AN EVALUATION OF PLANT COMMUNITY STRUCTURE, FISH AND BENTHIC MEIO- AND MACROFAUNA AS SUCCESS CRITERIA FOR RECLAIMED WETLANDS

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AN EVALUATION OF PLANT COMMUNITY
STRUCTURE, FISH AND BENTHIC MEIO- AND MACROFAUNA
AS SUCCESS CRITERIA FOR RECLAIMED WETLANDS

FINAL REPORT

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PERSPECTIVE

An important component of FIPR's research program addresses two important questions related to restoring wetlands on surface-mined phosphate lands. Those questions are: "How do you reconstruct fully functional wetlands on reclaimed mined lands?" and "How can you tell when you have achieved that goal?" This project examined several parameters in natural marshes and reconstructed marshes of various ages with the aim of identifying meaningful and measurable indicators of wetland development that might be useful as success criteria. The project looked particularly at benthic invertebrates, fish, vegetation, organic sediment accumulation, and water chemistry.

For a success criterion to be useful it must be a good indicator of wetland development, and it should be easily and reliably measured. Nitrogen and phosphorus concentrations in the water column seemed to meet these requirements. N and P concentrations in the water were initially higher in newly constructed wetlands than in natural wetlands but declined in about five years to levels similar to natural wetlands. The changes in N and P appeared to be related to the establishment of emergent vegetation and the resulting accumulation of organic sediment which then led to an equilibrium in nutrient cycling within the system. The project found complications or problems with indices based on benthic macroinvertebrates, benthic meiofauna, or fish.

The reader is referred to the following related projects and reports:

Brown, M.T. and R.E. Tighe (Eds.). 1991. Techniques and Guidelines for Reclamation of Phosphate Mined Lands. FIPR Publication No. 03-044-095.

Cowell, B.C. 1997. Meiofauna and Macrofauna in Six Headwater Streams of the Alafia River, Florida. FIPR Publication No. 03-101-130.

Erwin, K.L., S. J. Doherty, M.T. Brown, and G.R. Best (Eds.). 1997. Evaluation of Constructed Wetlands on Phosphate Mined Lands in Florida. Final Report FIPR Project 92-03-103.

Richardson, S.G. and C.D. Johnson. 1998. Forested Wetland Restoration and Nuisance Plant Species Management on Phosphate Mined Lands in Florida. Proceedings of the 1998 National Meeting of the American Society for Surface Mining and Reclamation.

Steven G. Richardson
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EXECUTIVE SUMMARY

The current study was designed to address two major gaps in our knowledge of wetlands constructed on phosphate mined lands of Florida: 1) succession of ecosystem structure and function and 2) development of bioindicators for evaluating water quality. Four specific questions were addressed relative to ecosystem succession following construction:

1.   What is the relationship between plant community structure and ecosystem functioning?

2.   Is there a lag time between plant establishment and relative stability in individual ecosystem functional parameters?

3.   Does ecosystem structure and function ever approximate that of a natural marsh, and if so, how long does it take to be attained?

4.   Can ecosystem succession times be altered significantly via system design and initial mulching with organic substrates from natural marshes?

The database for addressing the questions was compiled from 24 marshes located in the central Florida phosphate region, equally divided between natural and constructed marshes ranging in age between just created to ten years of age. Often a subset of this database was used to address specific research questions.

The quantity and composition of system memory, autochthonously produced sediment from plant production, will ultimately stabilize ecosystem fluctuations in functional parameters to approximate those of natural systems, but there is a lag time between initial plant colonization of the newly constructed wetlands and development of sufficient sediment memory to approximate functional aspects of natural systems. It appears that this lag time can be approximately five years for constructed wetland ecosystems of central Florida.

Both the percent macrophyte cover and plant community species richness in constructed wetlands attained values similar to natural marshes approximately one year after construction. In spite of a similar structural appearance to natural wetlands in a short time period, the functional aspects of constructed wetlands took considerably longer to approach stability and were linked to the development of organic sediments from plant detritus. Litter density and mass per unit area accumulated linearly for 3.5 years following wetland flooding, then reached a plateau. Thus, the peak in detritus mass lagged behind above ground plant biomass by approximately 1.5 years. As will be discussed below, it was only after attainment of predicatable sediment composition after 3.5 years that fluctuations in ecosystem physical-chemical and biotic parameters were dampened and interwetland variability declined.

Constructed wetlands were similar to natural wetlands in species richness, percent cover, biomass structure and hydric preferences of plant species within 1- 1.5 years following construction, but did exhibit high intersite variability, high intrasite diversity and more overall coverage by opportunistic species. Exotic plant species displayed higher coverage in constructed wetlands 1-4 years old, then declined to approximate the level found in natural systems after seven years of age. Algal production as indicated by chlorophyll a peaked shortly following inundation of newly created wetlands, then crashed as plant cover was quickly established.

Phosphorus peaked in constructed wetlands within the first 1.5 years, then decreased to approach the lower levels characteristic of natural wetlands between five and eight years after construction. Relative to phosphorus, the nitrogen peak in constructed wetlands was delayed by 2.5 years, peaking approximately four years following inundation. Values then declined to levels approximating those of natural wetlands at about five years following construction. During the first year following construction, nitrogen to phosphorus ratios declined steadily, after which they increased progressively for years two and three. The data are inconclusive due to the pronounced intersite variability, particularly in older sites. It is therefore unclear when N:P ratios may approximate those found in natural wetlands of the area.

Nutrient concentrations and cycling are strongly linked with sediment characteristics. Litter density and mass per unit area accumulated linearly for 3.5 years after wetland formation, then reached a plateau. The peak in detritus mass lagged behind above ground plant biomass by about 1.5 years. It is possible that water chemistry equilibrium is attained after sufficient organic matter has accumulated in the wetland.

The structure of benthic macroinvertebrate communities of constructed wetlands approximated that of natural wetlands three years following system innundation. Biotic structural and functional parameters displaying this trend included Morisita's Similarity Index, diversity, the importance of major feeding guilds, total benthic abundance and taxa richness. Based on a confidence interval analysis, constructed wetlands did not exhibit a trend of increasing stability relative to natural wetlands within the age range (0-8 years) of systems studied.

It is suggested that wetlands constructed on phosphate mined lands of central Florida pass through three distinct development stages. The first is an initial lag phase lasting approximately two years when planktonic primary production is rapidly replaced by emergent plant species. The second phase of development focuses around a trophic surge and decline lasting approximately three years. It is during this phase that phosphorus values peak, then decline to lower relatively stable levels, and nitrogen is progressively accumulted by the ecosystem, especially in plant tissue and the sediments. Throughout this period, organic matter is progressively accumulated in the ecosystem associated with detrital production from emergent vegetation. The final development phase, equilibrium, is acheived only after sediment composition approximates that found in natural systems and

can act to dampen system variability in chemical cycling. This phase is attained at approximately five years following construction, although individual physical-chemical and biological parameters stabilize between three and six years. For most parameters, once equilibrium is attained, values approximate those found in natural wetlands of the area.

Design parameters involving wetland morphology, location with respect to natural wetlands and substrate selection were explored for relationships with water quality, chemical partitioning and accrual within the wetland and plant community structure. The data suggested that a threshold mean depth that allows macrophytes to root in the contoured substrate, as opposed to growing as a floating mat, was the parameter with the most control over water quality in the marsh. Shore development, side slopes and the relative amount of wetland area to upland watershed influenced water chemistry. Plant community structure appeared to be heavily influenced by the donor wetland organic matter used as mulch. The effects of mulching over water chemistry were not significant. Bottom roughness, proximity to natural wetlands, and total water area displayed little control on chemistry or botany over the range of conditions covered by the current database. Although not examined quantitatively, water level fluctuations may be a significant controlling variable for water chemistry and both plant and benthic invertebrate community structure through changes in oxygen regime and redox conditions.

In addition to an assessment of the currently accepted biotic standard for approximating water quality, benthic macroinvertebrates, we examined the possibility of developing water quality indices based on chydorid cladocerans and fish. Although the structure of benthic invertebrate communities in constructed wetlands approximates that of natural wetland within three years of inundation, the reduced range of dissolved oxygen values found in all wetlands, regardless of origin, significantly reduces the utility of benthic invertebrates as water quality indicators. With index reliance almost exclusively on differential taxon tolerance to oxygen, the reduced range of oxygen values found in wetlands and the exceeding low and generally stressful dissolved oxygen values characterzing all wetlands regardless of human impact, the utility of using conventional water quality indices based on benthic invertebrate assemblages from lakes and streams is highly questionable. The current database failed to provide an adequate water quality index based on benthos that could be used for assessment of wetlands of central Florida.

Recognizing the pitfalls inherent in using benthic macroinvertebrates, we also examined the utility of chydorid cladocerans as water quality indicators for central Florida wetlands. While chydorids have proven to be excellent water quality indicators in lakes, their applicability to wetland assessment is hindered by the highly pulsed nature of their populations and the fact that dissolved oxygen levels vary little among wetlands regardless of their origin. Qualitative analyses based on the presence or absence of species suggested that cladoceran assemblages of some constructed wetlands mimic those of some natural wetlands, but the range of assemblages found in constructed wetlands was narrower than those of natural wetlands.

Fish communities were also assessed in both constructed and natural wetlands quarterly for one year using throw traps. Total species richness was slightly higher in constructed wetlands and it appeared that species composition was not limited by recruitment following inundation. Smaller water level fluctuations in natural marshes relative to those of constructed wetlands likely accounted for seasonal differences in fish assemblages. It was not possible to construct a water quality model based on the current fish database.

A final aspect of this research project was estimation of the sample size necessary to characterize benthic invertebrate, chydorid cladoceran and fish communities in central Florida wetlands. It was suggested that six throw trap samples and at least nine core samples are required to assess species richness or density of fish and benthic invertebrates, respectively, in a central Florida wetland. If, however, the goal is to test hypotheses regarding biotic differences between constructed and natural wetlands, the number of samples to statistically assess abundance differences increases to 28 for fish, 156 for benthic invertebrates and 2,450 for chydorid cladocerans. Assessment of species richness for these three biotic parameters requires 56, 40 and 76 samples, respectively.

This study raised a number of questions regarding current practices of monitoring water quality in constructed wetlands and assessment of how closely structural and functional aspects of constructed wetlands approximate those of natural wetlands of central Florida. It appears that water quality indices developed for benthic invertebrates may have limited application to wetland monitoring, regardless of whether such systems are natural or constructed on phosphate mined lands. It also places in question the utility of monitoring benthic invertebrates in constructed wetlands younger than three years, when only those older than three years ever approximate a community structure that resembles natural systems. Attempts to develop alternative biotic indices from chydorid cladocerans and fish proved unsuccessful.

It is recommended that a standard methodology and protocol be developed for monitoring water quality in constructed wetlands, It is unclear when sampling should begin following construction and for how long it should be continued. In addition, both the number of samples per sampling event and the sampling methodology should be standardized.

It is critical that the relationship between wetland design and ecosystem structure and function be delineated. It is essential that all design considerations begin with an understanding of the purpose for which the system is being designed. While the current study has suggested that hydrological regime plays a major controlling influence over plant composition, it is unclear how plant composition, in turn, affects the functioning of contructed wetlands. Finally, it is recommended that the importance of habitat heterogeneity in wetland design in affecting the multipurpose use of constructed wetlands be examined.

INTRODUCTION TO THE STUDY

RESEARCH QUESTIONS

As a result of legislative action in 1975 (Florida Administrative Code Section 16C-16, 16C-17), landscape reclamation is required on all lands mined for phosphate in the State of Florida. Between 1975 and 1991, for example, approximately 40,000 ha of land in Florida were strip mined for phosphate and thus subject to provisions of this legislation. In excess of 2,700 ha of mitigation wetlands have been constructed as part of the land reclamation effort, but the degree to which these systems mimic both the structural and functional properties of natural wetlands is poorly known.

There has been a great diversity in the type of wetland created on phosphate mined lands. The basic design has ranged from relatively steep sided systems with a large proportion of open water areas to shallow swales that quickly become colonized throughout by emergent macrophytes. As a presumed aid in system succession, some constructed wetlands have been lined with a layer of organic sediment collected from donor wetlands in areas to be mined, while the bottoms of other wetlands were left as bare inorganic sediments. Some wetlands were planted with select taxa immediately upon flooding, while plant succession via volunteer taxa was allowed to proceed unimpeded in others. Finally, pronounced interbasin differences in hydrological regime are common among constructed wetlands throughout the phosphate region.

The current study was designed to address two major gaps in our knowledge of wetlands constructed on phosphate mined lands of Florida:

1. Succession of ecosystem structure and function

2. Development of bioindicators for evaluating water quality

Succession of Ecosystem Structure and Function.

Far too often it is assumed that ecosystem biotic structure is equatable with ecosystem function. It has long been assumed that once aquatic macrophytes colonize a constructed wetland and the overall structure of the plant community approximates that of a natural system, then all functional aspects of the wetland ecosystem have been attained as well. The structure and composition of lacustrine soils do not come instantly with macrophyte colonization but are built slowly by successive generations of plants and their associated detritus. Nutrient cycles in wetlands are strongly dependent on transformations taking place in aquatic soils, and there is likely to be a pronounced lag time between when macrophytes colonize constructed wetlands and the soils formed from plant detritus are fully formed. It follows, then, that chemical cycles in contructed wetlands are controlled by system memory, the aquatic soils, and are likely to oscillate greatly until soils are fully formed.

Many faunal groups including benthic macroinvertebrates, perhaps the most important group of bioindicators of water quality, depend on wetland soils/sediments as either a habitat or food source, and thus their community structure should also change as soil structure slowly develops during ecosystem succession. The relationship of higher organisms, including fish, to wetland successional development is also poorly known.

The current project was interested in four questions related to successional trends in the structure and function of constructed wetland ecosystems on phosphate mined lands:

1. What is the relationship between plant community structure and ecosystem functioning?

2. Is there a lag time between plant establishment and relative stability in individual ecosystem functional parameters?

3. Does ecosystem structure and function ever approximate that of a natural marsh, and if so, how long does it take to be attained?

4. Can ecosystem succession times be altered significantly via system design and initial mulching with organic substrates from natural marshes?

Development of Bioindicators.

Demonstration of the value of bioindicators for defining water quality was one of the first research thrusts almost a century ago in the newly emerging field of limnology. The greatest attention was paid to the use of macroinvertebrates, especially chironomid larvae, and the highly predictive relationship between individual chironomid species, and profundal oxygen concentration in lakes during summer were used to define the universally accepted classification of lake trophic state (oligotrophic, mesotrophic, eutrophic). It was firmly established that benthic invertebrate communities could be used to integrate water quality conditions over the life of parameter sensitive organisms, often several months, and the bioindicator system was quickly established as the legal basis of pollution assessment.

At the same time as these lake studies were being conducted in northern Germany and Sweden, limnologists in central Europe began quantifying the relationship between the distribution of macroinvertebrates and relative pollution (oxygen stress) levels in streams. By the 1920's, the importance of benthic invertebrates as water quality indicators was firmly established for streams as well leading to the legal standards currently in place for monitoring water quality.

It has long been assumed that bioindicator schemes that have been developed for lakes and streams can also be applied to monitoring water quality in wetlands. Monitoring of benthic macroinvertebrates has been included as part of permit requirements for

constructed wetlands throughout North America without studies clearly establishing a quantitative relationship between benthic invertebrate community structure and water quality.

Benthic invertebrate species often display rigidly established breeding times during the year, and some species may have only one or two generations per year. The hydrological regime of wetlands is often very unpredictable, and periods of ecosystem desiccation may interrupt breeding patterns of species with moderate (weeks) to long (months) life cycles. In such cases, aquatic larvae would be destroyed, eliminating the possibility of successful reproduction of those species in the wetland without outside recruitment.

The vast majority of bioindicator schemes for determination of water quality in lakes and streams are based on differential sensitivity of individual taxa of benthic invertebrates to oxygen availability. Unlike lakes and streams, the range of dissolved oxygen reported in wetlands is very narrow, and values in most systems approach conditions considered physiologically stressful in other aquatic ecosystems. As there appears to be few invertebrate taxa endemic to wetlands, it is possible that benthic invertebrate communities in wetlands are physiologically stressed even without a pronounced human impact to the ecosystem.

Although a quantitative biotic index employing benthic macroinvertebrates to evaluate water quality in wetlands has not been developed, questions regarding the length of their life cycle and their general sensitivity to the range of conditions characterizing wetlands support the search for alternative biotic groups for development of biotic indices for wetlands. One possible group worth examining in this regard is chydorid cladocerans.

The Chydoridae are predominantly benthic cladoceran zooplankton, most of which browse on detrital material or scrape epiphytic algae. Chydorids possess a number of characteristics supporting their potential use as biological indicators in wetlands:

1. Short life cycle. The typical life cycle of a chydorid is three to seven days. In times of environmental stress, chitin covered resting eggs called ephippia are produced, accumulate in the sediments, and do not hatch until favorable conditions return. Such eggs can remain viable for years even under complete dessication. Thus, unpredictable hydrological events will not eliminate the chydorid community from the ecosystem, merely interrupt their life cyle temporarily.

2. Abundant populations. In general, zooplankton communities in vegetated aquatic ecosystems are over an order of magnitude more abundant than in open water communities.

3. Taxonomic ease. The taxonomy of North American chydorids is in reasonably good condition, and interspecific differences are clearly identifiable with moderate practice.

4. Environmental parameter sensitivity. Numerous studies have demonstrated the sensitive nature of chydorid cladocerans relative to trophic state, pH and conductivity, and quantitative models have been developed for using chydorids for evaluating water quality in lakes. Of additional significance is the fact that the chitinous shell of chydorids is preserved upon the death of the organism, thus permitting the reconstruction of past chydorid assemblages and therefore the water quality condition of the system at the time of production. Such reconstructions have proven extremely valuable as part of paleolimnological investigations of lakes throughout the world.

APPROACH

A total of 24 wetlands in central Florida were utilized for this investigation. The data base for 1989-90 was developed from ten herbaceous (marsh) wetlands constructed on phosphate mined lands and two natural wetlands, and six constructed and six natural wetlands comprised the 1990-1991 data base. Constructed wetlands ranged in age from newly created to in excess of ten years, and in design from shallow to relatively deep. Some were either initially mulched with donor wetland soils or left bare. Most wetlands were planted during the initial flooding period.

Two sampling regimes were employed during this investigation. Sixteen constructed wetlands were selected to represent a gradient of age from newly created to the oldest constructed wetlands on phosphate mined lands. In addition, a suite of natural wetlands were also selected as control sites. The constructed and natural wetlands in the study were divided randomly into two subsets, each of which was sampled quarterly for one year. Two newly created wetlands were monitored throughout the study period to delineate further early successional trends.

A number of physical, chemical and biological parameters were monitored during this investigation. Dissolved oxygen, nitrogen, phosphorus, pH, specific conductivity, calcium lead, iron and copper were monitored in the water column as measures of water quality and plant production potential (trophic state) of the ecosystem. With the exception of dissolved oxygen, these parameters plus degree of compaction and inorganic and organic content were also measured from wetland soils forming in constructed wetlands in order to evaluate correspondance between water column and sediment conditions and the development of system memory during the course of ecosystem aging.

System autotrophic production trends were based on chlorophyll measurements from the water column (planktonic algae) and surveys of macrophyte communities. These data were not only useful for determining the relation between algal and macrophyte production relative to wetland age, but also the successional patterns displayed by the macrophytic community itself Detailed chemical analyses (Ca, Pb, Fe, Cu, TP and TKN)

were also conducted on dried macrophyte tissue samples from wetlands along the age gradient in order to see if the role of macrophytic vegetation as a potential chemical sink for the ecosystem changed with wetland age.

Our investigations of the heterotrophic components of the wetlands focused on two invertebrate assemblages, macroinvertebrates and chydorid cladocerans, and fish. In addition to delineating community trends in species composition and trophic relationships with wetland age for each of these parameters, considerable effort was also expended in defining appropriate sampling regimes and data analytical procedures for benthic invertebrates.

REPORT ORGANIZATION

This report is organized as a collection of published works on various phases of the investigation. Three documents dealt with successional trends in physical-chemical parameters and autotrophic components of constructed marshes:

J.H. Kiefer. 1991. Chemical Functions and Water Quality in Marshes Constructed on Phosphate Mined Lands in Central Florida. M.E. Thesis, University of Florida, Gainesville, FL. 226 pp.

Kiefer, J.H. and T.L. Crisman. 1992. Design Considerations Influencing Water Quality and Plant Community Structure in Reclaimed Freshwater Marshes. pp. 74-85. IN: F.J. Webb Jr. (ed). Proceedings of the Nineteenth Annual Conference on Wetlands Restoration and Creation. Hillsborough Community College, Tampa, FL.

Kiefer, J.H. and T.L. Crisman. 1993. Use of Botanical Indicators as Success Criteria for Constructed Freshwater Marshes - Form Versus Function. pp. 583-587. IN: M.C. Landin (ed.). Wetlands: Proceedings of the Thirteenth Annual Conference of the Society of Wetland Scientists. New Orleans, LA. South Central Chapter, Society of Wetland Scientists, Utica, MS. 990 pp.

Kiefer (1991) focused both on successional trends in chemical partitioning among the water column, plant tissue and developing wetland soils and trends in macrophyte community structure relative to wetland age. Particular emphasis was placed on relating inter wetland differences to both system age and design characteristics.

Three documents examined successional trends in macroinvertebrate and chydorid cladoceran communities with wetland age and appropriate sampling strategies to be employed for maximum information return:

D.L. Evans. 1996, Aquatic Macroinvertebrate Communities of Constructed and Natural Freshwater Marshes in Central Florida. Ph.D. Dissertation, University of Florida, Gainesville, FL. 233 pp.

Streever, W.J. and T.L. Crisman. 1992. The Effect of Inter-Wetland and Intra-Wetland Variability on Sampling Strategies in Faunal Studies Comparing Natural and Constructed Wetlands. pp. 183-196. IN: F.J. Webb Jr. (ed). Proceedings of the Nineteenth Annual Conference on Wetlands Restoration and Creation. Hillsborough Community College, Tampa, FL.

W.J. Streever and T.L. Crisman. 1993. A Preliminary Comparison of Meiobenthic Cladoceran Assemblages in Natural and Constructed Wetlands in Central Florida. Wetlands 13:229-236.

These investigations sought not only to document community trends for both invertebrate groups relative to wetland age but also whether such data would be useful predictors of water quality and by extension, wetland success.

Finally, two papers examined fish populations from constructed marshes and the development of a PC based software package for analyzing species abundance from aquatic ecosystems:

Streever, W.J. and T.L. Crisman. 1993. A Comparison of Fish Populations From Natural and Constructed Freshwater Marshes in Central Florida. Journal Freshwater Ecology 8: 149-153.

Streever, W.J. and T.L. Crisman. 1993. CAS: A New Software Package for Analyzing Species Abundance Data. pp. 217-226. IN: F.J. Webb Jr. (ed). Proceedings of the Twentieth Annual Conference on Wetlands Restoration and Creation. Hillsborough Community College, Tampa, FL.

Abstract of Thesis Presented to the Graduate School of the University of
Florida in Partial Fulfillment of the Requirements for the Degree of
Master of Engineering

CHEMICAL FUNCTIONS AND WATER QUALITY IN MARSHES
CONSTRUCTED ON PHOSPHATE MINED LANDS IN CENTRAL FLORIDA

BY

John H. Kiefer II

December 1991

Chairman: Dr. Thomas L. Crisman
Major Department: Environmental Engineering Sciences

The purpose of this thesis was to ascertain if the chemical functions performed by

constructed marshes attain similarity to natural marshes and, if so, how long this takes to

occur. Ten reclaimed marshes, 0-9 years old, and two natural marshes, located in Central

Florida, were surveyed for a year.

The way marshes partition calcium, lead, iron, copper, nitrogen and phosphorus

was measured in four wetland compartments including the water, plants, detritius and

inorganic sediments. Physical characteristics of the plants, detritus and inorganic

sediments were also examined. The chemical concentrations and physical parameters

measured in the non-water compartments were compared to several water quality

indicators including: calcium, lead, iron, copper, nitrogen and phosphorus concentrations;

pH; specific conductivity; and dissolved oxygen. Design parameters involving wetland

morphology, location with respect to natural ecosystems, and substrate selection were

explored for relationships with water quality.

The data suggested that the chemical functions and water quality in reclaimed marshes five years or older were similar to natural marshes. The timing of this depended on constructed marsh dynamics which appeared to consist of three steps, including: 1) an initial lag lasting approximately two years, 2) a trophic surge and decline lasting about three years, and 3) an equilibrium phase. This equilibrium coincided with the accrual of a dense detrital layer in the marsh. Mean depth, the amount of area covered by emergent macrophytes, shore development, and side slope were the design parameters that correlated most strongly with water quality.

Design Considerations Influencing Water Quality and Plant Community Structure in Reclaimed Freshwater Marshes

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Abstract

The water column was analyzed for one year at ten reclaimed marshes on phosphate mined land. Parameters tested were total phosphorus, Kjeldahl nitrogen, calcium, iron, copper, and lead in the water column, inorganic sediments, detritus, macrophyte tissue, specific conductivity, pH and dissolved oxygen. Two natural marshes located in central Florida were sampled concurrently for baseline comparisons. Marshes less than three years old lacked a significant detrital layer and water quality was a time-dependent factor as these systems stabilized. Therefore, only seven marshes greater than three years old were assessed for design considerations. Only sites inundated year round were sampled. Design parameters involving wetland morphology, location with, respect to undisturbed ecosystems, and substrate selection were explored for relationships with water quality, chemical partitioning and accrual within the wetland, and plant community structure. The data suggested that a threshold mean depth that allows macrophytes to root in the contoured substrate, as opposed to growing as a floating mat, was the parameter with the most control over water quality. and normal chemical relationships within the marsh. Shore development, side slopes, and the relative amount of wetland area to upland watershed influenced wetland chemistry. Plant community structure appeared to be heavily impacted by mulching. Bottom roughness, proximity to natural wetlands, volume development, and total area displayed little control on chemistry or botany over the ranges set by this database. The effects of mulching on water quality were not significant. Water level fluctuations effects were not evaluated, but may have been significant.

Introduction

Natural marshes usually perform several valuable functions within a landscape, affecting water quality and quantity and providing wildlife habitat. Industries which cannot avoid destroying natural marshes are required to mitigate for their toss by constructing replacement wetlands. Wetlands designed with different morphologies offer an opportunity to explore what factors contribute most substantially to marsh function. This information can be used to develop engineering strategies to build wetlands that perform at a desired level. Improvements may also be made at existing marshes. For example, a wetland next to a highway may provide excellent duck habitat, but because of its location, is devoid of such birds. If this wetland is destroyed, it may make sense to construct one in its place that may have features less useful to ducks,. but render it more capable of treating runoff pollutants from the nearby road. It is necessary to

understand what factors influence water quality and habitat structure in order to accomplish such goals.

Stochastic elements, coupled with a tendency for self-organization, render simple cause and effect relationships between, design parameters and desired marsh function difficult to model. Many studies have been published on the chemical functions performed by natural marshes. Chemical dynamics within a marsh depend on complex feedback loops driven by microbes, plants, sediment structure, and aquatic loading to the system (Kadlec, 1987). Studies on the chemical functions of constructed marshes have focused on wastewater treatment wetlands rather than on systems built to restore a full compliment of wetland functions. Most of the design studies focused on the effects of hydraulic distribution and residence time on treatment efficiencies (Hammer; 1989). In the present study, we explored the control certain morphological variables had on the water quality, chemical partitioning, and plant communities within freshwater marshes constructed on phosphate mined lands in central Florida.

Study Sites

Ten constructed and two natural marshes located in central Florida were sampled (Figure 1). Seven of the reclaimed marshes were old enough to have reached chemical equilibrium and were used in the design analysis. These marshes ranged in area from 2.4-83.4 ha and were supported by drainage basins 1.4-6.7 times as large as the wetland (Table 1). All marshes were constructed on mined lands on overburden soils. Four of the marshes were mulched with an organic topsoil from a donor wetland. All the marshes studied exhibited a hydroperiod close to 100% at the sampling stations, but then represented a wide range of morphological features (Table 1).

Materials and Methods

Two adjacent sampling stations typical of the marsh were established in each site. The stations consisted of blocks approximately 100-500 m². Grab samples were taken during 3 sampling periods, beginning in January and -ending September, 1990, from each station for the analysis of chlorophyll-a, pheophytin-a, TKN and total phosphorus (TP) in the water column. Dissolved oxygen, pH, and temperature were measured in situ during these collections with a YSI model 54 DO meter and Fischer pH pen. Four grab samples (two from each station) were taken during a single sampling event in October for the analysis of total Ca, Pb, Fe, and Cu. Above ground plant tissues, detrital cores, and sediment cores were collected at this time for the analysis of the above referenced metals, TP and TKN.

Pigment determinations were made in accordance with the spectrophotometric and trichromatic procedures of Standard Methods 1002G.1 and 1002G.3 (APHA,

Figure 1. Florida phosphate deposits and study sites (adapted from Blakey, 1973).

■ Northern-pebble district

▨ Hardrock district

▦ Central-pebble district

● Natural marsh

✗ Reclaimed marsh

0 20 40 60 80 100

Miles
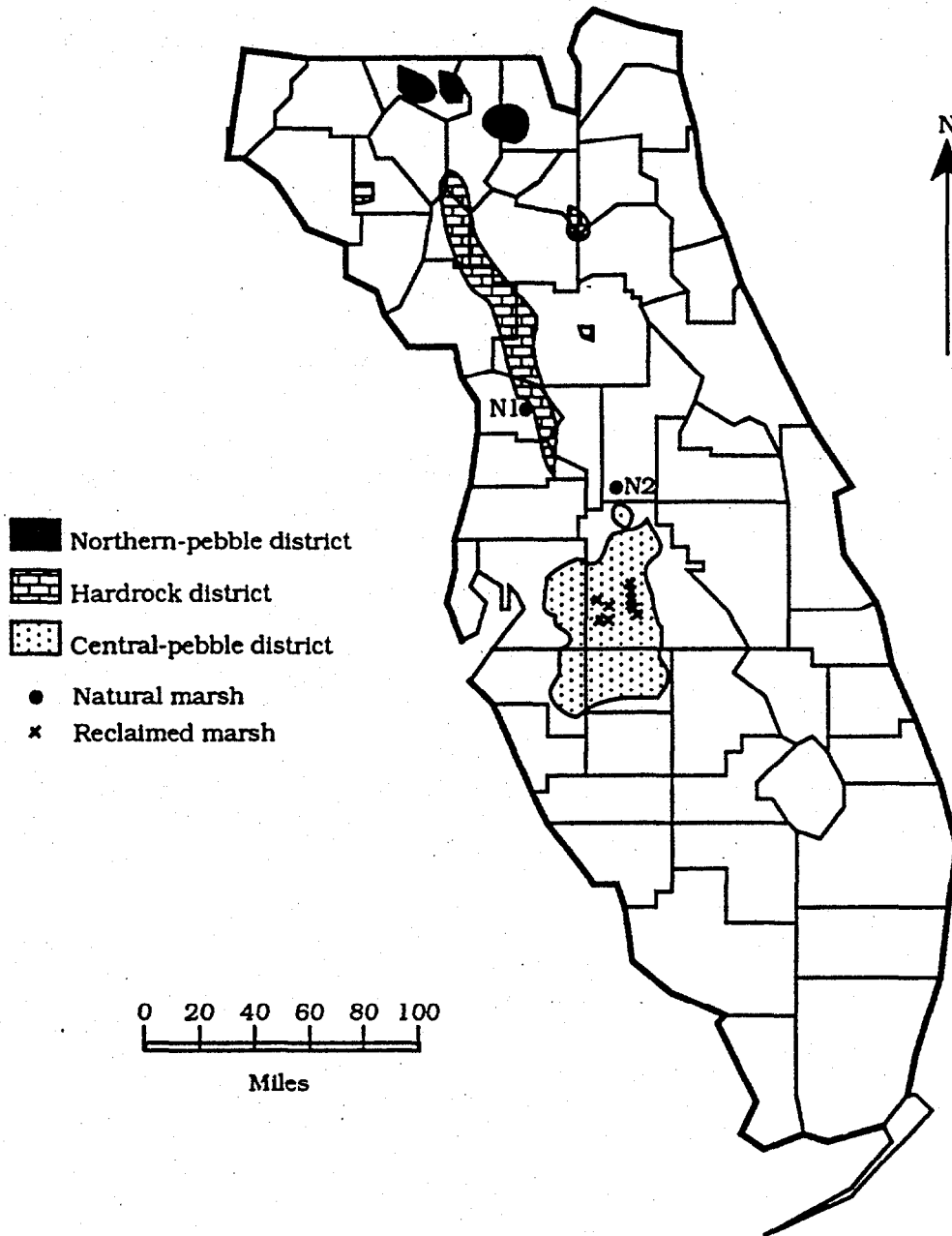
Table 1. Design parameters.

Marsh	TA (Ha)	Z (cm)	zmax (cm)	R	S (cm/m)	mulch	Sd	Ao/Av	Au/Aw
CS83	2.4	58 (2)	79 (3)	1.18	4.00 (2)	yes	2.1	0.34	3.0 (3)
CS84	10.5	40	69	1.65	1.24	no	2.3	2.49	1.7
CS85	6.9	63 (1)	86 (2)	1.20	3.42 (3)	yes	2.2	0.04	6.7 (1)
CS86	24.3	35	71	1.05	1.67	no	2.6	0.31	1.6
CS87	5.3	33	86 (1)	1.05	4.42 (1)	no	3.2 (1)	1.37	4.0 (2)
FG82	60.7	34	53	1.17	1.63	yes	2.3	0.27	1.4
FG86	83.4	23	28	1.75	0.52	yes	1.8	0.11	1.5

() rank in parenthesis

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1975). Values reported are functional chlorophyll-a corrected for interferences due to turbidity and other pigments. Percent transmittance was read on a Milton Roy Spectronic 20. TP was determined via EPA method 365.4, TKN by method 351.2, and the metals by method 287 (EPA, 1979). Percent transmittance was read on the Milton Roy Spec 20 for TP and a Technicon Autoanalyzer II for TKN. Total metal concentrationa were read on a Perkin-Elmer model 5000 atomic adsorption spectrophotometer. Chemical procedures were performed using equipment maintained by a Florida Department of Health and Rehabilitative Service certified lab at the University of Florida's Department of Environmental Engineering. Five percent of all samples were run as duplicates. If duplicates differed significantly, the analysis was repeated.

Water and Air Research, Inc. conducted a single vegetation survey in each permanent station. Transects were established perpendicular to the wetland edge with methodology following Florida Department of Environmental Regulation (FDER) specifications. Species and percent cover of each type were determined within 0.25 m^2 frames at 6 m intervals. Herbaceous vegetation was classified as ground cover (< 50 cm high) and lower understory (> 50 cm). This data was used to calculate species richness, Shannon-Weaver diversity, Simpson's dominance index and Morisita's Index of overlap (Horn, 1966). Supplemental surveys were made during November to determine percent cover by community type defined by vertical structure of the member plants (Bed1 = Pontedaria&Sagittaria; Bed2 = Typha; Bed3 = low mixed herbaceous; Bed4 = open water). Core samples were collected to determine the height and density of the detrital layer and the depth of visual organic matter in the sandy substrate. Plant tissues were collected from the communities above within 30.5 diameter sampling rings for the analysis of above ground biomass. Percent sand, silt and clay in the sediments were determined using ASTM method D 422-63 (ASTM, 1985). Percent organic matter was determined on dry sediments by incineration at 550 degrees C in a muffle furnace.

Design variables explored included total wetland area (TA), the areal ratio of permanent open water pools to emergent macrophyte beds (Ao/Av), mean depth (z), maximum depth (zmax), initial substrate composition (overburden, mulch), bottom roughness (R), bottom slope (s), proximity to natural water bodies (eprox, cprox), ratio of upland watershed to wetland area (Au/Aw), volume development (Vd), and shore development (Sd).

Z, zmax, R, s, substrate, and Vd were derived from data collected at the stations surveyed for the chemical parameters. They may not be indicative of the wetland as a whole and are microscale design parameters. If wetland water chemistry is heterogeneous, they may not be representative of the entire marsh. Stations were picked to be as representative of the marsh as possible. Depths were measured in January at 6.0 m intervals along the full length of each station. Therefore, z and zmax are not annual means and, based on when they were measured, are close to seasonal low water levels. The depth data here are useful for exploratory purposes.

R was defined as the ratio of the actual bottom length to that of a second order regression line fit through the bottom geometry, High values of R indicate a high degree of bottom irregularity. S was defined as the slope of alinear regression

fit to the bottom configuration of the first 20 m from shore (cm/m). Vd measures the similarity of the water volume contained by the depression of a cone with the

same surface area and a depth equal to zmax. A conical depression scores 1.6 and most natural lakes score above this value (Wetzel, 1983).

Macroscale parameters (based on the entire marsh) included TA, Ao/Av, Au/Aw, eprox, cprox, and Sd. Eprox is the shortest distance between the two water bodies and cprox is the shortest distance from the wetland's centroid to the edge of the nearest natural water body. Sd measures the amount of shoreline in comparison to the circumference of circular water body of identical surface area. This parameter can be used as an assessment of edge effects on a water body (Wetzel, 1983).

Results and Discussion

This study incorporated several wetlands of various ages. Differences in water quality potentially depend on temporal, as well as site-specific characteristics. Therefore, the water quality of the marshes was compared by running a cluster analysis on a set of variables, including conductivity; mean aqueous Ca, Pb, Cu, Fe, TP, and TKN concentrations; mean surface and bottom DO concentrations; and pH.

The cluster analysis assigned wetlands to three major groups; (1.) constructed marshes less than a year old, (2.) constructed marshes with water quality similar to the natural marshes surveyed (N1, N2), and (3.) constructed marshes greater than two years old with water quality dissimilar to N1 and N2. The three wetlands in Group 3 were CS83, CS85, and CS87. Group 1 wetlands were assumed to be immature and were not included in the design analysis.

Of the seven wetlands used in the design analysis, Group 3 wetlands ranked in the top three for Au/Wa, s and zmax. CS85 and CS83 ranked 1 and 2 respectively, for z (Table 2). All three wetlands are steep sloped, deep bowls supported by large watersheds (Table 1). Potentially high nutrient loads from upland runoff combined with the deep water to favor the development of floating mats of vegetation, as opposed to vegetation growing from the contoured substrate. CS83 was completely covered by such a mat, CS85 had a shallow lip of emergents about 5 m wide between the mat and shore, and CS87 had about 15 m lip. The floating mats seem to impact water quality. These mats differ from natural mat communities because they were devoid of any hydric soil material and the roots were fully exposed to the water column,

Wetland species of emergent macrophytes transfer atmospheric oxygen to the flooded anoxic substrate in which they grow (Mendelssohn and Postek, 1982; Bacha and Hossner, 1977; Taylor et al., 1984). This establishes an interlaced network of oxidizing zones in a reducing matrix. Sediment pore water is hydrodynamically connected to surface water allowing diffusion (Wetzel, 1982; Stumm and Morgan, 1981), and enables substrate redox conditions to affect nitrogen-denitrification coupling (Bowden, 1987). Reddy (1983) showed NO_3 and NH_4 aqueous partitions

Table 2. Water quality.

Parameter	CS83	CS84	CS85	CS86	CS87	FG82	FG86	N1	N2
DO-surface	2.2 (8)	4.3	1.1 (9)	5.3	2.9 (7)	4.5	5.1	4.5	5.0
DO-bottom	0.7 (9)	5.2	0.8 (8)	1.3	2.5	2.3	3.1	1.5	2.8
cond. (umho)	228	147	69	143	247 (1)	246	119	82	64
Ca (ppm)	20.2	27.8	8.0	9.5	37.8 (1)	23.9	10.9	20.0	2.5
Cu (ppb)	2.7	6.2	5.3	17.6	3.5	0.0	2.6	2.6	3.5
Fe (ppm)	3.1 (3)	1.4	4.0 (1)	2.1	3.5 (2)	0.6	0.8	0.5	0.8
Pb (ppb)	18.8	12.5	0.0	12.5	25.0 (1)	6.3	12.5	25.0	12.5
TKN (ppm)	4.5	2.0	5.9 (1)	3.2	4.6 (3)	1.4	4.7	1.5	2.4
TP (ppm)	1.6	1.1	2.4 (2)	1.8	5.2 (1)	0.2	2.4	0.2	0.2
pH	7.2	7.8	6.4	7.6	7.6	8.0	7.4	6.9	6.5
chla.(mg/m3)	2.4	2.7	18.4 (1)	13.1	4.7	1.7	13.2	1.9	10.2

() rank in parenthesis

of 0.11.5% and 3.4% in floating plant systems verses water column partitions of 0.1% in emergent systems for both.   Root redox can also effect the solubility of metals in wetlands. (Bacha and Hossner, 1977; Taylor et al., 1984). Phosphorus sometimes behaves similarly to the metals because of its tendency to complex with iron in most natural waters. This may have been the case in Reddy's (1983) study where floating systems retained 64% of TP dosed to the system in the water column, and emergent beds sequestered only 40% in aqueous solution.

CS83, CS85, and CS87 demonstrated individual characteristics worth further exploration. CS83 showed deficiencies in the areal accumulation of TP, TKN, and Pb which was attributable to low littermass.   Without emergents, the aqueous chemistry of flooded systems probably reflects sediment conditions more strongly. CS83 had low sediment chemical loading, and this could be contributing to its low areal accumulations.   . Without any rooted macrophytes, CS83 displayed higher percentages of TP, TKN, Pb and Fe in its water column than marshes of similar. age.

CS85 had a higher sediment area Fe loading than CS83 (which is typical of the other wetlands). With little emergent bed, Fe transferred into the non-sediment compartments of CS85 at levels much higher than at the other marshes. However, comparisons of partitioning percentages revealed that CS85 had enough of an emergent lip to prevent widely aberrant behavior in nutrient and metal partitioning.

The floating mats of CS83 and CS85 probably contributed to depressed oxygen regimes as these wetlands ranked last in average surface and bottom DO (Table 2).   Such mats shield the water surface from sunlight and wind, thereby reducing algal and diffusional oxygen supply to the water column. The deep water also reduces diffusional effects with decreased surface to volume ratio.

CS87 clustered with the two other systems as a marsh with aberrant water quality. It was a system with a floating mat, but was not dominated by this mat. Its aberrance was probably derived elsewhere. CS87's Sd was unique and this wetland was a lobate body fed by trenchlike fingers (Table 1). Its conductivity, Ca, Pb, Fe, TKN, and TP concentrations ranked first or second (Table 2). During the field surveys of this marsh shortly after rainfall events, its water became cloudy. Several other marshes were surveyed after similar events and no cloudiness was observed. It would appear that CS87's lobate shape and channelization allowed runoff laden with particulates and with dissolved metals and nutrients to overwhelm the filtering capacity of this wetland. Brown (1988) found that wetlands with channels retained less nutrients than unchannelized systems, and steep sloped channels eroded more, providing even less retention. This appeared to be the case for CS87.

Pearson product moment correlations were used to assess the potential sensitivity of measured parameters to design factors.   All Pearson correlations discussed have $p < 0.05$. S potentially enhanced Fe concentrations within the water column. ($r=0.87$), macrophyte tissue ($r=0.77$), and detritus ($r=-0.79$).   S may have increased the percent of TP partitioned into the water column ($r=0.89$), as well as the total amount of Fe ($r=0.84$) and TP ($r=0.90$) in the water column,   It also potentially repressed surface DO ($r=-0.81$). S correlated with zmax ($r=0.83$). Fe and TP are ubiquitous in the soils of the area and their 'enrichment in the wetland

waters appeared to be a function of a combination of erosional and redox factors related to steep slopes declining to deep water.

Sd also correlated with aqueous TP concentrations (r=0.83). Partitioning of TKN (r=0.87), TP (t-=0.86), and Cu (r=0.90) into macrophytes appeared to be enhanced by Sd, while the detrital partitioning of these chemicals correlated negatively (r=-0.85, -0.84, -0.82, respectively). Areal loadings of Ca (r=0.78, Fe (r=0.82), TKN (r=0.84) and TP (r=0.83) increased in the macrophytes with Sd. It is difficult to explain why Sd would enhance macrophyte loadings seemingly at the expense of the detritus. It is possible that the fingered lobes resulting in high Sd decrease the hydraulic contact time of upland runoff through the detritus. The post depositional accumulation of nutrients and metals on the detritus is a function of the hydraulic residence time (Hammer, 1989).

Z correlated negatively with surface DO (r=-0.86). Areal loads of Fe (r=0.88) and TKN (r=0.82) increased with z. In the case of Fe, this was probably caused by lowered redox conditions. A survey of 57 marshes constructed to control acid-mine drainage found that deeper wetlands removed less Fe and Mn from the influent (Wieder, 1989). Z apparently limited the thickness of the detrital layer (r=0.82). The elevation at which detrital accumulation balances oxidation is probably dependent on the height of the water column because diffusional oxygen transfer is not appreciably enhanced by wind mixing in densely vegetated marshes. Therefore, thicker accumulations would be expected in deeper water.

Ao/Av correlated strongly with sediment Pb (r=0.87) and Cu (r=0.83) concentrations and areal loading of Ca (r=0.88), Cu (r=0.92), and Pb (r=0.87) in the sediments. Bottom DO increased with Ao/Av (r=0.79). This implies a well mixed water column which may advect a higher flux of metals to the sediment/water interface. Open water is more susceptible then emergent beds to mixing caused by wind, and such mixing is a function of the fetch and depth of the water column (Wetzel, 1983). It stands to reason that more open water would result in greater top to bottom mixing. Sediments often act as chemical sinks in open water habitats (Jeanne, 1977).

TA, and R did not display any meaningful correlations with water quality or plant community structure. Both of these parameters are probably important for wildlife considerations. It is possible roughness organizes micro-habitat for plant assemblages on a scale inappropriate for the methodology used. A greater diversity of wildlife are likely to utilize constructed wetlands that are large and/or are connected to larger habitat areas by wildlife corridors. This study did not explore such issues. Proximity to natural water bodies can also potentially effect wildlife usage. Eprox and Cprox yielded no interpretable correlations.

Mulching, the placement of topsoil/detritus from a natural donor marsh, was assessed by an analysis of variance of selected parameters. When referring to the NOVA comparisons, the word "significant" will be used exclusively to refer to differences in means that have less than a 5% chance of resulting purely by chance. CS83 was deleted from the mulch analysis because it contained no emergent bedding and was somewhat dysfunctional. Six marshes remained for this analysis; three mulched (FG82, FG86, CS85) and three unmulched (CS84, CS86, CS87).

Constructed wetlands are typically mulched to enhance the establishment of diverse macrophyte communities. ANOVA showed mulching significantly reduce the coverage of beds dominated by Typha. This supports Erwin and Best (1985). Mulched beds average 18% Typha dominated beds (Bed2), while unmulched systems average 55% coverage. Pontedaria communities (Bed1) covered 51% of the area in mulched marshes and 28% of unmulched systems. Diversity appears to be greater in mulched systems, as well with miscellaneous. communities (Bed3) occupying 21% and 1% of mulched and unmulched area, respectively. Unmulched areas showed slightly higher unvegetated areas within the vegetated zones than mulched marshes (16% versus 10%).

ANOVA showed the thickness of the litter in mulched and overburden systems to be similar (17.0 cm and 13.3 cm; respectively). However, mulching may have led to significant differences in litter density and detrital mass/area. Litter density was twice as high, and littermass was over 2.5 times as high in mulched marshes.

Most chemicals are partitioned predominately in the wetland substrate, including the ditritus (Faulkner and Richardson, 1989). Detrital accumulation goes through a positive feedback stage where increase organic matter imposes a BOD, causing a shift to a less efficient breakdown of organic matter (Feijtel et al., 1989). These facts, coupled with the possibility that mulching accelerated the formation of a dense, bulky litter, imply that mulching may impact water chemistry via chemical partitioning and stimulation of detrital accumulation and microbial communities.

Few such relationships were found at the 95% confidence level. However, mulched marshes. consistently displayed lower mean partition percentages and concentrations for most chemicals in the water column. Concentrations and partitions were similarly raised in the detritus and repressed in the plants with few exceptions. DO levels were slightly lower in mulched wetlands. Mulching may cause the wetland's ability to reduce chemical accumulation in the water column and living plants by prolonging storage in the detrital mass. This possibility is based both on partition percentages and areal concentrations. Enhanced functionality is implied because concentration and partition percentage changed.

Conclusions

1. High nutrient loads in the water column, caused by a combination of steep side slopes; high upland to wetland area ratios, and excessive shore development, promoted the formation of floating plant communities in wetlands with the greatest mean depths. Such wetlands demonstrated inferior water quality and did, not consistently partition nutrients and metals among the detritus, living biomass and water column similarly to systems (natural and constructed) that were dominated by emergent plants rooted in the substrate.

    Design vegetation hydroperiod and depth should allow an emergent to develop. This depth is a function of nutrient loads to the water column, because floating communities persist only under high nutrient regimes.

Therefore, the system should be evaluated in the context of its surrounding land use and topography. Wetlands with the following criteria performed similar to natural marshes: (1.) seasonal low water (SLW) less than 35 cm, (2.) S less than 2%, (3.) Sd less than 3, (4.) Au/Aw less than 2. It would be inappropriate to use these guidelines as absolute limits because wetlands display large, overlapping hydroperiod and depth ranges that support emergent macrophytes. For example, SLW greater than 35 cm may be appropriate for wetlands supported by relatively small basins. Conversely, SLW at 35 cm may form a floating community if the watershed consists entirely of intensive agriculture or paved surfaces with high nutrient runoff.

2.   It may be desirable to design wetlands near the upper limit of depth allowing for the development of emergent growth because of the ultimate thickness of the detrital layer and system respiration were potentially limited by mean depth.

3.   Mulching significantly accelerates the development of a dense, bulky detrital layer and a diverse botanical community. Mulch may have augmented wetland chemical functions, but this conclusion remains tentative.

4.   Within the bounds of this study, TA, Vd, R, and proximity were insignificant controls over water quality or plant community structure. Microtopography should be explored in greater detail and may have been addressed at an appropriate scale by this study.

5.   Ao/Av was not a major control over water quality, but may have resulted in higher sediment loads for the wetland with the highest ratio (2.5). More research is needed concerning the effects of Ao/Av because factors such as fetch and continuity probably influence wave mixing and will affect wetland performance. Wetlands with less than 30% ponded areas performed well.

6.   Hydroperiod significantly controls seasonal water quality patterns and impacts overall chemical function.

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USE OF BOTANICAL INDICATORS AS SUCCESS CRITERIA FOR CONSTRUCTED FRESHWATER MARSHES -- FORM VERSUS FUNCTION

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Abstract: Botanical communities in 11 marshes constructed on phosphate mined lands were compared to those in 11 natural marshes located in central Florida. A subset of 10 constructed and 2 natural marshes was surveyed for nutrients and metals in the soils, macrophytes and water column. Parameters describing soil structure, primary production, and morphological factors were measured. This data was used to establish which wetlands provided chemical functions similar to reference marshes. Reclaimed marshes with natural levels of percent cover, diversity, dominance, and species richness often did not display a high degree of overlap with natural marshes as measured by Morisita's index. Some constructed wetlands which provided low botanical similarity to reference marshes performed chemical functions like natural, systems. Similarity of form did not, uniformly imply similarity of function. This could be attributed to the temporal dynamics of developing ecosystems and the high variability in the plant communities studied.

INTRODUCTION

The degree of success attained by constructed marshes is typically assessed by monitoring the macrophyte communities and comparing marsh flora at the species level. Apparently, it is assumed that because community composition is largely a function of aquatic chemistry, soil type, hydrologic regime and interactions among the biota, that floristic similarity to a reference marsh is a conservative measure that integrates the functional attributes of a reclamation project. This assumption models ecosystem development as a non-random, or completely deterministic, phenomenon. Similarity in communities can occur on different hierarchies above the species level such as in biomass structure, where different species with similar adaptations can occupy similar niches and provide comparable habitat (Mueller-Dombois and Ellenberg 1974). The purpose of this study was to assess the relevance of floristic overlap as a functional marker in constructed and natural freshwater marshes and to explore any hierarchical aspects of similarity in the plant communities within these two types of ecosystems.

STUDY SITE

Twenty-two freshwater, herbaceous wetlands located in central

Florida were surveyed. Eleven were strip-mine reclamation
projects constructed by phosphate mining companies. Nine were
natural marshes located within the phosphate district. Two
additional reference marshes were located outside the district but
within the same climatic zone *in* the central peninsular region
(Beaver et al. 1981, Dohwender and Harris 1975).   The systems
ranged from 2 to 200 acres.

METHODS

    Permanent sampling sites were established in the 22 marshes,
one site per marsh.   The size of the site ranged from 500-2000
square. meters and was dependent on the variability of habitat
within the location of the access point. Sites were picked to be
as representative of the marsh as possible. Transects were run
perpendicular to the shore and vegetative cover was determined by
species within 0.5 m2 rectangular quadrats on 5 meter intervals.
Transects were run until the all the major species within the site
were accounted for.

    Ten of the constructed sites, 0-8 years old, were sampled
during 1990 on three events for water quality parameters (TP, TKN,
Ca, Cu, Pb, Fe, specific conductivity, surface DO, bottom DO, pH
and chlorophyll-a).   Two samples were collected per event, giving
a total of 6 samples per wetland. Sediment, detritus and above
ground macrophyte tissues were collected on a one time basis and
analyzed for the nutrients and metals above. Soil bulk density,
above ground biomass, biomass structure, water depths, and wetland
morphology were also measured during this collection period. All
collection methods and laboratory analyses were performed in
accordance with standard procedures (APHA et al. 1975, EPA 1979,
ASTM 1985).   The two natural marshes, N1 and N2, outside of the
phosphate district were sampled as reference marshes concurrently
with the ten constructed marshes for the parameters discussed
above.   The combination of data collected above was used to
calculate the amounts and percentages of nutrients and metals
partitioned into various above ground wetland compartments.

    Analysis of variance (ANOVA) was used as a univariate
comparison for selected means. Differences were considered
significant at the $p < 0.05$ level. Multivariate comparisons were
facilitated by using principal components and clustering
techniques.   Constructed wetlands were analyzed for potential
temporal fluctuations by sorting these systems by age and
exploring the data for meaningful clusters and trends.

RESULTS AND DISCUSSION

    cumulative species curves were generated as a function of the
number of sites sampled and these curves indicated that a more
extensive sampling effort would be necessary to account for the
variability of herbaceous hydrophytes in the region. Seventy-one
species were found in the 11 constructed sites verses 69 in 11
reference sites.   Logarithmic equations fit to the species
accumulation curves (r=0.98) suggested that the constructed sites
had greater inter-site variability than the reference marshes. If
one assumes a pool of 200 common herbaceous macrophyte species are

available in the region, the species accumulation equations showed that a sampling effort of 100 constructed sites, verses 140 natural sites, would be necessary to fully assess if the systems supported these species. Therefore, this study was inadequate to assess the overall floristic overlap of constructed and reference marshes of the area although it is acceptable to check calculated similarity values for correlation with the functional attributes of each site.

Significant differences appeared in the overall intra-site diversity as measured by the Shannon-Weaver information index (C,constructed = 2.08; N,natural = 1.76). Species richness was similar (C-15.6, N=16.2), but significant differences occurred in, Simpson's dominance index (C=0.17, N=0.27) which measures the odds of drawing the same species from a site on consecutive random pulls (Colinvaux 1986). Developing ecosystems often display higher diversities than mature areas because they need time for direct competition to organize or Select species for optimal resource utilization. The constructed wetlands supported similar species numbers but none of the species were able to outcompete and dominate the other forms to the extent found in the natural marshes over the timeframe explored.

Morisita's index measures the odds of pulling the same species from two different sites on consecutive samples relative to accomplishing this within the sites (Horn 1966). Nine of the natural marshes were compared to two other reference marshes, N1 and N2, as reference sites. Florida regulatory agencies-often require wetlands to exhibit a score of 0.60 or higher as an indicator of a successful project and this value was used in this study as the cutoff for referring to a pair of sites as similar. None of the natural marshes were scored greater than 0.18 when compared to Nl. Four were similar to N2. N1 was the most diverse natural system in the study (H'=2.45). N2 and the marshes most similar to it were dominated by <u>Pontedaria cordata</u>, which allowed for the high overlap scores. N6 displayed the closest similarity, 0.92, to N2. This system, although it had an acceptable floristic composition, was bisected by State Road 62 and was surrounded by cattle pasture to the south of the road. Low similarities to N1 could be attributed to the high diversity of this marsh.

Two of the constructed sites were similar to the reference marshes (83/N1=0.65, 90/N2=0.91). 83 was similar to N1 because both areas contained significant amounts of <u>Hydrocotyle umbellata</u>. 90 was similar to N2 because of a prevalence of <u>Pontederia cordata</u> at both sites. Neither of these marshes were partitioning chemicals similarly to natural marshes at the time of the study. Nl and N2 partitioned over 80% of the areal accumulation of nutrients and metals in the detritus, less than 5% in the water column and approximately 10-15% in above ground plant tissues. Properly designed marshes over 4 years old partitioned chemicals similar to the reference sites. 83 partitioned over 15% into the water column because of design flaws, while 90 partitioned over 50% into the water column because this system was too young to have developed an appreciable detrital layer. Overlap as measured by Morisita's index failed to correlate with proper chemical partitioning in freshwater marshes.

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Wetland jurisdictional assessments are made based on plant species preference for wet soil conditions. The United States Army Corps of Engineers classifies plants according to the probability that the plants occur in wetlands. Means, weighted according to percent relative frequency, of the midrange of these probabilities were used to generate a hydrophytic index based on the species at each site. A score of 100 indicates a wetland completely dominated by obligate hydrophytes, while a score of 0 indicates an area completely dominated by upland species; No significant differences occurred between the constructed and reference marshes, with mean scores of 96.4 and 96.3 respectively.

The biomass structure of the intensively surveyed sites was defined by the above ground macrophyte biomass, chlorophyll-a in the water column, total percent cover, percent open water, and percent areal cover of communities with different maximum strata (separated into communities dominated by plants less than 50 cm, 50-150 cm, and greater than 150 cm high). This hierarchy of comparison was intended to provide information on the habitat qualities of the site and was evaluated by using the EML multivariate agglomerative clustering technique (SAS 1985). The 12 marshes clustered into 3 major groups. Group1 included N1 and the three oldest of constructed marshes (82, 83, and 84). Group2 included 87 and 86, two marshes dominated by Typha communities. Group3 included N2 and the remaining five reclaimed marshes. From this comparison, and the similarity of the mean hydrophytic index scores, it is apparent that the sites have attained a hierarchical similarity to the reference marshes above the species level.

Developing ecosystems usually have higher percentages of opportunistic species than do mature communities. The constructed systems had significantly higher percent relative cover of small floating plants (e.g. Wolfiella, Wolfia, Azolla etc.) than the reference marshes (43.9% verses 17.2%). These plants have excellent dispersal mechanisms. More research is needed into why they thrived at the constructed sites. No correlations with open water (shade) or water quality could be found. Hydrological differences in the sites may have played a role or this could be attributed to stochastic factors involving dispersal mechanisms and primacy of establishment.

Exotic species prefer disturbed or developing ecosystems. Exotic species richness was similar at the sites (C=1.0, N=1.1), but percent relative cover was higher in the constructed areas (C=7.4, N=1.3). Exotic cover peaked 1-4 years after construction and gradually declined to natural levels after 7 years. This surge and decline pattern is typical of exotic species reacting to a spiked (as opposed to a continuous) disturbance in natural ecosystems. This pattern and the cover of other opportunistic species indicate that the constructed systems are developing ecosystems.

The high variability and temporal trends of such systems make floristic overlap indices difficult to use as functional markers when evaluating design procedures and the proper construction and maintenance of reclaimed wetlands. It may be more appropriate to

compare trends in these sites over a period of several years to expected trends in natural wetlands reacting to a temporary disturbance. In general, the constructed systems reacted like developing ecosystems with higher variability between sites higher diversity within sites, and more cover by opportunistic species than the reference marshes surveyed. This study was not adequate to address the floristic overlap of constructed and natural marshes in the region, but similarities were found in the species richness, percent cover, biomass structure and hydric preferences of the plants present.

ACKNOWLEDGMENTS

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Abstract of Dissertation Presented to the Graduate School
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Requirements for the Degree of Doctor of Philosophy

AQUATIC MACROINVERTEBRATE COMMUNITIES OF CONSTRUCTED AND
NATURAL FRESHWATER MARSHES IN CENTRAL FLORIDA

By

David L. Evans

August, 1996

Chairperson: Dr. Thomas L. Crisman
Major Department: Environmental Engineering Sciences

Macroinvertebrate communities of 8 natural wetlands
and 11 constructed wetlands of varying age were sampled
quarterly for 1 year. Shifts in macroinvertebrate
assemblages occurring during constructed wetland development
were estimated using data from constructed wetlands of
varying age. Macroinvertebrate assemblages of natural and
constructed wetlands were compared and the efficacy of using
macroinvertebrates as indicators of constructed and natural
wetland function was evaluated.

Natural wetlands exhibited qualities more similar to
those of older constructed wetlands (>3 years old) than to
the qualities of newly constructed wetlands, providing
evidence of directional changes over time. Parameters
showing this pattern included macroinvertebrate community
structure (Morisita's Similarity Index), structure of feeding
guild, community macroindicators (total organism abundance,

taxa richness, and diversity), degree of siltation, and vegetation density.

Based on a confidence interval analysis, constructed wetlands did not exhibit a trend of increasing stability relative to natural wetlands within the age range studied (8 years).

General hydrological characteristics, which can influence macroinvertebrate communities, did not demonstrate the described pattern. For example, some of the natural wetlands studied went through a dry down phase during the dry season, whereas all constructed wetlands were constantly inundated. Consequently, natural wetlands supported a higher number of semi-terrestrial invertebrate species than constructed wetlands.

Sediments in constructed wetlands with gently sloping banks (<1.5 percent) were lower in silt content than constructed wetlands with steeper slopes. Sediments of older constructed wetlands tended to be lower in silt content than sediments of newly constructed wetlands, but results were insignificant ($p=0.0601$, R-square=0.76). Siltation may be reduced over time as constructed wetland banks are vegetated and become more stabilized.

The time and resources *required* to collect a statistically valid sample size may be prohibitive because macroinvertebrate populations are highly variable. High variability imposes constraints on the reliability of

interpretations based on population size estimates. However, I conclude that a reduced sampling effort directed toward the collection of key subsets of the macroinvertebrate community can be sufficient to detect the effects of major influential factors (hydrology, vegetation, water quality). Even when the sampling effort is limited, semi-terrestrial dipteran species and gastropods may be particularly useful indicators of wetland function (hydrology and energy flow, respectively).

The widely fluctuating conditions characteristic of newly constructed wetlands and the highly variable population sizes in constructed wetlands imply that aquatic macroinvertebrate monitoring of constructed sites prior to the third year after initial inundation may be unnecessary.

The Effect of Inter-Wetland and Intra-Wetland Variability on Sampling Strategies in Faunal Studies Comparing Natural and Constructed Wetlands

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Abstract

Studies comparing natural and constructed wetland faunal communities entail sampling on two levels, yet sampling strategy is seldom formally planned or recognized. Choice of faunal collection sites within the study-wetlands constitutes the first level of sampling, while choice of study-wetlands from the many wetlands available in any geographical region constitutes the second level of sampling. Parameter-area curves comparing the parameter of interest to increasing sampling effort may guide the choice of faunal collection sites within wetlands, while more rigorous statistical techniques may guide the choice of study-wetlands. In both cases, cost considerations may dictate sample sizes smaller than the ideal. Sampling strategy may affect similarity and diversity values. Conceptual diagrams and data from a study comparing fish, chydorid cladoceran, and benthic invertebrate communities in 10 natural and 12 constructed freshwater marshes in central Florida's phosphate mining district, will demonstrate potential sampling pitfalls.

Introduction

Phosphate companies in Florida construct wetlands to mitigate losses of natural wetlands during strip mining. The degree to which constructed wetlands mimic natural wetlands is of interest to government regulators, conversation groups, and the phosphate mining industry. A study initiated in 1989 and now nearing completion, evaluates similarities and differences in fish, benthic invertebrates, macroinvertebrates, and chydorid cladoceran communities of 10 natural and 12 constructed marshes in central Florida's pebble phosphate mining district (Figure 1).

During the planning stages of our study, we were aware that numerous decisions regarding the sampling regime would affect the study's outcome. For example, choice of benthic sampling technique has been shown to favor collection of certain species (Tsu and Breedlove, 1978; Flannagan, 1981; Benfield et al., 1983). Likewise, frequency and duration of sampling must be considered in order to detect short-term and long-term temporal changes in aquatic populations. Green (1987) and lsom (1986) have provided extensive reviews of sampling strategy levels associated with aquatic ecosystem studies. Rather than revisiting each of the numerous levels of sampling strategies, the intention of this paper to review two levels of particular importance in studies designed to compare the fauna of natural
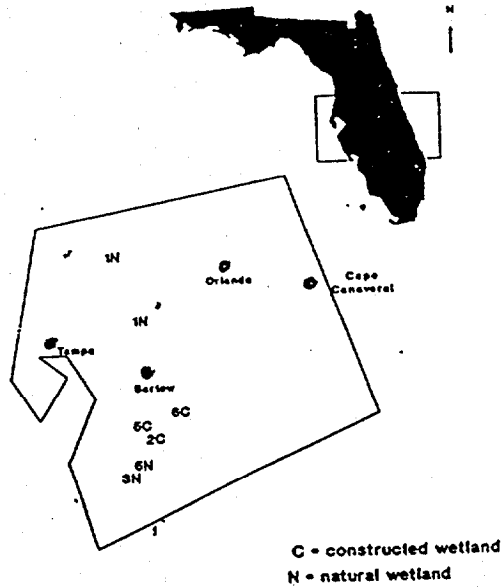
Figure 1. Approximate location of study sties.
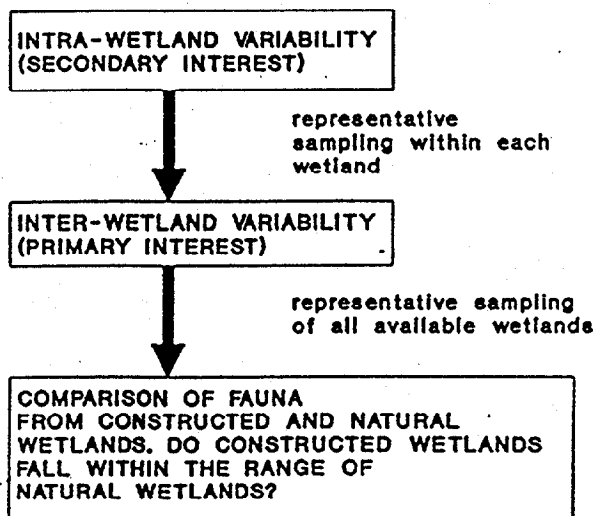
Figure 2. Levels of interest in studies comparing the fauna of constructed and natural wetlands.

and wetlands. The first level, choice of faunal collection sites within each of the study wetlands, is driven by intro-wetland variability, while the second level, choice of study-wetlands, is driven by inter-wetland variability (Figure 2). Since the primary question in the comparative wetland studies considered here regards differences between natural and constructed wetlands, the effect of inter-wetland variability on sampling effort should be considered more rigorously.   However, intra-wetland variability must be examined also to insure representative sampling.


Faunal Collection Sites Within Wetlands: Intra-Wetland Variability.

The relationship between faunal communities and habitat heterogeneity is widely recognized within many ecosystems (MacArthur et al, 1966; Recher, 1969; Murdoch et al., 1972; August, 1983). In intertidal wetlands of southwestern Britain, Warick et al. (1991) showed that both static variables, such as sediment grain size and organic content, and dynamic variables, such as current velocities and wave energy, influenced faunal community structure. Learner (1989) and Wrubleski (1990) found relationships between chironomid populations and vegetation in marshes. Phosphate industry reports on the Agrico Fort Green Reclamation Project near Mulberry, Florida, noted that invertebrate communities were partitioned differently among mocrophyte communities in both a constructed and a natural reference wetland (Erwin, 1987; 1988).   Our own data suggest that fish may also be partitioned among vegetation. The sailfin molly (Poecilia latipinna) was represented in 5 of 7 samples collected from a Pontederia cordata community, but no P. latipinna were found in any of 7 samples collected from an adjacent Hydrocotyle sp community.

Due to such intra-wetland variability, two questions regarding faunal collection sites within each wetland must be addressed: How many sites within each wetland should be sampled, and how should these sites be chosen?   Obviously, multiple replicates must be collected from each wetland in order to account for intra-wetland variability, and some method of insuring that a sample size is truly representative of the entire wetland must be employed.

A simple approach involves the construction of parameter-area curves. Observations of the parameter of interest are collected, and all observations are plotted against measurements representing the parameter of interest until the resulting curve reaches an asymptote (Brower et al., 1990).   Fish sampling data from a 'constructed marsh in central Florida will serve as an example. To measure species richness of fish populations, a throw trap was repeatedly deployed in the wetland, and the cumulative species richness of fish was plotted against number of throw-trap replications until the curve reached an asymptote (Figure 3). Since the asymptote for the wetland occurred after three observations were collected, only three throw-trap replicates were required to be reasonably certain of having obtained a representative sample.   If the asymptote for another wetland does not occur until 15 observations have been collected, then we must collect 15 samples to feel reasonably certain of having obtained a representative sample for that wetland. Also, required sample size may vary when different parameters or different faunal communities are sampled.   For example, the parameter-area curve for fish species

richness reaches it asymptote at a smaller number of replicates than the parameter-area curve of benthic invertebrate species richness (Figure 4).
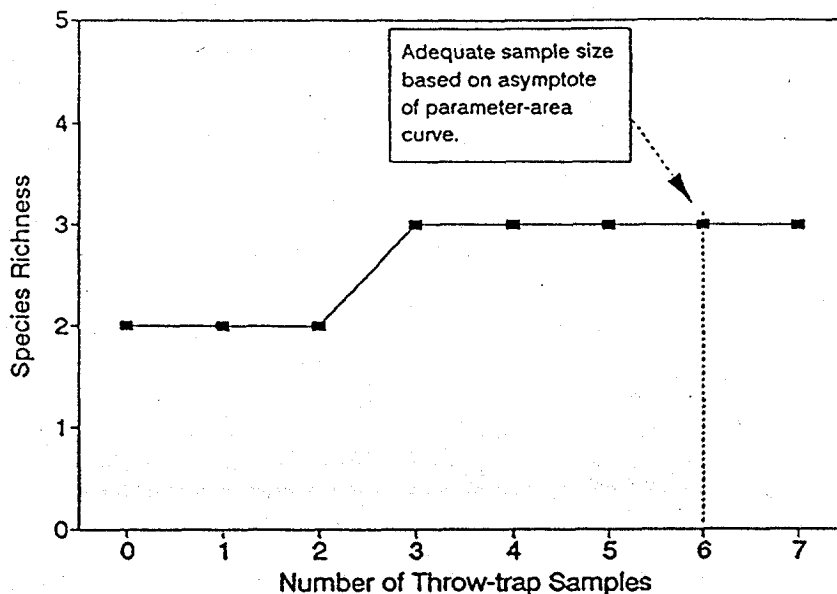
Figure 3. Parameter-area curve used to determine the sampling effort required to estimate species richness of fish communities within each wetland.
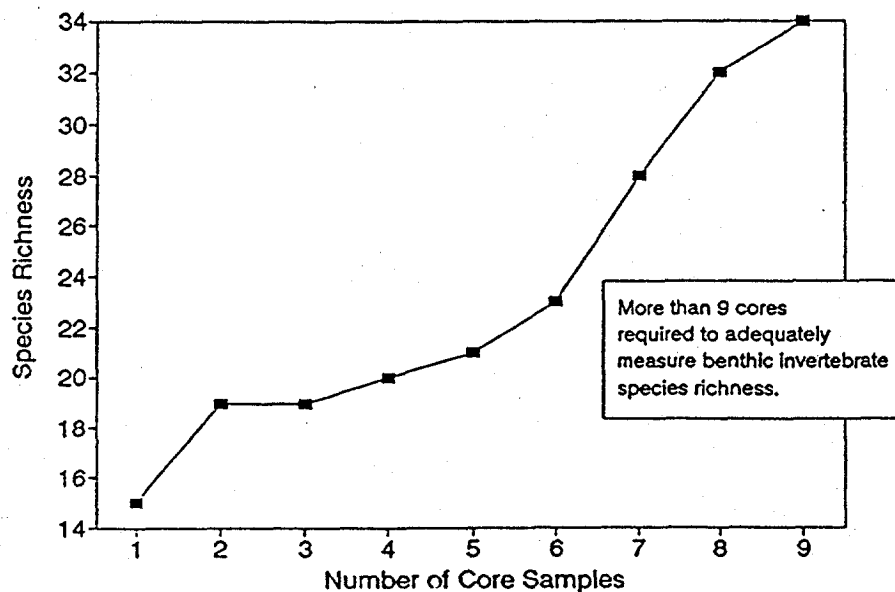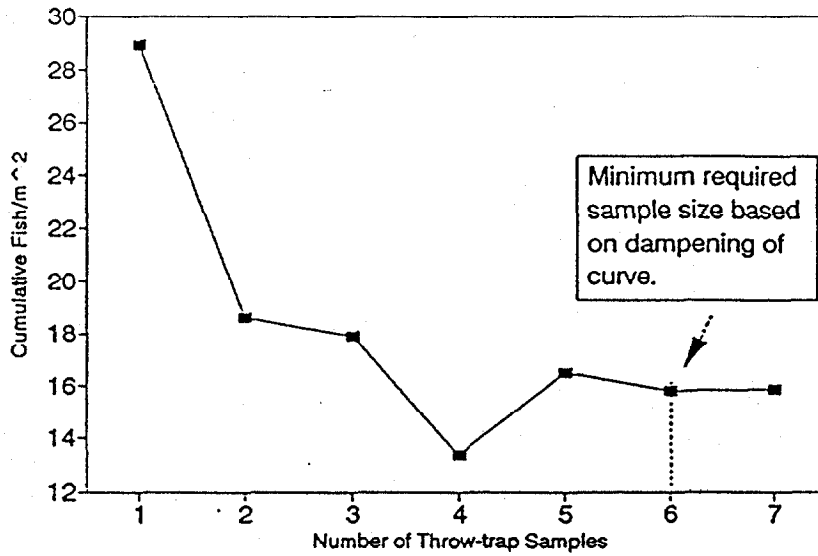

Figure 4. Parameter-area curve used to determine the sampling effort required to estimate species richness of benthic invertebrates within each wetland.

Similarly, performance curves may be constructed for parameters such as mean abundance which are not in themselves cumulative. In order to estimate the required number of replicates for fair representation of fish abundance within a wetland, we plotted the number of trapping events against mean abundance of all previous throws (Figure 5). After six throw trap samples had been collected, the curve dampened out and we were reasonably certain that the mean abundance of fish, as measured by the cumulative mean, was representative of the wetland. Again, required sample size may vary among wetlands and among parameters. A performance curve constructed to estimate the required number of core sample replicates for fair representation of benthic invertebrate abundance suggests that greater than nine cores will have to be collected (Figure 6).

Parameter-area curves and performance curves are not free of problems. Rare events, such as rare species or small areas of unusually high population densities, may be missed. The ordering of the sample replicates may affect the shape of the curve. Because some subjectivity is involved in choosing either the asymptote of the parameter-area curve or the dampened portion of the performance curve, determination of the number of samples that must be collected remains somewhat subjective. Thus, one researcher might sample a smaller number of faunal collection sites within each wetland than another researcher working in the same wetland. Choice of the smaller sample size will reduce sampling effort on a per wetland basis, but the cumulative values for each wetland may be less representative than the cumulative values from a larger sampling effort. Total project cost may, in fact, increase due to an artificial increase in inter-wetland variability, which will, in turn, require the sampling of a greater number of study-wetlands.

In some cases, parameter-area curves or performance curves may indicate that an impractically large sampling effort is required. In wetlands that support numerous plant communities or where other environmental gradients are present, habitat heterogeneity may lead to excessive intra-wetland variability and impractically large sampling effort requirements. In some cases, sampling effort may be reduced by limiting sampling to select habitats within each wetland. For example, all samples might be collected from Pontederia cordata. This approach affects the comparison study in two ways: 1.) only wetlands which support Pontederia cordata communities may be sampled, and 2.) the results of the study will be restricted to Pontederia cordata communities unless some justification allows extension of the results to other communities. Furthermore, variability related to non-plant habitat heterogeneity or chance, may still result in a need for excessive sampling effort. If variability between samples cannot be reduced, the possibility of abandoning examination of a parameter or faunal community should be considered.

Figure 5. Performance curve used to determine the sampling effort required to estimate the fish density within each wetland.
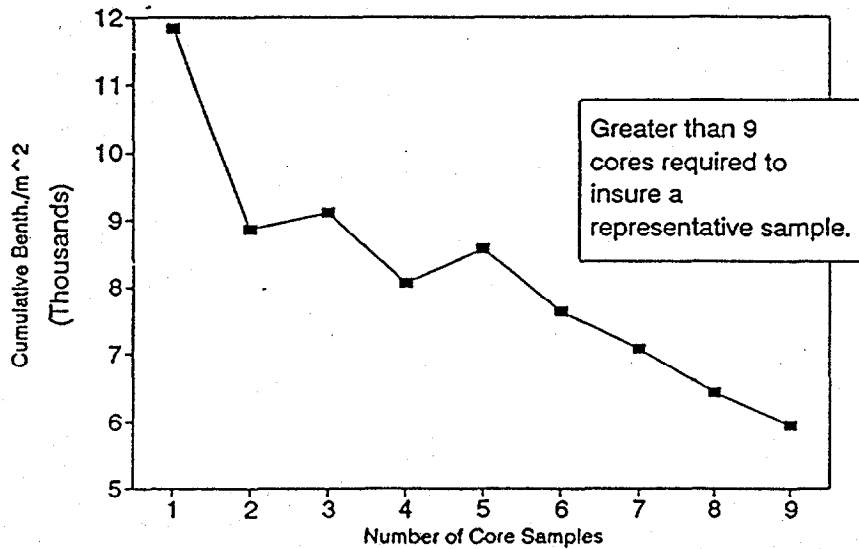
Figure 6. Performance curve used to determine the sampling effort required to estimate benthic invertebrate density within each wetland.

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Choice of Study-Wetlands: Inter-Wetland Variability

Just as intra-wetland variability stipulates the need for replicate sampling within each wetland, inter-wetland variability stipulates a need for replicate sampling among wetlands. Studies that compare natural wetlands to single "reference" wetlands ignore this principle and, thus, may produce misleading results.

The numerous wetland topology schemes in the literature illustrate the extent of inter-wetland variability. Brown et al (1985), listed 8 types of freshwater marshes in Florida: flag marshes, sawgrass marshes, arrowhead marshes, non-grass herbs marshes, cattail marshes, spike-rush marshes, bulrush marshes, and maidencane marshes. Mixing of these types within a single wetland frequently occurs. Additional wetland topology schemes based on parameters other than vegetation have been devised: Ervin (1990) listed five types of nonforested, freshwater wetlands which appear to be based on hydrology, while the U.S. Fish and Wildlife Service (Cowardin et al., 1979) defined wetlands on the basis of soils and hydroperiod, as well as vegetation.

In order to reduce background variation in wetland fauna, due to obvious differences in wetland type, wetlands could be assigned to various strata in the stratified-random sampling scheme, or a study could concentrate on select types of wetlands. However, wetlands may vary in ways that make assignment to discrete strata difficult. Vegetation types may be mixed, as noted above, and physical structure may not be readily apparent. In addition, subtle differences not addressed in typical wetland classification schemes may have dramatic effects on wetland fauna. For example, input of small quantities of chemical pesticides may have major impacts on invertebrate communities, or a wetland's size may control the taxonomic composition of the wetland fauna (LeClaire and Franz, 1990; Moler and Franz, 1987). The amount of faunal variability directly related to such subtle differences depends to some extent on the faunal community of interest. Small imputs of chemical pesticides may have little direct impact on a population of fish, yet the impact on pesticide-sensitive chydorid cladocerans may be extreme.

If wetland type or particular wetland features are normalized, care must be taken to implement identical controls in both natural and constructed sites. If only isolated marshes dominated by <u>Pontederia cordata</u> are to be included in the sample set of natural marshes, then only isolated marshes dominated by <u>Pontederia cordata</u> should be included in the sample set of constructed marshes. Although this may seem to be an obvious point during planning phases, field conditions often make matching of differences between natural and constructed sites difficult, or impossible. For example, it may not be possible to control for contour intervals when comparing older constructed marshes, where steep contours are the norm with natural marshes (Owen et al., 1989). In this situation, two options are possible: attempts to control for contour during choice of study-wetlands could be abandoned, or constructed wetlands with steep contours could be excluded from the study. If all steeply-contoured constructed wetlands are excluded from the study, the exclusion should be acknowledged: the comparison is no longer one of natural and constructed marshes, but rather of constructed and natural marshes without a steep grade. Thus, the objective of the study has 'become more limited than was originally intended.

Where possible, controlling for obvious differences among wetland types remains a sensible approach to limiting background variability, but uncontrollable background variation will still require replicate sampling. That is, similar wetland topology does not justify the use of a single natural "reference" wetland in a comparison study evaluating the. fauna of a constructed wetland.   Data on fish abundance from eight natural wetlands will serve as an example.   All of these wetlands are classified as isolated, herbaceous wetlands.   All samples were collected from <u>Pontederia cordata</u> communities, and the mean values of seven throw-trap replications in each wetland were considered$_2$ representative of that wetland.   Mean abundances of fish ranged from 0 fish/m to 50 fish/m (S.D. = 17.9).   Depending on the desire level of risk and precision, and assuming that variances among fish populations in constructed wetlands will not be appreciatively higher than those found in natural wetlands, the standard deviation' can be used to estimate the necessary sample size (the number of study-wetlands)- required to compare the true mean abundance of fish in these wetlands to those of nearby constructed wetlands, using straightforward univariate statistical tests. If a precision of ± 20 fish with risk coefficenients of a = 0.05 and B = 0.2 is accepted, about 28 wetlands would have to be sampled (Marks, 1982).

Sample size estimated, based on data from our study of natural and constructed wetlands in the phosphate district, have been computed for various other parameters (Table 1).

These computations indicate that the magnitude of the estimated sampling effort may be impractical.   For example, our data indicate that a comparison of mean chydorid cladoceran abundance in natural and constructed wetlands will require the sampling of over 2,450 wetlands.   The highly variable abundance of chydorid cladocerans, which exhibit irregular reproductive pulses during which populations increase by several orders of magnitude, partially accounts for the requirement of such a large sample number.

Given the high variability that exists among certain faunal communities, studies attempting to evaluate certain differences between constructed and natural wetlands may require unrealistically large sample sizes. If an individual parameter of a particular faunal community exhibits high variability, univariate statistical analysis may not be possible.   Resorting to multivariate statistical analysis may prove useful in some situations, but it is often inadequate when the intention of a study is to test hypotheses rather than to describe communities.   Furthermore, multivariate analysis frequently requires sample sites beyond those obtainable in most wetland studies.

Sampling Effects on Shannon-Weiner and Morisita Index Values

The literature is replete with warnings regarding the over-dependence upon and misuse of similarity and diversity indices (Brillouin, 1962; Pielou, 1966; Hurlbert, 1971; Wolda, 1981), yet both are commonly used to compare constructed and natural wetlands (Caldwell, 1989). As a rule, diversity and similarity values should not be accepted as "black box" representations of wetlands. In the context of this paper, the effect of sampling techniques on, similarity and diversity values should be considered whenever these values are applied.

Table 1.   Estimates of sample sizes required to statistically test hypotheses relating constructed and natural wetlands.   In all cases, projected critical values are: $\alpha = 0.05$ and $B = 0.02$.

Parameter of Interest	Standard Deviation (based on n samples from pilot data)	Precision	Estimated Sample Size
Mean fish abundance per m^2	17.9 (8)	± 20 fish	28 wetlands
Mean fish species richness	1.3 (8)	±1 species	56 wetlands
Mean benthic invertebrate abundance per m^2	3321 (6)	± 1500 invertebrates per m^2	156 wetlands
Mean benthic invertebrate species richness	10.74 (6)	± 10 invertebrate species	40 wetlands
Mean chydorid abundance per m^2	20,000 (12)	± 2,000 chydorids per m^2	> 2 4 5 0
Mean chydorid species richness	1.54 (12)	±1 chydorid species	76 wetlands

Similarity values may be impacted by sample size and species diversity. Of the many similarity indices available, the Morisita Index is the least affected by sample size and diversity (Wolda, 1981).   However, even the Morisita Index produces erroneous results when samples are not representative.   Like most similarity indices, the Morisita Index gives values of between 0 (minimal similarity) and approximately 1 (maximal theoretical similarity).   However, to account for sampling error, the maximum practical similarity should be represented by the value found when a wetland is compared to itself.   The sampling procedure that is

accepted for the study should be repeated twice within the same wetland, the value obtained should be accepted as the highest similarity possible, given the sampling procedure used.

Table 2.   Chydorid cladocera collections from modified Whiteside Samplers at 2 locations within a single constructed wetland.   The sites were approximately 15 m apart.

Species	Site A	Site B	Total
Alona monacantha	2	0	2
Chydorus sphaericus	10	14	24
Kurzia latisma	2	6	8
Dunhevedia americana	0	4	4
Biapterus karua	4	12	16
Biapterus affinis	8	4	12
Total Chydorids	26	40	66

Similarity indices computed from chydroid cladoceran data (Table 2) collected more collected from a natural wetland will provide an example.   To modified Whiteside samplers (Whiteside, 1975) were positioned about 15 m apart and left overnight, and a Morisita Similarity Index value was computed to determine the similarity between the contents of the 2 traps:

between the contents of the 2 traps:

$I_M = [2\Sigma x_i y_i] / [(1_1 + 1_2)N_1 N_2] = 0.9197$

Where: I_M =   Morisita's Index of Similarity

x_i =   Number of individuals of species i in first sample.

y_i =   Number of individuals in species i in second sample.

N_1 =   Total number of individuals in first sample.

N_2 =   Total number of individuals in second sample.

$1_1 = \Sigma x_i(x_i - 1)] / [N_1(N_1 - 1)]$

$1_2 = \Sigma y_i(y_i - 1)] / [N_2(N_2 - 1)]$

Since the similarity value 0.9197 represents the similarity of two samples from the same wetland, 0.9197 should be considered as the highest expected similarity value. Thus, when chydorid communities are compared from different wetlands, any similarity values greater than 0.9197 indicate identical chydorid communities within the sensitivity of the sampling technique.

Like Morisita Similarity values, Shannon-Weiner Diversity values are commonly used to evaluate wetland communities. However, the Shannon-Weiner Diversity Index is not appropriate unless random samples of species abundance data are available (Pielou, 1966). When habitat heterogeneity leads to a non-random distribution of organisms within a community, and when organisms are sampled via quadrats or traps, samples cannot be considered random. In this case, the Brillouin (1962) Index will yield valid diversity measures of the sample, but not the community. Index values computed from the chydorids cladoceran data (both traps) discussed in the previous paragraph will provide an example of the differences between the indexes. The diversity values are computed as follows:

$$H' = \quad p_i \log p_i = 0.674$$

Where: H' = Shannon-Weiner diversity of the community.

p_i = The proporation of total number of individuals occurring in species i.

$$H = [\log(N!/n_1!n_2!...n_s!]/N = 0.616$$

Where: H = Brillouin Index of sample diversity.

N = Total number of individuals.

n = Number of individuals for species i.

The two diversity values are not equal, and in fact, only the Brillouin value is valid (Brower, 1990).

Summary

While decisions regarding study-wetland choice are driven by inter-wetland variability, decisions regarding faunal collection sites within each wetland are driven by intra-wetland variability. Pilot data and simple calculations may indicate that straightforward univariate statistical comparisons (hypothesis testing) of certain parameters will require unreasonably large sample sizes. Required sample sizes may be reduced by limiting the study to specific conditions at either the faunal collection site level, the study-wetland level, or both. Alternatively, less variable parameters may be examined, multivariate analysis may be employed, or the statistical analysis may be abandoned. When diversity or similarity indices are computed on wetlands data, the effects of sampling error on the indices should be considered. If samples are not demonstrably representative of the wetlands under comparison, conclusions drawn from index values apply only to the samples and not to the wetland communities,

Acknowledgements

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A PRELIMINARY COMPARISON OF MEIOBENTHIC CLADOCERAN ASSEMBLAGES IN NATURAL AND CONSTRUCTED WETLANDS IN CENTRAL FLORIDA

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Abstract: In Florida, phosphate mining companies that destroy natural wetlands are required to construct mitigation wetlands. Comparisons of plant communities are frequently used to determine the degree to which constructed wetlands mimic natural wetlands, but efforts to address similarity based on fauna are rare. Studies in lake littoral zones suggest that meiobenthic cladocerans possess characteristics that may make their use in wetland comparisons advantageous. In this study, meiobenthic cladocerans were sampled from 8 natural freshwater wetlands and 11 freshwater wetlands constructed on phosphate-mined lands. The pulsed nature of the cladoceran communities limits their value as a quantitative measure of the similarity of natural and constructed wetlands. Qualitative analyses based on the presence or absence of species suggest that cladoceran assemblages of some constructed wetlands mimic those of some natural wetlands, but the range of assemblages found in constructed wetlands is narrower than that found in natural wetlands.

Key Words: Chydoridae, cladocerans, constructed wetlands, marshes, meiobenthos, mitigation, phosphate mining, wetlands.

INTRODUCTION

Approximately 40,000 ha of land in Florida were strip mined for phosphate between 1975 and 1991 (Florida Department of Natural Resources 1992). Activity is heaviest in the Central-Pebble District east of Tampa, where pie-mining land cover is characterized by the presence of wetland, lake, and riparian communities interspersed with upland hardwoods,. pine, and palmetto scrub communities. In order to reach underlying phosphate deposits, mining operations temporarily displace 3 to 20 m of sand and clay overburden. Mining impacts can drastically alter the landscape, changing not only the biological communities and the appearance of the land, but also the substratum structure and the water table (Brooks 1991). These impacts have destroyed an estimated 5422 ha of wetlands in Florida, while an additional 2423 ha of wetlands have been approved for mining (Armstrong 1991).

Since 1975, the Florida Department of Natural Resources has required land reclamation following mining (Florida Administrative Code Section 16C-16, 16C-17). Approximately 2700 ha of mitigation wetlands have been constructed as part of the land reclamation effort (Armstrong 1991), but the degree to which these constructed systems mimic natural wetlands is poorly understood. Although the implied goal of mitigation permits frequently has been the establishment of hydrophyte communities similar to those found in nature (Erwin 1991), similarity of hydrophytes does not guarantee similarity of other features.

To gain a better understanding of the ability of constructed wetlands to mimic natural wetlands, hydrophyte communities should be examined in conjunction with several additional wetland community elements, including aquatic fauna, hydrology, nutrient cycling, and wildlife use. Due to the high variability that exists among both constructed and natural wetlands, studies evaluating the potential ability of constructed wetlands to imitate natural wetlands should examine numerous sites (Streever and Crisman 1993). Ideally, an attempt should be made to emphasize parameters that might later be used by state regulators-and industry consultants in the routine evaluation of constructed wetlands. As a first step toward such a multi-parameter, multi-site study, we examined meiobenthic cladoceran populations, consisting of chydorid cladocerans and allied taxa from 11 constructed and 8 natural freshwater wetland systems.

Historically, the use of aquatic fauna in the assessment of environmental conditions has centered around benthic macroinvertebrate communities, especially chironomids, in temperate zone lacustrine systems (Thienemann 1913, Deevey 1941; Brundin 1956, Brinkhurst 1974). In Florida, attempts have been made

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to construct quantitative models for assessing water quality in lakes based on bryozoans (Crisman et al 1986), *Chaoborus* (Crisman and Crisman 1989), and chydorids (Crisman 1980). Although numerous studies have demonstrated that invertebrate regression models developed for lakes are not applicable to Florida wetlands (Brightman 1976, Haack 1984, Crisman and Meier 1986, Pezeshski 1987), each of the above-mentioned invertebrate groups may be useful in comparisons of natural and constructed wetlands.

Studies in lake littoral zones suggest that meiobenthic cladocerans possess several characteristics that may make their use in wetland comparisons particularly advantageous. First, the ability of cladoceran eggs to survive desiccation, coupled with their relatively rapid life cycle, may allow rapid repopulation of wetland systems following water-level changes. Second, taxonomy of meiobenthic cladocerans is reasonably well known, and in general, identification is reliable to the species level. Third, large numbers of cladocerans may be easily collected from lake littoral zones. Finally, lake cladoceran assemblages are known to be sensitive to a wide range of environmental and ecological factors, including predation by fish and invertebrates, competitive interactions, and water chemistry (Smirnov 1974). Thus, numerous environmental features that may affect fauna directly or indirectly are integrated by cladoceran assemblages.

There is, however, at least one pitfall in the use of meiobenthic cladocerans as indicator organisms: cladoceran population densities are known to vary both seasonally and over shorter periods of time. In temperate zone lakes, population growth follows spring warming, with fluctuations that vary among species within a lake and among lakes for individual species (Smirnov 1974). To the best of our knowledge, meiobenthic cladoceran population density pulses have not been investigated in wetlands. Dramatic pulses in population densities may render the use of meiobenthic cladocerans as quantitative biotic indicators impractical. If individual species peak at different times, as has been found in lakes, comparisons based on relative densities of various species may also prove impractical. The possibility that any index based on meiobenthic cladocerans must rely solely on species presence-absence data had to be accepted at the outset of this study.

Thus, this study focuses on two complementary goals. (1) We evaluate the usefulness of meiobenthic assemblages for routine comparison of natural and constructed wetlands. Our evaluation criteria are based on ease of applicability and potential information content of assemblages. We look for strong species associations to facilitate wetland comparisons based on assemblages. We investigate changes in populations over time and intra-wetland differences in populations because these factors affect information content per sampling effort. (2) We attempt to identify patterns of similarity between natural and constructed wetlands on the basis of the meiobenthic cladoceran assemblages. From these patterns, we formulate a hypothesis regarding the ability of constructed wetlands to imitate natural wetlands.

METHODS

All 19 sites sampled in this study were palustrine emergent wetlands occurring in or near the Central-Pebble Phosphate District east of Tampa, Florida. The 11 constructed sites were built on phosphate-mined lands. by contouring overburden after phosphate deposits were removed. The majority of these wetlands were mulched with sediments collected from nearby natural wetlands. Hand-planting followed flooding at all constructed sites. Dates of flooding varied between 1982 and 1990.

Constructed wetland size ranged from approximately 3 to 80 ha, while natural wetlands ranged in size from approximately 3 to 50 ha. Vegetation in constructed wetlands consisted of mixed herbaceous wetlands species, including *Typha* spp., *Scirpus* spp., *Pontederia cordata* L., *Sagittaria lancifolia* L., and *Panicum hemitomon* Schult. Vegetation in the natural wetlands consisted primarily of *P. cordata, S. lancifolia,* and *P. hemitomon,* with *P. cordata* clearly dominating areal coverage.

Whiteside Pattern Samplers (Whiteside and Williams 1975) were used to collect meiobenthos from areas vegetated with *P. cordata* and *S. lancifolia* and with depths of 30 to 50 cm. Each sampler consisted of a plexiglass plate on which 9 bottles were mounted, with the stem of a funnel extending through the plate and into each bottle. Total surface area covered by each trap was 0.03 m². Samplers were left overnight with the funnel mouths down. Substratum beneath samplers generally consisted of leaf litter overlying peat or sand. Because meiobenthic cladocerans are known to migrate vertically at night, they are easily trapped within samplers of this design. Bottles were retrieved the morning after deployment, and the contents were sieved through a 64-μm sieve and preserved in 80% ethanol.

Silt disturbed during placement of traps frequently contaminated samplers. To separate the meiobenthos from contaminants, samples were picked under 20 x magnification in the laboratory. Picking times ranged between 1 and 20 hours per sample, depending upon sediment content. All meiobenthic cladocerans found in the samples were identified following schemes summarized by Brooks (1959), Smirnov (1974) and Pennak (1989); Three species, probably from the genus *Alona,* could not be identified and were thus numbered

1 to 3. Because *Chydorus sphaericus* and *Chydorus piger* Sars could not be consistently separated, both species were recorded as *C. sphaericus*. A total of 158 samples were processed.

In order to investigate the occurrence of population pulses, samples were collected monthly at constructed wetlands C9 and C10 from November 1989 through November 1991 and May 1990 through September 1991, respectively. Samples used to investigate intrawetland variability were collected quarterly by deploying 2 samplers, spaced 10 to 30 meters apart, at constructed wetlands C1, C2, C3, C4, C5, and C6. Quarterly samples were collected at all other wetlands for a minimum of 1 year during 1990-91 in order to identify patterns of similarity between natural and constructed wetland meiobenthic cladoceran populations. Due to seasonal drying of some wetlands, along with occasional accessibility problems, sampling was incomplete at several wetlands.

The pulsing of meiobenthic population densities rendered quantitative comparisons of assemblages impractical, so our search for associations among meiobenthic species, as well as our search for patterns of similarity, depended only upon the presence or absence of individual taxa. Because drying of wetlands and accessibility problems prevented winter sampling at some sites, only samples from spring, summer, and fall quarters were considered. Samples from each wetland were pooled, and the resulting assemblage was considered representative of the wetland.

Because this study was of an inductive nature, we felt that extensive statistical analysis was inappropriate. Thus, we limited statistical examination of the presence-absence data to the following 3 steps. (1) A principal component analysis was performed in an effort to identify species associations within all wetlands (Krzanowski 1988). (2) A similarity matrix based on Sorensen's Index (Southwood 1991) was constructed in order to identify patterns of meiobenthic community similarity among wetlands. The matrix consisted of 3 types of similarity relationships: one relationship compared constructed wetlands to other constructed wetlands, one compared constructed wetlands to natural wetlands, and one compared natural wetlands to other natural wetlands. A one-way ANOVA followed by a Student Newman-Keuls (SNK) multiple comparison procedure was applied to detect differences in mean similarities among the 3 relationship types (Marks 1990). (3) Dendrograms were constructed from average linkage cluster analysis and centroid cluster analysis in order to illustrate relative similarities between wetlands on the basis of their meiobenthic communities (Manly 1990). All computations were run using SAS (1985).
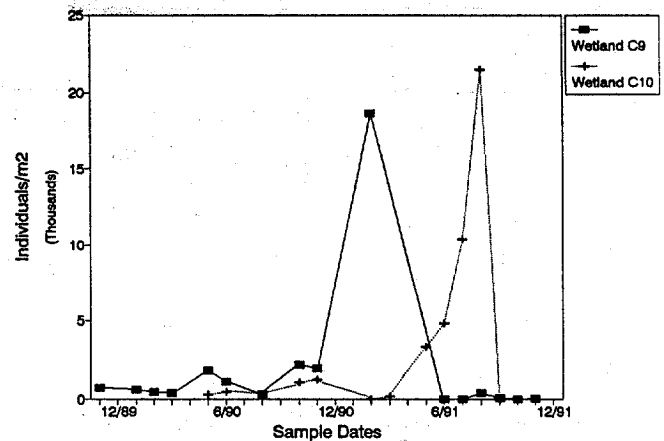

Figure 1. Population densities of meiobenthic cladocerans at constructed wetlands C9 and C10.

RESULTS AND DISCUSSION

Meiobenthos Temporal and Spatial Variability

At constructed wetlands C9, and C10, where samples were collected monthly, total meiobenthic cladoceran populations ranged from 0 to 18,648 and 0 to 21,481 cladocerans/m^2 respectively (Figure 1). The greatest single density change detected over a 1-month period occurred at C10 during August, 1991 with a decrease from 21,481 to 0 cladocerans/m^2. Population density pulses were not seasonally synchronized. For example, in February 1991, wetland C9 peaked at its greatest population density, 18,648 cladocerans/m^2, while wetland C10 supported 0 cladocerans/m^* during the same month. Likewise, wetland C10 consistently supported over 3000 cladocerans/m^2 in June, July, and August 1991, while populations at wetland C9 were below 300/m^2 during the same period. Just as total population density peaks and crashes occurred differentially among wetlands, population peaks and crashes of individual species within wetlands fluctuated differentially. For example, in May, June, and July 1991, *Alonella circumfimbriata* population densities at constructed wetland C10 varied from 1038 to 3047 to 1875 cladocerans/m^2, while *Kurzia latissima* populations varied from 2227 to 201 to 971 cladocerans/m^2. Because of this high temporal variation in population density and relative abundance of individual species, comparison of wetlands on the basis of a limited number of sampling dates may produce misleading results.

Since studies of lake littoral zones have shown differential meiobenthic population density pulses between lakes and between species within a lake (Smirnov 1974) the occurrence of similar non-synchronized pulses in wetland populations is not surprising. The pulsed nature of meiobenthic population densities requires frequent sampling if some idea of quantitative population parameters is desired. However, sediments
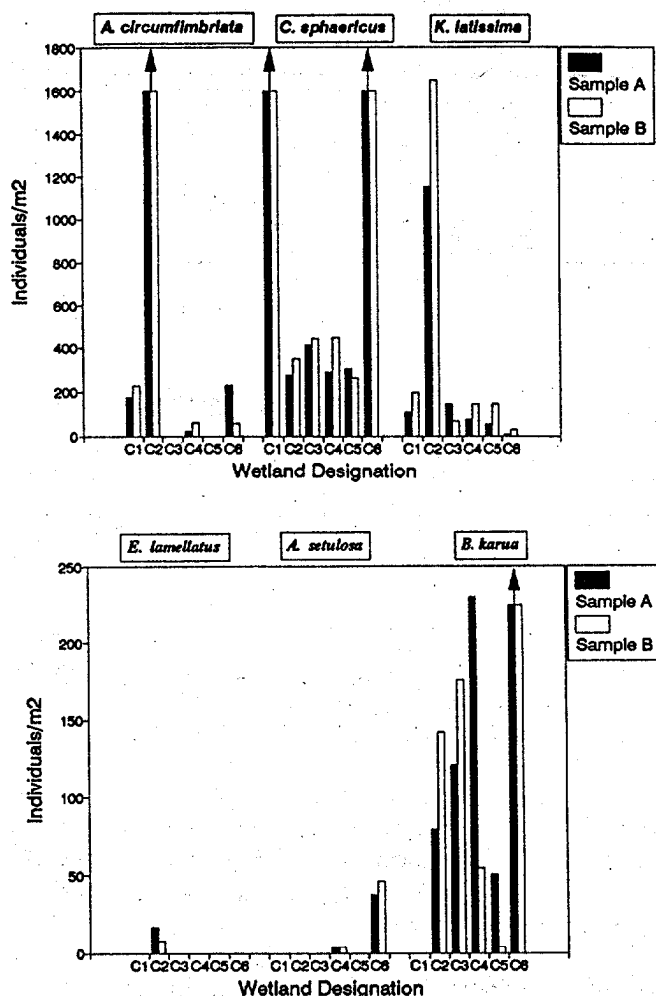
Figure 2. Mean population densities for 6 species of meiobenthic cladocerans showing intra-wetland variability. The upper graph plots densities for 3 frequently occurring species, while the lower graph plots densities for 3 infrequently occurring species. Samples A and B were separated by 10 to 30 m. Arrows indicate measurements off the scale of the graph. Although densities varied considerably between sites, the presence of a species in Sample A was always accompanied by the presence of the same species in Sample B.

that contaminated samplers deployed in the wetlands led to high sample processing costs and thus rendered frequent sampling impractical. Even relative densities could not be used due to differential peaking between species. Thus, we were forced to abandon a quantitative approach and consider only the presence or absence of species in our search for meiobenthic species associations and in our search for pattern among wetlands. Although abandonment of quantitative data in favor of nominal data carries with it a penalty of information loss, in this particular instance, the loss was unavoidable.

In future comparative wetland studies, quantifiable.

samples may be available in the form of chitinous subfossil remains found in the uppermost layer of wetland sediments. The use of such remains is well known in lake studies (Goulden 1966, Whiteside 1970, Crisman 1978). Since cladoceran remains in sediments are well preserved, sediment samples should be temporally integrated and thus quantifiable. However, since some portion of the sediment in the constructed wetlands included in this study was imported from nearby natural donor wetlands, and since it was impossible to separate the subfossil remains of the meiobenthos produced in constructed wetlands from those contained within imported sediment, a quantitative approach dependent upon the subfossil record was not possible in our study.

Paired samplers at constructed wetlands C1-C6 indicated that population densities of individual species varied at sampling locations separated by 10 to 30 m but that presence-absence trends were similar (Figure 2). Although our intra-wetland sampling was not extensive enough to rigorously establish a probability of obtaining identical samples from any location within a wetland, we felt that our limited results justified tentative acceptance of species presence-absence samples collected at a single location. Further research may determine the degree to which meiobenthic populations vary within individual wetlands.

Meiobenthic Assemblages

We found 31 species of meiobenthic cladocerans (Table 1). Species richness varied from a single species present at wetland N17 to 18 species present at wetland N 18 (Figure 3). Mean species richness for the 11 constructed wetlands was 6.6 (S.D. = 2.2) while the 8 natural wetlands had a mean species richness of 8.25 (S.D. = 5.5). No relationship was apparent between constructed wetland age and species richness. Although species composition varied among sites, certain species occurred in a greater number of wetlands than others (Table 1). In particular, *C. sphaericus* was found at all but 2 wetlands. Since *C. sphaericus* is known to occur almost ubiquitously throughout its range (Whiteside 1970), the frequent appearance of this taxon in our samples was not surprising.

Certain species occurred differentially in either natural or constructed wetlands (Table 1). For example, *K. latissima* occurred in all 11 constructed wetlands but was absent from all but 2 of the 8 natural wetlands sampled. Whiteside (1970) reports that *K. latissima* is typically found in bogs and ponds, and we can offer no explanation regarding its absence from the majority of natural wetlands in our study. Similarly, *Alonella excisa* occurred in 4 of the 8 natural wetlands sampled but was absent from all 11 constructed wetlands. Like

Table 1. Meiobenthic cladoceran taxa and their relative occurrences in constructed and natural wetlands.

Taxon	Constructed Wetlands	Natural Wetlands
Kurzia latissima (Kurz)	100.0%	25.0%
Chydorus sphaericus (Müller)	90.9%	87.5%
Alona circumfimbriata Megard	81.8%	50.0%
Biapertura karua (King)	63.6%	25.0%
Biapertura affinis (Leydig)	54.5%	25.0%
Dunhevidia crassa King	45.5%	62.5%
Ephemeroporus hybridus (Daday)	36.4%	62.5%
Ephemeroporus barroisi (Richard)	27.3%	37.5%
Pseudochydorus globosus Baird	18.2%	37.5%
Alona quadrangularis (Müller)	18.2%	25.0%
Alona setulosa Megard	18.2%	25.0%
Unknown #1	18.2%	12.5%
Pleuroxus laevis Sars	18.2%	12.5%
Alona monacantha Sars	18.2%	12.5%
Camptocercus rectirostris Schødler	9.1%	37.5%
Alona rustica Scott	9.1%	37.5%
Alona guttata Sars	9.1%	12.5%
Eurycercus lamellatus (Müller)	9.1%	12.5%
Leydigia leydigi (Schoedler)	9.1%	0.0%
Alona intermedia Sars	9.1%	0.0%
Alonella excisa (Fischer)	0.0%	50.0%
Oxyurella tenuicaudis (Sars)	0.0%	25.0%
Alonella dadayi Birge	0.0%	25.0%
Alonella exigua (Lillejeborg)	0.0%	25.0%
Unknown #2	0.0%	12.5%
Unknown #3	0.0%	12.5%
Ilyocryptus spinifer Herrick	0.0%	12.5%
Alona pulchella King	0.0%	12.5%
Acroperus harpae Baird	0.0%	12.5%
Pleuroxus denticulatus Birge	0.0%	12.5%
Alonella verrucosa Sars	0.0%	12.5%

K. latissima, A. excisa is commonly found in bogs and ponds (Whiteside 1970). Overall, 2 species were found within the constructed wetlands that were not found in the natural wetlands, but 11 species were found within natural wetlands that were not found in constructed wetlands. Limited ecological data on these species offers no definitive explanation for these patterns.

Natural wetland N16 contained 2 species, designated as Unknown 2 and 3, that were not found in any of the other 18 wetlands sampled. Natural wetland N14 also contained 2 species, *Alona pulchella* and *Ilyocryptus spinifer,* which were not found in any of the other wetlands. Of the 9 species found in only 1 of the 18 wetlands sampled, 7 were from natural wetlands and 2 were from constructed wetlands. Thus, rare species seemed to occur more frequently in natural wetlands than in constructed wetlands. The 2 constructed wetlands containing rare species were relatively young systems, both flooded in 1990. The occurrence of rare species in young systems may suggest that recruitment

of species over time plays a minor role in shaping assemblages within wetlands. The apparently limited role of recruitment in shaping assemblages is readily explained by the ability ofdrought-resistant cladoceran eggs to hatch from imported mulch.

Principal component analysis of meiobenthic cladoceran assemblages indicated that the first 3 principal components explained 19%, 17%, and 13% of the variability. No other components explained more than 10% of the total variability: 'Given the small percentage of variation explained, it seems likely that species associations are not particularly strong (Rexstad et al. 1988).

Comparison of Natural and Constructed
Wetland Assemblages

A one-way ANOVA and a SNK multiple comparison procedure compared the mean Sorensen similarity values for the three possible similarity relationships.of (1) constructed *vs.* constructed wetlands, (2) constructed vs. natural wetlands, and (3) natural *vs.* natural wet-
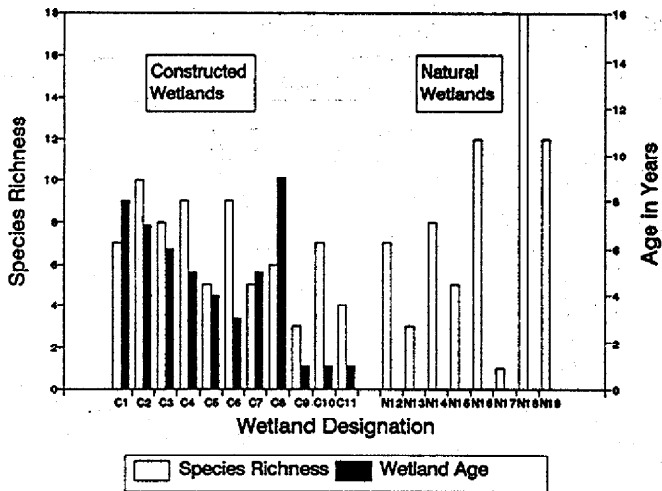
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Figure 3. Cladoceran species richness at 11 constructed and 8 natural wetlands. Age in years represents the approximate time since initial flooding of constructed wetlands.

lands. Results of the ANOVA indicated a significant difference in mean similarity values ($p < 0.0001$, $F=38.04$). The SNK multiple comparison procedure indicated that the mean index value comparing constructed *vs.* constructed wetlands was significantly greater ($p < 0.05$, $DF=168$) than the mean index values comparing constructed *vs.* natural wetlands and natural *vs.* natural wetlands (Table 2). Thus, constructed wetlands were most similar to other constructed wetlands, while natural wetlands were no more similar to one another than to constructed wetlands. Apparently meiobenthic cladoceran assemblages found in natural

Table 2. Mean Sorensen Similarity Index values for each of the three possible relationships between the natural and constructed wetlands. The Student-Newman-Keuls multiple comparison procedure indicates that the mean similarity value that results when constructed wetlands are compared to other constructed wetlands is higher than the mean similarity values resulting from comparisons of constructed wetlands to natural wetlands and natural wetlands to natural wetlands ($p < 0.05$). Mean similarity values resulting from comparisons of constructed wetlands to natural wetlands and natural wetlands to natural wetlands are not significantly different.

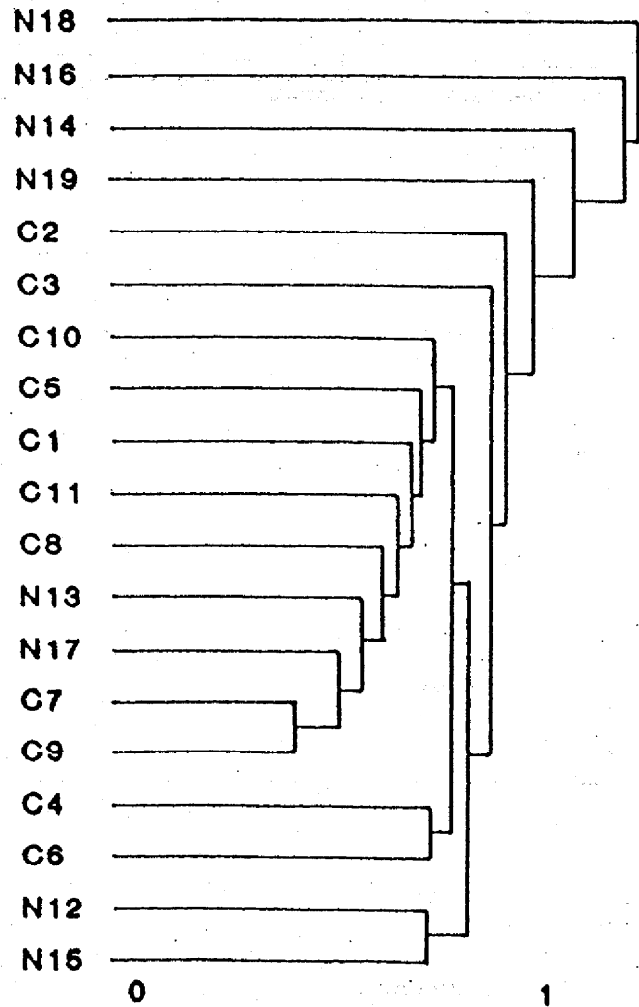| Similarity Comparison	Mean Sorensen Similarity Value
Constructed wetlands compared to other constructed wetlands	0.5339
Constructed wetlands compared to natural wetlands	0.3497
Natural wetlands compared to other natural wetlands	0.2975

Figure 4. Dendrogram showing the similarity between wetlands as revealed by centroid cluster analysis of meiobenthic cladoceran assemblages. The position of the vertical line on the distance axis indicates the relative distance at which clusters of wetlands were joined.

wetlands were more variable than those found in constructed wetlands.

The centroid cluster analysis produced a dendrogram that also indicated greater variability among natural wetlands than constructed wetlands (Figure 4). Since the centroid cluster and average linkage cluster methods produced similar results, only the centroid cluster analysis was considered. Nine of the 11 constructed wetlands and 4 of the 8 natural wetlands comprised a large central cluster, while constructed wetlands C2 and C3, along with natural wetlands N14, N16, N18, and N19 joined the other wetlands individually. Of the wetlands that joined individually, constructed wetlands C2 and C3 were less separated from the main cluster than were natural wetlands N14, N16, N18, and N19. Thus, 4 of the 8 natural wetlands fell outside the main group, while only 2 of the 11 constructed wetlands fell outside of the main group.

Given the differences in the range of assemblages that existed in the wetlands, it is tempting to speculate on factors that might control these assemblages. Competition for substratum and food, predation, water chemistry, hydroperiod, and temperature may all play a role. Due to the paucity of knowledge regarding the ecology of the species considered in this study and the lack of quantitative data on the meiobenthic assemblages, it is impossible to go beyond speculation or to suggest a relative importance of each of these factors. However, it seems reasonable to expect that more than one of these factors affects assemblages. In addition, these factors may affect meiobenthic populations directly by acting upon a meiobenthic cladoceran species, or indirectly by acting upon a predator, a competitor, or upon some resource of a meiobenthic cladoceran species. Because of the many factors affecting these assemblages, differences may not correspond to readily identifiable differences between wetlands. Instead, assemblage differences may be due to differences in a combination of factors, both biotic and abiotic, existing in a wetland. Thus, our analysis of meiobenthic cladoceran assemblages indicates that the environmental and ecological attributes that control the assemblages may be more variable among natural wetlands than among constructed wetlands.

CONCLUSION

The goals of this study included (1) evaluation of the usefulness of meiobenthic assemblages in the routine comparison of natural and constructed wetlands and (2) identification of patterns of similarity in the meiobenthic assemblages of 8 natural and 11 constructed wetlands. The cost of sample processing and wide temporal fluctuations in cladoceran densities suggest that meiobenthic cladoceran assemblages may not be particularly useful in the routine comparison of wetlands. Comparison of wetlands based only on species presence-absence data leads us to hypothesize that a broader range of biotic assemblages and the conditions that drive these assemblages occurs in natural wetlands than in constructed wetlands. This hypothesis requires testing using different wetlands and an indicator species that can be quantified.

If additional research supports this hypothesis, two practical implications may impact management decisions. (1) Because of the wide range of assemblages that occur in nature, a lack of similarity between a constructed wetland and .a single natural wetland does not necessarily indicate a lack of similarity between the constructed wetland and all natural wetlands. This point renders the use of natural "reference" wetlands problematical. (2) Because the full range of constructed wetland conditions reflected in the meiobenthic cla-

doceran assemblages represents only a subset of the range of natural wetland conditions, mitigation replacement of a large percentage of natural wetlands in any one area should be approached with caution. The possibility of a decreased range of wetland conditions should be considered in the determination of an acceptable degree of wetland destruction and mitigation in areas of intensive landscape alteration.

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A Comparison of Fish Populations from Natural and Constructed Freshwater Marshes in Central Florida

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ABSTRACT: In Florida, freshwater marshes are constructed as mitigation for wetland loss associated with phosphate strip mining, but little is known regarding the similarity of fish communities in natural and constructed marshes. Fish from five constructed and eight natural marshes were sampled quarterly via throw-traps for one year. Gambusia holbrooki, Heterandria formosa, Poecilia latipinna, Elassoma evergladei, Fundulus chrysotus, Jordanella floridae, Fundulus rubifrons, and unidentified juvenile centrarchids were found in both constructed and natural marshes, while Lucania goodei was found only in constructed marshes. A comparison between constructed and natural marsh populations showed that differences in mean abundance and biomass at $p < 0.05$ (Wilcoxon rank-sum test) were present for G. holbrooki and E. evergladei. Differences in populations may be attributed to differences in conditions found in constructed and natural marshes.

INTRODUCTION

Phosphate mining in Florida has led to the destruction of over 5000 ha of wetlands. Since 1975, legal requirements for land reclamation following mining have led to the construction of approximately 2700 ha of mitigation wetlands, many of which are freshwater, herbaceous marshes (Florida Department of Natural Resources 1992). Mitigation permits commonly require the establishment andmonitoring of various hydrophyte communities, but similarity of hydrophyte communities does not necessarily insure similarity of faunal communities. The need to monitor faunal similarity has been recognized by numerous groups (Erwin 1991, Jensen and Platts 1991). In the study reported here, fish communities of five constructed and eight natural freshwater herbaceous marshes were compared.

Fish communities in natural and constructed marshes may differ for at least two reasons: (1) recruitment of fish species to constructed marshes may be limited, and (2) environmental conditiona of constructed marshes may differ from those of natural marshes. Pathways of recruitment to both natural and constructed marshes are poorly understood. In many cases, individuals are recruited from adjacent water bodies which are connected permanently or seasonally to a particular marsh. However, many marshes exist without permanent surface water connections to other water bodies. Maintenance of fish populations in such isolated marshes must depend on either replacement of individuals through reproduction or on irregular events that provide temporary connections to external recruitment sources. Fish populations are frequently absent from isolated marshes subject to seasonal drying (Moler and Franz 1987).

Numerous environmental conditions might affect fish communities. Niche theory suggests that parameters such as temperature, dissolved oxygen, and pH will define a fundamental niche for fish species. This fundamental niche hypervolume will be reduced by competition and predation to a smaller realized niche (Hutchinson 1957). In light of niche theory, it would be reasonable to suspect that physical and chemical differences between constructed and natural marshes might lead to the development of different fish communities. For example, since marsh fish communities are heavily influenced by water level fluctuations (Kushlan 1980), hydroperiod differences between natural and constructed marshes may lead to differences in fish communities. Likewise, differences in substrata may lead to differences in water chemistry between constructed and natural marshes (Langis 1991, Kiefer 1992), which in turn may lead to differences in fish communities.

Rather than directly measuring differences in recruitment and environmental conditions for natural and constructed marshes, we reasoned that differences would be reflected in the communities existing in the

marshes. If lack of recruitment of species to constructed marshes is preventing development of "natural" fish communities, then certain species found in natural marshes should be entirely absent from constructed marshes. If different conditions exist in natural and constructed marshes, then different species, abundances, and biomasses should be found. If recruitment and conditions are the same in natural and constructed marshes, then fish communities should be similar.

METHODS AND MATERIALS

All marshes included in this study were freshwater, herbaceous marshes in Florida's Central-Pebble District east of Tampa. The marshes were selected on the basis of accessibility and subjective similarity of their hydrophyte communities. Constructed marsh size ranged from 3 to 80 ha, while natural marsh size ranged from 3 to 50 ha. Constructed marsh age ranged from two to ten years. Because fish may have distinct preferences for vegetation or depth, all samples were collected from stands of _Pontederia cordata_ with water depths of 0.1 - 0.4 m. Because seasonal differences have dramatic effects on marshes, samples were collected at three month intervals from June 1991 through March 1992.

A 0.24 m throw-trap was used to sample fish. Similar 1 m^2 throw-traps have been recommended for sampling small fish from vegetated marshes (Chick et al. 1992), and the efficiency of similar 1 m^2 throw-traps has been favorably compared to seining methods in herbaceous marshes (Freeman et al. 1984). Because substratum irregularity in constructed marshes led to difficulties in sealing a 1 m^2 throw-trap to the bottom, the smaller 0.24 m trap was used to sample all marshes. The throw-trap was deployed seven times at each sampling date in each marsh. With the exception of centrarchids, all fish were identified to species. Because juvenile centrarchids could not be reliably identified to species, no attempt was made to identify any centrarchids beyond family. All fish were counted, dried at 90 C for 72 hours, and weighed.

In order to address questions regarding differential recruitment of fish to constructed and natural marshes, all trap-throws for all quarters for each site were pooled. However, not all of the marshes could be sampled every quarter: two natural marshes were dry on the December 1991 sampling date, one constructed marsh was dry on the March 1992 sampling date, and one natural marsh could not be sampled after June 1991 because of nearby agricultural development. Thus, sampling effort for the comparison of fish species present at natural and constructed sites consisted of 28 or fewer trap-throws at each site.

In order to address questions regarding the abundance and biomass of fish, samples were compared on a seasonal basis; Because of concerns regarding the ability of seven trap-throws (the sampling effort for each quarter at each marsh) to census rare species or species adept at avoiding the throw-trap, those species collected in fewer than two trap-throws for any given date were not considered in the analysis. Marshes that could not be sampled for a given sample date were not included in the analysis. Abundances and biomasses were statistically compared via multiple two-tailed Wilcoxon rank-sum tests run on SAS 6.04, 1987. We used a repeated univariate, non-parametric test because of the small sample size and doubts regarding normality of distribution (Tabachnick and Fidel1 1983).

RESULTS AND DISCUSSION

Because constructed marshes are new features in the landscape, it seems reasonable to suspect that recruitment of fish species may limit species richness and therefore control community structure. If recruitment of a species to constructed marshes is limited, relative occurrence of that species in constructed marshes (that is, the number of constructed marshes in which the species is found) should be lower than the relative occurrence in natural marshes.

All species found within natural marshes also occurred within constructed marshes (Table 1). _L. goodei_ was found, in some constructed marshes but was absent from natural marshes. Total species richness was slightly higher in the constructed marshes than in the natural marshes. In all but one case, relative occurrence of each species in constructed marshes was higher than or equal to the relative occurrence in natural marshes. Only _E. evergladei_ occurred in a higher percentage of natural than constructed marshes. Thus, comparison of the relative numbers of

constructed and natural marshes in which each species occurs clearly indicates that constructed marsh fish communities do not suffer from poor recruitment of species, with the possible exception for E. evergladei. It should be noted that E. evergladei in constructed marshes never occurred in the high densities that were found in the natural marshes, which suggests that low numbers may be related to environmental conditions in the constructed marshes rather than differences in recruitment.

Since recruitment did not appear to limit the species available to the constructed marshes examined in this study, any difference in community structure between natural and constructed marshes should be attributable to differences in either biotic or abiotic environmental factors. In order to address this issue, mean abundances and biomasses from constructed marshes were compared to mean abundances and biomasses from natural marshes (Table 2). G. holbrooki was significantly more abundant ($p < 0.05$) in constructed marshes in June 1991 and in March 1992, and E. evergladei was significantly more abundant ($p < 0.05$) in natural marshes in June, September, and December 1991. In all cases, significant differences in mean abundances were accompanied by significant differences in mean biomasses. Because the Wilcoxon rank-sum test examines differences in assigned ranks rather than differences in actual abundance or biomass values, the equality of the significance values for abundance and biomass comparisons of any one species in a given quarter is not surprising.

Although it is beyond the scope of this study to explain the causes of differences that were detected, discussion of at least one possible cause is appropriate. Examination of mean abundance and biomass values, along with the accompanying significance values, suggests that some seasonal effect is present, Although we neglected to record water depths at each sampling event, subjective observations indicate that seasonal water level changes may account for seasonal fish community changes. As marshes dry during winter months, fish are concentrated into smaller areas where both predation and competition are intensified. Following reflooding, densities of fish are low (Kushlan 1980). All of the natural marshes sampled in this study were dry for a short period between December 1991 and March 1992. Because of the recent completion of the drying-reflooding cycle, the March 1992 sampling event produced only a few fish from the natural marshes. Conversely, none of the constructed marshes dried during the winter dry season, and thus measurable numbers of fish were found throughout the year. The significant difference found for G. holbrooki populations of natural and constructed marshes in the June 1991 and the March 1992 samples can be explained by these water level fluctuations. G. holbrooki was virtually eliminated from natural marshes during the dry season, but reasonably large populations remained in constructed marshes. Following reflooding, the few G. holbrooki remaining in the natural marshes required several months to increase their numbers to a level comparable to that of the constructed marsh populations. Although significant differences in H. formosa populations could not be detected at the small sample size of this study, similar population changes were apparently present.

Table 1. Relative occurances of fish species in five constrcuted and eight natural marshes, based on results of all trap-throws from four quarterly sampling events.

Fish Species	% of Constructed Marshes with Occurrence	% of Natural Marshes with Occurrence
Gambusia holbrooki	100.0	100.0
Heterandria formosa	100.0	100.0
Fundulus chrysotus	80.0	75.0
Jordanella floridae	80.0	25.0
Poecilia latipinna	60.0	12.5
Lucania goodei	60.0	0.0
Elassoma evergladei	40.0	87.5
Fundulus rubifrons	20.0	12.5
Centrarchids	20.0	12.5

Table 2. Mean abundances/m^2 and biomasses/m^2 for constructed and natural marshes, and the significance of the difference between constructed and natural marshes based on the Wilcoxon rank-sum test (DF = 1). Significance values greater than 0.05 are reported as not significant (NS). Fish which did not appear in at least two of seven trap throws were not included in this analysis due to concerns regarding the ability of seven trap throws to fairly census rare species or species adept at trap avoidance.

Fish Species	Mean Abundance/m^2 (Biomass in g/m^2) in Constructed Marshes	Mean Abundance/m^2 in (Biomass in g/m^2) in Natural Marshes	Significance Significance of difference in abundance/m^2 (Biomasses/m^2)†
June 1991			
Gambusia holbrooki	24.3 (0.614)	5.5 (0.101)	0.033 (0.012)
Heterandria formosa	16.6 (0.148)	7.8 (0.058)	NS (NS)
Poecilia latipinna	5.4 (0.444)	0.0 (0.000)	NS (NS)
Elassoma evergladei	0.0 (0.000)	4.2 (0.096)	0.021 (0.021)
Fundulus chrysotus	1.3 (0.549)	0.4 (0.082)	NS (NS)
September 1991			
Gambusia holbrooki	9.7 (0.348)	4.4 (0.256)	NS (NS)
Heterandria formosa	10.0 (0.098)	10.8 (0.157)	NS (NS)
Jordanella floridae	0.2 (0.082)	0.0 (0.000)	NS (NS)
Poecilia latipinna	7.2 (1.389)	0.0 (0.0)	NS (NS)
Elassoma evergladei	0.0 (0.000)	3.4 (0.118)	0.011 (0.011)
Fundulus chrysotus	1.5 (0.300)	0.5 (0.104)	NS (NS)
Lucania goodei	0.7 (0.04)	0.0 (0.000)	NS (NS)
Centrarchids	0.24 (0.104)	0.0 (0.000)	NS (NS)
December 1991			
Gambusia holbrooki	14.1 (0.554)	18.1 (0.601)	NS (NS)
Heterandria formosa	7.3 (0.136)	13.6 (0.132)	NS (NS)
Jordanella floridae	1.3 (0.062)	0.0 (0.000)	NS (NS)
Poecilia latipinna	3.3 (0.770)	0.0 (0.000)	NS (NS)
Elassoma evergladei	0.0 (0.000)	29.5 (0.608)	0.025 (0.025)
Fundulus chrysotus	1.1 (0.192)	3.3 (0.498)	NS (NS)
March 1992			
Gambusia holbrooki	8.3 (0.426)	0.0 (0.000)	0.016 (0.016)
Heterandria formosa	5.8 (0.132)	0.3 (0.009)	NS (NS)
Poecilia latipinna	2.2 (0.647)	0.0 (0.000)	NS (NS)
Jordanella floridae	0.7 (0.256)	0.0 (0.000)	0.069 (0.069)

† Probability > |z|.

Significant differences in populations of E. evergladei cannot be explained by seasonal fish community changes. Although E. evergladei was present in some of the constructed marshes (Table 1), it was never present in sufficient numbers to be trapped dependably during quarterly sampling events (Table 2). However, E. evergladei was consistently present in relatively high numbers in seven of the eight natural marshes, with the exception of the post-reflooding March 1992 sampling event. Furthermore, high mean abundances and biomasses of E. evergladei in the natural marshes indicate that conditions were favorable for its support. Conversely, in the two constructed marshes where E. evergladei, was present, its densities were insufficient to allow estimation using the sampling method employed. Three other taxa, P. latipinna, J. floridae, and centrarchids, were found with low frequency in the constructed marshes, but not in the natural marshes. Because of a lack of knowledge regarding the full spectrum of environmental conditions found in the study sites, we cannot offer any specific explanation for this pattern. We can, however, repeat our initial remarks regarding niche theory: if fish communities differ but species pools are similar, differential conditions must be present. Differences may be related to hydroperiod, as appeared to be the case for G. holbrooki, or they may be more subtle.

ACKNOWLEDGEMENTS

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CAS: A NEW SOFTWARE PACKAGE FOR ANALYZING SPECIES ABUNDANCE DATA

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ABSTRACT

Monitoring of the abundance and species composition of flora and fauna allows comparisons of constructed and natural wetlands. However, large species lists and extremely variable abundances complicate data management and comparative analyses. The Community Analysis System (CAS), a recently developed software package, simplifies these tasks. CAS facilitates data tabulation and report generation, multivariate analysis, and plot generation. Data tabulation and report generation allow creation of tables of raw data and accompanying statistics, such as diversity index values. Multivariate analysis capabilities allow classification (cluster analysis) and ordination. Plotting capabilities allow construction of dendrograms, three-dimensional ordination plots, and species area curves. In addition, recovery analysis, inversion of samples, and exclusion and pooling of taxonomic groups are readily performed. Although SAS and SPSS are capable of performing similar data manipulations and analyses, CAS offers an easily learned, user-friendly format that may allow for more' effective interpretation of data sets assembled during mitigation monitoring.

INTRODUCTION

Sampling of restored and created wetlands frequently results in large matrices of samples and taxa. Community Analysis System (CAS) software provides a convenient method of processing taxa-abundance data to produce summary reports, diversity measurements, species-area plots, similarity matrices, classification (cluster) dendrograms, and ordination plots, The purpose of this paper is to describe the capabilities of CAS and to discuss various potential applications of CAS in the wetlands creation and restoration field. Major sections of the program's primary menu (Figure 1) will be discussed separately. It is important to note that not all of the functions available in CAS will be discussed.

TAXONOMIC CATALOG UTILITIES

Taxonomic catalog utilities allow creation of lists of taxa contained in samples and accompanying identification codes which will be associated with each taxon. Entries may be made at any taxonomic level (Figure 2), and subsequent analyses may be made at any of the taxonomic levels in the catalog. For example, an initial analysis may indicate that certain plant species are absent from a particular site, while a secondary analysis may show that entire families are missing. Taxonomic catalogs may be printed as a summary report.
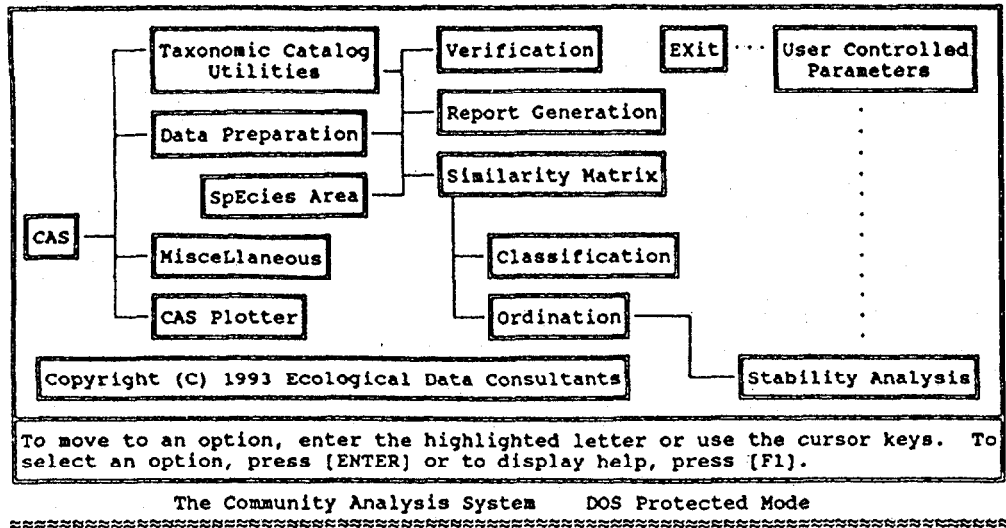
Figure 1. Primary CAS menu.
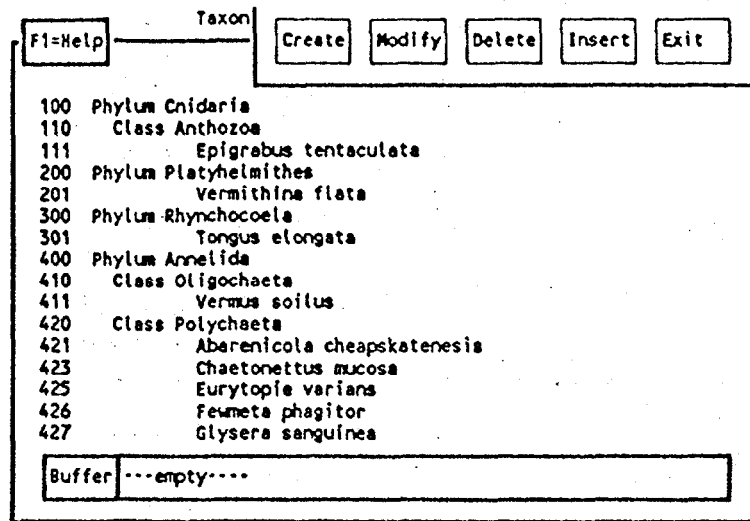
Figure 2. Taxonomic catalog.

DATA PREPARATION

Obviously, all data from a study must at some point be entered into the computer. By taking a "fill-in-the-blank" approach, the data preparation screens in CAS tend to minimize the chances of incorrectly entering data. The sample header screen (Figure 3) allows entry of data specific to each sample, while the data-entry screen (Figure 4) allows entry of taxon names and abundances. When the name of a taxon is keyed into the data entry screen, the identification code associated with that taxon automatically appears in the "ID" window. The abundance of the taxon is keyed into the "Value" window.

```
Please enter....              Press  (F1)  for help

Project :

Location :
Sampling Date    :
Replicate        :

Sample Name      :_____

Sample Size      :     1.00000
Sample Unit      :

Comment : ()
                        Last Modified on


Ctrl-C = abort                          Type Over
```

Figure 3. Sample header screen for data entry.

```
Project                           Catalog Entries        61
Location                          Taxa in search range   61
Date
Sample name  an example   Replicate=
Sample size       1.00000 Unit       =


ID        Name                           Value

ESC=Abort  F1=Help  HOME=Header  INS=Insert  DEL=Delete  END=Alter #  F10=Clear
```

Figure 4. Sample entry screen for data entry.

VERIFICATION

Every data set should be checked for errors prior to analysis. In general, this involves checking data sheets against data entered into the computer. Although this can be accomplished by comparing data sheets to entries as they appear on the screen, it is generally more effective to use print-outs of information which appears on the screen. The verification menu allows generation of printed reports which can be easily compared to data sheets.

REPORT GENERATION

The report generation menu may be used to create summary reports (Figure 5) containing information on taxa richness, total number of specimens collected, specimen densities, diversity, and evenness. The reports also summarize the absolute and relative contributions of each taxa present in the data set. Reports may be generated for individual samples or for collections of samples. Thus, reports could be generated for each of the plant. quadrats collected. in a wetland and for the pooled plant quadrats of the wetland. In addition, a matrix can be created to allow quick comparisons of various summary statistics between samples (Figure 6), and a report of the mean and standard deviation for each taxa in a number of samples can be created (Figure 7). Reports generated by CAS are formatted to allow direct insertion into a document with little or no revision. Pielou (1977) and May (1975) provide reviews of the mathematics of diversity and evenness.


```
                                                    Page  1
                     Cheapskate Bay

Project       =Cheapskate Bay Faunal Survey (CAS Example Dataset)
Location      =Station # 1
Date          =July 23,1981
Name          =St#1 Jul81
Replicate     =1
Size          =    0.09375
Sample Unit   =m-2
Catalog       =Cheapskt.TXC
Last Modified =Sep 21,1990

Sample Characteristics
  Number of Species                 :           11
  Density of All Species            :   1664.00000 /m-2
  Total numbers of specimens counted:    156.00000
  Diversity (Hprime -log base e)    :      1.69375
  Diversity (Hprime -log base 10)   :      0.73559
  Evenness (Hprime/Hmax)            :      0.70635
  Evenness (Scaled)                 :      0.65006
  Simpson's Index                   :      0.75087

          Taxa                  Observed   Percentage

Phylum Annelida
  Class Oligochaeta
          Vermus sollus         533.33333   32.05128
  Class Polychaeta
          Eurytopie varians     608.00000   36.53846
          Fewmeta phagitor       32.00000    1.92308
Phylum Mollusca
  Class Bivalvia
          Nucula interferrous    53.33333    3.20513
          Peeropig peeropig      10.66667    0.64103
          Uno donu               74.66667    4.48718
  Class Gastropoda
          Nastle vibovex         96.00000    5.76923
          Turida predaceous      21.33333    1.28205
Phylum Arthropoda
  Class Crustacea
    Order Amphipoda
          Copralithia phagus     64.00000    3.84615
          Gamma rush             10.66667    0.64103
  Class Insecta
          Chiromona extendor    160.00000    9.61538
```

Figure 5. Summary Report.

```
                                                    Page   1
                      Cheapskate Bay

  Sample Characteristics          St#1 Jul81   St#2 Jul81

  Number of Species         :         11           16
  Density of All Species    :    1664.00000   1845.33333
  Total numbers of specimens counted :  156.00000    173.00000
  Diversity (Hprime -log base e)  :    1.69375      1.55312
  Diversity (Hprime -log base 10) :    0.73559      0.67451
  Evenness (Hprime/Hmax)    :      0.70635      0.58851
  Evenness (Scaled)         :      0.65006      0.50176
  Simpson's Index           :      0.75087      0.63611

  Densities based on following units :      m-2          m-2
```

Figure 6. Data report for comparison between samples.


```
                                                        Page   1
                        Cheapskate Bay

              Taxa               Mean      Std.Dev.    N / N
                                        (Divisor = N)

Phylum Platyhelmithes
       Vermithina flata        5.33333     6.15840    2/ 4
Phylum Rhynchocoela
       Tongus elongata        18.66667    13.42193    3/ 4
Phylum Annelida
   Class Oligochaeta
       Vermus soilus         738.66667   549.26429    4/ 4
   Class Polychaeta
       Abarenicola
         cheapskatenesis     13.33333    10.21256    3/ 4
       Eurytopie varians    152.00000      ----      1/ 4
       Feumeta phagitor       8.00000      ----      1/ 4
       Pseudosludgia indicatus 45.33333   52.52724    2/ 4
       Sabellaria podunka    40.00000      ----      1/ 4
Phylum Mollusca
   Class Bivalvia
       Geoductus improbabilus 2.66667      ----      1/ 4
       Nucula interferrous   29.33333    28.05286    3/ 4
       Peeropig peeropig     98.66667   103.92020    4/ 4
       Tellina confusus       5.33333      ----      1/ 4
       Uno donu              40.00000    34.15000    4/ 4
   Class Gastropoda
       Nastia vibovex       120.00000    30.63767    4/ 4
       Polinices lunaformes   5.33333     6.15840    2/ 4
       Turida predaceous     24.00000     5.33333    4/ 4
Phylum Arthropoda
   Class Crustacea
     Order Amphipoda
       Coprolithia phagus   618.66667   651.91865    4/ 4
       Gamma rush            16.00000     6.15840    4/ 4
   Class Insecta
       Chiromona extendor    72.00000    84.15858    2/ 4
Phylum Echinodermata
   Class Holothuroidea
       Stychopus disgustus   32.00000      ----      1/ 4
Phylum Hemichordata
       Ptycho daru           45.33333    52.52724    2/ 4
```

Figure 7. Summary reports giving means and standard deviations for each taxon.

64

SPECIES AREA

Curves comparing sampling effort to species richness indicate the level of sampling required to characterize a community qualitatively (Southwood, 1991). The concept may be extended to include quantitative information by replacing species richness with diversity. Further insight may be gained by considering only the evenness component of diversity. In each case, the shape of the curve will be affected by the order in which samples are drawn from the data. By chance, the first sample might contain an unusually large number of, species, or it might contain only a few.

The species area menu in CAS readily computes species richness, diversity, and evenness curves. Samples are randomly drawn from the data set and richness, diversity, and evenness are computed. Problems with the effect of chance events on the shape of the curve are minimized by performing numerous iterations at each level of sampling effort, then reporting a mean and standard deviation. Data can be represented in a table or a figure (Figure 8).

SIMILARITY MATRIX

Similarity indices are used to compute a value which represents the degree to which two data sets are similar. In general, two identical data sets would be assigned a value of one, while two data sets with no species in common would be assigned a value of zero.
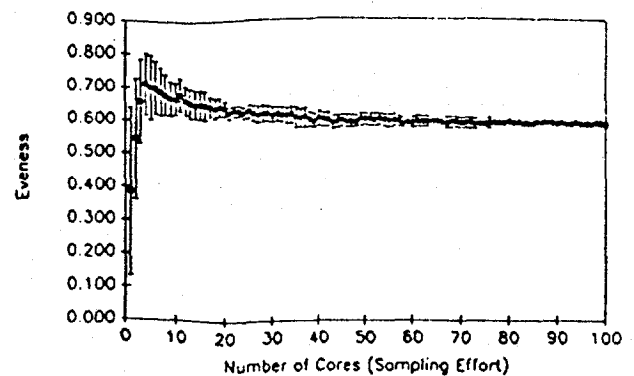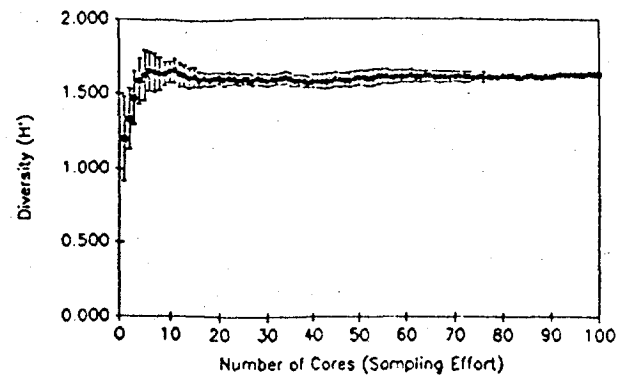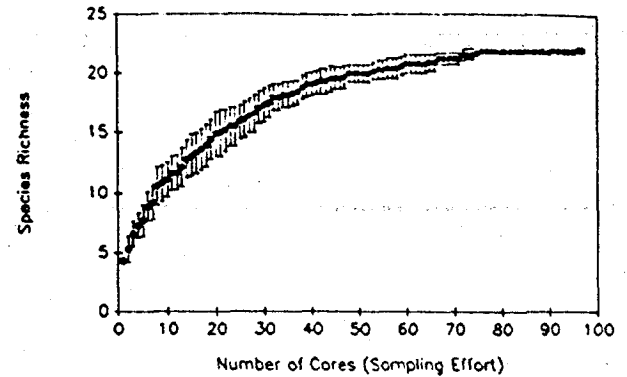
Figure 8. Curves for determining effect of increased sampling effort.

Because some similarity analyses are more effectively performed on transformed data, CAS provides various commonly-used transformation algorithms. CAS also allows choices between numerous qualitative (species presence-absence) and quantitative (species abundance) similarity indices (Figure 9). Output from the similarity analyses can be printed in a matrix (Figure 10) or saved for use in either classification or ordination. Several of the more commonly used similarity indices and the affect of index choice are reviewed by Bloom (1981).

CLASSIFICATION

The classification menu in CAS allows generation of dendrograms based upon similarities between samples (Figure 11). Samples which are most similar to one another in the dendrogram. Samples will be joined into clusters at some point along the similarity axis of the dendrogram. CAS allows the application of various clustering strategies, including nearest neighbor, furthest neighbor, group average, and flexibility. See Krzanowski (1990) or any comprehensive multivariate statistics text for further information regarding cluster analysis.

Proportional similarity	Coefficient of divergence
Canberra metric	Gower's Angular seperation
Gower's distance measure	Rho
Morisita's index (original)	Czekanowski's Qualitative index
Morista's index (Horn's mod.)	Jaccard's index
Euclidean distance	Simple matching
Mean character Distance	

—Press (F1) for help—

Figure 9. Available similarity indices.

ORDINATION

Like classification, ordination is used to show relationships between samples. In ordination, each species in a set of samples is assigned an axis in multi-dimensional space, and each sample is plotted against the axes [Krzanowski, 1990). Although this is easily pictured for three species (a three-dimensional plot), visualization of plots containing greater than three species is generally impossible.

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                                                        Page 1
                    Cheapskate Bay Stations 1..4

Transformation = None
Standardization = None
Index = Proportional Similarity


            Stat#1 Jul81 Stat#2 Jul81 Stat#3 Jul81

Stat#2 Jul81      0.16413
Stat#3 Jul81      0.31891        0.66228
Stat#4 Jul81      0.53061        0.16111      0.56596
```

Figure 10. Matrix of similarity.

CAS uses principal coordinate analysis to reduce the number of axes from the maximum (which is equal to the total number of species in the sample) to three. In the three-dimensional representation, the first axis represents the line through multivariate space which contains the greatest variability; the second axis represents the line through multivariate space which contains the second greatest variability; and, the third axis represents the line through multivariate space which contains the third greatest variability. Variability explained by each line is reported by CAS, and

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a "lollipop diagram" representing the three dimensional figure is produced (Figure 12).
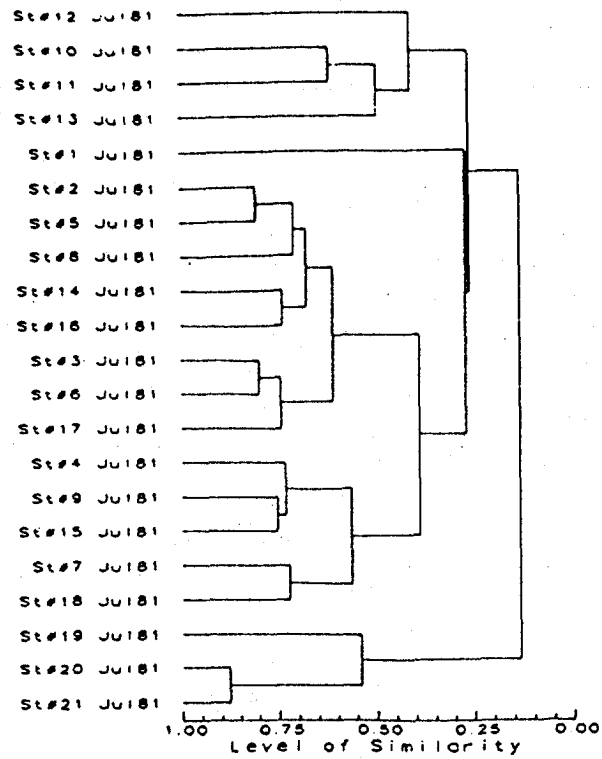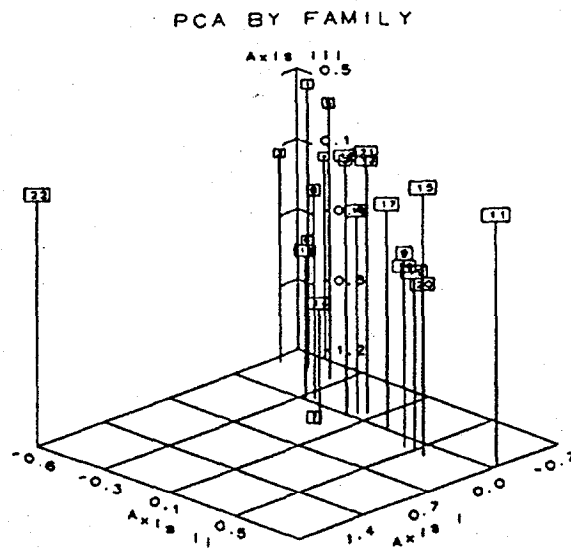

Figure 11. Cluster dendrogram

Figure 12. Principle coordinate Analysis plot.

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STABILITY ANALYSIS

The stability analysis menu provides a method of tracking the development of a community through multivariate space. If the plant community in a restored marsh is sampled over a period of several years, one might expect it to return to a "natural" condition. If data is available to indicate what constitutes a "natural" condition, the approach of the restored marsh to the end point "natural" condition can be plotted. Bloom (1980) explains stability analysis.

MISCELLANEOUS

The Miscellaneous Menu provides various data manipulation operations that may be useful for some analyses. For example, if a researcher is interested in identifying species assemblages, samples and species could be inverted. In an inverted data matrix, the samples become attributes of the species, thus allowing operations such as generation of classification dendrograms which will cluster species assemblages. If a researcher is interested in the effect of removing some taxon, such as Typha from a data set, this is easily accomplished with the "Drop species from one or more samples." option. Likewise, a researcher might be interested in the effect of merging two taxa, such as Taxodium distichum and Taxodium ascendens.

One further option available on the Miscellaneous Menu, "Convert spreadsheet tables to CAS data-sets." will be of importance to users who store data in Lotus or Quattro' spreadsheets. This option creates crude taxonomic catalogs and enters all data from the spreadsheet into CAS.

CAS PLOTTER

The CAS Plotter Menu generates figures from programs run in CAS. Figures 11 and 12 were generated by the CAS plotter. In general, figures made on CAS are suitable for direct insertion into reports or publications.

SUMMARY

GAS is not a substitute for major statistical software packages such as SAS and SPSS. Instead, CAS provides a user-friendly environment in which to explore ecological data with techniques commonly employed by ecologists. CAS also provides an efficient means of generating tables and figures for use in reports and publications. Although this paper is not intended as an advertisement for CAS software, readers should be aware that it is available from Ecological Data Consultants, Inc., P.O. Box 760, Archer, Florida 32618. A low-cost "demo disk" is available for users who wish to explore the program prior to purchase.

ACKNOWLEDGEMENTS

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CONCLUSIONS AND RECOMMENDATIONS

The current study was designed to address two major gaps in our knowledge of wetlands constructed on phosphate mined lands of Florida: 1) succession of ecosystem structure and function and 2) development of bioindicators for evaluating water quality. Four specific questions were addressed relative to ecosystem succession following construction:

1. What is the relationship between plant community structure and ecosystem functioning?

2. Is there a lag time between plant establishment and relative stability in individual ecosystem functional parameters?

3. Does ecosystem structure and function ever approximate that of a natural marsh, and if so, how long does it take to be attained?

4. Can ecosystem succession times be altered significantly via system design and initial mulching with organic substrates from natural marshes?

The database for addressing the questions was compiled from 24 marshes located in the central Florida phosphate region, equally divided between natural and constructed marshes ranging in age between just created to ten years of age. Often a subset of this database was used to address specific research questions.

1) Succession of ecosystem structure and function

1A) What is the relationship between plant community structure and ecosystem functioning?

Plant community structure plays an important role in regulating ecosystem functioning. It influences invertebrate community composition via its roles as substrate and food. It regulates total ecosystem functioning ultimately through the quality and quantity of detritus produced. Respiration rates of the plant community as well as decomposition rates of plant detritus control levels of dissolved oxygen present in the wetland and ultimately the biota of the system through individual species' tolerance of low oxygen conditions. The quantity and composition of system "memory" (i.e. autochthonously produced sediment from plant production) will ultimately stabilize ecosystem fluctuations in functional parameters to approximate those of natural systems, but there is a lag time between initial plant colonization of the newly constructed wetlands and development of sufficient sediment memory to approximate functional aspects of natural systems. It appears that this lag time can be approximately five years for constructed wetland ecosystems of central Florida.

1B) Is there a lag time between plant establishment and relative stability in individual ecosystem functional parameters?

Both the percent macrophyte cover and plant community species richness in constructed wetlands attained values similar to natural marshes approximately one year after construction. In spite of a similar structural appearance to natural wetlands in a short time period, the functional aspects of constructed wetlands took considerably longer to approach stability and were linked to the development of organic sediments from plant detritus. Litter density and mass per unit area accumulated linearly for 3.5 years following wetland flooding, then reached a plateau. Thus, the peak in detritus mass lagged behind above ground plant biomass by approximately 1.5 years. As will be discussed below, it was only after attainment of predicatable sediment composition after 3.5 years that fluctuations in ecosystem physical-chemical and biotic parameters were dampened and interwetland variability declined.

1C) Does ecosystem structure and function ever approximate that of a natural marsh, and if so, how long does it take to be attained?

Constructed wetlands were similar to natural wetlands in species richness, percent cover, biomass structure and hydric preferences of plant species within 1-1.5 years following construction, but did exhibit high intersite variability, high intrasite diversity and more overall coverage by opportunistic species. Exotic plant species displayed higher coverage in constructed wetlands 1-4 years old, then declined to approximate the level found in natural systems after seven years of age. Algal production as indicated by chlorophyll a peaked shortly following inundation of newly created wetlands, then dropped rapidly as plant cover was quickly established.

Phosphorus peaked in constructed wetlands within the first 1.5 years, then decreased to approach the lower levels characteristic of natural wetlands between five and eight years after construction. Relative to phosphorus, the nitrogen peak in constructed wetlands was delayed by 2.5 years, peaking approximately four years following inundation. Values then declined to levels approximating those of natural wetlands at about five years following construction. During the first year following construction, nitrogen to phosphorus ratios declined steadily, after which they increased progressively for years two and three. The data are inconclusive due to the pronounced intersite variability, particularly in older sites. It is therefore unclear when N:P ratios may approximate those found in natural wetlands of the area.

Nutrient concentrations and cycling are strongly linked with sediment characteristics. Litter density and mass per unit area accumulated linearly for 3.5 years after wetland formation, then reached a plateau. The peak in detritus mass lagged behind above ground plant biomass by about 1.5 years. It is possible that water chemistry equilibrium is attained after sufficient organic matter has accumulated in the wetland.

The structure of benthic macroinvertebrate communities of constructed wetlands approximated that of natural wetlands three years following system innundation. Biotic structural and functional parameters displaying this trend included Morisita's Similarity Index, diversity, the importance of major feeding guilds, total benthic abundance and taxa richness. Because of inter- and intra-wetland variation, confidence interval analysis of these parameters did not indicate a trend of increasing stability in constructed wetlands relative to natural wetlands withing the age range (0-8 years) of systems studied.

It is suggested that wetlands constructed on phosphate mined lands of central Florida pass through three distinct development stages. The first is an initial lag phase lasting approximately two years when planktonic primary production is rapidly replaced by emergent plant species. The second phase of development focuses around a trophic surge and decline lasting approximately three years. It is during this phase that phosphorus values peak, then decline to lower relatively stable levels, and nitrogen is progressively accumulted by the ecosystem, especially in plant tissue and the sediments. Throughout this period, organic matter is progressively accumulated in the ecosystem associated with detrital production from emergent vegetation. The final development phase, equilibrium, is acheived only after sediment composition approximates that found in natural systems and can act to dampen system variability in chemical cycling. This phase is attained at approximately five years following construction, although individual physical-chemical and biological parameters stabilize between three and six years. For most parameters, once equilibrium is attained, values approximate those found in natural wetlands of the area.

1D) Can ecosystem succession times be altered significantly via system design and initial mulching with organic substrates from natural marshes?

Design parameters involving wetland morphology, location with respect to natural wetlands and substrate selection were explored for relationships with water quality, chemical partitioning and accrual within the wetland and plant community structure. The data suggested that a threshold mean depth that allows macrophytes to root in the contoured substrate, as opposed to growing as a floating mat, was the parameter with the most control over water quality in the marsh. Shore development, side slopes and the relative amount of wetland area to upland watershed influenced water chemistry. Plant community structure appeared to be heavily controlled by mulching with donor wetland organic matter. The effects of mulching over water chemistry were not significant. Bottom roughness, proximity to natural wetlands, and total water area displayed little control on chemistry or botany over the range of conditions covered by the current database. Although not examined quantitatively, water level fluctuations may be a significant controlling variable for water chemistry and both plant and benthic invertebrate community structure through changes in oxygen regime and redox conditions.

2) Development of bioindicators for evaluating water quality

In addition to an assessment of the currently accepted biotic indicator for

approximating water quality, benthic macroinvertebrates, we examined the possibility of developing water quality indices based on chydorid cladocerans and fish. Although the structure of benthic invertebrate communities in constructed wetlands approximates that of natural wetland within three years of inundation, the reduced range of dissolved oxygen values found in all wetlands, regardless of origin, significantly reduces the utility of benthic invertebrates as water quality indicators. Such an index relies almost exclusively on differential taxon tolerance to oxygen. The exceeding low and generally stressful dissolved oxygen values characterizing all wetlands, regardless of human impact, renders the use of conventional water quality indices based on benthic invertebrate assemblages from lakes and streams highly questionable. The current database failed to provide an adequate water quality index based on benthos that could be used for assessment of wetlands of central Florida.

Recognizing the pitfalls inherent in using benthic macroinvertebrates, we also examined the utility of chydorid cladocerans as water quality indicators for central Florida wetlands. While chydorids have proven to be excellent water quality indicators in lakes, their applicability to wetland assessment is hindered by the highly pulsed nature of their populations and the fact that dissolved oxygen levels vary little among wetlands regardless of their origin. Qualitative analyses based on the presence or absence of species suggested that cladoceran assemblages of some constructed wetlands mimic those of some natural wetlands, but the range of assemblages found in constructed wetlands was narrower than those of natural wetlands.

Fish communities were also assessed in both constructed and natural wetlands quarterly for one year using throw traps. Total species richness was slightly higher in constructed wetlands and it appeared that species composition was not limited by recruitment following inundation. Smaller water level fluctuations in natural marshes relative to those of constructed wetlands likely accounted for seasonal differences in fish assemblages. It was not possible to construct a water quality model based on the current fish database.

A final aspect of this research project was estimation of the sample size necessary to characterize benthic invertebrate, chydorid cladoceran and fish communities in central Florida wetlands. It was suggested that six throw trap samples and at least nine core samples are required to assess species richness or density of fish and benthic invertebrates, respectively, in a central Florida wetland. If, however, the goal is to test hypotheses regarding biotic differences between constructed and natural wetlands, the number of samples to statistically assess abundance differences increases to 28 wetlands for fish, 156 for benthic invertebrates and 2,450 for chydorid cladocerans. Assessment of species richness for these three biotic parameters requires 56, 40 and 76 wetlands, respectively.

Recommendations

This study raised a number of questions regarding current practices of monitoring water quality in constructed wetlands and assessment of how closely structural and

functional aspects of constructed wetlands approximate those of natural wetlands of central Florida. It appears that water quality indices developed for benthic invertebrates may have limited application to wetland monitoring, regardless of whether such systems are natural or constructed on phosphate mined lands. It also places in question the utility of monitoring benthic invertebrates in constructed wetlands younger than three years, when only those older than three years ever approximate a community structure that resembles natural systems. Attempts to develop alternative biotic indices from chydorid cladocerans and fish proved unsuccessful.

It is recommended that a standard methodology and protocol be developed for monitoring water quality in constructed wetlands. Sampling should probably begin following trophic surge and continue until N and/or P decline to stable levels similar to those in natural wetlands. In addition, both the number of samples per sampling event and the sampling methodology should be standardized.

It is critical that the relationship between wetland design and ecosystem structure and function be delineated. It is essential that all design considerations begin with an understanding of the purpose for which the system is being designed. While the current study has suggested that hydrological regime plays a major controlling influence over plant composition, it is unclear how plant composition, in turn, affects the functioning of contructed wetlands. Finally, it is recommended that the importance of habitat heterogeneity in wetland design and its possible effects on the multipurpose use of constructed wetlands be examined.