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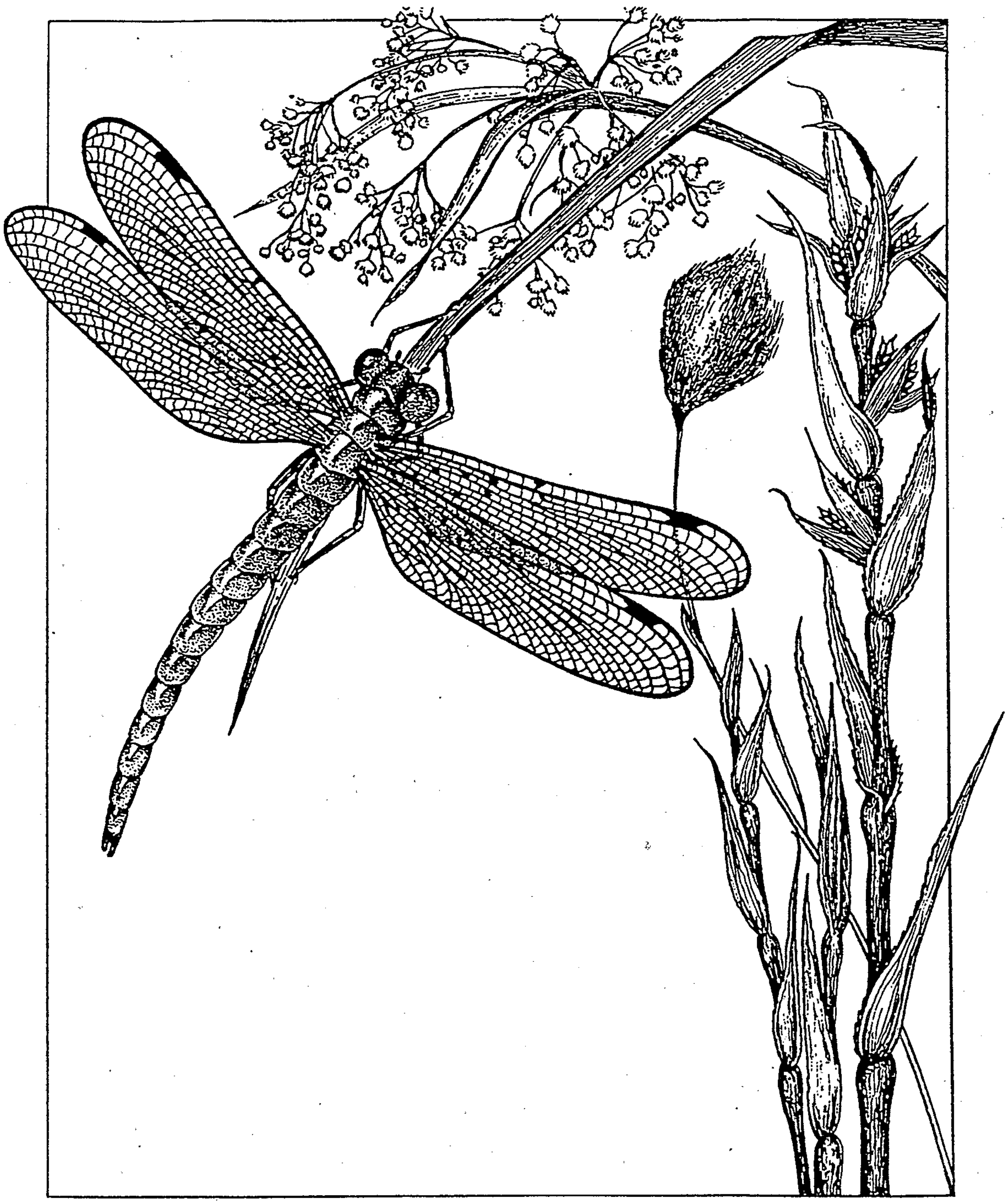
HABITAT QUALITY ASSESSMENT OF TWO WETLAND TREATMENT SYSTEMS IN MISSISSIPPI - A PILOT STUDY



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

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DISCLAIMER

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EXECUTIVE SUMMARY

The use of wetland treatment systems (WTS), or constructed wetlands, for treating municipal wastewater is increasing in the United States, but little is known about the ability of these systems to duplicate or sustain wetland functions. The pilot study was designed to examine methods and the usefulness of various wetland indicators for assessing the wildlife habitat quality in six WTS sites throughout the United States. This report focusses on two of those sites, one located near Collins and one near Ocean Springs, MS.

Vegetation, invertebrate, site morphology, water quality, and bird data were collected in the field or compiled from existing data sets. Various metrics were calculated as indicators of wetland condition and assessed for their utility in characterizing wildlife habitat quality. Wildlife habitat quality was assessed mainly with respect to birds. 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 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 gross deviations from indicator values from non-WTS. In addition, whole-effluent toxicity tests were conducted on influent and effluent water samples from each WTS. As an alternative to indicator analysis, the Wetland Evaluation Technique (WET) was tested for its effectiveness and reliability in assessing various wetland functions, including wildlife habitat, in WTS.

Comparisons of indicators for which data on non-WTS existed showed that indicator values from WTS were generally within the range of values found in non-WTS. Bird density, biochemical oxygen demand, and ammonia-nitrogen were above the range of values reported for non-WTS. The data suggest that the habitat condition of the two WTS studied is not grossly different than that of the general population of wetlands in the same region. The preliminary results, however, do not indicate actual habitat value because little is known about the habitat quality of non-WTS used in the comparisons.

In whole-effluent toxicity tests on *Ceriodaphnia dubia*, there were no significant effects on survival or reproduction in the Ocean Springs samples or the Collins effluent sample. However, 100% mortality occurred in influent water samples from the Collins site. Survival of fathead minnows (*Pimephales promelas*) was not significantly reduced in any samples. Determining the precise cause of the mortality of *Ceriodaphnia* and whether it is a risk to other forms of wildlife would require further testing of water, sediment, or animal and plant tissue.

Results of this study provide evidence that the WTS studied provide wildlife habitat as an ancillary benefit. For future assessment of wildlife habitat quality in WTS, it is recommended that indicators from the following categories be further tested and developed:

- o vegetation
- o invertebrates
- o site morphology

Invertebrate sampling should be expanded to include benthic invertebrates. Bird use is one potential 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 wetland condition. Use of existing water nutrient data, whole-effluent toxicity tests, and the WET analysis should be discontinued or should have low priority.

Water quality data is variable and difficult to interpret in terms of wildlife habitat. The collection of a smaller set of nutrient parameters, including dissolved oxygen and ammonia nitrogen, might be considered as part of field sampling (i.e., not acquired from existing data sets) to aid in interpretation of other data collected.

Whole-effluent testing does not provide information on effects to food chain function that may occur through bioaccumulation or on effects on wildlife of specific substances. Because documentation of effects is a long-term process and can become very expensive, effort should be focussed on contaminant testing of sediments and plant and animal tissues only in selected higher risk wetlands (e.g., those that receive some industrial inputs, where contaminants have been found in the past, or where routine biological monitoring indicates a potential problem).

For comparing wildlife habitat quality of WTS to non-WTS, future studies should include sampling 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 some guidelines for rating habitat quality on a continuum from high to low. 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. To take advantage of this purifying function, wetlands are often built specifically to treat water. 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 lead ... in the construction of wetlands for treating ...ucted wetland treatment systems ... are engineered complexes of ... and submergent vegetation, animal ... natural wetlands for the primary ...nt (Hammer and Bastian 1989). These ...ated wastewater and are designed to ...and (BOD), nutrient and metals con... ...her pollutants (Kadlec and Kadlec ... Wetland treatment systems fall into ...vegetated submerged-bed wetlands, in ...nonsoil substrate in the bed of the ... with plant roots; and 2) free water ...ost of the water flow is above ground ... 1988a, Reed et al. 1988).

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...ands, the focus of this study, are ...eral sections, or cells, separated by weirs which can ... 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 allows respiration by microorganisms. The most effective WTS are marshes that support herbaceous emergent and submergent plants (Hammer and Bastian 1989). Wetland treatment systems are being 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).

EPA Role

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. In addition to water quality, the general habitat quality and potential for risks to wildlife by substances entering in wastewater are of concern to the EPA (Davis and Montgomery 1987).

The ecological condition and habitat quality of WTS is of concern to the EPA because these systems attract wildlife and cannot be considered isolated operations. It is important for the EPA to develop methods for assessing and monitoring the ecological condition of WTS and to coordinate these methods with methods used for natural, restored, and created wetlands.

Assessing Wetland Function and Ecological Condition

Wetland treatment systems can duplicate structural aspects of some natural wetlands, but little is known about the replication of wetland functions. Wetlands usually perform one or more functions, depending upon their type, location, the local geology, topography, and hydrology, and other characteristics of the watershed. 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 for wastewater treatment and not necessarily 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 ecosystem viability, sustainability, and ability to serve one or more 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).

Use of Indicators

Ecological condition can be assessed and monitored on the basis of various wetland attributes, or indicators. Indicators are attributes which can be measured and used to assess and monitor ecological condition and to identify potential problems or failures (e.g., eutrophication, low species diversity, contamination, food chain malfunction). 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 efficient to identify a suite of indicators that best describes the overall condition of the resource.

Use of the Wetland Evaluation Technique

Rapid assessment techniques are another option for evaluating wetland ancillary functions and values. One of these techniques is the Wetland Evaluation Technique (WET) (Adamus et al. 1987). The evaluation is based on a computer analysis of answers to yes/no questions about a wetland. WET rates functions and values in terms of social significance, effectiveness, and opportunity. WET evaluates these three parameters by characterizing a wetland in terms of physical, chemical, and biological attributes, taking into account both internal, site-specific attributes and characteristics of the surrounding landscape (Adamus et al. 1987). Social significance indicates the value of a wetland to society based on its special designations, potential economic value, and strategic location. This assessment is designed to determine whether the wetland has specific characteristics that indirectly indicate it may be performing functions and values beneficial to society. Effectiveness indicates the capability of a wetland to perform a function due to its physical, chemical or biological characteristics. Opportunity indicates the chance a wetland has to perform a function based on inputs from the surrounding landscape (Adamus et al. 1987).

Ratings are based on the probability that the wetland serves a particular function. The following functions and values are rated:

Groundwater recharge
Groundwater discharge
Floodflow alteration
Sediment stabilization
Sediment/toxicant retention
Nutrient removal/transformation
Production export
Wildlife diversity/abundance
Wildlife diversity/abundance - breeding
Wildlife diversity/abundance - migration

Wildlife diversity/abundance - wintering
Aquatic diversity/abundance
Uniqueness/heritage
Recreation

WET was designed primarily for conducting an initial, rapid evaluation of wetland functions and values. The method is not intended to produce definitive ratings of wetland functions. The ratings represent only the likelihood that the functions are present. WET is intended to be used as a decision-making tool that is only one piece of the wetland evaluation process, so it does not replace the need for professional opinion and the use of other evaluation methods.

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 nutrients 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. The 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 habitat quality.

Wildlife use of WTS can expose animals 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. Some organisms can be affected through exposure, ingestion, or bioaccumulation of these substances. In some places, viral or bacterial diseases such as avian botulism can be promoted by draw-downs and other hydrologic manipulations. Detailed information about potential effects of wastewater on wetland animal communities, however, 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, D.C.). Because the use of WTS is increasing, knowledge of the ecological functions, ancillary roles, and potential problems of these systems is needed. It is also important to assess the level of sustainability of these systems as wildlife habitat over the long term.

This pilot study was designed as an exploratory effort for examining research methods, indicators, logistics, and capabilities. It was designed as a preliminary assessment of the wildlife habitat quality in WTS. It 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 can be used to design future research on the ancillary values of WTS.

The objectives of the study were to:

- assess the utility of methods and indicators for evaluating the wildlife habitat quality of WTS,
- identify any major differences in values of wildlife habitat indicators in WTS and non-WTS, and
- 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. A 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) Mississippi sites (this report); 2) Florida sites (titled Habitat Quality Assessment of Two Wetland Treatment Systems in Florida--A Pilot Study); and 3) western sites (titled Habitat Quality Assessment of Two Wetland Treatment Systems in the Arid West--A Pilot Study). This report presents results from only the Mississippi sites.

Pilot Study Overview

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

Site Selection

Six free water surface municipal WTS in the United States were chosen for sampling in 1991. The six sites, listed in Table 1, were chosen based on the following criteria:

- location in the arid and semi-arid West or the Southeast 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,
- the availability of water quality data for use in indicator analysis,
- permission to use the site, and
- interest of site operators and other groups in collaboration.

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

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

Assessment of Habitat Quality

Two general assessment techniques were evaluated for use in assessing wildlife habitat quality as an ancillary benefit. First, selected indicators of habitat quality were measured. Indicator data were acquired in the field, from existing data sets, and from aerial photographs. Second, an evaluation of wetland ancillary values, including function as wildlife habitat, was performed at half of the sites sampled using the Wetland Evaluation Technique (WET) (Adamus et al. 1987). 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 wetland dependent, thus more literature exists on wetland habitat requirements of birds than of mammals. 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).

Measurement of Indicators

A suite of indicators was chosen for measurement at the WTS sampled. Indicators were selected based on the likelihood that:

- o sample collection, processing, and labor costs would not exceed budget constraints,
- o data collection would not exceed available human resources,
- o adequate data could be collected within the 4-5 days spent at each site,
- o 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 site and consistent among sites.

Some of these criteria were unknown for some of the candidate indicators. One of the objectives of the study 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 are listed in Table 2. They are grouped into one of three data source categories:

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

Evaluation of Ancillary Values Using WET

The Wetland Evaluation Technique (WET) (Adamus et al. 1987) was conducted at three of the six sites - Collins, MS, Ocean Springs, MS, and Show Low, AZ. The intent was to test its utility in assessing wetland ancillary functions, including wildlife habitat value. WET was given low priority in the pilot study, and limited time at some of the larger sites prevented its completion.

Sampling Schedule

Field data were collected during July and August, 1991. Table 3 shows the field sampling schedule.

Table 2. Indicators of wetland habitat condition measured during the 1991 pilot study.

A. Indicators measured in the field:

Ecological Component	Indicators
Vegetation	-Species composition and percent coverage -Structural diversity and dominance -Species dominance -Species richness
Invertebrates	-Species and functional group composition and relative abundance -Species richness
Water	-Whole effluent toxicity tests on inflow and outflow
Birds	-Density -Relative abundance -Species richness

B. Indicators taken from aerial photographs:

Ecological Component	Indicators
Site morphology	-Wetland area -Distance of land/water interface in relation to wetland area -Distance of edge between selected cover types in relation to wetland area -Ratio of open water area to area covered by vegetation -Relative coverage of selected vegetation types

C. Indicators obtained from existing data sets:

Ecological Component	Indicators
Water	-pH -Dissolved oxygen -Biochemical oxygen demand -Total suspended solids -Ammonia nitrogen -Total Kjeldahl nitrogen -Total phosphorus -Fecal coliform bacteria

Table 3. 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

Mississippi Study

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

Site Descriptions

The general locations of the Mississippi sites are shown in Figure 1, and site design and arrangement of cells 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.

Collins, MS. The wetland, constructed in 1987, covers approximately 4.47 ha (10 acres) and supports primarily marsh vegetation. It contains two cells separated by a dike. The wetland treats domestic wastewater, which has received pre-treatment in an adjacent lagoon. Water enters the constructed cells and flows in a serpentine fashion around several dikes that protrude into the wetland. The first cell supports shoreline emergent vegetation and duckweed (*Lemna* spp.), whereas the second cell is heavily vegetated throughout with *Scirpus* spp. Average flow is 1514 m^3/day (0.4 mgd). Effluent meets water quality standards for tertiary treatment and flows into the nearby Okatoma Creek.

Ocean Springs, MS. The site supports two separate WTS - Phase I and Phase II. Phase II was planted in late 1990 and did not satisfy the criterion that the site be in operation for at least one year. Therefore site assessment was done only in Phase I. Phase I contains 2 cells and covers 9.28 ha (22 acres). The wetland, constructed in 1990, was designed for tertiary treatment of domestic wastewater from the town of Ocean Springs.

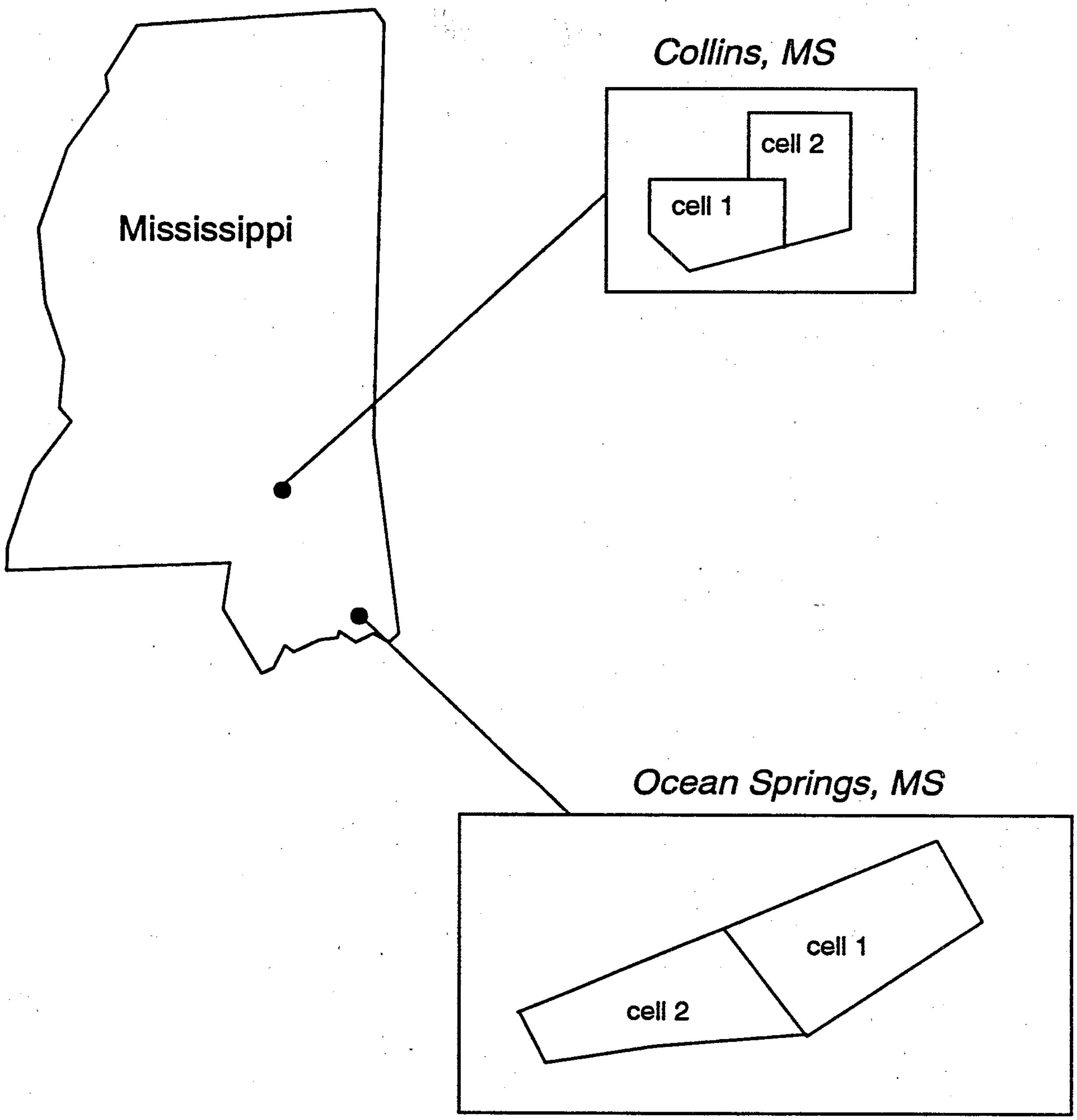

Figure 1. Location and general design of constructed wetland sites studied in Mississippi (Only Phase I of the Ocean Springs site is shown).

Approximate flow is 3610 m^3/day (0.95 mgd). The site supports marsh vegetation, primarily monospecific stands of *Typha* spp. and *Scirpus* spp. The WTS is located on the Mississippi Sandhill Crane National Wildlife Refuge and is operated under a 20-year memorandum of understanding between the Mississippi Gulf Coast Regional Wastewater Authority and the U.S. Fish and Wildlife Service (Hardy 1989).

Field and Laboratory Methods

This section describes the methods for all activities conducted during the field season in July and August, 1991 (Table 3), as well as laboratory analysis of water and invertebrate samples. The activities described are: site characterization; vegetation, invertebrate, water, and bird sampling; invertebrate and water laboratory analyses; and evaluation of ancillary values. Indicators measured in the field or calculated are listed in Table 2.

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 site, and recording wildlife species observed.

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 1) verify air photo interpretation, and 2) ensure that vegetation transects could be sited representatively.

Vegetation Sampling

Vegetation sampling was determined to be the highest priority task for the field study. It included transect establishment through major cover types at each site, cover estimation at points along transects, plant specimen preservation, and identification of unknown plants by local botanists. Data collected were used to calculate indicators listed in Table 2.

Transect establishment

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

- o parallel to the gradient of wastewater treatment, so that data could be stratified by wetland cell

- o through the major vegetation strata in each cell. 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

- o to intersect the 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.

At least two transects were established at each WTS site, beginning at the influent end and continuing toward the effluent end when possible. 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 unless a transect intersected an island within the wetland. Spacing of sampling points along transects depended on the length of the transect, the size of the wetland, and the sizes of vegetation patches along the transect. We sampled at least 40 points per WTS. Transect locations are shown on site maps in Appendix A.

<u>Cover estimation</u>

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

We recorded the scientific names for all species found within each plot and estimated absolute percent cover of each 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, so submerged species were often not recorded but were 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.

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 for 5-m^2 plots was scrub/shrub, and for 10m radius circular plots was forested. In addition, each species observed in plots was assigned to one of the following structural types (or layers):

- submerged
- emergent (or herbaceous)
- scrub/shrub
- forested
- floating-leaved
- dead.

Plant specimen preservation and identification

Unknown plants were collected, coded, and pressed for later identification. Professional botanists who identified unknown plants are listed in Appendix B.

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., Merritt and Cummins 1984, Tucker 1958, Smith et al. 1987, Brooks and Hughes 1988, Jeffries 1989, Voights 1976). This 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 protocols 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 (Wallace, personal communication, Environmental Consultants, Gainesville, FL).

We used rectangular kick nets with #30 mesh. 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.

Two people sampled each cell/habitat simultaneously. Effort was divided between the two members by dividing areas to be sampled in half. Sweeps were made along the wetland bottom, 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 an enamel pan, 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. Each person picked invertebrates for 30 minutes, which resulted in a 1-hour collection period for each sampling point. When all individuals had been picked from the sample, the timer was stopped while a new net sample was obtained.

When sample densities were very high, a representative proportion of each species in the nets was estimated and collected to comprise the sample. Thus, instead of spending excessive time collecting every individual, more time could be spent obtaining a new sample to try to obtain new species.

Invertebrates were identified by biologists at the University of Minnesota-Duluth (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 into 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 in plastic cubitainers at the inflow and outflow of each wetland and shipped on 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 determine whether toxicity is a problem at the sites studied so that it can be examined in future studies.

Upon arrival of water samples at ERL-Duluth, the following routine measurements for whole effluent toxicity testing were taken: alkalinity, hardness, ammonia, total residual chlorine, and temperature. Standard laboratory operating procedures of the National Effluent Toxicity Assessment Center (ERL-Duluth) were used (US EPA 1988b). 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 Mississippi wetlands 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 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.

Bird Use

Data on bird use of the wetlands were acquired from surveyors from the University of Southern Mississippi (Appendix B), who conducted systematic surveys weekly in 1991 during summer (May 29-June 26) and fall (October 16-November 13). Five surveys were conducted during each period. Counts were made from fixed survey points which were chosen to maximize visual coverage of the wetlands. Surveys began no later than 0730 Central Standard Time. For a 10-minute period at each survey point, all birds within a 50-meter radius of the point were identified to species and counted. Double counting of the same individuals from different survey points was avoided by placing survey points $\geq$ 100 m apart and/or by mentally keeping track of, as much as possible, which birds were already counted. Counts included birds within or flying immediately above the wetlands. The locations of survey points are shown on site maps in Appendix A.

Evaluation of Ancillary Values

The Wetland Evaluation Technique (WET) (Adamus et al. 1987) was conducted to test its utility in assessing wildlife habitat quality and other ancillary functions of WTS when applied in conjunction with field sampling. The WET evaluation was conducted as the final component of field work at each site because many of the questions require knowledge about the site that can be acquired during sampling. Many of the questions should be answered in the field, while others require the use of maps and soil surveys and consultation with local people familiar with the region. Two people answered the questionnaire together so that questions could be discussed and answered most accurately. Site managers were consulted about questions that could not be answered without additional knowledge of regional physical, geographical, climatological, and seasonal patterns.

Site Morphology

Color infrared photographs were taken of each site in summer 1991 by a local aerial survey company (Appendix B). One photo was taken of the Collins site at a scale of 1:3600; three overlapping photos were taken of Phase I at the Ocean Springs site at a scale of 1:4200. 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 ones could be consistently resolved based on different colors, shades, and textures on photographs and on ground truth mapping done during reconnaissance. Table 4 lists the cover types delineated at each site.

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. If floating-leaved

Table 4. Cover types delineated on air photos

Collins	Ocean Springs
Typha latifolia	*Typha latifolia*
Scirpus californicus	*Scirpus validus*
Alternanthera philoxeroides	*Pontederia cordata*
Polygonum punctatum	*Sagittaria lancifolia*
Small floating-leaved	Small floating-leaved
Mixed emergents	Open water
Open water	

plants could not be seen on the photos due to dense growth of overstory species, they were not included in polygons (even if it was highly probable that they were present). 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 2 (B). Calculations are described in the Data Analysis section below.

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 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. Water samples are collected 4-5 times per month from the influent and 7-10 times per month from the effluent at the Ocean Springs site and quarterly from the influent and effluent at the Collins site. Sampling schedules differ for the water parameters analyzed, so the set of parameters is not the same for every collection date. Samples from the Collins site are analyzed by Culpepper Laboratory in Jackson, MS; those from the Ocean Springs site are analyzed by the Mississippi Gulf Coast Regional Wastewater Authority Laboratory in Pascagoula, MS. Data from Ocean Springs are from October 1990 through September 1991; those from Collins are from December 1987 through June 1988 and, less completely, from July 1988 through September 1991. Table 5 shows the parameters for which data were available at each site.

Data Analysis

Vegetation, invertebrate, and site morphology data were summarized for each wetland and for each cell within the wetland by calculating descriptive statistics. Analysis of data for each cell was intended to show potential patterns in indicator values

Table 5. Water quality data available 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
Ocean Springs	x	x	x	x	x	x	x	x
Collins	x	x	x	x	x		x*	x

*in the form of total phosphate

along a wastewater treatment gradient. Bird counts were totaled for each wetland and are not reported by survey point.

Vegetation, invertebrate, and bird data were summarized using the Paradox database system. Programs were written in PAL, a Paradox database programming language. Water quality data acquired from site managers were analyzed with the Statistical Analysis System (SAS). 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 indicator (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 within a site. Species richness was defined as the total number of species sampled at a site. Average percent coverage for 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. Relative abundance of invertebrate species was calculated by totaling the number of individuals of each species and dividing by the total number of individuals of all species combined. The percentage of total number of individuals belonging to each functional group (percent relative abundance) was calculated similarly. The number of invertebrates collected per person hour was calculated for each cell and habitat type. Species richness was defined as the total number of species collected at each site.

Counts of each bird species were totaled for all survey points for the five surveys in each of the summer and fall periods. The average number of birds per visit during each survey period was calculated for each species by summing the totals for all stations and dividing by the number of visits (N=5). Calculation of an average count per visit eliminates biases due to multiple counting of the same individuals in successive weeks. Species richness was calculated by totaling the number of species detected during both survey periods. Relative abundance of bird species for each site and survey period was calculated by dividing the total counts of each species from all survey points by the total number of birds counted. Average density was calculated by summing the average number of each species per visit and dividing by wetland area as calculated from aerial photographs using ARC/INFO.

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

- o Wetland area was measured as the area within surrounding dikes.
- o 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.
- o Length of shoreline (land/water interface) was divided by wetland area to normalize the shoreline irregularity estimate.
- o Distance of cover/cover interface is the length of edge between cover types and is a measure of cover type interspersion.
- o 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.

Answers to the questions from the WET assessment were entered into a data set and run through the WET computer analysis to classify a wetland according to function (Adamus et al. 1987). The result was an assignment of a qualitative probability rating of high, moderate, or low to each function in terms of social significance, effectiveness, and opportunity. Results were interpreted qualitatively, using descriptions of interpretation in the WET manual as guidelines (Adamus et al. 1987). The evaluation of social significance normally consists of two levels of assessment. Level 2 is an optional step to refine the probability rating for uniqueness/heritage. It was therefore recommended that only level 1 be completed for this study (Adamus, personal communication, ManTech Environmental Technology, Inc., Corvallis, OR).

Literature Review

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. The two groups of data 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 gross 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). Quality assurance information was not available from bird surveyors. However, the same procedures were used at both 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 assurance/quality control (QA/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 QA/QC data was therefore unnecessary and was not done. However, the procedures for collection and evaluation of QA/QC vegetation data are discussed below. 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.

The following procedures were used to collect and evaluate QA/QC vegetation data. At 10% of sampling plots a QA/QC check was performed to determine how similarly the two field crew members were estimating percent coverage and identifying species. The decision to make a plot a QA/QC plot was usually made while sampling the plot just before it. Each person had 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 QA/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 QA/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.

Data QA/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 QA check. It was computed for each QA/QC sample jar by calculating the ratio of invertebrate taxa jointly observed to the total taxa observed and multiplying by 100. To compute count comparability for each QA/QC sample jar, the percent relative difference between the two people for each jointly recorded taxon was computed by subtracting one person's count from the other's and taking the absolute value. This value was then divided by the mean of the two counts, and the result was multiplied by 100. The percent relative difference was then subtracted from 100 to obtain count comparability. The mean percent count comparability was then calculated for the whole QA sample jar. Count and identification comparability for a whole site were obtained by calculating the means of all QA/QC sample jar means.

The mean identification comparability for invertebrates was 94.0%, and the mean count comparability was 99.0%. The values meet the data quality objectives of $\geq$85% established prior to the study for identification and count comparability. As previously mentioned, all vegetation percent cover estimates were made by both crew members, so evaluation of QA/QC data was not necessary in this study.

Quality assurance procedures were not used to evaluate the precision or accuracy in the identification, delineation, or digitizing of habitat types on aerial photographs. 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 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 assurance procedures were obtained from the laboratories that analyze water samples collected at the constructed wetland sites. It was decided, however, that a careful inspection of the data and quality assurance procedures was not necessary for pilot study objectives. Criteria for assessing data quality had not been developed, and the process would have been very time-consuming. Data were intended to be used regardless of laboratory protocols and measurement consistency among testing labs. Results of water quality analyses suggested that water quality should probably not be continued as an indicator of constructed wetland condition. One reason for this is that the effort and difficulty involved in collecting necessary information from each water lab and assessing data quality and consistency among labs would probably be too cumbersome for future studies. 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.

RESULTS AND DISCUSSION

Summary data are presented separately for each indicator group for each WTS. Results from the Ocean Springs site are for Phase I only. Discussion addresses 1) indicator and WET 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.

Summaries of Indicator Data

Vegetation

Plants sampled at the Collins site belonged to three structural layers: emergent, floating-leaved, and dead. Those at the Ocean Springs site belonged to four different layers - emergent, scrub/shrub, floating-leaved, and dead. The scrub/shrub component, however, was very small (1% average coverage per square meter) (Table 6) and probably does not contribute significantly to wildlife habitat value. Floating-leaved vegetation was the most dominant structural layer at both the Collins and Ocean Springs sites, but the emergent layer had the highest species richness (Table 6). The dead category at both sites was composed primarily of persistent emergent vegetation (*Scirpus* and *Typha*). Dead vegetation was considered in this study because it can contribute cover for waterfowl or nesting habitat for passerines that is different from that offered by live plants of the same species.

Emergent and floating-leaved plants were interspersed throughout the Ocean Springs WTS, but structure above the water surface was relatively uniform. At the Collins site, some structural diversity existed, but structural types were not well-interspersed. Cell 1 consisted primarily of open water, *Lemna* spp. and *Alternanthera philoxeroides*, while cell 2 consisted of robust *Scirpus californicus* and small floating-leaved species. Most of the non-dominant species were located on the periphery of the wetland. 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 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).

Table 6. Percent of the total number of plant species on each site and average percent cover per square meter comprising each vegetation structural layer at the Collins and Ocean Springs sites, 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
Collins site		
Emergent	77	26 ± 43.1
Floating-leaved	23	89 ± 79.7
Dead	--	17 ± 33.4
Ocean Springs site		
Emergent	79	39 ± 35.9
Floating-leaved	16	103 ± 82.0
Scrub/shrub	5	1 ± 3.4
Dead	--	25 ± 27.6

A structural component that is absent at both sites is submerged vegetation, which usually provides habitat for fish and invertebrates, which in turn are eaten by waterfowl and wading birds. The numbers and weights of macroinvertebrates per unit weight of macrophyte are positively related to the amount of surface area available as a substrate and the degree of leaf dissection (Krull 1970, Dvorak and Best 1982, Biochino and Biochino 1980). Floating-leaved plants at the WTS serve as habitat and substrate for fish and invertebrates to an extent, but substrate area is limited to the surface layer of the water column.

The particular species of plants are important when considering wildlife food preferences. Plant species sampled at each site and their average percent coverages per square meter are listed in Table 7. Average percent coverage estimates of plant species at the Ocean Springs site are likely to be slightly high because deepwater areas between subcells where no plants grew were not sampled (see Appendix A). 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 at both WTS were primarily floating-leaved species (Table 7), which are an important food item for many species of waterbirds. Also dominant at the Ocean Springs site were *Typha latifolia* and dead emergents. *Scirpus californicus* and dead emergents (composed primarily of *Scirpus*) were dominants only in cell 2 at the Collins site, but their percent coverage values for the whole site were also quite high - 19% and 17%, respectively (Table 7). *Typha* and *Scirpus* are important as cover for many species of birds but are consumed by few species of wildlife. All other species at both sites had average coverages of 8% or less.

Species richness (the number of species sampled) was 13 at the Collins site and 19 at the Ocean Springs site. For comparison, Table 8 shows species richness for several non-WTS marshes in Georgia and Florida. The numbers of plant species range from 9 to 68. 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 for 18 created and natural non-WTS in Florida in 1988; values ranged from 13 to 93 species. Species richness values from the WTS studied are generally at the low end of the range of the data compiled for non-WTS sites in the southeast region. The species of plants that can survive on a long-term basis in WTS is often dependent on varying tolerances to eutrophication (Hartog et al. 1989) and permanent flooding (Farnez and Bookhout 1982, Sjoberg and Danell 1983). In addition, native and perennial species, especially grasses and

Table 7. Frequency of occurrence, average percent cover per square meter ± standard deviation, and dominance indication (*) for each plant species sampled at the Collins and Ocean Springs sites, 1991. 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. Frequency of occurrence is the percent of sample points at which each species was present. Average percent coverage was rounded to zero if it was below 0.5%. The sum of percent coverages at a site can exceed 100 since species can have overlapping coverages. The total of number of points sampled was 59 at the Collins site and 52 at the Ocean Springs site.

Collins site

Species	Frequency of Occurrence	Average percent cover/m^2 +/- Std. Dev.	Dominance
Emergent			
Andropogon spp.	2	0 ± 0.1	
Digitaria sanguinalis	2	0 ± 2.6	
Echinochloa walteri	7	2 ± 7.4	
Justica ovata	5	1 ± 2.2	
Paspalum dilatatum	2	2 ± 11.7	
Paspalum notatum	2	0 ± 0.7	
Polygonum punctatum	5	1 ± 2.9	
Scirpus californicus	24	19 ± 35.3	
Typha latifolia	3	2 ± 12.0	
Unidentified grass	2	1 ± 8.5	
Floating-leaved			
Alternanthera philoxeroides	41	23 ± 35.6	*
Lemna spp.	97	47 ± 40.3	*
Wolffia spp.	24	20 ± 36.5	*
Dead			
Emergent	29	17 ± 33.4	

Table 7, continued

Ocean Springs site

Species	Frequency of Occurrence	Average percent cover/m^2	Dominance
Emergent			
Amphicarpum muhlenbergianum	4	1 ± 7.0	
Echinochloa walteri	4	1 ± 3.9	
Eclipta alba	2	0 ± 0.7	
Leptochloa fascicularis	2	0 ± 0.7	
Lilaeopsis caroliniana	4	4 ± 18.4	
Ludwigia decurrens	6	0 ± 1.9	
Polygonum densiflorum	8	1 ± 5.1	
Polygonum hydropiperoides	6	0 ± 2.0	
Rhynchospora inundata	4	0 ± 1.5	
Sacciolepis striata	2	1 ± 4.9	
Sagittaria lancifolia	2	1 ± 4.9	
Scirpus cyperinus	2	0 ± 0.1	
Scirpus validus	25	8 ± 19.3	
Typha latifolia	62	22 ± 25.6	*
Unidentified grass	2	0 ± 0.1	
Floating-leaved			
Lemna spp.	71	54 ± 43.0	*
Spriodela polyrhiza	35	26 ± 39.7	*
Wolffia spp.	33	23 ± 37.9	*
Scrub-shrub			
Salix nigra	8	1 ± 3.4	
Dead			
Emergent	79	25 ± 27.9	*
Submergent	2	0 ± 2.8	
Scrub/shrub	4	0 ± 0.7	

Table 8. Plant species richness at palustrine emergent non-WTS wetland sites in Georgia and Florida, 1983-1990.

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

* herbaceous layer only

sedges, may be replaced by more aggressive exotic, clonal, or annual species (commonly *Typha*, *Scirpus*, *Sagittaria*, and *Pontederia*) when areas are permanently flooded (e.g., Botts and Cowell 1988, McIntyre et al. 1988). Both of the WTS studied are eutrophic and permanently inundated, which might limit the number and species of plants that can persist. In addition, water depth at the Collins site is approximately 0.8-1.0 m through most of the wetland; banks are quite steep and the substrate is a firm clay. These factors can inhibit the establishment of seeds, the propagation of clonal species, or the aeration of roots of many species.

Vegetation is one of the most significant components of wildlife habitat. It is directly and indirectly related to wildlife habitat quality and is a major component of most free-water surface WTS. It has also been used frequently to characterize wetlands and habitat conditions and is often recommended as a wetland monitoring indicator (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). Sampling methods are well-developed, and sampling can be completed during one visit to a wetland. The continued use of indicators of vegetation for the assessment of habitat quality in WTS is recommended. 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. Evaluation of habitat quality should be based to a lesser extent on plant species richness alone. Species richness, however, is an important component of quantifying the overall value of plant species to wildlife at a site.

Invertebrates

Five person-hours was spent sampling invertebrates at the Collins site, and eight person-hours were spent at the Ocean Springs site. The total number of invertebrates collected was 1011 (202.2 per person-hour) at the Collins site and 3220 (402.5 per person-hour) at the Ocean Springs site. Thirty taxa were collected at the Collins site, and 50 taxa were collected at the Ocean Springs site.

Dominant insect orders were Hemiptera at the Collins site, and Hemiptera and Diptera at the Ocean Springs site (Table 9). The Hemipterans consisted primarily of immature Belostomatidae (25.4% of the site total) at the Collins site and *Trichocorixa* (31.7% of the site total) at the Ocean Springs site. Dipterans at the Ocean Springs site were comprised primarily of *Tanypus* larvae (19.6% of the site total). Also abundant were Coleopterans, which consisted primarily of *Tropisternus* at the Collins site (13.7% of the site total) and *Suphisellus* at the Ocean Springs site (11.6% of the site total). Dominant non-insect taxa were Oligochaeta, family Tubificidae, which comprised

Table 9. Aquatic invertebrate taxa and their relative abundances at the Collins and Ocean Springs sites, Mississippi, 1991. Total number of invertebrates collected was 1014 and 3220 at the Collins and Ocean Springs sites, respectively. Functional groups were not assigned for terrestrial, immature, and non-insect invertebrates. Percent relative abundance was rounded to 0.0 if less than 0.05.

COLLINS SITE

Taxon	Count	% Relative Abundance	Functional Group
Odonata			
Libellulidae - Pachydiplax, naiad	1	0.1	Predator
Orthoptera			
terrestrial	1	0.1	
Hemiptera			
Belostomatidae - Belostoma	49	4.8	Predator
- immature	258	25.5	
Corixidae - Trichocorixa	3	0.3	Predator
- immature	1	0.1	
Mesoveliidae - Mesovelia	18	1.8	Predator
Nepidae - Ranatra	2	0.2	Predator
Notonectidae - Notonecta	3	0.3	Predator
terrestrial	1	0.1	
Coleoptera			
Chrysomelidae - terrestrial	2	0.2	
Curculionidae - Tanysphyrus	1	0.1	Shredder
Dytiscidae - Cybister, larvae	14	1.4	Predator
- Hydrovatus	1	0.1	Predator
- Laccophilus	4	0.4	Predator
- Laccophilus, larvae	2	0.2	Predator
Hydrophilidae - Berosus	1	0.1	Piercer/collector
- Enochrus	25	2.5	Piercer
- Enochrus, larvae	4	0.4	Collector
- Paracymus, larvae	2	0.2	Predator
- Tropisternus	40	3.9	Piercer/collector
- Tropisternus, larvae	139	13.7	Predator
Noteridae - Hydrocanthus	76	7.5	Predator
- Hydrocanthus, larvae	17	1.7	Predator
- Suphisellus	13	1.3	Predator
Diptera			
Chironomidae - Chironomous/Einfeldia, larvae	3	0.3	Collector/shredder
- Goeldichironomous, larvae	36	3.6	Collector
- Tanypus, larvae	1	0.1	Predator

Table 9, continued

Taxon	Count	% Relative Abundance	Functional Group
Diptera			
Culicidae - Culex, larvae	1	0.1	Collector
Stratiomyidae - Odontomyia, larvae	27	2.7	Collector/scraper
- Odontomyia/Heriodiscus, larvae	40	3.9	Collector/scraper
Tabanidae - Tabanus, larvae	1	0.1	Predator
terrestrial	2	0.2	
Non-insects			
Oligochaeta			
Haplotaxida			
Tubificidae	220	21.7	
Arachnoidea			
Hyracarina			
Araneae, terrestrial	4	0.4	
OCEAN SPRINGS SITE			
Collembola			
Sminthuridae - Sminthurides	1	0.0	Collector/shredder
Ephemeroptera			
Baetidae - Callibaetis, naiad	1	0.0	Collector
Odonata			
Aeshnidae - Anax, naiad	5	0.2	Predator
Coenagrionidae - Enallagma, naiad	3	0.1	Predator
- Ischnura, naiad	51	1.6	Predator
- Ischnura	1	0.0	Predator
Libellulidae - Erythemis, naiad	3	0.1	Predator
- Plathemis, naiad	2	0.1	Predator
- Orthemis, naiad	6	0.2	Predator
- Pachydiplax, naiad	8	0.2	Predator
- Sympetrum	1	0.0	Predator
Hemiptera			
Belostomatidae - Belostoma	9	0.3	Predator
- immature	87	2.7	
Corixidae - Corisella	2	0.1	Predator
- Trichocorixa	1021	31.7	Predator
- immature	100	3.1	

Table 9, continued

Taxon	Count	% Relative Abundance	Functional Group
Hemiptera			
Mesoveliidae - *Mesovelia*	68	2.1	Predator
Nepidae - *Ranatra*	1	0.0	Predator
Notonectidae - *Notonecta*	1	0.0	Predator
terrestrial	1	0.0	
Lepidoptera			
Pyralidae - *Acentria*	1	0.0	Shredder
- *Crambus*, larvae	1	0.0	Shredder
Coleoptera			
Carabidae - terrestrial	1	0.0	
Chrysomelidae - terrestrial	1	0.0	
Dytiscidae - *Celina*, larvae	1	0.0	Predator
- *Cybister*	4	0.1	Predator
- *Laccophilus*	2	0.1	Predator
- *Uvarus*	1	0.1	Predator
Gyrinidae - *Dineutus*	1	0.1	Predator
Hydrophilidae - *Berosus*	12	0.4	Piercer/collector
- *Berosus*, larvae	39	1.2	Predator
- *Crenitis*, larvae	1	0.0	Predator
- *Enochrus*	36	1.1	Piercer
- *Enochrus*, larvae	4	0.1	Collector
- *Hydrophilus*	1	0.0	Piercer/collector
- *Paracymus*	1	0.0	Collector
- *Paracymus*, larvae	1	0.0	Predator
- *Tropisternus*	5	0.2	Piercer/collector
- *Tropisternus*, larvae	69	2.1	Predator
Noteridae - *Hydrocanthus*	69	2.1	Predator
- *Hydrocanthus*, larvae	51	1.6	Predator
- *Suphisellus*	373	11.6	Predator
Diptera			
Chironomidae - *Chironomous/Einfeldia*	204	6.3	Collector/shredder
- *Goeldochironomous*, larvae	23	0.7	Collector
- *Larsia*, larvae	1	0.0	Predator
- *Tanypus*, larvae	631	19.6	Predator/collector
Culicidae - *Culex*, larvae	13	0.4	Collector
- *Uranotaenia*, larvae	5	0.2	Collector
Stratiomyidae - *Odontomyia/Hedriodiscus*, larvae	23	0.7	Collector/scraper
Syrphidae - *Eristalis*, larvae	1	0.0	Collector
Tabanidae - *Tabanus*	1	0.0	Predator
- *Tabanus*, larvae	2	0.1	Predator
terrestrial, adult	6	0.2	
terrestrial, pupa	8	0.2	

Table 9, continued

Taxon	Count	% Relative Abundance	Functional Group
Non-insects			
Class Oligochaeta			
Haplotaxida			
Tubificidae	53	1.6	
Class Arachnoidea			
Hydracarina			
Aranae, terrestrial	16	0.5	
Class Hirudinea			
Rhynchobdellida			
Glossiphoniidae	4	0.1	
Class Gastropoda			
Pulmonata			
Lymnaeidae - *Lymnaea*	3	0.1	
Physidae - *Physella*	179	5.6	

21.7% of the total invertebrates collected at the Collins site and Gastropoda, family Physidae, which comprised 5.6% of invertebrates collected at the Ocean Springs site. The majority of taxa collected at both sites had relative abundances less than 1%.

Aquatic insect orders not represented in Table 9 are Plecoptera, Neuroptera, Megaloptera, Hymenoptera, and Trichoptera. 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. Although Odonata were represented by 8 species at the Ocean Springs site, they were nearly absent at the Collins site (only 1 individual was collected). Ephemeroptera, Collembola, and Lepidoptera were present only at the Ocean Springs site, each representing less than 0.1% of the total invertebrates collected. Most Ephemeroptera prefer a high concentration of dissolved oxygen; aquatic Collembolans have a spotty distribution and are most common in the early spring or late autumn (Pennak 1978).

Many species of Chironomids tolerate the low oxygen conditions in wetlands. (Adamus and Brandt 1990). Chironomid abundance was high only at the Ocean Springs site, although only two species of Chironomids contributed substantially to the count (Table 9). Numbers and species richness of collected Chironomids might have been low because benthic sampling was not conducted, so many benthic species were missed. Benthic sampling is recommended for future studies to assure accurate estimation of Chironomid abundance. 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).

A large portion of cell 1 at the Collins site contained alligatorweed (*Alternanthera philoxeroides*), which would be expected to be a more suitable substrate for Chironomids. Although few Chironomids were present, Tubificid worms were very abundant, occurring in masses within the floating alligatorweed. Tubificids are usually most concentrated in polluted waters with low dissolved oxygen, and some species are considered indicators of organic pollution (Pennak 1953). Most of the true aquatic Tubificids are able to thrive at low dissolved oxygen concentrations, and many can tolerate complete absence of oxygen for extended periods (Pennak 1953). Enrichment of the water at the Collins site helps to explain the abundance of Tubificids, although dissolved oxygen was not exceptionally low and was never recorded as absent at the site (see Table 23 in the Water Quality section below).

The number of invertebrates collected per hour is related to density. The highest collection rate by far occurred in cell 1A at the Ocean Springs site where water enters the WTS (Table 10). This is likely due to high nutrient concentrations, high productivity, and higher dissolved oxygen concentrations at the inlet to the wetland (see Table 23 in Water Quality section below). Macroinvertebrate abundance normally increases with increasing nutrient concentrations (Cyr and Downing 1988, Tucker 1958). Macroinvertebrate diversity, however, was lowest in cell 1A at the Ocean Springs site (Robert Knight, personal communication, CH2M Hill, Gainesville, FL). This was likely due to extremely high numbers of only a few genera: *Trichocorixa, Tanypus, and Chironomous/Einfeldia.*

The same pattern in abundance, however, was not observed at the Collins site (Table 10). The greatest influence on invertebrate abundance at the Collins site might have been due instead to habitat type. The highest collection rate at the Collins site occurred in a mixed community of plant species in cell 1C (Table 10). Alternatively, invertebrate abundance at the influent might be reduced by the factor affecting *Ceriodaphnia* survival (see Whole effluent toxicity section below). Because the same habitats were not sampled in every cell, the effects of cell number and habitat type are confounded in these analyses. For instance, the lowest abundance occurred in cell 2B in the *Scirpus* habitat, so it is not possible to determine whether the causal factor is the habitat or the degree of water treatment or both. Similarly, *Scirpus* habitat at the Ocean Springs site was sampled in cells 1A and 2A; the high collection rate could have been influenced either by habitat or by enrichment (Tables 10 and 11). 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 might provide information for designing WTS to enhance habitat quality.

Table 12 shows the percent relative abundance of invertebrate functional groups present at the Collins and Ocean Springs sites. A total of 487 invertebrates (48.2%) at the Collins site and 280 (8.7%) at the Ocean Springs site were not assigned functional groups. These invertebrates included immatures, terrestrial invertebrates, and non-insect invertebrates (i.e., Oligocheata, Gastropoda, Hirudinea). The high percentage at Collins was due primarily to Oligochaeta in the family Tubificidae and immature Belostomatidae (Table 9). Because the only adult Belostomatids collected at the Collins site were predators, the immature Belostomatids are likely predators also. Invertebrate communities at both sites were dominated by the predator functional group, while the shredder functional group was the least dominant.

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

Table 10. Number of invertebrates collected per person-hour in each cell at the Collins and Ocean Springs sites.

COLLINS SITE		OCEAN SPRINGS SITE	
Cell	No. invertebrates collected/hour	Cell	No. invertebrates collected/hour
1A	249.0	1A	1105.5
1B	Not sampled	1B	Not sampled
1C	365.0	1C	81.5
1D	Not sampled	1D	Not sampled
2A	114.0	2A	155.5
2B	62.0	2B	Not sampled
		2C	Not sampled
		2D	232.5

Table 11. Number of invertebrates collected per person-hour in each habitat type at the Collins and Ocean Springs sites.

COLLINS SITE		OCEAN SPRINGS SITE	
Habitat type	No. invertebrates collected/hour	Habitat type	No. invertebrates collected/hour
Emergent-*Scirpus*	88.0	Emergent-*Scirpus*	489.7
Emergent-*Typha*	272.0	Emergent-*Typha*	361.0
Emergent-other	226.0	Emergent-other	237.0
Emergent-mixed	265.0		

usually predator-based systems (H. 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.

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 in Lower Mississippi River abandoned channel and oxbow palustrine wetlands in 1984 ranged from 8 to 28 (Lowery et al. 1987). Genera richness for benthic invertebrates in Lower Mississippi River borrow pit palustrine wetlands ranged from 7 to 29 in 1981. The taxon level to which invertebrates are identified, as well as the collection techniques and group of invertebrates collected (e.g., nektonic, benthic) vary, so comparison is difficult. Nevertheless, it appears that genera richness of 30 and 50 for the Collins and Ocean Springs sites, respectively, are within the range of richness in non-WTS wetlands. 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.

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 less likely than birds or mammals to migrate from one wetland to another, they can be sampled in a relatively short time period, and they serve as an indicator of secondary productivity. Macroinvertebrates have been suggested as monitoring indicators by various scientists (Brooks et al. 1989, Brooks and Hughes 1988, Brown et al. 1989, Schwartz 1987, US EPA 1983). Continued development of this indicator for habitat evaluation in WTS is recommended and should include standardization of collection methods, expansion of collection techniques (e.g., sediment 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. 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.

Table 12. Relative abundances of invertebrate functional groups, Collins and Ocean Springs sites, Mississippi, 1991. Terrestrial, immature, and non-insect invertebrates were not assigned functional groups.

Collins Site

Functional Group	Relative Abundance
Not assigned	48.2%
Predator	34.2
Collector/scraper	6.5
Collector	4.1
Piercer/collector	4.1
Piercer	2.5
Collector/shredder	0.3
Shredder	0.1

Ocean Springs Site

Functional Group	Relative Abundance
Predator	56.0
Predator/collector	19.5
Not assigned	8.7
Collector/shredder	6.4
Scraper	5.6
Collector	1.5
Piercer	1.1
Collector/scraper	0.7
Piercer/collector	0.6
Shredder	0.1

Whole Effluent Toxicity Tests

There was no statistically significant toxicity effect at either site for the fathead minnow acute tests. Survival was 95% or more for all samples. Toxic effects on reproduction of *Ceriodaphnia dubia* were not observed or were not significant in the Ocean Springs samples or the Collins effluent sample. In the Collins influent sample, however, 100% mortality occurred within 96 hours in all replicates (Table 13). Measurements of each water sample performed by the Duluth Laboratory upon arrival of water samples are shown in Table 14, and initial and final chemistries for water samples and the controls are shown in Appendix D.

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

Sample	Mean young/original female (confidence interval)	Mean Survival (%)
Collins		
Influent	0 (n/a)	0
Effluent	31.1 (29.5-32.7)	100
Control	29.6 (27.4-31.8)	100
Ocean Springs		
Influent	22.7 (16.7-28.7)	90
Effluent	24.3 (21.3-27.3)	100
Control	19.6 (17.5-21.7)	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)
Collins				
Influent	30	170	5.5	<0.02
Effluent	35	150	6.4	<0.02
Ocean Springs				
Influent	43	167	<1	0.02
Effluent	51	182	<1	<0.02

* TRC=total residue chlorine

The precise cause of mortality of *Ceriodaphnia dubia* in the Collins site influent samples and its implications for wildlife should be further investigated. Wastewater entering the Collins WTS is pre-treated only with a lagoon, so its potential to contain substances harmful to some aquatic organisms could be greater than that for WTS that receive water that has gone through secondary treatment at a wastewater treatment plant prior to entering the wetland.

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 constructed wetlands do not receive water from industries. Although concentrations of toxic substances are likely to be absent or low in WTS, probably the greatest risk to wildlife from substances entering in wastewater, even in low concentrations, is through bioaccumulation. Benthic organisms inhabiting and feeding in contaminated sediments can uptake toxic substances bound in the sediments. Short-term whole-effluent tests of water will not indicate whether bioaccumulation is occurring, and, unless the harmful substance is entering the wetland at the time of sample collection, the test will not detect it. Furthermore, tissue analyses conducted to determine whether bioaccumulation is occurring will not be sufficient unless a connection between tissue levels of contaminants and adverse effects can be established. Nevertheless, it seems wise to monitor contaminant levels in sediments or tissues of invertebrates or fish in wetlands that are suspect (e.g., those that have past histories of user violations) or where the potential for contamination is greater (e.g., wetlands receiving industrial inputs). Determining whether the levels of specific substances found pose risks to higher forms of wildlife through ingestion, exposure, or bioaccumulation is then necessary. Early detection and correction is preferable to remedying a problem once it has occurred.

Although toxicity is a very important issue, it is not one that is related exclusively to wildlife habitat. Depending upon public use of the WTS, toxicity can become a human health issue as well. Whole-effluent toxicity tests are not recommended for future studies of wildlife habitat quality because they do not provide enough information for assessing risk of toxicity or effects of bioaccumulation in a system. The proper procedure for documenting the effects of harmful substances in WTS is a much more lengthy and expensive process than a general assessment of wildlife habitat quality and thus should be a separate activity in selected wetlands suspected as higher risks for the presence of 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 (e.g., 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).

Bird Use

A total of 35 species were detected at both the Collins and Ocean Springs sites during the survey period. Species richness was greater in fall at both sites. Wood ducks were the most abundant bird at the Collins site in both the summer and fall surveys, accounting for 27.5% and 31.1%, respectively, of the average total birds counted per survey (Table 15). At the Ocean Springs site, red-winged blackbirds had the highest relative abundance in the summer, while American coots had the highest in the fall (Table 16). The mean number of red-winged blackbirds per survey stayed about the same from summer to fall, but the mean number of coots almost quadrupled. Although some species were detected on both the summer and fall surveys, the bird communities were quite different for the two survey periods. Only 8 species at the Collins site and 7 species at the Ocean Springs site were detected on both the summer and fall surveys (Tables 15 and 16).

The total number of birds surveyed at Collins and Ocean Springs was 189 and 296, respectively, in the summer and 123 and 674, respectively, in the fall. This resulted in means of 37.8 and 26.4 birds per survey for the summer and fall periods, respectively, at the Collins site (Table 15) and 59.2 and 134.8 birds per survey for the summer and fall periods, respectively, at the Ocean Springs site (Table 16). The high counts at the Ocean Springs site in the fall were due to a large extent to a large number of American coots. The average density at the Collins site was 8.5 birds/ha in the summer and 5.9 birds/ha in the fall. At the Ocean Springs site, bird density was 6.4 per ha in summer and 14.5 per ha in fall.

Several bird species were detected at each site during the field work in mid-summer that were not detected during surveys. The probability of detecting less common birds or migrants was greater during field work because researchers spent four continuous 10-hour days on the site. In addition, more forest birds were detected during the field work, particularly at the Collins site, probably because the distance within which birds were considered was not limited as it was with the more systematic surveys. Additional birds seen or heard while conducting work at the Collins site were: northern cardinal, rufous-sided towhee, white-eyed vireo, yellow warbler, yellow-billed cuckoo, tufted titmouse, and indigo bunting. Additional birds seen or heard while conducting work at the Ocean Springs site were: cliff swallow, great egret, white ibis, turkey

Table 15. Mean number of birds of each species per survey (n=5) and their relative abundances in the summer and fall periods-Collins site. Total of relative abundance is not exactly 100 due to rounding error.

Species	Summer 1991 mean per survey (n=5)	Summer 1991 rel. abund. (%)	Fall 1991 mean per survey (n=5)	Fall 1991 rel. abund (%)
wood duck	10.4	27.5	8.2	31.1
red-winged blackbird	7.8	20.6	2.8	10.6
chimney swift	5.6	14.8	0	0
barn swallow	3.0	7.9	0	0
purple martin	1.6	4.2	0	0
common grackle	1.4	3.7	1.4	5.3
eastern kingbird	1.4	3.7	0	0
rough-winged swallow	1.4	3.7	0	0
eastern bluebird	1.2	3.2	0	0
common yellowthroat	0.8	2.1	0.4	1.5
semipalmated sandpiper	0.4	1.1	0	0
great egret	0.4	1.1	0.6	2.3
great crested flycatcher	0.4	1.1	0	0
green-backed heron	0.4	1.1	0	0
ruby-throated hummingbird	0.4	1.1	0	0
American crow	0.4	1.1	1.8	6.8
sharp-shinned hawk	0.2	0.5	0	0
northern mockingbird	0.2	0.5	0.2	0.8
blue jay	0.2	0.5	0	0
great blue heron	0.2	0.5	0.6	2.3
eastern phoebe	0	0	1.8	6.8
killdeer	0	0	1.4	5.3
ring-necked duck	0	0	1.2	4.5
blue-winged teal	0	0	1.0	3.8
American coot	0	0	0.8	3.0
yellow-rumped warbler	0	0	0.8	3.0
American widgeon	0	0	0.8	3.0
gadwall	0	0	0.6	2.3
marsh wren	0	0	0.4	1.5
belted kingfisher	0	0	0.4	1.5
Carolina wren	0	0	0.4	1.5
yellow-bellied sapsucker	0	0	0.2	0.8
common snipe	0	0	0.2	0.8
hooded merganser	0	0	0.2	0.8
field sparrow	0	0	0.2	0.8
Totals	37.8	100.0	26.4	100.1

Table 16. Mean number of birds of each species per survey and their relative abundances in the summer and fall periods - Ocean Springs site. Total of relative abundance is not exactly 100 due to rounding error.

	Summer 1991		Fall 1991	
Species	mean per survey	rel. abund. (%)	mean per survey	rel. abund (%)
red-winged blackbird	18.6	31.4	15.8	11.7
American coot	17.0	28.7	62.4	46.3
common gallinule	7.8	13.2	1.6	1.2
pied-billed grebe	4.0	6.8	0.8	0.6
mallard	2.4	4.1	0.6	0.4
common grackle	1.8	3.0	0	0
eastern kingbird	1.6	2.7	0	0
barn swallow	1.0	1.7	0	0
green-backed heron	1.0	1.7	0	0
rough-winged swallow	1.0	1.7	0	0
chimney swift	0.4	0.7	0	0
great blue heron	0.4	0.7	0	0
wood duck	0.4	0.7	0	0
killdeer	0.4	0.7	2.0	1.5
mourning dove	0.4	0.7	0	0
bobwhite quail	0.4	0.7	2.6	1.9
brown-headed cowbird	0.2	0.3	0	0
common nighthawk	0.2	0.3	0	0
orchard oriole	0.2	0.3	0	0
common snipe	0	0	12.2	9.1
blue-winged teal	0	0	10.0	7.4
swamp sparrow	0	0	8.6	6.4
sanderling	0	0	5.0	3.7
savannah sparrow	0	0	4.2	3.1
black-necked stilt	0	0	2.0	1.5
marsh wren	0	0	1.6	1.2
shoveler	0	0	1.2	0.9
sora	0	0	0.8	0.6
eastern meadowlark	0	0	0.8	0.6
lesser yellowlegs	0	0	0.8	0.6
snowy egret	0	0	0.4	0.3
northern harrier	0	0	0.4	0.3
kestrel	0	0	0.4	0.3
ring-necked duck	0	0	0.4	0.3
American widgeon	0	0	0.2	0.1
Totals	59.2	100.1	134.8	100.0

vulture, mottled duck, Canada goose, great crested flycatcher, Mississippi kite, purple martin, little blue heron, and Mississippi sandhill crane (flying over the site).

The benefits of both WTS are probably increased because the wetlands occur in a landscape setting where wildlife can make use of a complex of habitats in a larger area. Although the Collins site lies at the edge of a residential area in a small town, it is surrounded by mature forest and is bordered on one side by a wooded stream, which provides habitat for forest passerines and wood ducks, which also use the wetland. The density of wood ducks at the site is also enhanced by nest boxes, which have been placed at several points throughout the wetland and are used heavily during nesting season. The Ocean Springs site is part of the Sandhill Crane National Wildlife Refuge, so ample wildlife habitat of various forms borders that site as well. The two WTS attract and provide resources for birds that are not wetland dependent. For example, the WTS produce insects which can serve as food for songbirds such as swallows, flycatchers, warblers, and nighthawks. Sixty-six percent of bird species at the Collins site and 51% at the Ocean Springs site were not wetland dependent.

At several southeastern palustrine non-WTS comparison wetlands, species richness ranged from 5 to 98 (values in the lower range were reported in studies that surveyed only waterfowl and wading birds) (Tables 17-19). Bird density ranged from 0.04 to 0.35 (Tables 17 and 19). Species richness at the WTS (35 at both sites) was within the range found in other types of wetlands. Densities at the WTS (5.9-14.5 birds per ha.) were much higher than those reported for comparison wetlands. This can probably be attributed in part to the observed high biological productivity in the WTS. Increased organic loading increases production of aquatic invertebrates and thus the abundance and diversity of songbirds (Hanowski and Niemi 1987) and waterfowl (Belanger and Couture 1988, Piest and Sowls 1985). Benefits to waterfowl and other species of wildlife from use of wastewater for habitat enhancement in California marshes were also reported by Cedarquist and Roche (1979) and Cedarquist (1980a, 1980b) for wastewater discharge to natural wetlands and by Demgen (1979) and Demgen and Nute (1979) for artificial wetlands.

Although the benefits are great, concerns exist about wildlife use of WTS. Some mortality of water birds from microbial diseases (Steiniger 1962, Dodge and Low 1972, Clegg and Hunt 1975) and from contaminants (Nero 1964) has been attributed to their use of WTS. Current concern about a nematode parasite (*Eustrongylides ignotus*) associated with eutrophic waters primarily in the southeastern United States, poses one reason for conservative evaluation of WTS as wildlife habitats. The parasite spends parts of its life cycle in aquatic worms (Oligochaeta) and fish and can cause mortality in fish-eating wading birds, particularly nestlings (Spalding, M. personal communication, University of Florida, Gainesville). Presently,

Table 17. Bird species richness and density in Florida non-WTS palustrine wetlands.

Site		Species richness	Year	Density (birds/ha)	Source
West of K-6	(FL)	40	1990	- -	Henigar & Ray 1990
Lake Hancock	(FL)	13	1988	0.26	Edelson and Collopy 1990
Lake Hancock	(FL)	15	1989	0.28	Edelson and Collopy 1990

- - data not available

Table 18. Waterfowl and wading bird species richness at non-WTS palustrine wetlands in Guntersville Reservoir, Alabama, 1988 (James et al., 1989).

Site	Species richness	Site	Species richness
Town Creek Embayment		Mud Creek Embayment	
Compartment 1	5	Compartment 1	11
Compartment 2	5	Comparment 2	8
Compartment 3	6	Comparment 3	8

Table 19. Bird species richness and density in Lower Mississippi River borrow pits (non-WTS palustrine wetlands), Mississippi, 1983 (US Army Engineer Mississippi River Commission, 1986). BP=borrow pit; pits are named by numbers.

Site	Species richness	Density (birds/ha)
BP1	98	0.06
BP14	98	0.20
BP15	61	0.35
BP19	79	0.04
BP22	65	0.04
BP25	68	0.22

the overall impact of these parasites on wading bird populations is uncertain.

Habitat requirements, life histories, and species assemblages of wetland birds are relatively well-known, although information on community-level response to particular stressors has been difficult to collect (Adamus and Brandt 1990). 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.

Birds, however, are very mobile, and their use of a wetland may be erratic, necessitating multiple surveys over a period of time. Because of their mobility, effects of contamination in birds, if detected, usually cannot be linked with certainty to the wetland in question. In addition, one cannot assume that the presence of birds means good habitat quality, particularly in regions where suitable habitats are diminishing and birds are forced to use available habitat regardless of its condition. Most bird species might be better as indicators of overall landscape conditions than of single wetland conditions (Adamus and Brandt 1990). The food resource (e.g., invertebrates, zooplankton, or fish) might be a more reliable indicator for assessing the faunal component of individual wetlands and is not as mobile. However, laboratory time and expense are required for identification of invertebrates or zooplankton.

Migratory seasons are the best time to assess optimal foraging and resting use by birds but not necessarily the best time to sample other indicators at the wetland, such as vegetation. If the goal of future monitoring is to assess a wetland in a short time period (1 day or less), estimation of bird use may be grossly biased. Depending upon the information desired, consideration might be given to conducting surveys in only one season instead of two to save expense and field effort. The results of this study showed that species richness changed very little between the summer and fall survey periods, but the species compositions changed considerably. Thus, species richness for only one season of surveys would be substantially lower than for two seasons of effort and would not represent bird use as accurately. In addition, the average number of birds per survey varied between the summer and fall periods and the direction of the change was different at the two WTS. If bird use is an indicator in future studies, research planners should consider:

- The amount of sampling effort that can be devoted versus that required to obtain an accurate representation of bird use, density, and diversity

- Establishment of a yearly sampling schedule that minimizes survey effort (i.e. repeat visits) while assuring that bird use is accurately characterized

- Data integration and reduction if multiple surveys are conducted

- Logistics and quality assurance 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

- Quantification of the relationship between bird abundance/density and other wildlife habitat indicator values

- Comparison of bird density and richness at a WTS with that found at surrounding reference wetlands

- Quantification of bird activity (breeding, feeding, resting) and the relative abundance of threatened, endangered, or keystone species.

Evaluation of Ancillary Values

Probability ratings of high, medium, and low assigned by the WET analysis to the various wetland functions are given in Table 20. The WET analysis provides ratings for functions other than wildlife habitat, which are also included in the table. Of greatest concern with regard to wildlife habitat are the ratings for effectiveness and opportunity that characterize the wetland and surrounding area in terms of physical, chemical and biological attributes. The four ratings for wildlife and aquatic diversity/abundance under effectiveness were high for the Collins site. At the Ocean Springs site, ratings were high for wildlife migration and wintering and low for wildlife breeding and aquatic diversity/abundance. The Collins site was also rated high under effectiveness for sediment/toxicant retention and nutrient removal/transformation; The Ocean Springs site was high for groundwater discharge, floodflow alteration, sediment stabilization, sediment/toxicant retention, and nutrient removal/transformation. Both sites were rated low for groundwater recharge. The majority of values under social significance were rated low at the Collins site and moderate at the Ocean Springs site, with the exception of wildlife diversity/abundance, which was moderate at the Collins site and high at the Ocean Springs site.

The use of WET in WTS presented some interpretation problems, which arose because questions did not pertain to the unique circumstances present in WTS. For instance, WET was designed primarily for wetlands, either natural or artificial, that function within a watershed. WTS, however, do not receive water from a typical watershed. Their artificial nature and the service they provide are unique circumstances that are not accommodated by some of the WET questions. Answering these questions requires assumptions and/or guesses that might affect the final outcome in unknown ways. The technique also requires a large amount of information on surrounding wetlands and a knowledge of the locality (e.g., geography, geology, land use, watershed characteristics). Information about the wetland in question in other times of the year is also necessary. Local people familiar with the area must be depended upon to answer questions regarding other wetlands and seasons.

The answers to many of the WET questions were uncertain or speculative, so the results are questionable. Unless field personnel are experienced with using the technique, it can be cumbersome and confusing. There is also the possibility that the answers given by the WET analysis may be taken at face value without consideration of other data collected or use of professional judgement. Continued use of WET for assessing WTS is therefore not recommended.

Other comprehensive evaluation methods exist and could be considered for testing in future research if a rapid assessment method is deemed necessary to complement indicator data. Some were designed for national use while others were designed for regional use. All methods have limitations and are based on assumptions and none have been validated extensively. An overview of the most commonly used methods is given by Adamus (1992).

Site Morphology

Physical features of artificial ponds, such as surface area and shoreline irregularity (or edge), influence waterfowl brood use (Belanger and Couture 1988, Lokemoen 1973, Mack and Flake 1980, and Hudson 1983). Belanger and Couture (1988) recommend that for good waterfowl habitat, artificial ponds should have $\geq$30% cover of emergent vegetation. Both of the Mississippi sites have more than 30% emergent vegetation cover. Emergent vegetation covered 32% (not including floating *Alternanthera philoxeroides*) of the wetland area at the Collins site and 71% at the Ocean Springs site.

Diversity, abundance, and density of wetland-dependent animals is usually higher when vegetation and water are well-interspersed. Weller and Frederickson (1973) noted a possible correlation between marsh-restricted bird species and percent open water or the number of open pools in emergent cover. Steel et al. (1956) reported larger duck nesting populations in broken than in solid emergent vegetation. 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) produce the greatest bird diversities and numbers (Weller and Frederickson 1973). 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. The Collins site ratio of water to

Table 20. WET ratings for the Collins and Ocean Springs sites. H=high; M=moderate; L=low; *=not evaluated by WET; Effect.=Effectiveness; Opport.=Opportunity; d/a=diversity/abundance.

Wetland Function	Social Significance	Effect.	Opport.
COLLINS			
Groundwater recharge	L	L	*
Groundwater discharge	L	M	*
Floodflow alteration	L	M	H
Sediment stabilization	M	M	*
Sediment/toxicant retention	L	H	H
Nutrient removal/transformation	L	H	H
Production export	*	M	*
Wildlife diversity/abundance	M	*	*
Wildlife d/a - breeding	*	H	*
Wildlife d/a - migration	*	H	*
Wildlife d/a - wintering	*	H	*
Aquatic diversity/abundance	M	H	*
Uniqueness/heritage	H	*	*
Recreation	L	*	*
OCEAN SPRINGS			
Groundwater recharge	M	L	*
Groundwater discharge	M	H	*
Floodflow alteration	M	H	M
Sediment stabilization	M	H	*
Sediment/toxicant retention	M	H	H
Nutrient removal/transformation	M	H	H
Production export	*	M	*
Wildlife diversity/abundance	H	*	*
Wildlife d/a - breeding	*	L	*
Wildlife d/a - migration	*	H	*
Wildlife d/a - wintering	*	H	*
Aquatic diversity/abundance	M	L	*
Uniqueness/heritage	H	*	*
Recreation	L	*	*

cover was slightly low (0.77) (Table 21), and open water pools were not interspersed with cover. Cell 1 contained more open water, making the ratio of water to cover more optimal, while cell 2 was densely vegetated with *Scirpus* spp. and *Lemna* spp., producing a ratio of zero (Table 21). The ratio of water to cover at the Ocean Springs site was only 0.09. The deepwater areas covered by duckweed at that site, however, might be better included as water, which would make the ratio slightly higher (0.10-0.15). The Ocean Springs site was planted as solid dense vegetation with deepwater areas between strips of *Typha* spp. and *Scirpus* spp. After a year of operation, however, some areas in the vegetated sections (particularly the *Scirpus* spp. are opening up, creating larger areas of shallow water for waterbird feeding and a better interspersion of water and cover.

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 large emergent plants such as *Typha* and *Scirpus*. At both of the WTS, surface water area covered by duckweed underneath emergent plants was sufficient to allow use by waterbirds for protection and feeding.

Land/water interface per hectare is a measure of edge. It is also another measure of the degree of interspersion of water and cover. Mack and Flake (1980) found that edge length was positively correlated with dabbling duck production in the prairie pothole region. Harris and others (1983) concluded that edge habitat is important to bird species diversity. Neither wetland has sinuous shoreline or islands. Land/water interface/ha was almost twice as great at the Collins site as at the Ocean Springs site (Table 21), probably due to the small dikes that extend into the Collins WTS (see Appendix A). The dikes at this site, however, cannot be used for nesting or protection most of the time because they are mowed regularly for maintenance purposes and are sometimes flooded in wetter years. At the Ocean Springs site, the only land/water interface is the rectangular border of the two cells.

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. Based on field observations, plant species were not well-interspersed overall at either of the Mississippi sites, partly because the number of dominant species (those that were discernable on aerial photos - Table 21) was low and partly because these sites, like many constructed wetland sites, were artificially planted. Planting is often done by placing different species next to each other in rectangular sections or long strips with straight sides, thus minimizing distance of edge

Table 21. 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 cell. Percent coverage of plants listed under the vegetated category can sum to more than the total vegetated percentage due to overlap of species.

COLLINS SITE

	Whole site	Cell 1	Cell 2
1. Wetland area (ha)*	4.473	2.849	1.624
2. Cover areas (ha) (percent of wetland area)*			
a. Vegetated	2.530 (57)	0.906 (32)	1.624 (100)
Scirpus californicus	0.973 (22)	0 (0)	0.973 (60)
Typha latifolia	0.071 (1)	0.071 (2)	0 (0)
Polygonum punctatum	0.043 (1)	0.043 (2)	0 (0)
Alternanthera philoxeroides	0.453 (10)	0.444 (16)	0.009 (1)
Small floating-leaved	2.036 (46)	0.404 (14)	1.624 (100)
Mixed emergents	0.375 (8)	0.327 (11)	0.048 (3)
b. Open Water	1.943 (43)	1.943 (68)	0 (0)
3. Land/water interface (m)	1835	1195	640
4. cover/cover interface (m)	1680	190	1490
5. Open water area:vegetated area*	0.77	2.14	0
6. Land/water interface: Wetland area (m/ha)*	410	419	394
7. Cover/cover interface: Wetland area (m/ha)*	376	67	917

(Table 21, continued)

OCEAN SPRINGS SITE	Whole site	Cell 1	Cell 2
1. Wetland area (ha)*	9.281	5.088	4.193
2. Cover areas (ha) (percent of wetland area)*			
a. Vegetated	8.530 (92)	4.677 (92)	3.853 (92)
Typha latifolia	5.177 (56)	2.650 (52)	2.527 (60)
Scirpus validus	1.217 (13)	0.897 (18)	0.320 (7)
Small floating-leaved	1.994 (21)	1.014 (20)	0.980 (23)
Sagittaria lancifolia	0.091 (1)	0.070 (1)	0.021 (1)
Pontederia cordata	0.051 (1)	0.046 (1)	0.005 (1)
b. Open Water	0.751 (8)	0.411 (8)	0.340 (8)
3. Land/water interface (m)	2134	1120	1013
4. Cover/cover interface (m)	2681	1392	1289
5. Open water area:vegetated area*	.09	.09	.09
6. Land/water interface: Wetland area (m/ha)*	230	220	242
7. Cover/cover interface: Wetland area (m/ha)*	289	274	307

(and thus interspersion) between species. Although this is often a practical design for treating wastewater, it may not be best for wildlife habitat. The whole-site ratio of cover/cover interface per ha was higher at the Collins site than at the Ocean Springs site, but the ratios of the two cells were more similar at the Ocean Springs site (Table 21). The low ratio at the Collins site cell 1 (67) was due to the sparse vegetation in that cell (only 32%) and the presence of only one major cover type. The distinction between *Alternanthera philoxeroides* and *Lemna* spp. was not made when measuring interface because both were floating-leaved species and not well-separable as two different cover types. The high ratio in cell 2 was due to the interspersion of dense *Scirpus* spp. and *Lemna* spp., which were two distinctly different cover types.

The size of a wetland is vital to maintaining a marsh fauna. To produce good waterfowl habitat, Belanger and Couture (1988) recommend that artificial ponds be $\geq$ 0.5 hectare. Both WTS studied meet this criterion. 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 some for nesting, others for feeding, and others for roosting and cover. The habitat value of the Collins wetland is very likely increased because of its setting adjacent to a forested stream, which provides necessary resources, including habitat, for songbirds and wood ducks that might not be present in the wetland alone. For assessing the value of a single wetland, the wetland area indicator, therefore, might be better expressed as the area 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 presence of other habitat types might also be important for evaluating wetland value in a landscape context, particularly for wildlife which is not wetland-dependent but which periodically makes use of wetlands. For instance, the Ocean Springs site lies within the Mississippi Sandhill Crane National Wildlife Refuge, and its value is enhanced because birds attracted to the refuge can use it.

Physical habitat features such as shoreline length, amount of edge, ratio of open water to vegetated area, and vegetation interspersion are good indicators of habitat quality because their relationships to wildlife production and/or use have been shown repeatedly. They can be obtained from maps or aerial photographs in a relatively short period of time and with less effort than field work. Some field ground truthing of vegetation types, however, is necessary for air photo interpretation. The indicators can be collected in every wetland of interest, and replicate samples and assessment of variability are not necessary. Comparisons with reference wetlands can be done to assess potential differences between natural and WTS. Aerial photos and maps can also be used to evaluate the larger landscape setting, which is of great importance in evaluating wildlife habitat, particularly when the wetland in question is small. There is some overlap between the kinds of information on cover types that can be obtained from photos and from vegetation sampling in the field. Cover estimation of the dominant cover types can be obtained from photos while field work might focus on determination of species richness. A wide variety of information can be obtained from photos and maps, and their use in the future is highly recommended. Cost of aerial photography, however, may be a limitation.

Water Quality

Water quality summary data are presented for the influent and effluent of each WTS in Table 22. Comparison data for non-WTS in the southeastern region are presented in Tables 23-27. Means for water quality indicators from the influent and effluent of the WTS were within the range of non-WTS for pH, TSS, DO, TKN, and TP.

Wetlands that receive water with low levels of TSS (less than 80 and never exceeding 200 mg/L) are more likely to support a greater diversity and/or abundance of fish and invertebrates (P. Adamus, personal communication, Mantech Environmental Technology, Inc., Corvallis, OR). Both WTS met these criteria. The TSS concentrations at both sites are periodically above 80 mg/L, but were never as high as 200 mg/L. In addition, both wetlands are very efficient at reducing TSS. Average effluent concentrations (12.8 mg/L at the Collins site and 10.9 mg/L at the Ocean Springs site) are much lower than influent concentrations.

Turbidity can affect fish and invertebrate populations indirectly by raising water temperature, leading to lower DO concentrations (Reed et al. 1983). Dissolved oxygen concentrations greater than 4 mg/L and 60% saturation are more likely to support a greater diversity and/or abundance of fish and invertebrates than wetlands with lower concentrations (Adamus, personal communication, Mantech Environmental Technology, Inc. Corvallis, OR). Concentrations of 2 and 4 mg/L, however, are not uncommon 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. Average dissolved oxygen at both WTS sites studied is 4.0 mg/L or above. Average effluent concentrations were lower than influent concentrations, with minimum concentrations of 3.6 mg/L at the Collins site and 1.2 mg/L at the Ocean Springs site (Table 22).

Total mean phosphorus values for the WTS (3.8-5.0 mg/L) (Table 22) were high compared to most non-WTS (0.02-2.10 mg/L)

(Tables 23, 24, 26, and 27), but were still lower than two of the natural wetlands studied by Brown (1991) in Florida, which had TP concentrations of 6.1 and 8.7 mg/L (Table 27).

Fecal coliform bacteria at the Collins site (58.8 influent; 56.4 effluent) (Table 22) was within the range of values reported for non-WTS. Fecal coliforms at the Ocean Springs site (249.3 influent; 112.7 effluent) were higher than those reported for non-WTS (<10-100) (Table 24), although few data were found for comparison. The effluent count at the Ocean Springs site is very close to the comparison range and is more than a 50% reduction of the influent count.

BOD and Ammonia-N concentrations in influent and effluent of the two WTS (1.1-4.7 mg/L for ammonia-N and 9.1-68.2 mg/L for BOD) (Table 22) were high in comparison with non-WTS (0.01-0.26 mg/L for ammonia-N and 2.3-7.4 mg/L for BOD) (Tables 23 and 24), although few data were found for comparison. Influent means at the WTS sites were about an order of magnitude greater than the highest comparison values for both parameters. It is likely, although not confirmed, that the results reported for ammonia-nitrogen at the WTS were actually for ammonium. Both WTS, however, were very efficient in removing BOD. Effluent BOD concentrations were 9.3 mg/L and 9.1 mg/L at the Collins and Ocean Springs sites, respectively, compared to 68.2 and 21.2 mg/L in the influent (Table 22). Ammonia-nitrogen was also removed within the WTS (from 4.1 to 1.1 mg/L at the Collins site and from 4.7 to 1.3 at the Ocean Springs site) (Table 22).

Variability is high for some water quality indicators (e.g., fecal coliform counts) and low for others (e.g. Ph, TP) as shown by the standard deviation (Table 22). For some indicators the magnitude of variability is not consistent between sites. Variability of TSS and NH_3-N is lower at the Collins than at the Ocean Springs site (Table 22).

Interpreting precisely what some water quality indicators mean for assessing wildlife habitat quality is difficult because the relationship to most important habitat components is indirect. The effects of water nutrient concentrations are reflected by community composition of plants, invertebrates, fish, etc., which are more directly related to wildlife habitat and are time-integrated measures. For instance, poor water quality (e.g., low water clarity, low oxygen concentrations) typically causes growth of competitive plant species over time, which often crowd out species most valuable to wildlife and produce little or no food (Atlantic Flyway Council 1972).

Another problem with using water quality parameters as indicators is that, because some of them are quite variable and the information obtained is not time-integrated, measurements need to be taken over time to have significant meaning. The measurements are usually available from site operators because discharge permits require monitoring of certain constituents in wastewater. However, evaluating the quality of these data

Table 22. Summaries of water quality data at the Collins and Ocean Springs 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.

COLLINS

Variable	I/E	N	Range	Mean	Std Dev
pH (SU)	I	14	6.6-8.4	7.4	0.6
Ph	E	16	6.2-7.5	7.0	0.4
TSS (mg/L)	I	16	16.4-123.0	75.1	31.2
TSS	E	15	8.4-16.4	12.8	2.9
DO (mg/L)	I	7	7.7-10.4	9.3	1.0
DO	E	7	3.6-6.9	4.9	1.1
BOD (mg/L)	I	16	27.5-121.3	68.2	37.1
BOD	E	15	7.3-11.7	9.3	1.4
NH3-N (mg/L)	I	7	3.2-4.8	4.0	0.5
NH3-N	E	7	0.8-1.3	1.1	0.2
TKN (mg/L)	I	Not Measured			
TKN	E	Not Measured			
TP (mg/L)	I	7	3.3-5.3	4.7	0.7
TP	E	7	2.6-4.9	3.8	0.8
Fec.Col. (no./100 Ml)	I	8	35-140	58.8	34.1
Fec.Col.	E	14	10-170	56.4	51.2

(Table 22, continued)

OCEAN SPRINGS

Variable	I/E	N	Range	Mean	Std Dev
Ph (S.U.)	I	44	7.3-10.8	8.7	0.9
Ph	E	67	6.3-9.9	7.3	0.6
TSS (mg/L)	I	48	2.0-128.0	37.8	28.5
TSS	E	67	1.0-90.0	10.9	13.9
DO (mg/L)	I	24	2.8-12.6	6.8	2.4
DO	E	68	1.2-9.3	4.0	2.6
BOD (mg/L)	I	47	4.8-61.8	21.2	12.1
BOD	E	67	3.0-51.0	9.1	6.2
NH3-N (mg/L)	I	11	0.1-13.2	4.7	4.8
NH3-N	E	68	0.1-4.4	1.3	1.1
TKN (mg/L)	I	11	4.4-20.2	11.4	5.4
TKN	E	24	1.3-8.3	3.8	1.5
TP (mg/L)	I	11	3.8-6.3	5.0	1.0
TP	E	Not Measured			
Fec.Col. (no./100 Ml)	I	47	13.0-1200.0	249.3	253.5
Fec.Col.	E	14	18.0-600.0	112.7	149.3

Table 23. 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; (from Erwin and Bartleson, 1985). pH in standard units, all other parameters in mg/L. DO=dissolved oxygen, BOD=biochemical oxygen demand, TP=total phosporus.

	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 24. Surface water quality mean values (depth=0.15-0.20m) from Nags Head non-WTS marsh ponds, NC, May 1987; (from Mac Pherson 1988). pH is in standard units; DO, NH_3-N, TKN, TP, and 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 25. Surface (0.3m depth) water quality means and ranges 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 size given in parentheses.

	pH	DO	TSS
Canadian Reach	6.9(3) 6.8-7.1	6.3(3) 4.8-7.4	10.7(3) 9.0-13.0
Crutcher Lake	7.9(3) 7.5-8.2	7.6(3) 5.7-8.9	13.7(3) 6.0-29.0
Catfish Chute	7.7(3) 7.6-7.8	4.5(3) 3.0-6.7	8.0(3) 6.0-12.0
Driver Bar	7.6(3) 7.4-7.9	6.3(3) 4.2-8.4	5.3(3) 4.0-6.0
Lake Whittington	7.5(3) 7.4-7.5	4.9(3) 4.0-5.6	25.7(3) 17.0-42.0
Yucatan Lake	7.4(3) 7.3-7.6	7.0(3) 5.9-7.6	8.0(3) 7.0-9.0
Raccourci Lake	7.7(3) 7.5-7.8	6.2(3) 5.3-6.8	5.0(3) 4.0-7.0
Deer Park Lake	7.2(6) 7.0-7.4	4.6(6) 2.3-6.0	7.2(6) 4.0-11.0

Table 26. Surface water quality mean values for non-WTS marsh sites in the Okefenokee Swamp; (reported by 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 ± .006
Mizell Prairie	3.87 3.72-4.15	4.88 2.12-6.60	0.031 ± .013
Mack's Island Rookery	4.17 3.93-4.77	2.50 0.60-7.10	0.052 ± .031
Reference	4.15 3.88-4.81	2.20 0.70-4.00	0.020 ± .002

Table 27. Water quality in created and natural herbaceous non-WTS marshes near Tampa, Florida, 1988; (from Brown, 1991). The letter "a" after the wetland number denotes a duplicate sample. TSS (total suspended solids), TP (total phosphorus), and TKN (total Kjeldahl nitrogen) in mg/L. Values are based on one sample.

	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
103a	13.0	0.05	0.65
103	10.0	0.06	1.30
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

requires extensive and time-consuming review and evaluation of standard operating and quality assurance procedures used by laboratories that conduct analyses on each wetland's water samples. If comparisons among wetlands are to be done in future studies, laboratory procedures and quality control measures should be the same for all laboratories. Also, the particular water quality parameters measured differ from one site to another and are not necessarily collected at the same frequency. Data management and record-keeping by site operators varies, and it is sometimes difficult to acquire specific data. Also, there is some discrepancy among laboratories about exactly which metric is measured and what it is called (e.g., ammonia vs. ammonium, total phosphorus vs. total phosphorous as phosphate). The continued use of existing data sets for acquiring indicator information is therefore not recommended. Sampling of some water quality indicators, such as dissolved oxygen, ammonia, or suspended solids, during field sampling might be useful for interpreting other field data collected at the same time (e.g., abundance and diversity of invertebrate and plant species).

CONCLUSIONS AND RECOMMENDATIONS

The use of partially treated wastewater for creation of wetlands has great potential. It is an efficient reuse of water, eliminates chemical treatment, can be very cost-effective, and can be beneficial to wildlife. For these reasons, assessment of the wildlife habitat quality and sustainability of these systems and development of methods for assessing and monitoring them is of interest to the EPA.

Table 28 contains a summary of the comparisons made between the two WTS studied and non-WTS in the Southeast. Overall, most of the indicator values from the two Mississippi WTS (for which comparison values were available) were within the range of values for non-WTS. Two of the water quality indicators - BOD and ammonia-N concentrations - had higher values in the two WTS than in the non-WTS used for comparison. The majority of water quality indicators, however, were within the range. Bird densities were generally higher in WTS than in non-WTS. The data suggest that the habitat condition of the two WTS studied is not grossly different than that of the general population of wetlands in the same region. The preliminary results, however, do not indicate actual habitat value because little is known about the habitat quality of comparison wetlands or the WTS as measured by the indicators used. Habitat quality was assessed only relevant to comparison wetlands. Guidelines for selecting comparison (i.e., reference) wetlands with good wildlife habitat are needed.

Table 28. General relationship of data from the WTS studied to the range of values reported for non-WTS in the southeast United States.

	Below Range	Within Range			Above Range
		Low	Middle	High	
Plant Species Richness		X			
Invertebrate Genera Richness				X	
Water Nutrient Concentrations				X	
Bird Species Richness			X		
Bird Density					X

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 29. Vegetation, invertebrate, and site morphology indicators are recommended for development for evaluating wildlife habitat quality in WTS. The use of birds as indicators is questionable, primarily because of their mobility.

Use of existing water nutrient data, whole-effluent tests, and the WET analysis are not recommended. Nutrient data can be variable, and problems exist with consistency of laboratory techniques, and quality assurance and control procedures. Acquisition and evaluation of QA/QC information is difficult and time consuming. Other indicators exist which are reliable and more directly related to wildlife habitat quality. The cost of toxicity testing is a limiting factor. Whole-effluent tests on water do not provide enough information about contamination because they do not provide time-integrated information. The discharge of harmful substances to WTS is likely a short-term or intermittent event, and toxicity in water could be missed by taking only one sample. Potential effects are better detected by testing for contamination in sediments or plant and animal tissues. Making the connection between levels found and actual effects on wildlife would then be necessary. Due to the length and expense of a rigorous testing program, toxicity testing should be done on selected wetlands suspected to be at risk from contamination or toxic inputs (e.g., 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). The WET analysis proved difficult to use in constructed wetlands because of their artificial nature and designated purpose. Many of the questions were not designed with these systems in mind, and thus were ambiguous and difficult to answer with certainty.

This study provided evidence that WTS provide wildlife habitat and that the two sites are used by a variety of bird species. 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:

o For making comparisons of WTS to non-WTS, it would be extremely beneficial if future studies 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, type of surrounding land use, and degree of human disturbance. Comparisons should be quantitative.

- In some landscapes, potential reference sites might in reality all be in marginal or poor condition. For assessing actual habitat quality, an alternative to reference site comparisons would be to develop guidelines for rating habitat quality. 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 each ecological component (e.g., vegetation, invertebrates, landscape). 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, are valuable for monitoring long term changes at a wetland and should not be abandoned 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, particularly new metrics for evaluating habitat in terms of vegetation, invertebrates, and site morphology. For example, invertebrate sampling should include specific techniques for collecting benthic invertebrates. It is recommended that new indicators be directly related to wildlife habitat rather than those that might only infer wildlife use through an indirect relation (e.g., nutrients, sediment type, hydrology). Indirectly related indicators might, however, be useful for supporting other data (e.g., hydrologic regime and sediment types can influence the species composition of plants).

- 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). Air photo analysis might be more accurate, particularly for large wetlands where time restricts thorough ground sampling of the whole wetland. Thus, more effort could be spent in the field sampling indicators that cannot be obtained from photos such as species composition, abundance, and richness.

Table 29. Summary of indicator suitability. Low, moderate, and high are relative ratings for the suite of indicators.

	Vegetation	Invertebrates	Landscape	Toxicity	Water Quality (nutrients)	Birds	WET
Sampling effort (over season)	small	small	small (photography)	small	large	large	small
Time required (sampling/analysis)	moderate	small (field); large (lab)	moderate-large (interpretation)	moderate	small-moderate	small-moderate	small-moderate
Expense (Sample collection & analysis)	low	moderate	high (for new photos)	moderate-high	low-high	moderate	low
Reliability of information for assessing habitat	high	moderate-high	high	high	low-moderate	moderate	uncertain
Recommend development for future studies	yes	yes	yes	no	no	possibly	no
Problems	none	inconsistent sampling methods	none	none	data quality difficult to evaluate; variability	bird mobility; logistics; contracts;	ambiguity; not designed for WTS;

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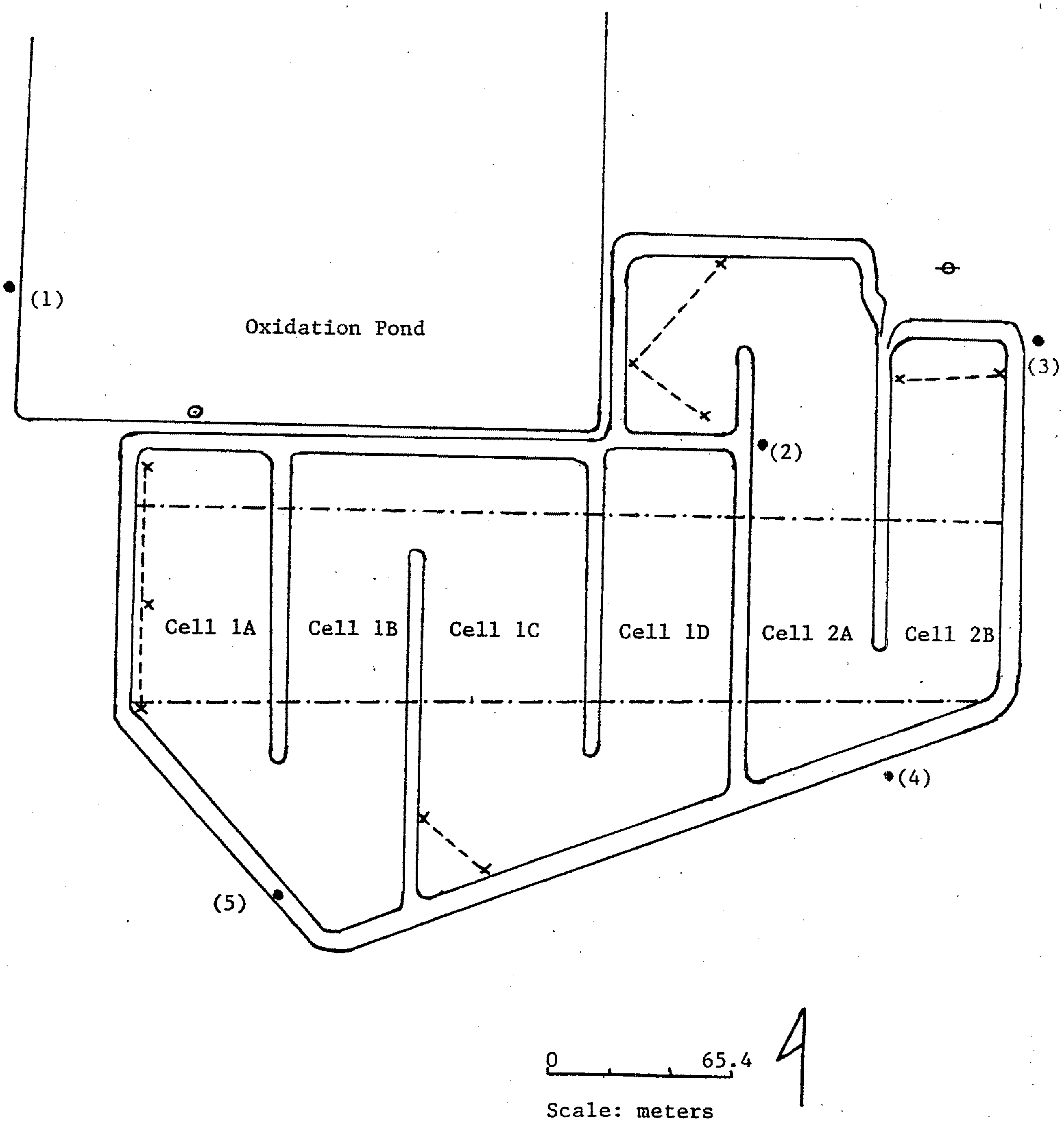
APPENDIX A. Site maps and sampling points

Maps provided by site operators of the Collins and Ocean Springs (Phase I) sites are included in this appendix. The following features are designated on each map: vegetation transect locations, invertebrate sample points, and bird survey points. Water sampling points (for whole effluent toxicity tests) are shown on the Collins map, but were off the boundaries of the map of Ocean Springs. The inflow sample at Ocean Springs was collected in a small maintenance building to the west of the wetland complex near the pre-treatment lagoon. The effluent sample was collected on the north end of the Phase II wetland to the northwest where water is routed after leaving Phase I. 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.

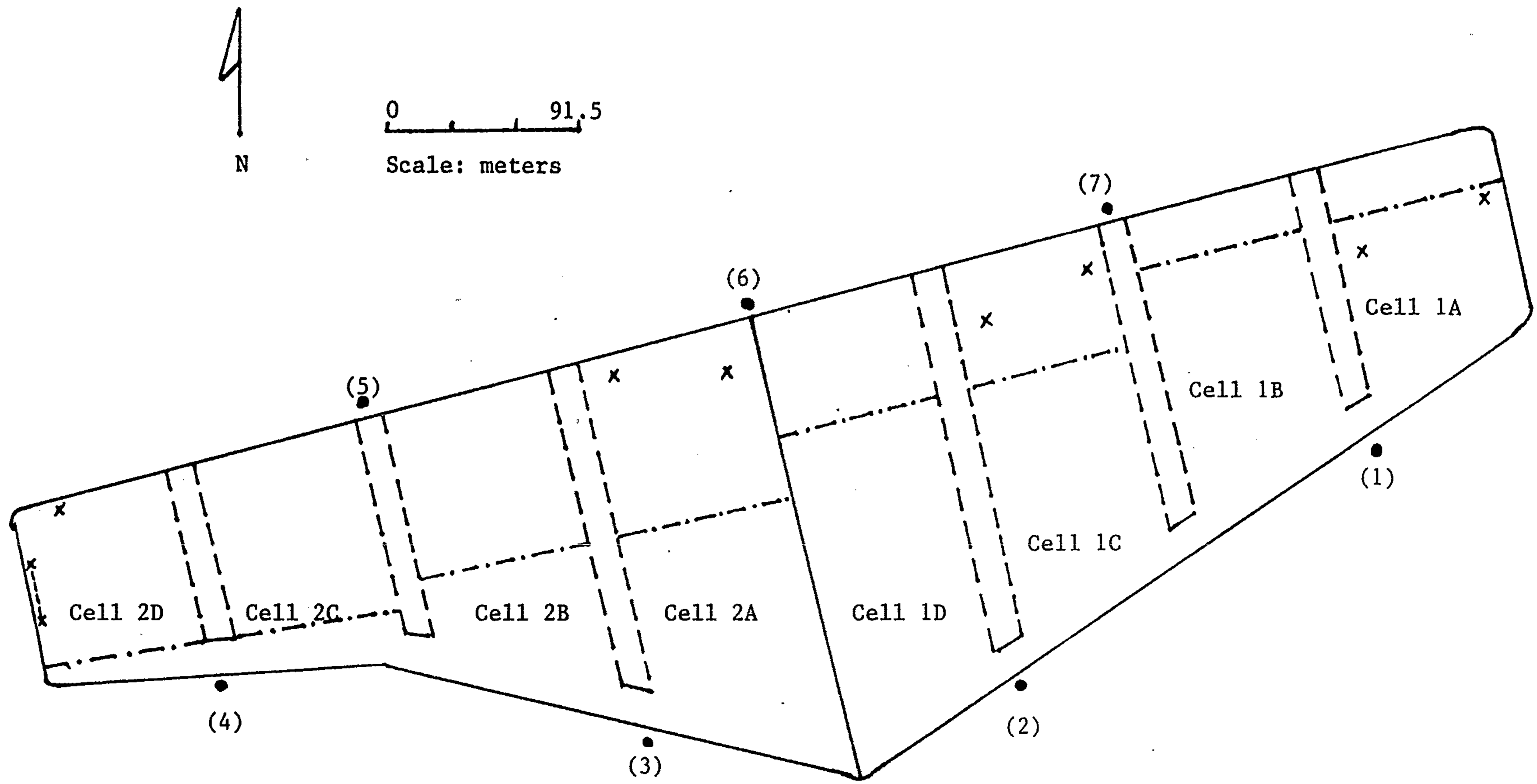
The key below describes the symbols and features found on maps in this appendix:

________	Dikes
---------¦ ---------¦	Deepwater areas
O	Influent sample collection point
o	Effluent sample collection point
.____.____.____	Vegetation transects
X or X----X	Invertebrate sample points
o (1) - (7)	Bird survey points

COLLINS WASTE TREATMENT WETLAND



OCEAN SPRINGS WASTE TREATMENT WETLAND



APPENDIX B. Site contacts and local experts consulted

COLLINS

Contacts:

Bob Hamill
Soil Conservation Service
601 7th Street
P.O. Box 487
Collins, MS 39428

V.O. Smith, Mayor
Collins, MS 39428

Botanists consulted:

Dr. Jean Wooten
University of Southern MS
Walker Science Bldg. Rm. 114
Hattiesburg, MS

Aerial Photography Company:

Harris Aerial Surveys
Lynn Harris
P.O. Box 246
Midway, AR 72651

Bird Surveyors:

Frank Moore; Jeffrey Clark
Department of Biological Sciences
University of Southern Mississippi
Southern Station, Box 5018
Hattiesburg, MS 39406-5018

Water Analysis Laboratories:

Culpepper Testing Laboratories, Jackson, MS
Contact is V.O. Smith, Mayor of Collins

OCEAN SPRINGS

Contact:

MS Gulf Coast Regional
Wastewater Authority
3103 Frederic Street
Pascagoula, MS 39567

Botanists consulted:

Dr. Bill Dunn
CH2M Hill
7201 NW 11th Place
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Gainesville, FL 32602

Aerial Photography Company:

Harris Aerial Surveys
Lynn Harris
P.O. Box 246
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Bird Surveyors:

Frank Moore; Wan Yong (Ocean Springs)
Department of Biological Sciences
University of Southern Mississippi
Southern Station, Box 5018
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Water Analysis Laboratories:

MS Gulf Coast Regional Wastewater Authority Laboratory,
Pascagoula, MS; Contact is Donald Scharr

APPENDIX C. Invertebrate Biologists and Identification Keys Used

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Invertebrate taxonomic keys used:

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APPENDIX D. **Water chemistry of replicates used for whole effluent toxicity tests.**

Ceriodaphnia dubia chronic test

Sample	Mean pH	pH Range	Mean Temp (°C)	Mean DO (mg/L)	Mean Conductiv. (umhos/cm)
Collins site					
		Initial Chemistries			
Influent	7.58	7.43-7.73	25.3	6.4	433
Effluent	7.18	7.13-7.23	25.2	8.2	410
Control	8.23	8.20-8.25	26.0	8.5	123
		Final Chemistries			
Influent	8.40	8.40	25.1	8.1	--
Effluent	8.37	8.35-8.40	24.7	8.3	--
Control	8.14	8.11-8.17	24.8	8.4	--
Ocean Springs site					
		Initial Chemistries			
Influent	9.06	8.99-9.13	25.2	8.6	449
Effluent	7.54	7.37-7.65	25.1	8.2	480
Control	8.09	7.96-8.16	25.7	8.6	120
		Final Chemistries			
Influent	8.46	8.43-8.48	25.2	8.3	--
Effluent	8.51	8.48-8.53	25.1	8.1	--
Control	8.14	7.97-8.25	25.4	8.0	--

(Appendix D, continued)

Fathead minnow acute tests

Sample	Mean pH	pH Range	Mean Temp (°C)	Mean DO (mg/L)	Mean Conductiv. (umhos/cm)
Collins site					
		Initial Chemistries			
Influent	7.58	7.43-7.73	25.3	6.4	433
Effluent	7.18	7.13-7.23	25.0	7.9	417
Control	8.22	8.20-8.25	26.1	8.6	118
		Final Chemistries			
Influent	8.44	--	25.0	8.6	--
Effluent	8.29	--	25.0	7.8	--
Control	8.04	--	25.0	7.9	--
Ocean Springs site					
		Initial Chemistries			
Influent	9.09	9.05-9.13	25.3	8.5	451
Effluent	7.49	7.37-7.61	25.2	7.9	479
Control	8.05	7.98-8.13	26.0	8.7	117
		Final Chemistries			
Influent	8.30	--	25.0	7.9	--
Effluent	8.33	--	25.1	8.2	--
Control	8.10	--	25.3	7.4	--