2, 0.09)	(53.5, 0.01)	(33.3, 0.02)
Eupatorium leptophyllum	(14.6, 0.00)			(25.0, 0.05)	
Eupatorium mohrii	(8.6, 0.05)				
Fuirena scirpoidea	(26.9, 0.00)	(25.0, 0.09)		(25.0, 0.05)	(36.2, 0.06)
Gaylussacia frondosa		(25.0, 0.06)			
Gratiola ramosa	(18.3, 0.00)			(41.7, 0.00)	
Hypericum chapmanii	(8.6, 0.04)	(45.5, 0.02)			
Hypericum fasciculatum	(38.2, 0.00)			(53.5, 0.00)	(44.4, 0.01)
Hypericum myrtifolium	(17.8, 0.01)		(47.7, 0.01)		
Hyptis alata	(10.8, 0.08)				
Ilex glabra	(46.2, 0.00)	(53.9, 0.01)	(78.6, 0.00)	(45.9, 0.02)	
Ilex myrtifolia	(17.1, 0.06)	(71.2, 0.01)			
Ipomoea sagittata	(11.0, 0.02)				
Lachnanthes caroliniana	(39.6, 0.00)	(71.2, 0.00)		(40.2, 0.08)	
Lachnocaulon anceps		(25.0, 0.08)			
Lobelia floridana		(25.0, 0.08)			
Lophiola aurea	(12.2, 0.01)	(62.5, 0.00)			
Ludwigia linifolia	(7.3, 0.05)				
Lycopodiella alopecuroides	(9.8, 0.02)	(25.0, 0.06)	(22.2, 0.07)		
Lyonia lucida	(24.4, 0.07)				
Nymphaea odorata	(6.2, 0.10)		(22.2, 0.08)		
Nymphoides aquatica	(12.2, 0.00)				
Panicum ensifolium	(13.1, 0.05)				
Panicum erectifolium	(41.5, 0.00)	(50.0, 0.01)	(33.3, 0.01)	(45.2, 0.01)	(36.2, 0.06)
Panicum hemitomon	(40.4, 0.01)			(79.2, 0.00)	
Panicum rigidulum	(17.1, 0.05)	(25.0, 0.08)			
Panicum tenerum	(14.6, 0.00)				(33.3, 0.02)

Table 3-22. Continued.

	Statewide	Panhandle	North	Central	South
Pinus elliottii	(33.9, 0.02)	(56.2, 0.04)		(37.0, 0.02)	(25.0, 0.07)
Pinus palustris	(8.6, 0.05)				
Pluchea foetida	(10.1, 0.06)			(25.0, 0.05)	
Pluchea rosea	(15.5, 0.07)			(33.3, 0.08)	
Polygala cymosa	(28.0, 0.00)	(37.5, 0.02)	(29.3, 0.06)	(33.3, 0.01)	
Polygala lutea	(7.3, 0.04)				
Proserpinaca pectinata	(12.4, 0.02)				
Rhexia alifanus	(13.4, 0.01)	(70.3, 0.00)			
Rhexia lutea	(13.4, 0.02)	(45.5, 0.02)	(22.2, 0.07)		
Rhexia mariana	(23.1, 0.00)		(47.7, 0.01)	(45.2, 0.01)	
Rhexia petiolata		(25.0, 0.07)			
Rhus copallinum		(33.1, 0.06)			
Rhynchospora corniculata					(25.0, 0.07)
Rhynchospora filifolia		(37.5, 0.02)			
Rhynchospora inundata	(17.2, 0.00)		(22.2, 0.08)		
Rhynchospora microcarpa				(28.8, 0.06)	
Rhynchospora wrightiana			(22.2, 0.08)		
Sabatia bartramii	(7.3, 0.05)				
Sagittaria graminea	(13.9, 0.08)			(41.7, 0.00)	
Sagittaria lancifolia	(17.2, 0.01)				(36.2, 0.06)
Salix caroliniana					(44.4, 0.01)
Sarracenia minor			(22.2, 0.07)		
Scleria baldwinii	(7.3, 0.04)				
Scleria georgiana	(7.3, 0.05)				
Scleria triglomerata	(7.3, 0.06)	(25.0, 0.08)			
Serenoa repens	(22.3, 0.05)	(37.5, 0.02)	(57.6, 0.01)		
Spartina bakeri	(7.3, 0.05)			(25.0, 0.05)	
Stillingia aquatica	(13.4, 0.01)				(36.2, 0.05)
Syngonanthus flavidulus	(12.2, 0.01)		(22.2, 0.08)		
Utricularia purpurea	(8.6, 0.06)				(25.0, 0.07)
Vaccinium corymbosum	(20.9, 0.02)	(33.1, 0.06)		(33.3, 0.02)	
Xyris ambigua	(11.0, 0.02)				
Xyris caroliniana	(7.3, 0.05)	(25.0, 0.08)			
Xyris elliottii	(18.3, 0.00)			(50.0, 0.00)	
Xyris jupicai	(8.6, 0.05)			(25.0, 0.04)	

were significantly correlated with LDI ($p < 0.01$). Two exceptions included the panhandle (statewide tolerant $r = 0.73$, panhandle tolerant $r = 0.72$) and central (statewide tolerant $r = 0.74$, central tolerant $r = 0.68$) tolerant indicator correlations.

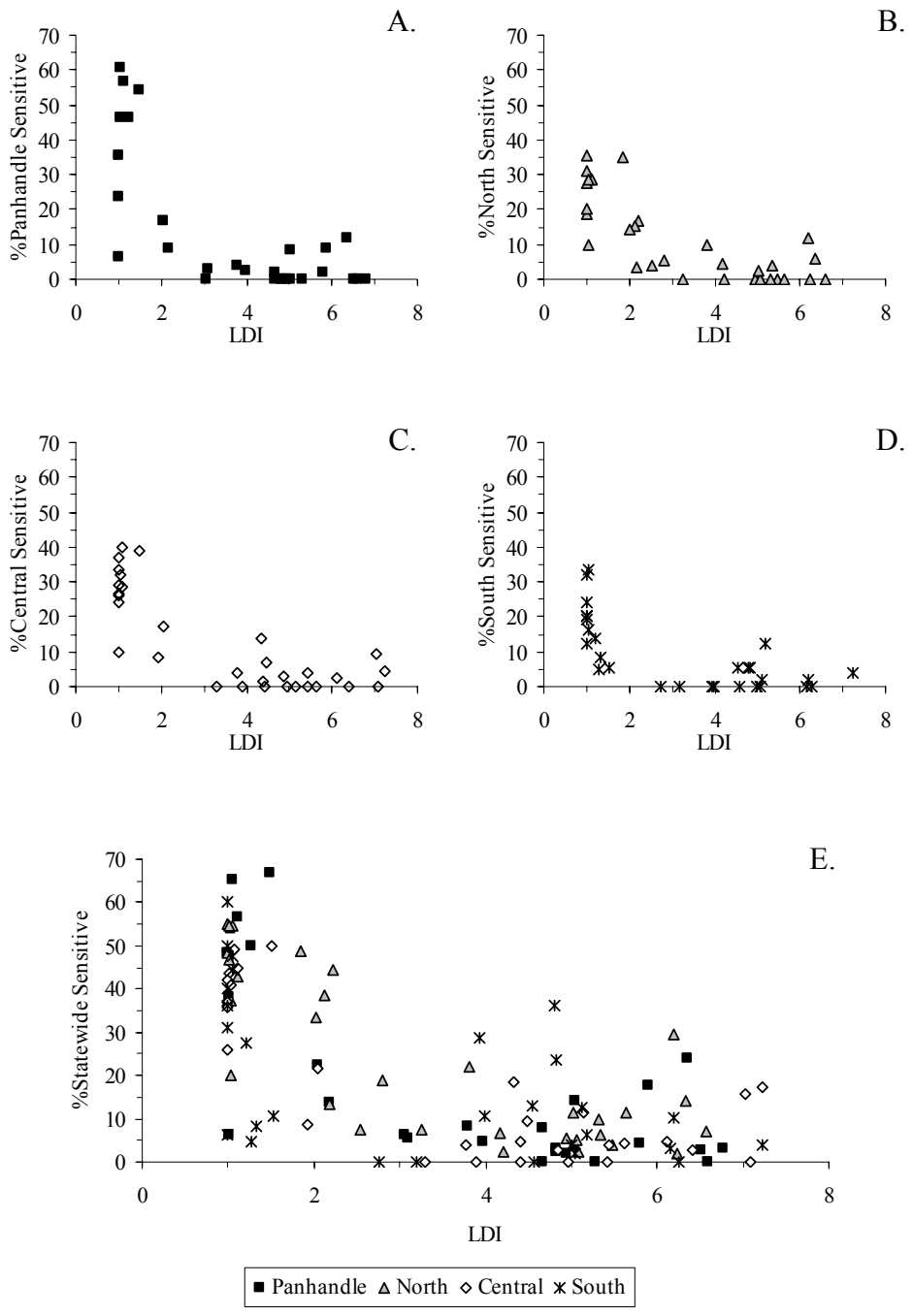


Figure 3-14. Macrophyte sensitive indicator species decreased with increasing development intensity (LDI). A) Panhandle sensitive indicator species at panhandle study wetlands. B) North sensitive indicator species at north study wetlands. C) Central sensitive indicator species at central study wetlands. D) South sensitive indicator species at south study wetlands. E) Statewide sensitive indicator species at all study wetlands.

Table 3-23. Statewide and regional macrophyte indicator species were significantly correlated with LDI ($p < 0.01$).

	n=	Statewide ISA Spearman's r	Regional ISA Spearman's r
Statewide			
Tolerant indicator species	118	0.75	
Sensitive indicator species	118	-0.66	
Panhandle			
Tolerant indicator species	28	0.73	0.72
Sensitive indicator species	28	-0.66	-0.68
North			
Tolerant indicator species	31	0.79	0.80
Sensitive indicator species	31	-0.76	-0.78
Central			
Tolerant indicator species	31	0.74	0.68
Sensitive indicator species	31	-0.67	-0.72
South			
Tolerant indicator species	28	0.78	0.86
Sensitive indicator species	28	-0.60	-0.71

It is important to note that shrub and tree species were included in the ISA for both tolerant and sensitive metrics. Metrics developed based on the macrophyte community composition included woody species rooted with the sampling quadrats, as structure was thought to play an important role in the biological condition of pondcypress domes. Excluding the tree and shrub layers would seemingly underscore their importance. However, trees comprised only a small percentage of the tolerant and sensitive indicator species lists (Tables 3-21 and 3-22). Three percent of the statewide tolerant indicator species were trees, 9% were shrubs, 14% vines, and 74% herbaceous (including herbs, sedges, grasses, etc.). The 2 statewide tolerant indicator tree species included the hardwood *Acer rubrum* and exotic *Sapium sebiferum* (Table 3-21). The 6 statewide shrub tolerant indicator species were of the genera *Aster* (a climbing species), an exotic *Ligustrum*, an exotic *Ludwigia*, *Rubus*, and *Sambucus*.

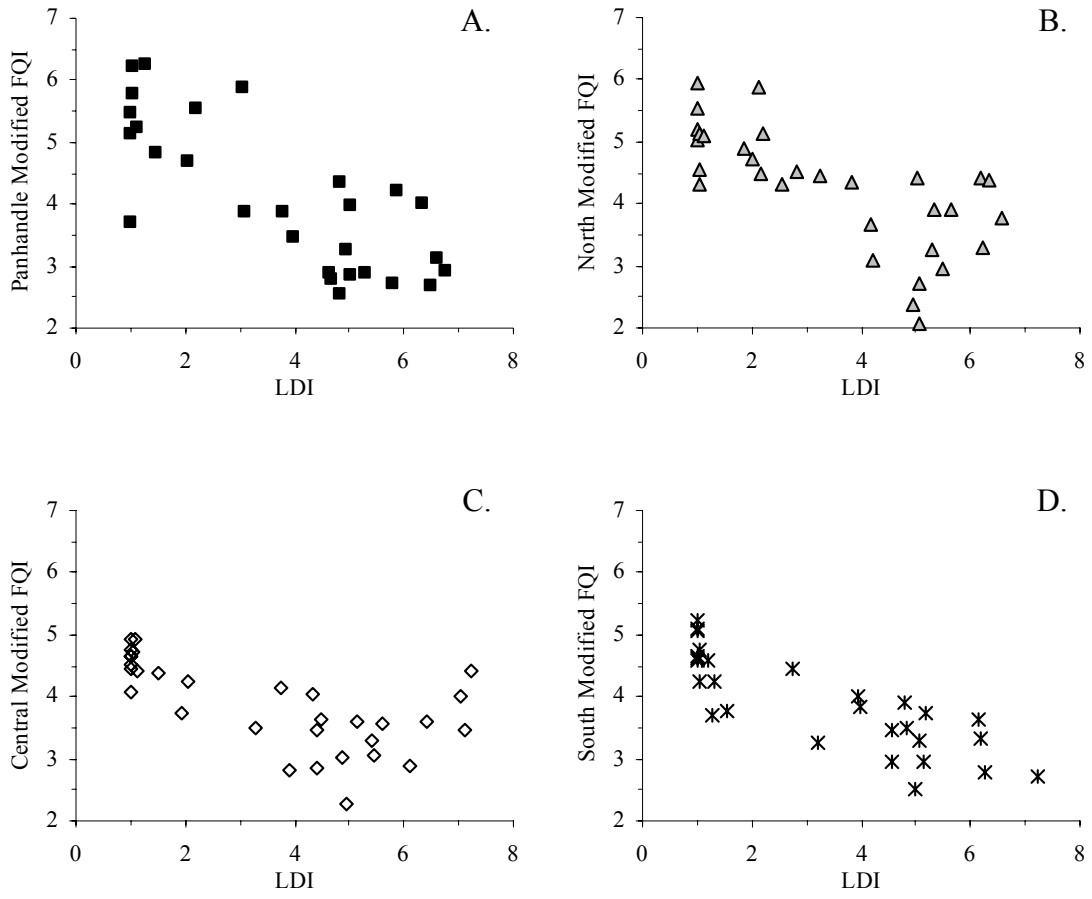


Figure 3-15. Modified FQI scores decreased with increasing development intensity (LDI). A) Modified FQI scores at panhandle study wetlands. B) Modified FQI scores at north study wetlands. C) Modified FQI scores at central study wetlands. D) Modified FQI scores at south study wetlands.

Modified Floristic Quality Index metric

Figure 3-15 shows modified FQI scores decreased with increasing landscape development intensity. Wetlands in the panhandle (maximum modified FQI = 6.25) and north (maximum modified FQI = 5.95) ecoregions had higher modified FQI scores versus wetlands in the central (maximum modified FQI = 4.93) and south (maximum modified FQI = 5.24) ecoregions. Statewide the modified FQI was significantly correlated with LDI ($|r| = 0.71$, $p < 0.001$; Table 3-18); and there was a significant difference between the mean modified FQI scores between low and high LDI groups ($W = 3771.0$, $p = <0.001$;

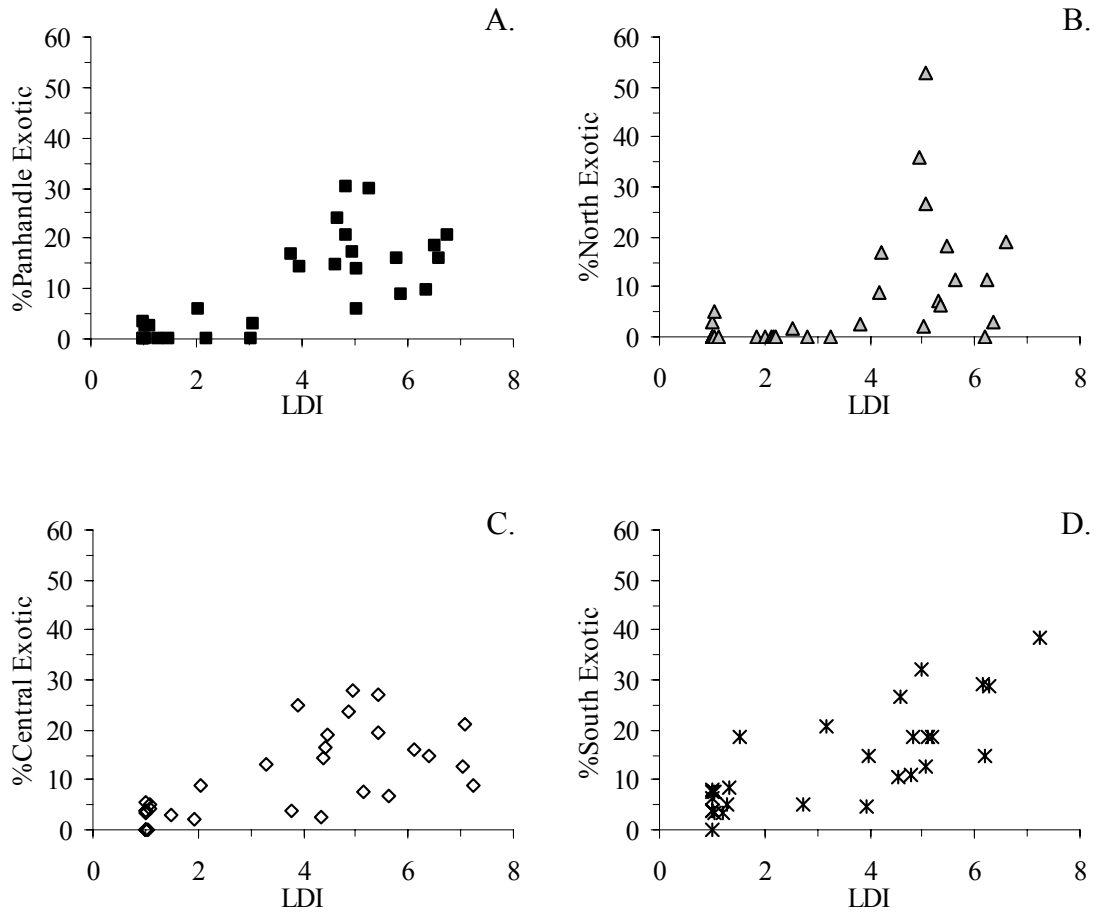


Figure 3-16. Exotic species increased with increasing development intensity (LDI). A) Exotic species at panhandle study wetlands. B) Exotic species at north study wetlands. C) Exotic species at central study wetlands. D) Exotic species at south study wetlands.

Table 3-19). In the low LDI groups, 87.5%, 100%, 92%, and 83% of the wetlands had a modified FQI score greater than 4.00 in the panhandle, north, central, and south ecoregions, respectively. Wetlands with a modified FQI score less than 4.00 accounted for 70% 82%, 74%, and 87.5% of the wetlands in the high LDI group in the panhandle, north, central, and south ecoregions, respectively.

Exotic species metric

Statewide, the percent exotic species was significantly correlated with development intensity in the surrounding landscape ($r = 0.65$, $p < 0.001$; Table 3-18). Figure 3-16 shows that the percent of exotic species increased with increasing LDI in each ecoregion. The north ecoregion hosted the wetland with the greatest percent exotic species, NA1 (52.6% exotic species). NA1 was surrounded by research facility growing experimental pasture species, 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 urban land use (residential and commercial). One apparent outlier in south ecoregion low LDI group was SU4 (exotics = 18.4%). All remaining wetlands in the low LDI group ($n = 40$) had less than 10% exotic species.

Statewide, the percent exotic species was significantly different between low and high LDI groups ($W = 1379.0$, $p < 0.001$; Table 3-19). Table 3-24 lists the 113 exotic species encountered throughout Florida and identifies the ecoregion(s) in which each species was found. Only 6 exotic species were found in all 4 ecoregions including *Commelina diffusa*, *Cuphea carthagenensis*, *Cynodon dactylon*, *Kyllinga brevifolia*, *Ludwigia peruviana*, and *Paspalum notatum*. Fourteen exotic species occurred in 3 of the 4 ecoregions.

Native perennial species metric

Of the 605 macrophyte species identified, 427 (71%) were classified as native perennials. Figure 3-17 shows that native perennial species decreased with increasing development intensity. Statewide there was a significant difference between the percent native perennial species between low and high LDI groups ($W = 3453.0$, $p < 0.001$; Table 3-19). The native perennial species metric was significantly correlated with LDI

Table 3-24. Exotic macrophyte species identified at 118 study wetlands.

Exotic Species	P	N	C	S	Exotic Species	P	N	C	S
Albizia julibrissin	★		★		*Kyllinga brevifolia*	★	★	★	★
Aloe vera		★			*Lantana camara*				★
Alternanthera philoxeroides	★	★	★		*Ligustrum japonicum*	★			
Alternanthera sessilis			★		*Ligustrum lucidum*		★		
Amaranthus blitum		★	★		*Ligustrum sinense*	★	★		
Amaranthus spinosus	★	★			*Lindernia crustacea*				★
Ardisia crenata	★				*Lolium perenne*		★		
Begonia cucullata		★	★		*Lonicera japonica*	★	★	★	
Bischofia javanica				★	*Ludwigia peruviana*	★	★	★	★
Blechum pyramidatum				★	*Lygodium japonicum*	★	★		★
Bromus catharticus		★			*Lygodium microphyllum*		★		★
Callisia repens				★	*Macroptilium lathyroides*			★	
Chenopodium album			★		*Melaleuca quinquenervia*				★
Chenopodium ambrosioides		★			*Melia azedarach*			★	
Cinnamomum camphora	★	★	★		*Melochia corchorifolia*	★		★	★
Citrus Xaurantium				★	*Momordica charantia*			★	★
Colocasia esculenta	★	★			*Morrenia odorata*			★	
Commelina diffusa	★	★	★	★	*Morus alba*		★		
Conyza bonariensis	★				*Murdannia nudiflora*			★	
Cuphea carthagenensis	★	★	★	★	*Nandina domestica*	★			
Cyclospermum leptophyllum				★	*Nephrolepis cordifolia*			★	★
Cynodon dactylon	★	★	★	★	*Oeceoclades maculata*				★
Cyperus iria	★				*Oxalis debilis*		★		
Cyperus lanceolatus	★	★	★		*Paederia foetida*		★	★	
Desmodium incanum			★		*Panicum maximum*			★	
Digitaria bicornis	★				*Panicum repens*	★	★		★
Dioscorea bulbifera		★	★		*Paspalidium geminatum*				★
Duchesnea indica	★				*Paspalum acuminatum*			★	
Echinochloa colona	★	★		★	*Paspalum notatum*	★	★	★	★
Echinochloa crusgalli		★			*Paspalum urvillei*	★	★	★	
Eichhornia crassipes			★		*Phalaris angusta*		★		
Eleusine indica	★	★			*Phyllanthus tenellus*				★
Eragrostis atrovirens				★	*Phyllanthus urinaria*	★	★	★	
Eugenia uniflora				★	*Plantago lanceolata*	★			
Hedychium coronarium			★		*Pouzolzia zeylanica*			★	
Hedyotis corymbosa			★	★	*Pueraria montana*	★			
Hemarthria altissima				★	*Rhodomyrtus tomentosa*				★
Hymenachne amplexicaulis			★	★	*Rhoeo discolor*				★
Imperata cylindrica			★		*Richardia brasiliensis*		★	★	
Ipomoea indica	★				*Richardia scabra*	★			
Ipomoea quamoclit	★				*Rumex crispus*	★	★		
Kummerowia striata	★				*Rumex obtusifolius*		★	★	

Table 3-24. Continued.

Rumex pulcher		★			*Tradescantia zebrina*				★
Sacciolepis indica			★	★	*Trifolium repens*	★	★		
Salvinia minima		★	★		*Urena lobata*			★	★
Sapium sebiferum	★	★	★		*Urtica dioica*	★			
Schinus terebinthifolius			★	★	*Verbena bonariensis*	★			
Senna obtusifolia	★		★	★	*Verbena brasiliensis*	★		★	
Senna pendula				★	*Viburnum odoratissimum*		★		
Solanum tampicense				★	*Vicia sativa*	★			
Solanum viarum			★	★	*Wedelia trilobata*				★
Sonchus asper			★		*Xanthosoma sagittifolium*	★			
Sorghum bicolor		★			*Xyris jupicai*	★		★	★
Spermacoce verticillata				★	*Youngia japonica*		★		
Sporobolus indicus	★	★		★	*Yucca aloifolia*			★	
Thelypteris dentata		★			*Zea mays*	★			
Tradescantia fluminensis	★								

(Spearman $|r| = 0.63$, $p < 0.001$; Table 3-18). Statewide 78% of the wetlands in the low LDI group had greater than 90% native perennial species; whereas, 75% of the wetlands in the high LDI group had less than 90% native perennial species.

Wetland status metric

Fifty-six percent of the macrophyte species identified were included in the wetland status metric, including 160 species designated as obligate and 180 species designated as facultative wet species. There were an additional 137 facultative, 62 facultative upland, and 49 upland species identified in the study wetlands. Seventeen species (of the 605 macrophyte species identified in this study) were not categorized by wetland status. Figure 3-18 shows wetland status species decreased with increasing development intensity in each ecoregion. The percent wetland status species was significantly different between LDI groups ($W = 3612.0$, $p < 0.001$; Table 3-19); and significantly correlated statewide with the LDI index (Spearman $|r| = 0.55$, $p < 0.001$; Table 3-18). Statewide 90% of the wetlands in the low LDI group had greater than 60% wetland status

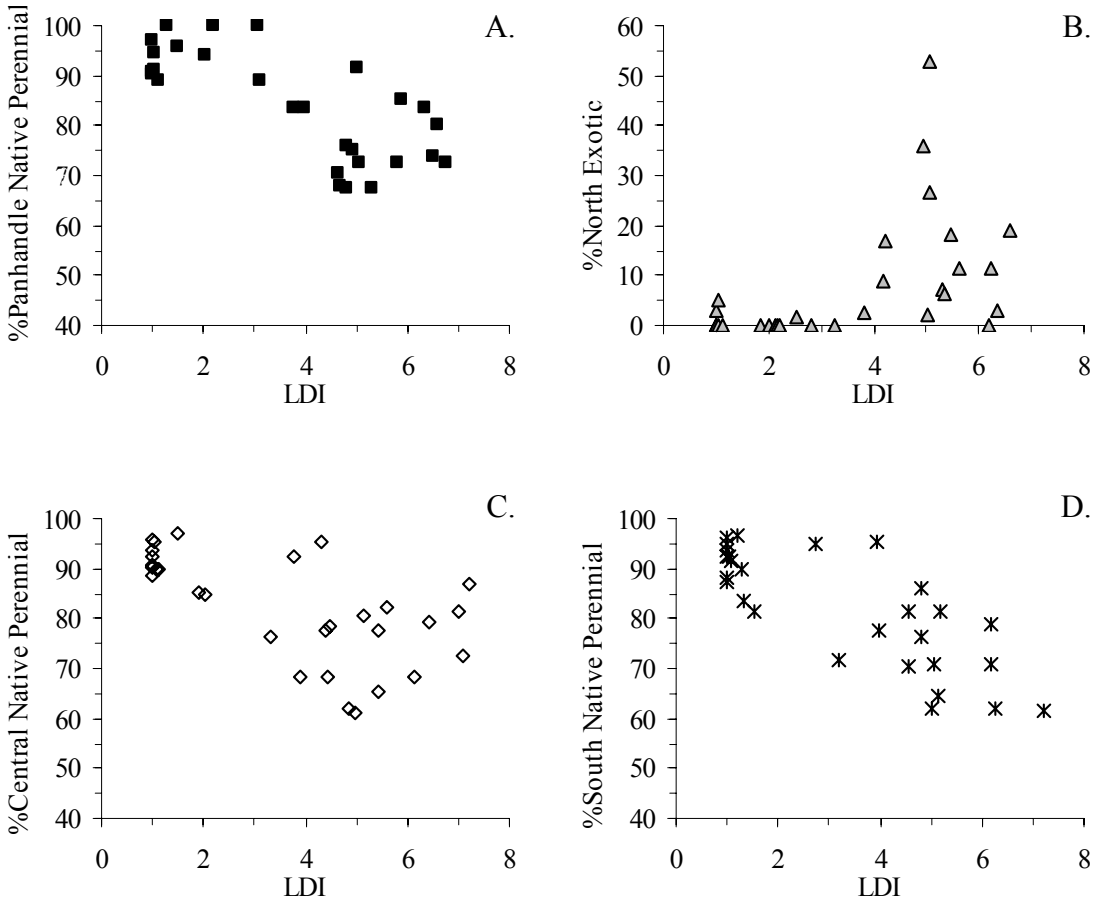


Figure 3-17. Native perennial species decreased with increasing development intensity (LDI). A) Panhandle study wetlands. B) North study wetlands. C) Central study wetlands. D) South study wetlands.

species, whereas 75% of the wetlands in the high LDI group had less than 60% wetland status species.

Macrophyte Wetland Condition Index

The 6 metrics described above were included in the macrophyte WCI. Figure 3-19 shows that both statewide and regional macrophyte WCI scores decrease with increasing development intensity. Table 3-25 compares the overall macrophyte WCI calculated statewide and regionally for the low LDI group (LDI < 2.0). A comparable statewide macrophyte WCI should equally score reference wetlands in each region; however, the

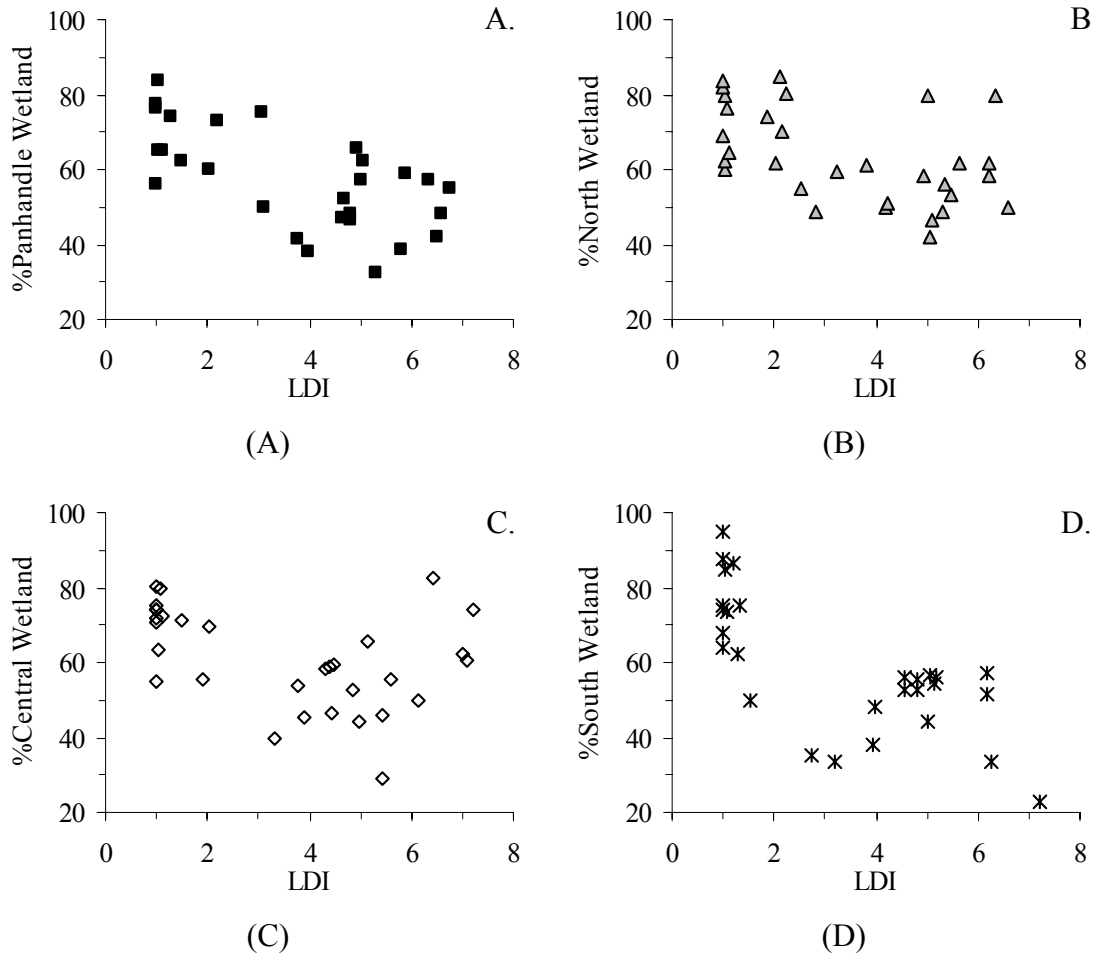


Figure 3-18. The percent wetland status species decreased with increasing development intensity (LDI). This trend was consistent for the (A) panhandle, (B) north, (C) central, and (D) south ecoregions.

south ecoregion had significantly different overall macrophyte WCI scores for the low LDI group compared to the panhandle, north, and central ecoregions. When calculated statewide, 5 of the 6 metrics had 1 or more ecoregion with significantly different metric scores. The north and central ecoregions had significantly different scores for the statewide tolerant indicator species, whereas the north and south ecoregions had significantly different scores for the statewide sensitive indicator species. The panhandle and north ecoregions were not significantly different from each other, but were significantly different from the central and south ecoregions for modified FQI scores;

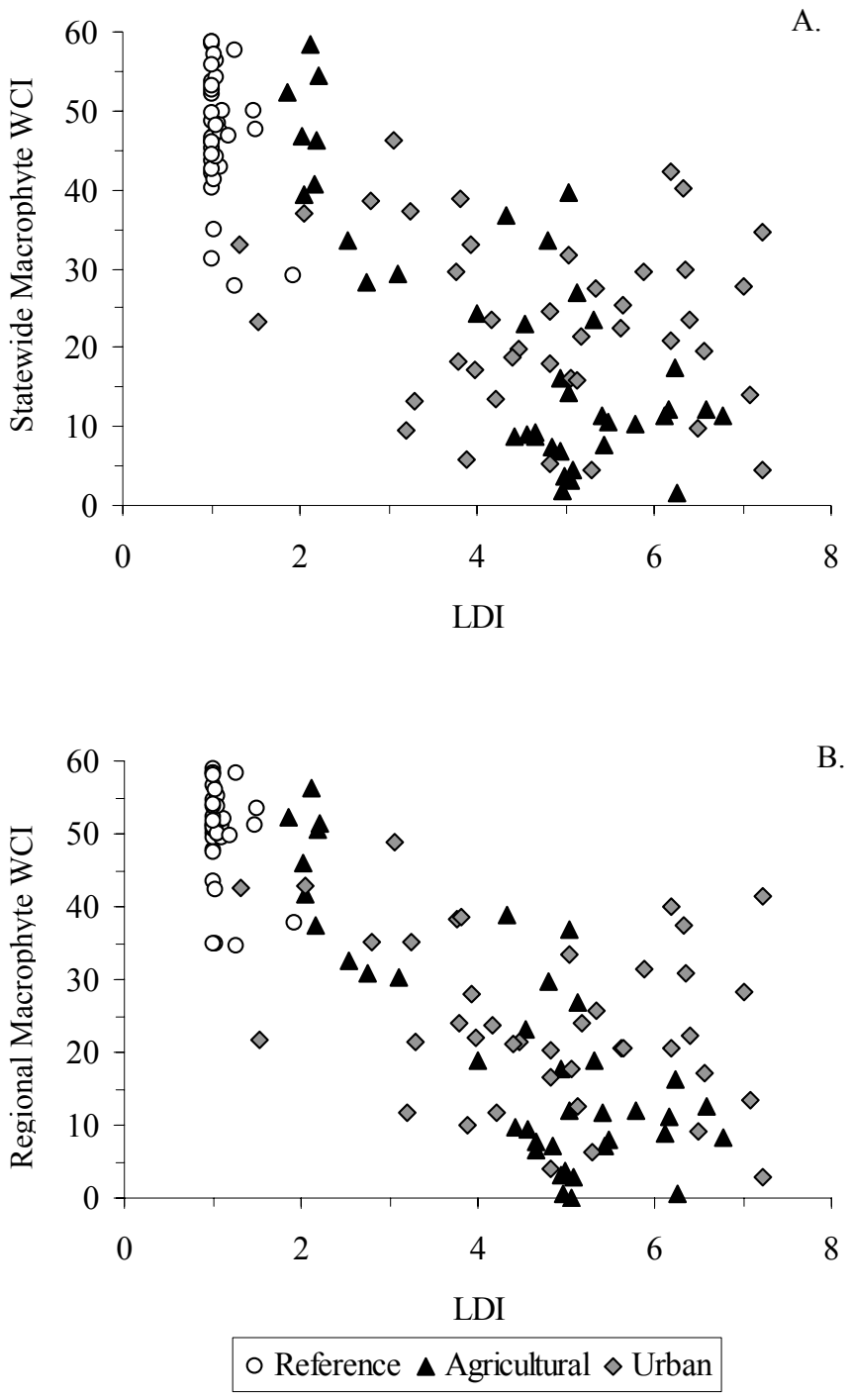


Figure 3-19. Macrophyte WCI scores decreased with increasing development intensity (LDI). A) Statewide B) Regional.

Table 3-25. Macrophyte WCI and metrics scored statewide and regionally for study wetlands in the low LDI group.

	Panhandle	North	Central	South
Statewide				
Macrophyte WCI	50.3 ± 8.6[a]	51.6 ± 8.4[a]	45.9 ± 3.4[ab]	42.2 ± 9.4[b]
State tolerant	8.1 ± 2.5[ab]	8.4 ± 2.2[a]	6.6 ± 1.2[b]	6.8 ± 2.3[ab]
State sensitive	8.8 ± 2.6[ab]	9.0 ± 1.3[a]	8.6 ± 0.7[ab]	6.7 ± 2.9[b]
Modified FQI	8.5 ± 2.2[a]	8.1 ± 1.4[a]	6.6 ± 0.9[b]	6.4 ± 1.7[b]
Exotic species	9.3 ± 1.0[a]	9.4 ± 1.1[a]	8.4 ± 1.3[a]	6.6 ± 1.9[b]
Native perennial species	8.6 ± 0.9[ab]	9.1 ± 1.0[a]	8.3 ± 0.6[b]	7.9 ± 1.1[b]
Wetland status species	7.0 ± 2.0[a]	7.5 ± 1.9[a]	7.4 ± 1.6[a]	7.7 ± 2.3[a]
Regional				
Macrophyte WCI	50.9 ± 7.2[a]	51.3 ± 7.9[a]	51.4 ± 4.0[a]	46.8 ± 9.8[a]
Regional tolerant	9.6 ± 1.0[a]	9.2 ± 1.3[ab]	9.0 ± 1.5[ab]	7.7 ± 2.7[b]
Regional sensitive	8.2 ± 2.6[a]	8.5 ± 1.7[a]	8.6 ± 1.6[a]	6.9 ± 2.4[a]
Modified FQI	7.6 ± 2.3[a]	7.9 ± 1.4[a]	8.7 ± 1.1[a]	7.7 ± 2.0[a]
Exotic species	9.3 ± 1.0a	9.5 ± 1.1[a]	8.3 ± 1.4[a]	8.3 ± 1.9[a]
Native perennial species	8.3 ± 1.0[a]	9.2 ± 0.9[a]	9.1 ± 0.7[a]	8.7 ± 1.2[a]
Wetland status species	8.0 ± 2.1[a]	7.0 ± 2.5[a]	7.8 ± 1.9[a]	7.6 ± 2.1[a]

Values represent the mean score ± the standard deviation
Ecoregions with similar letters were not significantly different ($p<0.05$)

suggesting that the panhandle and north ecoregions hosted more species with a narrower set of ecological conditions found in reference wetlands. The south ecoregion had significantly different statewide exotic species scores than the other 3 ecoregions. The scores for the statewide native perennial species were significantly different for the north ecoregion. When the macrophyte WCI was scored regionally, there was not a significant difference between mean scores for the low LDI group (Table 3-25). The only regionally scored metric with significantly different means scores for the low LDI group was the regional tolerant indicator species for the south ecoregion which was only significantly different from the panhandle ecoregion.

Table 3-26 shows similar results for the high LDI group. Within the macrophyte WCI calculated statewide, wetlands in the north ecoregion had significantly different macrophyte WCI scores, suggesting the north ecoregion high LDI wetlands had higher

Table 3-26. Macrophyte WCI and metrics scored statewide and regionally for study wetlands in the high LDI group.

	Panhandle	North	Central	South
Statewide				
Macrophyte WCI	20.4 ± 13.1[a]	29.4 ± 15.9[b]	18.9 ± 11.0[a]	17.6 ± 10.1[a]
State tolerant	2.1 ± 1.9[ab]	3.3 ± 2.8[a]	2.0 ± 1.6[b]	3.3 ± 1.9[ab]
State sensitive	2.5 ± 2.0[ab]	4.1 ± 2.4[a]	2.2 ± 2.1[ab]	2.9 ± 2.7[b]
Modified FQI	3.2 ± 3.2[a]	4.3 ± 2.8[a]	2.8 ± 1.8[b]	2.5 ± 1.8[b]
Exotic species	4.0 ± 3.0[a]	6.3 ± 3.7[a]	3.6 ± 2.4[a]	2.7 ± 2.4[b]
Native perennial species	5.2 ± 2.7[ab]	6.6 ± 3.2[a]	4.3 ± 2.6[b]	3.9 ± 2.9[b]
Wetland status species	3.3 ± 2.4[a]	4.8 ± 2.6[a]	4.0 ± 2.5[a]	2.3 ± 1.8[a]
Regional				
Macrophyte WCI	21.3 ± 14.5[ab]	27.1 ± 16.1[a]	20.6 ± 12.8[ab]	16.6 ± 9.6[b]
Regional tolerant	5.5 ± 3.7[a]	3.9 ± 3.2[ab]	4.2 ± 3.6[ab]	1.8 ± 1.4[b]
Regional sensitive	1.3 ± 1.7[a]	2.1 ± 2.4[a]	1.6 ± 2.0[a]	1.3 ± 1.8[a]
Modified FQI	2.8 ± 2.8[a]	4.4 ± 2.7[a]	3.3 ± 2.5[a]	2.9 ± 2.1[a]
Exotic species	3.9 ± 3.0[a]	6.4 ± 3.6[a]	3.2 ± 2.5[a]	3.7 ± 2.9[a]
Native perennial species	4.2 ± 3.2[a]	6.8 ± 3.2[a]	4.5 ± 2.9[a]	4.3 ± 3.2[a]
Wetland status species	3.7 ± 2.8[a]	3.5 ± 3.3[a]	3.7 ± 2.9[a]	2.6 ± 1.8[a]

Ecoregions with similar letters were not significantly different ($p<0.05$).
Values represent the mean score ± the standard deviation.

ecological integrity than wetlands of other ecoregions. Five of the 6 metrics calculated statewide had at least 1 ecoregion with significantly different mean scores in the high LDI group. Only the percent wetland status species metric did not have significantly different scores for both statewide and regional calculations among ecoregions. For the regional macrophyte WCI calculations, the north and south ecoregions had significantly different mean macrophyte WCI scores. Additionally, only the regional tolerant indicator species metric for the panhandle and south ecoregions had significantly different mean scores for all of the regionally calculated metrics.

Correlations between macrophyte WCI and 6 metrics with LDI were strong ($|r| > 0.50$, $p < 0.01$) for all of the metrics statewide and regionally (Table 3-27), except for the central ecoregion wetland status species ($|r| = 0.39$, $p = 0.03$). This metric was still significantly correlated at the more flexible $p < 0.05$ level. Three of the 4 ecoregions,

Table 3-27. Spearman correlations between the macrophyte WCI, metrics, and LDI.

	Spearman's r	p-value
Statewide		
Macrophyte WCI	-0.73	<0.0001
Tolerant indicator species	0.75	<0.0001
Sensitive indicator species	-0.66	<0.0001
Modified FQI	-0.71	<0.0001
Exotic species	0.65	<0.0001
Native perennial species	-0.63	<0.0001
Wetland status species	-0.55	<0.0001
Panhandle		
Macrophyte WCI	-0.74	<0.0001
Tolerant indicator species	0.72	<0.0001
Sensitive indicator species	-0.68	<0.0001
Modified FQI	-0.68	<0.0001
Exotic species	0.72	<0.0001
Native perennial species	-0.67	0.0001
Wetland status species	-0.60	0.0007
North		
Macrophyte WCI	-0.74	<0.0001
Tolerant indicator species	0.80	<0.0001
Sensitive indicator species	-0.78	<0.0001
Modified FQI	-0.75	<0.0001
Exotic species	0.65	<0.0001
Native perennial species	-0.65	<0.0001
Wetland status species	-0.55	0.0015
Central		
Macrophyte WCI	-0.73	<0.0001
Tolerant indicator species	0.68	<0.0001
Sensitive indicator species	-0.72	<0.0001
Modified FQI	-0.68	<0.0001
Exotic species	0.70	<0.0001
Native perennial species	-0.66	<0.0001
Wetland status species	-0.39	0.0309
South		
Macrophyte WCI	-0.88	<0.0001
Tolerant indicator species	0.86	<0.0001
Sensitive indicator species	-0.71	<0.0001
Modified FQI	-0.86	<0.0001
Exotic species	0.80	<0.0001
Native perennial species	-0.80	<0.0001
Wetland status species	-0.69	<0.0001

including the panhandle, north, and south ecoregions had stronger WCI correlations with LDI than both the statewide and central ecoregion WCIs.

Cluster Analysis

Cluster analysis determined 5 categories of wetlands based on macrophyte community composition. Clusters were roughly defined 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. Figure 3-20 shows that based on regional macrophyte WCI 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 each

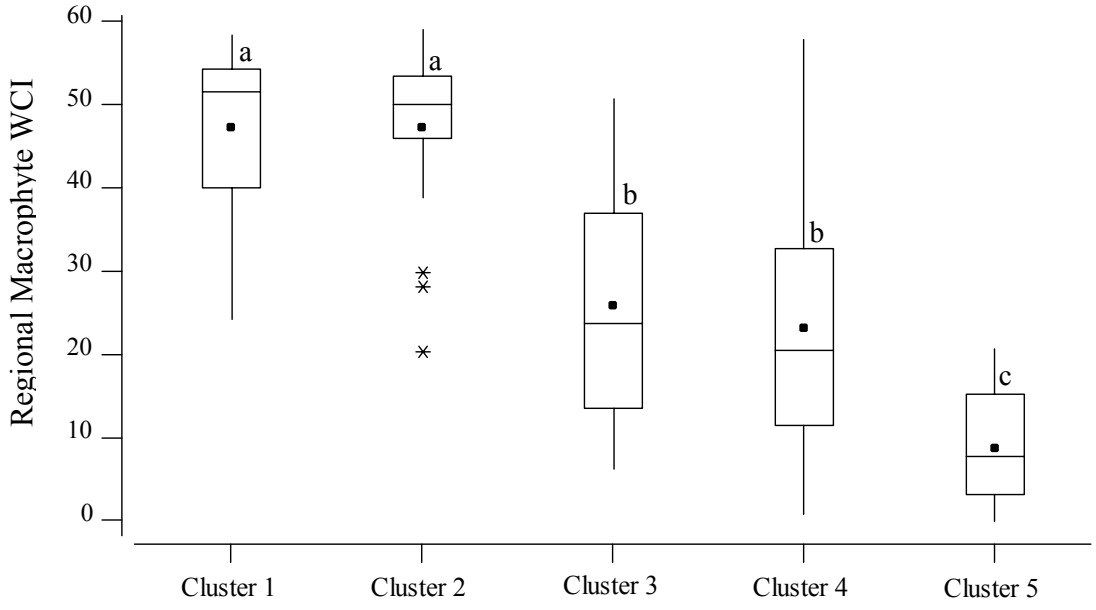


Figure 3-20. Regional macrophyte WCI scores for 5 wetland clusters based on macrophyte community composition.

Table 3-28. Macrophyte WCI scores and LDI values for wetland clusters based on macrophyte community composition.

Cluster	Statewide Macrophyte WCI	Regional Macrophyte WCI	LDI
1	46.8 ± 11.1[a]	47.4 ± 9.9[a]	2.3 ± 1.9[a]
2	44.5 ± 7.7[a]	47.4 ± 9.7[a]	1.7 ± 1.3[a]
3	24.9 ± 12.2[b]	25.9 ± 12.6[b]	4.4 ± 1.9[bc]
4	21.9 ± 13.8[b]	23.2 ± 16.2[b]	4.0± 2.0[b]
5	10.6 ± 7.0[c]	8.8 ± 6.6[c]	5.2 ± 0.5[c]

Clusters with similar letters within columns were not significantly different ($p<0.05$).

other. Cluster 5 was significantly different from all other clusters. Identical results were obtained using statewide macrophyte WCI scores. Table 3-28 provides means and standard deviations for cluster statewide macrophyte WCI scores, regional macrophyte WCI scores, and LDI.

Macroinvertebrates

Statewide 79 wetlands were sampled for the macroinvertebrate assemblage, with 118 species, representing 169 genera, 85 families, 24 orders, 9 classes, and 5 phyla. The most common macroinvertebrate genera identified were *Polypedilum, Dero*, and *Goeldichironomous,* comprising 19%, 18%, and 8% of all the individual macroinvertebrates identified to the genus or lower taxonomic level, respectively. Four genera, *Polypedilum, Dero*, *Goeldichironomous,* and *Kiefferulus,* were found at over 50% of the study wetlands. Of the genera encountered, 81 genera (48%) occurred at a minimum of 5% of the sample wetlands ($n \geq 4$). Approximately one-third of the genera identified (53 genera or 31%) were encountered at only one wetland.

The most common families identified included Chironomidae, Naididae, Enchytraeidae, and Culicidae, representing 39, 19, 4, and 4% of the individuals

identified, and occurring at 99, 81, 52, and 56% of the study wetlands, respectively. Macroinvertebrates in the family Chironomidae were further divides into the subfamilies Chironominae (89% of Chironomidae), Tanypodinae (10% of Chironomidae), and Orthocladiinae (1% of Chironomidae). Six orders were found at over 50% of the wetlands sampled, including Diptera (47% of individual identified to the order taxonomic level or lower), Tubificida (24%), Coleoptera (6%), Basommatophora (5%), Odonata (4%), and Hemiptera (3%). The most common classes of macroinvertebrates identified included Insecta (63%), Oligochaeta (24%), Gastropoda (6%), and Crustacea (5%), all occurring at over 50% of the study wetlands. Five phylum were identified, including Arthropoda, Annelida, Mollusca, Platyhelminthes, and Nemertea, with Arthropoda, Annelida, Mollusca found at 100, 92, and 65% of the wetlands sampled, respectively.

In the panhandle ecoregion, 13 wetlands were sampled hosting 84 genera representing 48 families and 17 orders. In the north ecoregion 15 wetlands were sampled with 87 genera (58 families and 20 orders) encountered. The central ecoregion included 25 wetlands with 109 genera (60 families and 23 orders) recognized. The south ecoregion had 26 sample wetlands with 105 genera (in 60 families and 21 orders) identified.

Summary Statistics

Richness (R), evenness (E), Shannon diversity (H), and Simpson's index (S) were calculated for each sample wetland (Appendix F). Richness ranged from 1 genera (*Pristina*) at PR4 (embedded in a low-intensity silvicultural land use in a National Forest), to 26 genera at PA3 (surrounded with row crops). Species evenness ranged from 0.00 at PR4 to 0.97 at PR5 (surrounded by upland forest). Shannon diversity ranged from 0 at PR4 to 0.92 at CA7 (surrounded by silvicultural operations and pasture with cattle).

Table 3-29. Macroinvertebrate richness, evenness, and diversity among *a priori* land use categories.

	Reference	Agricultural	Urban
Richness (R)	14 ± 6[a]	14 ± 6[a]	13 ± 5[a]
Evenness (E)	0.69 ± 0.24[a]	0.69 ± 0.12[a]	0.68 ± 0.15[a]
Shannon Diversity (H)	0.70 ± 0.26[a]	0.72 ± 0.15[a]	0.70 ± 0.17[a]
Simpson's Index (S)	1.83 ± 0.75[a]	1.81 ± 0.54[a]	1.73 ± 0.55[a]
Beta Diversity	8.0	7.8	8.5
Gamma Diversity	114	110	111

Categories with similar letters were not significantly different (Fisher's LSD, α=0.05)
Values represent mean ± standard deviation

Simpson's index was highest at CR10 at 2.79. Table 3-29 summarizes the richness, evenness, and diversity calculations by *a priori* land use category. No significant differences were found in richness, evenness, Shannon diversity, or Simpson's index among the 3 *a priori* land use categories. Beta and gamma diversity were also similar among *a priori* land use categories.

Table 3-30 shows that the no significant differences were found for richness, evenness, Shannon diversity, or Simpson's index between wetlands in low and high LDI groups. Beta and gamma diversity were higher for the high LDI groups with beta diversity at 10.9 and gamma diversity at 146 for the high LDI group and beta diversity at 8.7 and gamma diversity at 124 for the low LDI group wetlands.

Compositional Analysis

MRPP was used to test the similarity of macroinvertebrate genera composition across all ecoregions (panhandle versus north versus central versus south) as well as for multiple pair wise ecoregion comparisons (panhandle versus north, panhandle versus central, panhandle versus south, north versus central, north versus south, and central versus south). Among all wetlands, the comparison across all groups and the multiple pair wise comparisons suggested macroinvertebrate community composition at the

Table 3-30. Macroinvertebrate richness, evenness, and diversity for LDI groups.

	Low LDI	High LDI	W^	p`
Richness (R)	14 ± 6	13 ± 5	661	0.33
Evenness (E)	0.68 ± 0.22	0.69 ± 0.14	687	0.47
Shannon Diversity (H)	0.70 ± 0.24	0.71 ± 0.17	695	0.52
Simpson's Index (S)	1.82 ± 0.71	1.77 ± 0.56	678	0.42
Beta Diversity	8.7	10.9		
Gamma Diversity	124	146		

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

genera level was significantly different(at the $\alpha = 0.05$ level), with the exception of the pair wise comparison between the panhandle and north ecoregions (Table 3-31). The macroinvertebrate community composition of the panhandle and north ecoregions was not significantly different for all tests, including among all wetlands and for reference, agricultural, and urban wetlands independently.

In reference wetlands, the south ecoregion had a significantly different macroinvertebrate community composition as compared to all other ecoregions (panhandle versus south $T = -3.2$, $p = 0.00$; north versus south $T = -3.3$, $p = 0.00$; central versus south $T = -2.1$, $p = 0.03$). There were not significant differences in macroinvertebrate community composition between agricultural wetlands in any neighboring ecoregions; however, macroinvertebrate community composition in the south ecoregion was significantly different from both the panhandle and north ecoregions; as were the central and panhandle ecoregions. The only ecoregions with significantly different macroinvertebrate community composition among urban wetlands were the panhandle and south ecoregions.

Table 3-31. Macroinvertebrate community composition similarity among *a priori* land use categories and ecoregions.

	Sites (n)	T^	A`	p#
All Wetlands				
All regions (P vs N vs C vs S)	79	-7.1	0.10	0.00*
Panhandle vs north	28	0.5	-0.01	0.67
Panhandle vs central	38	-3.6	0.05	0.00*
Panhandle vs south	39	-8.5	0.13	0.00*
North vs central	40	-2.4	0.04	0.02*
North vs south	41	-6.6	0.10	0.00*
Central vs south	51	-3.2	0.04	0.00*
Reference wetlands				
All regions (P vs N vs C vs S)	29	-3.2	0.13	0.00*
Panhandle vs north	12	-0.2	0.01	0.35
Panhandle vs central	14	-0.7	0.03	0.21
Panhandle vs south	15	-3.8	0.17	0.00*
North vs central	14	-0.5	0.03	0.27
North vs south	15	-3.3	0.15	0.00*
Central vs south	17	-2.1	0.08	0.03*
Agricultural wetlands				
All regions (P vs N vs C vs S)	24	-2.7	0.12	0.01*
Panhandle vs north	8	-0.5	0.04	0.30
Panhandle vs central	12	-2.1	0.10	0.03*
Panhandle vs south	12	-3.3	0.18	0.00*
North vs central	12	-1.0	0.04	0.17
North vs south	12	-2.3	0.12	0.02*
Central vs south	16	-0.5	0.02	0.30
Urban wetlands				
All regions (P vs N vs C vs S)	26	-1.2	0.05	0.11
Panhandle vs north	8	-1.0	0.09	0.16
Panhandle vs central	12	-0.6	0.03	0.27
Panhandle vs south	12	-2.1	0.09	0.03*
North vs central	14	0.7	-0.03	0.75
North vs south	14	-1.6	0.07	0.07
Central vs south	18	-1.1	0.03	0.14

*A high |T| value and significant p-value ($p<0.05$) suggests a difference in species composition

^ T = the MRPP test statistic

`A = the chance corrected within-group agreement

#p = the significance value.

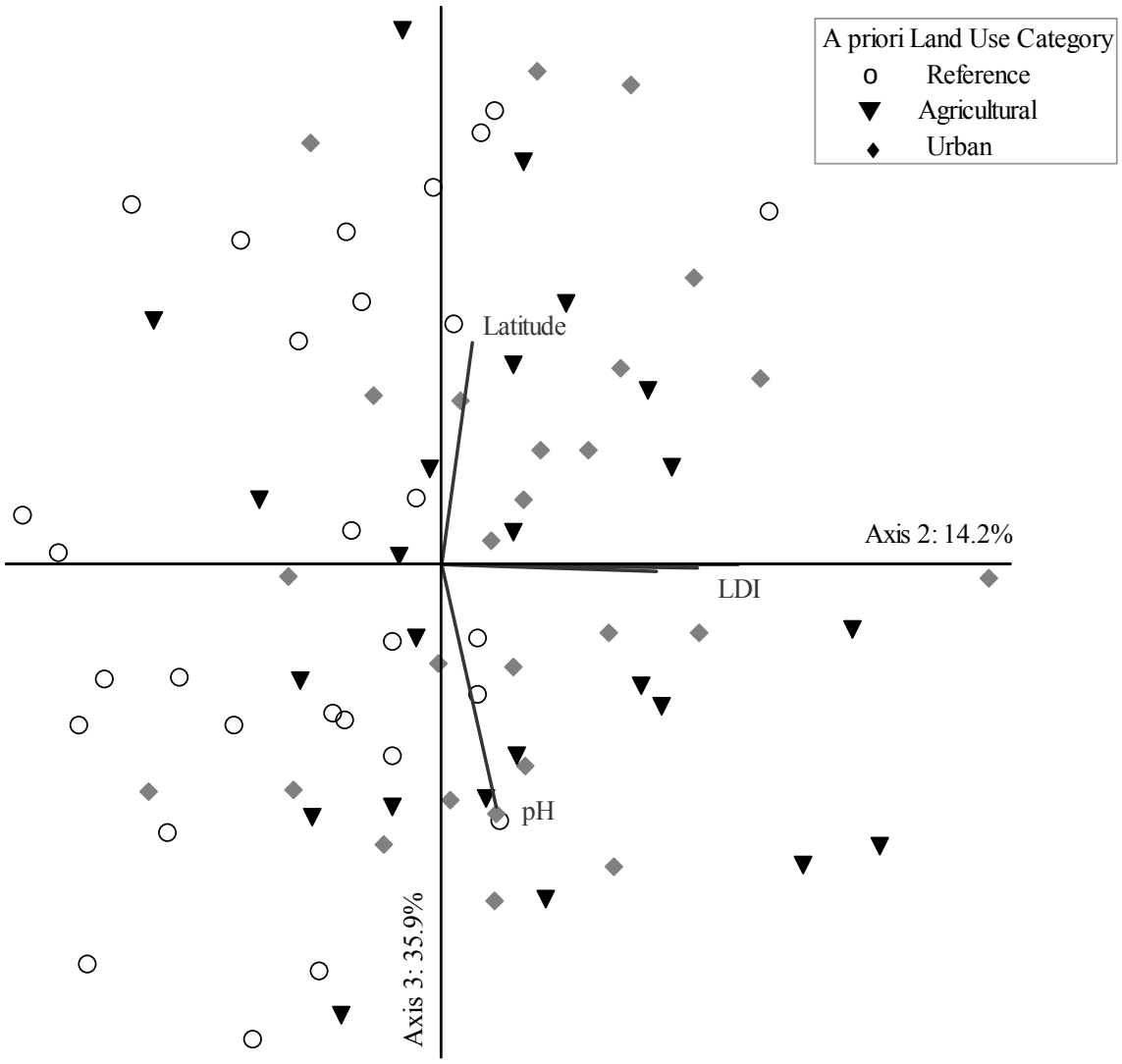


Figure 3-21. NMS ordination bi-plot for 79 wetlands in macroinvertebrate genus space with an overlay of environmental parameters. Latitude, LDI, and water column pH (shown as radiating vectors), were significantly correlated with the NMS axes based on macroinvertebrate community composition. Vector length represents the strength of the correlation, and the angle represents the direction of maximum change. Axis 2 explained 14.2% variance, axis 3 explained 35.9% variance, and axis 1 (not shown) represented an additional 18.9% variance.

Table 3-32. Pearson's r-squared correlations between environmental variables and NMS ordination axes based on macroinvertebrate community composition.

	Axis 1	Axis 2	Axis 3
Incremental r^2	18.9%	14.2%	35.9%
Cumulative r^2	18.9%	33.0%	68.9%
Latitude	0.17	0.03	0.22
Longitude	0.05	0.02	0.09
LDI	0.01	0.25	0.01
Log(DO)	0.01	0.13	0.00
Log(Temperature)	0.03	0.05	0.07
Log(Color)	0.02	0.02	0.10
Log(Turbidity)	0.00	0.09	0.02
pH	0.00	0.06	0.24
Log(Water ammonia-N)	0.04	0.08	0.01
Log(Water nitrate/nitrite-N)	0.01	0.02	0.05
Log(Water TKN)	0.05	0.08	0.00
Log(Water TP)	0.02	0.18	0.01
Arcsin Sqrt (Soil moisture)	0.02	0.04	0.09
Log(Soil TP)	0.10	0.04	0.01

Community Composition

Macroinvertebrate community composition was summarized in an NMS ordination to relate changes in macroinvertebrate community composition with environmental variables. Figure 3-21 shows a 2 dimensional bi-plot of the NMS axes. Overlays of significant environmental variables include water column pH, LDI, and latitude. Table 3-32 provides the Pearson's r-squared correlation coefficients between environmental variables and NMS ordination axes. A three dimensional solution was constructed with an overall stress of 19.8 with a final stability of 0.04. Axis 1 explained 18.9% of the variance and was not correlated with any measured environmental parameters. Axis 2 explained 14.2% variance and was correlated with LDI; axis 3 explained 35.9% variance and was correlated with latitude and water column pH.

Metric Selection

Over 20 of the candidate metrics were significantly correlated with LDI (Spearman's $|r| > 0.30$, $p < 0.01$). Table 3-33 provides the statewide Spearman correlation values between the 6 macroinvertebrate metrics and LDI, water column pH, dissolved oxygen, and water column TP. Macroinvertebrate metrics selected for inclusion represented tolerance, community balance, and functional group metrics. Tolerance metrics included the tolerant indicator genera, sensitive indicator genera, and Florida Index. Community balance metrics included Mollusca (phylum taxonomic level) and Noteridae (family taxonomic level). One functional groups metric was included, scrapers. The percent of tolerant indicator genera, Mollusca, and scrapers increased with increasing development intensity, whereas sensitive indicator genera, Florida Index, and Noteridae decreased with increasing development intensity. Table 3-34 shows that scores of selected metrics and the macroinvertebrate WCI were significantly different between low and high LDI groups ($p < 0.05$).

Table 3-33. Spearman correlations between macroinvertebrate metrics and the macroinvertebrate WCI with LDI, pH, log(DO), and log(TP).

Macroinvertebrate Metrics	LDI	pH	Log(DO)	Log(TP)
Tolerance metrics				
Tolerant indicator genera	0.51	0.62	-0.25	
Sensitive indicator genera	-0.47		0.39	-0.37
Florida Index	-0.35		0.35	-0.24
Community balance				
Mollusca	0.33	0.54	-0.28	
Noteridae	-0.34			
Functional group				
Scraper	0.30			
Macroinvertebrate WCI	-0.62	-0.56	0.48	-0.34
All correlations shown are significant ($p<0.05$)				

Table 3-34. Macroinvertebrate metric and WCI scores between LDI groups.

Metric	Low LDI	High LDI	W^	p`
Tolerant indicator species	4.4 ± 11.8	14.2 ± 15.5	904.0	<0.001
Sensitive indicator species	15.5 ± 20.3	2.2 ± 4.4	1679.0	<0.001
Florida Index	2.1 ± 2.3	0.9 ± 1.2	1572.0	0.008
Mollusca	2.0 ± 3.5	9.7 ± 13.4	1025.0	0.003
Noteridae	1.8 ± 3.7	0.2 ± 0.7	1518.0	0.012
Scrapers	4.2 ± 6.7	10.9 ± 12.8	1046.0	0.006
Macroinvertebrate WCI	36.8 ± 10.0	22.3 ± 8.4	1878.5	<0.001

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

Tolerance metrics

Multiple ISA using genus-level abundance data were completed at different LDI breaks, starting at 1.0 and continuing through 7.0, at 0.25 step increments. Table 3-35 shows the results of the iterative ISA calculations. The statewide tolerant indicator genera were established at an LDI break of 4.0, and included 6 genera, *Goeldichironomus*, *Micromenetus*, *Microvelia*, *Physella*, *Tropisternus*, and *Tanypus* (Table 3-36).

Figure 3-22 shows tolerant indicator genera increased with increasing development intensity. Two outliers were apparent in the low LDI group, including SU4 (tolerant = 61%) and SA8 (tolerant = 31%). Wetlands in the high LDI group with the highest percent tolerant indicator genera included 4 central (CA9, CA3, CU5, CA2) and 1 north (NA4) ecoregion wetland.. All 5 wetlands were surrounded by different land uses, including citrus crops (CA9), pullet farm spray field (CA3), residential and commercial (CU5), pasture (CA2), and row crops (NA4).

Table 3-35. Macroinvertebrate ISA calculations over a range of LDI values. Highlighted areas indicate the LDI value selected for sensitive and tolerant indicator species.

LDI	Low LDI n* =	High LDI n* =	Sensitive r^	Sensitive p`	Tolerant r^	Tolerant p`	No. Sensitive Indicators	No. Tolerant Indicators
1	10	69	-0.39	0.00	xx	xx	8	0
1.25	26	53	-0.43	<0.0001	0.47	<0.0001	16	3
1.5	30	49	-0.45	<0.0001	0.46	<0.0001	11	5
1.75	32	47	-0.46	<0.0001	0.48	<0.0001	14	5
2	33	46	-0.48	<0.0001	0.49	<0.0001	11	3
2.25	35	44	-0.52	<0.0001	0.49	<0.0001	12	3
2.5	35	44	-0.52	<0.0001	0.49	<0.0001	12	3
2.75	36	43	-0.54	<0.0001	0.48	<0.0001	13	4
3	37	42	-0.54	<0.0001	0.48	<0.0001	13	4
3.25	38	41	-0.53	<0.0001	0.47	<0.0001	13	4
3.5	39	40	-0.47	<0.0001	0.49	<0.0001	9	5
3.75	39	40	-0.47	<0.0001	0.49	<0.0001	9	5
4	42	37	-0.52	<0.0001	0.50	<0.0001	9	6
4.25	43	36	-0.53	<0.0001	0.50	<0.0001	7	5
4.5	46	33	-0.48	<0.0001	0.46	<0.0001	8	2
4.75	50	29	-0.40	0.00	0.51	<0.0001	7	4
5	56	23	-0.31	0.01	0.43	<0.0001	3	5
5.25	61	18	-0.20	0.07	0.55	<0.0001	3	5
5.5	64	15	-0.21	0.07	0.20	0.08	3	5
5.75	66	13	-0.04	0.70	0.13	0.25	1	3
6	66	13	-0.04	0.70	0.13	0.25	1	3
6.25	70	9	xx	xx	0.06	0.58	0	7
6.5	72	7	0.03	0.82	0.21	0.07	1	13
6.75	74	5	xx	xx	0.19	0.10	0	13
7	75	4	xx	xx	0.38	0.00	0	6

*n = number of sites.
^r = Spearman's r correlation coefficient of indicator species versus LDI.
`p = significance value.

Table 3-36. Macroinvertebrate tolerant indicator genera. Tolerant indicator genera calculated at an LDI break of 4.0 were significant ($p < 0.10$).

Phylum	Class	Order	Family	Genera	Indicator Value	p-value
Arthropoda	Insecta	Coleoptera	Hydrophilidae	*Tropisternus*	17.2	0.053
		Diptera	Orthocladiinae	*Goeldichironomus*	56.9	0.001
			Chironomidae	*Tanypus*	8.1	0.098
		Heteroptera	Veliidae	*Microvelia*	12.9	0.033
Mollusca	Gastropoda	Basommatophora	Planorbidae	*Micromenetus*	22.5	0.084
			Physidae	*Physella*	21.5	0.002

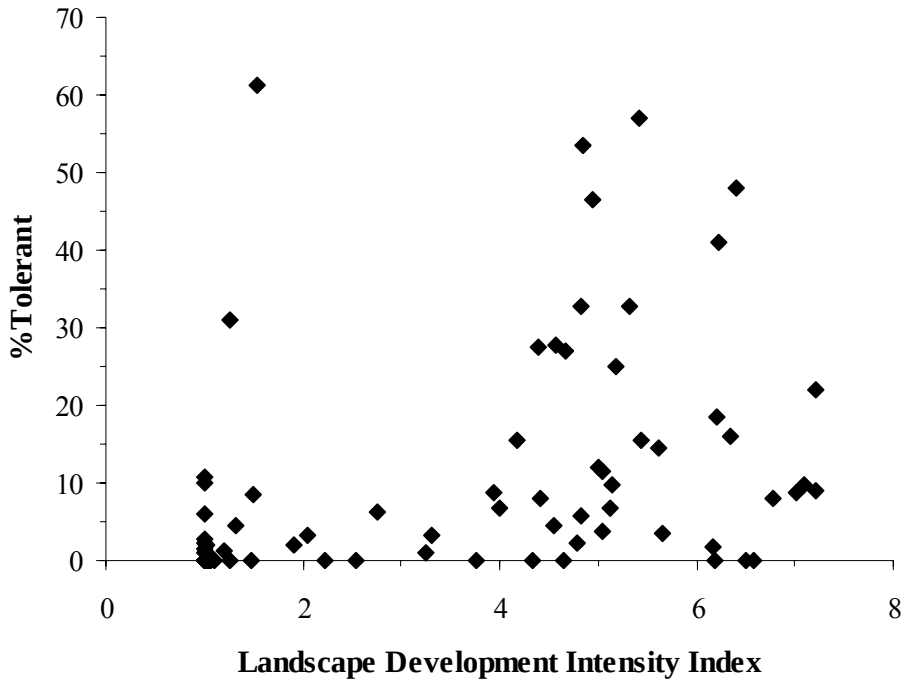


Figure 3-22. Tolerant macroinvertebrate indicator genera increased with increasing development intensity (LDI).

The 14 sensitive indicator genera (Table 3-37) were calculated at an LDI break of 1.75. The 5 sensitive indicator genera with the highest indicator values included *Bersous*, *Hydrocanthus*, *Larsia, Pristina*, and *Pristinella*. Sensitive indicator genera included macroinvertebrates in 2 phyla, Annelida and Arthropoda. The phylum Annelida was represented by 2 genera of aquatic worms, *Prisina* and *Pristinella*, both in the family Naididae, order Haplotaxida, class Oligochaeta. The 12 remaining sensitive indicator genera fell within the phylum Arthropoda, representing 2 classes Arachnida (including a water mite); and Insecta, aquatic insects in 3 orders including Coleoptera (5 genera of beetles), Diptera (4 genera of true flies), and Trichoptera (2 genera of caddis flies).

Figure 3-23 shows sensitive indicator genera decreased with increasing development intensity in the landscape. Two outliers occurred, 1 in each LDI group. All of the macroinvertebrates identified at PR4 (in the low LDI group) were sensitive

Table 3-37. Sensitive macroinvertebrate indicator genera. Sensitive indicator genera calculated at an LDI break of 1.75 were significant ($p < 0.10$).

Phylum	Class	Order	Family	Genera	Indicator Value	p-value
Annelida	Oligochaeta	Haplotaxida	Naididae	*Pristina*	35.5	0.008
				Pristinella	25.9	0.094
Arthropoda	Arachnida	Acariformes	Hydrachnidae	*Hydrachna*	9.4	0.049
	Insecta	Coleoptera	Dytiscidae	*Laccophilus*	8.9	0.061
			Haliplidae	*Haliplus*	8.8	0.055
			Hydrophilidae	*Berosus*	28.8	0.003
			Noteridae	*Hydrocanthus*	28.5	0.004
				Suphis	12.5	0.048
		Diptera	Chironomidae	*Larsia*	24.8	0.011
				Paramerina	14	0.062
				Zavreliella	17.3	0.008
			Orthocladiinae	*Dicrotendipes*	14.8	0.065
		Trichoptera	Leptoceridae	*Oecetis*	12.5	0.023
			Hydroptilidae	*Oxyethira*	12.5	0.032

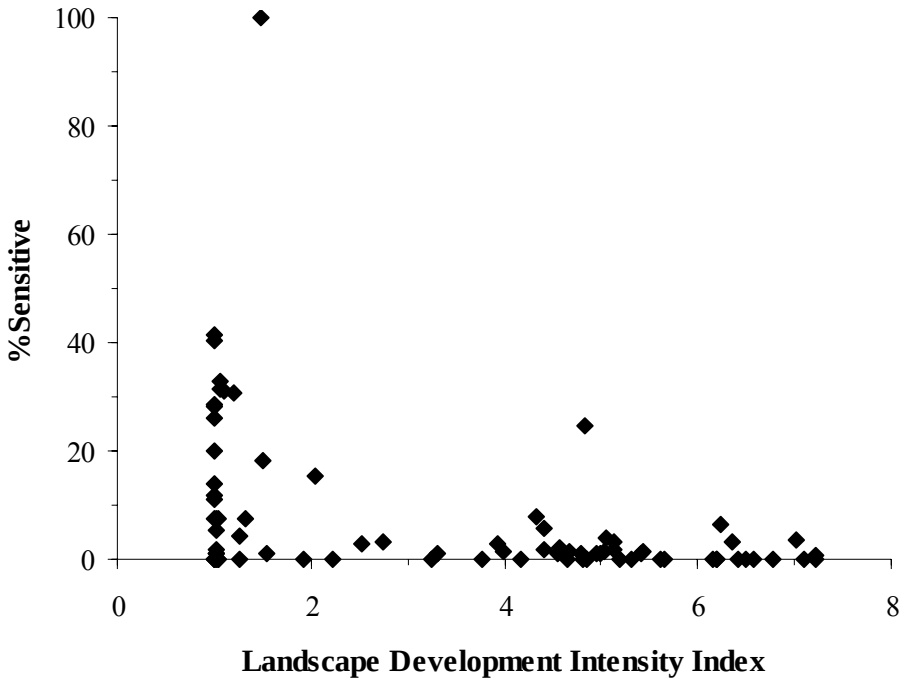


Figure 3-23. Sensitive macroinvertebrate indicator genera decreased with increasing development intensity (LDI).

indicator genera (100%). Wetlands hosting the next highest sensitive macroinvertebrate indicator genera were NR3 with only 42% sensitive indicator genera and CR11 with 41%. The outlier in the high LDI group with a low presence of sensitive indicator genera was SU2 (sensitive = 25%).

The third tolerance metric was the Florida Index, an index based on the relative pollution tolerance of macroinvertebrates identified in a water body (USEPA 2002c; Beck 1954). Calculations for the Florida Index included scoring Class I organisms, which were considered least tolerant, and Class II organisms, which were considered intolerant of pollution. Mixed taxonomic levels were included in the Florida Index from species (example: *Polypedilum halterale*) to genus (example: all species of *Elimia*) to family (example: all species of Gammaridae) to order (example: all species of Plecoptera; USEPA 2002c; Beck 1954). The Florida Index value was expected to decrease with

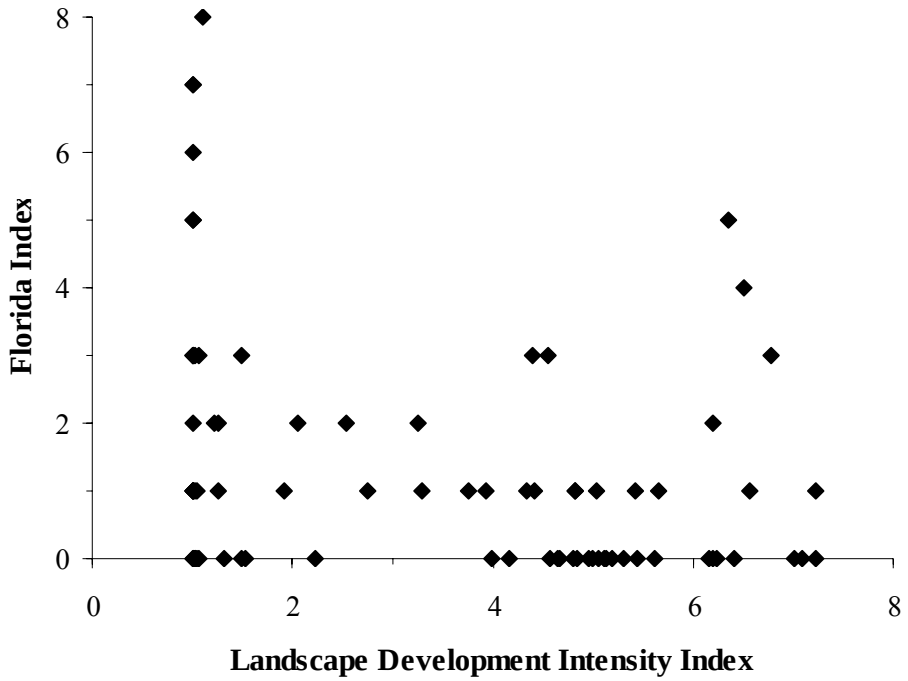


Figure 3-24. Florida Index scores decreased with increasing development intensity (LDI).

increasing development intensity in the surrounding landscape (Barbour et al. 1996a). Figure 3-24 shows that the Florida Index score generally decreased with increasing development intensity in the surrounding landscape, with 6 outliers including 4 urban and 2 agricultural wetlands. The 5 highest scoring wetlands in the low LDI group included wetlands in each ecoregion including CR4 (Florida Index = 8), SR8 (Florida Index = 7), NR3 (Florida Index = 7), CR11 (Florida Index = 6), and PR8 (Florida Index = 5).

Community balance metrics

Two community balance metrics were incorporated into the macroinvertebrate WCI including percent Mollusca and percent Noteridae. The percent of individuals in the phylum Mollusca was significantly correlated with the LDI (Table 3-33) and significantly differentiated between low and high LDI groups (Table 3-34). Figure 3-25 shows that the percent of macroinvertebrates in the phylum Mollusca increased with increasing

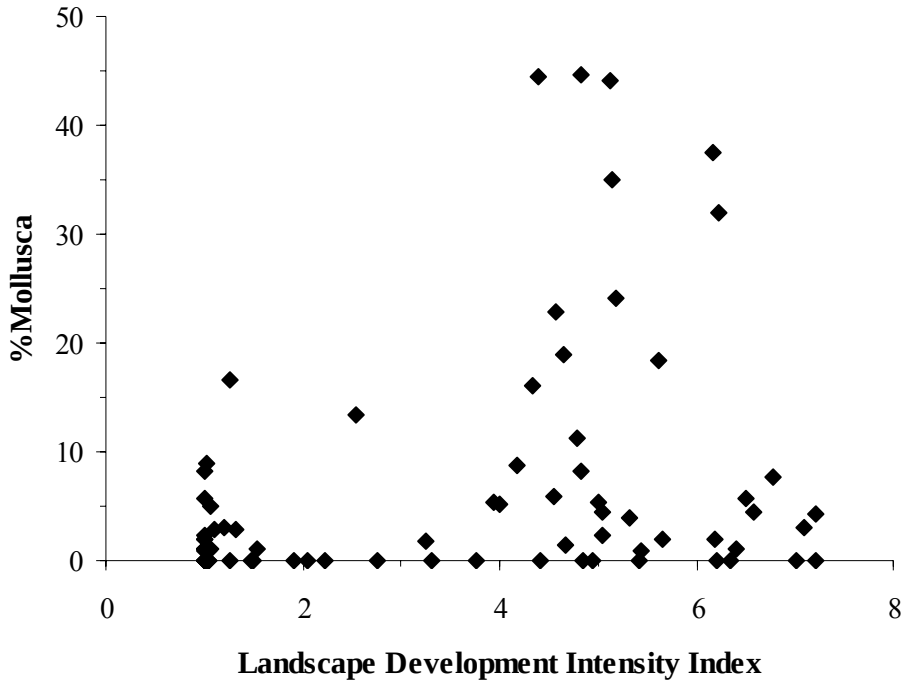


Figure 3-25. Macroinvertebrates in the phylum Mollusca increased with increasing development intensity (LDI).

development intensity. Macroinvertebrates were identified in 3 classes within the order Mollusca, including Bivalva, Gastropoda, and Plecypoda. Nearly two-thirds of the wetlands hosted macroinvertebrates in the phylum Mollusca (n=51). In 5 wetlands over one-third of the macroinvertebrates that were identified belonged to the phylum Mollusca, including SU2 (44.7%), CU8 (44.4%), SU7 (44.1%), SA4 (37.5%), and CA5 (34.9%). In the low LDI group, 4 wetlands had greater than 5% of the identified macroinvertebrates in the phylum Mollusca, including SA8 (16.7%), PR7 (9.0%), SR4 (8.1%), and CR3 (5.8%).

Figure 3-26 shows that the percent of macroinvertebrates in the family Noteridae decreased with increasing landscape development intensity as expected (Barbour et al 1996b). Macroinvertebrates in the family Noteridae never made up more than 15% of the

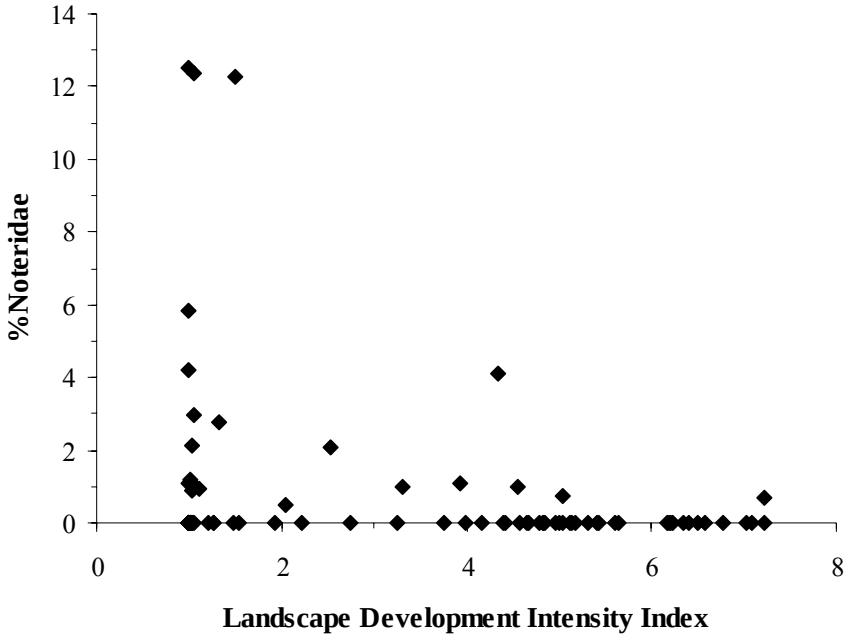


Figure 3-26. Macroinvertebrates in the family Noteridae decreased with increasing development intensity (LDI).

individuals identified to the family taxonomic level or lower at any of the sample wetlands. The family Noteridae falls within order Coleoptera, class Insecta, phylum Arthropoda. The family Noteridae (burrowing water beetles), typically inhabit the shallow margins of standing or slow flowing streams, or in lentic habitats act as climbers associated with vascular macrophytes or as burrowers (Williams and Feltmate 1992; White and Brigham 1996). Both the larvae and adults of macroinvertebrates in the family Noteridae are aquatic (Peckarsky et al. 1993), one of only 5 families within Coleoptera with both life stages being aquatic. As a general rule, the percent of macroinvertebrates in the order Coleoptera was found to decrease with increasing development surrounding Florida streams (Barbour et al. 1996b). One apparent outlier in the high LDI group was CA4 (4.1%).

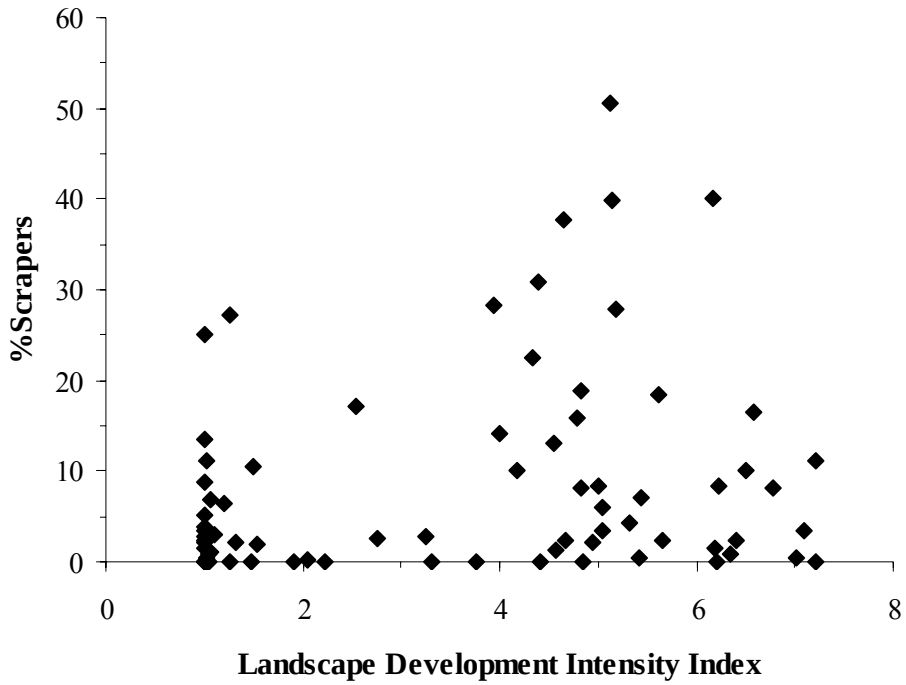


Figure 3-27. Macroinvertebrates that belong to the scraper functional feeding group increased with increasing development intensity (LDI).

Functional group metrics

One functional group metric was selected for inclusion in the macroinvertebrate WCI, the percent scrapers functional feeding group. Figure 3-27 shows the percent macroinvertebrates in the scraper functional feeding group increased with increasing landscape development intensity (LDI). The scraper functional feeding group included macroinvertebrates that scrape periphyton from mineral and organic surfaces and those that browse or graze algal materials. Two outliers in the low LDI group included SA8 (27%), and CR3 (25%). Five wetlands with the highest percent scrapers were found in among all 4 ecoregions and represented wetlands embedded in a mix of urban and agricultural land uses, including SU7 (51%), SA4 (40%), CA5 (40%), PA6 (38%), and CU8 (31%). Nearly one-quarter of the sample wetlands (n = 19) did not have scrapers identified in the macroinvertebrate samples.

Macroinvertebrate Wetland Condition Index

The 6 metrics described above were included in the macroinvertebrate WCI. Figure 3-28 shows the relationship between the macroinvertebrate WCI and LDI. Potential scores for the macroinvertebrate WCI range from 0-60, with higher values representing reference wetlands. Actual scores ranged from 5.2 at SU5 (deeply flooded swamp surrounded by industrial land use, LDI = 5.2), to 57.0 at SR3 (surrounded by native pine flatwoods, LDI = 1.0). Ranges varied regionally, though the regional scores were not significantly different for either low or high LDI groups. The highest scores in each ecoregion included 4 reference wetland, PR8 (40.7), NR3 (52.8), CR6 (50.4), and

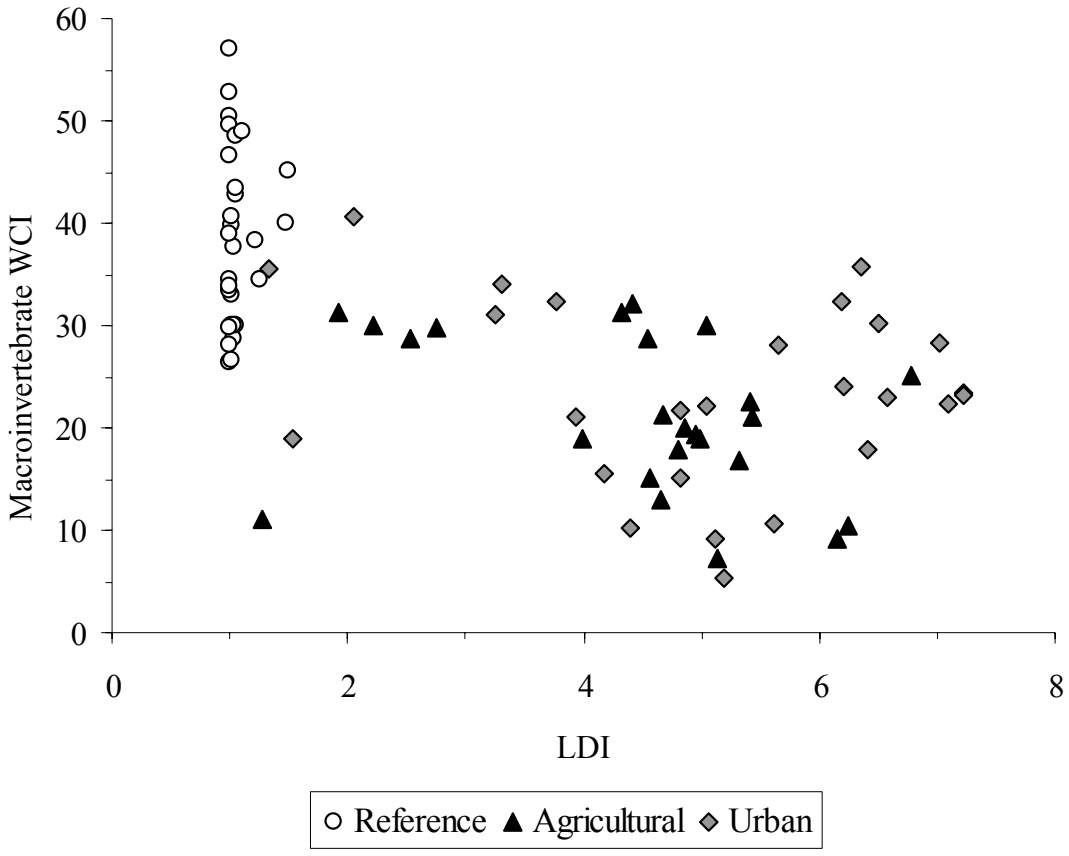


Figure 3-28. Macroinvertebrate WCI scores decreased with increasing landscape development intensity index (LDI).

SR8 (57.0), in the panhandle, north, central, and south ecoregions, respectively. The lowest scoring wetlands in both the panhandle and central ecoregions were embedded in pasture, including PA6 (12.9) and CA5 (7.2). In the north ecoregion NA4 (surrounded by row crops) scored 10.4. In the south SU5 (surrounded by urban land use) received the lowest score overall of 5.3. The macroinvertebrate WCI was significantly correlated with the LDI index (Spearman's $|r| = 0.62$, $p < 0.001$). A Kruskal-Wallis test suggested a significant difference ($H = 36.0$, $p < 0.001$) among median macroinvertebrate WCI scores for study wetlands in *a priori* land use categories.

Cluster Analysis

Cluster analysis determined 5 categories based on macroinvertebrate community composition. Clusters were explained by regions, *a priori* land use categories, and water level including: 1: south to central low development intensity; 2: mixed region low development intensity; 3: north central to panhandle middle development intensity; 4: northern to panhandle middle development intensity; and 5: high development intensity and southern Everglades. Figure 3-29 shows that based on macroinvertebrate WCI scores, clusters 1 and 2 were not significantly different from one another, but were significantly different from cluster 5. Clusters 3 and 4 were significantly different from cluster 1 and cluster 5. Table 3-38 provides means and standard deviations for macroinvertebrate WCI and LDI scores of the 5 clusters.

Wetland Condition Index

In total 19 metrics were used to construct the WCI, including 7 metrics based on the diatom assemblage, 6 metrics based on the macrophyte assemblage, and 6 metrics based on the macroinvertebrate assemblage. Table 3-39 lists the WCI scores for 118 isolated forested wetlands for each assemblage. Scores ranged from 0-70 for the diatom

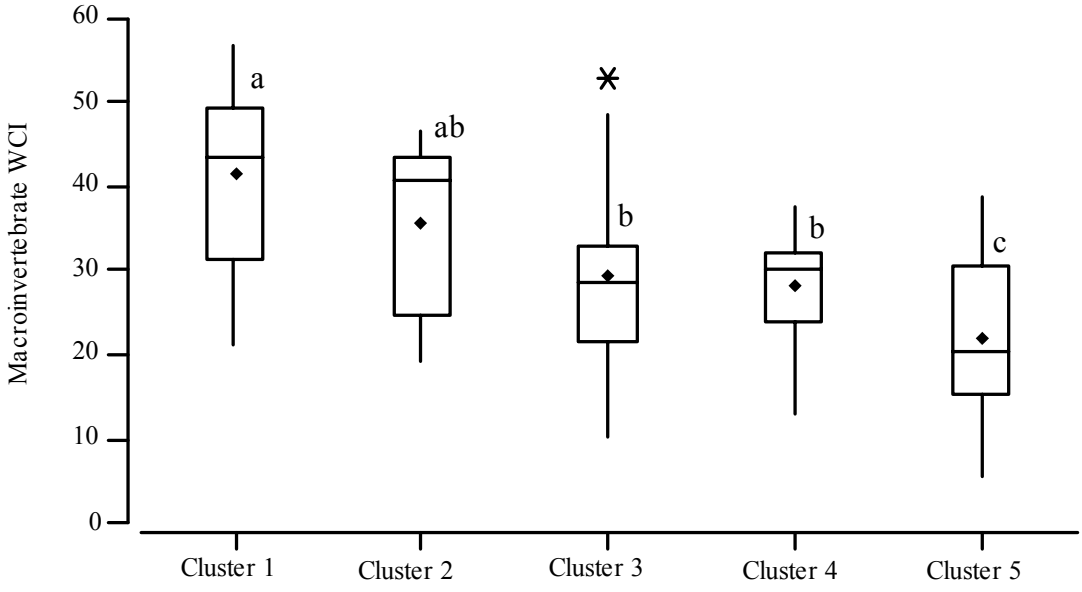


Figure 3-29. Macroinvertebrate WCI scores for 5 wetland clusters based on macroinvertebrate community composition. Clusters with similar letters were not significantly different (Fisher's LSD, $p<0.05$).

WCI and from 0-60 for both the macrophyte and macroinvertebrate WCI. Different wetlands received the highest and lowest scores for each WCI. The highest scores for each WCI were found at wetlands in the south and central ecoregions, including SR2 (diatom WCI = 68.9), CR11 (macrophyte WCI =59.0), and SR8 (macroinvertebrate WCI= 57.0). Minimum WCI scores were found among 3 different ecoregions, includingCA3 (diatom WCI = 7.9), NA1 (macrophyte WCI = 0.0), and SU2 (macroinvertebrate WCI = 5.3). Within the north and central ecoregions some wetlands received the ecoregion maximum scores for multiple assemblages, including NR3 (diatom WCI = 66.8; macroinvertebrate WCI =52.8) and CR6 (diatom WCI = 65.5; macroinvertebrate WCI = 50.4). In the panhandle ecoregion, PU4 received minimum WCI scores for both the diatom and macrophyte assemblages (diatom WCI = 10.5; macrophyte WCI = 4.0). Figure 3-30 shows a three dimensional scatter plot for the 50

Table 3-38. Macroinvertebrate WCI scores and LDI values for wetland clusters based on macroinvertebrate community composition.

	Cluster 1	Cluster 2	Cluster 3	Cluster 4	Cluster 5
Mi-WCI	41.6 ± 10.7[a]	35.5 ± 10.8[ab]	29.3 ± 11.8[b]	28.3 ± 6.2[b]	21.9 ± 9.7[c]
LDI	2.0 ± 1.5[a]	3.1 ± 2.5[ab]	3.4 ± 2.3[ab]	3.4± 2.3[ab]	4.0 ± 2.1[b]

Clusters with similar letters were not significantly different ($p<0.05$)

wetlands receiving scores for all three assemblages. The maximum diatom WCI for the 50 wetlands graphed was 68.9 (of 70) at SR2. The maximum macrophyte WCI was 58.4 (of 60) at PR6 (LDI = 1.3), and the maximum macroinvertebrate WCI was 52.8 (of 60) at NR3 (LDI = 1.0). The 1:1:1 line is shown for convenience in interpretation.

Figure 3-31 shows 2 dimensional comparisons of wetlands scored with multiple assemblages, including (A) 50 wetlands with diatom and macrophyte WCI scores, (B) 50 wetlands with diatom and macroinvertebrate WCI scores, and (C) 79 wetlands with macrophyte and macroinvertebrate WCI scores. The most obvious outlier between the diatom and macrophyte WCI scores (Figure 3-31 A) was at CU6, a wetland surrounded by a golf course that had been developed within the past five years. CU6 had low scores for the diatom (15.1 of 70) and macroinvertebrate (23.4 of 60) but a higher score for the macrophyte (41.5 of 60) WCI.

The 19 metrics incorporated into the WCI were compared across different assemblages (Table 3-40). Of the 120 potential metric comparisons (42 among diatom and macrophyte metrics, 42 among diatom and macroinvertebrate metrics, and 36 among macrophyte and macroinvertebrate metrics), 53% of the comparisons were significantly correlated at the $p < 0.01$ level. An additional 20% of the potential comparisons were significantly correlated at the more flexible $p < 0.05$ level; and an additional 6% at the more flexible $p < 0.10$ level. Less than one-quarter of the comparisons among metrics of

Table 3-39. WCI scores for 118 wetlands based on three assemblages including diatoms, macrophytes, and macroinvertebrates.

Site Code	Diatom WCI	Macrophyte WCI	MacroinvertebrateWCI
PA1	-	30.4	-
PA2	38.1	11.9	30.1
PA3	34.9	8.3	25.1
PA4	-	12.6	-
PA5	51.1	6.5	21.2
PA6	28.2	7.7	12.9
PA7	-	17.7	-
PA8	-	50.6	-
PA9	-	12.1	-
PA10	-	41.7	-
PR1	61.1	55.9	37.6
PR2	-	50.5	-
PR3	-	49.5	-
PR4	64.5	51.2	40.0
PR5	58.0	53.6	30.0
PR6	63.9	58.4	34.4
PR7	-	34.8	26.5
PR8	-	53.6	40.7
PU1	-	6.2	-
PU2	-	31.5	-
PU3	33.1	31.0	35.7
PU4	10.5	4.0	21.6
PU5	-	22.1	-
PU6	-	16.5	-
PU7	-	24.1	-
PU8	-	33.6	-
PU9	-	48.8	-
PU10	-	9.2	30.2
NA1	-	0.0	-
NA2	-	3.0	-
NA3	-	56.4	-
NA4	33.8	16.3	10.4

Table 3-39. Continued.

Site Code	Diatom WCI	Macrophyte WCI	Macroinvertebrate WCI
NA5	-	2.9	-
NA6	56.3	18.8	16.9
NA7	-	37.0	-
NA8	-	46.0	-
NA9	-	37.3	-
NA10	-	51.5	30.0
NA11	-	32.6	28.7
NA12	-	8.0	-
NR1	-	52.0	-
NR2	65.8	34.8	30.0
NR3	66.8	58.2	52.8
NR4	58.3	42.2	39.8
NR5	-	52.3	-
NR6	57.9	55.0	48.6
NR7	-	52.3	-
NR8	-	58.4	30.0
NR9	-	56.7	33.0
NU1	-	35.2	-
NU2	24.1	23.7	15.5
NU3	-	25.6	-
NU4	54.5	35.1	31.0
NU5	60.0	40.1	24.0
NU6	48.8	20.7	28.0
NU7	-	11.8	-
NU8	-	38.6	-
NU9	-	37.5	-
NU10	-	17.2	23.0
CA1	-	8.9	-
CA2	10.6	0.7	19.4
CA3	7.9	7.1	20.0
CA4	56.9	38.8	31.3
CA5	43.6	26.9	7.2
CA6	22.7	7.1	21.1
CA7	-	9.8	32.1

Table 3-39. Continued.

Site Code	Diatom WCI	Macrophyte WCI	Macroinvertebrate WCI
CA8	-	37.7	31.3
CA9	-	11.8	22.6
CR1	-	51.0	-
CR2	-	49.9	-
CR3	57.7	47.6	33.9
CR4	57.8	51.2	48.9
CR5	43.8	43.5	29.7
CR6	65.5	54.6	50.4
CR7	-	51.7	-
CR8	-	54.3	28.8
CR9	-	49.4	34.4
CR10	-	53.5	45.0
CR11	-	59.0	49.5
CU1	61.1	42.9	40.6
CU2	-	10.0	-
CU3	28.5	13.5	22.3
CU4	-	21.4	-
CU5	21.5	22.3	17.8
CU6	15.1	41.5	23.4
CU7	-	20.7	10.6
CU8	-	21.1	10.1
CU9	-	28.3	28.3
CU10	-	38.3	32.3
CU11	-	21.3	34.1
SA1	-	0.7	-
SA2	34.1	9.4	15.0
SA3	47.9	23.1	28.6
SA4	15.8	11.3	9.1
SA5	46.3	18.9	19.0
SA6	31.9	3.7	19.0
SA7	-	30.8	29.8
SA8	-	34.5	11.0
SA9	-	29.8	17.9
SR1	66.8	54.1	33.4

Table 3-39. Continued.

Site Code	Diatom WCI	Macrophyte WCI	Macroinvertebrate WCI
SR2	68.9	50.8	46.6
SR3	51.6	51.2	26.4
SR4	43.7	57.9	28.2
SR5	39.4	49.8	38.4
SR6	41.0	51.8	39.0
SR7	-	49.9	42.7
SR8	-	47.5	57.0
SR9	-	50.1	43.4
SU1	17.2	17.8	22.1
SU2	46.2	20.3	15.2
SU3	31.7	42.6	35.4
SU4	42.3	21.8	18.9
SU5	38.9	23.9	5.3
SU6	46.1	28.1	21.0
SU7	-	12.5	9.1
SU8	-	2.7	23.3
SU9	-	20.4	32.3
SU10	-	11.7	-

different assemblages were not significantly correlated (22%). The strongest correlation among metrics of different assemblages was between the diatom sensitive indicator genera and the macrophyte sensitive indicator species (Pearson's $r = 0.74$, $p < 0.01$). Twelve of the metric comparisons between the diatom and macrophyte assemblages were strongly significant ($|r| \geq 0.60$, $p < 0.01$). Only 2 of the comparisons between the diatom and macrophyte assemblages were not significantly correlated ($p < 0.10$).

Comparisons between the diatom and macroinvertebrate metrics were not as strong, with less than 50% of the metrics significantly correlated ($p < 0.10$). Only 6 diatom and macroinvertebrate metrics were correlated at the more stringent $p < 0.01$ level. The

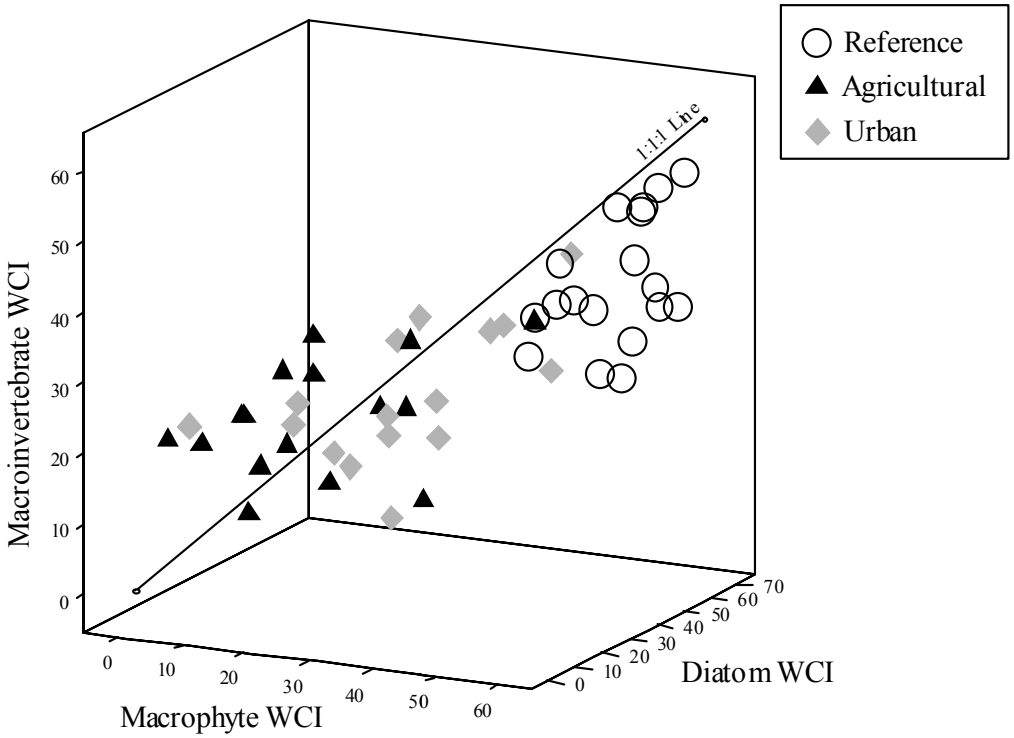


Figure 3-30. Three dimensional scatter plot of the WCI based on three assemblages, including diatoms, macrophytes, and macroinvertebrates.

macroinvertebrate metrics Noteridae and scrapers were not significantly correlated with any of the diatom metrics. Correlations were stronger among the macrophyte and macroinvertebrate comparisons, with 94% of the comparisons significantly correlated ($p < 0.10$). In fact, 20 of the metric comparisons (56%) were correlated at the strictest significance level of $p < 0.01$. Two metric comparisons between the macrophyte and macroinvertebrate metrics were not significantly correlated, including the macrophyte wetland status and macroinvertebrate Mollusca metrics and between the macrophyte exotic and macroinvertebrate scrapers metrics.

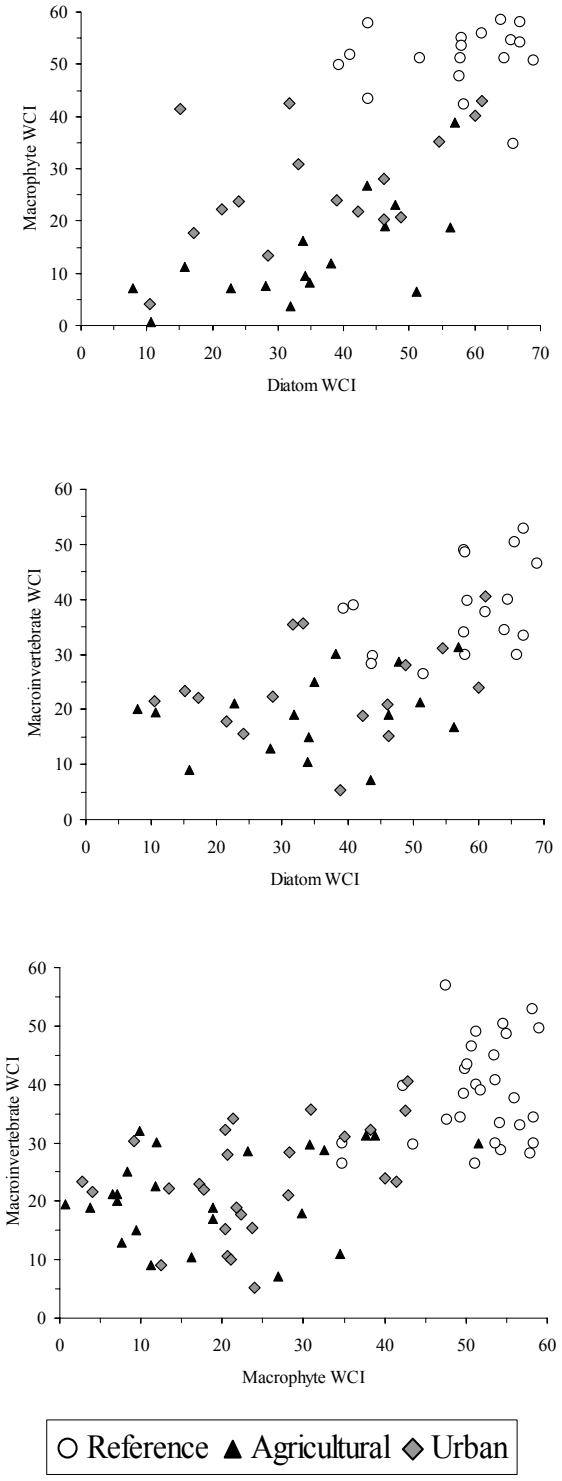


Figure 3-31. Scatterplots of WCI scores for wetlands based on diatom, macrophyte, and macroinvertebrate assemblages. A) Diatom and macrophyte WCI scores (n = 50 wetlands). B) Diatom and macroinvertebrate WCI scores (n = 50). C) Macrophyte and macroinvertebrate WCI scores (n = 79).

Table 3-40. Pearson correlations among 19 metrics.

	Macrophytes					
Diatoms	Tolerant	Sensitive	MFQI	Exotic	Native Perennial	Wetland Status
Tolerant	0.60*	-0.52*	-0.52*	0.49*	-0.60*	-0.26#
Sensitive	-0.59*	0.74*	0.68*	-0.53*	0.57*	0.42*
Pollution Class 1	0.60*	-0.50*	-0.51*	0.59*	-0.60*	-0.36^
Nitrogen Class 3	0.54*	-0.55*	-0.46*	0.61*	-0.58*	
Saprobity Class 4	0.51*	-0.47*	-0.44*	0.61*	-0.60*	-0.34^
pH Class 3	0.57*	-0.60*	-0.63*	0.62*	-0.58*	-0.39*
Oxygen Class 1	-0.57*	0.68*	0.56*	-0.57*	0.60*	

	Macroinvertebrates					
Diatoms	Tolerant	Sensitive	FL Index	Mollusca	Noteridae	Scrapers
Tolerant	0.49*	0.29^	-0.27#			
Sensitive	-0.48*	0.38*	0.40*	-0.33^		
Pollution Class 1	0.25#	-0.29^				
Nitrogen Class 3	0.33^	-0.29^	-0.30^			
Saprobity Class 4		-0.30^				
pH Class 3	0.50*	-0.30^	-0.25#			
Oxygen Class 1	-0.45*	0.32^	0.35^	-0.33^		

	Macroinvertebrates					
Macrophytes	Tolerant	Sensitive	FL Index	Mollusca	Noteridae	Scrapers
Tolerant	0.47*	-0.41*	-0.30*	0.27^	-0.25^	0.23^
Sensitive	-0.44*	0.54*	0.31*	-0.30*	0.31*	-0.27^
MFQI	-0.47*	0.34*	0.28^	-0.33*	0.21#	-0.30*
Exotic	0.48*	-0.32*	0.28^	0.27^	-0.23^	
Native Perennial	-0.45*	0.33*	0.29*	-0.30*	0.27^	-0.22#
Wetland Status	-0.35*	0.37*	0.28^		0.26^	-0.22#

* $p < 0.01$ ^ $p < 0.05$ # $p < 0.10$

CHAPTER 4
DISCUSSION

While previous research has identified responses of wetland ecosystems to individual changes (such as increased nutrients or altered hydrology), few have combined multiple biotic components, environmental parameters, and landscape development intensity in an attempt to quantify ecological integrity. The contribution of this research to our understanding of changes in the community composition of isolated forested wetlands (based on the diatom, macrophyte, and macroinvertebrate assemblages) in relation to different development intensities in the surrounding landscape can be summarized in 6 main points.

First, the richness, evenness, and diversity of each assemblage were not sensitive to different land uses or development intensities in the surrounding landscape. Second, biological indicators along with physical and chemical parameters were useful in defining biological integrity. Third, the variable turnover times and sensitivities of the 3 assemblages (diatoms, macrophytes, macroinvertebrates) suggest that a multi-metric multi-assemblage Wetland Condition Index (WCI) has more merit than a WCI based on a single assemblage. Fourth, regionalization may strengthen the WCI. Fifth, a WCI independent of wetland type may be feasible, given the strong likeness of the forested WCI to the marsh Index of Wetland Condition (IWC) (Lane 2003). Sixth, urban wetlands exhibit a different vector of change than do agricultural wetlands, and while the WCI suggests low biological integrity of both agricultural and urban wetlands, these wetlands do provide services and do work in the environment.

Richness, Evenness, and Diversity

Measures of richness, evenness, and diversity of the diatom, macrophyte, and macroinvertebrate assemblage were not sensitive to difference in land use or development intensity in the surrounding landscape. For both the diatom and macroinvertebrate assemblages neither *a priori* land use classification nor categories of landscape development intensity showed significant differences in richness, evenness, or diversity calculations. Differences in macrophyte evenness and diversity between reference and agricultural wetlands (Table 3-13; 3-14) may be attributable to both direct (for example grazing by domestic cattle) or more indirect (increased nutrients from fertilizer carried in run-off from surrounding agricultural fields) activities in the surrounding landscape. However, macrophyte evenness and diversity were higher for wetlands surrounded by more developed land uses, contrasting earlier findings on decreases in plant diversity from grazing pressures (Blanch and Brock 1994; Grace and Jutila 1999) and nutrient enrichment (Bedford et al. 1999). Mitsch and Gosselink (1993) report that freshwater forested wetlands have low species diversity, so perhaps macrophyte species that enter wetlands in developed landscape are merely taking advantage of available habitat and are in fact increasing the overall species diversity.

The increased incidence of exotic species have long been associated with disturbed ecosystems (Cronk and Fennessy 2001; Galatowitsch 1999b), suggesting more specifically that as anthropogenic development intensity increases, the incidence of exotic species may escalate. An increase in the frequency of exotic species has been attributed to drainage and hydrologic alterations (Hobbs and Heunneke 1992; David 1999; Galatowitsch et al. 1999b), increased human development (Cronk and Fennessy 2001), and ecosystem scale alterations such as clear-cut harvests (Devine 1998). Within

the study wetlands, the percent of exotic macrophyte species increased with increasing development intensity in the surrounding landscape (Figure 3-16). The influx of exotic species added to, rather than diminished, the species evenness and diversity within the isolated forested wetlands sampled.

Describing Biological Integrity

Biological indicators along with chemical and physical parameters were useful in determining the biological integrity of isolated forested wetlands. For the purposes of this study, biological integrity has been defined quantitatively with the WCI. The WCI incorporates 19 metrics from 3 different species assemblages (diatoms, macrophytes, and macroinvertebrates). Correlations between the diatom, macrophyte, and macroinvertebrate WCIs 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 changes in community composition were captured by the WCI. 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 WCI defined and detected deviations from the condition of reference wetlands based on community composition. Each of the 19 WCI metrics addressed some disparity from the assumed range of feasible conditions.

For all 3 assemblages, the tolerant indicator species metric demonstrated the strongest correlation with LDI (Tables 3-7; 3-18; 3-33), suggesting that the presence of a suite of taxa characteristic of wetlands with low 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.

Clustering the isolated forested wetlands based on the 3 assemblages separately suggested that differences in some agricultural and urban development intensities may be too subtle to detect with compositional data (Figures 3-10; 3-20; 3-29). Furthermore, greater variability in the macroinvertebrate assemblage of reference wetlands as compared to that of agricultural and urban wetlands (Table 3-31) suggested that perturbations to the driving energies in isolated wetlands may result in a convergence of the taxa present. Indeed, the natural compositional variability inherent among reference wetlands may be lost with increased development intensity in the surrounding landscape.

While the WCI can not be used to predict changes in the physical and chemical parameters of a wetland, its strength lies in providing an overview of biological integrity through the integration of changes in community composition from cumulative effects. Among *a priori* land use categories, differences in water and soil parameters were apparent (including dissolved oxygen, color, turbidity, water column pH, specific conductivity, water ammonia-nitrogen, water TKN, water TP, soil moisture, soil organic matter, and soil TP; Table 3-1). When soil and water parameters were used to explain variation in the community composition of each assemblage, water column pH was universally identified (Tables 3-6; 3-17; and 3-32). Additionally, total phosphorus concentrations explained some of the variance in both the diatom and macrophyte assemblages. Perhaps preservation and restoration strategies could focus on limiting

activities that influence changes to water column pH and total phosphorus loading to wetlands in order to promote biological integrity.

Merits of a Multi-Metric Multi-Assemblage WCI

The variable turnover times and sensitivities of the 3 different assemblages (diatoms, macrophytes, macroinvertebrates) suggest that a multi-metric multi-assemblage WCI has more merit than a WCI based on a single assemblage. Diatoms have short life cycles and live within the physical and chemical environment of the water column. As such, they act as integrators of ecosystem condition, with a rapid reaction time to environmental perturbation. Changes to the physical, chemical, and/or biological characteristics of a wetland influence the intricate interactions diatoms have with their environment. Macrophytes have longer life cycles than diatoms and macroinvertebrates, and as such they act as integrators of both present and historic changes in driving energies. The reaction time of the macrophyte assemblage to changes in driving energies is likely slower and more buffered than that of the other species assemblages. Unlike the diatom and macrophyte assemblages, macroinvertebrates may be able to abandon unsuitable habitats, and so their occurrence may reflect only the suitability of the recent wetland environment. Many macroinvertebrates have short life cycles, and many have multiple generations per year. Still others require over wintering in saturated soils or on wetland vegetation, suggesting an intimate relationship between macroinvertebrates and their individual environment.

The WCI can be used to infer influences in temporal and spatial changes to which a particular wetland has been exposed. For example, diatoms have rapid turnover times and may react immediately to shifts in driving energies. On the other hand, perennial macrophytes may respond to changes over a longer period of time, particularly in the

case of the woody mid- and over-story species. While the macrophyte WCI score may remain relatively high in a recently enriched wetland, the diatom WCI score may reflect lower biological integrity. While macrophytes assimilate nutrients for growth, this process has a longer time frame than the rapid growth rate typical of the algal assemblage. An explosion of algal growth may in turns alter the available food resources within a wetland affecting other assemblages, for example there may be an increase in macroinvertebrate algae scrapers, and decline in the macroinvertebrate WCI score.

While agreement in the ranking of the biological condition of study wetlands using the WCI was anticipated, discrepancies among the ranking from the different assemblages may provide greater 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 WCI; though there were no obvious outliers when the three assemblages were compared (Figure 3-30). While the *a priori* reference wetlands were generally differentiated from the agricultural and urban wetlands, differences between the agricultural and urban land uses were not as apparent (Figure 3-31).

Many of the metrics of the different assemblages were significantly correlated (Table 3-40), none were correlated at the threshold ($|r| > 0.90$) used to exclude candidate metrics from inclusion in the WCI. Perhaps the value of each of the 19 selected metrics is inherent in its differentiation between categories of landscape development intensity (Tables 3-8, 3-19, and 3-34). Diatom and macrophyte metrics were strongly correlated with one another, and yet diatom and macroinvertebrate metrics were not, reinforcing the value of including various species assemblages in an assessment of biological integrity.

Perhaps, with regular biological monitoring it may be possible to further explore the temporal effects of changing development intensity.

A Case for Regionalization

The climate of Florida is considered humid subtropical, though pronounced differences occur in the local climate across Florida, such as differences in the amount of annual rainfall, seasonal maximum temperatures, and number of freeze days (Fernald and Purdum 1992; Lane 2000). The latitudinal range of the wetlands sampled in this study was 31.0ºN at PA4 in Escambia County the western most county in the Florida panhandle, to 26.0ºN at SR4 in Collier County in southwest Florida. The longitudinal range was from 87.5ºW at PA4 to 80.1ºW at SU8 in Palm Beach County, along the southeastern coast of Florida. Despite the broad latitudinal and longitudinal ranges of sample wetlands throughout Florida, statewide significant difference in water and soil parameters among *a priori* land use categories were detected (Table 3-1), suggesting the statewide scale may be appropriate for a physical and chemical assessment of wetland condition.

The influence of latitude and longitude was reflected in the compositional difference of all 3 assemblages found among the Florida ecoregions (Table 3-5, 3-15, and 3-31). Latitude and longitude were significantly correlated with macrophyte community composition (Figure 3-11), and latitude explained partial variance in macroinvertebrate community composition (Figure 3-21). In addition, wetlands in the Florida Everglades were outliers in many of the diatom metrics (Figures 3-3; 3-5; 3-7; 3-8), and the southern Everglades wetlands formed distinguished clusters based on diatom (Figure 3-10) and macroinvertebrate (Figure 3-29) community composition.

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, they may be more affected by development in the surrounding landscape (such as compounded secondary effects) than their panhandle and far north ecoregion counterparts.

Regionalization was explored for the macrophyte assemblage because of the sufficient number of wetlands sampled within each ecoregion. There were clear regional differences in the statewide macrophyte WCI scores for wetlands in the low LDI group, which down scored the reference wetlands of the south and central ecoregion (Table 3-25). This led to the use of regionalized scoring of the macrophyte metrics. While the ease and utility of a single statewide WCI would seemingly prevail over 4 regional indices, 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.

Differences in the macroinvertebrate community composition among ecoregions may be of importance in improving the macroinvertebrate WCI. For example, none of the sample wetlands in the panhandle ecoregion hosted macroinvertebrates in the order Trichoptera (caddis flies), whereas no wetland in the north ecoregion hosted macroinvertebrates in the order Ephemeroptera (mayflies). While both of these orders

are characteristic of lotic environments (Edmunds and Waltz 1996), the absence of an entire order from ecoregions suggests the value of regionalization of the WCI.

WCI Independent of Wetland Type

Recent work by Lane (2003) presents a 24 metric Index of Wetland Condition (IWC) for Florida marshes based on the community composition of the diatom, macrophyte, and macroinvertebrate assemblages. The IWC was created based on 75 isolated depressional freshwater marshes surrounded by undeveloped (n=34) and agricultural (n=40) land uses throughout peninsular Florida. Of the 14 metrics based on the diatom assemblage, the forested WCI and the marsh IWC share 7 metrics, including all of the diatom WCI metrics. Two of these 7 shared metrics were based on tolerant and sensitive indicator species analyses, which were determined separately for each wetland type. Shared species were limited between wetland types, as the tolerant indicator species list had only 2 mutual species (*Navicula confervacea* and *N. minima*), of 12 species for the forested WCI and 21 species for the marsh IWC. Similarly, the sensitive indicator species lists shared only 5 species (*Eunotia flexuosa, E. naegelii, E. rhomboidea, Frustulia rhomboids*, and *F. rhomboids crassinervia*), of 18 species for the forested WCI and 22 species for the marsh IWC. The 5 remaining diatom metrics were based on autecological relationships, including pollution class 1 (Bahls 1993), nitrogen class 3 (van Dam et al. 1994), saprobity class 3 (van Dan et al. 1994), pH class 3 (van Dan et al. 1994), and dissolved oxygen class 1 (van Dan et al. 1994). Of the remaining 7 marsh IWC diatom metrics, 5 were considered too similar to selected forested WCI metrics and were excluded to avoid redundancy. The final 2 marsh IWC diatom metrics were based on salinity class (van Dam et al. 1994), and were not significantly correlated with LDI for forested wetlands.

Five macrophyte metrics were incorporated into the marsh IWC, and variants of these were included in the metrics of the forested WCI. Tolerant and sensitive indicator species lists were constructed separately for each wetland type. Of the 46 statewide tolerant macrophyte indicator species for the marsh IWC, 28 also occur on the statewide tolerant macrophyte indicator species list for the forested WCI (of 61 species). Similarly, 20 statewide sensitive macrophyte indicator species were shared for the marsh IWC (of 36) and the forested WCI (of 69). The 3 additional metrics included in the marsh IWC were percent exotic species, annual to perennial ratio, and a metric based on scores from a Floristic Quality Assessment Index (similar to the one conducted in this study, but specific to marshes). In the forested WCI, a variant of the annual to perennial ratio was used, the percent native perennial species (to account for anticipated conditions at urban wetlands). The sixth forested WCI metric was the percent wetland status species.

There was less similarity between the 5 macroinvertebrate marsh IWC metrics and the 6 forested macroinvertebrate WCI metrics. Tolerant and sensitive indicator metrics were constructed separately for each wetland type and were included in both indices. Three tolerant indicator genera occurred on both lists (*Goeldichironomus, Micromenetus,* and *Physella*), and only 1 sensitive indicator genera was shared (*Larisa*). The marsh IWC included 3 additional metrics: %Predators, %Odonata, and %Orthocladiinae. The forested WCI included 4 different metrics: Florida Index, %Mollusca, %Noteridae, and %Scrapers.

Overall, the marsh IWC and the forested WCI were similar, with many shared metrics. Some additional variability between selected metrics was expected as the forested WCI included 2 additional sources of variability (wetlands in the panhandle

ecoregion and urban land uses, which were not included in the sample wetlands for the marsh IWC). Perhaps the strong similarity of metrics suggests that a universal assessment index could be constructed regardless of wetland type. However, it would likely be necessary to maintain independent indicator species lists specific to wetland type.

Wetland Value

Urban wetlands appear to exhibit a different vector of change than do agricultural wetlands; however the WCI did not significantly differentiate between agricultural and urban wetlands (Figures 3-30; 3-31). The LDI also did not specifically differentiate between these land use categories, as Landscape Development Coefficients (LDC) for agricultural land uses range from 2.6 (unimproved pasture) to 6.6 (high intensity agriculture) (Table 2-3). Urban LDCs overlap that range with variants of Open Space/Recreational land uses ranging from 2.1 (low intensity) to 4.8 (middle intensity) to 6.9 (high intensity). Similarly, other measures of anthropogenic influence like the Wetland Rapid Assessment Procedure and the Minnesota disturbance index (Appendix B) did not clearly differentiate between agricultural and urban land uses.

The main conclusion we can draw from the WCI is that both agricultural and urban wetlands have lowered biological integrity. 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 perhaps 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 in highly developed landscape matrices. While wetlands do exist in highly urbanized areas, the do not appear to closely resemble wetlands in undeveloped landscapes.

Under current Florida law, mitigation ratios for urban wetlands will be small, and some people may question the idea of keeping urban wetlands of marginal 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 restoring a wetland with the possibility of successfully meeting mitigation criteria. However, off-site 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. The continued existence of urban wetlands is crucial (for the maintenance of biotic diversity, buffering pollution and contamination to protect nearby environments, increasing oxygen production in an urban center, etc.). While the WCI 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 and therefore no services or free work. Wetlands with the lowest biological integrity could have scored a 0 for the WCI, and yet only 1 agricultural wetland did (NA1, for the macrophyte assemblage). There is no doubt that the intensity of human land use across the landscape plays a role in the loss of biological integrity of wetlands, however we should reconsider our willingness to remove all of the biological integrity of a wetland by otherwise erasing its existence by filling.

Limitations and Further Research

Several limitations to this study should be noted, including sampling methods and drought conditions. One water sample was collected to represent the water environment of the entire sample wetland, and water samples were taken at a range of times throughout the day. While water samples were always taken first when a crew arrived at a sample wetland, the time of day the crew arrived fluctuated. Additionally, while an attempt was made to avoid taking water samples immediately following extreme rain event, there is the possibility that the sample was taken during a period of time without rain. Therefore, there was no consistency as to recent weather conditions when water samples were taken. There were also strict requirements of preservation, temperature control, and shipping protocols associated with the water samples. When these requirements were breached the sample had to be discarded.

Similarly, one composite soil sample was taken for each sample wetland, and bulk density was not measured, which complicates the use of soil nutrient data. As well, generally wetlands were visited only once, with a complete sample effort lasting just one day. This provided a mere snapshot of wetland condition. Revisits were conducted at some wetlands to collect water, soil, algae, or macroinvertebrates in the case of dry conditions on the initial visit or a discarded sample (generally for quality control reasons). Visiting these wetlands only once or twice did not allow insight into seasonal or yearly variations in the assemblages. As an additional confounding factor, Florida experienced drought conditions in 2001, and the macrophyte assemblage at many wetlands was sampled without standing water, which allowed many flood intolerant species to encroach into the sample wetlands.

While the WCI has satisfactorily distinguished between wetlands embedded in an array of land uses with varying development intensities, much needs to be done to insure accuracy and usability. First, seasonal and yearly variation should be identified for the study wetlands. Wetlands are pulsing systems, and as such wetland organisms must adapt to wide fluctuations in hydrology, temperature, salinity, and dissolved oxygen (Evans et al. 1999; Leslie et al. 1999; Sharitz and Batzer 1999). A new set of wetlands should be sampled and scored based on the WCI to test the reliability of this index. The WCI was limited to nineteen metrics due to the redundant nature of many of the candidate metrics, as well as the high variability of species composition within the dataset. A larger sample size could improve the significance of the WCI based on ecoregions for metrics such as indicator species analysis. Regionalization may be an important step in refining the WCI, as this study was somewhat limited to a statewide approach due to small sample sizes within each ecoregion.

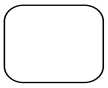
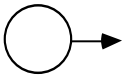
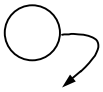
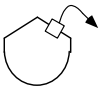
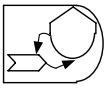
Conclusions

The use of 3 separate species assemblages for a biological assessment of isolated forested wetland provided a useful tool for detecting changes in biological integrity associated with changes in the driving energies of a wetland measured through landscape development intensity. While richness, evenness, and diversity measures were not particularly sensitive to changes in landscape development intensity, biological indicators along with physical and chemical parameters were useful in defining biological integrity. In the future a multi-metric multi-assemblage WCI could be constructed for all freshwater wetlands throughout the state of Florida, with specific indicator species and metric scores based on Florida ecoregions. While the WCI suggests 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 WCI should not be used as a surrogate for wetland value, but an objective, quantitative means of comparing changes in community composition along gradients of development intensity.

APPENDIX A
ENERGY CIRCUIT LANGUAGE

Table A-1. Symbols used in energy circuit diagramming.

Symbol	Name	Description
	System Boundary	Defines the system being diagrammed. Lines that cross the system boundary indicate inflows and outflows of the system.
	Energy Circuit	A pathway which has a flow proportional to the quantity in the storage or source upstream.
	Source	Outside source of energy delivering forces according to a program controlled from outside; a forcing function.
	Flow Limited Source	Outside source of energy with a flow that is externally controlled.
	Storage Tank	A compartment of energy storage within the system storing a quantity as the balance of inflows and outflows.
	Sensor	The sensor (tiny square box on storage) suggests the storage tank controls some other flow but does not supply the main energy for it.
	Producer	Unit that collects and transforms low-quality energy under control interactions of high-quality flows.
	Consumer	Unit that transforms energy quality, stores it, and feeds it back autocatalytically to improve inflow.
	Heat Sink	Dispersion of potential energy into heat that accompanies all real transformation processes and storages; loss of potential energy from further use by the system.

APPENDIX B
QUANTIFYING ANTHROPOGENIC INFLUENCE

Table B-1. LDI, WRAP, and Minnesota disturbance index scores for 118 wetlands.

Site Code	LDI	WRAP	Minnesota Disturbance Index	Site Code	LDI	WRAP	Minnesota Disturbance Index
SA1	6.3	6.9	16	CA1	6.1	5.8	16
SA2	4.6	4.6	14	CA2	5.0	6.3	9
SA3	4.6	5.5	9	CA3	4.9	8.1	14
SA4	6.2	7.5	17	CA4	4.3	3.4	5
SA5	4.0	4.5	11	CA5	5.1	5.5	13
SA6	5.0	6.0	9	CA6	5.4	4.8	17
SA7	2.8	2.6	2	CA7	4.4	5.7	8
SA8	1.3	3.6	5	CA8	1.9	2.7	9
SA9	4.8	3.2	6	CA9	5.4	6.3	17
SR1	1.0	1.0	0	CR1	1.0	1.1	5
SR2	1.0	1.2	0	CR2	1.0	1.8	1
SR3	1.0	1.4	3	CR3	1.0	1.5	2
SR4	1.0	1.0	1	CR4	1.1	1.4	0
SR5	1.2	1.0	1	CR5	1.0	1.0	0
SR6	1.0	1.5	2	CR6	1.0	1.3	0
SR7	1.1	1.8	3	CR7	1.1	1.8	6
SR8	1.0	1.7	2	CR8	1.0	3.0	2
SR9	1.0	3.2	3	CR9	1.0	1.3	2
SU1	5.0	6.2	14	CR10	1.5	1.8	5
SU2	4.8	6.0	8	CR11	1.0	1.5	0
SU3	1.3	5.9	15	CU1	2.1	2.4	7
SU4	1.5	4.9	8	CU2	3.9	6.0	7
SU5	5.2	6.3	17	CU3	7.1	6.4	17
SU6	3.9	5.5	15	CU4	4.5	6.0	13
SU7	5.1	7.1	13	CU5	6.4	6.1	12
SU8	7.2	8.4	17	CU6	7.2	6.0	13
SU9	6.2	4.6	16	CU7	5.6	6.4	16
SU10	3.2	6.0	8	CU8	4.4	5.1	16
				CU9	7.0	7.2	14
				CU10	3.8	5.8	10
				CU11	3.3	6.2	15

Table B-1 Continued.

Site Code	LDI	WRAP	Minnesota Disturbance Index	Site Code	LDI	WRAP	Minnesota Disturbance Index
NA1	5.1	6.9	13	PA1	3.1	4.7	9
NA2	4.9	6.3	9	PA2	5.0	6.2	18
NA3	2.1	2.9	4	PA3	6.8	6.1	19
NA4	6.2	6.8	19	PA4	6.6	6.8	18
NA5	5.1	7.5	17	PA5	4.7	6.7	19
NA6	5.3	5.5	16	PA6	4.7	6.0	17
NA7	5.0	4.3	7	PA7	4.9	6.6	17
NA8	2.0	3.1	4	PA8	2.2	4.9	10
NA9	2.2	4.2	5	PA9	5.8	5.4	7
NA10	2.2	3.0	6	PA10	2.0	4.9	6
NA11	2.5	3.9	9	PR1	1.0	1.6	0
NA12	5.5	5.1	7	PR2	1.0	1.3	0
NR1	1.1	2.1	1	PR3	1.1	1.9	0
NR2	1.0	1.5	1	PR4	1.5	1.7	2
NR3	1.0	1.2	1	PR5	1.0	1.0	0
NR4	1.0	1.2	2	PR6	1.3	1.3	0
NR5	1.0	1.4	1	PR7	1.0	1.3	0
NR6	1.1	1.1	0	PR8	1.0	1.2	0
NR7	1.8	1.8	1	PU1	5.3	8.3	17
NR8	1.0	2.5	0	PU2	5.9	5.3	9
NR9	1.0	1.9	1	PU3	6.3	5.6	12
NU1	2.8	3.9	7	PU4	4.8	7.5	17
NU2	4.2	4.4	9	PU5	4.0	6.4	13
NU3	5.3	6.6	10	PU6	4.8	5.1	8
NU4	3.2	5.4	11	PU7	3.8	5.4	10
NU5	6.2	6.2	9	PU8	5.0	5.4	9
NU6	5.6	4.8	15	PU9	3.1	3.9	5
NU7	4.2	5.3	10	PU10	6.5	8.6	17
NU8	3.8	4.8	11				
NU9	6.3	5.9	12				
NU10	6.6	6.4	17				

APPENDIX C
STANDARD OPERATING PROCEDURES

Standard Operating Procedures (SOPs) have been included for sampling methods employed for the entire project, which included more data collection than that included in this dissertation. These additional methods were included in Appendix C to provide readers with a complete picture of field methodology, and to understand the order of events during field sampling. Data omitted from this dissertation include tree basal area along transects, fisheye canopy photography, algae analysis of epiphyton, metaphyton, and phytoplankton, and dry benthic algae sampling. Canopy photo analysis was explored in an earlier thesis by Spurrier (2000). Laboratory identification of additional algae samples was not completed due to the enormous expense associated with enumeration and identification of each sample.

While vegetation zone descriptions are provided for soil sampling, these procedures were initially created for use in freshwater isolated marshes (Lane 2003) . Zonation for soil samples was only employed at 3 of 118 sample wetlands that were characterized by open centers (where no canopy trees occurred in the deep pooled center area of these 3 wetlands). As such, soil samples were taken in both the outer forested zone and the inner marsh zone and analyzed separately. As suggested in the soil SOPs, the soil data values were weighted based on the area occupied by each vegetation zone.

SOPs for Isolated Forested Wetlands

- Water Quality
- Site/Habitat Characterization
- Vegetation
  - Herbaceous – 1 x 5 m quadrats along transect
  - Trees – variable area plots every 10 meters along transect
- Algae
  - Wet Sites Only
  - Dry Sites – benthic algae only
- Macroinvertebrates – wet sites only
- Soils

ORDER OF FIELD EVENTS FOR ISOLATED CYPRESS DOMES:

1. Water quality is ALWAYS taken first. Two field crew members take the water samples; one records the data while the other takes the sample. The data are recorded on BOTH the FDEP lab submittal form and on the wetland characterization form.
2. While two crewmembers are collecting water, the other(s) unloads the vehicle and prepares the field equipment.
3. After the water samples are obtained (follow SOP for water quality), complete the Site/Habitat Characterization Data Sheet & WRAP assessment.
4. When completed, start the vegetation transects. This includes delineating the wetland and running all four transects (follow SOP for vegetation).
5. The remaining field crew should:
6. Collect algae samples (follow SOP for algae)
7. Collect macroinvertebrates (follow SOP for macroinvertebrates)
8. Collect soil samples (follow SOP for soil)
9. Take site photographs
10. Establish stakes for canopy photos

CHECKLIST OF MATERIALS/FIELD EQUIPMENT:

- Miscellaneous
  - SOPs
  - Large cooler with frozen ice bottles for soils and vegetation
  - Camera
  - 3.5” floppy disks
  - Waders
  - Garmin III - GPS unit
  - Florida Gazetteer
  - Machete
  - Aerial photo & FLUCCS codes of site
- Water Quality
  - Small cooler with ice
  - YSI meter (DO/temp)
  - 2 500-mL bottles – turbidity/color/conductivity/pH & NH_3/NO_X/TKN/TP
  - Pipette for H_2SO_4

- Bottle of 1:1 H_2SO_4
- Clear tape
- FedEx air bills
- Zip-loc bags
- FDEP Central Lab submittal form

- Vegetation Transects
  - 2 100m transect tapes
  - 1 m PVC
  - 2-3 compasses
  - 2 pieces of 1.5 m rebar
  - Clipboards
  - Field data sheets – a minimum of 8 per site
  - Site Characterization & WRAP sheets – 1 per person per site
  - Pencils & sharpie
  - Bag for unknown plants
  - Plant press, newspaper, and cardboard
  - Masking tape
  - Field ID manuals
  - Aerial photos
  - Prism for basal area
  - Hand lens
  - Index cards
- Macroinvertebrates
  - US Std 30 mesh sweep net
  - Large 1 gallon jar for sample
  - Bottle of formalin for preserving sample
- Algae
  - Collecting jars – 3 100-mL pea cups & 1 1-L sample bottle
  - Collection jar with bottom missing for benthic algae
  - Large pipette – aka turkey baste
  - Knife
  - Zip-lock freezer bags
  - Masking tape
  - Sharpie – black permanent maker
  - Falcon phytoplankton sampler – aka 50-mL centrifuge tube
  - Bottle brush & scraper
  - M3 preservative
  - Pipette for M3 preservative
  - 1-L deionized water for dry sites
- Canopy Photos
  - Digital camera
  - Spare batteries
  - Film disks
  - Tripod
  - Compass
  - Height pole

- Soils
  - 3-inch diameter PVC pipe
  - Knife
  - Piece of wood
  - Dampened hammer
  - Buckets
  - Small & medium sized freezer Zip-loc bags
  - Stainless steel spoon
  - Permanent marker
  - De-ionized water

SOPs for Forested Wetlands: WATER QUALITY

1. The YSI meter must be on and calibrated for 15 minutes before using.
2. *Always take this collection first*!!
3. Water samples can ONLY be collected Monday, Tuesday, Wednesday, or Thursday. Samples are sent to FDEP Central Lab overnight.
4. One 500 mL bottle for turbidity/color/conductivity/pH and one 500 mL bottle for NH_3/NO_x/TKN/TP are collected per site. These have labels provided by FDEP.
5. Only take water samples if the water depth is greater than 10 cm.
6. Carefully enter the water without stirring up organic material and silt.
7. Remove cap from each 500 mL bottle without touching the lip or interior surfaces.
8. Rinse the bottle three times in wetland water, dumping the water away from collection area.
9. Place the sample bottle upside down in the standing wetland water.
10. Carefully tilt back end into the water and press on bottom of bottle to allow water to slowly flow inside.
11. Be deliberate, making sure that no suspended organic matter enters the sample bottle. If organic materials do enter the sample, dump the sample and begin again.
12. When all exiting air bubbles have stopped, carefully lift bottle out of water.
13. Repeat, so both 500 mL bottles are full.
14. After the water is collected, take dissolved oxygen and temperature readings using the YSI meter. Take measurements within the top 10 cm of the water column. Apply constant, gentle motion to the dissolved oxygen probe, as the meter is consuming oxygen during measurement. Measure water depth.
15. Preserve only the bottle for NH_3/NO_x/TKN/TP, using 2 mL of 1:1 H_2SO_4 per 500 mL sample.
16. Place both sample bottles on ice in a six-pack cooler. For transport reasons, the ice should be in a bag atop the samples.
17. Fill out the FDEP Central Lab Sample Submittal Form. Place in a zip-lock bag in cooler.
18. If 2 sites are sampled in 1 day, place all 4 sample bottles in the cooler along with the forms (one form is sufficient if properly filled out).
19. Tape the cooler shut and make sure the air bill is filled out properly. Call 1-800-GO-FEDEX to find a nearby office.

20. If you cannot get to FedEx in time, dump the samples taken. Repeat procedure another day.

SOPs for Forested Wetlands: VEGETATION

1. Using a compass, locate the 4 cardinal point directions (north, south, east, and west). The 4 transects will begin at each cardinal point running from the edge of the wetland into the interior/middle of the wetland. These 4 transects will intersect in the middle and divide the wetland into 4 approximately equal sections.
2. At the beginning of each transect, delineate the edge of the wetland using a combination of wetland plants and hydric soils. Be conservative on the side of the wetland.
3. Establish the transect using the meter tapes. Start with 0 meters at the wetland edge, and increase distance towards the wetland interior.
4. Use a separate field data sheet for each cardinal direction. If the number of species located on a transect exceeds the number of columns on the data sheet, start a new data sheet.
5. Creating quadrats that are 0.5 m on either side of the transect (1 m wide) and 5 m long, record all species present within these elongated quadrats.
6. Plant species names are recorded on the data sheets using the full genus and species names. Each unknown species is given a unique ID code using the transect location (ex. N-1).
7. Voucher specimens for all unknown species are collected, being sure to get plant inflorescence and roots, tagged with properly labeled masking tape, and put into a labeled collection bag. Note the color of the inflorescence on the label, as the flowers often do not preserve well. Index cards can be used to protect especially sensitive parts.
8. All collected plants are identified in the field on the day of sampling and placed in a plant press for further clarification and identification. Plant nomenclature follows FDEP's *Florida Wetland Plant Identification Manual* (Tobe et al. 1998).
9. At each 10 m along each transect, starting at 10 m, 20 m, 30 m, etc., tree basal area will be recorded. Use the data sheet for basal area, and record basal area per species using variable area plots and a 10 factor prism. Hold the prism at eye level, with a bent elbow and count the number of trees per species that fall within the variable area plot. The prism shall be centered over the sampling point at all times, with the field person rotating around the prism so that the entire circular area (360^{o}) around the point of sampling is included.
10. As the sun lowers on the horizon, take canopy photos at 1 point along each transect. Placement of tripod will be 10 paces out from the center of the wetland along each transect. In those instances when the cypress dome is a "hole in the doughnut" and there are no cypress trees in the center of the dome, tripod placement will be 10 paces along the transect out from the ring of trees. Follow directions according to *A Manual for the Analysis of Hemispherical Photography* (Rich 1989).
11. At each photograph spot, insert a wooden stake so that photo sites can be revisited in the future.

12. Center the tripod over the stake with the top of the lens cap at the height of the provided height pole [which is at breast height 1.3 m]. The top of the camera should face south, so that the photographer's back is to the north.
13. Level the tripod, so that the bubble on top of the lens cap is centered within the circle.
14. Turn the camera on to automatic. Set the camera to XGA Fine, using the dial at the front right and the button on the back of the camera.
15. Zoom out the camera all of the way so that the back display shows the canopy as a circle, surrounded by a dark/black border.
16. Record the photo number for the position (ex. north, south, east, or west).
17. Complete the canopy photo data sheet, noting time of day, cloud condition, surrounding vegetation, etc.

SOPs for Forested Wetlands: ALGAE

AT WET SITES:

- Separate samples by substrates of the site you are working on (i.e. epiphyton, benthic algae, metaphyton, and phytoplankton).
- For each substrate, collect 10 aliquots, and keep each substrate type separate in their own collection jars. At the end of the collection there should be between 100-120 mL of wetland algae-water mix in the cups, except for phytoplankton which should have approximately 1000 mL.
- Rinse all sample equipment in wetland water prior to sampling.

- EPIPHYTON – divide appropriately among herbaceous and woody debris based of the proportion of the area of wetland of each:
    1. For herbaceous vegetation:
       Cut plant stems under water and place in zip-lock bag with wetland water; shake and knead vigorously in zip-loc bag; use turkey baste to extract 10 mL of algal suspension and place in labeled pea cup; distribute the aliquots appropriately throughout the different vegetation/habitat zones.
    2. For woody debris (roots, snags):
       Using a brush, brush the wood for algae. Place brush in bag with water and shake algae off of the brush. Pipette the algae into the collection jar;
       -or-
       Use bottomless pea cup to isolate a spot on the debris. Use turkey baste to stir algae from surface of debris; extract 10 mL of algal suspension and place in pea cup; since woody and herbaceous are within the same 10 aliquots, they must be divided appropriately between the two.

- BENTHIC ALGAE:
    1. Use bottomless pea cup to isolate a spot on the sediment;
    2. Use turkey baste to gently stir algae from the surface of the sediment;
    3. Extract 10 mL of algal suspension and place in pea cup.

- METAPHYTON:
    1. Collect approximately 100 mL of wetland water in a pea cup;

2. Using your fingers, collect a thumbnail size portion of the algal mat;
3. Obtain aliquots from 10 different areas of the wetland.

PHYTOPLANKTON (1 L is collected):

1. In total use the 50 mL centrifuge tube (x2) to collect ten 100 mL aliquots;
2. Divide aliquots proportionately between the major vegetation zones;
3. Rinse tube with wetland water. With the cap on the tube, lower into the water column and then remove the cap to allow the tube to fill with water;
4. Cap the tube under the water and then bring tube out of water;
5. Carefully pour contents into the dark algae collection bottle;
6. Since the tube is only 50 mL, you will need to do this twice in each of the 10 aliquots.

- Preserve the samples.
  - Add 5 mL of M^3 per 100 mL of algal suspension in pea cups.
  - Add 20 mL of M^3 per 1 L of algal suspension in column algae bottle.
- Properly label collection jars, identifying site, date, collector, and sample type.
- Carefully clean equipment with deionized water to avoid cross-contamination at future sites.
- When at the Center for Wetlands, clean all equipment with Clorox/water solution.
- Return full collection jars to room 8 at the Center for Wetlands to await laboratory analysis.

AT DRY SITES ONLY BENTHIC ALGAE IS TO BE COLLECTED:

- Use bottomless pea cup to isolate a spot on the sediment.
- Extract the upper 0.5 cm of soil into an additional pea cup, this depth is marked on the collection pea cup.
- Add 100 mL deionized water and stir well with turkey baste.
- Extract 10 mL of algal suspension and place in sample pea cup.
- Repeat, so that you have collected 10 aliquots representative of the vegetation/habitat zones in the wetland.
- Preserve the sample with 5 mL of M^3 per 100 mL of algal suspension.
- Properly label collection jar, identifying site, date, collector, and sample type.
- Carefully clean equipment with DI water to avoid cross-contamination at future sites. When at the Center for Wetlands, clean all equipment with Clorox/water solution.
- Return full collection jars to room 8 at the Center for Wetlands.

DIRECTIONS FOR M^3 FIXATIVE PREPARATION:

<u>Materials</u>

10 g Iodine
5 g Potassium iodide (KI)
50 mL Glacial acetic acid
250 mL Formalin (37% W/W formaldehyde)
1 L Deionized water

Methods
1. Dissolve 10 g iodine in a small quantity of deionized water to aid in solution of iodine.
2. Dissolve 5 g potassium iodide, 10 g dissolved iodine (from step 1), 50 mL glacial acetic acid and 250 mL formalin (37% W/W formaldehyde) in 1 L deionized water.
3. Store in the dark.
(p.10-8, Standard Methods for the Examination of Water and Wastewater, 17th edition)

SOPs for Forested Wetlands: MACROINVERTEBRATES

- There are to be 20 sweeps (evenly divided into the vegetation/habitat zones) to send to FDEP unpicked for identification.
- Always do your sweeps in undisturbed areas where you have not walked through yet.
- A single sweep is one net width and two net lengths to equal 0.5 m^2.
- Using a U.S. Standard 30 mesh net, sweep from the bottom of the substrate up the plant stalks. Use your hands to strip the plant of all material into the net. If you are in a forested site, use a brush to clean any snags and roots of material.
- Vigorously sample the area repeatedly (3 times) to ensure good coverage.
- Dip net into water repeatedly, without letting the sample out, to try and sift the muck and silt through the net.
- Do not sample in the muck!
- Place the contents of each sweep into the 3.8 L jar. When all 20 samples are complete, preserve the sample by adding Formalin at a rate of 10% of the sample volume. Seal the jar. Shake to ensure thorough mixing.
- Place masking tape over the lid to prevent leakage during travel & shipment. Properly label the jar with the site name, date, and collector.
- Thoroughly clean all equipment off with water.
- Return samples to room 8 at the Center for Wetlands for later shipment to the FDEP.

DIRECTIONS FOR BUFFERED FORMALIN PREPARATION:
Materials
Sodium bicarbonate (sodium borate may also be used)
Formalin (37% W/W formaldehyde)
pH meter

Methods
Note that formalin in the common name for 37% W/W formaldehyde. "Formalin" and "Buffered Formalin" are 2 separate things in this recipe.
1. Calibrate the pH meter (directions follow). The pH electrode and temperature probe should rest in a beaker of deionized water between measurements. Rinse the electrode and probe off with a spray bottle of deionized water before submerging in other solutions.
2. Select a container for preparing the buffered formalin. Usually this is simply the plastic container that the formalin was shipped in.
3. Fill the container with formalin to just below (1-2 cm) where the top of the desired buffered formalin solution level.
4. Scoop a small amount of sodium bicarbonate into the container, close, and shake vigorously (at least 1 minute to ensure proper mixing). All of the sodium bicarbonate may not dissolve into the formalin, as this is a supersaturated solution.

5. Measure the pH of the resulting solution.
6. Repeat steps 4 and 5 until the pH is at least 7.5 but not higher than 8.0.
7. If desired, transfer the buffered formalin to smaller containers for field use.
8. Make sure that all containers are clearly labeled "Buffered Formalin pH 7.5-8.0."
9. When disposing of used buffered formalin, deposit it in an appropriately labeled waste container. The waste container should have a yellow Hazardous Materials sticker, call the Environmental Health and Safety department at 392-1591.

Original recipe from the Florida Department of Environmental Protection Biology Section, Standard Operating Procedure (SOP) #IZ-10, "Preparation of Buffered Formalin." Expanded upon by Melissa Yonteck on May 2, 2001.

CALIBRATION OF pH METER (Hanna Instruments HI 9025C)

Materials

pH meter
pH 4.00 buffer solution
pH 7.00 buffer solution
3 beakers
deionized water bottle

Methods

*When not being used, rest the electrode and probe in a beaker of deionized water. Rinse the electrode and probe off with a spray bottle of deionized water before submerging in other solutions.

1. Pour small quantities of the pH 4.00 and pH 7.00 buffer solutions into each of 2 clean beakers.
2. Immerse the pH electrode and temperature probe into the pH 4.0 buffer solution, stir briefly. The electrode and probe should be close together, and they should be submerged approximately 4 cm (1½ inch) into the solution.
3. Press the CAL key. The "CAL" and buffer indicators will be displayed. The secondary LCD display should read "4.01." If not, adjust it using the "□□c" key.
4. After the pH reading becomes stable, the "READY" and "CON" indicators will blink. Once this happens, press the CON key to confirm the calibration.
5. After rinsing with deionized water, immerse the pH electrode and temperature probe into the pH 7.00 solution, and stir briefly.
6. Select the second buffer value ("7.01") on the secondary display using the "□□c" key.
7. After the pH reading becomes stable, the "READY" and "CON" indicators will blink. Once this happens, press the CON key to confirm the calibration.
8. Press "CAL" key to end calibration process and begin measuring.
9. When finished using the pH meter, pat the electrode and probe dry with a KimWipe. Place the pH electrode in the yellow/orange cap with a small amount of deionized water.

The pH meter should be recalibrates:

- whenever the pH electrode or temperature probe is replaced
- at least once a month
- after testing aggressive chemicals

- if greatest accuracy is required
whenever the batteries have been replaced.

SOPs for Forested Wetlands: SOIL

1. The wetland will be visually divided into vegetation zones. Cores will be taken within each zone and combined, so that each vegetation zone has one cumulative soil sample. The number of cores taken per zone is generally 4, 1 along each transect. In some cases there will be fewer than 4 cores per vegetation zone as the zones may not all fall along the established transects.
2. To sample soils:
   a. Clean off the detritus from the site that will be sampled. This means removal of plant material that appears less than 6 months old, or the recognizable fallen plant material.
   b. Place the 7.9 cm diameter PVC pipe on the soil surface at the sample location.
   c. Using the knife carefully cut a circular shape around the sampling pipe, so that the pipe will easily slide through the soil and roots. This reduces soil compaction.
   d. Using the dampened hammer, gently pound the sampling pipe into the soil. Hammer the core 10 cm into the soil. There is a black line indicating this depth on the soil core.
   e. With the core in place, dig down to the bottom of the core and extract the core into a bucket that has been marked with the name of the vegetation zone.
3. Repeat along each transect for each vegetation zone, making sure that the 10 cm soil sample is placed into the properly marked bucket (to assure vegetation zones are not mixed).
4. Thoroughly mix each bucket of soil with the large stainless steel spoon. Clean and rinse the spoon with de-ionized water between buckets.
5. Gather several quart-sized freezer zip-loc bags and a permanent marker. Label each small zip-loc bag with the site name, vegetation zone, number of cores, the bag number (i.e. 4 of 7), date, and name of collector.
6. Using a clean stainless steel spoon, take enough randomly selected spoonfuls of soil to fill the labeled quart size Zip-loc bag.
7. Seal the Zip-loc bag and place in a larger zip-loc freezer bag labeled with the site name, date, and number of smaller bags contained. Place in cooler and ice down.
8. Since the resultant nutrient and % organic matter will be weighted based on the % area of each vegetation zone, it is imperative that the vegetation zones marked on the soil bags are also marked on the vegetation zone map that is part of the wetland characterization sheet. Do not forget to include the approximate % of each area in the wetland.
9. Rinse field equipment with deionized water.
10. Return the samples to the Center for Wetlands, and store in the refrigerator in the back lab pending laboratory analysis.

APPENDIX D
COEFFICIENT OF CONSERVATISM SCORES

Table D-1. Coefficient of Conservatism (CC) scores for 561 macrophytes identified in isolated depressional freshwater forested wetlands in Florida.

Species	CC	Species	CC
Acalypha gracilens	3.3	*Begonia cucullata*	1.5
Acer rubrum	5.2	*Berchemia scandens*	5.1
Acrostichum danaeifolium	6.2	*Betula nigra*	4.8
Agalinis filifolia	6.7	*Bidens alba*	1.0
Agrostis hyemalis	5.4	*Bidens discoidea*	4.8
Albizia julibrissin	0.0	*Bidens mitis*	3.8
Aloe vera	0.0	*Bignonia capreolata*	4.8
Alternanthera philoxeroides	0.0	*Bischofia javanica*	0.0
Alternanthera sessilis	0.7	*Blechnum serrulatum*	5.5
Amaranthus australis	2.6	*Blechum pyramidatum*	0.0
Amaranthus blitum	0.0	*Boehmeria cylindrica*	4.5
Amaranthus spinosus	0.0	*Boltonia diffusa*	3.8
Ambrosia artemisiifolia	0.7	*Bromus catharticus*	0.0
Ampelopsis arborea	3.3	*Bulbostylis stenophylla*	4.4
Amphicarpum muhlenbergianum	5.0	*Callicarpa americana*	2.4
Andropogon glomeratus	3.1	*Callisia repens*	0.0
Andropogon virginicus	2.6	*Campsis radicans*	3.3
Annona glabra	6.8	*Canna flaccida*	5.7
Anthaenantia villosa	7.1	*Caperonia castaneifolia*	2.4
Apios americana	3.1	*Carex debilis*	6.5
Ardisia crenata	1.0	*Carex frankii*	6.0
Ardisia escallonioides	0.0	*Carex gigantea*	6.4
Aristida beyrichiana	9.8	*Carex glaucescens*	7.1
Aristida patula	6.3	*Carex longii*	3.6
Aristida purpurascens	6.0	*Carex striata*	5.7
Aristida spiciformis	6.4	*Carex verrucosa*	7.1
Asplenium platyneuron	4.8	*Carphephorus odoratissimus*	7.6
Aster carolinianus	6.9	*Carphephorus paniculatus*	6.0
Aster dumosus	3.6	*Celtis laevigata*	5.0
Aster elliottii	4.2	*Centella asiatica*	1.9
Aster pilosus	5.4	*Cephalanthus occidentalis*	6.0
Aster subulatus	4.5	*Cercis canadensis*	4.0
Aster tenuifolius	7.1	*Chamaecrista fasciculata*	0.0
Axonopus fissifolius	2.8	*Chamaecrista nictitans*	2.9
Axonopus furcatus	2.4	*Chamaesyce hypericifolia*	0.0
Azolla caroliniana	2.6	*Chaptalia tomentosa*	7.9
Baccharis halimifolia	2.1	*Chenopodium album*	0.0
Bacopa caroliniana	6.0	*Chiococca alba*	5.6
Bacopa monnieri	4.3	*Chrysobalanus icaco*	6.3

Table D-1. Continued.

Species	CC	Species	CC
Cicuta maculata	5.0	*Diodia teres*	1.9
Cinnamomum camphora	0.2	*Diodia virginiana*	2.4
Cirsium nuttallii	4.8	*Dioscorea bulbifera*	0.0
Cissus trifoliata	4.2	*Diospyros virginiana*	4.0
Citrus Xaurantium	0.0	*Drosera brevifolia*	6.7
Cladium jamaicense	5.5	*Drosera capillaris*	6.7
Cleistes bifaria	7.1	*Drymaria cordata*	1.2
Clethra alnifolia	5.2	*Duchesnea indica*	3.6
Cliftonia monophylla	5.0	*Dulichium arundinaceum*	6.8
Coelorachis cylindrica	5.6	*Echinochloa colona*	0.7
Coelorachis rugosa	6.3	*Echinochloa crusgalli*	0.0
Coelorachis tuberculosa	6.5	*Echinochloa walteri*	3.1
Colocasia esculenta	0.0	*Eclipta prostrata*	1.7
Commelina diffusa	1.7	*Eleocharis baldwinii*	2.1
Commelina erecta	4.8	*Eleocharis flavescens*	3.6
Commelina virginica	4.8	*Eleocharis interstincta*	5.5
Conoclinium coelestinum	4.3	*Eleocharis microcarpa*	3.0
Conyza canadensis	0.3	*Eleocharis vivipara*	2.4
Cornus foemina	4.8	*Elephantopus nudatus*	4.0
Crataegus viridis	8.6	*Eleusine indica*	0.0
Crinum americanum	7.6	*Elymus virginicus*	4.0
Ctenium aromaticum	10.0	*Eragrostis atrovirens*	1.8
Cuphea carthagenensis	1.4	*Erechtites hieracifolia*	2.1
Cyclospermum leptophyllum	1.2	*Erianthus giganteus*	6.0
Cynanchum scoparium	4.8	*Erigeron quercifolius*	2.9
Cynodon dactylon	0.0	*Erigeron strigosus*	2.4
Cyperus croceus	1.8	*Erigeron vernus*	4.3
Cyperus distinctus	3.8	*Eriocaulon compressum*	6.7
Cyperus erythrorhizos	4.2	*Eriocaulon decangulare*	6.7
Cyperus haspan	2.6	*Eriocaulon ravenelii*	4.8
Cyperus iria	1.2	*Eryngium prostratum*	4.0
Cyperus lanceolatus	2.4	*Eugenia uniflora*	0.0
Cyperus odoratus	3.6	*Eupatorium capillifolium*	0.5
Cyperus polystachyos	2.4	*Eupatorium leptophyllum*	3.6
Cyperus retrorsus	1.7	*Eupatorium mohrii*	5.5
Cyperus surinamensis	1.9	*Eupatorium rotundifolium*	6.2
Cyperus virens	3.9	*Eupatorium serotinum*	4.8
Cyrilla racemiflora	4.5	*Eustachys glauca*	2.4
Desmodium incanum	0.0	*Eustachys petraea*	0.0
Desmodium lineatum	6.0	*Euthamia caroliniana*	2.6
Desmodium paniculatum	3.6	*Euthamia minor*	3.6
Dichondra caroliniensis	1.9	*Ficus aurea*	5.7
Digitaria bicornis	0.0	*Fimbristylis dichotoma*	4.0
Digitaria ciliaris	0.3	*Fraxinus caroliniana*	7.1
Digitaria serotina	1.8	*Fuirena scirpoidea*	3.8

Table D-1. Continued.

Species	CC	Species	CC
Galactia elliottii	3.8	*Ixora chinensis*	0.0
Galactia volubilis	3.6	*Jacquemontia tamnifolia*	0.0
Galium hispidulum	3.3	*Juncus coriaceus*	5.1
Galium tinctorium	3.1	*Juncus dichotomus*	2.9
Galium uniflorum	5.1	*Juncus effusus*	1.9
Gaylussacia frondosa	6.7	*Juncus marginatus*	2.4
Gaylussacia mosieri	7.4	*Juncus megacephalus*	3.3
Gelsemium sempervirens	4.0	*Juncus polycephalus*	3.3
Gnaphalium falcatum	1.9	*Juncus repens*	5.2
Gnaphalium obtusifolium	2.4	*Juncus tenuis*	2.4
Gordonia lasianthus	6.7	*Juniperus virginiana*	5.2
Gratiola ramosa	5.0	*Justicia angusta*	6.0
Gratiola virginiana	7.1	*Justicia ovata*	5.5
Habenaria repens	4.8	*Kummerowia striata*	0.0
Hedychium coronarium	0.0	*Kyllinga brevifolia*	0.3
Hedyotis corymbosa	2.0	*Kyllinga pumila*	3.3
Hedyotis uniflora	3.6	*Lachnanthes caroliniana*	3.1
Hemarthria altissima	0.0	*Lachnocaulon anceps*	5.5
Hydrocotyle bonariensis	3.3	*Lachnocaulon engleri*	4.8
Hydrocotyle ranunculoides	3.1	*Lachnocaulon minus*	6.0
Hydrocotyle umbellata	2.9	*Lactuca graminifolia*	2.7
Hydrocotyle verticillata	3.1	*Lantana camara*	0.0
Hymenachne amplexicaulis	0.0	*Leersia hexandra*	4.8
Hypericum brachyphylum	6.8	*Lemna minor*	1.0
Hypericum chapmanii	7.1	*Lepidium virginicum*	0.2
Hypericum cistifolium	5.0	*Leptochloa uninervia*	3.0
Hypericum fasciculatum	5.7	*Leucothoe axillaris*	6.0
Hypericum gallioides	6.0	*Leucothoe racemosa*	6.2
Hypericum hypericoides	4.0	*Ligustrum japonicum*	0.0
Hypericum mutilum	3.6	*Ligustrum lucidum*	0.0
Hypericum myrtifolium	5.5	*Ligustrum sinense*	0.0
Hypoxis curtissii	6.0	*Limnobium spongia*	4.8
Hyptis alata	4.3	*Linaria canadensis*	0.3
Hyptis mutabilis	0.0	*Lindernia crustacea*	0.6
Ilex cassine	8.1	*Lindernia grandiflora*	3.6
Ilex coriacea	6.0	*Liquidambar styraciflua*	3.3
Ilex glabra	4.3	*Litsea aestivalis*	9.8
Ilex myrtifolia	8.3	*Lobelia floridana*	6.5
Ilex opaca	6.0	*Lolium perenne*	0.0
Ilex vomitoria	4.8	*Lonicera japonica*	0.0
Ilex x attenuata	7.1	*Lophiola aurea*	6.5
Ipomoea indica	0.6	*Ludwigia alata*	4.5
Ipomoea sagittata	5.4	*Ludwigia curtissii*	4.4
Iris hexagona	7.1	*Ludwigia hirtella*	6.0
Itea virginica	7.9	*Ludwigia linifolia*	4.5

Table D-1. Continued.

Species	CC	Species	CC
Ludwigia maritima	3.3	*Nymphaea odorata*	5.5
Ludwigia microcarpa	3.1	*Nymphoides aquatica*	5.7
Ludwigia octovalvis	2.4	*Nyssa aquatica*	3.6
Ludwigia palustris	4.0	*Nyssa biflora*	7.4
Ludwigia peruviana	1.2	*Oeceoclades maculata*	0.4
Ludwigia repens	2.9	*Oplismenus hirtellus*	3.3
Ludwigia virgata	3.9	*Osmunda cinnamomea*	5.5
Luziola fluitans	4.8	*Osmunda regalis*	6.9
Lycopodiella alopecuroides	6.7	*Oxalis corniculata*	1.2
Lycopodiella prostrata	7.1	*Oxalis debilis*	0.0
Lycopus rubellus	5.2	*Oxypolis filiformis*	6.7
Lycopus virginicus	5.2	*Paederia foetida*	0.0
Lygodium japonicum	0.0	*Panicum aciculare*	4.8
Lygodium microphyllum	0.0	*Panicum chamaelonche*	4.8
Lyonia ligustrina	6.9	*Panicum ciliatum*	4.5
Lyonia lucida	6.0	*Panicum commutatum*	4.5
Lythrum alatum	3.0	*Panicum dichotomum*	4.0
Magnolia grandiflora	3.6	*Panicum ensifolium*	5.0
Magnolia virginiana	8.1	*Panicum erectifolium*	5.7
Malus angustifolia	6.0	*Panicum hemitomon*	5.0
Matelea floridana	6.7	*Panicum repens*	0.0
Mecardonia acuminata	3.9	*Panicum rigidulum*	4.5
Melaleuca quinquenervia	0.0	*Panicum scabriusculum*	5.0
Melia azedarach	0.0	*Panicum sphaerocarpon*	5.1
Melochia corchorifolia	1.5	*Panicum spretum*	5.4
Melothria pendula	1.8	*Panicum tenerum*	5.0
Micranthemum glomeratum	3.6	*Panicum tenue*	4.2
Micranthemum umbrosum	4.3	*Panicum verrucosum*	4.3
Micromeria brownei	4.8	*Parietaria floridana*	1.8
Mikania scandens	2.4	*Parthenocissus quinquefolia*	3.0
Mitchella repens	6.7	*Paspalidium geminatum*	3.6
Mitreola petiolata	5.4	*Paspalum acuminatum*	2.0
Mitreola sessilifolia	5.4	*Paspalum conjugatum*	3.1
Modiola caroliniana	3.2	*Paspalum laeve*	3.8
Momordica charantia	0.0	*Paspalum monostachyum*	9.1
Morrenia odorata	0.0	*Paspalum notatum*	0.0
Morus alba	1.2	*Paspalum plicatulum*	2.4
Morus rubra	3.6	*Paspalum repens*	4.0
Myrica cerifera	3.1	*Paspalum setaceum*	2.1
Myrica heterophyla	7.9	*Paspalum urvillei*	1.2
Myrica inodora	9.0	*Passiflora incarnata*	3.0
Nandina domestica	0.0	*Passiflora suberosa*	3.0
Nephrolepis biserrata	5.2	*Peltandra virginica*	3.6
Nephrolepis exaltata	4.8	*Pentodon pentandrus*	6.0
Nuphar luteum	5.2	*Persea borbonia*	6.3

Table D-1. Continued.

Species	CC	Species	CC
Persea palustris	7.4	*Quercus phellos*	7.4
Phalaris angusta	0.0	*Quercus virginiana*	4.2
Phanopyrum gymnocarpon	6.0	*Rapanea punctata*	5.2
Phlebodium aureum	6.8	*Rhexia alifanus*	6.9
Photinia pyrifolia	5.7	*Rhexia lutea*	6.5
Phyla nodiflora	1.4	*Rhexia mariana*	3.8
Phyllanthus tenellus	0.0	*Rhexia nashii*	6.2
Phyllanthus urinaria	0.0	*Rhexia petiolata*	6.2
Physalis angulata	1.2	*Rhexia virginica*	5.0
Phytolacca americana	1.2	*Rhododendron viscosum*	7.6
Pieris phillyreifolia	9.5	*Rhodomyrtus tomentosa*	0.0
Pinus clausa	5.6	*Rhoeo discolor*	0.0
Pinus elliottii	4.0	*Rhus copallinum*	2.4
Pinus palustris	7.1	*Rhynchospora capitelatta*	6.0
Pinus serotina	7.1	*Rhynchospora cephalantha*	4.3
Pinus taeda	3.3	*Rhynchospora chalarocephala*	4.8
Plantago lanceolata	1.2	*Rhynchospora chapmanii*	6.0
Pluchea camphorata	4.3	*Rhynchospora colorata*	5.5
Pluchea carolinensis	3.6	*Rhynchospora corniculata*	6.0
Pluchea foetida	3.8	*Rhynchospora decurrens*	6.3
Pluchea longifolia	2.8	*Rhynchospora fascicularis*	4.5
Pluchea odorata	3.8	*Rhynchospora filifolia*	6.0
Pluchea rosea	3.6	*Rhynchospora gracilenta*	6.0
Polygala cymosa	9.0	*Rhynchospora inundata*	6.0
Polygala lutea	3.6	*Rhynchospora latifolia*	6.9
Polygonum hydropiperoides	2.6	*Rhynchospora microcarpa*	4.5
Polygonum punctatum	2.6	*Rhynchospora microcephala*	4.8
Polygonum sagittatum	4.8	*Rhynchospora miliacea*	7.1
Polypremum procumbens	1.2	*Rhynchospora odorata*	6.7
Pontederia cordata	5.0	*Rhynchospora plumosa*	6.4
Populus deltoides	1.2	*Rhynchospora pusilla*	6.7
Pouzolzia zeylanica	0.4	*Rhynchospora tracyi*	8.3
Proserpinaca palustris	3.8	*Rhynchospora wrightiana*	7.1
Proserpinaca pectinata	3.8	*Richardia brasiliensis*	0.0
Prunus caroliniana	3.0	*Rivina humilis*	1.2
Prunus serotina	3.6	*Rosa carolina*	7.1
Psilotum nudum	3.6	*Rosa palustris*	6.9
Psychotria nervosa	3.6	*Rubus argutus*	2.1
Psychotria sulzneri	3.6	*Rubus cuneifolius*	1.9
Pteridium aquilinum	3.6	*Rubus trivialis*	1.9
Ptilimnium capillaceum	3.1	*Ruellia caroliniensis*	4.3
Pueraria montana	0.0	*Rumex crispus*	0.2
Quercus geminata	5.2	*Rumex obtusifolius*	0.7
Quercus laurifolia	3.6	*Rumex pulcher*	0.6
Quercus nigra	2.1	*Sabal palmetto*	4.5

Table D-1. Continued.

Species	CC	Species	CC
Sabatia bartramii	6.8	*Solanum americanum*	1.4
Sacciolepis indica	1.9	*Solanum capsicoides*	1.4
Sacciolepis striata	3.6	*Solanum carolinense*	1.2
Sageretia minutiflora	7.9	*Solanum nigrum*	0.0
Sagittaria graminea	5.5	*Solanum tampicense*	0.7
Sagittaria lancifolia	4.5	*Solanum viarum*	0.0
Sagittaria latifolia	5.0	*Solidago canadensis*	3.0
Salix caroliniana	2.1	*Solidago fistulosa*	3.6
Salix nigra	3.3	*Solidago gigantea*	3.2
Salvia lyrata	0.0	*Solidago latissimifolia*	1.8
Sambucus canadensis	1.7	*Solidago sempervirens*	5.0
Samolus ebracteatus	5.7	*Sonchus asper*	1.5
Sapium sebiferum	0.0	*Sorghum bicolor*	0.0
Sarcostemma clausum	2.4	*Spartina bakeri*	5.5
Sarracenia flava	9.3	*Spermacoce assurgens*	1.2
Sarracenia minor	4.8	*Spermacoce verticillata*	0.0
Saururus cernuus	5.5	*Sporobolus floridanus*	7.1
Schinus terebinthifolius	0.0	*Sporobolus indicus*	0.2
Scirpus cyperinus	4.5	*Stachys floridana*	1.4
Scleria baldwinii	6.7	*Stenotaphrum secundatum*	0.8
Scleria georgiana	6.2	*Stillingia aquatica*	7.4
Scleria reticularis	5.1	*Styrax americanus*	6.9
Scleria triglomerata	4.8	*Syngonanthus flavidulus*	5.2
Scoparia dulcis	2.4	*Taxodium ascendens*	8.8
Scutellaria integrifolia	5.7	*Thalia geniculata*	6.2
Senecio glabellus	4.0	*Thelypteris dentata*	6.0
Senna obtusifolia	0.0	*Thelypteris hispidula*	4.5
Senna pendula	0.0	*Thelypteris interrupta*	5.2
Serenoa repens	4.5	*Thelypteris kunthii*	5.2
Sesbania herbacea	1.0	*Thelypteris palustris*	3.6
Sesbania vesicaria	0.5	*Tilia americana*	5.5
Setaria parviflora	3.1	*Toxicodendron radicans*	1.9
Seymeria cassioides	6.0	*Tradescantia fluminensis*	0.0
Sida acuta	1.0	*Tradescantia ohiensis*	0.9
Sida rhombifolia	1.0	*Tradescantia zebrina*	0.0
Sideroxylon celastrinum	6.0	*Triadenum virginicum*	5.0
Sideroxylon reclinatum	6.0	*Trifolium repens*	0.0
Smilax auriculata	3.8	*Tripsacum dactyloides*	4.0
Smilax bona-nox	2.6	*Typha domingensis*	1.2
Smilax glauca	3.3	*Typha latifolia*	1.2
Smilax laurifolia	5.2	*Ulmus americana*	7.4
Smilax rotundifolia	3.2	*Urena lobata*	0.0
Smilax smallii	4.5	*Urochloa mutica*	0.0
Smilax tamnoides	3.6	*Utricularia gibba*	3.6
Smilax walteri	6.0	*Utricularia purpurea*	6.7

Table D-1. Continued.

Species	CC	Species	CC
Vaccinium arboreum	6.4	*Vitis shuttleworthii*	1.2
Vaccinium corymbosum	5.7	*Vittaria lineata*	1.2
Vaccinium darrowii	6.2	*Waltheria indica*	2.4
Vaccinium elliottii	6.7	*Wedelia trilobata*	0.0
Vaccinium myrsinites	4.8	*Woodwardia areolata*	5.7
Valeriana scandens	7.1	*Woodwardia virginica*	4.8
Verbena bonariensis	0.0	*Xanthosoma sagittifolium*	0.0
Verbena brasiliensis	0.0	*Xyris ambigua*	5.7
Viburnum nudum	3.6	*Xyris caroliniana*	5.7
Viburnum obovatum	1.2	*Xyris elliottii*	5.7
Viburnum odoratissimum	0.0	*Xyris fimbriata*	5.7
Vicia sativa	0.4	*Xyris jupicai*	1.7
Vigna luteola	3.6	*Xyris platylepis*	3.6
Viola lanceolata	4.8	*Youngia japonica*	0.0
Vitis aestivalis	2.9	*Yucca aloifolia*	1.2
Vitis cinerea	2.0	*Zea mays*	0.0
Vitis rotundifolia	2.1		

APPENDIX E
CANDIDATE METRICS

Table E-1. Candidate metrics based on the diatom assemblage.

	N	P	A
Tolerance Metrics			
Indicator Species Sensitive Taxa - Abundance	1	1	1
Indicator Species Sensitive Taxa - Presence/Absence	1	1	1
Indicator Species Tolerant Taxa - Abundance	1	1	1
Indicator Species Tolerant Taxa - Presence/Absence	1	1	1
Community Composition Metrics			
Richness	1		
Evenness	1		
Shannon Diversity	1		
Simpson's Index	1		
Autecological Metrics			
Morphological Guild - Erect	1	1	1
Morphological Guild - Stalked	1	1	1
Morphological Guild - Unattached	1	1	1
Morphological Guild - Prostrate/Adnate	1	1	1
Morphological Guild - Variable	1	1	1
Motility - Highly Motile	1	1	1
Motility - Moderately Motile	1	1	1
Motility - Highly & Moderately Motile	1	1	1
Motility - Not Motile	1	1	1
Motility - Variable	1	1	1
Pollution Tolerance - Very Tolerant	1	1	1
Pollution Tolerance - Moderately Tolerant	1	1	1
Pollution Tolerance - Very & Moderately Tolerant	1	1	1
Pollution Tolerance - Sensitive / Intolerant	1	1	1
Dissolved Oxygen Class 1 (Bahls 1993)	1	1	1
Dissolved Oxygen Class 2 (Bahls 1993)	1	1	1
Dissolved Oxygen Class 3 (Bahls 1993)	1	1	1
Dissolved Oxygen Class 4 (Bahls 1993)	1	1	1
Dissolved Oxygen Class 5 (Bahls 1993)	1	1	1
Wet/Dry Preference Class 1 (van Dam et al. 1994)	1	1	1
Wet/Dry Preference Class 2 (van Dam et al. 1994)	1	1	1

Table E-1. Continued.

	N	P	A
Autecological Metrics - *continued*			
Wet/Dry Preference Class 3 (van Dam et al. 1994)	1	1	1
Wet/Dry Preference Class 4 (van Dam et al. 1994)	1	1	1
Wet/Dry Preference Class 5 (van Dam et al. 1994)	1	1	1
Nitrogen Metabolism Class 1 (van Dam et al. 1994)	1	1	1
Nitrogen Metabolism Class 2 (van Dam et al. 1994)	1	1	1
Nitrogen Metabolism Class 3 (van Dam et al. 1994)	1	1	1
Nitrogen Metabolism Class 4 (van Dam et al. 1994)	1	1	1
pH Class 1 (van Dam et al. 1994)	1	1	1
pH Class 2 (van Dam et al. 1994)	1	1	1
pH Class 3 (van Dam et al. 1994)	1	1	1
pH Class 4 (van Dam et al. 1994)	1	1	1
pH Class 5 (van Dam et al. 1994)	1	1	1
pH Class 6 (van Dam et al. 1994)	1	1	1
Salinity Class 1 (van Dam et al. 1994)	1	1	1
Salinity Class 2 (van Dam et al. 1994)	1	1	1
Salinity Class 3 (van Dam et al. 1994)	1	1	1
Salinity Class 4 (van Dam et al. 1994)	1	1	1
Saprobity Class 1 (van Dam et al. 1994)	1	1	1
Saprobity Class 2 (van Dam et al. 1994)	1	1	1
Saprobity Class 3 (van Dam et al. 1994)	1	1	1
Saprobity Class 4 (van Dam et al. 1994)	1	1	1
Saprobity Class 5 (van Dam et al. 1994)	1	1	1
Trophic Class 1 (van Dam et al. 1994)	1	1	1
Trophic Class 2 (van Dam et al. 1994)	1	1	1
Trophic Class 1 & 2 (van Dam et al. 1994)	1	1	1
Trophic Class 3 (van Dam et al. 1994)	1	1	1
Trophic Class 4 (van Dam et al. 1994)	1	1	1
Trophic Class 5 (van Dam et al. 1994)	1	1	1
Trophic Class 6 (van Dam et al. 1994)	1	1	1
Trophic Class 7 (van Dam et al. 1994)	1	1	1

Table E-2. Candidate metrics based on the macrophyte assemblage. Metrics were calculated in multiple forms including N - number, P - percent, A - abundance, F - frequency of occurrence, and O - other.

	N	P	A	F	O
Wetland Plant Status					
Obligate Species	1	1	1	1	
Facultative Wetland Species	1	1	1	1	
Facultative Species	1	1	1	1	
Facultative Upland Species	1	1	1	1	
Upland Species	1	1	1	1	
Obligate + Facultative Wetland Species	1	1	1	1	
Obligate + Facultative Wetland + Facultative Species	1	1	1	1	
Facultative Upland + Upland Species	1	1	1	1	
Upland + Facultative Upland + Facultative Species	1	1	1	1	
Plant Growth Form & Taxa Metrics					
Graminoid Species	1	1	1	1	
Carex sp.	1	1	1	1	
Herbaceous Species	1	1	1	1	
Species in Asteraceae	1	1	1	1	
Polygonum sp.	1	1	1	1	
Graminoids to Herbaceous	1		1	1	
Vine Species	1	1	1	1	
Vines that are Woody	1	1	1	1	
Shrub Species	1	1	1	1	
Tree Species	1	1	1	1	
Tree and Shrub Species	1	1	1	1	
Salix sp.	1	1	1	1	
Hardwoods	1	1	1	1	
Trees as Hardwoods	1	1	1	1	
Nyssa sp.	1	1	1	1	
Trees as *Nyssa* sp.	1	1	1	1	
Acer rubrum	1	1	1	1	
Trees as *Acer rubrum*	1	1	1	1	
Trees as Conifers	1	1	1	1	
Trees as *Taxodium* sp.	1	1	1	1	
Native Evergreen Shrubs	1	1	1	1	
Native Ferns	1	1	1	1	
Native Perennial Graminoids	1	1	1	1	
Native Perennial Herbs	1	1	1	1	

Table E-2. Continued.

	N	P	A	F	O
Indicator Species					
Sensitive (R:I Ratio, where R equals the number of Reference Sites and I equals number of agricultural and urban sites a species was found at)	1	1	1	1	
Tolerants (R:I Ratio, where R equals the number of Reference Sites and I equals number of agricultural and urban sites a species was found at)	1	1	1	1	
TWINSPAN Sensitive Species	1	1	1	1	
TWINSPAN Sensitive Species by Presence Absence	1	1	1	1	
TWINSPAN Tolerant Species	1	1	1	1	
TWINSPAN Tolerant Species by Presence Absence	1	1	1	1	
TWINSPAN Sensitive Species, Excluding Exotic Species	1	1	1	1	
TWINSPAN Sensitive Species by Presence Absence, Excluding Exotic Species	1	1	1	1	
TWINSPAN Tolerant Species, Excluding Exotic Species	1	1	1	1	
TWINSPAN Tolerant Species by Presence Absence, Excluding Exotic Species	1	1	1	1	
Indicator Species Sensitive Taxa – Occurrence	1	1	1	1	
Indicator Species Sensitive Taxa - Presence/Absence	1	1	1	1	
Indicator Species Tolerant Taxa – Occurrence	1	1	1	1	
Indicator Species Tolerant Taxa - Presence/Absence	1	1	1	1	
Indicator Species Sensitive Taxa, Excluding Exotic Species – Occurrence	1	1	1	1	
Indicator Species Sensitive Taxa, Excluding Exotic Species - Presence/Absence	1	1	1	1	
Indicator Species Tolerant Taxa, Excluding Exotic Species – Occurrence	1	1	1	1	
Indicator Species Tolerant Taxa, Excluding Exotic Species - Presence/Absence	1	1	1	1	
Modified FQI Score	1		1		
Exotic Species Metric	1	1	1	1	
Longevity Metrics					
Annuals	1	1	1	1	
Native Annuals	1	1	1	1	
Perennials	1	1	1	1	
Native Perennials	1	1	1	1	

Table E-2. Continued.

	N	P	A	F	O
Longevity Metrics - *continued*					
Annual to Perennial Ratio			1	1	
Native Annual to Native Perennial Ratio			1	1	
Richness Metrics					
Species Richness by Site					1
Species Richness by Quadrat					1
Species Richness by Occurrence					1
Species Richness by Transect					1
Mean Site Evenness					1
Dominant Species				1	
Log (Proportion of Dominant Species)			1		
Vascular Genera	1				
Nonvascular Genera	1				

Table E-3. Candidate metrics based on the macroinvertebrate assemblage. Metrics were calculated in multiple forms including N – number, P – percent, A – abundance, and T – number of taxa.

	N	P	A	T
Tolerance Metrics				
Indictor Species Analysis				
Sensitive Taxa - Abundance	1	1	1	
Sensitive Taxa - Presence/Absence	1	1	1	
Tolerant Taxa - Abundance	1	1	1	
Tolerant Taxa - Presence/Absence	1	1	1	
Florida Index	1			
Lake Condition Index	1			
Community Structure & Balance Metrics				
Mixed Taxonomic Levels				
Crustacea + Mollusca	1	1	1	
Dominant Taxa	1	1	1	
Exotic Richness				1
Taxa Richness	1			1
Tubificida/Insecta	1			
Phylum				
Phylum Richness	1			1
Annelida	1	1	1	
Arthropoda	1	1	1	
Mollusca	1	1	1	
Platyhelminthes	1	1	1	
Class				
Class Richness	1			1
Arachnida	1	1	1	
Bivalva	1	1	1	
Crustacea	1	1	1	
Gastropoda	1	1	1	1
Insecta	1	1	1	
Oligochaeta	1	1	1	1
Plecypoda	1	1	1	1
Turbellaria	1	1	1	1
Order				
Order Richness	1			1
Acariforrmes	1	1	1	
Amphipoda	1	1	1	1
Anostraca	1	1	1	
Basommatophora	1	1	1	
Coleoptera	1	1	1	
Collembola	1	1	1	
Decapoda	1	1	1	1
Diptera	1	1	1	1

Table E-3. Continued.

	N	P	A	T
Community Structure & Balance Metrics - *continued*				
Order - *continued*				
Diptera - non-Chironomid	1	1	1	
Ephemeroptera	1	1	1	1
Ephemeroptera + Plecoptera + Trichoptera	1	1	1	1
Ephemeroptera + Trichoptera + Odonata	1	1	1	1
Haplotaxida	1	1	1	
Hemiptera	1	1	1	1
Heteroptera	1	1	1	
Hoplonemertea	1	1	1	
Isopoda	1	1	1	1
Lepidoptera	1	1	1	
Lumbriculida	1	1	1	
Megaloptera	1	1	1	
Mesogastropoda	1	1	1	
Odonata	1	1	1	1
Oribatei	1	1	1	
Plecoptera	1	1	1	1
Trichoptera	1	1	1	1
Tricladida	1	1	1	
Tromibidiformes	1	1	1	1
Tubificida	1	1	1	
Veneroida	1	1	1	
Zygoptera	1	1	1	1
Family				
Family Richness	1			1
Aeshnidae	1	1	1	
Ancylidae	1	1	1	
Arrenuridae	1	1	1	
Asellidae	1	1	1	
Baetidae	1	1	1	
Belostomatidae	1	1	1	
Cambaridae	1	1	1	
Ceratopogonidae	1	1	1	
Chaoboridae	1	1	1	
Chironomidae	1	1	1	1
Coenagrionidae	1	1	1	
Corixidae	1	1	1	
Crangonyctidae	1	1	1	

Table E-3. Continued.

	N	P	A	T
Community Structure & Balance Metrics - *continued*				
Family - *continued*				
Culicidae	1	1	1	
Curulionidae	1	1	1	
Dryopidae	1	1	1	
Dytiscidae	1	1	1	
Enchytraeidae	1	1	1	
Haliplidae	1	1	1	
Helodidae	1	1	1	
Hydrophilidae	1	1	1	
Libellulidae	1	1	1	
Lumbriculidae	1	1	1	
Naididae	1	1	1	
Noteridae	1	1	1	
Notonectidae	1	1	1	
Physidae	1	1	1	
Planorbidae	1	1	1	
Tabanidae	1	1	1	
Tipulidae	1	1	1	
Tubificidae	1	1	1	
Sub-Families of Chironomidae				
Chironominae	1	1	1	
Orthocladiinae	1	1	1	1
Tanypodinae	1	1	1	
Ratio Tanypodinae/Orthocladiinae	1			
Ratio Chironominae/Orthocladiinae	1			
Ratio (Tanypodinae + Chironominae)/Orthocladiinae	1			
Genus				
Genus Richness	1			1
Ablabesmyia	1	1	1	
Anopheles	1	1	1	
Arrenurus	1	1	1	
Atrichopogon	1	1	1	
Beardius	1	1	1	
Belostoma	1	1	1	
Berosus	1	1	1	
Bratislavia	1	1	1	
Buenoa	1	1	1	
Caecidotea	1	1	1	
Callibaetis	1	1	1	
Chaoborus	1	1	1	
Chironomus	1	1	1	
Crangonyx	1	1	1	

Table E-3. Continued.

		N	P	A	T
Community Structure & Balance Metrics - *continued*					
	Genus - *continued*				
		Culex	1	1	1
		Dero	1	1	1
		Desmopachria	1	1	1
		Eclipidrilus	1	1	1
		Goeldichironomus	1	1	1
		Haemonais	1	1	1
		Hydrocanthus	1	1	1
		Hydrochus	1	1	1
		Ischnura	1	1	1
		Kiefferulus	1	1	1
		Labrundinia	1	1	1
		Larsia	1	1	1
		Micromenetus	1	1	1
		Monopelopia	1	1	1
		Notonecta	1	1	1
		Ochlerotatus	1	1	1
		Pachydiplax	1	1	1
		Pachydrus	1	1	1
		Parachironomus	1	1	1
		Paramerina	1	1	1
		Pelonomus	1	1	1
		Physella	1	1	1
		Polypedilum	1	1	1
		Pristina	1	1	1
		Pristinella	1	1	1
		Scirtes	1	1	1
		Tanytarsus	1	1	1
		Tropisternus	1	1	1
		Zavreliella	1	1	1
Functional Feeding Group Metrics					
	Browsers and Grazers of Periphyton		1	1	1
	Collector-Filterers/Suspension Feeders		1	1	1
	Collector-Gatherers/Deposit Feeders		1	1	1
	Macrophyte Piercers		1	1	1
	Macrophyte Shredders		1	1	1
	Parasites		1	1	1
	Periphyton Scrapers		1	1	1
	Predators & Carnivores		1	1	1
	Scavenger (animals)		1	1	1

APPENDIX F
SUMMARY STATISTICS

Table F-1. Summary statistics of richness (R), evenness (E), Shannon diversity (H), and Simpson's index (S) for the diatom assemblage (genus level).

Site	R	E	H	S	Site	R	E	H	S
PR1	12	0.82	2.03	0.82	CA3	18	0.63	1.81	0.73
PR4	16	0.63	1.75	0.69	CA4	17	0.78	2.21	0.86
PR5	12	0.71	1.77	0.77	CA5	22	0.73	2.27	0.83
PR6	11	0.60	1.44	0.68	CA6	13	0.89	2.28	0.87
NR2	22	0.68	2.09	0.77	SA2	31	0.75	2.57	0.85
NR3	13	0.76	1.96	0.79	SA3	25	0.84	2.69	0.90
NR4	12	0.83	2.07	0.84	SA4	20	0.78	2.33	0.84
NR6	16	0.81	2.25	0.87	SA5	24	0.61	1.95	0.73
CR3	30	0.84	2.87	0.92	SA6	20	0.86	2.56	0.90
CR4	22	0.74	2.28	0.85	PU3	28	0.83	2.78	0.90
CR5	26	0.81	2.63	0.86	PU4	20	0.72	2.17	0.82
CR6	9	0.66	1.45	0.68	NU2	23	0.83	2.60	0.89
SR1	19	0.65	1.90	0.77	NU4	14	0.66	1.73	0.73
SR2	13	0.59	1.50	0.63	NU5	12	0.57	1.41	0.58
SR3	26	0.81	2.63	0.90	NU6	23	0.74	2.31	0.85
SR4	22	0.77	2.37	0.86	CU1	9	0.76	1.66	0.78
SR5	35	0.84	2.98	0.93	CU3	39	0.79	2.89	0.90
SR6	36	0.81	2.89	0.91	CU5	31	0.69	2.37	0.78
PA2	12	0.80	1.98	0.83	CU6	26	0.84	2.73	0.91
PA3	10	0.70	1.62	0.73	SU1	17	0.62	1.77	0.72
PA5	19	0.74	2.17	0.83	SU2	28	0.88	2.92	0.93
PA6	17	0.74	2.08	0.83	SU3	24	0.73	2.31	0.84
NA4	34	0.78	2.75	0.89	SU4	21	0.62	1.87	0.67
NA6	14	0.64	1.68	0.70	SU5	23	0.83	2.59	0.89
CA2	14	0.76	2.00	0.80	SU6	15	0.59	1.59	0.70

Table F-2. Summary statistics of richness (R), jackknife estimators of species richness ($Jack_1$, $Jack_2$), evenness (E), Shannon diversity (H), and Whittaker's beta diversity (β_W) for the macrophyte assemblage (species level).

Site	R	$Jack_1$	$Jack_2$	E	H	β_W
PR1	37	43	45	0.89	3.2	3.5
PR2	31	37	41	0.87	3.0	2.6
PR3	37	50	56	0.80	2.9	5.5
PR4	24	30	30	0.79	2.5	4.8
PR5	23	28	31	0.88	2.7	4.0
PR6	27	37	41	0.83	2.7	4.1
PR7	32	43	42	0.71	2.5	8.3
PR8	34	47	54	0.83	2.9	2.0
NR1	14	15	11	0.83	2.2	2.8
NR2	40	55	62	0.84	3.1	6.8
NR3	28	37	44	0.84	2.8	5.1
NR4	32	43	49	0.85	2.9	4.8
NR5	29	38	43	0.85	2.9	6.4
NR6	42	48	49	0.89	3.3	3.6
NR7	35	44	45	0.83	3.0	3.1
NR8	31	38	40	0.86	2.9	3.8
NR9	15	17	16	0.79	2.1	1.2
CR1	31	35	36	0.91	3.1	2.3
CR2	31	40	46	0.82	2.8	6.7
CR3	53	72	84	0.86	3.4	5.0
CR4	40	54	58	0.83	3.1	3.4
CR5	31	46	56	0.89	3.1	9.4
CR6	27	31	32	0.92	3.0	1.4
CR7	49	62	69	0.88	3.4	4.9
CR8	22	28	30	0.79	2.4	1.4
CR9	53	69	76	0.87	3.5	5.0
CR10	35	42	42	0.89	3.2	3.4
CR11	46	56	58	0.91	3.5	3.2
SR1	27	36	40	0.85	2.8	4.1
SR2	25	31	31	0.86	2.8	3.3
SR3	25	31	33	0.86	2.8	4.6
SR4	20	25	28	0.85	2.6	3.1
SR5	29	38	42	0.88	3.0	3.0
SR6	16	20	21	0.91	2.5	1.5
SR7	60	77	79	0.89	3.6	3.8
SR8	40	53	61	0.84	3.1	3.1
SR9	26	34	40	0.85	2.8	1.1

Table F-2. Continued.

Site	R	$Jack_1$	$Jack_2$	E	H	β_W
PA1	36	45	47	0.85	3.0	7.9
PA2	29	37	39	0.88	3.0	4.9
PA3	29	38	40	0.82	2.7	7.5
PA4	25	34	39	0.87	2.8	6.7
PA5	50	68	75	0.87	3.4	6.2
PA6	34	39	42	0.89	3.1	4.5
PA7	52	62	64	0.89	3.5	5.6
PA8	22	29	33	0.87	2.7	1.6
PA9	44	56	58	0.89	3.4	3.3
PA10	35	53	68	0.88	3.1	3.8
NA1	19	24	25	0.86	2.5	2.8
NA2	36	49	57	0.82	2.9	3.6
NA3	13	17	17	0.72	1.8	5.6
NA4	53	74	85	0.90	3.6	4.7
NA5	45	60	71	0.89	3.4	6.1
NA6	41	50	55	0.86	3.2	4.6
NA7	44	55	60	0.91	3.4	4.0
NA8	21	25	27	0.90	2.7	3.4
NA9	60	73	80	0.92	3.8	3.4
NA10	36	45	46	0.88	3.2	4.1
NA11	53	73	85	0.89	3.5	5.9
NA12	77	99	114	0.90	3.9	6.6
CA1	44	56	64	0.89	3.4	2.8
CA2	18	23	26	0.91	2.6	1.4
CA3	34	45	49	0.88	3.1	4.2
CA4	43	51	53	0.89	3.3	4.8
CA5	26	33	37	0.80	2.6	2.3
CA6	26	34	39	0.80	2.6	6.2
CA7	60	81	91	0.85	3.5	4.6
CA8	47	63	69	0.85	3.3	4.4
CA9	31	41	47	0.90	3.1	6.1
SA1	21	26	29	0.87	2.7	4.3
SA2	34	43	45	0.90	3.2	5.5
SA3	38	45	46	0.91	3.3	2.1
SA4	31	40	45	0.83	2.9	5.8
SA5	27	35	36	0.83	2.7	7.0
SA6	50	65	74	0.86	3.4	4.8
SA7	20	27	31	0.91	2.7	7.2
SA8	40	52	55	0.87	3.2	7.5
SA9	36	47	51	0.87	3.1	5.9

Table F-2. Continued.

Site	R	$Jack_1$	$Jack_2$	E	H	β_W
PU1	37	47	54	0.89	3.2	5.9
PU2	34	42	43	0.87	3.1	5.8
PU3	42	59	68	0.86	3.2	5.4
PU4	43	57	66	0.93	3.5	6.1
PU5	42	61	76	0.88	3.3	4.4
PU6	29	39	46	0.85	2.8	5.3
PU7	24	30	34	0.91	2.9	0.8
PU8	35	50	59	0.83	2.9	1.6
PU9	16	22	26	0.79	2.2	0.2
PU10	38	55	63	0.87	3.1	7.5
NU1	37	46	50	0.89	3.2	3.8
NU2	46	51	50	0.86	3.3	5.3
NU3	48	60	62	0.88	3.4	5.4
NU4	27	34	38	0.87	2.9	4.0
NU5	34	44	49	0.86	3.0	3.9
NU6	26	39	49	0.78	2.5	7.7
NU7	41	55	65	0.89	3.3	5.1
NU8	41	50	50	0.88	3.3	4.2
NU9	35	49	58	0.86	3.1	1.7
NU10	42	55	61	0.86	3.2	5.5
CU1	46	56	61	0.85	3.2	5.5
CU2	44	54	58	0.88	3.3	3.1
CU3	33	40	43	0.86	3.0	5.4
CU4	42	58	67	0.84	3.1	7.0
CU5	34	40	37	0.85	3.0	4.5
CU6	23	27	24	0.83	2.6	4.0
CU7	45	59	64	0.87	3.3	4.3
CU8	63	92	111	0.85	3.5	4.5
CU9	32	40	45	0.86	3.0	1.7
CU10	26	36	40	0.77	2.5	2.8
CU11	38	49	56	0.90	3.3	3.0
SU1	55	73	81	0.88	3.5	6.4
SU2	38	50	59	0.89	3.2	3.3
SU3	24	32	36	0.85	2.7	4.5
SU4	38	53	63	0.85	3.1	4.8
SU5	16	20	21	0.90	2.5	2.6
SU6	21	30	33	0.83	2.5	3.8
SU7	48	59	60	0.91	3.5	3.5
SU8	26	37	45	0.86	2.8	2.8
SU9	47	59	64	0.90	3.5	5.3
SU10	39	50	56	0.87	3.2	3.9

Table F-3. Summary statistics of richness (R), evenness (E), Shannon diversity (H), and Simpson's index (S) for the macroinvertebrate assemblage (genus level).

Site	R	E	H	S	Site	R	E	H	S
PR1	11	0.58	0.62	1.40	CA5	14	0.68	0.71	1.80
PR4	1	0.00	0.00	0.00	CA6	11	0.59	0.60	1.42
PR5	5	0.97	0.78	1.56	CA7	20	0.90	0.92	2.70
PR6	10	0.87	0.84	2.01	CA8	12	0.65	0.70	1.62
PR7	18	0.84	0.89	2.41	CA9	9	0.55	0.59	1.20
PR8	15	0.69	0.74	1.86	SA2	14	0.79	0.84	2.09
NR2	8	0.63	0.65	1.30	SA3	17	0.73	0.78	2.07
NR3	24	0.86	0.91	2.72	SA4	7	0.53	0.48	1.03
NR4	21	0.82	0.89	2.50	SA5	15	0.89	0.89	2.41
NR6	19	0.88	0.90	2.59	SA6	14	0.68	0.73	1.79
NR8	4	0.15	0.08	0.21	SA7	11	0.60	0.63	1.45
NR9	11	0.56	0.59	1.34	SA8	19	0.73	0.81	2.14
CR3	11	0.76	0.80	1.81	SA9	8	0.51	0.46	1.05
CR4	17	0.82	0.87	2.33	PU3	23	0.84	0.90	2.63
CR5	16	0.76	0.84	2.12	PU4	16	0.79	0.82	2.19
CR6	20	0.86	0.90	2.57	PU10	13	0.83	0.85	2.13
CR8	8	0.29	0.23	0.60	NU2	7	0.76	0.71	1.47
CR9	7	0.37	0.29	0.72	NU4	11	0.49	0.47	1.16
CR10	25	0.87	0.92	2.79	NU5	8	0.83	0.77	1.72
CR11	21	0.84	0.89	2.55	NU6	10	0.72	0.73	1.66
SR1	17	0.67	0.72	1.90	NU10	8	0.68	0.66	1.41
SR2	10	0.87	0.84	2.01	CU1	25	0.82	0.90	2.64
SR3	18	0.76	0.82	2.21	CU3	9	0.68	0.69	1.49
SR4	11	0.78	0.79	1.87	CU5	9	0.57	0.63	1.25
SR5	21	0.84	0.89	2.56	CU6	11	0.51	0.54	1.22
SR6	15	0.68	0.71	1.84	CU7	11	0.72	0.75	1.72
SR7	11	0.61	0.60	1.46	CU8	21	0.89	0.91	2.72
SR8	22	0.85	0.89	2.62	CU9	13	0.69	0.70	1.78
SR9	14	0.45	0.44	1.19	CU10	5	0.19	0.12	0.31
PA2	24	0.66	0.77	2.09	CU11	17	0.71	0.77	2.02
PA3	26	0.78	0.87	2.54	SU1	11	0.63	0.69	1.50
PA5	18	0.79	0.85	2.28	SU2	13	0.79	0.82	2.04
PA6	8	0.63	0.63	1.32	SU3	19	0.65	0.73	1.91
NA4	13	0.83	0.84	2.13	SU4	12	0.58	0.60	1.44
NA6	9	0.73	0.77	1.60	SU5	16	0.87	0.89	2.42
NA10	5	0.38	0.32	0.61	SU6	20	0.63	0.75	1.89
NA11	18	0.81	0.87	2.35	SU7	14	0.67	0.74	1.76
CA2	20	0.73	0.78	2.20	SU8	8	0.65	0.65	1.36
CA3	8	0.67	0.65	1.40	SU9	10	0.47	0.45	1.09
CA4	18	0.76	0.84	2.20					

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BIOGRAPHICAL SKETCH

Kelly Ann Chinners Reiss was born March, 12, 1976, in West Palm Beach, Florida. She grew up in the boat, on the beach, or otherwise outdoors. Her parents recount that Kelly and her sister Casey learned to swim before they could walk. Kelly spent weeks at a time camping, canoeing, kayaking, and whitewater rafting with her extended family, which sparked her intrigue in the natural world around her. She attended Palm Beach Lakes High School, lettered in 3 varsity sports (volleyball, soccer, and softball), and graduated as valedictorian in 1994.

Kelly completed her Bachelors of Science degree through the School of Forest Resources and Conservation at the University of Florida in 1998, graduating with honors. During her undergraduate years, Kelly was active in the Environmental Action Groups (serving as Chair of the Endangered Species Committee) and University Habitat for Humanity (serving as Co-President). She spent one formative summer in Asheville, North Carolina, working at ReCreation Experiences and honing her home repair, leadership, and rock climbing/belaying skills. In May of 1998 she began studying reclaimed wetlands in the Central Florida Phosphate District. She completed her Masters of Science degree in systems ecology through the Department of Environmental Engineering Sciences, College of Engineering, University of Florida, in 2000. More recently, Kelly has studied forested wetlands throughout Florida, where she glimpsed her first Florida black bear.