United States
Environmental Protection
Agency

Environmental Research
Laboratory
Corvallis, Oregon 97333

EPA/600/R-93/222
November 1993

Research and Development

EPA

HABITAT QUALITY ASSESSMENT OF TWO WETLAND TREATMENT SYSTEMS IN FLORIDA -- A PILOT STUDY

HABITAT QUALITY ASSESSMENT OF TWO WETLAND TREATMENT SYSTEMS IN FLORIDA--A PILOT STUDY

By:

Lynne S. McAllister
ManTech Environmental Technology, Inc.
USEPA, Environmental Research Laboratory
Corvallis, OR 97333

Project Officer:

Mary E. Kentula
U.S. Environmental Protection Agency

U.S. Environmental Protection Agency
Environmental Research Laboratory
200 SW 35th Street
Corvallis, OR 97333

DISCLAIMER

The information in this document has been funded wholly or in part by the United States Environmental Protection Agency under contract number 68-C8-0006 to ManTech Environmental Technology, Inc. and 68-C8-0056 to AScI Corporation, Duluth, MN. It has been subjected to the Agency's peer and administrative review, and it has been approved for publication as an EPA document. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

This document should be cited as:

McAllister, L.S. 1993. Habitat quality assessment of two wetland treatment systems in Florida--A pilot study. EPA/600/R-93/222. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon.

CONTENTS

TABLES

FIGURES

ACKNOWLEDGEMENTS

Numerous individuals contributed to the completion of this research project, and, although I cannot list all of them, I am greatly appreciative of their efforts. I am especially grateful to Jane Schuler, who served as half of the field team and worked many long, difficult days in the field to complete data collection on schedule, and to JoEllen Honea, who made a substantial contribution to the final document by conducting extensive literature searches, developing the computer data base, performing all the data analyses, and preparing tables for the report.

Richard Olson served as the technical lead and provided guidance throughout the project. Robert Bastian, Richard Olson, and Robert Knight conceptualized the research approach and initiated project planning. I am grateful to Paul Adamus, who provided literature for supporting material, training in the Wetland Evaluation Technique, and advice and guidance in the planning and analysis stages of the project. Arthur Sherman provided important documentation on sampling and quality assurance procedures. Cindy Hagley, Debra Taylor, and Bill Sanville at the EPA Environmental Research Laboratory in Duluth, MN, were very helpful in planning and preparing for the field season.

Janelle Eskuri received water samples from the field and conducted whole effluent toxicity tests at EPA's Duluth Environmental Research Laboratory. Janelle Eskuri, Teresa Norberg-King, and Lara Anderson prepared documentation of whole effluent test methods and results, which is incorporated in this report. Nan Allen at the University of Minnesota-Duluth identified and enumerated all the invertebrates, prepared documentation, and provided data for the final report. Ann Hershey conducted the data quality checks for invertebrate quality assurance. Brenda Huntley digitized cover types on aerial photographs, conducted all the Geographic Information System work, and prepared the site location maps. Robert Gibson and Ted Ernst wrote data analysis programs and provided data base and software operation support. Kristina Miller assisted with word processing and editing and prepared Figure 1.

I thank the site managers -- Alan Oyler at the Bureau of Wastewater, Orlando, FL, and David Hill with Wastewater Operations in Lakeland, FL -- for permission to sample at their wastewater wetlands, for providing existing water quality data, and for their cooperation throughout the project. I am grateful to Seth Blitch, Jim Burney, Dr. Bill Dunn, and Mike Mahler for the time they saved the field crew by identifying pressed plant specimens. Dr. Peter Frederick and Steven McGehee from the University of Florida in Gainesville conducted aerial bird surveys and provided data and supporting documentation for this report. Dave Hill, the Lakeland site manager, conducted several informal ground bird surveys at the Lakeland site. Post, Buckley, Schuh, and Jernigan, Inc., provided species lists of biota at the Orlando site. The time and effort spent by numerous individuals who contributed data for use in the discussion are greatly appreciated.

Bill Ainslie, Robert Knight, and Glenn Guntenspergen provided technical review for the manuscript. Deborah Coffey and Allan Deutsch provided quality assurance and editorial reviews, respectively. All reviewers provided constructive comments and suggestions for the final draft.

EXECUTIVE SUMMARY

The use of constructed wetland treatment systems (WTS) for treating municipal wastewater is increasing in the United States, but little documentation exists concerning the ability of these systems to duplicate or sustain wetland functions. A pilot study was designed primarily to examine methods and the usefulness of various wetland indicators for assessing wildlife habitat as a wetland function. The study took place at six WTS sites throughout the United States. This report focusses on two of those sites, located in Florida, one near Orlando and one near Lakeland. Results from the other four sites are presented in two separate reports, one covering two sites in Mississippi and the other covering two sites in the arid West.

Data for vegetation, macroinvertebrate, site morphology, water quality, and bird use were collected in the field or compiled from existing data sets. To examine the wildlife habitat function, various metrics were calculated and assessed for their usefulness as indicators of wildlife habitat quality. Wildlife habitat quality was assessed mainly with respect to bird habitat. Indicator values were compared with ranges of values of the same indicators from wetlands in the southeastern United States not used for wastewater treatment (non-WTS). Comparison data from non-WTS sites were found in the literature. Comparisons were meant to provide a very preliminary examination of the wildlife habitat condition of the two WTS studied by identifying any obvious deviations from indicator values from non-WTS. In addition to indicator testing, whole-effluent toxicity tests were conducted on influent and effluent water samples from each WTS to determine whether contaminants are entering the WTS and potentially affecting biota.

Comparisons of habitat indicators for which data from non-WTS existed showed that indicator values from the two Florida WTS were generally within the range of values found in non-WTS in the southeastern U.S. Macroinvertebrate genera richness and bird species richness were within the upper part of the range or above the range of values reported for non-WTS. Foraging wading bird densities were in the lower range of densities calculated from simultaneous surveys at a nearby non-WTS marsh system. However, the WTS appear to be important as nesting habitat for wading birds. Concentrations of various water parameters were at the low to middle portion of the range of values for non-WTS. Plant species richness was at the high end of the range of values for non-WTS. Survival and reproduction of *Ceriodaphnia dubia* and *Pimephales promelas* were not significantly reduced ($P<0.05$) in whole-effluent toxicity tests at the two WTS studied.

These preliminary results provide evidence that the habitat condition of the two WTS studied is comparable with that of non-WTS in the same region and suggest that the two WTS provide favorable wildlife habitat as an ancillary benefit. Both sites are large, contain remote areas with a variety of habitats, and support breeding bird colonies, which is evidence that wastewater treatment and wildlife habitat enhancement are compatible.

It is recommended that, for future assessment of wildlife habitat quality in WTS, indicators from the following categories be further tested and developed:

- vegetation
- macroinvertebrates
- site morphology

Bird use may be a suitable indicator of the faunal component of a WTS, but further consideration should be given to reduction of sampling effort, collection of more specific metrics, and the direct relevance of bird use to habitat quality. Macroinvertebrate sampling should be expanded to include benthic macroinvertebrates. Use of existing water nutrient data and whole-effluent toxicity tests should have low priority for evaluating and monitoring the wildlife habitat function of WTS.

Water quality data are difficult to interpret consistently in terms of wildlife habitat quality. Data quality and comparability are difficult to assure when using existing data, which are collected and analyzed differently among sites and are intended for purposes other than evaluating habitat quality. The collection of a smaller set of water parameters during the field effort, such as dissolved oxygen, turbidity, and ammonia nitrogen, might provide information on system stressors, which can be used to help explain the status of other indicators.

The whole-effluent toxicity testing methods were successful and confidence in the results was high. Single, whole-effluent tests, however, do not provide time-integrated information about the effects of specific substances in wastewater on wildlife. Because documentation of effects is a long-term process and can become very expensive, toxicity testing should be a separate activity. It should be done routinely to detect potential contamination in suspect wetlands. Suspect wetlands could be those receiving industrial discharges, where contaminants have been found in the past, or where routine biological monitoring indicates a potential problem requiring further investigation.

Future studies comparing wildlife habitat quality of WTS to non-WTS should include simultaneous measurement of selected indicators at nearby reference sites (non-WTS) so that confounding factors are minimized and systematic comparisons can be made between WTS and non-WTS. For assessing the actual habitat quality of WTS, however, it is necessary to establish guidelines and criteria for rating habitat quality to avoid the possibility that habitat quality assessment is based on comparisons with suboptimal wetlands. Data reduction and assessment techniques, possibly including development of habitat quality indices, should be explored in future studies so that various indicators can be aggregated and conclusions about overall habitat quality can be derived from more rigorous analyses.

INTRODUCTION

Freshwater, brackish, and saltwater wetlands often serve as natural water purifiers for wastewater from point and non-point sources. Wetlands designed specifically for treating water are often built to take advantage of this purifying function. Recent declines in federal funds allocated to municipal pollution control, as well as water pollution control mandates under the Clean Water Act for both municipal and industrial point source dischargers, have led to an increase in the construction of wetlands for treating wastewater. Municipal constructed wetland treatment systems (hereafter referred to as WTS) are engineered complexes of saturated substrates, emergent and submergent vegetation, animal life, and water that simulate natural wetlands for the primary purpose of wastewater treatment (Hammer and Bastian 1989). These systems receive partially treated wastewater and are designed to reduce biochemical oxygen demand (BOD), nutrient and metals concentrations, and levels of other pollutants (Kadlec and Kadlec 1979, Nixon and Lee 1986). WTS are used for a variety of purposes, including treatment of municipal and home wastewater (US EPA 1988a, Conway and Murtha 1989), acid mine drainage (Brodie et al. 1989), landfill and industrial wastewater (Staubitz et al. 1989), nonpoint source pollution (Dickerman et al. 1985, Costello 1989), and urban stormwater (King County 1986).

Wetland treatment systems fall into two general categories: 1) vegetated submerged-bed wetlands, in which water moves through a soil or rock substrate in the bed of the system where it makes contact with plant roots; and 2) free water surface wetlands, in which most of the water flow is above ground over saturated soils (US EPA 1988a, Reed et al. 1988). Free water surface wetlands were the focus of this study because they are designed to replicate natural wetland systems. They are usually constructed with several sections, or cells, separated by weirs which can be used to control water level and flow rate. Water is treated primarily through assimilation of nutrients and other pollutants by microorganisms in the substrate and attached to plant roots. Plant species selected for these systems often contain large amounts of aerenchyma and are efficient in translocating oxygen from the atmosphere to their root zones, which facilitates respiration by microorganisms.

Role of EPA in WTS Operations

One objective of the Clean Water Act is to restore and maintain the physical, chemical, and biological integrity of waters of the United States through the elimination of discharges of pollutants (Yocum et al. 1989). Under the Clean Water Act, most natural wetlands are considered to be waters of the United States. The EPA is responsible for implementing the Clean Water Act and associated regulations on the discharge of wastewaters to the Nation's waters. Discharges must meet requirements set in a

National Pollutant Discharge Elimination System (NPDES) permit issued by EPA or a delegated state (Davis and Montgomery 1987). Presently, WTS usually are not considered "waters of the United States" (Bastian et al. 1989) and therefore discharges to these systems are not regulated by EPA under the Clean Water Act. However, discharges from WTS to waters of the United States must meet NPDES requirements. Therefore, EPA must evaluate the capability of WTS to meet water quality standards under Sections 401 and 402 of the Clean Water Act. Water monitoring programs are in place at WTS which discharge to waters of the U.S.

In addition to its regulatory role, the EPA is interested in the ancillary functions of WTS, including wildlife habitat. WTS attract wildlife and therefore cannot be considered isolated operations. Habitat quality concerns include potential risks to wildlife by substances entering in wastewater (Davis and Montgomery 1987). Because many wildlife species are mobile, conditions in WTS can influence wildlife health and use in a network of wetlands, including wetlands that are waters of the U.S. It is therefore important for the EPA to develop methods for assessing and monitoring the condition of WTS and to coordinate these methods with methods used for natural, restored, and created wetlands.

Assessing Wetland Function and Ecological Condition

While WTS can duplicate structural aspects of some natural wetlands, little is known about the replication of wetland functions. The functions that wetlands perform depend upon wetland type, location, the local geology, topography, and hydrology, and other watershed characteristics. Typical wetland functions include: wildlife habitat, recreation, nutrient and pollutant assimilation and retention, detritus and dissolved nutrient and organic matter production, reduction of downstream sedimentation, floodwater retention, and groundwater recharge. With the exception of nutrient removal, wetland functions are normally considered "ancillary", or supplemental in WTS because these systems are designed primarily for wastewater treatment and secondarily for other purposes.

Wetland treatment systems can and do provide various ancillary functions, but concerns exist about potential contamination and effects on wetland ecological condition caused by additions of wastewater (Godfrey et al. 1985, US EPA 1984, Mudroch and Capobianco 1979). The ecological condition, or "health" of a wetland refers to its biological integrity, sustainability, and ability to serve multiple functions. A "healthy" wetland exhibits structures and functions necessary to sustain itself and is free of most known stressors or problems (Rapport 1989, Schaeffer et al. 1988).

Ecological condition can be assessed and monitored on the basis of various wetland attributes, or indicators. Indicators are characteristics of the environment that, when measured, quantify a habitat characteristic, magnitude of stress, degree of exposure to the stressor, or degree of ecological response to the exposure (Hunsaker and Carpenter 1990). Indicators can be measured or quantified through field sampling, remote sensing, or analysis of existing data. Many potentially valuable indicators exist for assessing and monitoring a resource, but it is most desirable to identify a suite of indicators that best describes the overall condition of the resource.

Factors Affecting Habitat Quality

Wetland treatment systems often provide wildlife habitat as an ancillary function (Piest and Sowls 1985, Sather 1989). Nutrient additions usually increase net primary productivity (Guntenspergen and Stearns 1985) and promote waterfowl production (Cedarquist 1979). Alternatively, extremely high nutrient concentrations or loadings and lack of variation in water depth can encourage establishment of macrophyte monocultures with lower habitat value (Fetter et al. 1978, Kadlec and Bevis 1990). Nutrient enrichment in eutrophic and hypereutrophic systems can cause algal blooms, resulting in highly variable dissolved oxygen concentrations and reduced light penetration. The latter condition greatly affects plant species diversity and distribution, particularly of submergent species. Species composition and extent of aquatic macrophytes can affect the abundance and diversity of aquatic invertebrates (Dvorak and Best 1982, Reid 1985, Voights 1976); subsequently, plant-invertebrate associations influence use by waterfowl (Krull 1970, Teels et al. 1976). Wetland morphology, location, and hydrologic regime also interact to influence the quality of habitat that develops.

Wildlife using WTS can be exposed to pollutants. Although municipal discharges to wetlands are regulated by state and federal agencies, and industrial discharges are not recommended for WTS, occasional exceptions and/or violations of regulations can result in at least temporary discharge of potentially harmful substances to WTS. This creates potential for some organisms to be affected through exposure, ingestion, or bioaccumulation of substances. Detailed information about wetland animal communities in WTS is lacking in the literature (Brennan 1985).

Research Objectives

There have been no comprehensive, large-scale studies of the ecological condition and wildlife use of WTS (Bastian, personal communication, U.S. EPA, Washington, DC). A pilot study was designed as an exploratory effort for examining research methods, indicators, logistics, and capabilities for conducting preliminary assessments of wildlife habitat as an ancillary function of WTS. The study was not intended to provide probability samples to statistically characterize a defined population of WTS. However, many of the conclusions about wildlife habitat quality drawn from the data collected in this study can be used to design future research.

Because WTS are not considered waters of the U.S., the issue of jurisdiction during and after the operational phase is complex. Results of this study should not be used to make inferences regarding jurisdiction of WTS or to provide support for mitigation credit for WTS under Section 404. There is still inadequate knowledge of the ability of WTS to replace wetland functions.

The objectives of the study were:

- to assess the usefulness of methods and indicators for evaluating the wildlife habitat quality of WTS,
- to identify any major differences in values of wildlife habitat indicators in WTS and non-WTS, and
- to provide baseline data and identify approaches for a more focussed follow-up project that will provide specific information for developing measures of the wildlife habitat quality of WTS.

METHODS

The pilot study included sampling and habitat quality assessment at six WTS in the United States (Table 1). The same general framework and study design was used for conducting work at all sites. Pilot study results, however, are reported in three separate EPA documents, each dealing with two sites: 1) Florida sites (this report); 2) Mississippi sites (McAllister 1992); and 3) western sites (McAllister 1993).

Pilot Study Overview

This section discusses activities concerning the design of the overall pilot study, including selection of the six WTS sites studied, habitat quality assessment techniques, the indicators chosen for measurement, and the field sampling schedule.

Site Selection and Sampling Schedule

Six free water surface municipal WTS in the United States were chosen for sampling in 1991. The sites were chosen based on the following criteria:

- location in the Southeast or in the arid and semi-arid West, so that WTS in two different geographic and climatic regions of the country could be studied,
- representing a range of sizes,
- representing a range of ages and in operation for at least one year,
- availability of water quality data for use in indicator analysis,
- permission to use the site, and
- interest in collaboration by site operators and other groups.

Field data were collected in July and August 1991 according to the schedule shown in Table 2.

Table 1. Names, locations, construction dates, and sizes of WTS sampled in the pilot study.

Site name	Location	Year built	Size (ha)
Orlando	Orlando, FL	1987	486.00
Lakeland	Lakeland, FL	1987	498.00
Collins	Collins, MS	1987	4.47
Ocean Springs	Ocean Springs, MS	1990	9.28
Show Low	Show Low, AZ	1980	284.00
Incline Village	Incline Village, NV	1985	198.00

Table 2. Pilot study field sampling schedule.

Sampling location	Dates
Incline Village, NV	July 8-12
Show Low, AZ	July 19-23
Ocean Springs, MS	July 30-Aug. 3
Collins, MS	Aug. 6-9
Orlando, FL	Aug. 14-19
Lakeland, FL	Aug. 19-23

Habitat Quality Assessment

Habitat quality was assessed mainly with respect to birds because birds were used as a faunal indicator in the project. More species of birds than of mammals are dependent on wetlands, thus more literature exists on wetland habitat requirements of birds. Many of the habitat components necessary for birds are also beneficial to mammals (e.g., cover extent and diversity, food resources, a landscape habitat mosaic).

Two general assessment techniques were evaluated in the pilot study for use in assessing wildlife habitat quality as an ancillary benefit. One technique was the measurement of selected indicators of habitat quality. A suite of indicators was chosen for measurement at the WTS sampled. Indicators were selected based on the likelihood that

- sample collection, processing, and labor costs would not exceed budget constraints,
- data collection would not exceed available human resources,

- adequate data could be collected within the 4-5 days spent at each site,

- chosen indicators could be used to effectively characterize and evaluate wildlife habitat quality,

- required sampling would minimize environmental impact, and

- variability of collected data would be low within a site and consistent among sites.

Some of these criteria were unknown for some of the candidate indicators (e.g., ability of indicators to characterize and evaluate wildlife habitat quality, data variability, adequate number of data). One of the objectives of the study, however, was to test the indicators by determining their ease of measurement and the quality of data obtained in relation to logistics involved in collecting them. Indicators chosen for testing are listed in Table 3. They are grouped into one of three data source categories:

- data collected in the field
- data acquired from aerial photographs
- existing data sets and records kept for each site

The other assessment technique, performed at only half the sites, was the use of the Wetland Evaluation Technique (WET) (Adamus et al. 1987), a rapid assessment technique for evaluating wetland ancillary values, including wildlife habitat. WET was given low priority in the pilot study, and limited time at some of the larger sites prevented its completion. It was therefore not conducted at either of the Florida sites. Its use in WTS is discussed in more detail by McAllister (1992, 1993).

Florida Study

The remainder of this document addresses only the Florida portion of the pilot study. This section contains site descriptions, and field and laboratory methods.

Site Descriptions

The general locations and designs of the Florida sites are shown in Figure 1. Additional site details are shown on site maps in Appendix A. In addition, a management/operations contact is given for each site in Appendix B. Each site is briefly described below.

Table 3. Indicators of wetland habitat condition measured during the 1991 pilot study.

Ecological Component	Indicators
A. Indicators measured in the field:	
Vegetation	-Species composition and percent coverage -Structural diversity and dominance -Species dominance -Species richness
Invertebrates	-Species and functional group composition and relative abundance -Genera richness
Water	-Whole effluent toxicity tests on inflow and outflow
Birds	-Density -Species richness
B. Indicators taken from aerial photographs:	
Site morphology	-Wetland area -Distance of land/water interface per hectare -Distance of edge between selected cover types per hectare -Ratio of open water area to area covered by vegetation -Relative coverage of selected vegetation types
C. Indicators obtained from existing data sets:	
Water	-pH -Dissolved oxygen -Biochemical oxygen demand -Total suspended solids -Ammonia nitrogen -Total Kjeldahl nitrogen -Total phosphorus -Fecal coliform bacteria

Orlando, FL The 494-ha WTS was constructed in 1987 in an active cattle pasture to provide additional nutrient removal for tertiary treated domestic wastewater from the City of Orlando's Iron Bridge Regional Water Pollution Control Facility. The nutrient sensitivity of Florida's rivers and lakes necessitates advanced tertiary treatment before water can be discharged into the St. John's River, approximately 1.3 km away. The project is one of the first large-scale artificial wetlands designed to treat domestic wastewater as well as to provide wildlife habitat. The WTS has 17 cells and a lake. The design flow is 75,700 m^3/day (20 mgd), which results in an average residence time of water in the WTS of 30 days. Most of the necessary water treatment is accomplished in cells 1-12, which are composed primarily of bulrush and cattail. Cells 13-17, which are designed and managed to provide habitat diversity for wildlife, are composed of communities of mixed vegetation, including palm and cypress swamps. In the off-hunting season, the site serves as a recreational park for area residents. Post, Buckley, Schuh, and Jernigan, a consultant for the city of Orlando, regularly collects water, flora, and fauna data and conducts monitoring and management at the site.

Lakeland, FL This WTS receives secondary treated water and provides tertiary treatment for the city of Lakeland, FL. The treatment plant in Lakeland primarily handles domestic wastewater but also receives some wastes from packaging and food industries, photo finishing, and linen services (Dave Hill, personal communication, Wastewater Operations, Lakeland, FL). The WTS has a design flow of 86,400 m^3/day (14 mgd), and a design detention time of 80-100 days (Dave Hill, personal communication, Wastewater Operations, Lakeland, FL). The Lakeland plant treats domestic and industrial wastewater before discharging water to the WTS. The site is located 97 km west of Orlando on an old phosphate mine. Built in 1987, the WTS is comprised of seven cells and covers 498 ha, making it the largest free water surface WTS in the U.S. at the time of construction. It supports herbaceous wetland plants, primarily cattails, and very dense shrub communities, which have colonized the site since shortly after its construction. The sediment is a very fine silt or clay that is settling out slowly, so the substrate is soft and unstable, and the bottom contour is irregular. Many areas of the site, particularly in cells 4-7, are deep and more characteristic of lakes than of palustrine wetlands. Cell 6, the deepest cell, is over $\geq$14 m deep. This and other cells are possibly receiving groundwater. Islands in cell 5 support large wading bird rookeries. Water monitoring data are collected routinely at the site.

Field and Laboratory Methods

This section describes the methods for all activities conducted during the field season in July and August, 1991 (Table 2), as well as laboratory analysis of water and invertebrate

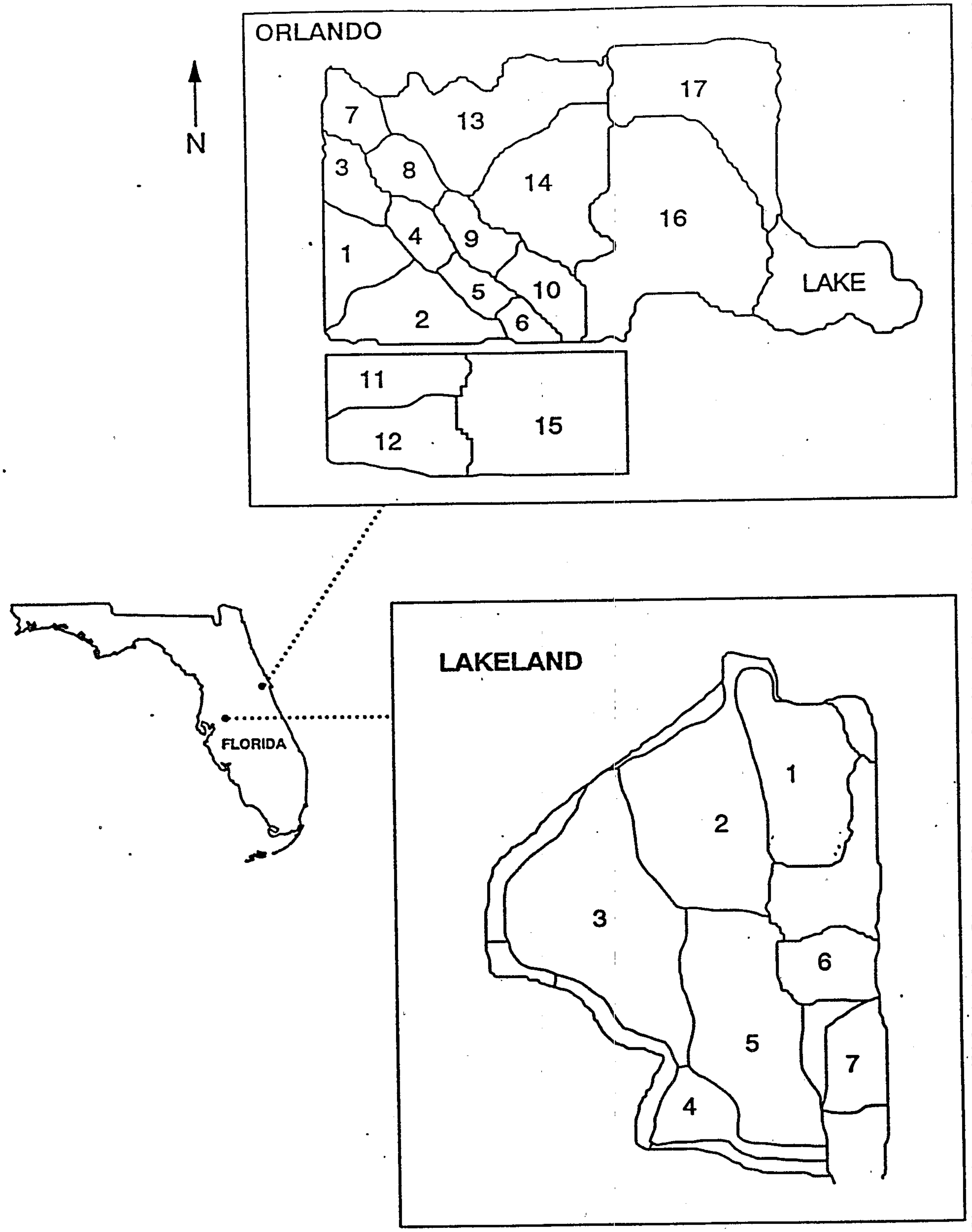


Figure 1. Location and general design of the constructed wetland sites studied in Florida. The cells of each WTS are numbered for reference.

samples. The activities described are: site characterization; vegetation, invertebrate, water, and bird sampling; and invertebrate and water laboratory analyses. Indicators measured in the field or calculated are listed in Table 3.

Site Characterization

Site characterization included gathering information about the layout of the site and the distribution of major vegetation types, photographing the major habitat types on the site, and recording wildlife species observed during the several days spent sampling the site.

The first task at each site was to drive and/or walk around the entire site and along all interconnecting dikes to roughly map the locations of major vegetation types, open water, bare soil, roads, and rookeries visible from the dikes. Cover type boundaries were delineated on available site maps. This exercise provided cover maps of dominant plant species to verify air photo interpretation and to ensure that vegetation transects could be sited representatively.

Vegetation Sampling

Because of the dense shrub community and unstable substrate at the Lakeland site, vegetation was sampled only at the Orlando site. Vegetation sampling included transect establishment through major cover types, cover estimation at points along transects, plant specimen preservation, and identification of unknown plants by local botanists. Collected data were used to calculate indicators listed in Table 3. At the Lakeland site, effort was devoted to mapping plant communities as ground truth for aerial photo interpretation.

Transect establishment

Transect placement required a great deal of judgement based on the initial site survey and the distribution of vegetation types. In general, transects were placed:

- through the major vegetation strata at the site. Major strata were defined for this study as: emergent-*Typha*, emergent-*Scirpus*, emergent-other dominant, emergent-mixed species, submerged, floating-leaved, scrub/shrub, forested, and open water; and

- to intersect a diversity of dominant plant species represented within each stratum.

Wetland area, accessibility to vegetation sample points, and configuration and size of plant communities were factors considered when determining the length of individual transects and the number of sample points along transects.

Due to the large size of the Orlando WTS, sampling was not conducted in all cells. Cells 1-10 were dominated by a relatively uniform cover of either *Typha* spp. or *Scirpus* spp. Therefore one transect was established in each of those communities -- cell 2 and cell 9, respectively -- to characterize percent cover. In addition, one transect was established in the mixed community in cell 13, and one transect was established in the forested community of cell 16 to characterize a variety of plant species and strata more typical of cells 13-17. A greater number of points were sampled in the mixed community than in the *Typha* and *Scirpus* communities of cells 2 and 9 to characterize a greater variety of plant communities and structures (Appendix A). Transects began at the wetland edge (i.e., where hydrophytic plants or hydric soils were present) and extended into the wetland. Upland habitats were not sampled. Sixty-five points were sampled at the Orlando WTS. Sampling points were spaced 20 m apart along all transects. Cells 11, 12, and 15 to the south of the entrance road were not sampled.

<u>Cover estimation</u>

One, two, or three plots were established at each sample point along transects, depending upon the structural types of vegetation present. A 1-m^2 quadrat was used for sampling herbaceous vegetation (emergent, submergent, floating-leaved); a 5-m^2 quadrat was used for shrubs (0.5-6.0 m tall, including tree seedlings and saplings); and a 10 m radius circular plot was used to sample trees (>12.5 cm diameter at breast height and $\geq$6 m tall).

Scientific names were recorded for all species found within each plot, and absolute percent cover of each species was estimated as close as possible to the following categories: 1%, 5%, 10%, 20%, 35%, 50%, 65%, 80%, 90%, 99%, or 100%. The estimate was made of the undisturbed canopy of all plant species that fell within the plot, even if plants were rooted outside of the plot. No effort was made to adjust for discontinuities in the canopy of species with open growth habits or in the coverage of small floating-leaved species such as *Lemna* and *Wolffia*. Because species can overlap each other, the sum of cover percentages often exceeded 100%. The estimates included only vegetation that was visible. The percent cover of submerged species were therefore often not recorded, but submerged vegetation was noted as being present. Both members of the field crew discussed cover percentages for each species in each plot and together agreed on an estimate. Unknown plants were collected, coded, and pressed for later identification. Professional botanists who identified unknown plants are listed in Appendix B.

For herbaceous plots (1-m^2), the strata was the cover type that predominated in the plot. Strata types were emergent-*Typha*, emergent-*Scirpus*, emergent-other dominant, emergent-mixed species, submerged, floating-leaved, and open water. The strata type was scrub/shrub for 5-m^2 plots and forested for 10 m radius circular plots. In addition, each species observed in plots was assigned to one of the following structural types (or layers):

- submergent
- emergent (or herbaceous)
- scrub/shrub
- forested
- floating-leaved
- dead (standing or fallen).

Invertebrate Sampling and Identification

A semi-quantitative dip-net sampling method was used for collecting invertebrates. Collection techniques were qualitative, but the picking of invertebrates from nets was timed so that the numbers of invertebrates could be expressed per unit time and in relative abundances. This approach has been used in various forms to make general assessments and to determine relative abundance of the taxa of aquatic insects (e.g., Plafkin et al. 1989, Merritt and Cummins 1984, Tucker 1958, Smith et al. 1987, Brooks and Hughes 1988, Jeffries 1989, Voights 1976). The semi-quantitative method was chosen because the objective of the pilot study was to determine richness and relative abundance of taxa found at the time of sampling. Study objectives did not require statistical comparisons among sites or sampling points, so quantitative samples per unit area were not necessary. The semi-quantitative net method requires less time, labor, and equipment and has been shown to sample more taxa than quantitative methods such as Hester-Dendy samplers and sediment cores (Peter M. Wallace, personal communication, Environmental Consultants, Gainesville, FL).

Sample points were distributed among the wetland cells and within major vegetation strata. Locations of invertebrate sampling points are shown on the site maps in Appendix A. Because both WTS were large, not all wetland cell/habitat combinations were sampled. Where several adjacent cells supported similar plant communities, a subsample of cells and habitats was arbitrarily selected for sampling.

At the Orlando site, invertebrates were sampled in only half of the first ten cells because the plant communities of those cells were very similar. Samples were also collected from cells 13, 14, and 17 to characterize invertebrates in the mixed plant and forested communities. Two field crew members sampled each cell/habitat simultaneously. Effort was divided between the two people by dividing areas to be sampled in half. At the Lakeland site, invertebrates were sampled in all cells except cell 6. In all except cell 1, invertebrates were sampled from a raft because the substrate was too unstable to walk on and the water was usually over 5 feet deep. One person netted invertebrates, primarily in the upper water column around plant stems, while the other person steadied the raft.

Sweeps were made with a rectangular kick net (#30 mesh) along the wetland bottom (Orlando site only), around plant stems, and along the surface where floating-leaved species were present. After several sweeps with a kick net in one habitat, contents of the nets were placed into enamel pans, a timer was started, and invertebrates were picked out by hand or with forceps. Specimens were placed into 95% ethyl alcohol preservative in prelabelled glass jars. When all individuals had been picked from the sample, the timer was stopped while a new net sample was obtained. The two field crew members picked invertebrates for a total of 30 minutes, which resulted in a 1-hour collection period for each sampling point.

Invertebrates were shipped to the University of Minnesota-Duluth for identification (Appendix C). Collection jars were emptied into a glass pan and sorted by life stage and order/family. Individuals were identified to family and genus using a microscope and the taxonomic keys listed in Appendix C. Each genus was placed in one of the following functional groups: shredder, collector, predator, scraper, and piercer (Merritt and Cummins 1984). In some cases, Merritt and Cummins list two functional groups for a genus so both were specified when data were recorded. All functional groups except piercers are defined by Vannote and others (1980). Merritt & Cummins (1984) define piercers as insects that suck unrecognizable fluids from vascular hydrophytes. Functional groups were not assigned to terrestrial invertebrates or to immatures that could be identified only to family.

Invertebrates of the class Oligochaeta (aquatic earthworms) were keyed only to family based on external characteristics and were counted by totaling the terminal ends collected and dividing by two. Functional groups were not assigned to Oligochaeta. Chironomids were divided into groups based on external features. A few individuals from each group were then mounted on microscope slides for identification to genus. The total count for each genus was the total in the group. Partial invertebrates were counted if a head was present with the exception of snails for which whole shells were counted regardless of whether the animal was present.

Whole Effluent Toxicity Testing

One-liter water grab samples were collected at the inflow, where water had not yet received wetland treatment, and at the outflow, after water had undergone advanced treatment in the WTS (collection locations are shown in Appendix A). The purpose of the two sampling locations was to determine whether toxic substances were entering the WTS and, if so, whether they were being removed from the water during residence in the WTS. Samples were shipped in ice to the Environmental Research Laboratory in Duluth (ERL-Duluth). Samples arrived at the laboratory for acute and chronic whole effluent toxicity tests the day after they were collected ($\leq$ 36 hours). The purpose of the tests was to identify sites where toxicity might be a problem so that more intensive studies could be done in the future if necessary.

Standard laboratory operating procedures of the National Effluent Toxicity Assessment Center (ERL-Duluth) (US EPA 1988b) were used for making routine measures and for conducting toxicity tests. At ERL-Duluth, water samples underwent the following routine measurements for whole effluent toxicity testing: alkalinity, hardness, ammonia, total residual chlorine, and temperature.

Chronic toxicity tests were conducted over a period of 7 days with renewal of test solutions every other day. Lake Superior water was used for a performance control, and undiluted influent and effluent samples from the Florida WTS sites were tested. Aliquots of each sample were slowly warmed to 25° C prior to use. *Ceriodaphnia dubia* (water flea) six hours old or less were obtained from the ERL-Duluth culture. Ten replicates for each sample and the control were used. Each replicate contained one organism in 15 ml of test solution in a 1-oz. polystyrene plastic cup. Block randomization was used. The *Ceriodaphnia dubia* were fed daily with 100 uL of a yeast-cerophyll-trout food mixture and 100 uL of algae, *Selenastrum capricornutum*. Initial measurements of pH, temperature, conductivity, and dissolved oxygen were taken after each sample was warmed and prior to each renewal. The mean young produced per original female and the mean percent survival were recorded after seven days.

Fathead minnows (*Pimephales promelas*) 24 hours old or less were obtained from the ERL-Duluth culture for acute tests. Two replicates for each sample and the control (Lake Superior water) were used. Each replicate contained ten fish in 15 ml of test solution in a 1-oz. polystyrene plastic cup. The test was not renewed and the fish were not fed. The mean number of surviving minnows was determined after 96 hours and expressed as a percentage of the total at the beginning of the test. Fathead minnow tests were not conducted on the Lakeland water samples.

Bird Surveys

Data on waterbird use of the wetlands for foraging and nesting were acquired from surveyors from the University of Florida in Gainesville (Appendix B). Low altitude (60 m) aerial surveys were conducted once per month in October and December, 1991, and in February, March, April, and May, 1992. In addition, surveys were conducted at some of the relatively undisturbed natural wetlands in the St. Johns River marshes just to the east of the Orlando site to look for differences in waterbird use between the WTS and the natural wetlands. No similar natural wetlands were located near the Lakeland site for making comparisons. Surveys were scheduled to provide information about seasonal variation in waterbird use of the WTS. Surveys were conducted between 0700 and 1200 EST as close to the 15th of each month as weather and aircraft availability would permit. The Orlando site was always surveyed first, the St. Johns marshes second, and the Lakeland site last.

The number of foraging and nesting wading birds seen at each WTS was estimated on each survey flight. Surveys were conducted on east-west transects to provide 100% coverage of the sites. Transect boundaries were pre-determined to avoid double counting of birds on successive transect passes. Two observers conducted surveys from a Cessna 172 aircraft at an altitude of 200 feet. Observers sat on opposite sides of the aircraft and counted birds on their side. Each observer tallied the number of each wading bird species seen on each transect, and tallies from both observers were summed over all transects. Large flocks of mixed species were circled and counted several times to confirm the numbers before continuing on the transect. Waterbirds in colonies at the WTS were counted on March, April, and May aerial surveys. Counts were confirmed by repeatedly counting nesting birds at both low and high altitudes. Wading birds that could not be distinguished to species from flying altitude (e.g., snowy egrets and immature little blue herons) were lumped together as unidentified small white herons, following the methods of Hoffman et al. (1990).

In addition to aerial surveys, wading birds were followed on foraging flights from colonies at the Lakeland and Orlando WTS sites to examine the relative importance of the WTS to breeding wading birds. Randomly-selected breeding birds from colonies were followed to the first site at which they landed and foraged. Following was done by one observer and the pilot in a Cessna 172. Locations of foraging sites were noted using U.S. Geological Survey topographic maps and, on most flights, a Trimble TransPac Global Positioning System with a 100-m accuracy. Following flights concentrated on white ibises, great egrets, and snowy egrets.

Dave Hill, the site manager at Lakeland, conducted five ground counts at the Lakeland site between November 1991 and June 1992. Counts were made during the same period as the aerial surveys. Although only one of the counts was done on the same day as an aerial count, the ground counts provide a cursory field verification of aerial counts of breeding wading birds, as well as information on the presence of species not visible from the air. All individuals of each bird species were counted between 0900 and 1200 hours from fixed survey points along the dikes in ten designated areas, comprising approximately 15% of the total site area. The breeding islands in cell 5 were given the most complete areal coverage. These areas were also surveyed on every date, whereas other areas were surveyed on only two or three of the dates.

Site Morphology

Color infrared photographs were taken of each site in summer 1991 by a local aerial survey company (Appendix B). Photos overlapped with a scale of approximately 1:5000. Photos were encased in mylar and the major cover types at each site were hand delineated on the mylar and labeled. Delineation varied depending upon the plant communities present and which could be consistently resolved based on photographs and ground truth mapping done during reconnaissance. Table 4 lists the cover types delineated at each site. Dikes were considered upland and were not delineated on photos.

When vegetation was sparse but all of the same type, small interstitial gaps in cover were ignored and the area was delineated with only one polygon. If two vegetation types were distributed evenly over the same area, the polygon was labelled as both types and the area was counted twice. This often occurred when floating-leaved plants formed a solid cover over the water surface within a sparse stand of *Typha* or *Scirpus*. Therefore, the sum of the areas of different vegetation types at a site can exceed the total vegetated area. Polygons were electronically digitized. Data were entered into the ARC/INFO Geographic Information System (GIS) and estimates were calculated for the indicators listed in Table 3 (B). Calculations are described in the Data Analysis section below.

Table 4. Cover types delineated on air photos.

Orlando	Lakeland
Typha spp.	*Typha* spp.
Scirpus spp.	Upland/Grasses
Other emergent	Dying willows
Scrub/shrub	Scrub/shrub
Forested	Forested
Floating-leaved	Floating-leaved
Dead trees	Dead shrubs
Submerged	Live/dead shrubs mixed
Open water	Open water
	Bare ground

Acquisition and Use of Existing Data on Water Quality

Under state and federal regulations, constructed wetland operators are required to sample certain water quality parameters routinely to demonstrate compliance with standards set for discharge to streams. Managers at some sites acquire data beyond what is required and most acquire data on influent to, as well as effluent from, the wetland for their own performance records. Table 5 shows the parameters for which data were available at each site.

At the Orlando site, water samples are collected on three successive days per month at ten collection stations on the site. Data for 1989 and 1990 from the influent station in cell 1 and the effluent station in cell 17 were summarized for this study. When data were recorded as less than the detection limit, the number half way between zero and the detection limit was used. For fecal coliform bacteria counts, however, data were entered as whole numbers, so an entry of <1 was considered to be zero. For two samples, fecal coliform counts were entered as <10, and both entries were deleted from the data set before conducting analyses. When a parameter was not measured (e.g., due to instrument malfunction), data were entered as missing.

Table 5. The types of water quality data obtained from each site. Ph=Ph (standard units); DO=dissolved oxygen (mg/L); BOD=biochemical oxygen demand (mg/L); TSS=total suspended solids (mg/L); NH3-N=ammonia nitrogen (mg/L); TKN=total Kjeldahl nitrogen (mg/L); TP=total phosphorus (mg/L); TFC=total fecal coliforms (# colonies per 100 Ml).

Site	Parameter							
	pH	DO	BOD	TSS	NH3-N	TKN	TP	TFC
Orlando		x	x	x	x	x	x	x
Lakeland	x	x	x	x	x	x	x	x

At the Lakeland site, water samples are collected three to five times per month at eight stations within the WTS, and monthly averages are calculated for each station and water parameter. Monthly averages from the influent and effluent stations from January 1990 through July 1991 were summarized for this study. Fecal coliform bacteria data were available only for 1991.

Samples from the Orlando site are analyzed by the City of Orlando Bureau of Wastewater Laboratory, Orlando, FL; those from the Lakeland site are analyzed by the City of Lakeland Wastewater Treatment Laboratory, Lakeland, FL (Appendix B).

Data Analysis

Vegetation, invertebrate, and site morphology data were summarized by calculating descriptive statistics for each WTS and for the cells within the WTS where data were collected. Analysis of data by cell was intended to show potential patterns in indicator values along a wastewater treatment gradient.

Vegetation and water quality data were summarized using SAS, and invertebrate data were summarized using the Paradox database system. Air photo data were analyzed with the ARC/INFO geographic information system.

Water quality data from each site were summarized by calculating the mean, range, and standard deviation for each parameter (e.g., pH, total P, etc.) from the inflow and outflow points of the wetlands. Water quality indicators were summarized for all sampling dates included in the time frames specified in the subsection, Acquisition and Use of Existing Data on Water Quality above.

Vegetation data were analyzed for each site and for each cell where vegetation was sampled at a site. Species richness was defined as the total number of species sampled at a site. Average percent cover of a given plant species was calculated by summing cover estimates at all sample points and dividing by the total number of sample points. Structural diversity of vegetation was evaluated by counting the number of structural layers present at a site. Structural dominance was assessed by 1) calculating the average percent coverage of each structural layer per site and 2) calculating the percentage of species sampled belonging to each layer. Dominant species were determined by ranking all species at a site in descending order based on their average percent coverage and then summing the average percent coverage values for each species in order of the ranking until 50% was exceeded. All species contributing to the 50% threshold and any additional species with an average coverage of 20% or more were considered dominants.

Analyses on invertebrates were made by first totaling the number of individuals of each species from each sampling point. Percent relative abundance of invertebrate species at each site was calculated by totaling the number of individuals of each species and dividing by the total number of individuals of all species combined. The percent relative abundance of individuals belonging to each functional group was calculated similarly. The number of invertebrates collected per person hour was calculated for each cell and habitat type in which invertebrates were sampled. Species richness was defined as the total number of species collected at each site.

Bird densities were calculated by dividing the total numbers of wading birds seen on any given survey flight by the total area of each WTS. The area of each WTS was measured by scanning the WTS boundaries from topographic maps (1:24,000) and measuring the interior area using SPYGLASS software. Species richness was the total number of species detected on all surveys. Although cattle egrets are considered terrestrial, they were seen at all wetlands and were breeding at the Orlando site and were therefore included in survey counts.

Indicators were calculated from physical habitat features that had been digitized and entered into a GIS. Calculations were made for each entire wetland and for each cell within the wetland as follows:

- Wetland area was measured as the area within surrounding dikes.
- Distance of the land/water interface is the total length of shoreline in a wetland and is a measure of shoreline irregularity or development; for this calculation, the area of floating-leaved plants was considered water.
- Length of shoreline (land/water interface) was divided by wetland area to normalize the shoreline irregularity estimate.
- Distance of cover/cover interface is the length of edge between cover types and is a measure of cover type interspersion.
- The length of edge between different cover types was divided by wetland area to normalize the estimate of cover type interspersion.
- The area of open water (no vegetation) was divided by vegetated area (including floating-leaved plants) to obtain an index of the relative amounts of the two cover types.
- Relative coverage of selected cover types (Table 4) was calculated by dividing the area of each cover type by the total wetland area.

Survival and reproduction in whole-effluent toxicity tests were tested against the controls using Dunnett's multiple t-test for the chronic tests and a t-test for the acute tests ($P<0.05$).

Comparison Data from the Literature

The indicator values obtained from the two WTS were compared to data from non-WTS obtained from the literature to put the information from WTS in the context of what was known about wetlands in the region. Comparison data were obtained for plant species richness, percent cover, invertebrate genera richness, surface water quality, and bird species richness and density. Data from WTS and non-WTS were compared to get a preliminary idea of where the indicator values for WTS lie in relation to the range of indicator values from other types of wetlands. Data from palustrine systems, primarily marshes, in the southeastern United States were used for comparisons. Comparison wetlands were natural, created, restored, and enhanced. No further attempt was made to match comparison sites to the WTS sites studied. Comparisons were intended to be very broad and preliminary and to identify any large differences in indicator values between WTS and non-WTS.

Comparison data were obtained from published documents and personal communication or records from the southeastern United States. A library search produced a few journal articles and agency reports, but many published reports did not contain the detailed data required for summarizing the indicators of interest, and it was difficult to find data on many specific indicators. Therefore, regional scientists and resource managers were contacted directly and asked to provide relevant data.

Quality Assurance

Three types of indicator data were used during this study: (1) data collected in the field (vegetation, invertebrates, bird use, whole-effluent toxicity); (2) data derived from maps and aerial photographs (site morphology); and (3) existing data (water quality) (Table 2). Laboratory analytical data quality procedures and data quality objectives (DQOs) for whole effluent toxicity testing were based on the ERL-Duluth Quality Assurance Plans and Standard Operating Procedures (US EPA 1988b). Detailed quality assurance information was not available from bird surveyors. However, the same two observers were present during all flights, and they sat in the same positions during all flights. On each flight, the Orlando site was surveyed first, the St. Johns marshes second, and the Lakeland site last. This procedure was implemented to control the time of day effects within sites.

At all vegetation plots, both members of the field crew discussed cover percentages for each species in a plot and together agreed on an estimate. Precision and accuracy were assessed for identification and percent cover estimation of plants so that, in case the team members had to identify or estimate percent coverage separately, the quality control (QC) exercises would indicate the degree of precision and accuracy in estimates. Because solo work was unnecessary during the 1991 field season, all estimates were made by both crew members together. Evaluation of QC data was therefore not necessary for interpreting data from this study but was calculated as a reference for future studies, if needed.

The following procedures were used to collect and evaluate QC vegetation data. A QC check was performed at 10% of sampling plots to determine how similarly the two field crew members were estimating percent coverage and identifying species. The decision to make a plot a QC plot was usually made while sampling the plot just before it. Each person prepared a data sheet and estimated cover percentages separately without any interaction with the other crew member. Percent cover precision was computed by calculating the mean difference between percent cover values recorded by two team members for each jointly recorded species (i.e., recorded by both team members in the same plot). For each team member, percent cover estimates were summed across all QC plots, by species, for each species that was jointly recorded. Mean percent cover estimates for each species and team member were derived by dividing the percent cover sums by the number of QC plots in which each species was jointly recorded. The mean difference was simply the difference in the mean percentages for each team member. Cover precision for the site was the mean precision for all species. Plant recognition comparability was calculated by counting the number of species in each QC plot that were jointly recorded, dividing by the total number of species observed in each plot, summing the quotients, dividing the sum by the total number of plots, and multiplying by 100.

Because vegetation was not sampled at the Lakeland site, QC calculations were done only for the Orlando site. Plant recognition comparability was 91%, and the mean percent cover comparability was 97.3% Both values meet the data quality objective of 85% set prior to the study. It is recommended that the QA/QC exercises continue to be part of future field work so that, in the event that crew members must work alone, a record of the precision of data collected will be available.

Data QC was also performed in the laboratory at University of Minnesota-Duluth to check the precision and accuracy of the identification and counts of invertebrates. Contents of 10% of the sample jars (of sites combined) were re-identified and recounted by a second person. Subsequently, discrepancies were resolved through discussion and comparison of results obtained using different keys. Invertebrate identification comparability represents the number of taxa both people jointly observed and identified during the QC check. It was computed for each QC sample jar by calculating the ratio of invertebrate taxa jointly observed to the total taxa observed and multiplying by 100. Identification comparability for both sites combined was obtained by calculating the mean of all QC sample jars. The mean identification comparability for invertebrates was 96%. This value meets the identification comparability objective of ≥85% established prior to the study.

The reconnaissance portion of field work included vegetation mapping, which served as the best guide and accuracy check for delineation of cover types on aerial photos. One of the field crew members interpreted and delineated cover types on all photos so that precision was maximized.

Existing water quality data were evaluated to determine the usefulness of water quality variables as indicators, not to draw conclusions about constructed wetland performance or to use the data in subsequent analyses. Standard operating procedures and quality control procedures were obtained from the laboratories that analyze water samples collected at the constructed wetland sites. All laboratories follow specified protocols for sample handling and custody, calibration, analytical procedures, preventative maintenance, and data reduction and validation. The laboratories also incorporate QC checks into analyses using duplicates, spiked samples, split samples, external performance standards, internal standards, reagent checks, and calibration standards. Protocols and data quality objectives, however, varied among labs. Because data from WTS sites were not being compared, data were used regardless of laboratory protocols and measurement consistency among testing labs.

RESULTS AND DISCUSSION

Summary data are presented separately for each indicator group for each WTS. Discussion addresses 1) indicator suitability for future research, 2) wildlife habitat quality, based primarily on comparisons to non-WTS data from the literature, and 3) recommendations for follow-up studies. It is recognized that study methods (e.g., sample design and intensity), wetland size, and various other factors confound comparisons with literature data. Comparisons, however, are used simply for establishing a context for making general postulations about the ecological condition of the two WTS studied and for generating hypotheses for future research.

Vegetation

Because much of the Lakeland site was inaccessible for sampling, vegetation was sampled quantitatively only at the Orlando site. Plants sampled at the Orlando site belonged to all six structural layers identified: emergent, submergent, floating-leaved, scrub/shrub, forested, and dead. The emergent layer was the most dominant, with an average percent coverage of 81% and containing 67% of the species sampled at the site (Table 6). The floating-leaved and dead layers were also common, with average percent coverages of 52% and 34%, respectively. Although the scrub/shrub layer had one of the lowest percent coverages, it had the second highest species richness (Table 6). Conversely, the high percent cover of floating-leaved species was due to only 11% of the species sampled. The dead category was composed primarily of persistent emergent vegetation (*Scirpus* and *Typha*). Dead vegetation was evaluated separately because it can contribute cover for waterfowl or nesting habitat for passerines that is different from cover of live plants of the same species.

Structural types were the most diverse and well-interspersed in the mixed marsh hardwood swamp habitats at the Orlando site (cells 13-17, Fig. 1)). These cells are designed and managed to provide habitat, and the varied vegetation structures result in habitats for a variety of species and activities (e.g., feeding, roosting, nesting). The deep marsh areas (cells 1-12) were designed primarily for water treatment and are comprised predominantly of bulrush and cattails with interspersions of floating-leaved species, shrubs, and a few palm groves. Vegetation structure in those cells is more uniform, but scattered openings in an otherwise dense growth of vegetation provide protected areas and, in a few areas where snags or shrubs are present, serve as bird rookery sites.

Wildlife use of a habitat for nesting and cover is usually considered to be more dependent on the structure of vegetation than on the species of vegetation (Beecher 1942, Weller and Spatcher

Table 6. Percent of the total number of plant species (out of a total of 63 species) and average percent cover per square meter ($\pm$ one standard deviation) comprising each vegetation structural layer at the Orlando site, 1991. Species of dead vegetation were not determined, so the percent of species column is blank for dead vegetation. Percent coverage can exceed 100 since structural layers can overlap.

Structural layer	Percent of species	Average percent cover/m^2
Emergent	67	81 $\pm$ 78
Submergent	10	12 $\pm$ 28
Floating-leaved	11	52 $\pm$ 64
Scrub/shrub	11	8 $\pm$ 24
Forested	3	7 $\pm$ 18
Dead	--	34 $\pm$ 37

1965, Swift et al. 1984). Well-interspersed vegetation structures are often associated with high diversity and abundance of wetland dependent birds. Complex plant zonation results in an increase in the number of niches available for breeding birds (Swanson and Meyer 1977, Weller 1978, Dwyer et al. 1979, Ruwaldt et al. 1979, Roth 1976). The diversity and interspersion of plant community structures in cells 13-17 at the Orlando site likely satisfy the needs of many wetland-dependent wildlife species.

The particular species of plants are important when considering wildlife food preferences. Plant species sampled and their average percent coverages per square meter are listed for the Orlando site in Table 7. Standard deviations of average percent cover are high due to patchiness in species distribution, which often results in highly variable cover values. The interspersion and patchiness of plant species, however, can enhance wildlife habitat.

Dominant species are those whose percent coverages comprise the first 50% of vegetative cover on a site when ranked in descending order, plus any species whose coverages are 20% or greater. Dominant cover types at the Orlando site were dead emergents and *Lemna* spp. Species that were common but which were not classified as dominants were *Salvinia rotundifolia* (17.9%), *Typha* spp. (16.2%), and *Hydrocotyle umbellata* (13.7%) (Table 7). The majority of plant species identified had average percent site coverages of less than 1%. Dominant species varied to some degree from one cell to another. Dead emergents were dominant in all cells sampled. *Lemna* spp. was dominant in three of the four cells sampled, but comprised an average of only 2% of the cover in the hardwood swamp community of cell 16. Also dominant in cell 16 was *Hydrocotyle umbellata*, *Ludwigia peruviana*, *Panicum* spp. and *Salvinia rotundifolia*. Aside from dead emergents and several floating-leaved species, *Typha* spp. was dominant in cell 2, and *Scirpus californicus* was dominant in cell 9. Floating-leaved species are consumed by many species of waterbirds. *Typha* spp. and *Scirpus* spp. are important as cover for many species of birds but are consumed by few species of wildlife.

Species richness (the number of species sampled) at the Orlando site was 63. For comparison, the numbers of plant species at several non-WTS marshes in Georgia and Florida ranged from 9 to 68 (Table 8). In addition, plant species richness in 25 non-WTS Lower Mississippi River borrow pits ranged from 65 to 196 for the period 1981-1983 (Buglewitz et al. 1988). Brown (1991) reported plant species richness between 13 and 93 for 18 created and natural non-WTS in Florida in 1988. Erwin (1991) reported plant species richness values between 7 and 39 for wetland mitigation habitats in Florida. Plant species richness at the Orlando WTS was within the range of values reported for non-WTS. Post, Buckley, Schuh, and Jernigan, Inc., the City of Orlando contractor which conducts regular sampling on the site, has identified 150 plant and tree

Table 7. Frequency of occurrence and average percent cover per square meter ± standard deviation for each plant species sampled at the Orlando site, 1991. Frequency of occurrence is the percent of sample points at which each species was present. Average percent coverage was rounded to 0.0 if it was below 0.05%. The sum of percent coverages at a site can exceed 100 since species can have overlapping coverages. The total number of points sampled was 65.

Species	Frequency of Occurrence (%)	Average percent cover/m^2 +/Std. Dev.
Emergent		
Aeschynomene indica	2	0.2 ± 1.2
Alternanthera philoxeroides	3	0.7 ± 4.5
Alternanthera sessilis	2	0.0 ± 0.1
Andropogon virginicus	2	0.2 ± 1.2
Bacopa monnieri	5	0.2 ± 1.4
Bidens spp.	2	0.2 ± 1.2
Carex albolutescens	2	0.0 ± 0.1
Carex spp.	2	0.1 ± 0.6
Centella asiatica	12	0.7 ± 2.3
Conoclinium coelestinum	2	0.1 ± 0.6
Commelina diffusa	6	1.6 ±11.2
Cyperus haspan	2	0.0 ± 0.1
Cyperus odoratus	6	0.3 ± 1.5
Dichromena colorata	3	0.2 ± 0.9
Diodea virginiana	5	0.2 ± 1.2
Echinochloa spp.	2	0.0 ± 0.1
Eleocharis spp.	2	0.3 ± 2.5
Galium spp.	9	0.2 ± 0.9
Galium tinctorium	2	0.1 ± 0.6
Juncus effusus	5	0.5 ± 2.1
Ludwigia peruviana	6	5.6 ±22.4
Ludwigia repens	2	0.1 ± 0.6
Lycopus rubellus	3	0.2 ± 1.2
Mixed grasses	2	0.5 ± 4.3
Mikania scandens	9	2.8 ± 13.7
Myrica cerifera	2	0.8 ± 6.2
Panicum repens	25	8.5 ± 22.9
Panicum spp.	5	4.4 ± 20.4
Paspalum urvillei	5	0.4 ± 1.8
Phyla spp.	12	2.5 ± 11.7
Pontederia cordata	5	2.7 ± 14.7
Polygonum punctatum	12	1.6 ± 5.7
Rhynchospora spp.	2	0.5 ± 4.3
Sagittaria lancifolia	8	1.3 ± 8.2
Scirpus americanus	5	1.2 ± 8.1
Scirpus californicus	6	5.5 ± 21.9
Setaria geniculata	2	0.2 ± 1.2

Table 7, continued

Species	Frequency of Occurrence (%)	Average percent cover/m² +/Std. Dev.
Smilax bona-nox	2	0.1 ± 0.6
Typha spp.	34	16.2 ± 29.5
Unidentified emergent 1	2	0.1 ± 0.6
Unidentified emergent 2	2	0.0 ± 0.1
Unidentified emergent 3	2	0.2 ± 1.2
Submerged		
Ceratophyllum demersum	26	4.6 ± 14.5
Chara spp.	6	0.1 ± 0.2
Najas guadalupensis	11	2.3 ± 10.4
Utricularia foliosa	9	3.9 ± 15.4
Utricularia spp.	6	1.0 ± 6.3
Floating-leaved		
Azolla caroliniana	20	5.8 ± 18.7
Eichhornia crassipes	3	1.7 ± 12.4
Hydrocotyle ranunculoides	11	3.9 ± 15.6
Hydrocotyle umbellata	28	13.7 ± 27.5
Lemna spp.	54	26.7 ±35.7
Limnobium spongia	17	5.4 ± 18.3
Salvinia rotundifolia	51	17.9 ± 30.7
Wolffia spp.	3	1.3 ± 7.5
Scrub/shrub		
Boehmeria cylindrica	3	0.2 ± 1.4
Eupatorium capillifolium	6	0.8 ± 4.5
Eupatorium filamentosa	2	0.2 ± 1.2
Sabal minor	5	0.5 ± 2.6
Unidentified shrub	2	0.1 ± 0.6
Urena lobata	2	0.3 ± 2.5
Liquidambar styraciflua (seedling)	3	0.2 ± 1.4
Forested		
Nyssa sylvatica	3	0.5 ± 2.8
Sabal palmetto	15	6.4 ± 18.1
Dead		
Emergent	55	30.4 ± 36.9
Scrub/shrub	2	0.2 ± 1.2
Forested	18	1.7 ± 4.7

Table 8. Plant species richness at palustrine emergent non-WTS wetland sites in Georgia and Florida, 1983-1990. * Marsh communities sampled; + Herbaceous layer only.

Site	Species Richness	Collection Year	Source
Little Cooter Prairie, GA	12	1984	Greening and Gerritsen 1987
Mizell Prairie, GA	10	1984	Greening and Gerritsen 1987
Mack's Island, GA	10	1984	Greening and Gerritsen 1987
Rookery, GA	9	1984	Greening and Gerritsen 1987
Agrico Swamp*, FL	24	1983	Erwin and Best 1985
Agrico Swamp*, FL	26	1984	Erwin and Best 1985
Agrico Swamp West*, FL	61	1987	Erwin 1988
Agrico Swamp East*, FL	35	1990	Erwin 1990
West of K-6, FL	16	1990	Henigar and Ray 1990
Hookers Prairie, FL	29	1990	Donovan 1990a
Bradley Junction, FL	42+	1990	Donovan 1990b
Horse Creek, FL	68	1990	Wallace 1990

species at the Orlando site since its construction (personal communication, Post, Buckley, Schuh, and Jernigan, Inc., Winter Park, FL), which places the site above most non-WTS with respect to plant species richness.

Vegetation is one of the most significant components of wildlife habitat, and the continued development of indicators of vegetation for the assessment of habitat quality in WTS is recommended. The methods used in this study were effective for characterizing vegetative structure, and percent cover of plant species. Identification of unknown species by regional botanists assured accuracy and eliminated the need for later laboratory processing of samples. Sampling was possible during a single site visit at the peak of plant growth. Other researchers have also found vegetation indicators to be effective for wetland monitoring (e.g., Aust et al. 1988; Brooks et al. 1989; Brooks and Hughes 1988; Brown et al. 1989; US EPA 1983; Sherman, personal communication, J.D. White Company, Vancouver, WA).

Because the Orlando site was large, additional sampling time might have better characterized the diversity of plant communities. Adequate sampling time should be allowed, depending on site size and community diversity. Alternatives to transect sampling may be necessary for inaccessible sites, such as the Lakeland site. Some vegetation indicators can be obtained from aerial photographs, and field verification can then be accomplished with minimal time and effort during field visits.

Because structural diversity is an important component of wildlife habitat quality, future work could include development of methods for quantifying structure, particularly within the emergent category, which is usually dominant in WTS. Short and Williamson (1986) describe one method for measuring the relative structural diversity of terrestrial habitats using the Habitat Layer Index, originally developed for use with the Habitat Evaluation Procedure (HEP) (U.S. Fish and Wildlife Service 1980). It may be possible to test and develop methods such as this for wetland habitats. Evaluation of habitat quality should focus less on plant species richness and cover types. Species-specific information, however, can be used to extract various metrics such as the abundance of wildlife food plants or rare and sensitive plants.

Invertebrates

Eight and one-half person-hours were spent sampling invertebrates at the Orlando site, and six person-hours were spent at the Lakeland site. The total number of invertebrates collected was 785 (92.4 per person-hour) at the Orlando site and 1639 (273.2 per person-hour) at the Lakeland site. Forty-nine taxa were collected at the Orlando site, and 52 taxa were collected at the Lakeland site.

Table 9. Aquatic invertebrate taxa and their relative abundances at the Orlando and Lakeland Florida sites, 1991. Total number of invertebrates collected was 785 and 1639 at the Orlando and Lakeland sites, respectively. Functional groups were not assigned for terrestrial, pupal, and non-insect invertebrates.

ORLANDO SITE

Taxon	Count	% Relative Abundance	Functional Group
Ephemeroptera			
Baetidae - *Callibaetis*, naiad	1	0.1	Collector
Caenidae - *Caenis*, naiad	3	0.4	Collector/scraper
Odonata			
Libellulidae - *Erythemis*, naiad	42	5.4	Predator
- *Libellula*, naiad	2	0.3	Predator
- *Pachydiplax*, naiad	2	0.3	Predator
- *Perithemis*, naiad	1	0.1	Predator
Coenagrionidae - *Enallagma*, naiad	3	0.4	Predator
- *Ischnura*, naiad	9	1.1	Predator
- *Telebasis*, naiad	35	4.5	Predator
Hemiptera			
Belostomatidae - *Belostoma*	3	0.4	Predator
- immature, naiad	14	1.8	Predator
Corixidae - *Trichocorixa*	1	0.1	Predator
Gerridae - *Limnoporus*	3	0.4	Predator
Hebridae - *Lipogomphus*, naiad	2	0.3	Predator
Mesoveliidae - *Mesovelia*	24	3.1	Predator
Naucoridae - immature, naiad	37	4.7	Predator
- *Pelocoris*	46	5.9	Predator
Nepidae - *Ranatra*, naiad	2	0.3	Predator
Pleidae - *Paraplea*	1	0.1	Predator
Veliidae - *Microvelia*	2	0.3	Predator
Veliidae - *Microvelia*, naiad	3	0.4	Predator
Coleoptera			
Curculionidae - *Tanysphyrus*	1	0.1	Shredder
Dytiscidae - *Cybister*, larvae	4	0.5	Predator
- *Hydrovatus*	1	0.1	Predator
- *Laccophilus*	4	0.5	Predator
Haliplidae - *Peltodytes*	104	13.2	Piercer/shredder
Hydrophilidae - *Devallus*	2	0.3	Collector
- *Enochrus*	1	0.1	Piercer
- *Tropisternus*	7	0.9	Piercer/collector
- *Tropisternus*, larvae	8	1.0	Predator
Noteridae - *Hydrocanthus*	17	2.2	Predator
- *Hydrocanthus*, larvae	8	1.0	Predator
- *Suphisellus*	1	0.1	Predator
Scirtidae - *Scirtes*, larvae	2	0.3	Scraper/collector/shredder/piercer

Table 9, continued

Taxon	Count	% Relative Abundance	Functional Group
Diptera			
Ceratonpogonidae - *Palpomyia*, larvae	1	0.1	Predator/collector
Chironomidae - *Ablabesmyia*, larvae	6	0.8	Predator/collector
- *Chironomus/Einfeldia*	1	0.1	Collector/shredder
- *Glyptotendipes*, larvae	5	0.6	Collector/shredder
- *Nimbocera*, larvae	6	0.8	Collector
- *Polypedium*, larvae	4	0.5	Shredder/collector/predator
Culicidae - *Anopheles*, larvae	1	0.1	Collector
Stratiomyidae - *Odontomyia/Heriodiscus*, larvae	1	0.1	Collector/scraper
Tabanidae - *Chrysops*, larvae	1	0.1	Collector
Homoptera			
terrestrial	3	0.4	
Non-insects			
Class Crustacea			
Order Amphipoda			
Talitridae - *Hyalella azteca*	71	9.0	
Order Decapoda			
Astacidae - *Cambarellus*	26	3.3	
- *Procambarus*	3	0.4	
Palaemonidae - *Palaemonetes*	145	18.5	
Class Arachnoidea			
Hydracarina			
Aranae, terrestrial	1	0.1	
Class Gastropoda			
Pulmonata			
Ampullariidae - *Pomacea*	5	0.6	
Physidae - *Physella*	5	0.6	
Planorbidae - *Gyraulus*	36	4.6	
- *Helisoma*	66	8.4	
Class Hirundinea			
Rhynchobdelida			
Glossiphonidae - *Hellobdella*	2	0.3	

Table 9, continued

LAKELAND SITE

Taxon	Count	% Relative Abundance	Functional Group
Ephemeroptera			
Baetidae - *Callibaetis*, naiad	5	0.4	Collector
Caenidae - *Caenis*, naiad	2	0.1	Collector/scraper
Odonata			
Aeshnidae - *Anax*, naiad	1	0.1	Predator
Coenagrionidae - *Enallagma*, naiad	74	4.5	Predator
- *Ischnura*, naiad	113	6.9	Predator
- *Nehalennia*, naiad	13	0.8	Predator
- *Telebasis*, naiad	54	3.3	Predator
Libellulidae - *Erythemis*, naiad	8	0.5	Predator
- *Miathyria*, naiad	4	0.2	Predator
- *Pachydiplax*, naiad	15	0.9	Predator
Hemiptera			
Belostomatidae - *Belostoma*	9	0.5	Predator
- immature, naiad	36	2.2	Predator
Gerridae - *Limnoporus*	17	1.0	Predator
- *Rheumatobates*	2	0.1	Predator
Mesoveliidae - *Mesovelia*	29	1.8	Predator
Naucoridae - *Pelocoris*	55	3.4	Predator
- immature, naiad	37	2.3	Predator
Nepidae - *Ranatra*	1	0.1	Predator
- *Ranatra*, naiad	5	0.3	Predator
Veliidae - *Microvelia*	2	0.1	Predator
Lepidoptera			
Pyralidae - *Crambus*, larvae	1	0.1	Shredder
Coleoptera			
Curculionidae - *Hyperodes*	1	0.1	Shredder
- *Tanysphyrus*	1	0.1	Shredder
Dryopidae - *Pelonomus*	1	0.1	Collector/scraper
Dytiscidae - *Celina*	1	0.1	Predator
- *Cybister*, larvae	6	0.4	Predator
- *Copelatus*	1	0.1	Predator
- *Laccophilus*	3	0.2	Predator
Gyrinidae - *Dineutus*	1	0.1	Predator
Haliplidae - *Peltodytes*	2	0.1	Piercer/shredder
Hydrophilidae - *Derallus*	1	0.1	Predator
- *Enochrus*	11	0.7	Piercer
- *Helobata*	2	0.1	Collector
- *Paracymus*	2	0.1	Collector
- *Tropisternus*	7	0.4	Piercer/collector
- *Tropisternus*, larvae	1	0.1	Predator
Noteridae - *Hydrocanthus*	28	1.7	Predator
- *Hydrocanthus*, larvae	8	0.5	Predator
- *Suphisellus*	7	0.4	Predator
Scirtidae - *Scirtes*, larvae	64	3.9	Scraper/collector/shredder/piercer

Table 9, continued

Taxon	Count	% Relative Abundance	Functional Group
Diptera			
Chironomidae - *Chironomus/Einfeldia*, larvae	19	1.2	Collector/shredder
- *Endochironomus*, larvae	3	0.2	Collector/shredder
- *Glyptotendipes*, larvae	3	0.2	Collector/shredder
- *Larsia*, larvae	1	0.1	Predator
- *Polypedium*, larvae	2	0.1	Shredder/collector/predator
Culicidae - *Anopheles*, larvae	1	0.1	Collector
- *Uranotenia*, larvae	1	0.1	Collector
Stratiomyidae - *Odontomyia/Hedriodiscus*, larvae	12	0.7	Collector/scraper
pupa	1	0.1	
Non-insects			
Class Crustacea			
Order Decapoda			
Astacidae - *Cambarellus*	3	0.2	
Palaemonidae - *Palaemonetes*	299	18.2	
Order Amphipoda			
Talitridae - *Hyalella azteca*	495	30.2	
Class Arachnoidea			
Hydracarina			
Aranae, terrestrial	1	0.1	
Class Hirudinea			
Pharyngobdellida			
Erpobdellidae	6	0.4	
Class Gastropoda			
Pulmonata			
Lymnaeidae - *Lymnaea*	2	0.1	
Physidae - *Physella*	114	7.0	
Planorbidae - *Gyraulus*	10	0.6	
- *Helisoma*	35	2.1	

Non-insect invertebrates were dominant at both Florida sites, particularly at the Lakeland site (Table 9). Dominant non-insect invertebrates were primarily from the class Crustacea, order Decapoda, at the Orlando site and orders Decapoda and Amphipoda at the Lakeland site. *Palaemonetes* in the family Palaemonidae were the dominant decapods at both sites, comprising 18.5% of the Orlando site total and 18.2% of the Lakeland site total. At the Lakeland site, Amphipoda were also numerous and were composed entirely of *Hyalella azteca* of the family Talitridae (30.2% of the site total).

Coleoptera was the most numerous insect order. It was found in abundance only at the Orlando site and consisted primarily of *Peltodytes* in the family Haliplidae (13.2% of the site total). The majority of taxa collected at both sites had relative abundances less than 1%.

Aquatic insect orders not represented in Table 9 are Collembola, Plecoptera, Neuroptera, Megaloptera, Hymenoptera, and Trichoptera. Aquatic Collembolans have a spotty distribution and are most common in the early spring or late autumn (Pennak 1978). Plecoptera are usually associated with clean, cool running waters or large oligotrophic lakes (Merritt and Cummins 1984). Aquatic Neuroptera comprise only one family, the larvae of which are associated with fresh water sponges. Large numbers of these and the Megaloptera are rarely seen because they are short-lived and many species are nocturnal (Merritt and Cummins 1984). These characteristics may partially explain the absence of some aquatic insect orders in the WTS samples. Ephemeroptera numbers were low at both sites. Most Ephemeroptera prefer a high concentration of dissolved oxygen (Pennak 1978).

Many species of Chironomids tolerate the low oxygen conditions in wetlands (Adamus and Brandt 1990) and are often an important component of a wetland's macroinvertebrate community. Chironomid abundance and species richness was relatively low at both sites (Table 9), but benthic sampling was not conducted. Benthic sampling is recommended for future studies to assure accurate estimation of all invertebrate groups. Ratios of the number of invertebrate species tolerant of low oxygen to those that are intolerant have often been used to indicate ecological status of surface waters, and could be tested for use in wetlands (Adamus and Brandt 1990).

The number of invertebrates collected per hour is related to density. The highest collection rates occurred at the Lakeland site, particularly in cells 1 and 5 (Table 10). Macroinvertebrate abundance normally increases with increasing nutrient concentrations (Cyr and Downing 1988, Tucker 1958). High abundance in cell 1 is likely due to higher nutrient concentrations and high productivity nearer to the inlet of the wetland (see Table 17 in Water Quality section below). The high abundance in cell 5 could be caused by water enrichment by the colonial waterbirds that breed on the islands in that cell.

Table 10. Number of invertebrates collected per person-hour in each cell at the Orlando and Lakeland sites. Cells 3, 5, 7, 8, 9, 11, 12, 15, and 16 at the Orlando site, and cell 6 at the Lakeland site were not sampled.

ORLANDO SITE		LAKELAND SITE	
Cell	# invertebrates collected/hour	Cell	# invertebrates collected/hour
1	91.0	1	371.0
2	96.0	2	194.0
4	96.0	3	93.0
6	141.0	4	310.0
10	68.0	5	401.0
13	74.0	7	271.0
14	101.3		
17	67.0		

Table 11. Number of invertebrates collected per person-hour in each habitat type at the Orlando and Lakeland sites.

ORLANDO SITE		LAKELAND SITE	
Habitat type	# invertebrates collected/hour	Habitat type	# invertebrates collected/hour
Emergent-*Scirpus*	141.0	Emergent-mixed	282.5
Emergent-*Typha*	79.5	Emergent-*Typha*	255.0
Emergent-other	96.0	Scrub/shrub	310.0
Emergent-mixed	89.3		
Forested	64.0		

The high collection rate in cell 6 at the Orlando site may be more related to the habitat than the position of the cell. The sample was the only one at that site that was taken in a *Scirpus* spp. habitat, where the collection rate of 141/hr is noticeably higher than in other habitats at the site (Table 11). Invertebrate collection rates were also highest in the *Scirpus* spp. community at the Ocean Springs WTS studied in Mississippi (McAllister 1992).

Because the same habitats were not sampled in every cell, the effects of cell number and habitat type are confounded in these analyses. Although it is often difficult to find the same habitat types in every cell, a more sound experimental design and analysis of variance of the influences on invertebrate abundance should be attempted in future studies. Identifying the factor that explains most of the variance in invertebrate abundance or the plant community that is most suitable for both habitat and water treatment might provide information for designing WTS to enhance habitat quality.

Table 12 shows the percent relative abundance of invertebrate functional groups present at the Orlando and Lakeland sites. A total of 363 invertebrates (46.2%) at the Orlando site and 965 (58.9%) at the Lakeland site were not assigned functional groups. These invertebrates included immatures, terrestrial invertebrates, and non-insect invertebrates. The high percentage of unassigned functional groups was due to the high abundance of crustaceans at both sites (Table 9). Predators were the most dominant functional insect group at both sites (35.0% relative abundance at the Orlando site and 32.5% at the Lakeland site). At the Orlando site, the piercer/shredder group comprised 13.2% of the total, the result of a high abundance of *Peltodytes*. All other functional groups comprised less than 4% of the total at each site.

The distribution of invertebrates among functional feeding groups is difficult to evaluate because the functional evaluation of Vannote et al. (1980) is based on lotic systems. Wetlands are usually predator-based systems (Hoke Howard, personal communication, U.S. EPA Region 4, Athens, GA). Although no current protocol exists for evaluating the viability of a macroinvertebrate community of a wetland, comparisons of functional group composition between reference wetlands and the wetland in question are sometimes made to identify differences in community structure. In contrasting a reference wetland to another wetland, biologists in Region IV have observed the elimination in impacted wetlands of certain taxonomic groups such as amphipods and odonates (H. Howard, personal communication, U.S. EPA Region IV, Athens, GA). This kind of comparison might be considered for functional groups in future studies if reference sites are sampled simultaneously.

For comparison, genera richness of invertebrates varied from 25 to 41 in four non-WTS palustrine wetlands in North Carolina (MacPherson 1988). In addition, genera richness for invertebrates

Table 12. Relative abundances of invertebrate functional groups, Orlando and Lakeland sites, Florida, 1991. Terrestrial and non-insect invertebrates were not assigned functional groups.

Orlando Site

Functional Group	Relative Abundance
Not assigned	46.2%
Predator	35.0
Predator/collector	0.9
Collector/scraper	0.5
Collector	1.4
Piercer/collector	0.9
Piercer	0.1
Piercer/shredder	13.2
Collector/shredder	0.8
Shredder	0.1
Shredder/collector/predator	0.5
Shredder/collector/scraper/piercer	0.3

Lakeland Site

Functional Group	Relative Abundance
Not assigned	58.9%
Predator	32.5
Collector	0.7
Collector/scraper	0.9
Collector/shredder	1.5
Shredder	0.2
Shredder/collector/predator	0.1
Shredder/collector/scraper/piercer	3.9
Piercer	0.7
Piercer/shredder	0.1
Piercer/collector	0.4

in several Lower Mississippi River abandoned channel and oxbow palustrine wetlands in 1984 ranged from 8 to 28 (Lowery et al. 1987). Erwin (1991) reported invertebrate species richness between 7 and 44 for various sections and habitats in wetland mitigation sites in Florida. Genera richness for benthic invertebrates in several Lower Mississippi River borrow pit palustrine wetlands ranged from 7 to 29 in 1981 (Cobb et al. 1984).

The taxon level to which invertebrates are identified, the collection techniques, and the group of invertebrates collected (e.g., nektonic, benthic) vary, so comparison is difficult. Nevertheless, genera richness values of 49 and 52 for the Orlando and Lakeland sites, respectively, appear to be high compared to the range of richness values from non-WTS in the same region. Data on invertebrate abundance as determined with the Timed Qualitative Sampling Technique were not found for comparisons, so invertebrate abundance in the two WTS studied in relation to that in non-WTS could not be assessed.

Continued development of macroinvertebrates for habitat evaluation in WTS is recommended. In this study, the semi-quantitative sampling method was simple to implement, required minimal equipment, and could be adapted for difficult sampling situations, such as at the Lakeland site. It can easily be done in a variety of selected habitats during a single site visit.

Macroinvertebrates have been suggested as monitoring indicators by various other researchers (Brooks et al. 1989, Brooks and Hughes 1988, Brown et al. 1989, Schwartz 1987, US EPA 1983). Macroinvertebrates are important to habitat quality and system function because they serve as a major food source for waterbirds, fish, reptiles, and amphibians, and they are a critical link between primary production/detrital resources of systems and higher order consumers (Murkin and Batt 1987, Murkin and Wrubleski 1987). Because of their relatively low position on the food chain, invertebrates can serve as indicators of food chain function and its implications for higher organisms. Invertebrates are also less likely than birds or mammals to migrate from one wetland to another.

Further development of macroinvertebrate indicators should include standardization of collection methods, expansion of collection techniques (e.g., sampling for benthic invertebrates), looking for relationships between invertebrate abundance and bird use, adherence to a rigorous experimental design, and simultaneous sampling at reference sites.

The time and cost involved in identification might be a limiting factor in future monitoring work, and approaches should be explored for simplifying the identification process, such as sorting by gross morphological characteristics, order, or family. A courser level of sorting can often be done in the field after collection, does not require collection and laboratory identification (Robert Knight, personal communication, CH2M Hill, Gainesville, FL), and might provide sufficient information for assessing habitat quality. The abundance of specific taxonomic groups, such as Chironomids or Oligochaetes, could also be tested as indicators of environmental conditions at a site. Functional group data might be useful for comparisons with reference wetlands and for future development of protocols for assessment of invertebrate community viability in wetlands, but their usefulness as an effective indicator at this time is uncertain.

Whole Effluent Toxicity Tests

There were no statistically significant toxicity effects ($P<0.05$) at either site for the *Ceriodaphnia* acute or chronic tests or for the fathead minnow acute tests on the Orlando water sample (a fathead minnow test was not done for the Lakeland sample). Survival was 80% or more for all samples (Table 13). Measurements of each water sample performed by the Duluth Laboratory are shown in Table 14, and initial and final chemistries for water samples and the controls are shown in Appendix D.

Toxic heavy metals, primarily from industrial sources, and organic contaminants are sometimes present in municipal wastewater (US EPA 1984, Hicks and Stober 1989, Richardson and Nichols 1985). Their concentrations are typically reduced by approximately 30-95% in secondary treatment before entering a wetland (Richardson and Nichols 1985). In addition, most WTS do not receive water from industries. The wastewater treatment plant in Lakeland does receive wastes from industries in the area, and some heavy metals have been detected in the WTS. Some wetland influent silver, cadmium, and zinc concentrations exceed the Florida standards for class III waters, although the wetland brings the effluent averages into compliance with the standards (Post, Buckley, Schuh, and Jernigan, Inc. 1992). Although there is no indication that any of the average metal concentrations are increasing through time, the concentrations are of concern to the city of Lakeland.

The field sampling, sample transfers, and laboratory procedures involved in whole-effluent toxicity testing were successful in the pilot study. The method is feasible logistically, and the data are of high quality for identifying potential problems requiring further testing. Single, whole-effluent toxicity tests, however, will not be effective unless a contaminant is entering the wetland at the time of sample collection. To increase the probability of detecting potential contaminants, whole-effluent tests should be conducted on a routine basis and should be one of several initial assessments focussed on wetlands suspected as higher risks for the presence of contaminants, such as the Lakeland site. Another initial test could be the measurement of sediment concentrations of

Table 13. Reproduction and survival of *Ceriodaphnia dubia*.

Sample	Mean young/original female (95% confidence interval)	Mean Survival (%)
Orlando		
Influent	26.4 (20.2-32.8)	100
Effluent	25.6 (21.4-29.8)	100
Control	25.2 (18.5-31.9)	100
Lakeland		
Influent	19.9 (11.8-28.0)	80
Effluent	25.7 (19.1-32.3)	90
Control	20.9 (12.2-29.6)	100

Table 14. Measurements on water samples performed by ERL-Duluth immediately upon arrival of samples at the laboratory.

Sample	Hardness (mg/L as $CaCO_3$)	Alkalinity (mg/L as $CaCO_3$)	Ammonia $N:NH_3$ (mg/L)	TRC* (mg/L)
Orlando				
Influent	119	85	<1	0.03
Effluent	93	80	<1	0.02
Lakeland				
Influent	180	130	<1	0.04
Effluent	360	85	<1	0.70

* TRC=total residue chlorine

contaminants.

Suspect wetlands might be those where toxic substances or metals have been found in the past, where wastewater treatment plant user violations have occurred in the past, or where routine sampling suggests possible problems. Signs of possible problems might be a sharp reduction in invertebrates present, signs of stress or disease in birds that use the WTS, or a combination of indicator measurements that suggests a marked decrease in wetland integrity from one year to the next.

If contamination is detected in initial tests, a full examination should follow to measure tissue concentrations of contaminants in aquatic organisms, to determine whether bioaccumulation is occurring, and to relate tissue levels of contaminants to adverse effects on wildlife. A full examination, however, is a much more lengthy and expensive process than a general assessment of wildlife habitat quality and thus should be a separate activity. Results of these preliminary whole-effluent tests should not be used to evaluate wildlife habitat quality until a full examination of the contamination can be conducted and wildlife risks can be assessed.

Bird Use

This section provides summaries of: 1) wading birds surveyed on six surveys at the two WTS and at natural marshes near the St. Johns River; 2) breeding wading bird foraging use of WTS and surrounding wetlands based on following flight data; and 3) results of ground surveys conducted at the Lakeland site. Aerial bird surveys were designed to explore the range and variability of wading bird use of WTS and to compare bird use of nearby natural wetlands with WTS. Cursory ground surveys were conducted to provide ancillary data about use by species that were not surveyed from the air. Species composition and abundance were highly variable on a monthly basis. Between-site consistency was low, while differences within sites was high from month to month. Also, variation in numbers of each species at each site in the winter and spring periods was very high. Results therefore cannot be used to determine whether WTS are more or less attractive than natural wetlands. However, several observations are notable.

Aerial Surveys

A total of ten species of waterbirds were detected on aerial surveys at the Orlando site, and nine species were detected at the Lakeland site. Species richness varied from a low of three at the Lakeland site in December to a high of eight at both sites during the March, April, and May surveys (Table 15). Species richness was higher at the Orlando site than at the Lakeland site on four of the six surveys. Species richness, however, is difficult to estimate by aerial surveys, so the accuracy of the above figures is uncertain.

The highest counts of foraging and breeding wading birds at the two WTS sites occurred at the Lakeland site where breeding wood storks and great egrets were abundant during the March and April surveys in the cell 5 rookeries. Great egrets were also abundant in October at the Orlando site and in February and May at the Lakeland site. Wading bird densities at the WTS sites varied from a low of 0.04 birds/ha at the Lakeland site in October 1991 to a high of 0.71 birds/ha at the Lakeland site in April 1992 (Table 15).

In comparison, ten wading bird species were surveyed at the natural St. Johns marshes over the survey period. Species richness varied from four to nine for each survey. Foraging and breeding wading bird densities were higher at the St. Johns Marshes than at the two WTS for the December, February, and March surveys (Table 15). The high densities were largely due to flocks of white ibis. When white ibis are removed from the analysis, differences in overall and species-specific densities among the three sites are much less discernable (Frederick and McGehee, personal communication, University of Florida, Gainesville, FL). Densities varied from 0.02 birds/ha in October to 1.64 birds/ha in February. Among all three sites surveyed, densities of foraging and breeding birds combined were highest in April and May at the Lakeland site, due to large numbers of nesting wood storks and great egrets, and in October at the Orlando site due to large numbers of non-nesting great and snowy egrets.

In addition to surveyed species, the Iron Bridge site attracted large numbers of blue-winged teal, including 520 on the December survey. On the March survey, 45 mottled ducks were seen at the Orlando site, while 111 were seen at the St. Johns Marshes. Ducks were not found in large numbers at the Lakeland site. Cormorants and anhingas were consistently seen at both WTS during surveys, and roosting black and turkey vultures were noted at the Orlando WTS.

Wading birds formed colonies at both of the WTS but not in the St. Johns Marshes. Breeding at both sites was apparently successful, although nest checks and counts of immature birds were not conducted. The Lakeland site had the largest colonies, composed primarily of wood storks and great egrets. The highest colony counts at the site included 190 wood storks and 180 great egrets in March, and 272 wood storks and 173 great egrets in April. At the peak of nesting activity, surveyors counted between 145 and 188 wood stork nests, 155-173 great egret nests, 12 snowy egret nests, 8 white ibis nests, and 235 double-crested cormorant nests on the islands in cell 5 (the ranges of estimates result from differences between aerial and ground estimates). The wood stork

Table 15. Numbers and densities of wading birds detected during six aerial surveys at the Orlando and Lakeland sites, FL, 1991-1992. Areas surveyed: Orlando site = 498.4 ha; Lakeland site = 714.0 ha; St. Johns = 1675.0 ha. GE = great egret; SN = snowy egret; WI = white ibis; GBH = great blue heron; GI = glossy ibis; LBH = little blue heron; TCH = tri-colored heron; BCNH = black-crowned night heron; WS = wood stork; CE = cattle egret; UWH = unidentified small white heron.

Survey Date	Site	GE	SN	WI	GBH	GI	LBH	TCH	BCNH	WS	CE	UWH	Total	Density-Birds/ha
10-23-91	Orlando	107	88	6	1	0	30	17	0	0	13	5	267	0.54
	Lakeland	7	3	1	0	0	0	0	0	0	4	13	28	0.04
	St. Johns	22	7	3	5	0	0	0	0	0	0	2	39	0.02
12-17-91	Orlando	10	33	0	10	15	8	1	0	0	0	0	77	0.15
	Lakeland	66	66	0	2	0	0	0	0	0	0	0	134	0.19
	St. Johns	261	168	1034	50	26	42	24	11	0	4	0	1620	0.97
2-24-92	Orlando	42	11	59	11	0	1	3	1	0	0	1	129	0.26
	Lakeland	126	15	4	18	0	0	0	0	0	2	82	247	0.35
	St. Johns	182	214	2031	4	288	22	1	0	0	0	6	2748	1.64
3-18-92	Orlando	16	3	106	8	0	2	2	0	1	0	0	138	0.28
	Lakeland	22	11	8	2	7	4	0	0	12	25	0	91	0.13
	with colony counts	202	11	8	2	7	4	0	0	202	25	0	461	0.65
	St. Johns	141	89	860	6	82	2	5	2	40	0	0	1227	0.73
4-15-92	Orlando	38	26	35	5	1	1	4	0	19	0	1	130	0.26
	Lakeland	16	10	0	2	0	1	0	0	13	0	0	42	0.06
	with colony counts	189	27	2	8	0	1	0	0	285	0	0	512	0.72
	St. Johns	121	76	133	10	0	4	0	0	3	4	4	355	0.21
5-19-92	Orlando	18	4	9	8	0	4	0	0	0	4	8	55	0.11
	with colony counts	32	4	9	8	0	4	0	0	0	59	8	124	0.25
	Lakeland	37	29	9	1	0	1	2	0	2	21	8	110	0.15
	with colony counts	112	29	9	1	0	1	2	0	77	21	8	260	0.36
	St. Johns	43	19	307	3	5	7	4	0	6	39	5	438	0.26

and great egret nesting attempts at Lakeland were apparently successful. Although nest success was not monitored, many chicks of both species were seen fledging from the islands in late May and early June. The success of snowy egret nests is less certain.

During a separate May 7 observation at the Orlando site, 13 nesting great egrets were counted in a palm grove in the southwest corner of the site, and 5 nests were counted in palms in the northeast corner. The birds probably began nesting in the last two weeks of April. On the May 18 survey flight, 55 pairs of nesting cattle egrets were counted in the center of the WTS. Because the May survey was the last one conducted, it is not known whether this colony grew, but a late May initiation and early summer building of the colony would be characteristic for cattle egrets (Frederick and McGehee, personal communication, University of Florida, Gainesville, FL).

The persistence of breeding colonies at both WTS suggests that the sites are valuable as nesting habitat. Birds are attracted by the deep, permanent water at the WTS sites. At natural wetlands in central Florida, water depth is less predictable from year to year. With a dependable water supply, WTS may have great importance as breeding sites in an area where many breeding wetlands are being lost or annual conditions are variable.

When breeding birds are removed from the survey counts, the value of the St. Johns Marshes as foraging habitat is apparent. Foraging bird densities were higher at those marshes on four of the six surveys (Table 15; Figure 2). The high densities in the December, February, and March surveys are due primarily to large flocks of foraging white ibises. White and glossy ibis appeared to prefer the St. Johns marshes over the WTS sites for foraging (Figure 3). Very few white ibises were surveyed at the Lakeland site. Large flocks of shorebirds were also noted at the St. Johns sites but not at either of the WTS sites. Of the three sites surveyed, it appears that the shallower St. Johns Marshes are most effective in attracting shallow-water and moist soil foragers such as ibis and shorebirds (Frederick and McGehee, personal communication, University of Florida, Gainesville, FL). The St. Johns Marshes also appeared to be more important than the two WTS for foraging great and snowy egrets and wood storks (Table 15).

Although the St. Johns Marshes are used to a greater degree by foraging birds than the two WTS, the foraging wading bird densities at all three sites surveyed appear to be high in comparison with other tropical and subtropical wetlands that have been surveyed using similar methods (Figure 4). Two of these comparison wetland types, mangrove areas of the Everglades and the freshwater Everglades, are known to attract high concentrations and a large proportion of the southeastern wading bird populations (Bancroft 1989, Bancroft et al. 1992). When numbers of breeding birds are included in density calculations, the densities at the Lakeland

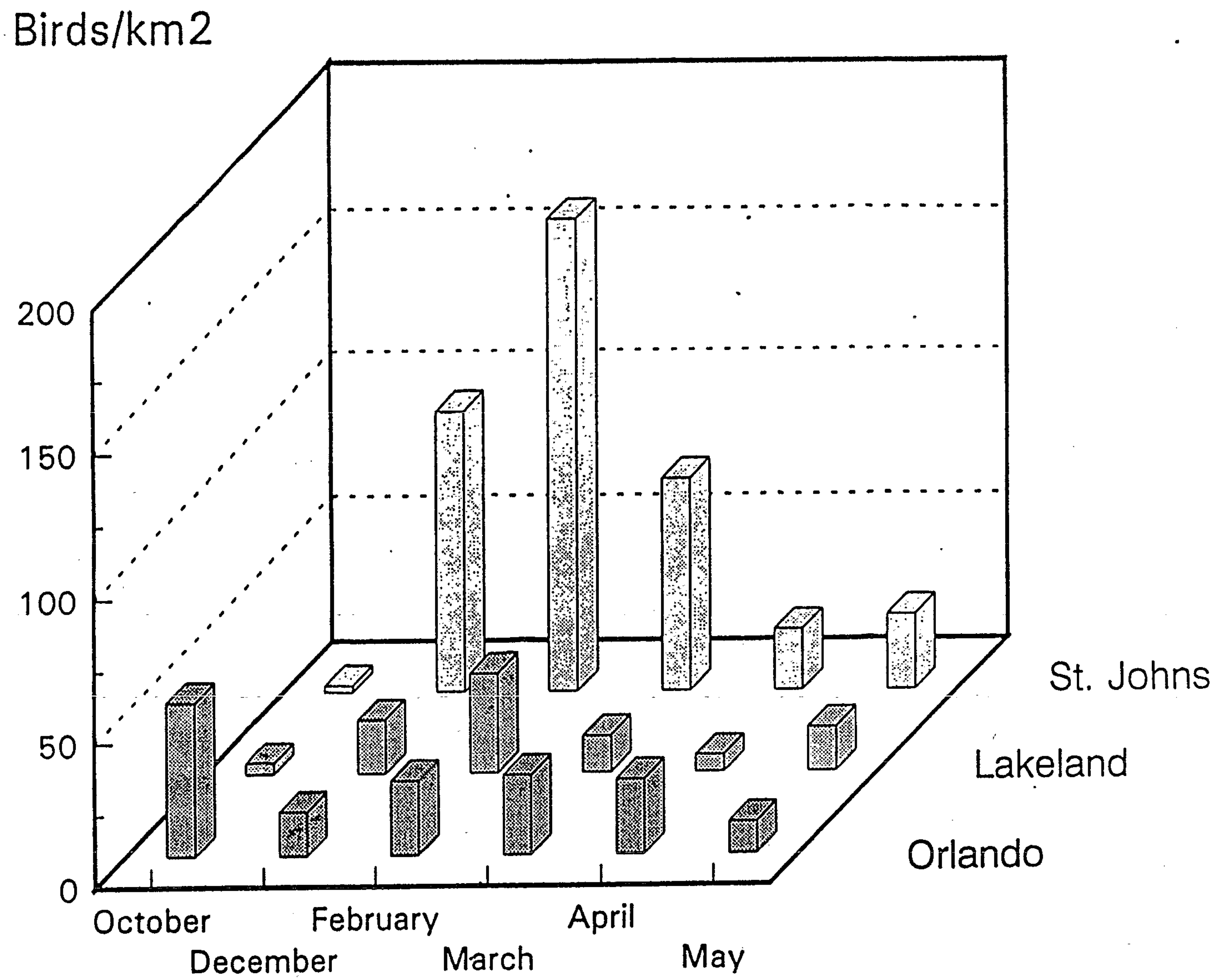


Figure 2. Monthly densities of foraging wading birds at the WTS and St. Johns sites in central Florida (adapted from Frederick and McGehee, personal communication, University of Florida, Gainesville, FL).

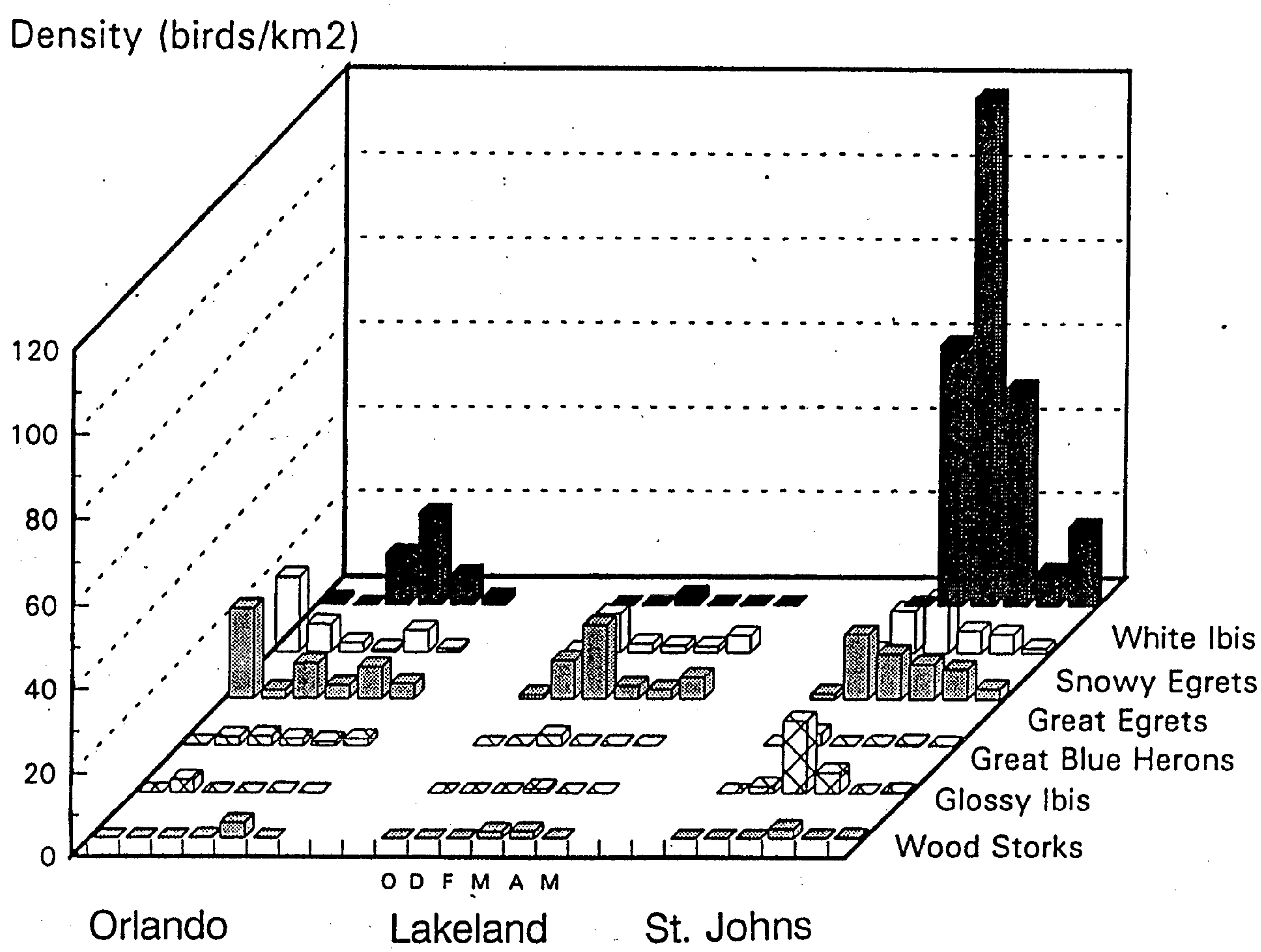


Figure 3. Comparison of densities of six foraging wading bird species at the WTS and St. Johns sites in central Florida (adapted from Frederick and McGehee, personal communication, University of Florida, Gainesville, FL). Letters on the X-axis indicate the months of surveys at each site.

site are much higher for the last three surveys and are more similar to the St. Johns Marshes, with the exception of the February survey. These observations suggest that the habitat value for foraging and breeding wading birds at the two WTS sites is high. The value of a wetland to wildlife is also greatly influenced by its position in the landscape and the habitat requirements that are available on a larger scale. The value of the Orlando site is enhanced by its position near the productive St. Johns Marshes because nesting birds are in close proximity to several important feeding areas.

Following Flights

Surveyors followed a total of 166 birds from colonies at the two WTS, Lake Mary Jane (27 km southwest of the Orlando site), and Homeland (14 km southeast of the Lakeland site). Most of the flights followed great egrets, snowy egrets, and white ibises.

Birds from the two natural wetlands did not travel to the WTS sites to forage. However, this was probably because sufficient foraging wetlands existed close to the natural wetlands or because the species of birds followed typically do not travel more than 20 km to feed (Frederick and Collopy 1988, Bancroft et al. 1991). All of the flights from Lake Mary Jane and Homeland were less than the distance from those sites to the WTS sites. Therefore, the results of following flights are inconclusive with respect to bird preference of WTS versus non-WTS.

The WTS sites were valuable as foraging areas to resident nesting great and snowy egrets. Foraging flight destinations were within the WTS sites for 65% of the flights at the Lakeland site and 46% of the flights at the Orlando site. Of the flights that ended elsewhere, the majority from the Lakeland site ended at phosphate pits and artificial ponds and ditches, while most from the Orlando site ended at natural wetlands. This is consistent with the types of wetlands available in the vicinity of each of the WTS sites.

Ground Counts and Ancillary Bird Data

Ground surveys, conducted independently of aerial surveys at the Lakeland site, provided cursory data on non-wading bird species richness and relative abundance. During the five ground surveys at the Lakeland site, 57 species were detected (Appendix E). Six additional species were detected during the field visit in August. Total birds counted in the 10 designated areas at the site ranged from a low of 574 in December, 1991, to a high of 1011 in June, 1992. The most abundant species on the first three ground counts (November, December, and January) were double-crested cormorants, which were seen mostly on the breeding islands in cell 5. On the

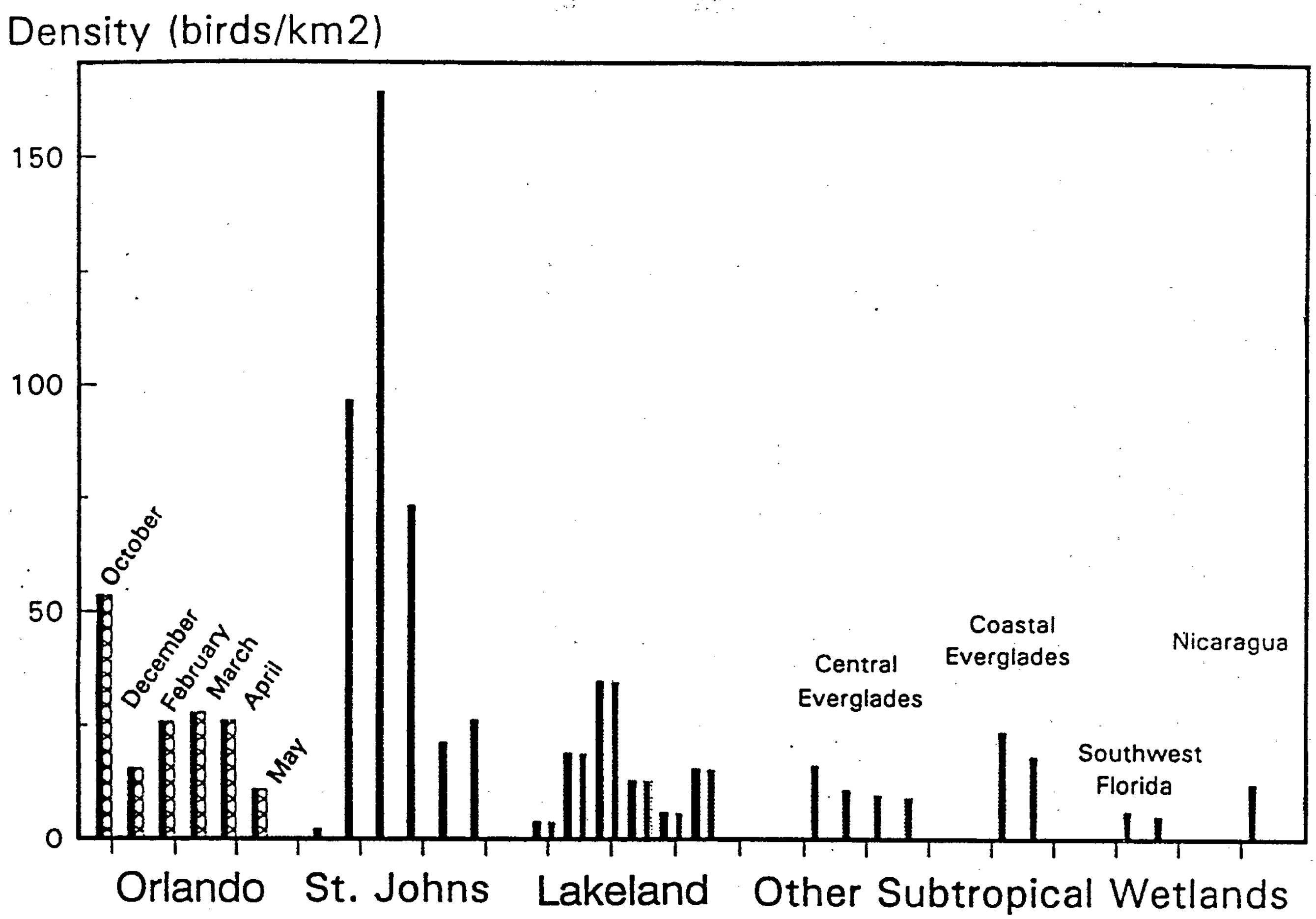


Figure 4. Comparison of non-breeding wading bird densities at the WTS and St. Johns sites with the results of similar surveys at subtropical non-WTS wetlands (adapted from Frederick and McGehee, personal communication, University of Florida, Gainesville, FL). Bars for the Orlando, St. Johns, and Lakeland sites represent months of the 1991 surveys, as shown for Orlando. Bars for the remaining sites represent results from multiple surveys. Data are from Hoffman et al. (1990) for the central Everglades, Jelks (1991) for southwest Florida, Robin Bjork and George Powell (unpublished, personal communication, P. Frederick, University of Florida, Gainesville, FL) for the coastal Everglades, and Peter Frederick and Marilyn Spalding (unpublished, personal communication) for the Miskito Coast of Nicaragua.

March and June surveys, the colonial nesting species -- wood stork, great egret, anhinga, snowy egret, and double-crested cormorant -- were most numerous.

Because ground and aerial counts were usually not simultaneous and because ground counts did not provide complete areal coverage of the site, it is difficult to use one type of survey to evaluate the accuracy of the other. However, it would be preferable to do this kind of evaluation in future studies to estimate the error associated with aerial surveys. The only ground count that was conducted on the same day as an aerial survey was the December 17 count. More species of wading birds were detected on that count than on the aerial survey, but fewer individuals of each species were counted. This suggests that species identification may be a problem on aerial surveys, while bird counts may be difficult from the ground when not all areas of the site are visible. If all areas of the site are not accessible on the ground, which is the case at the Lakeland site, one approach would be to obtain separate aerial counts in a smaller area that is also visible from the ground so that comparisons and error estimates can be made using both types of counts.

A total of 141 bird species have been recorded (as of 1991) for the Orlando site by Post, Buckley, Schuh, and Jernigan, Inc., the city of Orlando site contractor (Appendix E). The Orlando list represents a greater amount of time and effort devoted to inventory of the biota present at the site and is likely to be more complete than the Lakeland site list. Species seen during routine field work in August, 1991, are indicated on the lists.

At several southeastern palustrine non-WTS comparison wetlands, species richness ranged from 13 to 98 (Edelson and Collopy 1990, Henigar and Ray 1990, U.S. Army Engineer Mississippi River Commission, 1986). Richness at the Orlando site is above this range of values, while richness at the Lakeland site falls in the upper half of the range.

Bird Indicator Discussion

The intensity of bird use at the two WTS can probably be attributed to the large size of the sites, the availability and diversity of suitable habitats within the sites, and the observed high biological productivity. Both WTS have been stocked with fish, which is likely an important food resource for colonial nesting birds at the WTS, as evidenced by destinations within the WTS of many of the bird following flights. Sixteen species of fish have been observed at the Orlando site (personal communication, Post, Buckley, Schuh, and Jernigan, Inc., Winter Park, FL), including Florida gar, largemouth bass, bluegill, and bullhead. Apple snails (*Pomacea* spp.), food of the snail kite, were also very abundant. Bird species richness is high, which suggests a good habitat diversity and suitability. Benefits to wildlife from use of wastewater for habitat enhancement have been reported in numerous other cases (e.g., Cedarquist and Roche 1979, Cedarquist 1980a, 1980b, Demgen 1979, Demgen and Nute 1979, Wilhelm et al. 1988).

It is possible that bird use indicators may be useful for future habitat assessments, but the time and level of effort required for bird surveys should be considered. Birds are very mobile, and their use of a wetland may be erratic and/or seasonal. To adequately characterize bird use, multiple surveys, throughout at least the spring and fall migratory seasons, are necessary. Advanced planning must assure that necessary work contracts, personnel, and funding are arranged on time so that surveys coincide with the annual cycles of bird use.

Birds are more visible and audible than other faunal components and are easily identified by trained biologists, which makes bird use a relatively reliable measurement in many cases. Information on birds is sometimes useful for assessing other system components, such as the types of food resources that might be present in the wetland or the presence of habitat features required by certain species. Because birds are mobile and often use a complex of wetlands, most species might be better as indicators of overall landscape conditions than of single wetland conditions (Adamus and Brandt 1990).

If traditional bird surveys are continued in future studies and monitoring efforts, the following should also be considered:

- At least some effort should be devoted to ground counts to: verify aerial survey data; estimate the error associated with aerial counts; and survey other groups of birds that are not detectable from the air, such as passerines and shorebirds. Ground counts should be done immediately prior to aerial counts and should maximize areal coverage of a site. This level of effort, however, may be too great for sites as large as the two WTS visited in Florida.

- The amount of sampling effort that can be devoted versus that required to obtain an accurate representation of bird use, density, and diversity should be evaluated. If the level of effort possible is insufficient to make accurate estimates, then objectives should be re-evaluated, surveys re-designed, or resources allocated to other indicators.

- Experienced aerial surveyors might not always be available; if inexperienced surveyors are used, the quality of data may be questionable.

- Indicators such as bird activity (breeding, feeding, resting) and the presence of threatened, endangered, or keystone species should be considered to provide more information about the types of habitat present and its value to species of interest in a particular region.

- A plan for data integration and reduction should be designed for summarizing results of multiple surveys. Analysis by taxonomic group (e.g., waterfowl, shorebirds, passerines) or feeding guilds should also be considered for a more detailed assessment of habitat quality.

- Logistics and QA issues involved in coordinating bird surveys with other agencies, universities, or organizations, and conflicts that might arise due to diverging interests in the kinds of data collected should be anticipated.

- Surveys should be conducted at nearby non-WTS reference wetlands, or habitat quality criteria should be developed for use as a "gauge" when making habitat quality assessments.

A method for rapid estimation of the habitat importance of specific wetlands is currently being developed and tested by Adamus (1993) and may prove useful when many wetlands are being assessed on a regular basis. The procedure, which emphasizes biodiversity and an ecosystems approach, estimates the number of bird species likely to occur regularly in a particular wetland and uses this to assign importance to the wetland. Development of a procedure such as this for use in WTS may be a feasible alternative to traditional bird surveys.

Site Morphology

Diversity, abundance, and density of wetland-dependent animals is usually higher when vegetation and water are well-interspersed (Steel et al. 1956, Weller and Frederickson 1973). Weller and Frederickson (1973) concluded that marshes with 50-70 percent open water that is well interspersed with emergent vegetation (or a ratio of water to cover of 1.00-2.33) produced the highest bird diversities and numbers. Weller and Spatcher (1965) noted that maximum bird species richness and abundance occurred when a well-interspersed water:cover ratio of 50:50 (or 1.00) existed. Based on these findings, the most optimal ratios of open water to vegetated area occurred in the lake at the Orlando site (ratio of 1.59) and in cells 5 and 6 at the Lakeland site (ratios of 1.05 and 1.29, respectively) (Table 16). Some of the land:water ratios at the Orlando site and in cell 3 at the Lakeland site, however, may be biased low because areas of small floating-leaved plants were not considered to be open water when the GIS analysis was done.

Cells 1 and 2 at the Lakeland site were almost entirely vegetated, primarily with scrub/shrub.

The open water category primarily describes large expanses of open water with no vegetation (i.e., those that are visible on photos); it is not the total amount of water present. Waterbirds can use areas covered by small floating-leaved plants and areas under the canopies of shrubs, trees, and large emergent plants, such as *Typha* and *Scirpus*. At both of the WTS, surface water underneath other vegetation, but not visible on photographs, was sufficient to allow use by waterbirds for protection and feeding. These areas, particularly abundant in the forested areas at the Orlando site, provided habitat for a diversity of birds.

Land/water interface per hectare is a measure of edge. It is also another measure of the degree of interspersion of water and cover. Harris and others (1983) concluded that edge habitat is important to bird species diversity. Numerous dikes at both sites and islands at the Lakeland site contribute to the amount of edge habitat available. The amount of land/water interface in relation to wetland area, however, is relatively low, averaging only 107.0 m/ha at the Orlando site and 67.0 m/ha at the Lakeland site. This is a result of the large area of most cells at both WTS in relation to shoreline. Although no land/water interface data were available from non-WTS for comparison, the land:water interface of two WTS studied in Mississippi were 410 and 230 m/ha (McAllister 1992). These WTS had construction similar to the Florida WTS but were much smaller. The incorporation of peninsulas, islands, or additional cells in the design of large WTS would result in a greater amount of shoreline per unit area of wetland.

The interface between different cover types is another measure of interspersion and edge. Wetlands with moderate to high vegetation richness and interspersion can support a greater density and species richness of aquatic animals than those with low interspersion (Weinstein and Brooks 1983, Rozas and Odum 1987). Weller and Spatcher (1965) noted that many marsh bird species nested near water-cover interfaces or the interface of two cover types. At the two WTS, plant species were observed to be diverse and well-interspersed. Plant communities have been allowed to develop naturally at the Lakeland site. Plant communities in the mixed marsh and hardwood swamp communities at the Orlando site are managed to provide wildlife habitat diversity. The cover/cover interface in these communities at the Orlando site was 387.4 m/ha and 443.7 m/ha, respectively, which is higher than the deep marsh, the lake, and the overall site. The cover/cover interface per ha was also very high in cell 4 at the Lakeland site (499.1 m/ha) (Table 16). The cover/cover ratio in Cell 5, which contains the bird rookeries, was 373.1, which is also high relative to the overall site.

The large areas of both WTS indicate that the sites have

Table 16. Landscape data acquired from aerial photographs. Indicators are marked with an asterisk. Numbers in parentheses are percentages of total wetland area of the site or habitat type. Percentages were rounded to zero if less than 0.5%. Percent coverage of plants listed under the vegetated category can sum to more than the total vegetated percentage due to overlap of species.

ORLANDO SITE

	Whole Site	Deep Marsh	Mixed Marsh	Hardwood Swamp	Lake
1. Wetland area (ha)*	453.08	163.19	144.46	110.24	35.20
2. Cover areas (ha) (percent of wetland area)*					
a. Vegetated	385.37 (85)	157.49 (97)	108.78 (75)	105.55 (96)	13.57 (3)
Scirpus spp.	31.48 (7)	28.20 (17)	3.17 (2)	0.12 (0)	0.00 (0)
Typha spp.	218.22 (48)	94.63 (58)	40.17 (28)	79.37 (72)	4.06 (12)
Other emergents	29.78 (6)	2.73 (2)	22.37 (15)	4.50 (4)	0.18 (0)
Floating-leaved	106.38 (23)	68.71 (42)	28.05 (19)	9.62 (9)	0.00 (0)
Submerged	16.57 (4)	2.59 (2)	3.85 (3)	1.28 (1)	8.85 (25)
Scrub/shrub	25.37 (6)	11.45 (7)	6.15 (4)	7.67 (7)	0.11 (0)
Forested	33.04 (7)	5.84 (4)	18.32 (13)	8.71 (8)	0.00 (0)

(Table 16, continued)

	Whole Site	Deep Marsh	Mixed Marsh	Hardwood Swamp	Lake
b. Open Water	66.99 (15)	5.72 (4)	35.67 (25)	4.69 (4)	21.63 (61)
3. Land/water interface (m)	48,462	22,217	10,731	10,949	4565
4. cover/cover interface (m)	160,031	45,797	55,960	48,917	9348
5. Open water area:vegetated area*	0.17	0.04	0.33	0.04	1.59
6. Land/water interface: Wetland area (m/ha)*	107.0	136.1	74.3	99.3	129.7
7. Cover/cover interface: Wetland area (m/ha)*	353.2	280.6	387.4	443.7	265.6

LAKELAND SITE

	Whole Site	Cell 1	Cell 2	Cell 3	Cell 4	Cell 5	Cell 6	Cell 7
1. Wetland area (ha)*	459.91	70.07	61.80	148.16	25.08	93.76	28.47	32.57
2. Cover areas (ha) (percent of wetland area)*								
a. Vegetated	372.01 (81)	70.07 (100)	58.70 (95)	143.28 (97)	20.29 (81)	45.80 (49)	12.43 (44)	21.44 (66)
Typha spp.	134.29 (29)	15.64 (22)	1.43 (2)	98.00 (66)	0.39 (2)	18.15 (19)	0.00 (0)	0.68 (2)

55

Table 16, continued

	Whole Site	Cell 1	Cell 2	Cell 3	Cell 4	Cell 5	Cell 6	Cell 7
Floating-leaved	128.21 (28)	9.66 (14)	4.00 (6)	100.53 (68)	0.62 (2)	13.37 (14)	0.00 (0)	0.01 (0)
Scrub/shrub	190.25 (41)	47.43 (68)	44.22 (72)	41.62 (28)	19.58 (78)	19.17 (20)	8.18 (29)	10.05 (31)
Forested	23.80 (5)	4.02 (6)	6.29 (10)	2.77 (2)	0.06 (0)	3.46 (4)	7.20 (25)	0.00 (0)
Upland grasses	30.75 (7)	3.25 (5)	6.49 (10)	0.00 (0)	0.00 (0)	6.22 (7)	4.12 (14)	10.68 (33)
b. Open Water	84.78 (18)	0.00 (0)	0.00 (0)	4.88 (3)	4.79 (19)	47.96 (51)	16.04 (56)	11.11 (34)
c. Bare Ground	3.12 (1)	0.00 (0)	3.10 (5)	0.00 (0)	0.00 (0)	0.00 (0)	0.00 (0)	0.02 (0)
3. Land/water interface (m)	30,812	4085	4583	6050	3578	7459	2498	2559
4. Cover/cover interface (m)	108,617	12,872	12,315	21,043	12,518	34,982	6932	7955
5. Open water area:vegetated area*	0.23	0.00	0.00	0.03	0.24	1.05	1.29	0.52
6. Land/water interface: Wetland area (m/ha)*	67.0	58.3	74.2	40.8	142.7	79.6	87.7	78.6
7. Cover/cover interface: Wetland area (m/ha)*	236.2	183.7	199.3	142.0	499.1	373.1	243.5	244.2

potential to provide many of the habitat requirements of birds. Because wetlands are numerous in the southeastern United States, however, it is not essential that the WTS provide all the habitat requirements of wildlife. Large wetlands or complexes of wetlands types and upland areas may be necessary for fulfilling all wildlife needs or for attracting birds (Weller 1978). Birds, in particular, can move between different wetlands (i.e., within a wetland complex), using certain locales for nesting, feeding, roosting, and cover. The habitat value of the Orlando WTS, for example, is likely increased because of its setting adjacent to the St. Johns Marshes, a productive feeding area for wading birds in central Florida. Site morphology indicators might provide more information if calculated for a complex of wetlands in a watershed or within a chosen distance from the wetland in question so that single wetlands can be assessed in the landscape context and not as isolated entities. The landscape setting and its influence on wildlife habitat quality is also a very important consideration when choosing the most appropriate construction site for a WTS.

The use of aerial photographs for obtaining site morphology indicators is highly recommended for future assessment and monitoring. Aerial photography can easily be arranged through regional photographic companies as long as photo specifications are clearly defined. Photographs should be timed to coincide with field sampling of other indicators. This is often problematic, particularly in the southeastern U.S., because late summer haze obstructs visibility and photo quality. Photographers often will not fly on hazy days, which can potentially delay photography until autumn.

Physical habitat features such as shoreline length, amount of edge, ratio of open water to vegetated area, and vegetation interspersion and structural diversity are good indicators of habitat quality because their relationships to wildlife production and/or use have been demonstrated. Data interpretation is therefore facilitated by using guidelines found in the literature. Site morphology measurements can be obtained from maps or aerial photographs in a relatively short time and with less effort than field work. They can be taken in every wetland of interest, and replicate samples and assessment of variability are not necessary. Some field ground truthing of vegetation types, however, is necessary for air photo interpretation.

Aerial photos and maps can also be used to evaluate the larger landscape setting, which is of great importance in evaluating wildlife habitat. Photo interpretation and field sampling should be used interactively to maximize the information obtained. Estimation of the dominant structural layers can be obtained from photos while field work might focus on gathering data on cover types and species richness. Methods for evaluating vegetative structure using aerial photographs have been described (Short and Williamson 1986) and may be adaptable for WTS. One limitation of using landscape indicators is the high cost of aerial photography. Current existing photos, if available, may be an alternative.

Water Quality

Water quality can influence the biological components of wetland systems, such as plant and animal abundance and species diversity. Water quality data are presented for both WTS in Table 17. With the exception of total phosphorus at the Lakeland site, water quality values are generally within the range of values for non-WTS, and there is some indication that water quality at the WTS is superior to that in non-WTS in the southeastern U.S. Both WTS have achieved permitted effluent requirements (Jackson 1989). There is evidence from data collected at the Orlando site that nutrient concentrations lower than those attained through conventional advanced waste treatment processes can be achieved in WTS (Swindell and Jackson 1990).

Average TSS concentrations at the two WTS ranged from 5.02 to 8.10 mg/L. The lowest concentration recorded was <0.02 mg/L in the effluent at the Orlando site. Neither site ever had TSS concentrations over 21 mg/L, and the TSS concentrations were reduced from the influent to the effluent points (Table 17). For comparison, TSS concentrations in non-WTS wetlands near the lower Mississippi River and in created and natural marshes in central Florida ranged from 1.0-25.7 mg/L (Tables 18 and 19). Values at the WTS fell within the lower part of this range, and the lowest WTS concentration was lower than any of the values found for non-WTS. Wetlands that receive water with TSS levels less than 80 mg/L and never more than 200 mg/L are more likely to support a greater diversity and/or abundance of fish and invertebrates (Adamus, personal communication, ManTech Environmental Technology, Inc., Corvallis, OR). The two WTS clearly fall into this category.

Average DO concentrations at the WTS ranged from 1.47 to 8.09 mg/L (Table 17). The average concentration dropped by over 3 mg/L between the influent and effluent at the Orlando site and increased only slightly from the influent to the effluent ends of the Lakeland WTS. Average DO concentrations for non-WTS ranged from 1.0 to 12.1 mg/L (Tables 18, 20, 21, 22). Most non-WTS values, however, were between 2 and 8 mg/L, and some of the highest of those were recorded in abandoned channel and oxbow lakes (Table 18). The average effluent DO of 1.47 mg/L at the Orlando site is low in comparison to other wetlands in the region and does not meet the site permit requirement of 3.5 mg/L (Jackson 1989), even when the standard deviation is added. Dissolved oxygen concentrations of 2 and 4 mg/L are common in many Florida streams and swamps (Dierberg and Brezonik 1984, Friedemann and Hand 1989, Hampson, 1989). Consequently, low DO often naturally limits the richness of invertebrates (Ziser 1978) and fish (Tonn and Magnuson 1982) in wetlands. There was no evidence from this study, however, that the richness of aquatic life was limited at the Orlando site.

Table 17. Summaries of water quality data at the Orlando and Lakeland sites. I=influent; E=effluent; N=number of samples; SU=standard units; TSS=total suspended solids; DO=dissolved oxygen; BOD=biochemical oxygen demand (5-day); NH3-N=ammonia nitrogen; TKN=total Kjeldahl nitrogen; TP=total phosphorus; Fec.Col.=fecal coliform bacteria. Standard error was calculated at the Lakeland site because the values used to calculate means were monthly averages.

ORLANDO

Variable	I/E	N	Range	Mean	Std Dev
pH	Not measured				
pH	Not measured				
TSS (mg/L)	I	72	1.80-17.00	8.10	3.83
TSS	E	71	0.10-20.60	5.02	4.66
DO (mg/L)	I	51	0.20-11.60	4.72	3.38
DO	E	69	0.01- 6.00	1.47	1.68
BOD (mg/L)	I	71	0.30-32.20	4.57	4.62
BOD	E	71	0.20- 5.00	2.36	1.08
NH3-N (mg/L)	I	71	0.05- 8.74	2.36	1.97
NH3-N	E	71	0.00- 0.80	0.11	0.15
TKN (mg/L)	I	70	0.59- 9.10	3.15	2.07
TKN	E	71	0.32- 1.64	0.91	0.22
TP (mg/L)	I	71	0.15- 3.30	0.68	0.50
TP	E	71	0.02- 0.24	0.08	0.04
Fec.Col. (no./100 mL)	I	71	0.00-75.00	5.66	11.99
Fec.Col.	E	70	0.00-180.00	53.26	43.19

(Table 17, continued)

LAKELAND

Variable	I/E	N	Range	Mean	Std Err
pH (S.U.)	I	19	7.10- 7.50	7.33	*
pH	E	19	7.60- 8.40	7.97	*
TSS (mg/L)	I	19	2.00-14.00	7.68	0.75
TSS	E	19	4.00- 9.00	6.00	0.30
DO (mg/L)	I	19	6.50- 9.80	7.92	0.19
DO	E	19	6.20-10.50	8.09	0.26
BOD (mg/L)	I	19	3.00- 8.00	4.74	0.34
BOD	E	19	3.00- 6.00	3.89	0.20
NH3-N (mg/L)	I	19	0.20- 4.35	1.36	0.25
NH3-N	E	19	0.06- 0.30	0.17	0.02
TKN (mg/L)	I	19	2.20- 5.90	3.46	0.30
TKN	E	19	0.96- 1.88	1.42	0.06
TP (mg/L)	I	19	5.70-13.05	8.36	0.42
TP	E	19	1.97- 5.02	4.15	0.19
Fec.Col. (no./100 mL)	I	7	1.00- 2.00	1.14	0.14
Fec.Col.	E	7	17.00-61.00	33.43	6.48

* pH means were calculated by taking the log of the average hydrogen ion concentration; standard errors were not considered meaningful.

Table 18. Surface (0.3 m depth) water quality means, ranges, and sample sizes from eight Lower Mississippi River non-WTS abandoned channel and oxbow lakes, 1984 (Lowery et al., 1987) pH in standard units, DO (dissolved oxygen and TSS (total suspended solids) in mg/L. Sample sizes were 3 for each measurement at all lakes except Deer Park, where sample sizes were 6.

	pH	DO	TSS
Canadian Reach	6.9 6.8-7.1	6.3 4.8-7.4	10.7 9.0-13.0
Crutcher Lake	7.9 7.5-8.2	7.6 5.7-8.9	13.7 6.0-29.0
Catfish Chute	7.7 7.6-7.8	4.5 3.0-6.7	8.0 6.0-12.0
Driver Bar	7.6 7.4-7.9	6.3 4.2-8.4	5.3 4.0-6.0
Lake Whittington	7.5 7.4-7.5	4.9 4.0-5.6	25.7 17.0-42.0
Yucatan Lake	7.4 7.3-7.6	7.0 5.9-7.6	8.0 7.0-9.0
Raccourci Lake	7.7 7.5-7.8	6.2 5.3-6.8	5.0 4.0-7.0
Deer Park Lake	7.2 7.0-7.4	4.6 2.3-6.0	7.2 4.0-11.0

Table 19. Water quality in created and natural herbaceous non-WTS marshes near Tampa, Florida, 1988 (Brown 1991). Values represent one sample; "a" denotes a duplicate sample. TSS (total suspended solids), TP (total phosphorus), and TKN (total Kjeldahl nitrogen) in mg/L.

	TSS	TP	TKN
Natural Wetlands			
107	11.0	0.05	2.20
108	800.0	1.50	15.00
108a	4.0	0.05	2.20
110	66.0	0.11	2.50
201	5.0	0.05	1.90
206	3.0	2.10	6.40
207	270.0	6.10	10.00
207a	280.0	8.70	13.00
Created Wetlands			
101	21.0	0.42	1.40
102	13.0	0.13	1.30
103	10.0	0.06	1.30
103a	13.0	0.05	0.65
104	1.0	0.05	1.20
105	24.0	0.19	2.40
106	20.0	0.18	1.10
204	43.0	0.44	3.90
204a	4.0	0.05	7.50
205	50.0	0.05	1.20
208	59.0	0.05	6.60

Average total phosphorus concentrations at the WTS ranged from 0.08 to 8.36 (Table 17). These values are within the range of values found for non-WTS, which ranged from 0.02 to 8.70 mg/L (Tables 19-22). The mean at the Orlando site fell at the low end of the range, while the Lakeland site mean was toward the upper end of the range. Nevertheless, the phosphorus removal rate within the Lakeland site averaged 50%, which is the expected rate for the site, based on the original site design (Jackson 1989). Jackson (1989) reported 60% removal of phosphorus for the first year of operation at the Lakeland site. Also, the Florida Department of Environmental Regulation operation permit and the U.S. EPA permit do not state a limit for total phosphorus because of the nature of the Lakeland system and because the Alafia River, which receives effluent from the WTS, is not phosphorus limited (Post, Buckley, Schuh, and Jernigan, Inc. 1992).

Average fecal coliform bacteria counts at the WTS sites (1.14-53.26 per 100 mL) (Table 17) fell generally in the lower range of values reported for non-WTS (<10-100) (Table 21). The variability of fecal coliform data, however, is high at the Orlando site, which makes comparison difficult.

The average ammonia nitrogen concentrations at the WTS (0.11-2.36 mg/L) (Table 17) were within or above the range found for non-WTS, although only four comparison values were found and may not be completely appropriate because they are from ponds in North Carolina (Table 21). Average BOD concentrations (2.36-4.74 mg/L) (Table 17) were in the lower range of values found for non-WTS (2.3-7.4 mg/L) (Tables 20 and 21). Average TKN values at the WTS ranged from 0.91 to 3.46 mg/L (Table 17) and fall in the lower range of values found for non-WTS (0.65-15.00 mg/L) (Tables 19 and 21).

Interpreting precisely what some water quality indicators mean for assessing wildlife habitat quality is difficult because the relationships between water quality and habitat quality are usually indirect. Water quality influences community composition of plants, invertebrates, and fish, which are more direct measures of habitat quality and better integrators of conditions important to wildlife than is water quality. In addition, the influences of water quality on habitat are not always consistent. Relationships between nutrient concentrations and wildlife habitat quality often are not applicable under a variety of environmental conditions.

In addition, water quality parameters are often variable, and many measurements must be taken over time to accurately characterize conditions on the site. In a monitoring program, available resources and logistics may not permit the number of measurements required. Use of existing data is also problematic. The measurements are usually readily available from site operators because discharge permits require monitoring of certain constituents in wastewater. However, data management and record-

Table 20. Surface water quality means (N=4) and ranges for Agrico Swamp non-WTS (reclaimed phosphate mine, marsh and swamp habitat) and an open water area in a nearby non-WTS natural marsh in Florida, 1982 (Erwin and Bartleson 1985). pH in standard units, all other parameters in mg/L. DO=dissolved oxygen, BOD=biochemical oxygen demand, TP=total phosphorus.

	pH	DO	BOD	TP
Agrico Swamp	8.9 7.9-9.8	12.1 7.8-15.8	7.4 2.4-15.0	0.31 0.07-0.46
Natural Marsh	5.9 5.7-6.0	1.0 0.8-1.8	2.3 1.2-3.0	0.32 0.10-0.39

Table 21. Surface water quality mean values (depth=0.15-0.29 m) from Nags Head non-WTS marsh ponds, NC, May 1987 (MacPherson 1988). pH in standard units; Dissolved oxygen (DO), ammonia-nitrogen (NH_3-N), total Kjeldahl nitrogen (TKN), total phosphorus (TP), and biochemical oxygen demand (BOD) in mg/L; fecal coliforms are number per 100 mL. Number of stations sampled (N) is given in parentheses. Ranges given where N>1.

	pH	DO	NH_3-N	TKN	TP	BOD	Fecal Coliform
Center Pond	6.2(5) 6.1-6.3	4.9(5) 4.1-6.4	0.03(1)	0.9(1)	0.11(1)	5.9(1)	<10(1)
Pond #13	6.1(4) 6.0-6.1	3.3(4) 1.8-6.0	0.26(1)	1.2(1)	1.30(1)	3.9(1)	<10(1)
Clear Pond	5.3(5) 5.2-5.5	2.0(5) 1.4-3.5	0.01(1)	1.2(1)	0.23(1)	2.5(1)	20(1)
Frog Pen Pond	6.0(4) 5.9-6.1	2.0(4) 1.5-2.9	0.11(1)	1.1(1)	0.29(1)	3.0(1)	100(1)

Table 22. Surface water quality mean values for non-WTS marsh sites in the Okefenokee Swamp (Greening and Gerritsen 1987). pH in standard units, DO (dissolved oxygen) and TP (total phosphorus) in mg/L. Sample size was not reported. Range is given for pH and DO; standard error given for TP.

	pH	DO	TP
Little Cooter Prairie	4.08 3.87-4.53	5.09 1.3-7.2	0.021 ±0.006
Mizell Prairie	3.87 3.72-4.15	4.88 2.1-6.6	0.031 ±0.013
Mack's Island Rookery	4.17 3.93-4.77	2.50 0.6-7.1	0.052 ±0.031
Reference	4.15 3.88-4.81	2.20 0.7-4.0	0.020 ±0.002

keeping by site operators can vary, making it potentially difficult to acquire specific data and to be certain that all data have been obtained. There is also some discrepancy among laboratories and individuals about exactly which metric is measured and what it is called (e.g., ammonia vs. ammonium, total phosphorus vs. total phosphorous as phosphate).

Proper evaluation of acquired data requires review and evaluation of standard operating and quality assurance procedures used by field crews and each analytical laboratory. This may be too subjective and lengthy a procedure for routine monitoring. Interpretation and comparison of data can be difficult if methods, collection frequencies, or intended uses of the data vary from one site to another. Because future studies could involve statistical comparisons, precision and consistency in collection and analysis methods are important and would be difficult to achieve using existing data sets. In addition, it is difficult to find non-WTS for which comparable amounts of existing data are available. For these reasons, the use of existing water quality data sets is not recommended. However, sampling of some water quality indicators, such as dissolved oxygen, ammonia, or suspended solids, during field sampling might provide information on system stressors. This information can be used to interpret indicator data collected at the same time and to determine the reasons for the status of a particular habitat indicator.

CONCLUSIONS AND RECOMMENDATIONS

Based on indicators measured in this study, wetlands used for treating wastewater also appear to provide suitable wildlife habitat in central Florida. Wetland treatment systems are an efficient reuse of water for environmental enhancement; they eliminate some of the chemical treatment; they can be very cost-effective; and they can be beneficial to wildlife. Wildlife habitat is most often an ancillary function of WTS, and the wetlands vary greatly in the habitat values they provide. Much of the variation can be attributed to whether wildlife habitat features are considered when the wetlands are designed, funding available for incorporating specific features, such as islands, wildlife food plants, irregular shoreline, varying depths, and vegetation interspersion, and the degree of management and monitoring of habitats once the wetland is operating.

Table 23 contains a summary of the comparisons between the two WTS studied and non-WTS in the Southeast. Overall, most of the indicator values from the two Florida WTS (for which comparison values were available) were in the mid to high portion of the range of values for non-WTS. None of the indicator values from the two WTS studies were below the range of values for non-WTS. With the exception of ammonia nitrogen, which was at the high end or above the range, the water quality indicator values were in the low to mid range of values for non-WTS. Foraging wading bird densities were higher than densities in some non-WTS in the tropics but were lower than those in the nearby St. Johns Marshes. The two WTS appear to be important as breeding habitat for several species of wading birds, including the endangered wood stork.

The available data suggest that the two WTS provide wildlife habitat similar or superior in quality to that of non-WTS in the same region. Habitat quality was assessed in relation to non-WTS comparison wetlands, but little is known about the habitat quality of comparison wetlands. Guidelines are needed for selecting comparison (i.e., reference) wetlands with good wildlife habitat or for developing criteria for defining good habitat to serve as a gauge for ranking habitat quality.

A summary of the indicators used in this study, including sampling effort, expense, reliability of information collected, direct relevance to wildlife habitat quality, and recommendation for development in future studies, is given in Table 24. Vegetation, invertebrate, and site morphology indicators are recommended for development for evaluating wildlife habitat quality in WTS. Birds may also be good indicators, but focus should be on relating bird numbers to habitat quality or on exploring indicators that may be more informative than bird numbers for assessing habitat quality, such as bird feeding activity or brood counts.

Table 23. General relationship of data from the WTS studied in Florida to the range of values reported for non-WTS in the southeast United States.

	Below Range	Within Range: Low	Within Range: Middle	Within Range: High	Above Range
Plant Species Richness				X	
Invertebrate Genera Richness					X
Water Nutrient Concentrations		X ------	X		
Bird Species Richness				X ------	X
Non-breeding Wading Bird Density			X		

Use of existing water quality data was not considered effective for making general assessments of wildlife habitat quality in WTS, and should be given lower priority in future indicator development. Water quality data can be variable. The water quality constituents sampled, collection frequencies, collection methods, and intended uses of the data vary from one site to another. Laboratory techniques vary among laboratories, and information on QC protocols at each laboratory may be time-consuming to acquire and assess with the same subjectivity in different geographic areas. Nutrient concentrations do not have consistent, direct relationships with wildlife habitat quality that can be applied with certainty under a variety of environmental conditions. Other indicators, such as vegetation structural diversity, number of nests, number of singing birds, relative abundance of wildlife food plants, invertebrate and fish abundance, and site morphology characteristics are more directly related to wildlife habitat and may be more reliable indicators.

Toxicity testing can be expensive, particularly beyond the whole-effluent level of testing. In addition, single, whole-effluent tests do not provide time-integrated information, information about the effects of specific substances in wastewater on wildlife, or the cause of the problem. The discharge of harmful substances to WTS is likely a short-term or intermittent event, and

Table 24. Summary of indicator suitability for assessing the wildlife habitat quality of WTS. Low, moderate, and high are relative ratings for the suite of indicators.

	Vegetation	Invertebrates	Landscape	Whole-Effluent Toxicity	Existing Water Quality Data	Birds
Sampling effort	moderate	moderate	low (photography)	low	low (obtaining existing data)	large (multiple aerial surveys)
Analysis effort	low-moderate	large	moderate-large (interpretation)	moderate	low-moderate	low-moderate
Expense (Sample collection & analysis)	low	moderate	high (for new photos)	moderate-high	low	moderate
Reliability of information for assessing habitat	high	moderate-high	high	low-moderate	low	moderate
Recommend development for future studies	yes	yes	yes	possibly in some cases	no	possibly
Problems	none	standardization of sampling methods	none	none	data quality difficult to evaluate; variability	bird mobility; logistics; contracts

toxicity in water could be missed by taking only one sample.

Whole-effluent tests should therefore be conducted on a routine basis as initial testing for contaminants in selected wetlands suspected to be at risk from contamination or toxic inputs, such as wetlands that receive industrial discharges, where user violations have occurred in the past, or where other data collected indicate a potential problem requiring further investigation. Determining the source of any substances found and making the connections between the levels found and actual effects on wildlife would then be necessary.

Some topics regarding wildlife habitat quality (e.g., how to measure it, how to evaluate it) require further study. The following are suggestions for future studies:

- For comparing WTS with non-WTS, future studies should include simultaneous sampling on nearby reference (non-WTS) wetlands so that results from both types of wetlands are more directly comparable and confounding factors are minimized. It is not possible to assess collected data if comparison values are unavailable or unreliable. Comparison with literature values might be sufficient for preliminary studies, but to put in context the indicator values from WTS and to make valid conclusions about the quality of wildlife habitat, the best data for comparison are those that are collected at the same time, in close proximity, on similar classes of wetlands, and with the same sampling techniques.

  Reference wetlands should be natural, enhanced, or restored wetlands that are not used for wastewater treatment. Created wetlands should not be used for comparisons because there is not enough information to show that they duplicate wetland functions on a long-term basis (Kusler and Kentula 1990). Establishing appropriate criteria for selection of reference wetlands will require further thought. One approach would be to establish guidelines for selection of reference sites that represent "good" habitat quality. Data collected can be used as a gauge against which measurements or an aggregation of measurements taken at WTS can be rated. Reference wetlands should also be as similar as possible to the WTS in question with respect to size, wetland classification, location, and type of surrounding land use. Comparisons should be quantitative.

- In addition to comparison with non-WTS reference sites in the same region, guidelines should be developed for rating habitat quality. In some landscapes, potential reference sites might all be in marginal or poor condition. Using suboptimal reference sites as a gauge for assessing habitat quality is not desirable or wise. Although it allows an assessment of habitat value relative to the predominant condition in a region, it can weaken the overall concept of good wildlife habitat. Guidelines should be performance standards that are applied on the basis of best professional judgment and provide for flexibility for dealing with environmental uncertainty (Chapman 1991).

- Future work should also focus on developing means for assessing and reducing data. Developing assessment methods can identify potential stressors, or causes of condition, which can then be used to establish a gauge for rating habitat value. Data reduction involves combining information from a group of indicators or from data on multiple species to form a single indicator, or index. For instance, species diversity incorporates richness and abundance of all species into a single value. A similar index might be developed for vegetation structural diversity based on the number of vegetation layers and their relative coverages. Multivariate analyses are also useful for analyzing combined data and forming indices. Species-specific data, however, can be used to identify stressors or to monitor long term changes at a wetland and should not be overlooked in favor of indices.

- The suite of indicators for this study was limited by level of funding, labor, and logistical constraints. Future studies could assess the usefulness of indicators that were not examined in this study, including new metrics for evaluating habitat in terms of vegetation, invertebrates, site morphology and bird use. New indicators might include benthic invertebrates, basin slope, average water depth, water permanence, size and configuration of open water areas, and degree of human disturbance. At this stage, future work should focus on development of biological indicators that are directly related to wildlife habitat rather than on attributes that might only infer wildlife use through an indirect relation (e.g., nutrients, sediment type, hydrologic regime). Indirectly-related indicators, however, can be useful for identifying ecosystem stressors and the reasons for the status of a particular biological indicator (e.g., hydrologic regime and sediment types can influence the plant communities that develop).

- If bird use is retained as an indicator, a greater focus should be placed on bird activity (breeding, feeding, roosting, resting) in the WTS and the presence of threatened, endangered, or keystone species.

- The elimination of some indicators, if different indicators provide essentially the same information, would save money and time in sampling and analysis. For instance, some vegetation indicators measured in the field can easily be obtained from air photos (e.g., structural diversity, relative coverage of each structural type). Development of remotely-sensed indicators should be further explored, particularly for large wetlands, such as the two WTS in Florida, where time restricts thorough ground sampling of the wetland.

This pilot study provided evidence that the two WTS provide favorable wildlife habitat, comparable to that of non-WTS in the same geographic region. A dependable water supply at both wetlands helps ensure permanent, deep water, which makes the sites attractive as nesting sites for several wading bird species. Wildlife habitat at both sites has been enhanced while maintaining effective water treatment, which is evidence that the two interests are compatible.

LITERATURE CITED

Adamus, P.R. 1993. Irrigated Wetlands of the Colorado Plateau: Information Synthesis and Habitat Evaluation Method. EPA/600/R-93/071. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR.

Adamus, P.R. and K. Brandt. 1990. Impacts on Quality of Inland Wetlands of the United States: a Survey of Indicators, Techniques, and Applications of Community-Level Biomonitoring Data. EPA/600/3-90/073. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR.

Adamus, P.R., E.J. Clairain, R.D. Smith, and R.E. Young. 1987. Wetland Evaluation Technique (WET), Vol. II: Methodology. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR, and Department of the Army, Vicksburg, MS.

Aust, M.W., S.F. Mader, and R. Lea. 1988. Abiotic changes of a tupelo-cypress swamp following helicopter and rubber-tired skidder timber harvest. Fifth Southern Silvicultural Research Conference, Memphis, TN.

Bancroft, G.T. 1989. Status and conservation of wading birds in the Everglades. American Birds 43:1258-1265.

Bancroft, G.T., S.D. Jewell, and A.M. Strong. 1991. Foraging and nesting ecology of herons in the lower Everglades relative to water conditions. South Florida Water Management District, West Palm Beach, FL.

Bancroft, G.T., W. Hoffman, and R. Sawicki. 1992. The importance of the Water Conservation Areas in the Everglades to the endangered Wood Stork (*Mycteria americana*). Conservation Biology 6:392-98.

Bastian, R.K., P.E. Shanaghan, and B.P. Thompson. 1989. Use of wetlands for municipal wastewater treatment and disposal - regulatory issues and EPA policies. Pages 265-278 IN D.A. Hammer (Ed.), Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural. Lewis Publishers, Inc., Chelsea, MI.

Beecher, W.J. 1942. Nesting Birds and the Vegetation Substrates. Chicago Ornithological Society, Chicago, IL.

Brennan, K.M. 1985. Effects of wastewater on wetland animal communities. Pages 199-223 IN P.J. Godfrey, E.R. Kaynor, S. Pelczarski, and J. Benforado (Eds.), Ecological Considerations in Wetlands Treatment of Municipal Wastewaters. Van Nostrand Reinhold Company, New York, NY.

Brodie, G.A., D.A. Hammer, and D.A. Tomljanovich. 1989. Treatment of acid drainage with a constructed wetland at the Tennessee Valley

Authority 950 Coal Mine. Pages 201-209 IN D.A. Hammer (Ed.), Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural. Lewis Publishers, Inc., Chelsea, MI.

Brooks, R.P., D.E. Arnold, E.D. Bellis, C.S. Keener, and M.J. Croonquist. 1989. A methodology for biological monitoring of cumulative impacts on wetland, stream, and riparian components of watersheds. Proceedings of the International Wetlands Symposium, Charleston, SC. Association of State Wetland Managers, Inc., Berne, NY.

Brooks, R.P. and R.M. Hughes. 1988. Guidelines for assessing the biotic communities of freshwater wetlands. Pages 276-282 IN J.A. Kusler, M.L. Quammen, and G. Brooks (Eds.), Proceedings of the National Wetland Symposium: Mitigation of Impacts and Losses. Association of State Wetland Managers, Berne, NY.

Brown, M.T. 1991. Evaluating Created Wetlands through comparisons with natural wetlands. EPA/600/3-91/058. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR.

Brown, M.T., J. Schaefer, and K. Brandt. 1989. Buffer zones for water, wetlands, and wildlife in the east central Florida region. Center for Wetlands, University of Florida, Gainesville, FL.

Buglewitz, E.G., W.A. Mitchell, J.E. Scott, M. Smith, and W.L. King. 1988. A physical description of main stem levee borrow pits along the lower Mississippi River. U.S. Army Corps of Engineers, Mississippi River Commission, Vicksburg, MS.

Cedarquist, N.W. 1979. Suisun Marsh management study, progress report on the feasibility of using wastewater for duck club management. U.S. Department of Energy, Water and Power Resources Service, Sacramento, CA.

Cedarquist, N.W. 1980a. Suisun Marsh management study, progress report on the feasibility of using wastewater for duck club management. U.S. Department of Interior, Water and Power Resources Service, Sacramento, CA.

Cedarquist, N.W. 1980b. Suisun Marsh management study, 1979-1980 progress report on the feasibility of using wastewater for duck club management. U.S. Department of Interior, Water and Power Resources Service, Sacramento, CA.

Cedarquist, N.W. and W.M. Roche. 1979. Reclamation and reuse of wastewater in the Suisun Marsh of California. Proceedings of the Water Reuse Symposium, Vol. 1. American Water Works Association Research Foundation, Denver, CO.

Chapman, P.M. 1991. Environmental quality criteria: what type should we be developing? Environmental Science and Technology 25:1353-1359.

Cobb, S.P., C.H. Pennington, J.A. Baker, and J.E. Scott. 1984. Fishery and ecological investigations of main stem levee borrow pits along the lower Mississippi River. U.S. Army Corps of Engineers, Mississippi River Commission, Vicksburg, MS.

Conway, T.E. and J.M. Murtha. 1989. The Iselin Marsh Pond Meadow. Pages 139-144 IN D.A. Hammer (Ed.), Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural. Lewis Publishers, Inc., Chelsea, MI.

Costello, C.J. 1989. Wetlands treatment of dairy animal wastes in Irish drumlin landscape. Pages 702-709 IN D.A. Hammer (Ed.), Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural. Lewis Publishers, Inc., Chelsea, MI.

Cyr, H. and J.A. Downing. 1988. Empirical relationships of phytomacrofaunal abundance to plant biomass and macrophyte bed characteristics. Canadian Journal of Fisheries and Aquatic Sciences 45:976-984.

Davis, D.G. and J.C. Montgomery. 1987. EPA's regulatory and policy considerations on wetlands and municipal wastewater treatment. Pages 69-70 IN K.R. Reddy and W.H. Smith (Eds.), Aquatic Plants for Water Treatment and Recovery. Magnolia Publishing Inc., Orlando, FL.

Demgen, F.C. 1979. Wetlands creation for habitat and treatment at Mt. View Sanitary District, California. Pages 61-73 IN R.K. Bastian and S.C. Reed (Project Officers), Aquaculture Systems for Wastewater Treatment: Seminar Proceedings and Engineering Assessment. EPA 430/9-80-006. U.S. Environmental Protection Agency, Office of Water Program Operations, Municipal Construction Division, Washington, DC.

Demgen, F.C. and J.W. Nute. 1979. Wetlands creation using secondary treated wastewater. Pages 727-739 IN American Water Works Association Research Foundation Water Reuse Symposium, Vol. I. American Water Works Association Research Foundation, Washington, DC.

Dickerman, J.A., A.J. Stewart, and J.C. Lance. 1985. The impact of wetlands on the movement of water and nonpoint pollutants from agricultural watersheds. A report to the Soil Conservation Service. U.S. Department of Agriculture, Agricultural Research Service, Water Quality and Watershed Research Laboratory, Durant, OK.

Dierberg, F.E. and P.L. Brezonik. 1984. Water chemistry of a Florida cypress dome. Pages 34-50 IN K.C. Ewel and H.T. Odum (Eds.), Cypress Swamps. University Presses of Florida, Gainesville, FL.

Donovan, D.B. 1990a. First Annual Report, Monitoring of Wetland Vegetation at IMCF Section 12 Hookers Prairie Reclamation Site Polk County, FL. IMC Fertilizer, Inc., Bartow FL.

Donovan, D.B. 1990b. Forth Annual Report, N.E. 7/12 Reclaimed Stream. IMC Fertilizer, Inc., Bartow, FL.

Dvorak, J. and E.P.H. Best. 1982. Macroinvertebrate communities associated with the macrophytes of Lake Vechten: structural and functional relationships. Hydrobiologia 95:115-26.

Dwyer, T. J., G. L Krapu, and D. M. Janke. 1979. Use of prairie pothole habitat by breeding mallards. Journal of Wildlife Management 43:526-531.

Edelson, N.A. and M.W. Collopy. 1990. Foraging ecology of wading birds using an altered landscape in central Florida. Florida Institute of Phosphate Research, Bartow, FL.

Erwin, K.L. 1988. Fort Green Reclamation Project, 6th annual report for Agrico Chemical Company, Mulberry, FL. Erwin Consulting Ecologist, Inc., Fort Myers, FL.

Erwin, K.L. 1990. Payne Creek Reclamation Project, 3rd annual report for Agrico Chemical Company, Mulberry, FL. Erwin Consulting Ecologist, Inc., Fort Myers, FL.

Erwin, K.L. 1991. An evaluation of wetland mitigation in the South Florida Water Management District - Volume II. Report to South Florida Water Management District, West Palm Beach, FL. Fort Myers, FL.

Erwin, K.L. and F.D. Bartleson. 1985. Water quality within a central Florida phosphate surface mined reclaimed wetland. Pages 74-85 IN F.J. Webb (Ed.), Proceedings of the Twelfth Annual Conference on Wetland Restoration and Creation. Hillsborough Community College, Tampa, FL.

Erwin, K.L. and G.R. Best. 1985. Marsh community development in a central Florida phosphate surface-mined reclaimed wetland. Wetlands 5:155-66.

Fetter, C.W., Jr., W.E. Sloey, and F.L. Spangler. 1978. Use of a natural marsh for wastewater polishing. Journal of the Water Pollution Control Federation 50:290-307.

Frederick, P.C. and M.W. Collopy. 1988. Reproductive ecology of wading birds in relation to water conditions in the Florida Everglades. Florida Fish and Wildlife Cooperative Research Unit, Technical Report No. 30. U.S. Fish and Wildlife Service, Gainesville, FL.

Friedemann, M. and J. Hand. 1989. Typical water quality values for Florida's lakes, streams and estuaries. Florida Department of Environmental Regulation, Tallahassee, FL.

Greening, H.S. and J. Gerritsen. 1987. Changes in macrophyte community structure following drought in the Okefenokee Swamp, Georgia, U.S.A. Aquatic Biology 28:113-128.

Godfrey, P.J., E.R. Kaynor, S. Pelczarski, and J. Benforado (Eds.). 1985. Ecological Considerations in Wetlands Treatment of Municipal Wastewaters. Van Nostrand Reinhold Company, New York, NY.

Guntenspergen, G.R. and F. Stearns. 1985. Ecological perspectives on wetland systems. Pages 69-97 IN P.J. Godfrey, E.R. Kaynor, S. Pelczarski, and J. Benforado (Eds.), Ecological Considerations in Wetlands Treatment of Municipal Wastewaters. Van Nostrand Reinhold, New York, NY.

Hammer, D.A. and R.K. Bastian. 1989. Wetlands ecosystems: natural water purifiers? Pages 5-19 IN D.A. Hammer (Ed.), Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural. Lewis Publishers, Chelsea, MI.

Hampson, P.S. 1989. Dissolved oxygen concentrations in a central Florida wetlands stream. Pages 149-159 IN D.W. Fisk (Ed.), Proceedings of the Symposium on Wetlands: Concerns and Successes, Tampa, FL. American Water Resources Association, Bethesda, MD.

Harris, H.J., M.S. Milligan, and G.A. Fewless. 1983. Diversity: quantification and ecological evaluation in freshwater marshes. Biological Conservation 27:99-110.

Henigar and Ray Engineering Associates, Inc. 1990. A qualitative and quantitative assessment of the West-of-K6 reclamation unit, Hillsborough County, FL. Prepared for IMC Fertilizer, Inc., Bartow, FL.

Hicks, D.B. and Q.J. Stober. 1989. Monitoring of constructed wetlands for wastewater. Pages 447-455 IN D.H. Hammer (Ed.), Constructed Wetlands for Wastewater Treatment, Municipal, Industrial and Agricultural. Lewis Publishers, Inc., Chelsea, MI.

Hoffman, W., G.T. Bancroft, and R.J. Sawicki. 1990. Wading bird populations and distributions in the Water Conservation Areas of the Everglades: 1985-1988. South Florida Water Management District, West Palm Beach, FL.

Hunsaker, C.T. and D.E. Carpenter, Eds. 1990. Ecological indicators for the Environmental Monitoring and Assessment Program. U.S. Environmental Protection Agency, Atmospheric Research and Exposure Laboratory, Research Triangle Park, NC.

Jackson, J. 1989. Man-made wetlands for wastewater treatment: two case studies. Pages 574-580 IN D.A. Hammer (Ed.), Constructed Wetlands for Wastewater Treatment: Municipal, Industrial, and Agricultural. Lewis Publishers, Inc., Chelsea, MI.

Jeffries, M. 1989. Measuring Talling's element of chance in pond populations. Freshwater Biology 20:383-93.

Jelks, H. 1991. The distribution of foraging wading birds in relation to the physical and biological characteristics of freshwater wetlands in southwest Florida. MS Thesis, University of Florida, Gainesville, FL.

Kadlec, R.H. and F.B. Bevis. 1990. Wetlands and wastewater: Kinross, Michigan. Wetlands 10(1):77-92.

Kadlec, R.H. and J.A. Kadlec. 1979. Wetlands and water quality. Pages 436-456 IN P.E. Greeson, J.R Clark, and J.E. Clark (Eds.), Wetland Functions and Values: The State of Our Understanding. American Water Resources Association, Minneapolis, MN.

King County. 1986. The Use of Wetlands for Stormwater Storage and Nonpoint Pollution Control: A Review of the Literature. Resource Planning Section, Department of Planning and Community Development, King County, WA.

Krull, J.N. 1970. Aquatic plant macroinvertebrate associations and waterfowl. Journal of Wildlife Management 34:707-718.

Kusler, J.A. and M.E. Kentula (Eds.). 1990. Wetland Creation and Restoration: The Status of the Science. Island Press, Washington, DC.

Lowery, D.R., M.P. Taylor, R.L. Warden, and F.H. Taylor. 1987. Fish and benthic communities of eight lower Mississippi River floodplain lakes. Lower Mississippi River Environmental Program Report 6. Mississippi River Commission, Vicksburg, MS.

MacPherson, T.F. 1988. Benthic macroinvertebrates of selected ponds in the Nags Head Woods Ecological Preserve. The Association of Southeastern Biologists Bulletin 35(4):181-188.

McAllister, L.S. 1992. Habitat quality assessment of two wetland treatment systems in Mississippi - A pilot study. EPA/600/R-92-229. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR.

McAllister, L.S. 1993. Habitat quality assessment of two wetland treatment systems in the arid West - A pilot study. EPA/600/R-93-117. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR.

Merritt, R.W. and K.W. Cummins (Eds.). 1984. An Introduction to the Aquatic Insects of North America, Second Edition. Kendall/Hunt Publishing Company, Dubuque, IA.

Mudroch, A. and J.A. Capobianco. 1979. Effects of treated effluent on a natural marsh. Journal of Water Pollution Control Federation 51(9):2243-2256.

Murkin, H.R. and B.D.J. Batt. 1987. Interactions of vertebrates and invertebrates in peatlands and marshes. Memoirs of the Entomological Society of Canada Vol. 40.

Murkin, H.R. and D.A. Wrubleski. 1987. Aquatic invertebrates of freshwater wetlands: function and ecology. Pages 239-49 IN D.D. Hook, W.H. McKee Jr., H.K. Smith, J. Gregory, V.G. Burell, Jr., M.R. DeVoe, R.E. Sojka, S. Gilbert, R. Banks, L.H. Stolzy, D. Brooks, T.D. Matthews and T.H. Shear (Eds.), The Ecology and Management of Wetlands, Vol. I: Ecology of Wetlands. Croom Helm, London.

Nixon, S.W. and V. Lee. 1986. Wetlands and water quality: a regional view of recent research in the United States on the role of freshwater and saltwater wetlands as sources, sinks, and transformers of nitrogen, phosphorus, and various heavy metals. Technical Report Y-86-2. U.S. Army Corps of Engineers, Vicksburg, MS.

Pennak, R.W. 1978. Freshwater Invertebrates of the United States. Second Edition. John Wiley and Sons, Inc., New York, NY.

Piest, L.A. and L.K. Sowls. 1985. Breeding duck use of a sewage marsh in Arizona. Journal of Wildlife Management 49:580-585.

Plafkin, J.L., M.T. Barbour, K.D. Porter, S.K. Gross, and R.M. Hughes. 1989. Rapid bioassessment protocols for use in streams and rivers: benthic macroinvertebrates and fish. EPA/444/4-89-001. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

Post, Buckley, Schuh, and Jernigan, Inc. 1992. Performance Review for City of Lakeland's Glendale Wetland Treatment System: Monitoring report for years 1987 through 1991. Prepared for City of Lakeland Department of Public Works. PBS&J, Winter Park, FL.

Rapport, D.J. 1989. What constitutes ecosystem health? Perspectives in Biology and Medicine 33(1):120-132.

Reed, S.C., E.J. Middlebrooks, and R.W. Crites. 1988. Natural Systems for Waste Management and Treatment. McGraw-Hill, New York, NY.

Reid, F.A. 1985. Wetland invertebrates in relation to hydrology and water chemistry. Pages 72-79 IN M.D. Knighton (Ed.), Water Impoundments for Wildlife: a Habitat Management Workshop. General

Technical Report NC-100. North Central Forest Experiment Station, St. Paul, MN.

Richardson, C.J. and D.S. Nichols. 1985. Ecological analysis of wastewater management criteria in wetland ecosystems. Pages 351-391 IN P.J. Godfrey, E.R. Kaynor, S. Pelczarski, and J. Benforado (Eds), Ecological Considerations in Wetlands Treatment of Municipal Wastewaters. Van Nostrand Reinhold Company, New York, NY.

Roth, R.R. 1976. Spatial heterogeneity and bird species diversity. Ecology 57:773-82.

Rozas, L.P., and W.E. Odum. 1987. The role of submerged aquatic vegetation in influencing the abundance of nekton on contiguous tidal freshwater marshes. Journal of Experimental Marine Biology and Ecology 114:289-300.

Ruwaldt, J.J., Jr., L.D. Flake, and J.M. Gates. 1979. Waterfowl pair use of natural manmade wetlands in South Dakota. Journal of Wildlife Management 43:375-383.

Sather, J.H. 1989. Ancillary benefits of wetlands constructed primarily for wastewater treatment. Pages 353-358 IN D.A. Hammer (Ed.), Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural. Lewis Publishers, Inc., Chelsea, MI.

Schaeffer, D.J., E.E. Herricks, and H.W. Kerster. 1988. Ecosystem health I: measuring ecosystem health. Environmental Management 12(4):445-455.

Schwartz, L.N. 1987. Regulation of wastewater discharge to Florida wetlands. Pages 951-958 IN K.R. Reddy and W.H. Smith (Eds.), Aquatic Plants for Water Treatment and Resource Recovery. Magnolia Publishing, Inc., Orlando, FL.

Short, H.L. and S.C. Williamson. 1986. Evaluating the structure of habitat for wildlife. Pages 97-104 IN J. Verner, M.L. Morrison, and C.J. Ralph (Eds.), Wildlife 2000: modeling habitat relationships of terrestrial vertebrates. University of Wisconsin Press, Madison, WI.

Smith, B.D., P.S. Maitland, and S.M. Pennock. 1987. A comparative study of water level regimes and littoral benthic communities in Scottish Locks. Biological Conservation 39:291-316.

Staubitz, W.W., J.M. Surface, T.S. Steenhuis, J.H. Peverly, M.J. Lavine, N.C. Weeks, W.E. Sanford, and R.J. Kopka. 1989. Potential use of constructed wetlands to treat landfill leachate. Pages 735-742 IN D.A. Hammer (Ed.), Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural. Lewis Publishers, Inc., Chelsea, MI.

Steel, P.E., P.D. Dalke, and E.G. Bizeau. 1956. Duck production at

Gray's Lake, Idaho, 1949-51. Journal of Wildlife Management 20:279-85.

Swanson, G.A. and M.I. Meyer. 1977. Impact of fluctuating water levels on feeding ecology of breeding blue-winged teal. Journal of Wildlife Management 41:426-433.

Swift, B.L., J.S. Larson, and R.M. DeGraaf. 1984. Relationship of breeding bird density and diversity to habitat variables in forested wetlands. Wilson Bulletin 96:48-59.

Swindell, C.E. and J.A. Jackson. 1990. Constructed wetlands design and operation to maximize nutrient removal capabilities. Pages 107-114 IN P.F. Cooper and B.C. Findlater (Eds.), Constructed Wetlands in Water Pollution Control. Pergamon Press, Oxford, NY.

Teels, B.M., G. Anding, D.H. Arner, E.D. Norwood, and D.E. Wesley. 1976. Aquatic plant, invertebrate and waterfowl associations in Mississippi. Proceedings of the Southeast Association of Game Fish Commission 30:610-616.

Tonn, W.M. and J.J. Magnuson. 1982. Patterns in the species composition and richness of fish assemblages in northern Wisconsin lakes. Ecology 63:1149-1166.

Tucker, D.S. 1958. The distribution of some fresh-water invertebrates in ponds in relation to annual fluctuations in the chemical composition of the water. Journal of Animal Ecology 27:105-119.

U.S. Army Engineer Mississippi River Commission. 1986. Bird and Mammal Use of Main Stem Levee Borrow Pits along the Lower Mississippi River. Lower Mississippi River Environmental Program Report 3, Vicksburg, MS.

U.S. Environmental Protection Agency. 1983. The Effects of Wastewater Treatment Facilities on Wetlands in the Midwest. Appendix A: Technical Support Document. USEPA-905/3-83-002. U.S. Environmental Protection Agency, Region 5, Chicago, IL.

U.S. Environmental Protection Agency. 1984. The Ecological Impacts of Wastewater on Wetlands, An Annotated Bibliography. EPA 905/3-84-002. U.S. Environmental Protection Agency and U.S. Fish and Wildlife Service, Washington, DC.

U.S. Environmental Protection Agency. 1988a. Design Manual: Constructed Wetlands and Aquatic Plant systems for Municipal Wastewater Treatment. EPA/625/1-88/022. U.S. Environmental Protection Agency Center for Environmental Research Information, Cincinnati, OH.

U.S. Environmental Protection Agency. 1988b. Short-term Methods for Estimating the Chronic Toxicity of Effluents and Receiving

Waters to Marine and Estuarine Organisms. EPA-600/4-87-028. Environmental Monitoring and Support Laboratory, Cincinnati, OH.

U.S. Fish and Wildlife Service. 1980. Habitat Evaluation Procedures (HEP) Manual (102ESM). U.S. Fish and Wildlife Service, Washington, DC.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130-137.

Voights, D.K. 1976. Aquatic invertebrate abundance in relation to changing marsh conditions. American Midland Naturalist 95:313-322.

Wallace, P.M. 1990. Herbaceous vegetation monitoring of the IMC Horse Creek wetland reclamation site. IMC Fertilizer, Inc., Bartow, FL.

Weinstein, M.P. and H.A. Brooks. 1983. Comparative ecology of nekton residing in a tidal creek and adjacent seagrass meadow: community composition and structure. Marine Ecology Progress Series 12:15-17.

Weller, M.W. 1978. Management of freshwater marshes for wildlife. Pages 267-84 IN R.E. Good, D.F. Whigham, and R.L. Simpson (Eds.), Freshwater Wetlands: Ecological Processes and Management Potential. Academic Press, New York, NY.

Weller, M.W. and L.H. Frederickson. 1973. Avian ecology of a managed glacial marsh. Living Bird 12:269-91.

Weller, M.W. and C.E. Spatcher. 1965. Role of habitat in the distribution and abundance of marsh birds. Special Report No. 43. Iowa Agricultural Home Economics Experiment Station, Ames, IA.

Wilhelm, M., S.R. Lawry, and D.D. Hardy. 1988. Creation and management of wetlands using municipal wastewater in northern Arizona: a status report. Pages 114-120 IN J. Zelazny and J.S. Feierabend (Eds.), Increasing Our Wetland Resources. National Wildlife Federation, Washington, DC.

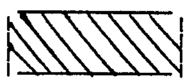
Yocum, T.G., R.A. Leidy, and C.A. Morris. 1989. Wetlands protection through impact avoidance: A discussion of the 404(b)(1) alternatives analysis. Wetlands 9(2):283-297.

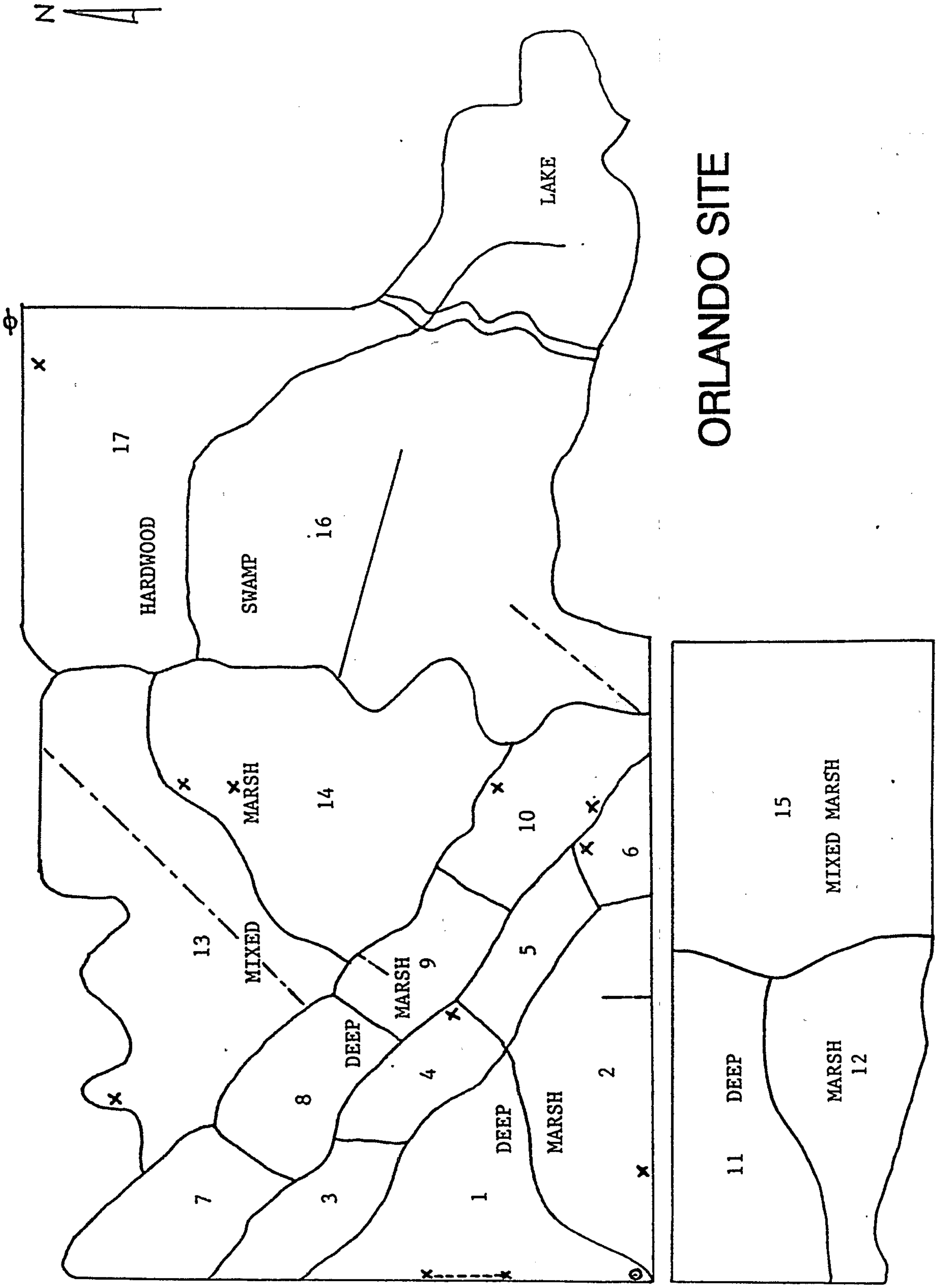
Ziser S.W. 1978. Seasonal variations in water chemistry and diversity of the phytophilic macroinvertebrates of three swamp communities in southeastern Louisiana. Southwestern Naturalist 23(4):545-562.

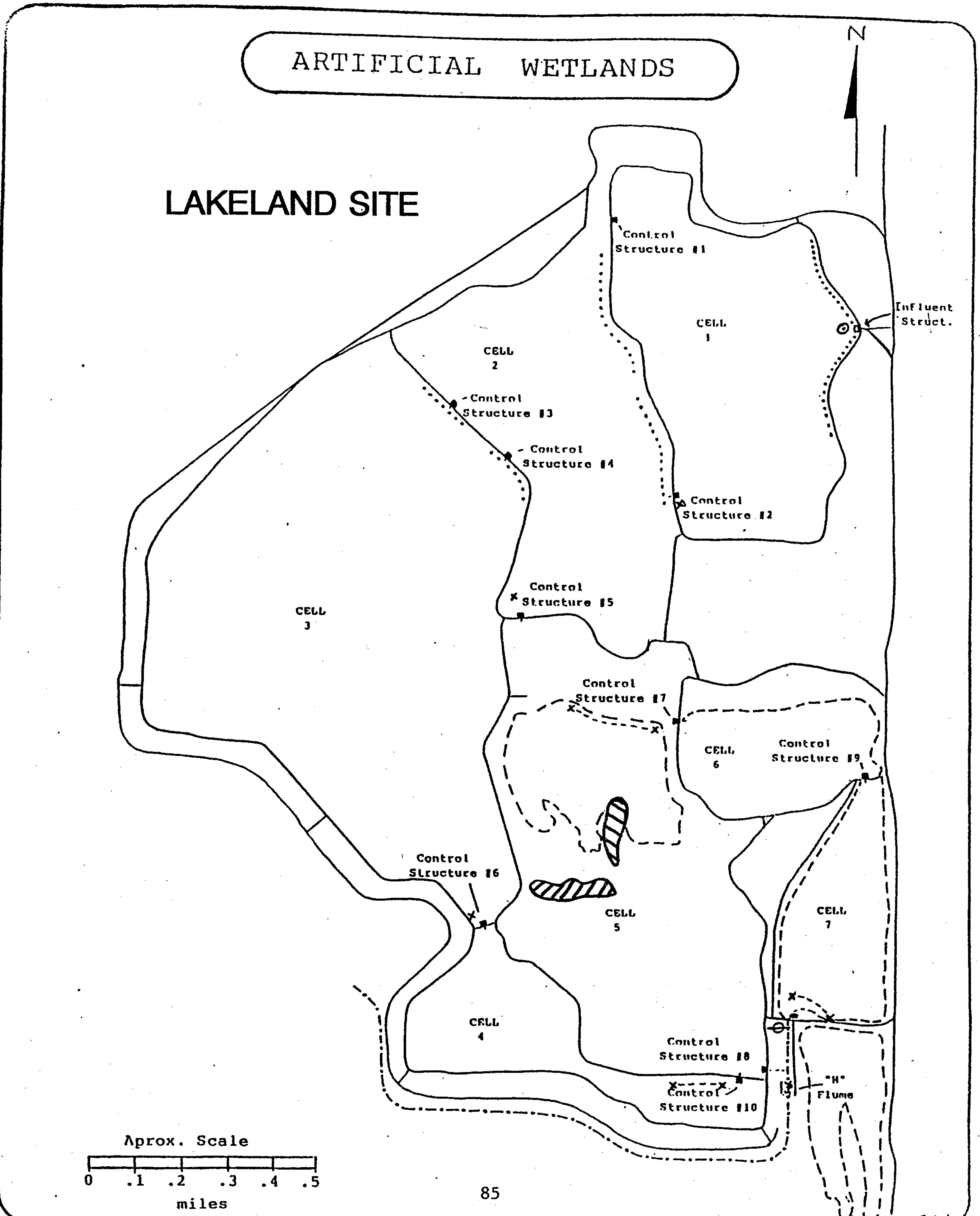
APPENDIX A. Site Maps and Sampling Points

Maps provided by site operators of the Orlando and Lakeland sites are included in this appendix. The treatment cells are numbered for reference. The following features are designated on each map: vegetation transect locations (Orlando site only), invertebrate sample locations, and whole effluent toxicity water sampling points. Some of the invertebrate samples were collected at a single spot in the wetland, designated by an X on the maps. When invertebrate densities were low, however, several net samples had to be collected to obtain 1/2 hour of collection time. Therefore, Xs connected by a dotted line represent places where samples consisting of several nettings were taken along a shoreline or the edge of vegetation from a single habitat type.

The key below describes the symbols and features found on maps in this appendix:

————	Dikes
⊙	Influent sample collection point
-⊖-	Effluent sample collection point
___.___.___	Vegetation transects
X or X----X	Invertebrate sample locations
	Rookery Islands


ORLANDO SITE
N
LAKE
17
HARDWOOD
SWAMP
16
MARSH
14
13
MIXED
10
6
MARSH
9
DEEP
8
5
4
2
DEEP
MARSH
1
3
7
15
MIXED MARSH
DEEP
11
MARSH
12


ARTIFICIAL WETLANDS
N
LAKELAND SITE
Control Structure #1
CELL 1
Influent Struct.
CELL 2
Control Structure #3
Control Structure #4
Control Structure #2
Control Structure #5
CELL 3
Control Structure #7
CELL 6
Control Structure #9
Control Structure #6
CELL 5
CELL 7
CELL 4
Control Structure #8
Control Structure #10
"H" Flume
Aprox. Scale
0 .1 .2 .3 .4 .5
miles
DMH

APPENDIX B. Site Contacts and Local Experts Consulted

ORLANDO

Site Contact:

Alan R. Oyler, P.E.
Assistant Bureau Chief - Bureau of Wastewater
Environmental Services Department - City of Orlando
5100 L.B. McLeod Road
Orlando, FL 32811

Botanists consulted:

Dr. Bill Dunn
CH2M Hill
7201 NW 11th Place
P.O. Box 1647
Gainesville, FL 32602-1647

Mike Mahler
Polk County Environmental Services
Bendurrac Road
Winter Haven, FL

Seth Blitch and Jim Burney
Post, Buckley, Schuh, and Jernigan, Inc.
Winter Park Plaza
1560 Orange Ave., Suite 700
Winter Park, FL 32789

Aerial Photography Company:

Kucera International, Inc.
Dick Connors/Larry Towles
3550 Drain Field Road
Lakeland, FL 33811

Bird Surveyors:

Dr. Peter C. Frederick and Steven M. McGehee
Department of Wildlife and Range Sciences
118 Newins-Ziegler Hall
University of Florida
Gainesville, FL 32611

Water Analysis Laboratories:

Bureau of Wastewater Laboratory, City of Orlando, FL
Contact: Alan Oyler -Orlando site contact

LAKELAND

Site Contact:

Dave Hill
Wastewater Operations
City of Lakeland Department of Public Works
1825 Glendale Street
Lakeland, FL 33803

Botanist consulted:

Dr. Bill Dunn
CH2M Hill
7201 NW 11th Place
P.O. Box 1647
Gainesville, FL 32602

Aerial Photography Company:

Kucera International, Inc.
Dick Connors/Larry Towles
3550 Drain Field Road
Lakeland, FL 33811

Bird Surveyors:

Dr. Peter C. Frederick and Steven M. McGehee
Department of Wildlife and Range Sciences
118 Newins-Ziegler Hall
University of Florida
Gainesville, FL 32611

Water Analysis Laboratory:

City of Lakeland Wastewater Treatment Laboratory, Lakeland, FL
Contact: Dave Hill, site manager

APPENDIX C. Invertebrate Biologists and Identification Keys Used

Biologists:

Nan Allen; Ann Hershey
221 Life Sciences Bldg. - Biology office
10 University Drive
University of Minnesota-Duluth
Duluth, MN 55812

Invertebrate taxonomic keys used:

Borror, D.J., C.A. Triplehorn, and N.F. Johnson. 1989. An Introduction to the Study of Insects. Sixth Edition. Sanders College Publishing. Philadelphia, PA.

Klemm, D.J. 1982. Leeches (Annelida:Hirudinea) of North America. EPA-600/3-82/025. Environmental Protection Agency Environmental Monitoring and Support Lab. Office of Research and Development, Cincinnati, OH.

Merritt, R.W. and K.W. Cummins. 1984. An Introduction to the Aquatic Insects of North America. Second Edition. Kendall Hunt Publishing Co., Dubuque, IA.

Pennak, R.W. 1978. Freshwater Invertebrates of the United States. Second Edition. John Wiley and Sons, Inc., New York, NY.

Pennak, R.W. 1989. Freshwater Invertebrates of the United States. Third Edition. John Wiley and Sons, Inc., New York, NY.

Usinger, R.L. (Ed.). 1968. Aquatic Insects of California, with North American Genera and California Species. University of California Press, Berkeley, CA.

Ward, H.B. and G.C. Whipple (Eds.). 1959. Fresh Water Biology. Second Edition. John Wiley and Sons, Inc., New York, NY.

Wiederholm, T. (Ed.). 1983. Chironomidae of the Holarctic Region. Part 1 Larvae. Entomologica Scandinavica Supplement No. 19. Borgstroms Tyckeri AB, Motala.

APPENDIX D. **Water Chemistry of Replicate Samples Used for Whole Effluent Toxicity Tests.**

Ceriodaphnia dubia chronic test[a]

Sample	Mean pH	pH Range	Mean Temp (°C)	Mean DO* (mg/L)	Mean Conductiv. (umhos/cm)
Orlando site					
		Initial Chemistries			
Influent	7.25	7.17-7.35	25.9	8.7	485
Effluent	7.25	7.14-7.35	25.7	9.0	401
Control	8.03	7.87-8.11	26.3	7.9	128
		Final Chemistries			
Influent	8.12	8.05-8.18	25.2	8.1	--
Effluent	8.16	8.12-8.22	25.4	8.2	--
Control	8.17	8.10-8.22	25.5	8.1	--
Lakeland site					
		Initial Chemistries			
Influent	7.22	7.20-7.26	25.5	9.3	1178
Effluent	7.67	7.63-7.70	25.5	9.1	668
Control	8.35	8.31-8.37	26.2	8.6	108
		Final Chemistries			
Influent	7.84	7.80-7.89	26.0	8.3	--
Effluent	8.11	8.05-8.18	26.0	8.3	--
Control	8.20	8.20-8.21	26.4	8.4	--

[a]means based on 10 replicates
*DO = dissolved oxygen
-- = not measured

(Appendix D, continued)

Fathead minnow acute tests[b] - Orlando site

Sample	Mean pH	pH Range	Mean Temp (°C)	Mean DO (mg/L)	Mean Conductiv. (umhos/cm)
			Initial Chemistries		
Influent	7.20	7.17-7.24	25.8	8.5	487
Effluent	7.20	7.14-7.27	25.5	8.8	399
Control	7.99	7.87-8.11	26.1	7.9	131
			Final Chemistries		
Influent	7.86	--	25.0	7.6	--
Effluent	7.96	--	25.2	7.8	--
Control	8.01	--	25.1	7.9	--

Note: a fathead minnow test was not conducted on the Lakeland sample.

[b]means based on two replicates
*DO = dissolved oxygen
-- = not measured

APPENDIX E. Bird Species Lists Based on Ground Counts and Inventories.

Bird species observed during informal ground counts and field sampling at the Lakeland site, 1991-1992. *=birds observed on 5 informal bird counts from November 1991 to June 1991 and during field sampling in 1991; + = birds observed only during field sampling.

* Pied-billed Grebe
American White Pelican
* Double-crested Cormorant
* Anhinga
American Bittern
* Least Bittern
* Great Blue Heron
* Great Egret
* Snowy Egret
* Little Blue Heron
+ Tri-colored Heron
* Cattle Egret
* Green-backed Heron
* Black-crowned Night Heron
* White Ibis
Glossy Ibis
Roseate Spoonbill
Wood Stork
Hooded Merganser
* Wood Duck
Mottled Duck
Mallard
Pintail
Blue-winged Teal
Black Vulture
* Turkey Vulture
* Osprey
Bald Eagle
Northern Harrier
Red-shouldered Hawk
American Kestrel
* Northern Bobwhite
* Common Moorhen
American Coot
Killdeer
Black-necked Stilt
Greater Yellowlegs
Lesser Yellowlegs
Spotted Sandpiper
Ring-billed Gull
Common Tern
+ Gull-billed Tern
* Mourning Dove
Common Ground Dove
* Belted Kingfisher
Eastern Phoebe
+ Eastern Kingbird
Tree Swallow
+ Barn Swallow
Blue Jay
American Crow
Tufted Titmouse
American Robin
* Northern Mockingbird
* Loggerhead Shrike
+ White-eyed Vireo
Yellow-throated Warbler
* Northern Cardinal
+ Rufous-sided Towhee
* Red-winged Blackbird
Eastern Meadowlark
* Boat-tailed Grackle
Common Grackle

Bird species recorded throughout the period of site operation (1987-1991) at the Orlando site (list provided by Post, Buckley, Schuh, and Jernigan, Inc., Winter Park, FL). * = bird species seen or heard during field sampling in 1991.

*Pied-billed Grebe
American White Pelican
*Double-crested Cormorant
*Anhinga
American Bittern
*Least Bittern
*Great Blue Heron
*Great Egret
*Snowy Egret
*Little Blue Heron
*Tricolored Heron
*Cattle Egret
*Green-backed Heron
Black-crowned Night-Heron
Yellow-crowned Night-Heron
*White Ibis
Glossy Ibis
Roseate Spoonbill
Wood Stork
Fulvous Whistling-Duck
Wood Duck
Green-winged Teal
*Mottled Duck
Mallard
Northern Pintail
Blue-winged Teal
Northern Shoveler
Gadwall
American Wigeon
Canvasback
Ring-necked Duck
Lesser Scaup
Common Goldeneye
Hooded Merganser
Ruddy Duck
*Black Vulture
*Turkey Vulture
Osprey
American Swallow-tailed Kite
*Snail Kite
Bald Eagle
Northern Harrier
Sharp-shinned Hawk
*Red-shouldered Hawk
Short-tailed Hawk
American Kestrel
Merlin
Peregrine Falcon
Wild Turkey
Northern Bobwhite
King Rail
Sora
Purple Gallinule
*Common Moorhen
*American Coot
*Limpkin
Sandhill Crane
Black-bellied Plover
Killdeer
American Oystercatcher
Black-necked Stilt
Greater Yellowlegs
Lesser Yellowlegs
Solitary Sandpiper
Spotted Sandpiper
Least Sandpiper
Dunlin
Long-billed Dowitcher
Common Snipe
Ring-billed Gull
Caspian Tern
Forster's Tern
*Mourning Dove
Common Ground-Dove
Common Barn-Owl
Eastern Screech-Owl
*Barred Owl
Common Nighthawk
Chuck-will's-widow
Chimney Swift
*Belted Kingfisher
Red-headed Woodpecker
*Red-bellied Woodpecker
Yellow-bellied Sapsucker
Downy Woodpecker
Northern Flicker
*Pileated Woodpecker
Eastern Phoebe
Great Crested Flycatcher
Eastern Kingbird
Purple Martin
Tree Swallow
*Barn Swallow
Blue Jay
American Crow
*Fish Crow
*Tufted Titmouse
Carolina Wren
House Wren
Sedge Wren
Marsh Wren
Ruby-crowned Kinglet
Blue-gray Gnatcatcher
Hermit Thrush
American Robin
Gray Catbird
Northern Mockingbird
Brown Thrasher
Water Pipit
Cedar Waxwing
*Loggerhead Shrike
European Starling
White-eyed Vireo
Solitary Vireo
Orange-crowned Warbler
Northern Parula
Yellow-rumped Warbler
Yellow-throated Warbler
Pine Warbler
Prairie Warbler
Palm Warbler
Black-and-white Warbler
American Redstart
Prothonotary Warbler
Ovenbird
*Common Yellowthroat
*Northern Cardinal
Indigo Bunting
Rufous-sided Towhee
Savannah Sparrow
Henslow's Sparrow
Song Sparrow
Swamp Sparrow
Bobolink
*Red-winged Blackbird
Eastern Meadowlark
*Boat-tailed Grackle
Common Grackle
Brown-headed Cowbird
American Goldfinch
House Sparrow